Terrestrial Current Landscape Condition (CLC) Assessment
San Juan National Forest, Colorado
USDA Forest Service – Rocky Mountain Region

Working Draft Report

Table of Contents

List of Figures

List of Tables

Chapter 1. Introduction

The San Juan and GMUG (Grand Mesa, Uncompahgre and Gunnison) National Forests

The Aquatic, Riparian and Wetlands (ARW), Terrestrial Current Landscape Condition (CLC) and Geographic Analysis Area (GAA) Assessments

The Aquatic, Riparian and Wetland Assessment (ARWA) and Current Landscape Condition (CLC) assessments describe the aquatic and terrestrial ecological characteristics of lands influenced directly and indirectly by Forest planning and management on the San Juan and Grand Mesa, Uncompaghre and Gunnison (GMUG) National Forests of western Colorado. In addition, this assessment describes the anthropogenic influences from European settlers and their relationship to these ecological characteristics (Winters et al. 2004).

These two forests encompass approximately 8,195 square miles (5.2 million acres) of the Rocky Mountain Region (Region 2) of the U.S. Department of Agriculture, Forest Service (Table 1-1). The analysis area is located near the geographic center of five western states: Wyoming, Utah, Colorado, Arizona and New Mexico (Fig 1-1).

Table 1-1 Area table for the GMUG and San Juan National Forests.

Forest Name	Acres	Sq. Miles	Percent
Grand Mesa	351,194	548	6.7%
Gunnison	1,796,022	2,806	34.0%
Uncompahgre	1,040,553	1,625	19.7%
GMUG Total	3,187,769	4,980	60.4%
San Juan	2,093,085	3,270	39.6%
Total	5,280,854	8,251	100.0%

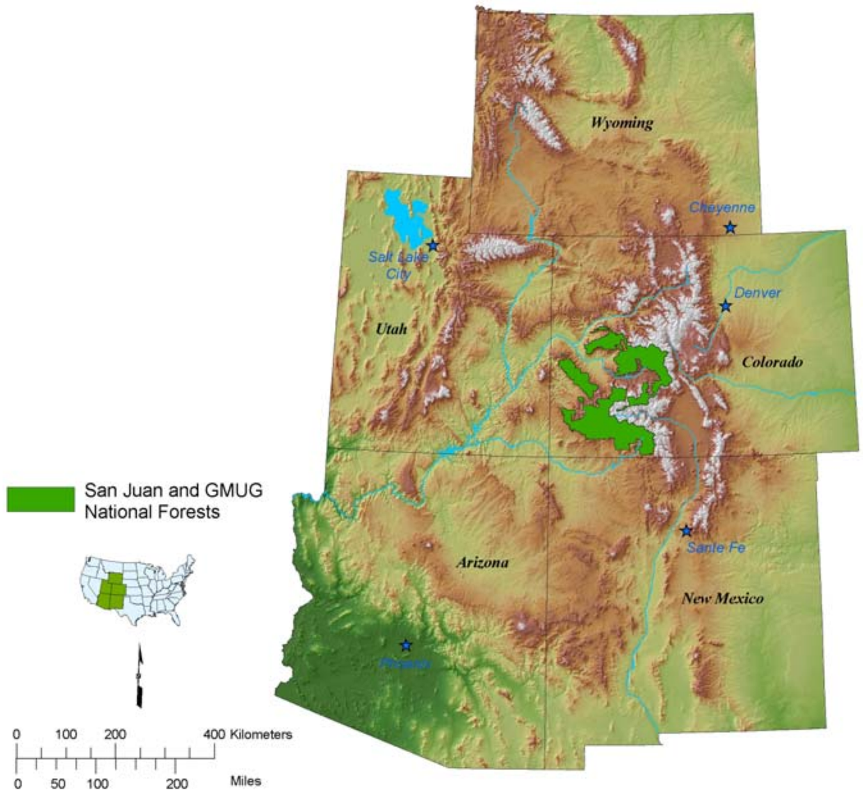


Figure 1-1 San Juan/GMUG National Forests in southwestern Colorado
The San Juan/GMUG National Forests cover about 8,195 square miles (5.2 million acres) of high desert plateau, upland and alpine areas of southwestern Colorado.

Located in the Upper Colorado River Basin, the GMUG National Forests fall along the transition between the western Colorado Plateau and the Southern Rocky Mountains physiographic provinces (Fig. 1-1).

The Species Conservation Project Assessment Process

Together, both the ARW and CLC assessments are constituents of the comprehensive Region 2 Species Conservation Project (SCP). The SCP process combines the results of Forest aquatic and terrestrial assessments in the region with species assessments to show species-ecosystem relationships and thus enhance immediate and long term species conservation efforts, both locally and regionally (Fig. 1-3). These aims will also serve to support both the credibility and legal defensibility of forest plans and project-level decisions.

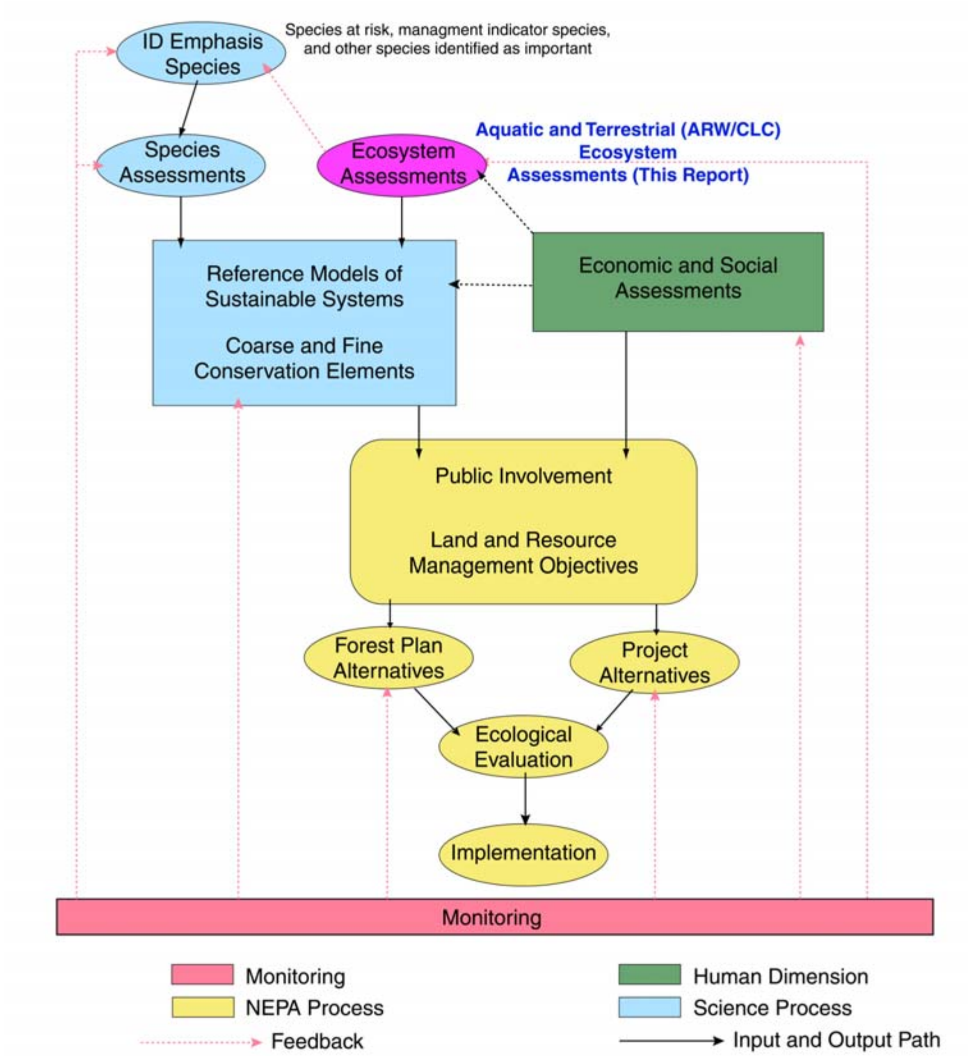


Figure 1-2 The Species Conservation Project (SCP) conceptual model.

The ARW and CLC Ecosystem Assessments presented here in this report are but one element of the overall SCP model in Region 2.

Relationship to Forest Planning

The National Forest Management Act of 1976 (90 Stat 2949 et seq; 16 U.S.C. 1601-1614 (NFMA), 1976) provides the basis for the development of the SCP process, its supporting elements and integration into both forest and project planning. The Act, in part, requires the Forest Service to "*…provide for diversity of plant and animal communities based on the suitability and capability of the specific land area in order to meet overall multiple-use objectives.*" In addition *"…fish and wildlife habitat shall be managed to maintain viable populations of existing native and desired non-native vertebrate species in the planning area."* The assessment process provides an approach that allows us to measure, evaluate and interpret both natural processes and human influences that support these goals. The resulting assessment documents are not planning documents because they do not resolve issues or determine policy (Jensen et al. 2001). Instead, the ARW and CLC assessments presented here are intended to contribute to the resolution of issues and provide a foundation for policy discussion and determination.

Thus, in the SCP process ARW and CLC assessments for SJ and GMUG will include:

1) Summaries of existing condition
2) Identification of important and unique habitat(s) that may influence the development of plan alternatives
3) Identification of risks and sensitivity of watersheds and vegetation communities
4) Delineate the distribution of species, habitat and communities from regional ecosystem perspectives.
5) Identification of areas suitable or critical for the maintenance and/or improvement of rare habitat and communities

These assessments do not:

1) Quantify the condition of plants and animal communities at a local or site level scale
2) Identify the thresholds for impacts
3) Provide results suitable to application at local or site-level project scales
4) Make changes in land allocation as specified by existing forest plans and area resource plans.
5) Serve as a decision document

Relationship to Program and Project Development

The assessments presented here may further amplify Forests activities by creation of a multi-scale approach that directly assists with prioritization of ecological land units including watersheds, vegetative communities and geomorphic settings.

Aquatic ARW assessments can:

1) Identify the highest priority watersheds for restoration

2) Identify reference watersheds and conditions to support monitoring
3) Characterize relative impacts to important resource values.
4) Assist in the development of funding requirements at the watershed level
5) Identify potential watersheds at risk and sensitive watersheds, and critical aquatic, riparian and wetland areas suitable for *program level* planning and activity

Terrestrial CLC assessments can:

1) Provide common baseline information of terrestrial vegetation dynamics and conditions
2) Provide a scientific basis for discussions with community leaders at local, state and federal levels regarding ecological processes and how they influence the landscape
3) Increase the effectiveness of the Accelerated Watershed Restoration Program (AWRP) by:
    - Improving fire risk classifications.
    - Providing a sound basis for management activity prioritization
    - Enhancing understanding of native disturbance processes that affect ecological processes
    - Providing scientific basis for planning
4) Provide multi-scale baseline information for project planning and provide mechanism to prioritize restoration work on the Forest

The ARW and CLC Assessments

Ecologists and land managers in Region 2 have developed protocols to guide the preparation of both ARW and CLC assessments that are elements of SCP. (Winters et al. 2004 and Regan et al 2004). These flexible protocols describe the structure of assessments and their goals.

ARW assessments are designed to characterize the influence of current and historic management activities on aquatic, riparian and wetland (ARW) ecosystems that include and extend beyond Forest Service administrative areas. These ARW assessments relate anthropogenic influences to specific variables that drive the function of aquatic, riparian and wetland systems. It provides managers with important insights into the sensitivity of ecosystem components to both natural and human disturbances. It reveals areas of opportunity and areas at risk. In particular the assessment addresses five key questions:

1) Where are the highest concentrations of wetlands and riparian areas?
2) What is the range of sensitivity of aquatic ecosystem components to anthropogenic influences?
3) What is the suitability of watersheds for species reintroduction?
4) What watersheds are important for fisheries and aquatic production?
5) What watersheds are most sensitive and at greatest risk to sediment production?

The GMUG National Forest has augmented the ARW assessment by applying a watershed sensitivity rating to the analysis. The watershed sensitivity rating is intended to supplement the ecological driver analysis completed under the official ARW assessment protocols. Ecological drivers are environmental factors that exert a major influence on the fitness of individual organisms and their populations, and help describe the physio-chemical template of an ecosystem (See Ecological Driver portion of this report). The watershed sensitivity rating identifies physical factors (geology, precipitation, soils) that determine how a watershed responds to disturbance (natural or management related). The sensitivity rating will be used in Forest and project planning to identify potential risks land management activities may pose to watershed health.

Current Landscape Condition (CLC) assessments examine current social, physical, biological and disturbance settings for a given area. These characteristics may be measured against historic settings to understand the influence of current and historic anthropogenic activities and their influence on terrestrial ecosystems. The measures also provide insight into opportunities for ecosystem maintenance and restoration.

CLC assessments are divided into four chapters:

- Chapter 1 provides an introduction describing concepts, setting and key components.
- Chapter 2 describes the geographic, physiographic and ecological settings, the assessment approach and scale.
- Chapter 3 provides an overview of assessment modules, assessment of specific ecological factors along with descriptions of data sources and analytical methods.
- Chapter 4 describes implementation criteria, interagency involvement, data management and timelines.

The GMUG Geographic Area Assessments

The GMUG current forest plan revision process includes the development of Geographic Area Assessments (GAA). These assessments cover five geographic areas (Fig. 1-4). The GAA are included in this report are an adjunct to the SCP process. Integration of GAA into the assessment process is intended to ensure forest plan revision and SCP efforts are complementary and well integrated.

GAAs describe the current conditions of lands and resources within each area and include a comparison of these conditions to desirable conditions.

GAAs will:

- Define key issues to focus the analysis
- Describe current conditions relating to key issues
- Outline historic conditions to help identify the type of changes
- Outline important trend and likely future conditions
- Synthesize and interpret information
- Define opportunities and include recommendations

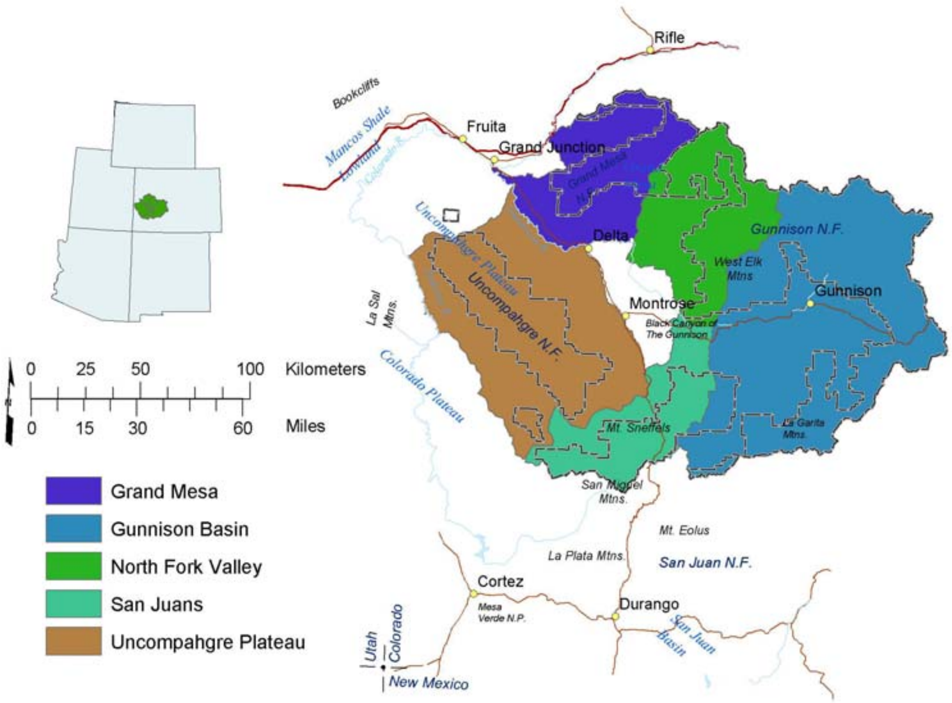


Figure 1-3 GMUG forest plan revision Geographic Areas Assessments (GAA).

Integration of ARW, CLC and GAA Elements

In some cases the ARW, CLC and GAA protocols consider ecological characteristics common to both aquatic and terrestrial ecosystems. Where these characteristics are common, then a single umbrella document has been prepared. In other instances, a given protocol may call for the reporting of characteristics that are distinct to that protocol and thus will be dealt with separately.

The ARW, CLC and GAA Multi-scale Geographic Setting

Both the ARW and CLC assessments are organized by geographic scale. Concepts of scale provide important variation in perspective and understanding of ecosystem function by following a multi-scale approach that characterizes local to regional ecological settings. The ARW protocol is based on a hierarchical arrangement of hydrologic units and the CLC protocol is based on a hierarchical arrangement of land cover (vegetation) based ecological units. These two scale frameworks are complimentary (Maxwell et al. 1995).

The ARW Scale Framework

The ARW protocol follows concepts defined by the National Hierarchical Framework of Aquatic Ecological Units in North America (Maxwell et al. 1995) and the National Watershed Boundary Dataset's Federal Standards for Delineation of Hydrologic Unit Boundaries (FGDC 2002).

The aquatic ecological unit hierarchy delineates aquatic ecosystems into seven hierarchical categories including: subzones, regions, subregions, basins, subbasins, watersheds and subwatersheds. The National Watershed Boundary Dataset defines hydrologic unit boundaries that may be adapted to fit the National Hierarchical Framework by defining four hierarchical scales. These four scale categories include, in descending order: basin, landscape, management and reach scales (Fig.1-5).

Scale	Summary Units	Biotic Processes	Spatial Resolution
Basin	*Landscape*	*Species Association*	*1000's of Sq. Miles*
Landscape	*4th Level HUB*	*Metapopulations*	*100's of Sq. Miles*
Management	*6th Level HUB*	*Populations*	*1000's of Acres*
Reach	*Site(s)*	*Individuals*	*Feet, Miles Acres*

Figure 1-4 The watershed-based multi-scale configuration for ARW assessments.

The basin scale is defined by major river drainages, including the Green and Colorado Rivers. The landscape scale provides regional context for analysis and the management scale sharpens the focus to local watersheds. Site level activity is applied at the reach scale.

Basin Scale

The San Juan and GMUG National Forests fall completely within the approximately 114,000 square miles of the Upper Colorado River Basin (Steeves and Nebert, 1994) (Fig 1-6). Principal rivers in that drain the basin include the Colorado, Green, Gunnison, San Juan and Dolores Rivers and a small portion of the Rio Grande Basin. Finer scales are important to measure anthropogenic influences, habitat quality and management opportunities.

Figure 1-5 San Juan/GMUG National Forests in the Upper Colorado River Basin.
The San Juan/GMUG National Forests are located completely within the approximately 114,000 square miles of the Upper Colorado River Basin.
The area comprises the basin scale.

Landscape Scale

The landscape scale is the aggregation of several 4th level hydrologic units (Figure 1-7), or sub-basins that intersect the San Juan and GMUG National Forests. Analysis at this scale considers the magnitude of anthropogenic influences, summarized to each of the eighteen 4th level HUBs that comprise the landscape (Table 1-2). In total, the area covers about 22,258 square miles of high desert, dry forests and alpine uplands of Western Colorado and Eastern Utah. Principal rivers that drain these lands include the Gunnison, San Juan, Uncompahgre, San Miguel, Mancos, Animas, Piedra and Dolores Rivers. Each drains ultimately into the Colorado River (Fig. 1-7).

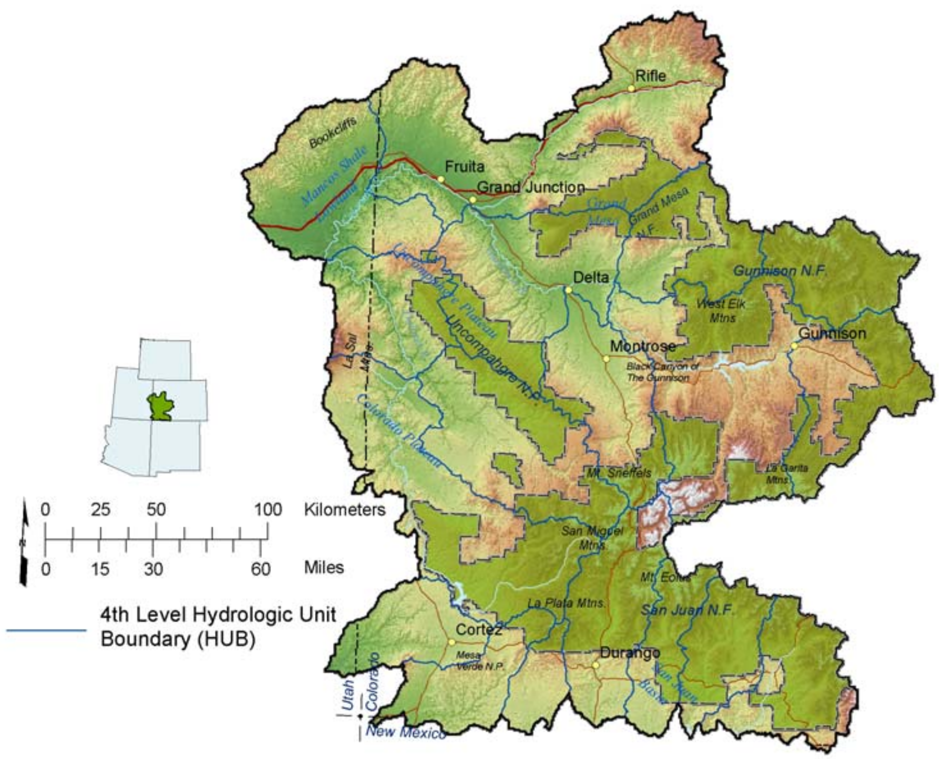


Figure 1-6 ARW landscape scale for the San Juan/GMUG National Forests. The ARW landscape scale for the San Juan/GMUG National Forests covers about 22,258 square miles (14,245,187 acres). The area is defined by aggregating eighteen 4th level watersheds that intersect the National Forests' administrative areas.

Table 1-2 Eighteen 4th level HUBs comprise the San Juan/GMUG ARW Landscape Scale.

Rank	HUB4	HUB4 Name	Acres	Sq. Miles	Percent
1	14010005	Colorado Headwaters-Plateau	1,998,348	3,122	14.0%
2	14020002	Upper Gunnison	1,543,036	2,411	10.8%
3	14030002	Upper Dolores	1,381,647	2,159	9.7%
4	14020005	Lower Gunnison	1,064,086	1,663	7.5%
5	14030003	San Miguel	995,742	1,556	7.0%
6	14030001	Westwater Canyon	933,861	1,459	6.6%
7	14020006	Uncompahgre	714,738	1,117	5.0%
8	14020003	Tomichi	705,059	1,102	4.9%
9	14080104	Animas. Colorado	702,036	1,097	4.9%
10	14020004	North Fork Gunnison	620,473	969	4.4%
11	14030004	Lower Dolores	591,992	925	4.2%
12	14080101	Upper San Juan	583,052	911	4.1%
13	14020001	East-Taylor	490,726	767	3.4%
14	14080202	Mcelmo	459,777	718	3.2%
15	14080107	Mancos	448,023	700	3.1%
16	14080102	Piedra	432,475	676	3.0%
17	14080101	Upper San Juan	370,102	578	2.6%
18	14080105	Middle San Juan	210,014	328	1.5%
			14,245,187	22,258	100.0%

Management Scale

The ARW management scale is based on 6th level sub-watersheds (HUBs). The gross management scale area is defined by the collection of 6th level watersheds, some portion of which fall within the Forests. These areas thus defined embrace watersheds along the margins of these National Forests and include all watersheds within the Forests.

Three hundred eighty-one (381) 6th level HUBs intersect or are adjacent to HUBs intersecting the Forests (Table 1-3 and Fig. 1-8). These 381 comprise the management scale HUBS for the Forests. These 381 HUBS range in size from a maximum area of 109,340 acres (170.8 sq. miles) to a minimum area of 1,736 acres (2.7 sq. miles). The average HUB area is 21,875 acres (34.2 sq. miles).

In cases where data simply do not exist, assessment team personnel will create or obtain the data from external sources. Where this is not possible, important data gaps are documented in the assessment.

Table 1-3 The 381 HUBs that define the ARW Management scale include HUBs distinct to each Forest, three common to each and one external HUB.

The HUB 140801010501 (East Fork Navajo River) falls between the east-most edge of the San Juan Forest and the continental divide.

Forest	Count	Acres	Hectares	Sq. Miles
San Juan	151	2,781,672	1,125,703	4,346
GMUG	226	5,431,815	2,198,177	8,487
Common To Both	3	107,822	43,634	168
External But Included	1	13,268	5,369	21
	381	8,334,577	3,372,883	13,023

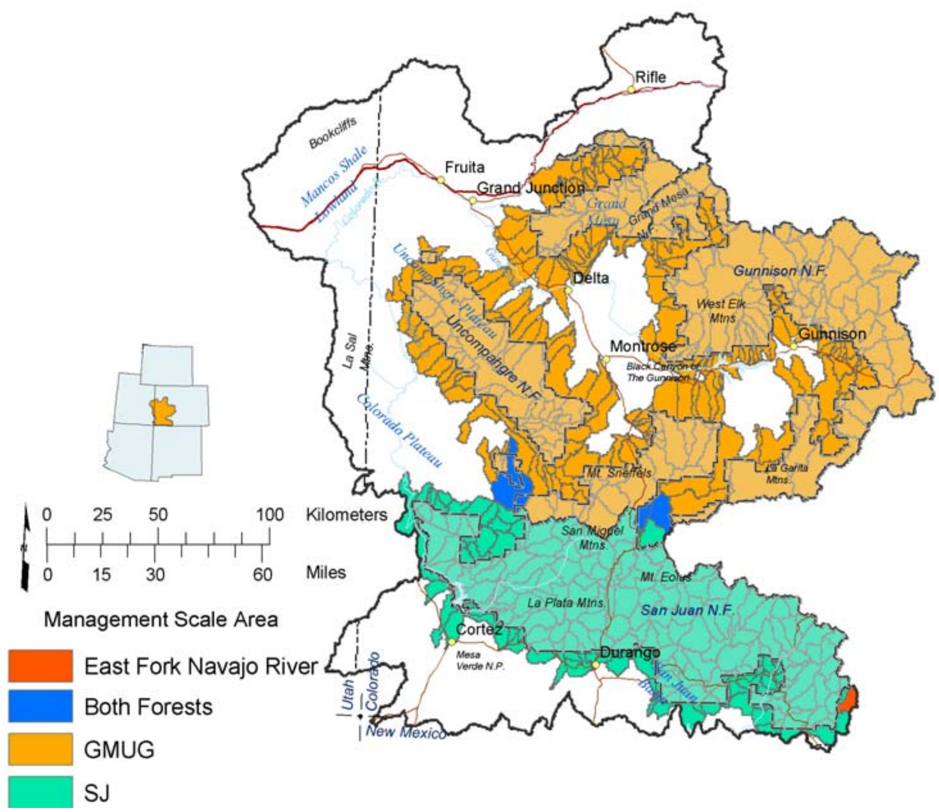


Figure 1-7 381 6th level HUBs make up the combined the San Juan/GMUG ARW management scale area.

Reach Scale

For the most part, reach scale data has not been included in this assessment and is beyond the scope of the analysis. Stream channel habitat and fish population data for selected streams on the GMUG are described in the Biological portion of the Anthropogenic influences section of this report.

The CLC Scale Framework

The CLC assessment protocol also uses a multi-scale hierarchical analysis framework of ecological units (ECOMAP, 1993). The ECOMAP mapping framework was designed to assist with forest-level analysis and planning (Bailey, 2004). This framework defines ecological units based on biotic and environmental factors that affect or express energy, moisture and nutrient gradients that regulate the structure and function of ecosystems. The descending hierarchy of region, sub-region, landscape and land-unit define the CLC scale framework applied in this assessment. These scales roughly correspond to the basin, landscape, management and reach scales described above in the ARW scale framework. The CLC landscape scale should not be confused with the ARW landscape scale. Figure 1-9 illustrates the hierarchical arrangement of scales that define CLC scale framework.

Scale	Summary Units	Biotic Processes	Spatial Resolution
Regional	*Province*	*Complexes*	*1000's of Sq. Miles*
Subregion	*Subsections*	*PNC*	*100's of Sq. Miles*
Landscape	*LTA/Forest*	*Plant Assoc.*	*1000's of Acres*
Landunit	*Patch*	*Individuals*	*Feet, Miles Acres*

Figure 1-8 The ECOMAP based hierarchical scale arrangement for the San Juan and GMUG CLC assessment.

The regional scale includes the Colorado Plateau and the Southern Rocky Mountains provinces. The subregion scale provides regional context for analysis and the landscape scale sharpens the focus to analysis at the forest level (similar in scope to the ARW management scale). Site level activity is applied at the landunit scale.

Regional Scale

Provinces are similar in scope to the basin scale of the ARW protocol and they provide further subdivision below province (Fig. 1-10). Provinces typically cover areas from 1,000 to 10,000 square miles.

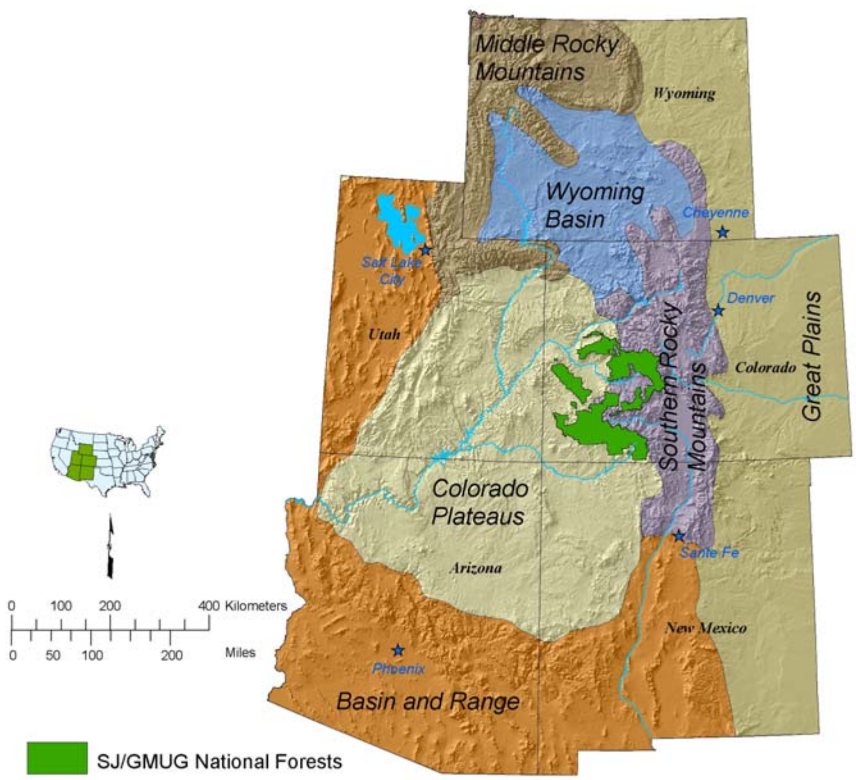


Figure 1-9 The San Juan and GMUG National Forests are included in the Colorado Plateau and Southern Rocky Mountains ECOMAP (1993) provinces.

Subregion Scale

The CLC subregion for the Forests is based on ECOMAP sections that intersect these forests and some contiguous sections that do not. The aim is to build an ecosystem container similar in scale and utility to the ARW Landscape scale.

Sections are the first terrestrial scale below the ECOMAP province and typically cover areas up to about 1000 square miles. They are described by characteristic geomorphology, geology, climate, soils, potential natural vegetation and potential natural communities. Forest management and other anthropogenic activities along with natural disturbance can affect the character and function of sections.

Six ECOMAP sections intersect the San Juan and GMUG National Forests (Fig 1-11). ECOMAP subsections are defined by the characteristics of geomorphology, geology and potential communities as sections, but subsections as ecological entities, are more responsive to changes in climate, soils, vegetative and animal community than sections. The area spanned by these six sections is more than adequate to the needs of most wide-ranging terrestrial species and brings into focus the larger complex of dry to wet vegetation communities of the Colorado Plateau and Southern Rocky Mountains provinces.

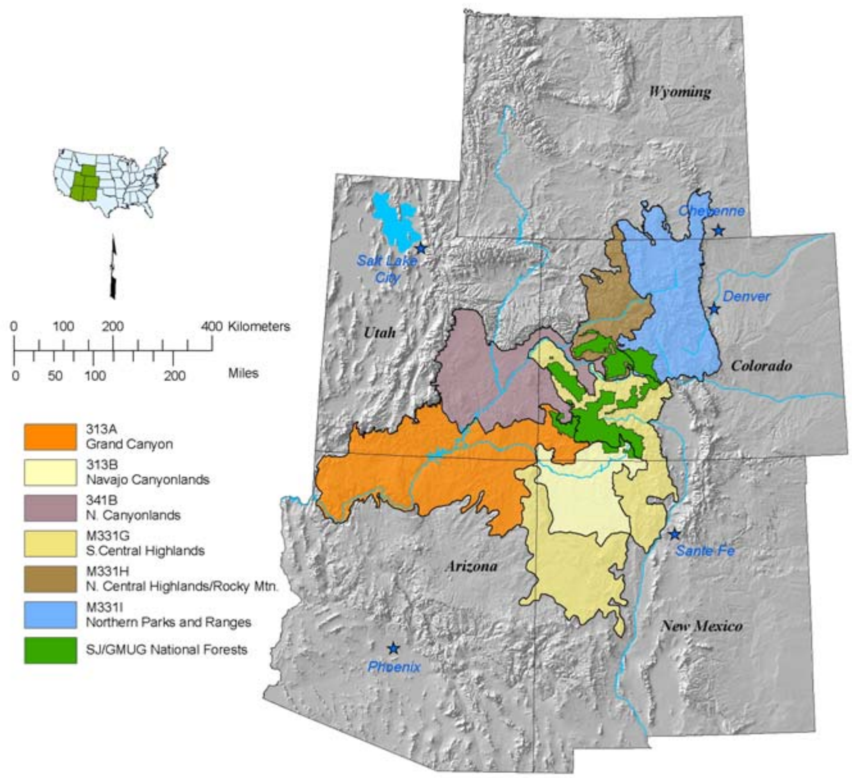


Figure 1-10 Six ECOMAP sections intersect the San Juan and GMUG National Forests.

The portions of sections far from these National Forests are minimally influenced by management on those Forests.

The broad geographic setting formed by the six sections intersecting the Forests requires some approach to trim away the portions of sections minimally influenced by Forest management and well beyond the scope of the analysis. As a consequence the assessment team aggregated relevant subsections to form a **subregion** scale. Twenty-seven subsections intersect the Forests (Fig. 1-12). These twenty-seven subsections are constituents of the sections and more directly relevant to National Forest management.

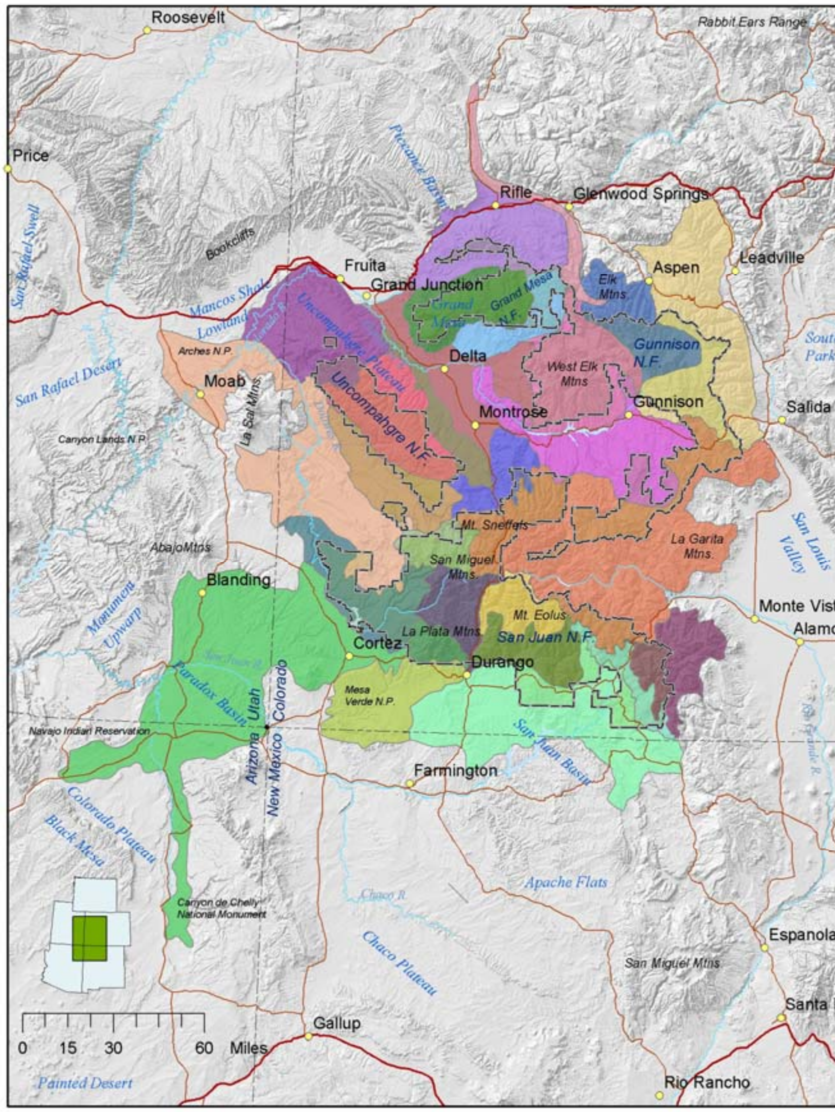


Figure 1-11 Twenty-seven subsections intersect the San Juan and GMUG National Forests.

The resulting region defined by these subsections is broad enough to provide adequate ecological context for the consideration of wide-ranging terrestrial species and ecological processes that could be influenced management of the Forests. The team added an additional nineteen subsections to ensure adequate consideration of systems that both influence the Forests and are influenced by Forest management.

Just as the aggregation of 4th level watersheds in the ARW assessment forms a landscape scale, these twenty-seven subsections plus the additional nineteen, form a **subregion** scale for this CLC assessment (Fig. 1-13 and Table 1-4).

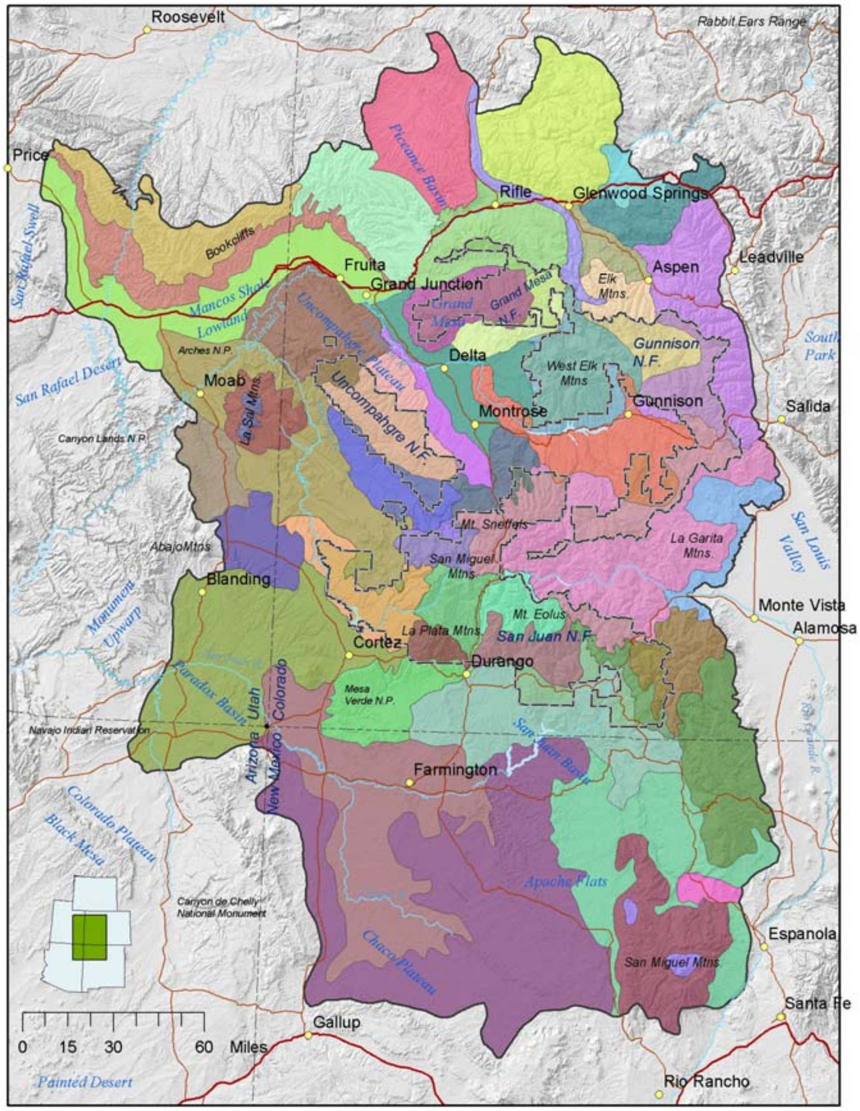


Figure 1-12 Nineteen subsections have been added to the twenty-seven subsections that intersect the San Juan and GMUG Forests.

These nineteen were added to take in important subsections and provide a comprehensive geographic context.

Table 1-4 The forty-six ECOMAP subsections aggregated to form the CLC *subregion* scale for the San Juan and GMUG CLC assessment.				
SubSection	Section Name	ACRES	HECTARES	SQMILES
313Aa	Grand Canyon	535,102	216,548	836
313Ab		1,995,409	807,513	3,118
313Ac		536,428	217,085	838
313Bc	Navajo Canyonlands	2,082,085	842,590	3,253
313Bd		3,516,363	1,423,022	5,494
313Bm		1,384,139	560,141	2,163
331J	Northern Rio Grande Basin	105,031	42,505	164
341Ba	Northern Canyon Lands	543,149	219,805	849
341Bb		46	18	0
341Bd		1,436,402	581,291	2,244
341Be		385,636	156,062	603
341Bg		229,280	92,786	358
341Bk		282,563	114,349	442
341Bl		48,827	19,760	76
341Bn		337,233	136,473	527
341Bo		1,233,952	499,363	1,928
M331Ga	South-Central Highlands	776,028	314,048	1,213
M331Gb		432,022	174,833	675
M331Gc		808,651	327,249	1,264
M331Gd		350,179	141,713	547
M331Gf		272,207	110,158	425
M331Gi		594,165	240,450	928
M331Gk		875,042	354,117	1,367
M331Gm		1,011,198	409,217	1,580
M331Gn		262,894	106,389	411
M331Go		1,749,144	707,853	2,733
M331Gq		259,061	104,838	405
M331Gr		307,765	124,548	481
M331Gt		161,396	65,314	252
M331Gu		2,270,872	918,989	3,548
M331Gv		433,767	175,539	678
M331Hd	North-Central Highlands and Rocky Mtns.	1,101,687	445,837	1,721
M331Hf		153,117	61,964	239
M331Hg		216,024	87,422	338
M331Hh		368,854	149,270	576
M331Hj		179,684	72,715	281
M331Hl		702,537	284,307	1,098
M331Hm		298,311	120,722	466
M331Hn		720,772	291,686	1,126
M331Hp		484,514	196,076	757

Table 1-4 The forty-six ECOMAP subsections aggregated to form the CLC *subregion* scale for the San Juan and GMUG CLC assessment.				
SubSection	Section Name	ACRES	HECTARES	SQMILES
M331Ik	Northern Parks and Ranges	1,219,353	493,455	1,905
M331Iw		474,395	191,981	741
M341Bb	Northern Canyonlands	761,659	308,232	1,190
M341Bc		893,921	361,757	1,397
M341Bd		699,305	282,999	1,093
M341Bg		631,752	255,661	987
		34,121,920	13,808,650	53,316

Landunit Scale

Forest planning and mangement at the Forest level requires a finer scale than subsections. Landtype Association (LTA) LTAs, typically ranging in area from 10 to 1000 acres provide a suitable summary unit for analysis at the landunit scale.

Here, in this assessment, while LTAs will be used to provide a broad context for plant associations they will not be used as a primary summary unit. Instead, landunit scale analysis on the Forests will use alternative approaches to LTAs. At the same time, LTAs will be used in the CLC assessment to define broad contexts for plant associations and communities. The San Juan Forest has defined a landunit scale equivalent referred to as the San Juan CLC landscape (Fig. 1-14). The GMUG National Forest defines their GA as a landunit scale equivalent (Fig. 1-15)

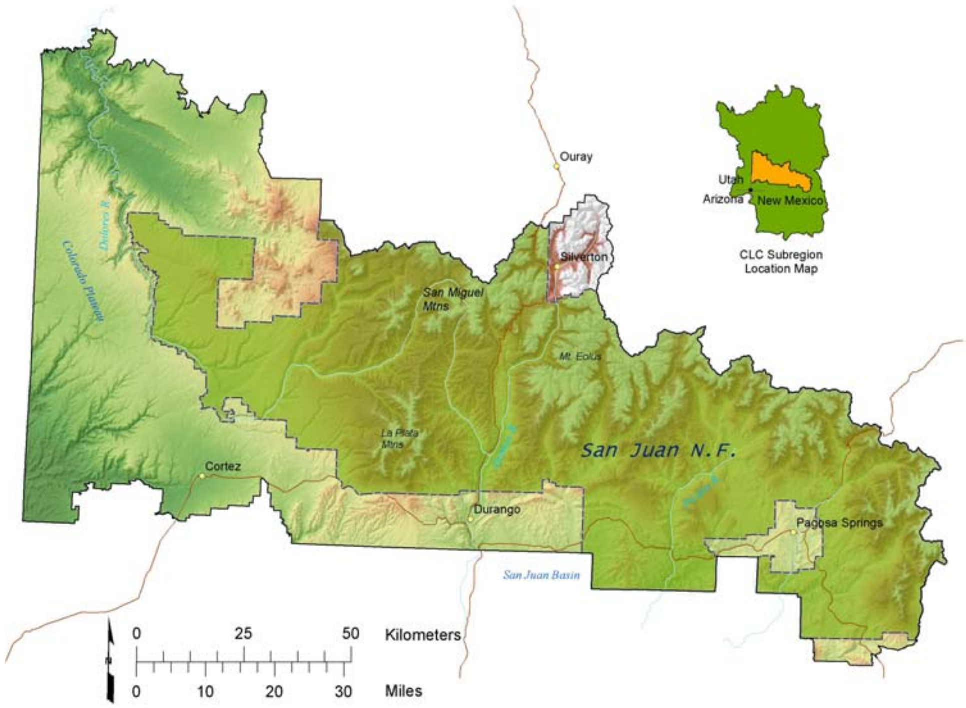


Figure 1-13 The San Juan National Forest CLC *landscape scale* area.

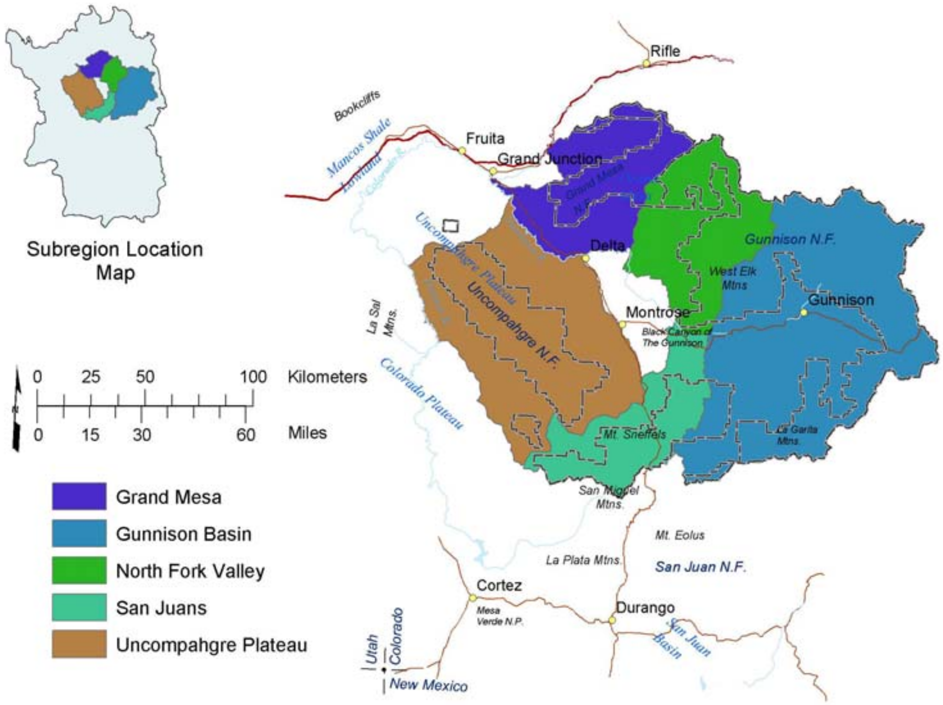


Figure 1-14 GMUG National Forest Geographic Area Scale.
The GMUG Forest will use their Geographic Area Assessment to define the landscape scale used in their terrestrial analysis.

Stand Scale

The stand scale is a fine scale level beyond the scope of this CLC assessment.

GMUG Geographic Area Assessment Framework.

Geographic Area Assessment (GAA) on the GMUG is the link between the broader-scale forest assessment and project-level analysis. Under the CLC portion of the assessment, current vegetation conditions, wildlife habitat structural stages, Potential Natural Vegetation (PNV), and natural and management influences on vegetation will be described. Natural disturbance regimes and management influences will be described by dominate cover type in the GA and then related to how those influences are affecting current vegetative and wildlife habitat conditions and trends in the future. Based upon current conditions and future trends, potential effects to various wildlife species dependent upon those habitats will be completed.

Combined Assessments Geographic Framework

The assessments Greater Study Area is defined by combining the ARW landscape scale with the CLC subregion scale (Fig. 1-16). The Greater Study Area extends from the Painted Desert of northeastern Arizona to the Rabbit Ears Range of westcentral Colorado. The area is about 275 miles wide and 350 miles long.

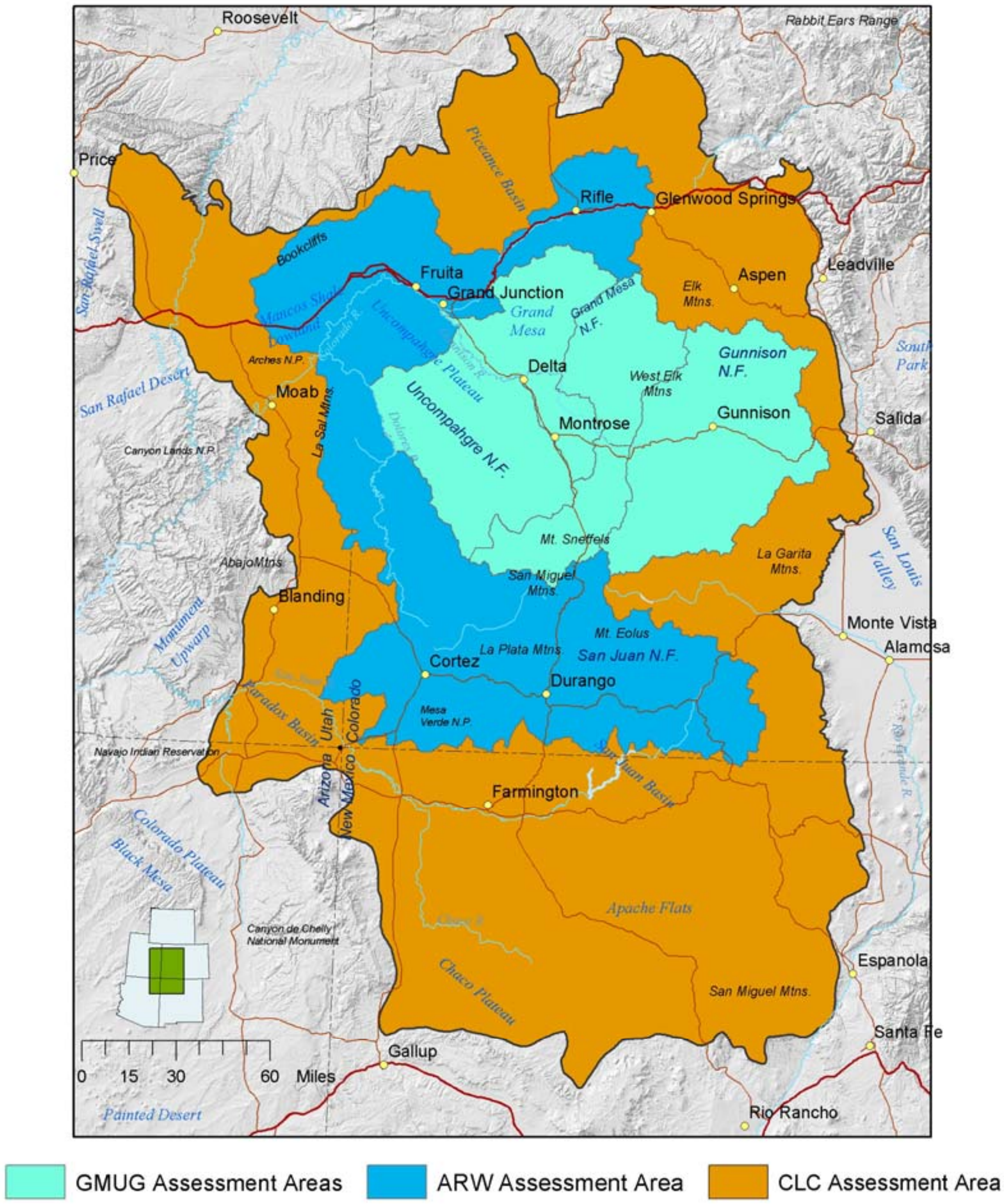


Figure 1-15 The Greater Study Area.

Bibliography

ECOMAP. 1993. National hierarchical framework of ecological units. Washington, DC: USDA Forest Service. 20 p.

Fenneman, N.M., and Johnson, D.W. 1946. Physical divisions of the United States. Washington, DC: U.S. Geological Survey Online: http://water.usgs.gov/lookup/getspatial?physio

Federal Geographic Data Committee (FGDC). 2002. Federal standards for delineation of hydrologic unit boundaries, FGDC proposal, version 1.0. Washington, D.C.

Frissell, C.A.; Liss, W.J.; Warren, C.E.; Hurley, M.D. 1986. A hierarchical framework for stream habitat classification: Viewing streams in a watershed context. Environmental Management. 10(2):199-214.

Jensen, M.E., I.A. Goodman, C.A. Frissell, C.K. Brewer, and P.S. Bourgeron. 2001. Ecological classification and mapping of aquatic ecosystems. In: A guidebook for integrated ecological assessments. Edited by: M.E. Jensen and P.S. Bourgeron. Springer Verlag, New York, NY.

Maxwell, J.R.; Edwards, C. J.; Jensen, M. E. 1995. A hierarchical framework of aquatic ecological units in North America (nearctic zone). General Technical Report. NC-176. St. Paul MN: U.S. Department of Agriculture, Forest Service, North-Central Forest Experimental Station. 72p.

McNab, W. Henry and Peter E. Avers. 1994. Ecological Subregions of the United States. WO-WSA-5. U.S. Department of Agriculture, Forest Service. Online: http://www.fs.fed.us/land/pubs/ecoregions/

Regan, Claudia; Dean Erhard; Greg Hayward; Dan Kashianl; Sigrid Resh and Jennifer Ross. 2004. Protocol for Developing Terrestrial Ecosystem Current Landscape Condition Assessments. U.S. Department of Agriculture, Forest Service, Region 2, Lakewood, CO.

Steeves, Peter and Douglas Nebert. 1994. Hydrologic units maps of the Conterminous United States. Reston, Virginia: U.S. Geological Survey Online: http://water.usgs.gov/lookup/getspatial?huc250k

Winters, David; Bryce Bohn; David Cooper; Greg Eaglin; Jeffrey Hamerlinck; Christine Hirsch; N. LeRoy Poff; Charles Quimby; Frank Rahel; Paul Rau; Dan Scaife; Dennis Staley and Ellen Wohl. 2004. Conceptual Framework and Protocols for Conducting Multiple Scale Aquatic, Riparian and Wetland Ecological Assessments. U.S. Department of Agriculture, Forest Service, Region 2, Lakewood, CO.

Chapter 2. Ecological and Socio-Economic Context of the Assessment Area

Socioeconomic and Anthropogenic Influences

Historical Context

Human occupation in the assessment area began as early as the late Pleistocene and early history includes paleo-indian, archaic, Anasazi and Ute cultures. The Anasazi occupied the area from about 1 A.D. to 1300 A.D. followed by the Ute culture from 1300 A.D. to present. Remains of the Anasazi people still influence local character and economy. Mesa Verde, Chaco Canyon and other important sites draw visitors and are subject of continued research (Blair, 1996). With settlement, foraging, introduced fire and agriculture by these ancients, the threads of anthropogenic influence were introduced into the landscape tapestry in the assessment area.

The importance and depth of human activity follow an ever increasing trajectory through subsequent phases of discovery, claim and use. This trajectory is punctuated by five recognized periods. The Pre-European Settlement and Migration Era of the Anasazi was followed by the European Settlement Era, the Conservation Era, the Post-World War II Energy Development Era and The Era of Tourism/Amenity Migration (Preston, 2004).

During the European Settlement Era, the important elements of the current land jurisdiction/ownership pattern took shape. Broad tracts of privately held low valley bottoms reflect settlement patterns driven by homesteading and land grants. Settlement and community development in this arid country, overall, correlates to access to water. Mining led to the settlement of upland communities like Telluride, Rico and Silverton. Patented mining claims proliferate in the upland areas and now represent substantial private in-holdings. The creation of reservations and the relocation of indigenous peoples dictate broad pattern, culture and legacy.

Agriculture, livestock management, timber harvests, mineral extraction and road building during this Era strongly influenced the character and ecology of the assessment area. By both ignorance and careless abuse, significant loss of soil and resource waste occurred in this period. These abuses along with other important changes in the national culture gave rise to the Conservation Era and the development of, in the words of Theodore Roosevelt: "…a New Nationalism that puts the national need before sectional or personal advantage." (Cotner et al., 1957).

During the Conservation Era, large tracts of public land, not yet under homestead, mining or other valid claim were reserved by the Federal Government and came under the management of principally the U.S. Forest Service, the Bureau of Land Management and the National Park Service. Large scale water projects were developed by the Bureau of Reclamation. By the middle of the 20th century, the land use and ownership pattern of the region was generally fixed (Fig 2-1).

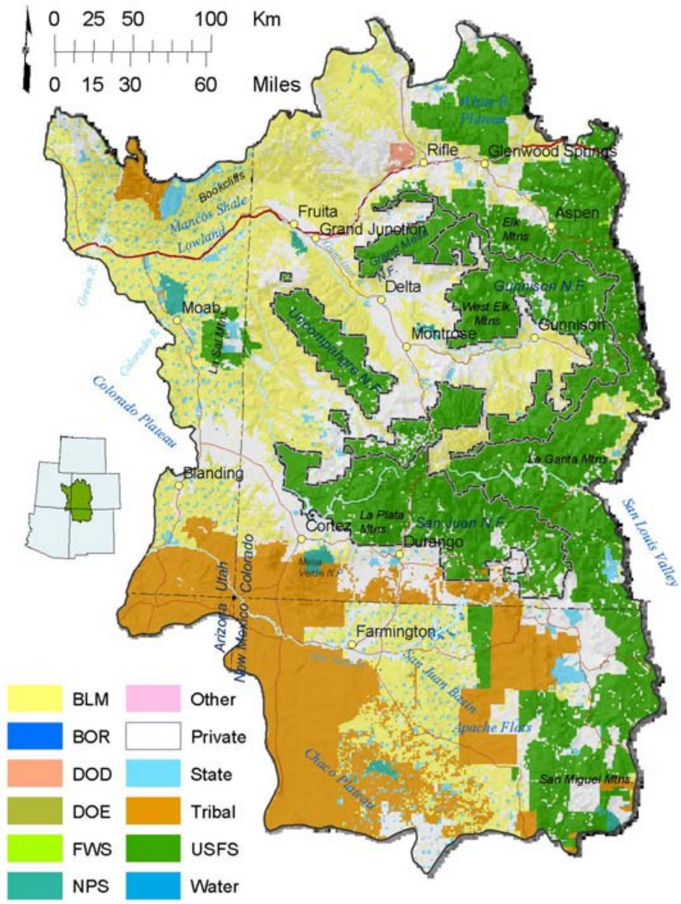


Figure 2-1 Jurisdiction/Land Ownership in the assessment area.

Private land pattern, established by homesteading and grants and mining claim, was established largely in the late 19th century. Large tracts of Federal land including the National Forests, National Parks, the Bureau of Land Management and Bureau of Reclamation were set aside during the Conservation Era of the early 20th century. Indian Reservations were established over the periods of settlement and conservation.

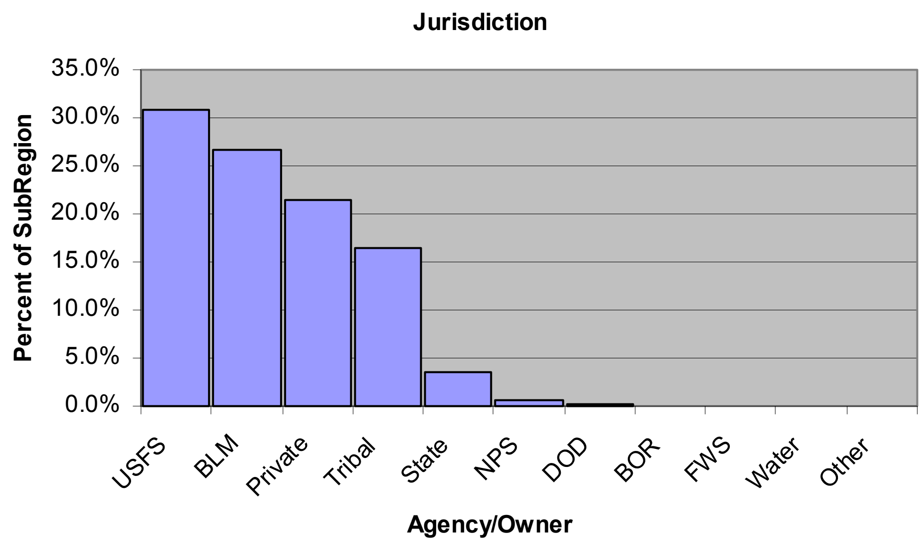
While many of the most severe effects settlement and land-use were mitigated during the Conservation Era, the prevailing land ethic remained oriented toward use and development. While perhaps more orderly and systematic, road building, timber harvests, grazing and mining continued throughout the Era and in some cases increased sharply with rises in local and national demand, especially during periods of world war.

Table 2-1 Ownership/Jurisdiction area table.

Jurisdiction	Acres	Hectares	Sq Miles	Pct	Sum Pct
USFS	10,515,806	4,255,596	16,431	30.8%	30.8%
BLM	9,123,884	3,692,304	14,256	26.7%	57.6%

Private	7,338,073	2,969,613	11,466	21.5%	79.1%
Tribal	5,646,754	2,285,160	8,823	16.6%	95.6%
State	1,175,076	475,536	1,836	3.4%	99.1%
NPS	229,352	92,815	358	0.7%	99.8%
DOD	55,547	22,479	87	0.2%	99.9%
BOR	25,807	10,444	40	0.1%	100.0%
FWS	2,981	1,206	5	0.0%	100.0%
Water	248	100	0	0.0%	100.0%
Other	32	13	0	0.0%	100.0%
	34,113,560	13,805,267	53,302	100.0%	

Figure 2-2 Ownership/Jurisdiction proportions.



After World War II, exploration and development of energy resources in the assessment area sharply increased. Oil, gas and uranium development on the eastern margins of the Colorado Plateau during the 1950's contributed to a booming economy in the region especially affecting Farmington, Cortez and Durango. Development of fields, mines, pipelines, roads and infrastructure along with increased settlement and community development placed increased demands on aquatic and terrestrial systems. Development of energy resources on the Southern Ute, Ute Mountain and Navajo Reservations strongly influenced the economies of tribal communities. Increases in local demand for wood product along with increased export opportunities promoted continued and expanding timber harvests. Interest in recreational opportunities on public lands in the region began to rise at this time.

A bust in the energy development economy punctuates the transition from the Energy Development Era to the current era of Tourism/Amenity Migration. The era marks an import shift in settlement pattern and demographics. It also marks a time of important transition. It marks a shift from a development orientation on public lands to expanded recreation and a greater focus ecological restoration.

Hecox and Ack (1996) characterize the transition stating that:

The Colorado Plateau is undergoing a profound economic and demographic transformation. A region settled by pioneers with an economy built largely upon natural resource commodity production is being transformed to a new western frontier – one being re-settled by a new type of pioneer, fleeing urban decay and traditional economic options. The economy is increasingly driven by service-based enterprises, led by tourism and recreation, but encompassing financial, legal and health services, retail trade, construction and technical professions, among others. The plateau's population is growing rapidly, as is the surrounding region – five of the country's fastest-growing metropolitan areas ring the plateau.

Importantly, transition in the regional economy, depending on amenities that draw tourism, recreation and modern settlers will then continue to depend on accessibility to public lands and services. Patterns of in-migration by professionals and retirees draw upon a shrinking pool of private lands. Moreover, management constraints and opportunities are strongly governed by an ownership pattern that is the legacy of historical patterns of settlement and development over time.

See http://ocs.fortlewis.edu/forestPlan/reports/Productive_Harmony_Analysis_7-25-05.pdf for complete report: **Productive Harmony Analysis Interpretive Framework for Social and Economic Assessment of Southwest Colorado Communities and San Juan Public Lands Michael Preston, Office of Community Services, Fort Lewis College 7/25/2005.**

Land Ownership Pattern

As mentioned above, in the subregion, the most significant land ownership patterns reflects patterns of land selection during the European Settlement Era and public withdrawal during the Conservation Era. During the Settlement Era, farmers, ranchers and those founding communities would naturally select gently sloping lowland areas lands. Consequently, the best of these lands were settled before withdrawal to form blocks of Forest, BLM and other Federal Lands later in the Conservation Era. The pattern is evident in Fig. 2-x, correlating slope to ownership in the greater assessment area.

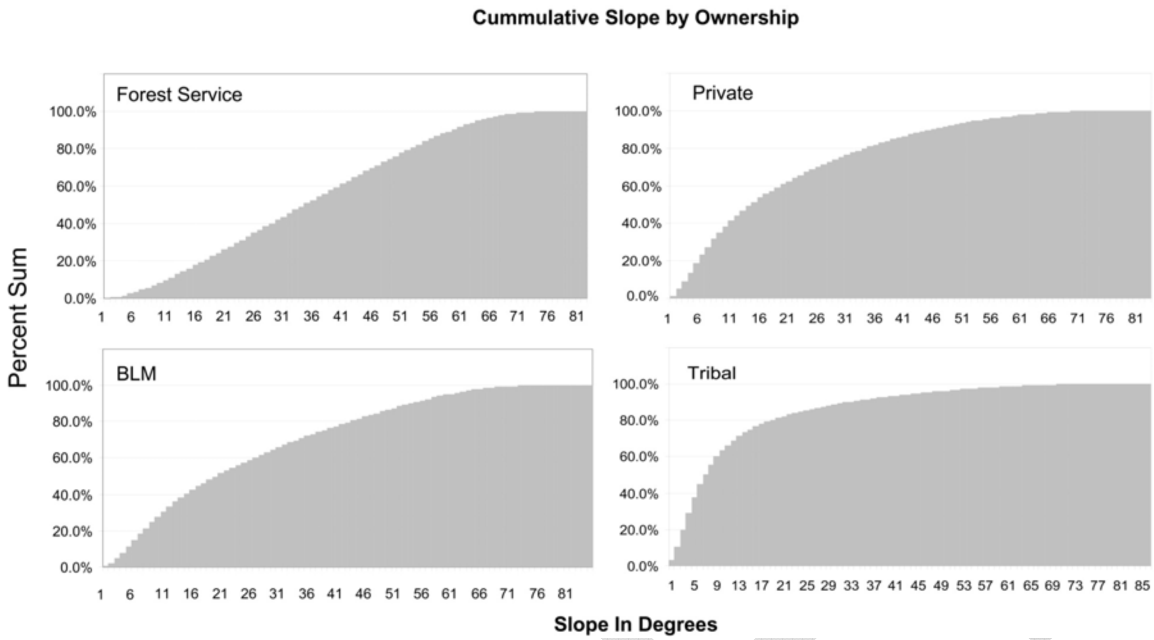


Figure 2-3 Slope on Forest Service lands.
Slope on Forest Service lands tends to be significantly higher than BLM, private and Tribal lands.

These steeper slopes are characteristic of the upland settings for Forest Service lands. The shallower slopes on private lands are indicative of the historic choice of valley bottoms, for settlement, housing and agriculture. Lower slopes on Tribal lands are indicative of selection criteria for reservations that included broad inland areas away from basin margins where foothills provide water.

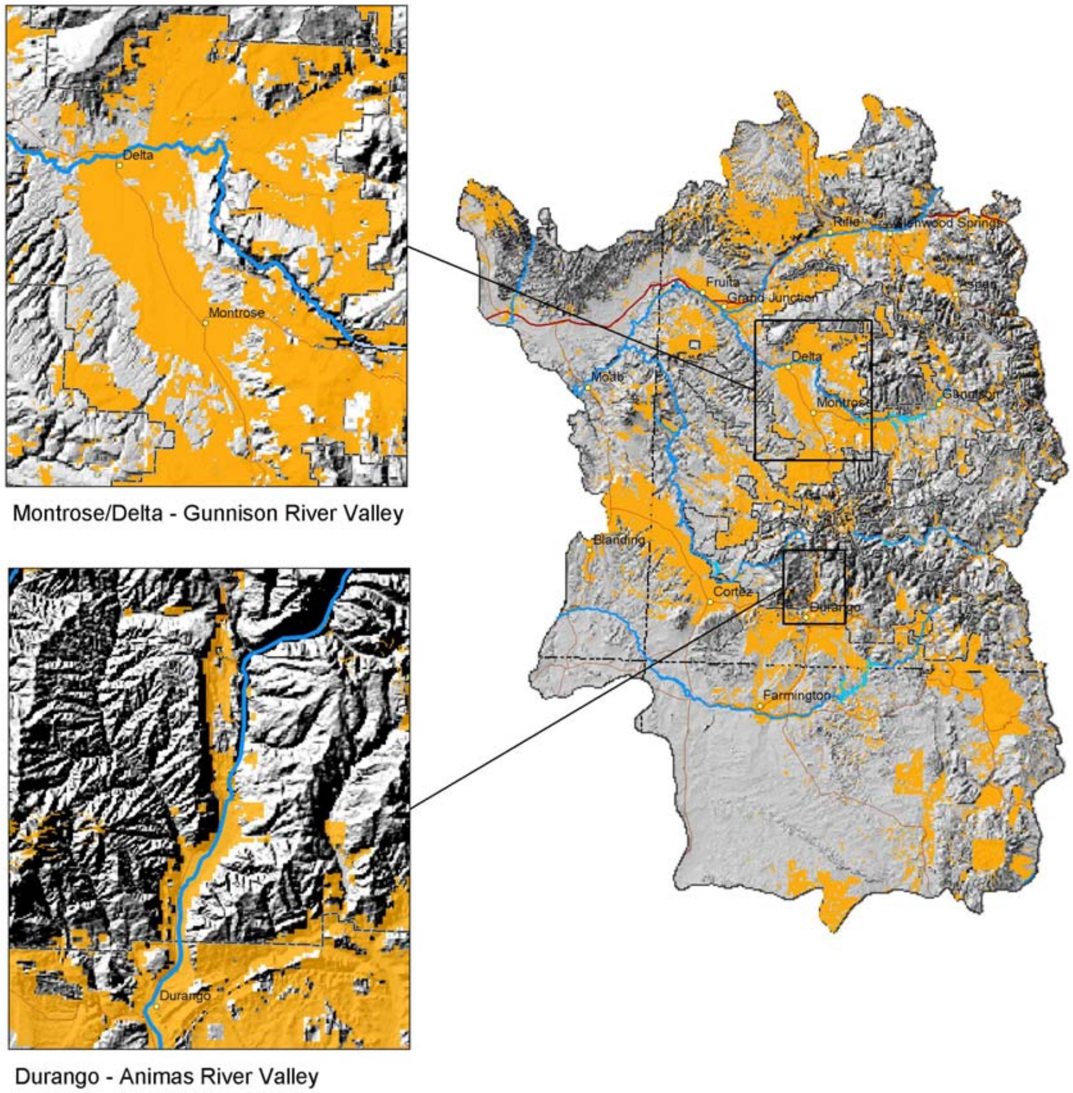


Figure 2-4 Private lands shown in orange superimposed on shaded relief.

Private lands pattern generally follow lowland and valley settings. The inset maps illustrate this.

Confinement of private lands, such as in the Animas River Valley, North of Durango constrains settlement in desirable regions and causes prices to rise.

The availability of private lands in the subregion is a key characteristic defining the shape of the newly emerging economy. Many of the more attractive and pleasing valleys in the subregion are rapidly filling to capacity with homes while room to expand is sharply limited by enclosing public lands (Fig. 2-x). The result is an economic shortage in supply with a corresponding rise in prices. As a consequence, communities are often stratified. The more desirable areas are the most expensive and tend to be occupied by service consumers. Service providers, earning lower wages tend to form communities in perhaps somewhat less desirable areas but with greater abundance of private lands. Moreover, increasing land prices in concert with increasing costs encourage both farmers and ranchers to sell their lands. This ongoing conversion of agricultural lands to residential uses strongly influences changes in the local economy and the character of land uses and expectation.

These changes thus alter the demands placed on public lands and their management. Fire managers now must be concerned with protection of homes in the public lands/urban interface occupying lands that were once largely open and agricultural.   Moreover, increased occupancy increases the risk that fires will start on private lands and propagate into public lands. Increased settlement will place increases in demands for water and infrastructure and recreational resources on public lands.

Moreover, the pattern of land ownership has important correspondence and influence on vegetative communities. Again, this is largely a result of the selection of agricultural lands in the 19th century and subsequent withdrawals in the early 20th century. (GAP reference here along with narrative describing GAP combined classes)   Importantly, historic allocations of land have created sharp distinctions in distributions of upland and desert vegetative communites.

Private lands, largely following valley bottom, lowland and foothills settings correlate strongly with deciduous oak, mountain shrub, woody riparian wetland and mountain grassland (Fig. 2-5).

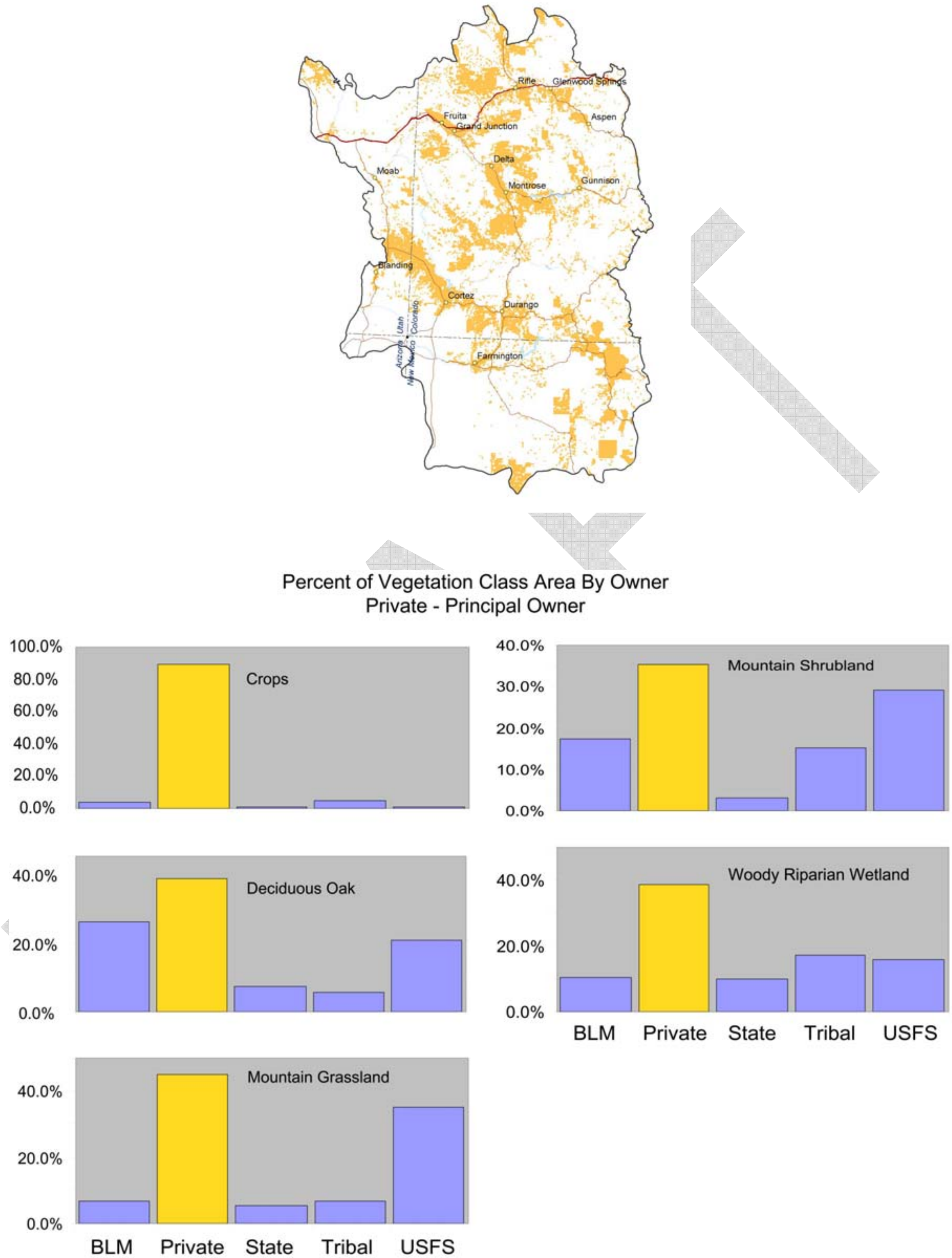


Figure 2-5 Significant proportions of lowland and valley bottom vegetation communities are found on private lands in the subregion.

While largely following valley bottom, lowland and foothills settings correlate strongly with deciduous oak, mountain shrub, woody riparian wetland and mountain grassland (Fig. 2-6).

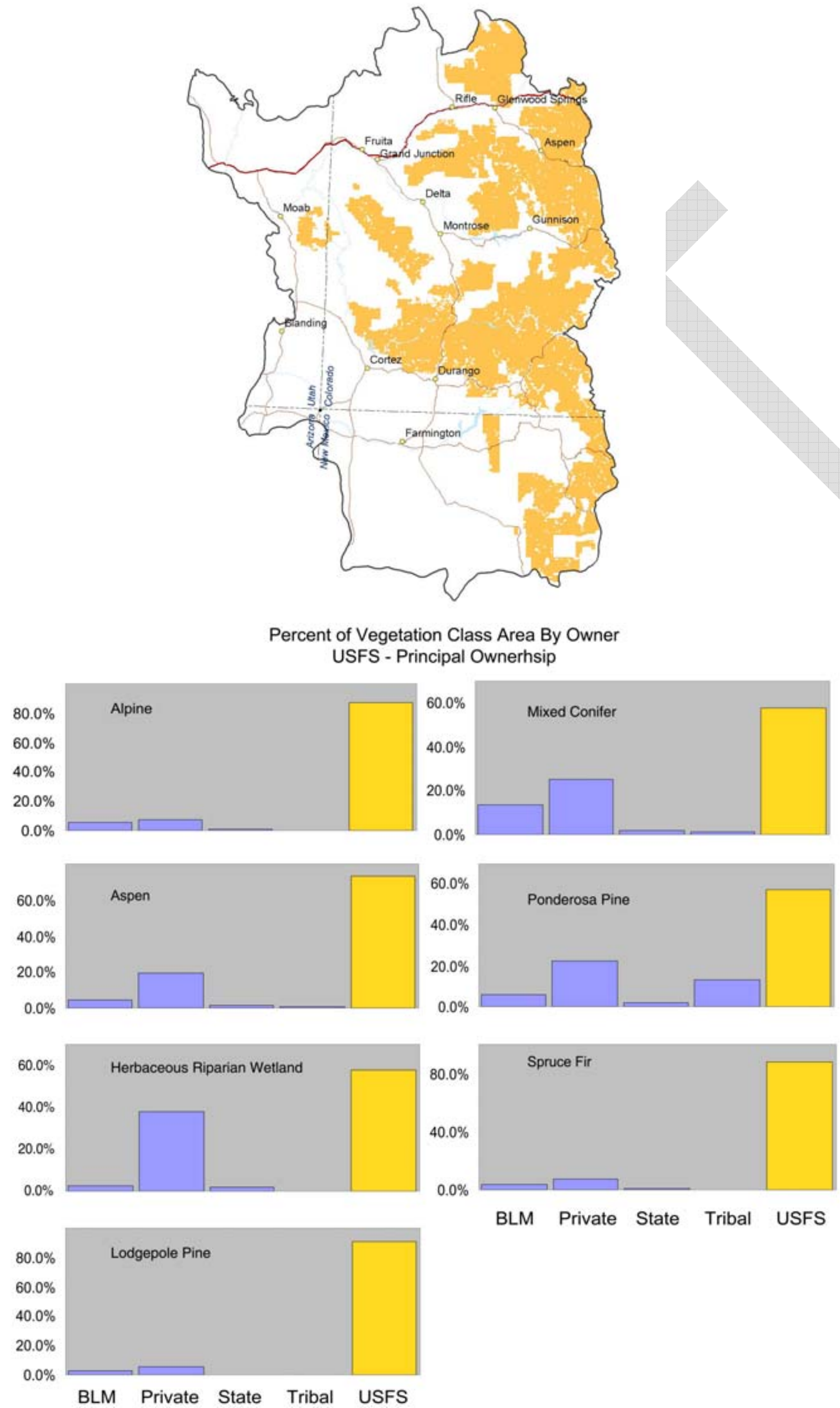


Figure 2-6 Strikingly significant proportions upland and alpine vegetation communities are found on U.S. Forest Service lands in the subregion.

Forest service ownership strongly correlate to uplands and upland forested systems (Fig. 2-6, above). Where proportions fall below sixty percent, much of the remaining proportion of the vegetation community is in private ownership. Importantly, nearly all of herbaceous riparian wetland and woody riparian wetland are under forest service management or under private ownership.

Alternatively, BLM lands are dominately comprised of dryland and desert vegetative communities (Fig. 2-7). Sixty to seventy percent of pinyon juniper and sagebrush communities are under BLM management or privately held. Furthermore, nearly thirty percent of desert grassland, greasewood and desert shrub are under BLM management.

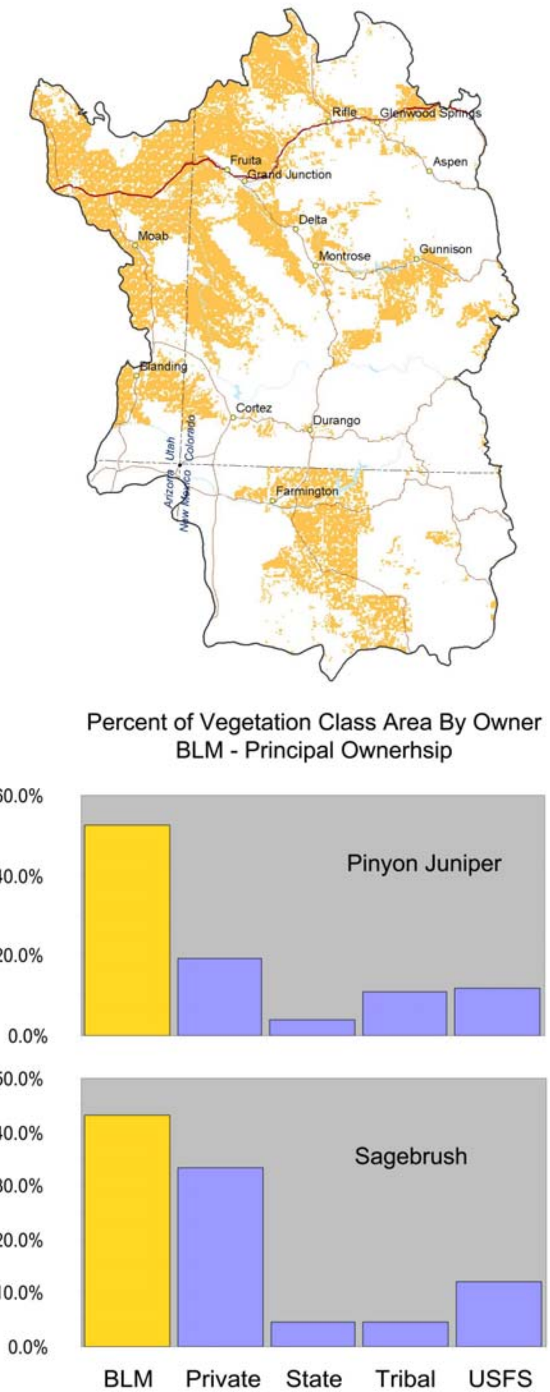
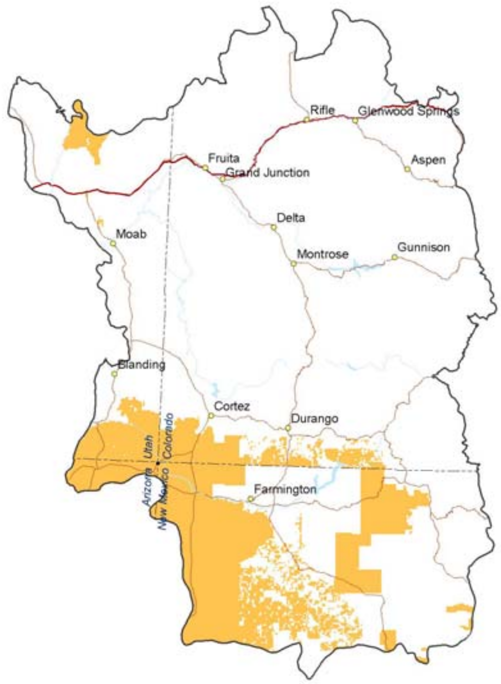


Figure 2-7 BLM lands in the subregion.

Finally, Tribal lands are largely comprised of dry-land and desert vegetative communities. Significantly, the majority of barren lands are Tribal.



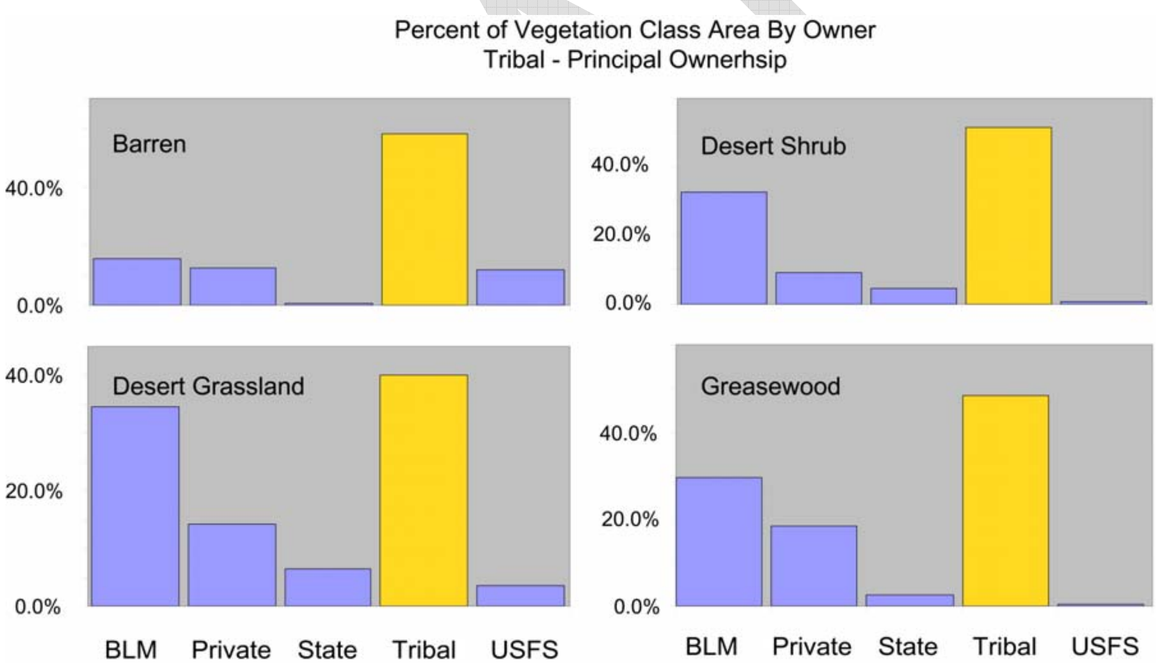


Figure 2-8 Tribal lands are substantially more barren than other ownerships. They, along with BLM lands contain much of the dry desert vegetative communities in the subregion.

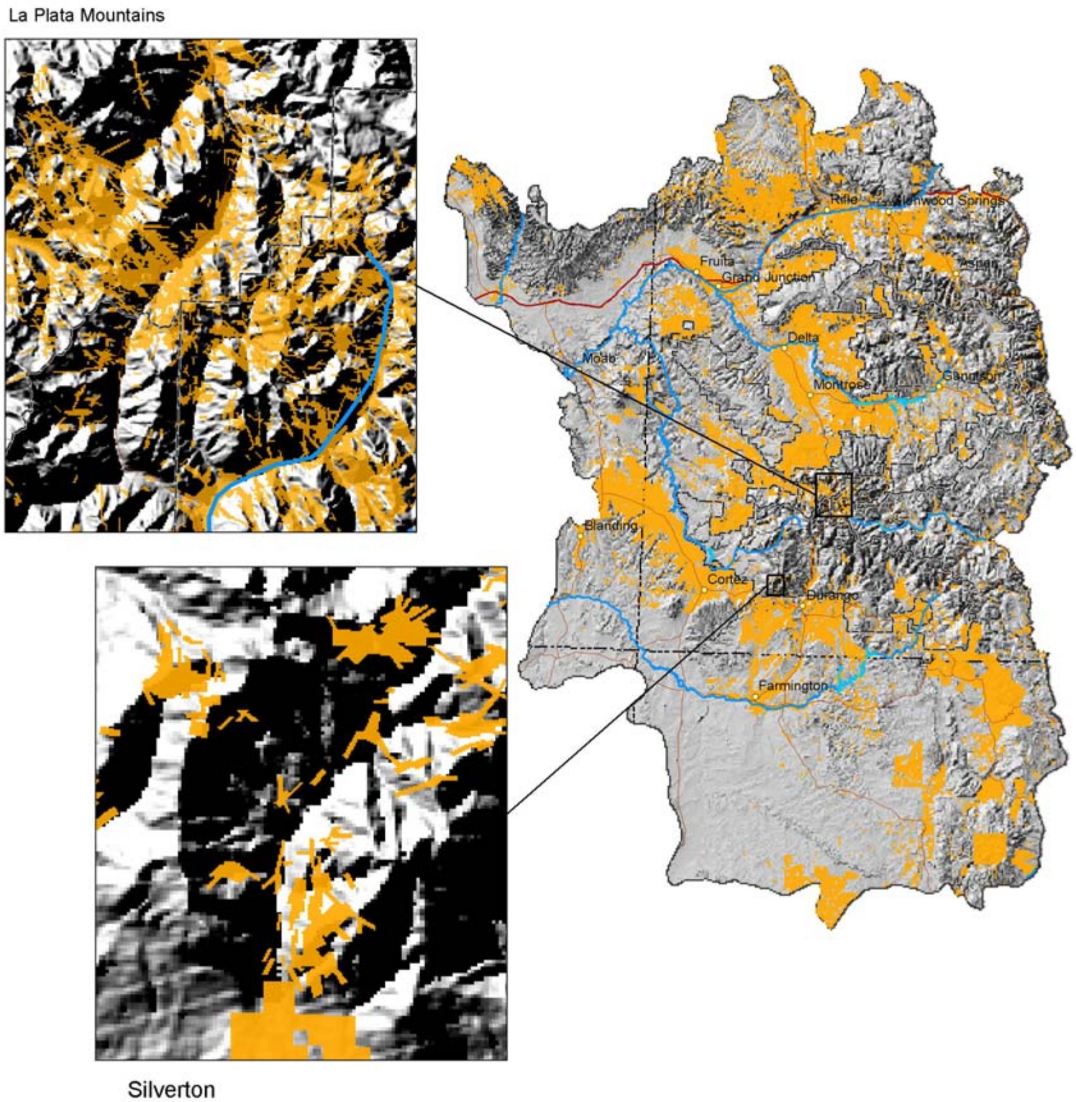


Figure 2-9 Important mineralized areas in the subregion
Important mineralized areas in the subregion were claimed and patented to become privately held lands during important historical periods of development. Locally these are important.

In the subregion mineralized areas are covered with patented mining claims. As in-holdings, contained in Federal land, these lands have always been of important interest. These lands have been of economic interest economically. Today some tracts attract interest as potential back-country home-sites. Overall they are of critical safety and environmental concern.

Population and Demographic Setting

Narrative to be done here. Also – the demographic data is from ESRI counties data because we're looking at four states.

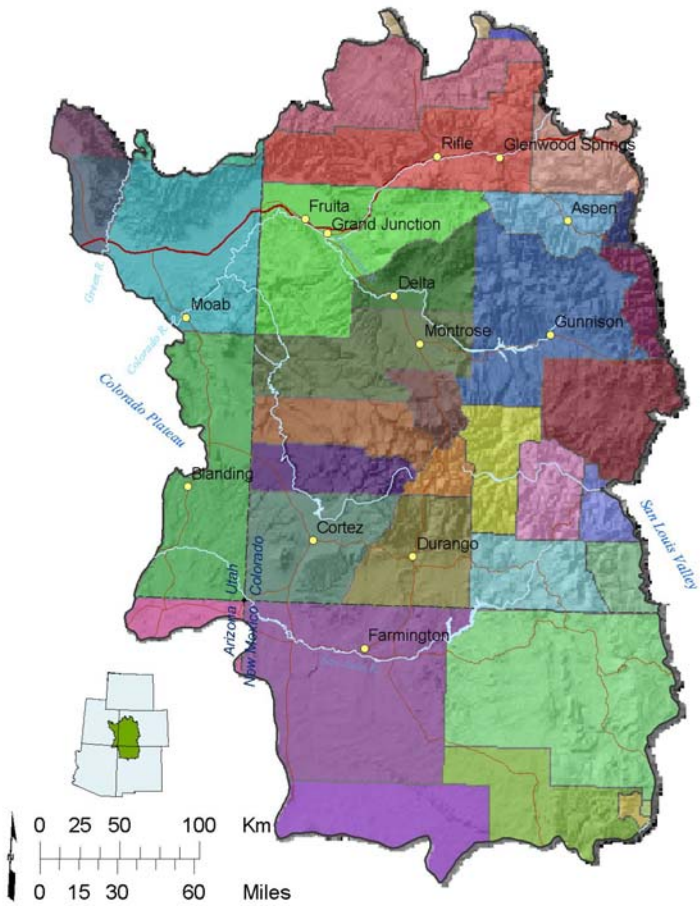


Figure 2-10 Thirty-eight counties comprise the Greater Assessment Area. Of these, seventeen have 90% percent or more of their area in the subregion. Of those eighteen, 14 are fully contained in the subregion.

Table 2-2 Counties in the Greater Assessment Area.

State Name	County	SubRegion %	County %
Arizona	Apache	1.0%	2.4%
Colorado	Moffat	0.2%	1.1%
	Routt	0.1%	1.4%
	Rio Grande	0.8%	23.8%
	Lake	0.3%	42.6%
	Conejos	1.1%	44.1%
	Chaffee	1.0%	51.3%
	Saguache	3.6%	60.5%
	Eagle	2.0%	61.8%
	Rio Blanco	3.9%	64.1%
	Garfield	5.6%	99.5%
	Mesa	6.2%	100.0%
	Gunnison	6.1%	100.0%
	Montrose	4.2%	100.0%
	Montezuma	3.9%	100.0%
	La Plata	3.2%	100.0%
	Archuleta	2.6%	100.0%
	San Miguel	2.4%	100.0%
	Delta	2.1%	100.0%
	Hinsdale	2.0%	100.0%
	Dolores	2.0%	100.0%
	Pitkin	1.8%	100.0%
	Mineral	1.7%	100.0%
	Ouray	1.0%	100.0%
	San Juan	0.8%	100.0%
New Mexico	Santa Fe	0.1%	0.9%
	McKinley	3.7%	35.7%
	Sandoval	3.5%	50.1%
	Rio Arriba	9.5%	86.4%
	San Juan	9.3%	88.9%
	Los Alamos	0.2%	99.7%
Utah	Uintah	0.2%	1.2%
	Emery	1.3%	15.1%
	Carbon	0.7%	25.5%
	San Juan	5.9%	39.8%
	Grand	6.2%	89.3%

Table 2-3 2000 population and population density by county.
Calculated subregion population estimate is given by product of Sq. Miles x Pop. Density.

State Name	County	Pop. 2000	Pop. Density	Sq. Miles	Calc. Pop.
Arizona	Apache	69,423	6.3	514.8	3,243
Colorado	Hinsdale	790	0.7	1,077.7	754
	Mineral	831	0.9	924.3	832
	San Juan	558	1.4	401.0	561
	Dolores	1,844	1.7	1,071.5	1,822
	Rio Blanco	5,986	1.9	2,055.5	3,905
	Saguache	5,917	1.9	1,928.7	3,665
	Moffat	13,184	2.7	52.7	142
	Gunnison	13,956	4.3	3,255.0	13,997
	San Miguel	6,594	5.1	1,291.2	6,585
	Conejos	8,400	6.6	560.1	3,697
	Ouray	3,742	6.8	553.2	3,762
	Archuleta	9,898	7.2	1,366.7	9,841
	Routt	19,690	8.4	60.6	509
	Montezuma	23,830	11.5	2,082.3	23,947
	Rio Grande	12,413	13.7	424.9	5,822
	Garfield	43,791	14.7	2,968.5	43,637
	Montrose	33,432	15.1	2,223.4	33,574
	Pitkin	14,872	15.7	949.4	14,905
	Chaffee	16,242	16.1	517.2	8,327
	Lake	7,812	20.2	164.8	3,329
	Delta	27,834	24.4	1,142.8	27,884
	Eagle	41,659	24.6	1,046.0	25,733
	La Plata	43,941	26.1	1,682.6	43,917
	Mesa	116,255	35.0	3,331.6	116,605
New Mexico	Rio Arriba	41,190	7.0	5,078.7	35,551
	McKinley	74,798	13.7	1,952.7	26,752
	San Juan	113,801	20.5	4,935.3	101,174
	Sandoval	89,908	24.2	1,862.9	45,082
	Santa Fe	129,292	67.2	30.2	2,030
	Los Alamos	18,343	182.7	100.1	18,291
Utah	San Juan	14,413	1.8	3,165.6	5,698
	Grand	8,485	2.3	3,279.9	7,544
	Emery	10,860	2.5	672.1	1,680
	Uintah	25,224	5.6	49.8	279
	Carbon	20,422	13.3	395.2	5,256
	Total	1,089,630			650,330
	Average	58,899	17.1		
	Max	129,292	182.7		
	Min	558	0.7		

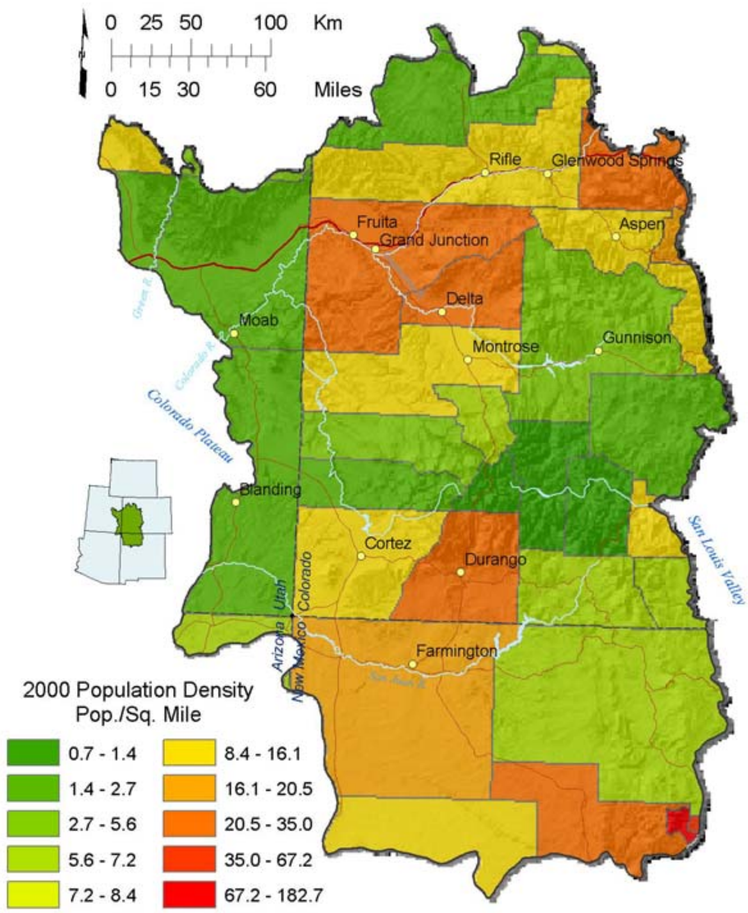


Figure 2-11 2000 population density by county.

High density counties and population centers reflect centers of economic development from agriculture, mining and energy development, recreation, and lately immigration of retiring baby-boomers. Low density counties reflect upland and desert areas.

Anthropogenic Activities and Use Categories

To be completed: -- This will simply outline the categories to be evaluated in the upcoming analysis.

Watershed Sensitivity
To be completed:

Summary of Important Information Gaps
To be completed:

Bibliography

Blair, Rob Managing Editor. 1996. The Western San Juan Moutains, Their Geology, Ecology and Human History. University Press of Colorado, Fort Lewis College Foundation, Niwot, Colorado.

Cotner, Robert C. John S. Ezell and Gilbert C. Fite. 1957. Readings in American History, Volume Two: 1865 to the Present. The Cambridge Press, Cambridge Mass.

Daly, Chris and Wayne Gibson, 2002. 103-Year High-Resolution Precipitation Climate Data Set for the Conterminous United States. Spatial Climate Analysis Service, Oregon Corvallis, Oregon. *Online:* <URL:ftp://ftp.ncdc.noaa.gov/pub/data/prism100>

Desert Research Institute (DRI). 2004. State Climate Extremes. Western Regional Climate Center. Online: http://www.wrcc.dri.edu/htmlfiles/state.extremes.html

ECOMAP. 1993. National hierarchical framework of ecological units. Unpublished administrative paper. Washington, DC: U.S. Department of Agriculture, Forest Service. 20 p. Ecoregions of the United States [map, rev. ed.]. Robert G. Bailey, cartog. 1994. Washington, DC: U.S. Department of Agriculture, Forest Service. Scale 1:7,500,000; colored

Environmental Systems Research Institute (ESRI). 2002. ESRI Data &Maps, Disk 7, ArcMap V8.2

Fenneman, N.M., and Johnson, D.W. 1946. Physical divisions of the United States. Washington, DC: U.S. Geological Survey. Online: http://water.usgs.gov/lookup/getspatial?physio

Hardy Colin C. and David L. Bunnell. 2001. Potential Natural Vegetation Groups, Version 2000. U.S. Dept. of Agriculture, Forest Service, Fire Sciences Laboratory, Rocky Mountain Research Station, Missoula, Montana. Online: http://www.fs.fed.us/fire/fuelman/pnv.htm

Hecox, Walter E. and Bradley L. Ack. 1996. Charting the Colorado Plateau, An Economic and Demographic Exploration. The Colorado College, Salt River Project (SRP) and Grand Canyon Trust, Flagstaff, AZ.

Maxwell, J.R.; Edwards, C. J.; Jensen, M. E. 1995. A hierarchical framework of aquatic ecological units in North America (nearctic zone). General Technical Report. NC-176. St. Paul MN: U.S. Department of Agriculture, Forest Service, North-Central Forest Experimental Station. 72p.

Maxwell, J.R.; Edwards, C. J.; Jensen, M. E. 1995b. A Map showing subzones, regions, and subregions of aquatic zoogeography in North America (Nearctic Zone). Great Lakes Ecological Assessment Online: http://www.ncrs.fs.fed.us/gla/elc/aqsubreg.htm.

McNab, W. Henry and Peter E. Avers. 1994. U.S. Department of Agriculture, Forest Service, Ecomap Team, Online: http://www.fs.fed.us/land/pubs/ecoregions/

Meyer, C. B. and D. H. Knight. 2001. Historic variablility for upland vegetation in the Bighorn National Forest, Wyoming. Unpublished report on file at Bighorn National Forest S.O., Sheridan, Wyoming.

Preston, Michael. 2004. Productive Harmony Analysis, Interpretive Framework for Social and Economic Assessment of Southwest Colorado Communities and San Juan Public Lands, DRAFT. Fort Lewis College, Office of Community Services, Unpublished. Prepared report for and on file at the San Juan Public Lands Center, U.S. Forest Service and Bureau of Land Management. Durango, Colorado.

Steeves, Peter and Douglas Nebert. 1994. Hydrologic units maps of the Conterminous United States. Reston, Virginia: U.S. Geological Survey Online: http://water.usgs.gov/lookup/getspatial?huc250k

Seyfert, Carl K. and Leslie A. Sirkin. 1974. Earth History and Plate Tectonics, An Introduction to Historical Geology. Harper and Row Pub. Inc. NY, NY.

U.S. Geological Survey. 1981. Geographic Names Information System (GNIS). Reston, Virginia. Online: http://nationalmap.usgs.gov/

U.S. Geological Survey. 2002. Elevations and Distances in the United States. USGS Eastern Region Geography, Reston VA. Online: http://mac.usgs.gov/mac/isb/pubs/booklets/elvadist/elvadist.html

U.S. Geological Survey. 2004. Colorado Drought Watch, Graphic Comparison of 2004 Streamflow and Volume to Historic Values for 23 Sites. U.S. Department of the Interior, U.S. Geological Survey, Water Resources of Colorado. Online: http://webserver.cr.usgs.gov/index.html

Chapter 3. Biogeographic Significance

The Colorado Plateau and the southern Rocky Mountains (Fig. 3-1) are the two dominant geographic features at the subregion scale that have an important influence on historical and current species distribution patterns. In addition, there are numerous smaller geographic features (basins, plateaus, mesas, and mountain ranges) that influence the spatial arrangement of species and vegetation communities at this scale. Some of the prominent geographic features include the San Juan Basin, Uncompahgre Plateau, Grand Mesa, and the San Juan Mountains. For this discussion, the influence of the two dominant geographic features at the subregion scale on biotic distributions will be addressed.

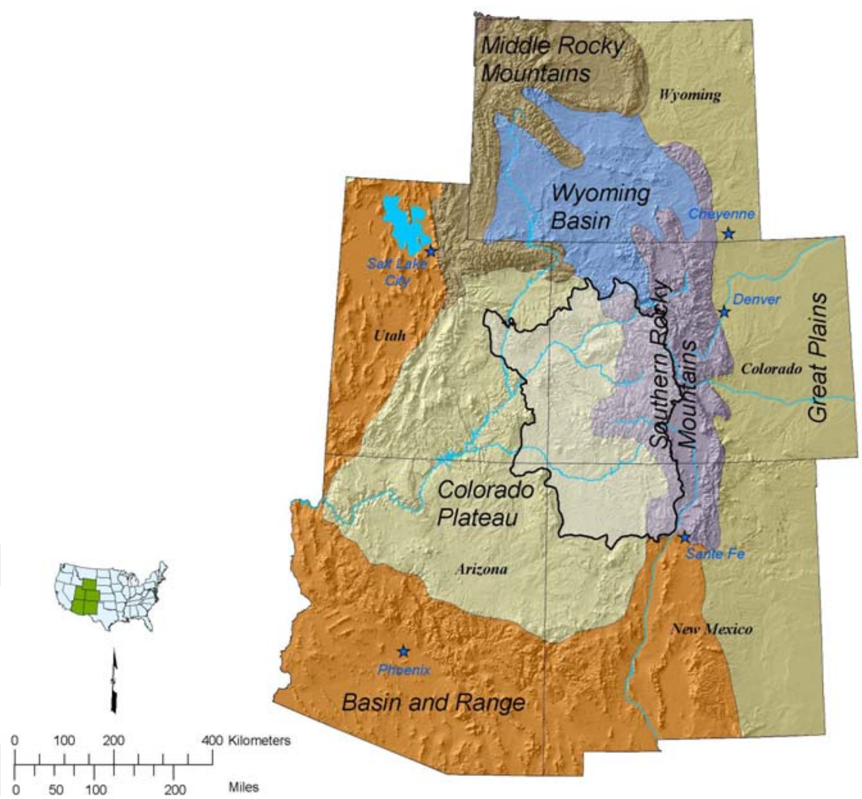


Figure 3-1 The Subregion scale of the assessment covers portions or the eastern Colorado Plateau and western Southern Rocky Mountains provinces.[1]

[1] After Fenneman, N.M., and Johnson, D.W. 1946. (Save this ref. till end of editing: II_E_Fig__Provinces4643.mxd)

The subregion scale of our analysis includes the northwestern and southwestern sections of the Colorado Plateau. The Colorado Plateau is a unique physiographic province in the western U.S. located between the Rocky Mountains to the east and north and the Basin and Range to the west and south. The Plateau consists of an area of approximately 83 million acres (129,697 square miles) with an average elevation of 1525 m (Anderson et al. 2000). One of the major biogeographical influences of the Plateau is its overwhelming role in monsoonal precipitation that is characteristic of the Southwest (Adams and Comrie 1997). The Plateau serves as a high elevation landmass that heats cool air during the summertime and creates a trough that creates monsoons. Generally in this region the cold temperatures from spring snowpack in the southern Rocky Mountains would prohibit monsoonal development if not for the presence of the Colorado Plateau. It has been hypothesized that snowpack on the Colorado Plateau during the Pleistocene prevented monsoonal precipitation and therefore the distribution of ponderosa pine (*Pinus ponderosa*) (Thompson et al. 1993).

The north central, northeastern, and parts of the southeastern region of the southern Rocky Mountains comprise our subregion scale. The Rocky Mountains are the largest interior mountain range in North America forming a north-south axis between 35-65º N latitude. This mountain range forms a significant 2,000-4,000 m barrier to the westerly atmospheric flow from the Pacific Ocean, creating classic orographic precipitation patterns that have significant influences on biotic distributions. In the Southwest however, the majority of this Pacific moisture is lost as it crosses the Sierra Nevada and intermountain west and moisture patterns are driven primarily by tropical and subtropical surface temperatures and ENSO (El Nino and the Southern Oscillation) circulation patterns (Kittel et al. 2002). The Rocky Mountains have considerable high species diversity in comparison to other adjacent regions due to its latitudinal and topographic diversity (Ricketts et al. 1999). In addition, many species in the southern Rockies represent relict populations of the once widely distributed Oroboreal flora during the Tertiary period and have similarities with species currently found only in Middle Asia (Weber 2003).

Historical Biogeographical Perspective

Understanding regional responses of species, populations, and vegetation types to historical climate and topography on both temporal and spatial scales is critical to formulating theories and testing hypotheses about species responses to present day natural and anthropogenic changes. The majority of this information in the Southwest comes from fossil pollen found in lake sediments, macrofossil assemblages of packrat (*Neotoma* spp.) middens, and charcoal deposits (Whitlock et al. 2002). For this biogeographical assessment, historical migration patterns of some tree species that are currently prevalent at the subregion scale are examined. Existing patterns of species distribution in relation to climate, topography, and other environmental drivers are assessed in Module 3: Existing Vegetation Conditions. The majority of historical biogeographical research conducted in the Southwest specifically applies to either the Colorado Plateau region or the southern Rocky

Mountains, both part of our subregion scale. While complete patterns at the subregion scale cannot be assessed through current literature due to a lack of scientific inquiry at our specific subregion scale, research for the general area is still extremely useful in understanding broad patterns of species migration over the past thousands of years.

Climate has long been considered a major environmental driver in determining biotic distribution patterns at regional scales (Woodward and Williams 1987, Humphries and Bourgeron 2003). During the late Pleistocene (50,000-12,000 B.P.), cooler temperatures (~3-5º C) in this region allowed species generally found at higher elevations to be commonly found at significantly lower elevations. For example, areas on the Colorado Plateau that are currently densely covered with pure stands of ponderosa pine were instead forested by Engelmann spruce (*Picea engelmanni*) and subalpine fir (*Abies lasiocarpa*), which are currently found hundreds of meters higher in their present range (Anderson et al. 2000). Similarly, low elevations that previously sustained montane forests during this timeframe are now composed of steppe and desert vegetation types (Whitlock et al. 2002). Different species assemblages were also found together during the Pleistocene, such as Engelmann spruce and sagebrush (*Artemisia* spp.) on the Colorado Plateau, indicating that species historically responded individualistically to changes in climate rather than as complete vegetation communities (Anderson et al. 2000).

Approximately 12,000 years ago, major environmental changes occurred resulting in a transition to the current interglacial period, the Holocene, which resulted in a major restructuring of biotic distributions (Allen et al. 1998). As the climate began to warm, vegetation became arranged by latitudinal and elevational gradients. Present day climate, vegetation patterns, and fire regimes at the subregion scale are the result of this global climatic shift. Evidence for the importance of fire as a natural disturbance agent in this region during this time period is seen from 1) the rapid spread of fire evolved ponderosa pine forests across the region where previously they were absent (Anderson 1989), 2) the abundance of charcoal deposits found in bog and lake sediments (Weng and Jackson 1999) and 3) changes in early successional tree species composition in relation to climatic patterns (Fall 1997).

Low Elevation Forests

During the late Pleistocene, data suggest that pinyon pine (*Pinus edulis*) occurred below 1770 m at 31-34º N, much farther south in the present location of modern southwestern deserts, and at lower elevations that its current distribution (Van Devender, 1986). In addition, packrat midden evidence indicates that pinyon pines grew in association with other species that they currently do not occur with today (Betancourt et al. 1990). Since approximately 12,000 B.P., pinyon pine has expanded northward to ~40º N and elevational ranges between ~1290-2900 m (Lanner and Van Devender 1998). It has been hypothesized that this expansion was most likely due to the activity of corvids that are known to disperse large, wingless seeds over long distances (Vander Wall and Balda 1981 in Lanner and Van

Devender 1998). Pinyon pine and other woodland species (e.g., *Juniperus* spp.) moved to their general current geographical distribution approximately 6,000 B.P. and have fluctuated both in latitude and elevation to climatic changes over the past thousand years with junipers being more tolerant of warm, drought conditions than pinyon pine (Thompson et al. 1993).

Middle Elevation Forests

Cooler and wetter temperatures of the late Pleistocene resulted in heavy snowfall and the development of glaciers at higher elevation, which prevented summertime heating and thermally induced low-pressure cells. These climatic conditions combined with a southerly jet stream and lower sea levels in the Gulfs of Mexico and California, may have inhibited the development of summer monsoon rains and the establishment of ponderosa pine until the late Holocene (Anderson et al. 2000). Since ponderosa pine trees are highly evolved to frequent, low intensity surface fire regimes in this subregion, changes in rain patterns and lightning would have had significant impacts on the distribution of this fire evolved species more so than other western conifers. A climate that is conducive to the establishment of understory vegetative growth and favorable to early summer fires, which established in the late Holocene, is consistent with current climatic conditions where ponderosa pine is presently distributed.

High Elevation Forests

Spruce (*Picea engelmannii* and *Picea pungens*) trees were found in portions of the southeastern Colorado Plateau during the Last Glacial Maximum (18,000 yr B.P.), but at elevations approximately 1000 m lower than their current distribution and were absent from its modern range in central Colorado at this time, probably because of cold and dry conditions (Thompson and Anderson 1997). During the early Holocene, 12,000 B.P., spruce dispersed northward to higher elevations and by 6000 B.P. spruce was approaching its current distribution limits. Pollen studies from central Colorado indicate that between 7000 and 4000 B.P. subalpine forests extended both lower in elevation, due to higher soil moisture availability, and higher in elevation due to warmer temperatures (Fall 1985). Pollen evidence indicates that treeline in the San Juan Mountains reached its current position approximately 3,000 B.P. as a result of area cooling (Carrara et al. 1984). The coexistence dynamics of subalpine fir and Engelmann spruce were different during the Holocene that today. Engelmann spruce was more dominant, while presently subalpine fir is the more dominant of the two species. The life history traits of Engelmann spruce being more prevalent in areas with wetter and warmer environments suggest that infrequent stand replacing fires were common during this timeframe. In addition, the dominance of aspen (*Populus tremuloides*) and lodgepole pine (*Pinus contorta*) pollen in intervals between spruce/fir pollen in lake sediments indicates that regions in central Colorado may have burned repeatedly with reforestation of spruce/fir during non-fire intervals (Hall 1997). Charcoal evidence in lake sediments for this study also supports the theory that fire permitted early successional aspen and lodgepole pine to become established in otherwise spruce/fir dominated forests (Hall 1997).

Summary

General conclusions can be drawn from the paleoecological history at the subregion scale. First, biotic distributions have been responding to climatic shifts for thousands of years as evident in shifting ecotonal boundaries and can be expected in the future in response to human land-use and global change (Walther et al. 2002). Second, shifts in biotic distributions do not necessarily occur as intact communities but rather communities have been disassembled and reassembled for thousands of years (Whitlock et al. 2002). Finally, fire regimes are closely tied to climatic regimes and have varied in the past with high frequency regimes being most prevalent during periods of drought historically (Swetnam and Betancourt 1998) and fluctuations in tree composition have been responding to fires over thousands of years (Hall 1997). All of these general patterns have implications for interpreting the historical range of variability (HVR) for different vegetation types at the subregion and landscape scales and for creating land management decisions based on HRV. See Module 3: Existing Vegetation Conditions for further discussion on this topic.

Literature Cited

Adams, D. K., and A. C. Comrie. 1997. The North American monsoon. Bulletin of the American Meteorological Society 78: 2197-2214.

Allen, C. D., J. L. Betancourt, and T. W. Swetnam. 1998. Landscape changes in the southwestern United States: Techniques, long-term data sets and trends. Pp. 71-84 In: Sisk, T. D., editor Perspectives on the Land Use History of North America: A context for understanding our changing environment. Biological Science Report USGS/BRD/BSR-1998-0003. U.S. Geological Survey, Biological Resources Division, Reston, VA.

Anderson, R. S. 1989. Develolpment of the southwestern podnerosa pine forests: What do we really know? Pp. 15-22 In: USDA Forest Service General Technical Report RM-185. Fort Collins, Colorado.

Anderson, R. S., J. L. Betancourt, J. I. Mead, R. H. Hevly, and D. P. Adam. 2000. Middle- and late-Wisconsin paleobotanic and paleoclimatic records from the southern Colorado Plateau, USA. Palaeogeography, Palaeoclimatology, Palaeoecology 158: 25-43.

Betancourt, J.L., T. R. Van Devender, and P. S. Martin. (eds.) 1990. Fossil packrat middens: the last 40,000 years of biotic change. University of Arizona Press, Tucson, Arizona.

Carrara, P. E., W. N. Mode, M. Rubin, and S. W. Robinson. 1984. Deglaciation and postglacial timberline in the San Juan Mountains, Colorado. Quaternary Research 21: 42-55.

Fall, P.L., 1985, Holocene dynamics of the subalpine forest in central Colorado. Pp. 31-46 In: Jacobs, B.F., Fall, P.L., and Davis, O.K. (Editors), Late Quaternary vegetation and climate of the American Southwest. American Association of Stratigraphic Palynologists, Contributions Series No. 16.

Fall, P. L. 1997. Fire history and composition of the subalpine forest of western Colorado during the Holocene. Journal of Biogeography 24: 309-325.

Fenneman, N.M., and Johnson, D.W. 1946. Physical divisions of the United States. Washington, DC: U.S. Geological Survey Online: http://water.usgs.gov/lookup/getspatial?physio

Humphries, H. C., and P. S. Bourgeron 2003. Environmental responses of Pinus ponderosa and associated species in the south-western USA. Journal of Biogeography 30: 257-276.

Kittel, T. G. F., P. E. Thornton, J. A. Royle, and T. N. Chase 2002. Climates of the Rocky Mountains: Historical and Future Patterns. Pp. 59-82 In: Baron, J. S. editor Rocky Mountain Futures: an ecological perspective. Island Press, Washington, D.C.

Lanner, R. M., and T. R. Van Devender 1998. The recent history of pinyon pines in the American Southwest. Pp. 171-182 In: Richardson, D. M. (ed.) Ecology and Biogeography of Pinus. Cambridge University Press. New York, New York

Ricketts, T. H., E. Dinerstein, D. M. Olson,, C. J. Loucks, W. Eichbaum, D. DellaSala, K. Kavanagh, P. Hedao, P. T. Hurley, K. M. Carney, R. Abell, and S. Walters. 1999. Terrestial ecoregions of North America: A conservation assessment. Island Press. Washington D. C.

Swetnam, T. W. and J. L. Betancourt. 1998. Mesoscale disturbance and ecological response to decadal climatic variability in the American Southwest. Journal of Climate 11: 3128-3147.

Thompson, R. S., and K. H. Anderson. 1997. Past climate and vegetation changes in the Southwest. Impacts of climate change and land use in the southwestern United States. U.S. Global Change Research Program Workshop. USGS. http://geochange.er.usgs.gov/sw/impacts/biology/pastclim/

Thompson, R.S., Whitlock, C., Bartlein, P.J., Harrison, S.P., and Spaulding, W.G. 1993. Climatic changes in the western United States since 18,000 yr

B.P. Pp. 468-513 In: Wright, H.E., Jr., Kutzbach, J.E., Webb, T. III, Ruddiman, W.F., Street-Perrott, F.A., and Bartlein, P.J. (Editors). University of Minnesota Press.

Vander Wall, S. B. and R. P. Balda 1981. Ecology and evolution of food storage behavior in conifer-seed-caching corvids. Zeitschrift fur Tierpsychologie 56: 217-242.

Van Devender, T. R. 1986. Late Quaternary history of pinyon-juniper-oak woodlands dominated by Pinus remota and Pinus edulis. Pp. 99-103 In: R. L. Everett, comp Proceedings-Pinyon-Juniper Coniference. USDA Forest Service. Ogden, Utah.

Walther, G.-R., E. Post, P. Convey, A. Menzel, C. Parmesan, T. J. C. Beebee, J.-M. Fromnentin, O. Hoegh-Guldberg, and F. Bairlein. 2002. Ecological responses to recent climate change. Nature 416: 389-395.

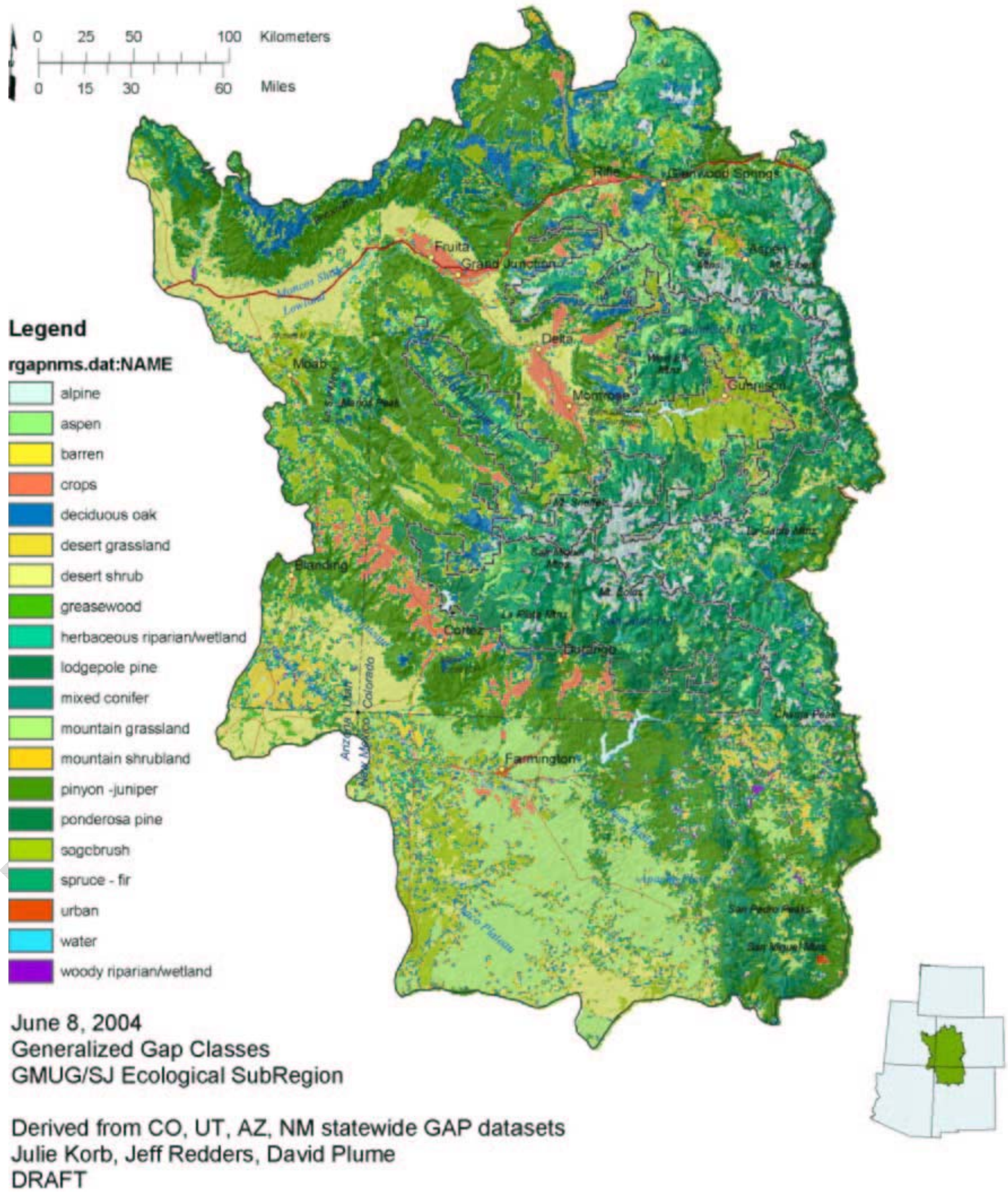
Weber, W. A. 2003. The middle Asian element in the southern Rocky Mountain flora of the western United States: a critical biogeographical review. Journal of Biogeography 30: 649-685.

Weng, C. and S. T. Jackson. 1999. Late Glacial and Holocene vegetation history and paleoclimate of the Kaibab Plateau, Arzona. Palaeogeography, Palaeoclimatology, Paleoecology 153: 179-201.

Whitlock, C., M. A. Reasoner, and C. H. Key 2002. Paleoenvironmental history of the Rocky Mountain Region during the past 20,000 years. Pp. 41-57 In: Baron, J. S. editor Rocky Mountain Futures: an ecological perspective. Island Press, Washington, D.C.

Woodward, F. I., and B. G. Williams 1987. Climate and plant distribution at global and local scales. Vegetatio 69: 189-197.

Chapter 4. Subregional Existing Vegetation Conditions



A. Sub-Regional-scale vegetation descriptions

Current Vegetative Description[2]

Vegetation communities at the Sub-Regional-scale were created by compiling existing GAP Analysis vegetation community classes from Colorado, New Mexico, Utah, and Arizona at the state level because currently no GAP Analysis exists across state borders in the Southwest; however, a Southwest Regional Gap Analysis is currently being created. We chose to use the Colorado GAP Analysis as our template for the Sub-Regional-scale GAP Analysis because the majority of the Sub-Region, 58%, is within Colorado, with only 26% in New Mexico, 14% in Utah, and 1% in Arizona. Our compiled Sub-Regional-scale GAP Analysis has 20 different categories, from an original 51 categories within Colorado GAP Analysis. Of these 20 categories, 17 vegetation communities have been categorized for the Sub-Regional-scale (Table 4-1).

Table 4-1 Generalized vegetation GAP vegetation classes in the subregion and proportion of sub-region.

Class	Name	Acres	SqMiles	Pct	SumPct
Vegetation	pinyon -juniper	7,265,426	11,352	21.3%	21.3%
	desert shrub	5,503,332	8,599	16.1%	37.4%
	spruce - fir	3,485,025	5,445	10.2%	47.6%
	sagebrush	2,965,204	4,633	8.7%	56.3%
	desert grassland	2,694,989	4,211	7.9%	64.2%
	aspen	2,272,984	3,552	6.7%	70.9%
	ponderosa pine	2,064,177	3,225	6.1%	76.9%
	deciduous oak	1,760,036	2,750	5.2%	82.1%
	alpine	1,606,281	2,510	4.7%	86.8%
	mixed conifer	1,209,315	1,890	3.5%	90.4%
	mountain grassland	589,306	921	1.7%	92.1%
	lodgepole pine	462,098	722	1.4%	93.4%
	barren	289,406	452	0.8%	94.3%
	mountain shrubland	222,042	347	0.7%	94.9%
	woody riparian/wetland	118,599	185	0.3%	95.3%
	greasewood	63,033	98	0.2%	95.5%
	herbaceous riparian/wetland	17,141	27	0.1%	95.5%
		32,588,394	50,919	95.5%	
Non-Vegetation	crops	1,398,025	2,184	4.1%	99.6%
	urban	64,334	101	0.2%	99.8%
	water	63,942	100	0.2%	100.0%
		1,526,301	2,385	4.5%	
		34,114,695	53,304	100.0%	

[2] Version: July 18, 2005

Julie Korb et al

Table 4-2 summarizes the distribution of GAP classes in the San Juan National Forests. The upland setting of the Forest is expressed in the dominance of upland categories including spruce-fire, aspen and alpine and the absence of classes such as desert grassland and greasewood.

Table 4-2 Generalized vegetation GAP vegetation classes in the San Juan National Forest. Percent values are relative to the Forest area.

Forest	Class	Name	Acres	SqMiles	Pct	SumPct
San Juan	Vegetation	spruce - fir	665,361	1,040	31.8%	31.8%
		ponderosa pine	543,573	849	26.0%	57.8%
		aspen	273,778	428	13.1%	70.8%
		alpine	252,406	394	12.1%	82.9%
		mixed conifer	145,760	228	7.0%	89.9%
		deciduous oak	65,747	103	3.1%	93.0%
		pinyon - juniper	54,924	86	2.6%	95.6%
		mountain grassland	32,274	50	1.5%	97.2%
		sagebrush	21,520	34	1.0%	98.2%
		barren	6,788	11	0.3%	98.5%
		mountain shrubland	3,032	5	0.1%	98.7%
		herbaceous riparian/wetland	1,520	2	0.1%	98.7%
		woody riparian/wetland	709	1	0.0%	98.8%
		greasewood	277	0	0.0%	98.8%
		Sub Total:	2,067,669	4,981	98.8%	
	Non-Veg	crops	15,523	24	0.7%	
		urban	72	0	0.0%	
		water	9,939	16	0.5%	
		Sub Total:	25,533	40	1.2%	
		Total:	2,093,203	5,021	100.0%	

Table 4-2 Generalized vegetation GAP vegetation classes in the San Juan National Forest.

It is important to recognize that GAP Analysis uses coarse-scale classification criteria and that each of the 17 vegetation communities that we have identified can be broken down into more detailed vegetation types. For example, the mixed conifer vegetation type can be broken down into warm, dry mixed conifer or the cool, moist mixed conifer forests. In addition, it is important to recognize that while our GAP Analysis map recognizes discrete patches of numerous vegetation types within the Sub-Region, there are in fact transition zones between these patches (ecotones) that play an important role in regional and landscape diversity and heterogeneity of the area.

Historical Range of Variability

Understanding the history of past environments and communities is very useful for managing future ecosystems. Historical ecology can help identify the extent to which current communities have deviated from historical community patterns and processes and identify which communities are in need of ecological restoration or management and susceptible to irreversible shifts to new successional trajectories (Holling and Meffe 1996). One of the primary goals of ecological restoration is to reverse the degradation of ecosystems by restoring their structure and function and natural disturbance regimes to conditions prior to degradation. Ecological restoration is also referred to as the process of renewing and maintaining ecosystem health, a current goal for ecosystem management whose main focus is on long-term ecosystem sustainability and diversity. In order to accomplish these goals, reference conditions (the historical or natural range of variability in ecological structures and processes that reflect evolutionary history, disturbance regimes, and abiotic and biotic conditions) must be determined prior to restoration or management (Fulé et al. 1997; Landres et al 1999). Reference conditions will be referred throughout the rest of this assessment as HRV (historical range of variability). In identifying HRV, it is also important to recognize a spatial and temporal context. Disturbances may create different successional stages of ecosystems at different spatial scales within the area to be managed and therefore produce a shifting mosaic of patches (White and Walker 1997). HRV is useful for creating a benchmark on which to evaluate current conditions, identifying the restoration potential of a site, and evaluating the success of restoration or management actions (White and Walker 1997). HRV can be determined from historical data from the site (e.g., old field notes, old photographs, etc.); contemporary data from current sites that are similar to the degraded sites; on-site information [e.g., paleoecological analysis (pollen, packrat middens, charcoal), fire scars, soil seed banks, dead and down woody material, soil characteristics, species composition]; traditional indigenous knowledge (e.g., understanding indigenous peoples burning, foraging, planting methods); and historical climatic data (Naveh 1998; Sauer 1998; Swetnam and Betancourt 1998; Swetnam et al. 1999; Gray et al. 2003).

In order for HRV assessment to be applicable and properly incorporated into management, site-specific HRV assessments need to be conducted. This HRV assessment is at the Sub-Regional-scale and therefore only represents generalized trends. Forest-scale HRV assessments will be discussed later in this module. It is important to recognize that at both the Sub-Regional and Forest-scales there are inherent variations in species composition, density, and age and vertical class structures between and within different vegetation communities. Many of these differences are the result of natural disturbances prior to European settlement (~1870) and land management legacies such as timber harvesting, livestock grazing, and fire suppression since European settlement. In addition, underlying any of these management legacies are the local environmental parameters (elevation, slope, aspect, substrate, and topographic position) that influence stand composition, structure, and function. These environmental parameters are the primary drivers that represent HRV. For example, a ponderosa pine stand on a 10-degree south facing slopes historically would have lower tree density and more shrub/grass interspaces than a ponderosa pine stand within a drainage facing north. Therefore management needs to reflect these inherent environmental differences.

A paucity of specific information for the entire Sub-Regional-scale exists and therefore generalizations must be made from site-specific HRV assessments surrounding and within the Sub-Region to the larger analysis area. More detailed studies are needed at the Sub-Regional-scale in order to properly assess how actions at the Forest-scale will influence Sub-Regional-scale patterns and processes within specific vegetation types.

Vegetation/HRV Description

The general vegetation/HRV descriptions will be discussed in order from the most to the least dominant within the Sub-Region. For some of the vegetation types there is a large discrepancy in dominance between the Sub-Region and Forest-scales.

Pinon-Juniper

The pinon-juniper vegetation type compromises 7,265,426 acres (21.3%) of the Sub-Region and 116,641 acres (4%) within the GMUG and 54,924 acres (3%) within the SJ National Forests (Fig. 4-1). This vegetation type is found between ~4500-8500 ft (1372-2591 m). At its lower boundary, this vegetation type grades into the sagebrush, desert shrub, and desert grassland types and at its upper boundary grades into the ponderosa pine vegetation type. Pinon pine (*Pinus edulis*), Utah juniper (*Juniperus osteosperma*) and Rocky Mountain (*Juniperus scopulorum*) dominate the pinon-juniper type. *Juniperus* spp. dominates lower elevation/xeric sites and pinon pine dominates at higher elevation/mesic sites.

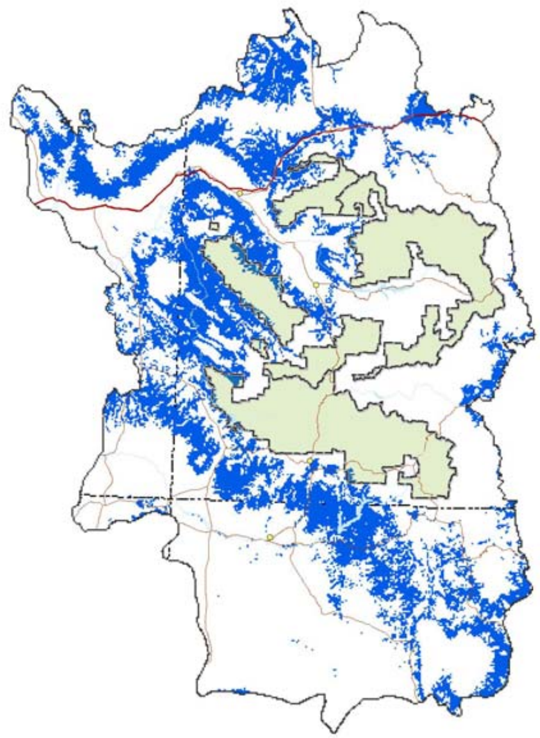


Figure 4-1 The Pinyon-Juniper GAP class in the subregion.
The Pinyon-Juniper GAP class in the subregioncovers 7,265,426 acres (21.3%) of the subregion.

There are two juniper types included in the pinon-juniper vegetation types. The juniper savanna type is located in northwestern New Mexico and on the east side of the Uncompahgre Plateau. This vegetation type occupies lower and warmer elevations than the pinon-juniper type and consists of widely spaced mature Utah juniper with shrubs and grasses (*Heterostipa comata*, *Hilaria jamesii*, and *Bouteloua gracilis*) (Romme et al. 2003f). Tree density found within the juniper savanna type is related to topography and soil type with grasses dominating gentle slopes and fine-textured soils. Increased tree densities and shrub encroachment has been correlated with livestock grazing, changes in fire regimes, and climate (West 1999). Areas that have experienced these changes are outside their HRV; however, specific information regarding HRV for this vegetation type is essentially unknown. The second juniper type is the Rocky Mountain juniper type that is found in association with coarse, dark-colored, rocky soils on steep slopes. This vegetation type is located in the Upper Gunnison Basin. Short-statured, sparse Rocky Mountain juniper along with sagebrush (*Artemisia tridentata*), mountain mahogany (*Cercocarpus* spp.), bitterbrush (*Purshia tridentata*), and sparse grasses (*Bouteloua gracilis*, *Festuca arizonica*, *Achnatherum hymenoides*) characterize this vegetation type. No specific information is available regarding HRV for this vegetation type. Livestock grazing and fire were likely rare in this vegetation type due to the steep slopes and paucity of herbaceous vegetation that is associated with this vegetation type and therefore these areas are probably inside their HRV.

The majority of pinon forests at the Sub-Regional-scale have previously been considered outside their HRV. This evaluation has been made from generalizing all pinyon-juniper forests as one vegetation type and not acknowledging the inherent diversity of stand structure and fire regimes for this forest type that dominates millions of hectares in the western United States. In addition, the methodological difficulty in determining HRV for this forest type (e.g., lack of fire scars, slow re-establishment after fire) in comparison to ponderosa pine has led to further discrepancies. A more thorough synthesis of current knowledge (Baker and Shinneman 2004) and site-specific studies at the Sub-Regional scale (Romme et al. 2003a; Eisenhart, 2004; Floyd et al. in press) presents a different viewpoint regarding the HRV of pinon-juniper forests for the analysis area. These assessments indicate that recent anthropogenic disturbances such as fire suppression, grazing, and recent climatic fluctuations must be put into a longer climatic timeframe to truly understand whether or not pinon-juniper forests are outside their HRV. For example, the large increase in pinon recruitment in the western U.S. during the last 20 years may be considered the result of anthropogenic disturbances. However, when this recruitment pulse is put into a larger climatic timescale, it shows that this pulse is the result of high seedling survivorship following the 1950's drought in the western U.S., one of the worst in the past 1000 years, and sustained from warm, wet springs since 1976 (Swetnam et al. 1999). The current drought and subsequent *Ips confusus* outbreaks in the Southwest over the past few years in many places is thinning out dense pinon stands that are the result of the post 1950's drought recruitment.

Baker and Shinneman (2004) and Romme and others (2003a) have identified three major types of pinon-juniper stand structures and fire regimes for the western U.S. All three of these major pinon-juniper types exist at the Sub-Regional-scale; however, more site-specific studies are needed throughout the analysis area and the forest to better understand their spatial distribution and abundance. It is impossible to assess percentages for each of these three pinon-juniper types on National Forest and BLM land due to a paucity of data. Based on the limited spatial data available, it is hypothesized that the pinon-juniper grass savanna is present but rare, the pinon-juniper shrub woodland has low abundance, and the pinon-juniper forest type is the most abundant on National Forest and BLM land within the analysis area. More research is necessary to support or refute this hypothesis.

The first pinon-juniper type is referred to as the pinon-juniper grass savanna. The pinon-juniper grass savanna's structure is characterized as having sparse trees, few shrubs, and dense grass and herbaceous cover with a frequent, low-severity surface fire that was sustained by significant grass cover (Romme et al. 2003a). Currently there are few areas that depict this stand structure and fire regime because of anthropogenic influences over the past ~120 years have moved this type outside its HRV. One known example within our Sub-Regional-scale is in northern New Mexico where pinon and juniper trees have become established in previously open grass/woodlands (Allen 1989). As bare ground was exposed from cattle grazing, pinon and juniper trees were able to get established and increase in numbers because of their ability to be more efficient competitors for water and nutrients than grasses.

The second pinon-juniper type is referred to as the pinon-juniper shrub woodland. The pinon-juniper shrub woodland's structure is characterized as having sparse to moderately dense herbs, shrubs, and trees all depending on the timing of the last fire and a moderately frequent, and a high-severity crown fire was sustained by shrubs and trees (Romme et al. 2003a). This pinon-juniper type is most prevalent in the Great Basin region and is widespread on both sides of the Uncompahgre Plateau at lower and middle elevations on deep soils developing from sandstone substrates (Romme et al. 2003f). These areas currently would have higher herb, shrub, and tree densities due to anthropogenic influences and therefore would be outside their HRV.

The third pinon-juniper type is referred to as the pinon-juniper forest. The pinon-juniper forest's structure is characterized as having dense trees, sparse to moderate shrubs, and few herbs with an infrequent, high-severity crown fire (Romme et al. 2003a). Pinon-juniper stands on Mesa Verde (Floyd et al. in press) and the Uncompahgre Plateau (Eisenhart 2004) are examples of this forest type. These areas are not outside their HRV. Numerous HRV studies for Mesa Verde show that the unprecedented frequency of stand replacing crown fires during the past decade do not necessarily indicate any departure from the natural fire regime which had a fire turnover time of ~400 years. Rather these studies suggest that previous wet years over the past 100 years have permitted increased tree densities and fuel continuity and ensuing fires during recent drought years most likely represents natural conditions at Mesa Verde (Floyd et al. in press). Structural evidence on the Uncompahgre Plateau also supports the theory that climate is the major driver in this forest type influencing cycles of tree establishment and mortality and thus fire and insect disturbances (Eisenhart 2004). In addition, paleoecological evidence shows current pinon distribution being just a few hundred years old in some areas of Colorado and not in equilibrium with modern climate, therefore making the assessment of whether recent tree expansion is the result of anthropogenic disturbances or long-term climatic fluctuations difficult (Allen et al. 1998).

Desert Shrub

The desert shrub vegetation type compromises 5,503,332 acres (16.13%) of the Sub-Region and only 331 acres within the GMUG and 0 acres within the SJ National Forests (Fig. 4-2). There is a significant paucity of research for this vegetation type at the Forest-scale, most likely because of it local rarity. Areas within the National Forests are of higher elevation, which limits the abundance of the desert shrub community. This vegetation type is more predominant farther west in Utah, Nevada, and parts of California. The desert shrub vegetation type is one of two semiarid shrub-steppe communities covering the majority of the low elevation areas within the Sub-Region. The dominant plants in this vegetation type are shadscale (*Atrixplex confertifolia*), fourwing saltbush (*Atriplex canescens*), saltbrush (*Atriplex gardneri*), rubber rabbitbrush (*Chrysothamnus nauseosus*), and horsebrush (*Tetradymia* spp.). All these shrubs are salt tolerant (halophytic) to varying degrees. Desert shrub is generally found on marine shales with poorly drained, saline soils (Floyd-Hanna et al. 1996). In areas with extreme concentrations of salt, these shrubs are generally unable to grow and bare ground is abundant or the greasewood vegetation type replaces them. Successional dynamics are limited due to the strict adaptations needed to survive in this harsh environment. Most

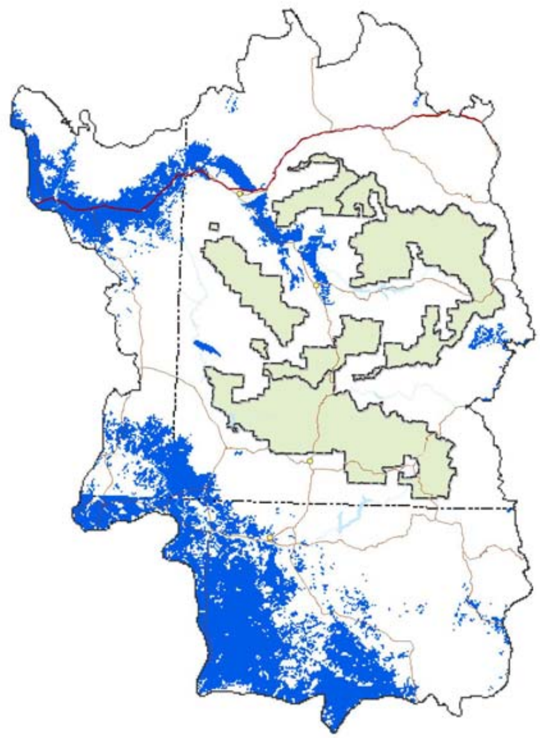


Figure 4-2 The Desert-Shrub GAP class in the subregion.
The Desert-Shrub GAP class in the subregion covers 5,503,332 acres (16.1%) of the subregion.

plants are members of the Chenopodiaceae with the vegetation compromised of monocultures due to specific plant adaptation needs and the gentle topographic gradients that permit competitive sorting (West 1983). Livestock grazing has had the most impact on this vegetation type since Euro-American settlement, altering the vegetation type towards non-palatable species. In addition, because of the lack of vegetation cover, this is one the few vegetation types in the western U.S. where fires were rare historically (West and Young 2000). Fires in recent years have emerged because of the establishment of non-native annuals, primarily cheatgrass (*Bromus tectorum*). Areas that have experienced grazing, increases in non-native annuals, and increases in fire frequency are outside their HRV. More site-specific studies are needed within the Sub-Region to get a better understanding of the spatial distribution of which desert shrub communities are outside their HRV.

The winterfat shrub steppe is also included within this vegetation type and is found within the Upper Gunnison Basin at higher elevations, 7,500-9,000 ft (2286-2896 m), than the desert shrub community described above (Johnston 1997). The winterfat shrub steppe is compromised of dwarf shrubs, primarily winterfat

(*Krasheninnikovia lanata*), and native grasses. Areas affected by historical grazing have experienced a shift in species composition towards grazing increasers [e.g., snakeweed (*Gutierrezia sarothrae),* rabbitbrush (*Chrysothamnus* spp.)] and are outside their HRV.

Spruce-Fir Forests

The spruce-fir vegetation type compromises 3,485,025 acres (10.22%) of the Sub-Region and 943,115 acres (30%) within the GMUG and 665,361 acres (32%) within the SJ National Forests (Fig 4-3). Spruce/fir forests are the highest elevational forests found within the Sub-Region between ~ 8200-11,000 ft (2500-3,380 m). Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) or corkbark fir (*Abies arizonica*) dominate this vegetation type. Corkbark fir reaches its northern distribution limit in the San Juan Mountains and is replaced by subalpine fir further north. Other species intermixed within the spruce/fir vegetation type include aspen (*Populus tremuloides*), blue spruce (*Picea pungens*), bristlecone pine (*Pinus aristata*), lodgepole pine (*Pinus contorta*), and other higher elevational tree species. A diversity of spruce/fir vegetation types exists at the Forest-scale, which is discussed later in this module.

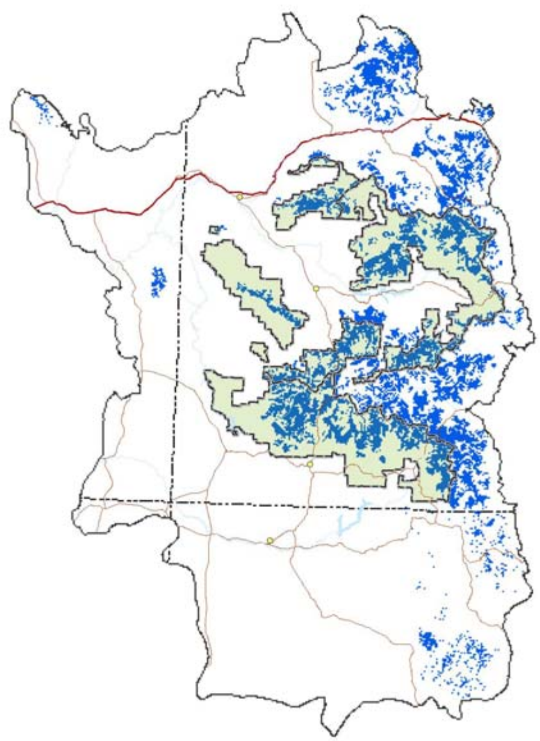


Figure 4-3 The Spruce-Fir GAP class in the subregion.
The Spruce-Fir GAP class in the subregion covers 3,485,025 acres (10.22%) of the subregion.

Stand dynamics within spruce/fir are strongly influenced by the autecologies of these two species. Both Engelmann spruce and subalpine fir are easily killed by fire. Subalpine fir is a short-lived species and rarely exceeds a lifespan of 250 years due to heart rot. Spruce has a longer longevity, often living over 500 years. Subalpine fir germinates successfully on fire prepared seedbeds and can exist under low light conditions better than Engelmann spruce. In contrast, Engelmann spruce is not an aggressive pioneer species (Bradley et al. 1992). Successional dynamics are also strongly influenced by site characteristics (elevation, topographic position, aspect, slope, soil type, and soil moisture). For example, spruce is generally more dominant in very wet or dry environments and fir in mesic environments (Peet 2000). A general successional model for this vegetation type includes four main stages. A stand initiation phase of herbaceous, seedling, and sapling establishment until 30-200 years, a stem exclusion phase comprising of dense tree cover until 150-300 years, a understory regeneration stage, which establishes two canopy layers and uneven age structure until 300-450 years, and a shifting mosaic stage that persists until a stand-replacing disturbance (Romme et al. 2003f).

Infrequent (~150-400 years), high intensity stand replacing crown fires coinciding with drought years characterize the fire regime in subalpine forests, although smaller, less intense fires may also occur in periods between larger fires (Veblen 2000; Kulakowski et al. 2003). These stand-replacing fires leave legacies of past disturbance and have large impacts on landscape patch dynamics at higher elevations. Successional development following major disturbances (fire, insect epidemics, blowdowns) can be highly variable depending on elevation, topographic position, aspect, soil substrate, slope, and disturbance interactions (Veblen et al. 1991; Peet 2000).

Determining HRV is more difficult in spruce/fir forests than other forest types because fire is generally lethal to trees leaving no fire scars and lags can exist between disturbance events and tree establishment. Therefore, post-fire age and size data combined are normally used to identify past disturbances (Veblen 1986). The majority of research in spruce/fir forests is near the Front Range in Colorado; however, a few studies within the Sub-Region exist (Baker and Veblen 1990; Veblen et al. 1994; Romme et al. 2003e; Kulakowski and Veblen 2004). Research in the San Juan Mountains and Grand Mesa, Colorado have shown that there has been a lack of stand replacing fires in spruce/fir forests since Euro-American settlement and that widespread stand-replacing fires occurred in the region around 1879 (Romme et al. 2003e; Kulakowski and Veblen 2004). However, visual accounts by Sudworth (1900) noted that some fires did occur in 1898 in previously burned over areas of spruce/fir forests on Grand Mesa and that the fires consumed regenerating aspen, downed logs, and patches of previously unburned spruce/fir. The general lack of stand-replacing fires in this region after Euro-American settlement could be interpreted as a result of fire suppression in the 20th century or it could be due to a lack of appropriate extreme weather conditions not occurring for stand-replacing fires to initiate. In addition, the long fire return intervals for this forest type (~150-400 years) suggests that spruce/fir forests are not outside their HRV (Kulakowski and Veblen 2004). Similar findings have been found on the White River Plateau (Veblen et al. 1994). Romme and others (2003e) have suggested however that the lack of fires in lower elevational forests during the 20th century may have prevented fires moving to higher elevational forests and therefore 20th century fire suppression would have had an effect on spruce/fir fire regimes in this region.

While spruce/fir forests may not be outside their HRV for fire frequency and intensity, patch size and patch number have significantly changed to smaller, more numerous patches as a result of roads and logging on a localized scale Romme and others (2003e). Manier and Laven (2002) found this same pattern for the entire western slope of the Rockies in Colorado. However, at the Sub-Regional-scale, changes in patch size and number would be minimal due to the limited amount of harvest in this forest type. In addition, this vegetation type historically consisted of large patches of different successional stages based upon past disturbances creating a mosaic of different spruce/fir structural classes and a heterogeneous landscape. More site-specific studies are needed within the Sub-Regional scale to better understand stand dynamics for this forest type, the spatial distribution of different successional stages across the landscape, and whether any stands are outside their HRV.

Sagebrush

The sagebrush vegetation type compromises 2,965,204 acres (8.7%) of the Sub-Region and 274,863 acres (9%) within the GMUG and 21,520 acres (1%) within the SJ National Forests (Fig. 4-4). The sagebrush vegetation type is the second semiarid shrub-steppe vegetation type covering the majority of the lower elevation areas within the Sub-Region. In addition, this vegetation type also occupies higher elevation sites above 7,500 ft (2286 m). *Artemisia tridentata* is acclimatized to cold temperatures, which is common for species that live in reas with high temperature seasonality. At increased elevations, canopy cover is denser and regeneration is often found in areas with sufficient snow cover or plant cover, with regeneration in lower elevations tending to occur in interspaces between shrubs (Loik and Redar 2003).

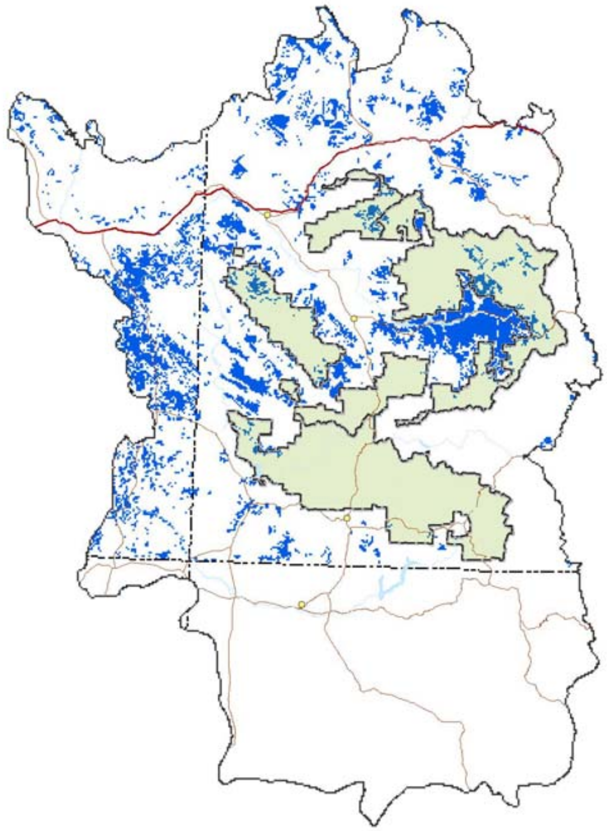


Figure 4-4 The Sagebrush GAP class in the subregion.
The Sagebrush GAP class in the subregion covers 2,965,204 acres (8.69%) of the subregion.

The Great Basin shrub steppe is found in lower elevations of the Sub-Region. This vegetation type is predominant in central and western Utah, northern Arizona, and throughout Nevada. Big sagebrush (*Artemisia tridentata* ssp. *tridentata*) along with grasses such as blue grama (*Bouteloua gracilis*), galleta (*Hilaria jamesii*) and other warm season sod grasses and forbs dominate this vegetation type. This vegetation type is also intermixed at lower elevations with pinon-juniper and ponderosa pine forests. Great Basin sagebrush is found in low areas with deep, well-drained soils that are sandstone or wind (eolian) derived (Floyd-Hanna et al. 1996). The shrub layer is generally less than 3.28 ft (1 m) tall and moderately spaced with interspaces of cryptogamic crusts. Big sagebrush generally occupies 70% of the plant cover regardless of its successional stage and can live to be 100 years of age (West and Young 2000). Big sagebrush does not resprout after being burned and regenerates solely through seeding with greatest survival in wetter than average years (Cawker 1980). Livestock grazing has increased soil erosion and the abundance of non-native annuals, particularly cheatgrass (*Bromus tectorum*). Sagebrush density generally increases with livestock grazing combined with a decrease in fire frequency (Daddy et al. 1988). In areas that burned recently with livestock grazing, non-native species are able to out compete native grasses and sagebrush from reestablishing. In addition, shrub species such as rabbitbrush (*Chrysothamnus* spp.), snakeweed

(*Gutierrezia sarothrae*), and horsebrush (*Tetradymia* spp.) increase with grazing and increased fire frequency because of their ability to resprout after fire (Tisdale and Hironaka 1981).

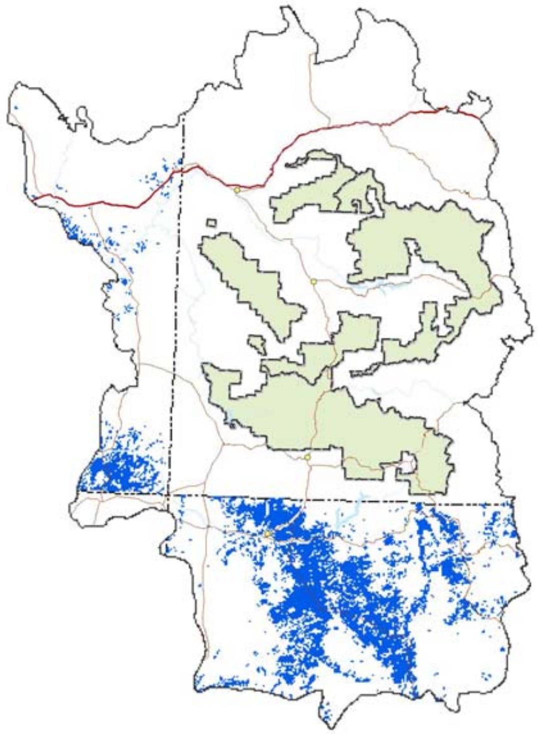
Sagebrush at higher elevations between 7,500-10,000 ft (2286-3048 m) is generally found on flat to rolling hills with well-drained clay soils and is characterized by dense shrubs with a significant herbaceous understory of bunch and sod grasses. This is a predominant vegetation type within the Upper Gunnison Basin, Colorado (Johnston 1997). Mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) is the most abundant species at higher elevations along with silver sage (*Artemisia cana*). Other shrubs species include rabbitbrush (*Chrysothamnus* spp.), bitterbrush (*Purshia tridentata*) and winterfat (*Krasheninnikovia lanata*).
Livestock grazing and increases in fire frequencies have had similar impacts as found in the Great Basin sagebrush type at lower elevations. A study by Wright and others (1979) suggested that pre-settement stand replacing fires occurred every 40-60 years, with smaller fires less often. Welch and Criddle (2003) have questioned fire frequencies of this interval and state that fires most likely were less frequent than often inferred for sagebrush communities based on sagebrush's longevity, highly flammable bark, low growth form, inability to resprout after fires, poor seed bank, and seeds lacking adaptations to high intensity fires (e.g., thick seed coat). Increases in non-native annuals have increased fire frequency, which allows these species to out compete native perennial grasses and sagebrush.

Areas with livestock grazing, increased non-natives and increaser shrub species, and increased fire frequency are outside their HRV. Successional dynamics indicate that these areas have crossed a critical threshold and have moved to an alternative community state or new equilibrium that can not be returned by removing livestock or decreasing fire frequency because of a significant decrease in soil organic matter leading to impoverished soils that are dominated by annual species (West and Young 2000; Norton et al. 2004).

Desert Grassland

The desert grassland vegetation type compromises 2,694,989 acres (7.9%) of the Sub-Region and is not present within the GMUG or the SJ National Forests (Fig. 4-5). Areas within the National Forests are of higher elevation, which limits the presence of desert grasslands. This vegetation type is predominant in lower elevational areas of Utah, Arizona, New Mexico, and Texas. The desert grassland community type includes typical semiarid grasslands and the grassland steppe. The grassland steppe community is comprised of different bunch and sod forming grasses and palatable dwarf shrubs found on deep, sandy soils in southeastern Utah (West and Young 2000). Fires were historically rare in the grassland steppe.

Figure 4-5 The Desert Grassland GAP class in the subregion.
The Desert Grassland GAP class in the subregion covers 2,694,989 acres (7.90%) of the subregion.



Bunch and sod forming grasses, forbs, small shrubs, and cryptogamic crusts, which help stabilizes soil from rain and wind erosion, dominate semiarid grasslands. Common grasses include three-awn (*Aristida* spp.), grama (*Bouteloua* spp), galleta (*Hillaria* spp.) and other cool and warm season grasses. Changes in precipitation, grazing intensity, and fire suppression are considered the most important factors influencing vegetation changes in semiarid grasslands (Rodriguez Iglesias and Kothmann 1997). Some of the semiarid grasslands within the Sub-Region are experiencing encroachment by shrubs and trees and are outside their HRV. A study by Allen (1989) illustrated encroachment of pinyon and juniper trees into adjacent grasslands in Bandelier National Monument, New Mexico.

Areas that have been affected by historical grazing have experienced a shift in species composition towards grazing increasers and non-native species similar to the desert shrub vegetation type. Many areas that have experienced grazing have been slow to recover and are outside their HRV.

Aspen

The aspen vegetation type compromises 2,272,984 acres (6.66%) of the Sub-Region and 648,763 acres (20%) within the GMUG and 273,778 acres (13%) within the SJ National Forests (Fig. 4-6). Aspen plays a crucial role in landscape diversity, spatial vegetation patterns, species habitat use and ecosystem processes (e.g., biogeochemical cycling) in an otherwise coniferous dominated landscape (Turner et al. 2003). Aspen is prevalent between ~ 6562-10,827 ft (2000-3300 m) and is generally found in areas with cool dry summers, cold winters, and deep, loamy soils with high nutrient availability within the Sub-Region. Aspen is an early colonizing, long-lived clonal species that depends on periodic disturbances for regeneration. The primary natural disturbance for regeneration is fire, although geomorphic events and wind can also initiate regeneration.

Two major aspen community types are present within the Sub-Region: stable and seral. Multilayered aspen stems (uneven aged) with no conifer invasion characterize stable aspen stands. These stands tend to be located at lower elevations in areas adjacent to ponderosa pine stands where the fire regime is frequent, although stable stands also exist at higher elevations (Romme et al. 2001). Frequent fires prevent conifer seedlings to reaching reproductive age, thereby eliminating a conifer seed source. Even in the absence of fire, these stands have remained stable because of a paucity of conifer seed sources, where at higher elevations seedlings could get established because of the prevalence for infrequent fires (Romme et al. 2003d). Studies in the San Juan Mountains also illustrated that stable stands were weakly associated with shale substrates rather than sandstone or igneous rock (Romme et al. 2003d). Fires within stables stands were most likely frequent surface fires that did not burn into the tree canopy. Fires of moderate intensity produce the highest amount of sprouting that allows stable aspen stands to persist (Parker and Parker 1983). The majority of aspen stands in the western U.S. is considered seral and is characterized by conifer invasion and lack multilayered (even aged) aspen stems (Mueggler 1985).

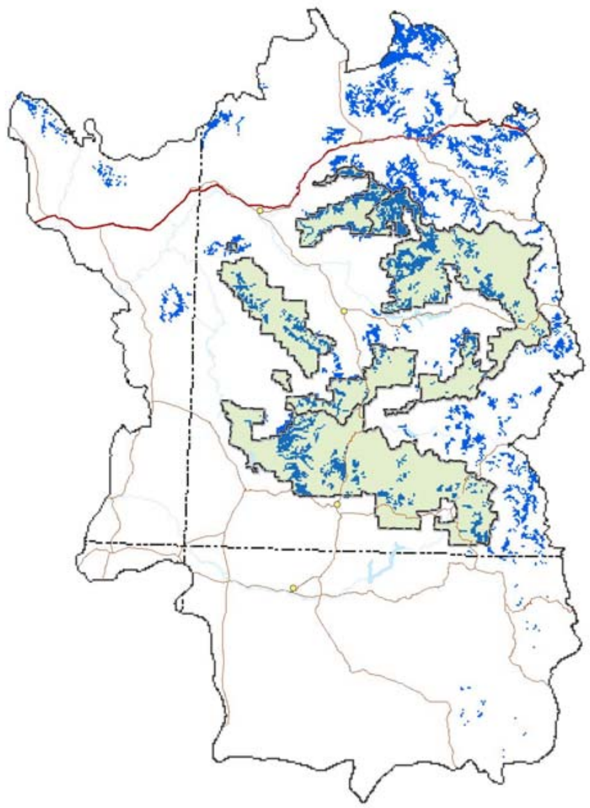


Figure 4-6 The Aspen GAP class in the subregion.
The Aspen GAP class in the subregion covers 2,272,984 acres (6.66%) of the subregion.

Seral stands are prevalent at higher elevations where spruce and fir become established normally decades after a significant disturbance event. In order for aspen to successfully regenerate, fire need to be of moderate to high intensity (Bartos et al. 1994). Seral aspen stands also exist lower in elevation in ponderosa pine and mixed conifer stands where white fir and Douglas-fir replace aspen.

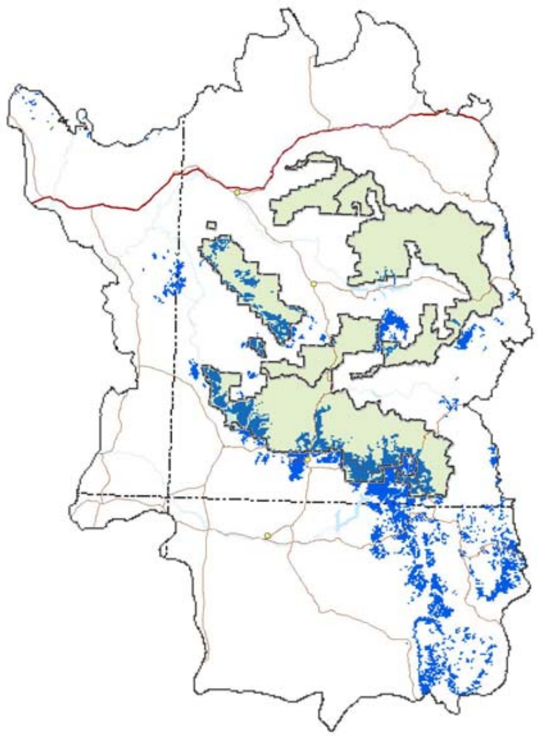
Numerous observations suggest that aspen abundance is experiencing widespread decline due to fire suppression and natural succession within the Sub-Region and therefore outside its HRV (Kay 1997; Bartos and Campbell 1998). Recent quantitative studies however illustrate a different perspective on the current status of aspen in the West (Manier and Laven 2002; Rogers 2002; Romme et al. 2003d; Elliott and Baker 2004; Kulakowski et al., in press). When assessing aspen decline, it is necessary to recognize that stable and seral aspen exist. Determining HRV is more difficult in aspen stands than other forest types because aspen trees do not create fire scars and fire is lethal to aspen. As a result, post-fire age cohorts must be used to identify fire regimes. Studies in the San Juan Mountains, Colorado indicate that some seral stands burned approximately every 70 years while other stands did not burn for over a century or longer (Romme et al. 2001). When assessing if seral aspen stands are outside of their HRV, it is important that stand age is taken into consideration. Numerous fires at the Sub-Regional-scale occurred between the late 1800's and early 1900's due to drought events during this timeframe. These fires allowed for numerous seral aspen stands to get reestablished in one localized period of time. Many of the stems in these stands currently are regarded as old (decadent) aspen, thus reaching their maximum longevity and natural succession to conifer trees is occurring. If one were to take a snapshot view of this current situation, it would appear that aspen are in fact declining due to fire suppression and other anthropogenic influences. Another interpretation of this situation, that incorporates a longer temporal scale, suggests that the lack of fire in the 20th century resembles fire activity in the late 1700's to early 1800's in the Southwest and therefore high densities of old aspen stands are most likely not outside their HRV; however, more detailed studies are needed to confirm or reject this hypothesis (Romme et al. 2003d; Kulakowski, in press).

Some studies indicate that on some level aspen loss is occurring within the Sub-Region either at the landscape scale (Rogers 2002), changes within stand age dynamics (fewer younger stand) in reference to pre-settlement stands (Romme et al. 2003d) or within seral stands of mixed conifer where conifer density and basal area has increased substantially between 1980-2001 on the Uncompahgre Plateau (Smith and Smith 2004). Other studies in the Sub-Region conclude that aspen is in fact not declining but rather increasing in spatial extent and abundance at the landscape scale. A study by Manier and Laven (2002) on the western slope of the Rockies in Colorado showed an increase in aspen over the past 100 years, along with an increase in conifer density, and a decrease in meadows, thus resulting in an overall decrease in patch diversity at the landscape scale and trends towards larger, contiguous forest patches and fewer group patches. Another study in the San Juans illustrated that aspen was increasing both in abundance and into new habitats (treeline and meadows), indicating a decrease in landscape patch diversity (Elliott and Baker 2004). Kulakowski and Veblen (2004) also determined that stable aspen stands on Grand Mesa were in fact increasing and that seral stands, where conifer encroachment was occurring, were returning within their HRV on the Mesa. In the San Juan Mountains stable stands, even without disturbances, exhibited continued recruitment with an uneven aged, multilayered canopy and were determined to be within their HRV (Romme et al. 2003d); a similar pattern was found on the Uncompahgre Plateau in stable stands (Smith and Smith 2004). While general trends indicate that stable and the majority of seral aspen stands at the Sub-Regional-scale are within their HRV, more site-specific studies are needed regarding HRV for aspen.

Ponderosa Pine

The ponderosa pine vegetation type compromises 2,064,177 acres (6.05%) of the Sub-Region and 198,087 acres (6%) within the GMUG and 543,573 acres (26%) within the SJ National Forests (Fig. 4-7). Pure ponderosa pine stands are found predominantly between 6,000-9,000 ft (1829-2743 m) on sandstone substrates. Ponderosa pine is the dominant tree species with occasional Rocky Mountain juniper at lower elevations and Douglas-fir at higher elevations. The understory in southwestern Colorado is dominated by shrubs with grasses intermixed. Gambel oak (*Quercus gambelii*) is the dominant shrub species along with snowberry (*Symphoricarpos rotundifolius*), Oregon grape (*Mahonia repens*), buckbrush (*Ceanothus fendleri*), gooseberry (*Ribes* spp.), and serviceberry (*Amelanchier utahens*is).

Figure 4-7 The Ponderosa Pine GAP class in the subregion.
The Ponderosa Pine GAP class in the subregion covers 2,064,177 acres (6.05%) of the subregion.



Identifying the HRV for ponderosa pine in the Southwest is much easier than for other forest types because of the ability for ponderosa pine trees to create fire scars, slow decomposition rates of dead and down woody material, and because of a massive tree irruption in 1919, which has lead to unprecedented dense stands in congruence with anthropogenic disturbances (Fulé et al. 1997). Fire statistics in this region since the 1600's show a clear pattern of an increasing number, size, and intensity of fires outside of the HRV for this forest type in the Southwest (Swetnam and Betancourt 1998). Many of the HRV studies for this region come from northern Arizona, although HRV studies also exist at the Sub-Regional-scale (Swetnam and Baisan 1996; Touchan et al. 1996; Brown and Shepperd 2003; Romme et al. 2003b). While a generalized HRV assessment is practical at the Sub-Regional-scale, it is important to recognize that other ponderosa pine HRV assessments in Colorado have shown that some ponderosa pine stands historically had stand-replacing mixed and high-severity fire regimes (Brown et al. 1999; Kaufmann et al. 2000), which likely occurred within our analysis area as well.

Quantitative reconstruction of pre-settlement (1870-1890) southwestern ponderosa pine forests indicates open, park-like stands. Covington and others (1997) determined that pre-settlement ponderosa pine stands consisted of approximately 60 stems/ha at the Gus Pearson Natural Area, Arizona. Romme and others (2003b) also found that ponderosa pine forests contained an average of 11 to 98 trees/ha during the late 1800s in the San Juan Mountains, Colorado. Similarly, several early National Forest inventories showed a range of 7 to 116 stems/ha in pre-settlement southwestern ponderosa pine forests in northern Arizona (Covington and Moore 1994). Pre-settlement southwestern ponderosa pine forests were regulated by a vital ecological attribute--fire. Low intensity surface fires carried by grass and shrubs recurred every 5-20 years in southwestern ponderosa pine ecosystems prior to Euro-American settlement and played a major role in regulating the structure, composition, and stability of these ecosystems (Swetnam and Baisan 1996, Fulé et al. 1997). Specifically, in the San Juan Mountains fire intervals are between 10-20 years (Romme et al. 2003b) and on the Uncompahgre Plateau between 10-25 years (Brown and Shepperd 2003). These frequent, low-intensity fires, along with shrub and grass competition, prevented dense ponderosa pine regeneration and maintained the open, park-like structure of pre-settlement ponderosa pine stands.

Three main anthropogenic influences are responsible for the dramatic alterations in the structure and function of ponderosa pine forest ecosystems: grazing, logging, and fire exclusion (Covington et al. 1997). In addition, climatic oscillations may have also altered ponderosa pine forest ecosystems over the past century (Covington and Moore 1994). Decadal-scale climatic variability is a primary driver for ecological processes. Shifts in climate towards warm, wet periods have been suggested as the causal mechanism for the pulse of pine recruitment in the early 1800's that corresponded to the longest intervals between fires in numerous areas in the Southwest, pine recruitment in 1919, and recruitment since 1976 due to anomalous warming of the tropical Pacific (Swetnam and Betancourt 1998). The latest recruitment since 1976 followed the worst drought in the Southwest over the past 1000 years during the 1950's. The increased stand density in ponderosa pine over the past ~120 years without a long-term climatic perspective may suggest that anthropogenic changes were the only underlying factor for this change in structure (Swetnam et al. 1999). In addition, shifts in climate can also be associated with changes in fire regimes. In the Southwest, a decrease in fire frequency between 1780-1830 coincided with a decrease in the El Nino-Southern Oscillation (ENSO) (Swetnam and Betancourt 1998). Interannual variability in moisture availability instead of drought alone plays a significant role in the occurrence of large fire years in the Southwest. Generally, large fires years in ponderosa pine are associated with one-two above-average winter and spring precipitation years followed by a drought year because the wet years permit enough fine fuels to accumulate and subsequently dry out (Swetnam and Baisan 1996).

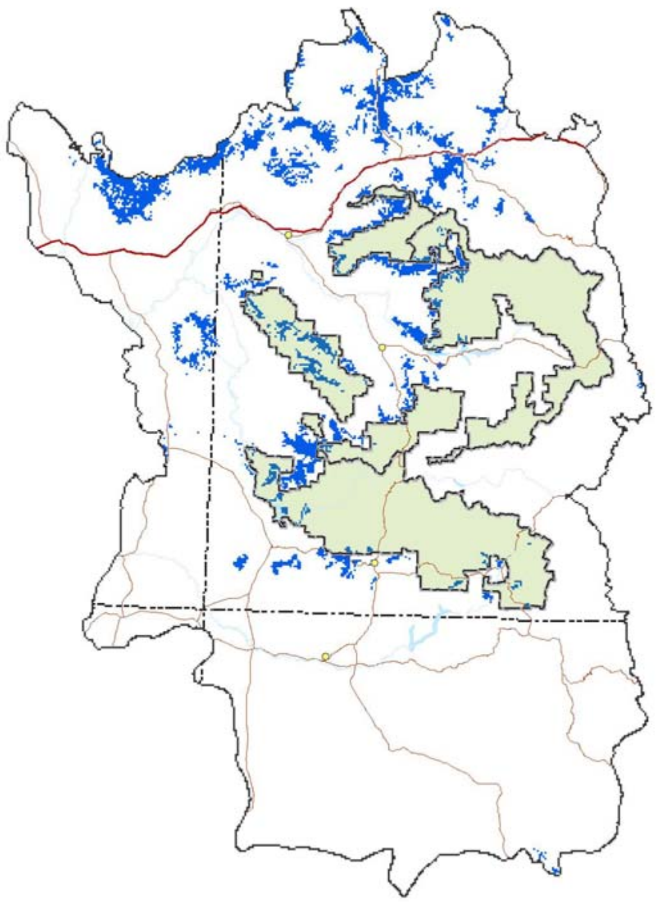
Anthropogenic effects associated with Euro-American settlement have resulted in numerous young, small trees; fewer old, large trees; increased forest fuel loads; lower herbaceous production and diversity; altered fire regimes; and changed wildlife habitats in southwestern ponderosa pine forests (Covington and Moore 1994; Covington et al. 1997; Fulé et al. 1997). As a consequence of both anthropogenic influences and climatic oscillations there has been an overall change in forest structure and function altering soil moisture availability, decomposition rates, tree health (e.g., carbon, water, nutrient, growth, and insect resistance), nutrient cycling, microbial populations, net primary production and susceptibility to invasive exotics (Covington and Sackett 1984; Wright 1996; Covington et al. 1997; Feeney et al. 1998; Crawford 2001). All of these factors indicate that a large portion of ponderosa pine stands within the Sub-Region-scale are outside of their HRV.

Ponderosa pine forests at the Sub-Regional-scale were not homogenous open park-stands prior to Euro-American settlement, but rather it appears that some dense stands of small ponderosa pine trees similar to the stands that dominate the landscape today existed. Therefore when identifying management and restoration goals for ponderosa pine, site-specific HRV assessments are necessary to maintain HRV on both a stand and landscape scale within the analysis area.

Deciduous Oak

The deciduous oak vegetation type compromises 1,760,036 acres (5.2%) of the Sub-Region and 207,282 acres (7%) within the GMUG and 65,747 acres (3%) within the SJ National Forests (Fig. 4-8). This vegetation type is often considered part of the Mountain Shrubland vegetation type but can be distinguished by the predominance of Gambel oak (*Quercus gambelii*) along with other shrubs such as serviceberry (*Amelanchier utahens*is) and snowberry (*Symphoricarpos rotundifolius*), grasses, and forbs. This vegetation type is prominent along the southern and western slopes of the San Juan Mountains and on the Uncompahgre Plateau.

Figure 4-8 Deciduous Oak GAP class in the subregion.
The Deciduous Oak GAP class in the subregion covers 1,760,036 acres (5.16%) of the subregion.



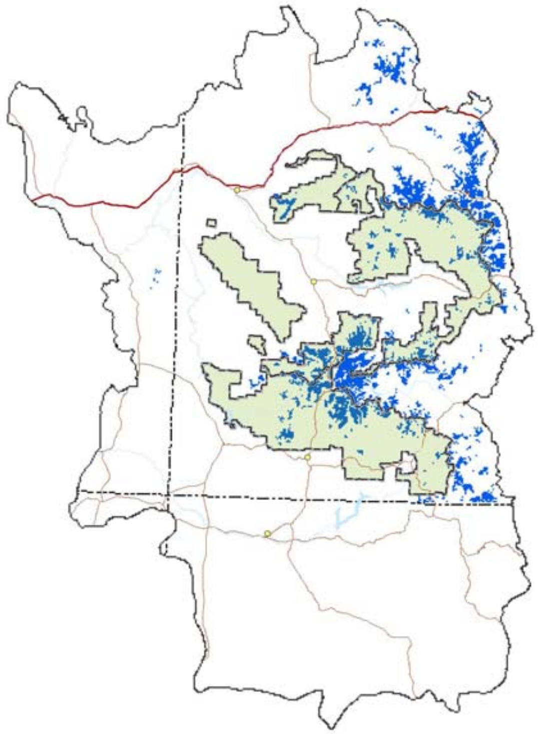
Deciduous oak can be found between 6,000-9,000 ft (1829-2743 m) and grows either as dense shrubby patches or trees. Gambel oak grows on a diversity of soil types from fine to heavy textured loams, deep alluvial sand, and coarse gravel (Christensen 1955). Gambel oak often forms large underground systems of lignotubers and rhizomes that are capable of resprouting rapidly and vigorously after fire. Often these clones are much older than the age of the aboveground stems. The ability of these underground systems to remain viable for long periods of time without disturbances has yet to be determined (Tiedemann et al. 1987). Serviceberry, the dominant shrub associated with the deciduous oak type also resprouts vigorously after fire. Fire plays a major role in the maintenance of this vegetation type as evident by the dominant shrub adaptations to fire. One study within the Sub-Region, in Mesa Verde National Park, has estimated a fire turnover time of approximately 100 years and that these fires are generally of high intensity removing the majority of aboveground biomass (Floyd et al. 2003). This vegetation type has been hypothesized to represent a recovery stage from fire and that without fire, trees such as pinon pine and Utah juniper at lower elevations and ponderosa pine, Douglas-fir and white fir at higher elevations would replace this vegetation type (Floyd-Hanna et al. 1996). Historically this vegetation type would be in different stages of recovery depending on the timing, frequency, and intensity of the disturbance and its relation to climatic fluctuations (Romme et al. 2003f). Increasing fire frequency within this vegetation type would lead to an increase in shrub density. The variability, proportion, and distribution of these different successional stages historically are unknown and therefore assessing whether this vegetation type is outside its HRV for the Sub-Region is difficult to assess.

Alpine

The alpine vegetation type compromises 1,606,281 acres (4.7%) of the Sub-Region and 397,266 acres (12%) within the GMUG and 252,406 acres (12%) within the SJ National Forests (Fig. 4-9). The alpine vegetation type is distinct from other vegetation types because it occupies the upper elevational limits of forests ~11,500 ft (3505 m) on high mountain summits, slopes, and ridges (Billings 2000). Climatic variables such as intense cold, wind, solar radiation, and snow dictate the types of vegetation that can exist. Dwarf shrubs, prostate herbaceous forbs, bunchgrasses, lichens, and mosses characterize the alpine vegetation type. Topography (aspect, slope, position) plays a strong role on the diversity of alpine vegetation types because of its influence on solar radiation, solifluction, and snow/water accumulation (Jamieson et al. 1996). A difference in snow accumulation is the most influential factor influencing alpine vegetation types. Topography creates a mosaic of different depths of snow accumulation, which influences growing season lengths and moisture abundance (Walker el al. 1993). Some general alpine vegetation types from little to high snow accumulation include fellfields (windswept areas), dry meadows (low snow), moist meadows (base of snowfields), dwarf shrublands (depressions), and wet meadows (flat snowmelt drainages) (Bowman et al. 2002).

The majority of alpine areas within the Sub-Region are either designated as wilderness or roadless and therefore significantly less disturbed than other vegetation types. However, alpine environments are highly susceptible to soil disturbance (compaction, erosion) and are slow to revegetate due a limited growing season, strong winds, drought, and high evaporation rates. Livestock grazing, mining, and recreation activities result in soil disturbance that subsequently alters species composition, abundance, biomass, nutrient cycling, and water availability (Redders 2003). Atmospheric depositions of pollutants also are a major disturbance to alpine environments. Elevated nitrogen deposition is present in the southern San Juan Mountains downwind of large coal-fired power plants. Nitrogen deposition results in changes to vegetation structure and function by favoring nitrogen responsive species (grasses) over non-nitrogen responsive species (Bowman et al. 1993). Identifying HRV assessments for the alpine vegetation is difficult because of a paucity of historical information. Areas of known soil disturbance and thus vegetation changes from livestock grazing, recreation, and mining are outside their HRV. More site-specific studies are needed to assess changes to the alpine from atmospheric deposition and climate change.

Figure 4-9 The Alpine GAP class in the subregion.
The Alpine GAP class in the subregion covers 1,606,281 acres (4.71%) of the subregion.



Mixed Conifer

The mixed conifer vegetation type compromises 1,209,315 acres (3.54%) of the Sub-Region and 92,296 acres (3%) within the GMUG and 145,760 acres (7%) within the SJ National Forests (Fig. 4-10). Mixed conifer forests are one of the most variable and complex forest types in the western United States. Numerous research studies have been conducted in mixed conifer forests of the Sierra Nevada and Cascade Ranges; however, there is a paucity of analogous information for southwestern mixed conifer forests (White and Vankat 1993; Wu 1999; Romme et al. 2003; Mast and Wolf 2004), which represent approximately 1 million ha of forested land in the Southwest (Dieterich 1983). Southwestern mixed conifer forests are found between the lower elevational ponderosa pine and the higher elevational spruce/fir forest type. In southwest Colorado and northwest New Mexico, two broad mixed conifer forest types have been recognized: the warm, dry mixed conifer and the cool, wet mixed conifer types. Being a continuum along an elevational gradient, the warm, dry mixed conifer tends to be more similar in community structure and function with the ponderosa pine forest type and the cool, wet mixed conifer more similar with the subalpine forest type (Romme et al. 2003c).

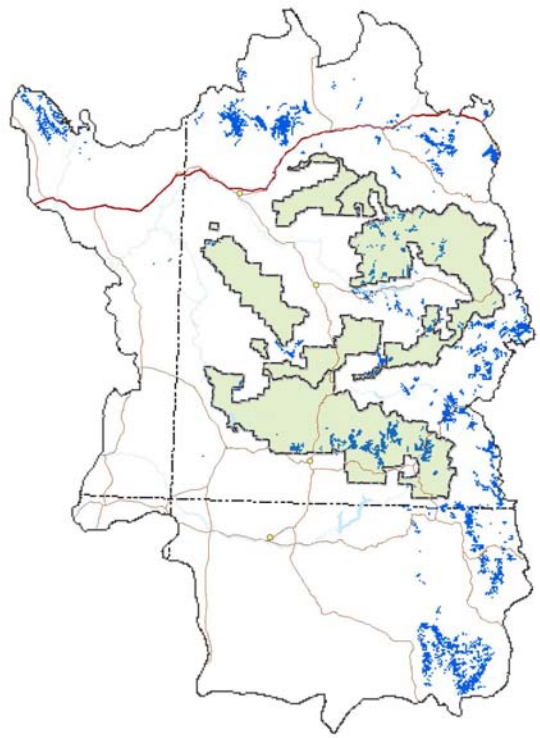


Figure 4-10 The Mixed Conifer GAP class in the subregion.
The Mixed Conifer GAP class in the subregion covers 1,209,315 acres (3.54%) of the subregion.

The warm-dry mixed conifer type is found between ~7500-9000 ft (2286-2743 m) and is dominated by ponderosa pine, Douglas-fir, white fir, and occasionally aspen. The abundance of individual tree species is dependent on local site characteristics (soil, aspect, slope, topographic position) and natural and anthropogenic disturbance history. Gambel oak, serviceberry, buckbrush, snowberry, mountain lover (*Paxistima myrsinites*), kinnikinnik (*Arctostaphylus uva-ursi*), grasses, and forbs dominate the understory. Different successional trajectories are present within the warm-dry mixed conifer type. For example, some warm-dry mixed conifer stands have originated from the deciduous oak vegetation type following a stand-replacing fire, while other stands formed following a moderate intensity burn, and many current stands are the result of natural succession and fire suppression and other anthropogenic disturbances combined (Redders 2004). General patterns of change since pre-settlement are well documented in stand structure within the warm, dry mixed conifer forest type in the western U.S. (White and Vankat 1993; Mast and Wolf 2004). Specifically, there has been a shift in species composition and abundance to shade tolerant species such as white fir (*Abies concolor*) and Douglas-fir (*Pseudotsuga menziesii*) at the expense of the shade intolerant but more fire resistant ponderosa pine and increased vertical and horizontal fuel continuity. This shift in stand structure has been attributed to an alteration in the natural fire regime (non-lethal fires at 20-50 yr intervals and rare lethal fires greater than 100 yr intervals) (Wu 1999; Romme et al. 2003c).

The cool, wet mixed conifer type is found between ~8,500-10,000 ft (2591-3048 m) and is dominated by Douglas-fir and white fir along with Engelmann spruce and subalpine fir with the occasional presence of aspen and blue spruce. The abundance of individual tree species is dependent on local site characteristics (soil, aspect, slope, topographic position) and natural and anthropogenic disturbance history. Serviceberry, snowberry, elderberry (*Sambucus microbotrys*) bush honeysuckle (*Distegia involucrata*), grasses, and shade loving forbs (Pyroloceae) dominate the understory. Similar patterns of change found within warm-dry mixed conifer are not found within the cool, wet mixed conifer forest type. Infrequent (greater than 100 yr intervals) moderate to high severity stand-replacing burns characterize the fire regime of these forests (Romme et al. 2003c). There is a significant lack of research in this forest type regarding HRV and extant stand dynamics; therefore, only broad generalizations can be made at the Sub-Regional-scale.

Changes in forest structure and processes in warm-dry mixed conifer forests indicate they are outside their HRV. In contrast, it appears that the majority of cool, wet mixed conifer forests are within their HRV, although more site-specific research is needed to confirm this statement.

Ten of seventeen GAP classes cover just over 90% percent of the subregion. These classes are listed and shown in Figures 3-1 to 3-10 above. The remaining seven classes, combined cover about 5% of the subregion. Combined the seventeen vegetation classes cover about 95% of the subregion. These are described in the following sections and are combined in Figure 4-11.

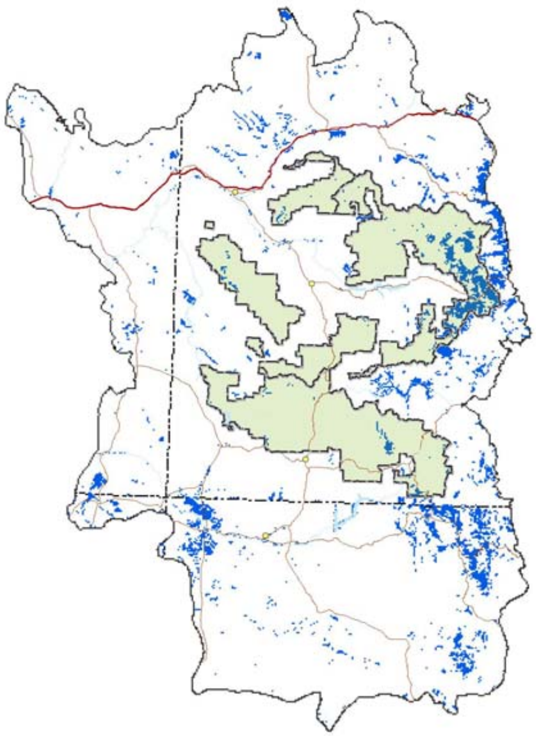


Figure 4-11 Seven GAP vegetation classes in the subregion combined.
Seven GAP vegetation classes in the subregion combined cover 1,761,625 acres (5.16%) of the subregion. These seven combined with the ten classes described above cover just over 95% of the subregion.

Mountain Grassland

The mountain grassland vegetation type compromises 589,306 acres (1.7%) of the Sub-Region and 6,956 acres within the GMUG and 32,274 acres within the SJ National Forests. Mountain grasslands are found interspersed between forested vegetation types between ~ 7000-10,500 ft (2286-3200 m). A variety of factors including topography, geology, soil, climate, and disturbances (fire, mass movement, and snow) are responsible for the presence of meadows in between forested vegetation (Debinski et al. 2000). Soil texture has been identified to be one of the most critical factors explaining the presence of meadows at lower elevation areas, generally being found on fine-textured alluvial or colluvial soils where adjacent forested areas are found on coarse-texture, rocky soils (Peet 2000). At higher elevations, soil moisture appears to influence the presence of meadows. Areas with excessive moisture near streams, slope bottoms, or on substrates that keep water at the surface are generally dominated by grasses, sedges, and forbs (Peet 2000). Three dominant mountain grassland types have been identified for Southwest Colorado and Northwest New Mexico: the Arizona fescue (*Festuca arizonica*) type that is associated with ponderosa pine and warm-dry mixed conifer; the Thurber fescue (*Festuca thurberi*) type that is associated with cool-moist mixed conifer and spruce fir; and, the Kentucky bluegrass (*Poa pratensis*) type that occurs at all elevations (Redders 2003b). Other native grass species associated with the Arizona fescue type include mountain muhley (*Muhlenbergia montana*), Parry oatgrass (*Danthonia parryi*), junegrass (*Koeleria macrantha*), bottlebrush squirreltail (*Elymus elymoides*), and muttongrass (*Poa fendleriana*). Other native grass species associated with the Thurber fescue type include junegrass, Arizona fescue, needle and thread (*Heterostipa comata*), blue wild rye (*Elymus glaucus*) and timber oatgrass (*Danthonia intermedia*).

Meadow/forest ecotones are areas that show rapid ecological change and therefore can serve as indicators of environmental changes through shifts in species composition and abundance (Harte and Shaw 1995). For example, tree encroachment of mountain meadows in surrounding areas of the Sub-Region has been well documented in juniper ecotones (Johnsen 1962), ponderosa pine ecotones (White 1985; Mast et al. 1997), aspen (Buell and Buell 1959) and the subalpine (Dunwiddle 1977). These encroachments are generally associated with changes in natural disturbance regimes, primarily fire (Turner 1975). Fire frequency intervals would be similar to adjacent forest type intervals with more frequent fires at mid elevation ponderosa pine and warm-dry mixed conifer forests and less frequent fires with cool-moist mixed conifer and spruce/fir forests. There is little evidence for fires occurring in meadows independent of fires in adjacent forested areas (Romme et al. 2003f). Livestock grazing since Euro-American settlement has also changed native species composition and abundance, altered natural disturbance processes (fire) and nutrient cycling, increased erosion, and increased non-native invasive species (Redders 2003b). Areas within the Sub-Region that have experienced changes to fire regimes, livestock grazing, introduction of non-native species, and tree invasions are outside their HRV. A paucity of historical quantitative data for mountain grasslands however makes it difficult to assess specific changes to community structure and function.

Lodgepole Pine

The lodgepole vegetation type compromises 462,098 acres (1.35%) of the Sub-Region and 236,601 acres (7%) within the GMUG and 0 acres within the SJ National Forests. Lodgepole pine generally occurs between 8,000-10,000 ft (2438-3048 m) and reaches it southern boundary of its range at the middle of the Gunnison Basin (Johnston 1997). Lodgepole pine south of this area within the Sub-Region were planted in the early 20th century following severe fires in the late 1800's due to a concern with the lack of natural tree regeneration (Romme 2003). Lodgepole pine has a lifespan of ~250 years, which is related to the frequency of stand-replacing fires that occur within this vegetation type (Mehl 1992). The understory of lodgepole pine is usually poorly developed with low species diversity due to a dense tree canopy cover and low soil fertility.

Lodgepole pine plays a similar successional role as aspen in higher elevational forests, but is found more predominantly in the northern latitudes and aspen is more dominant in southern latitudes (Allen et al. 1991). In environments where lodgepole pine is seral, shade tolerant species (Douglas-fir and subalpine fir) are able to replace lodgepole because of its intolerance to shade and lack of disturbances, generally fire, and inability to successfully get established without its mineral seedbed requirements (Bradley et al. 1992). A general successional model for lodgepole pine to Douglas-fir and spruce/fir stands includes the following stages. Following a stand-replacing fire, an initial herbaceous and shrub layer establishes. This stage is then followed by a dense stand of even-aged seedlings and saplings with serotinous stands regenerating faster than open-coned stands. Regeneration does not necessarily occur all at one time and is influenced by the distance to the nearest seed source, amount of competing herbaceous understory, soil texture, and availability of resources resulting in the initial cohort occurring over a period of 30-50 years (Peet 2000). Following this stand initiation stage, without a disturbance, a dense stand forms and density is decreased through competition mortality or low-moderate intensity disturbances (insects or fire), which opens up the stands allowing shade tolerant (Douglas-fir and subalpine fir) trees to invade openings. If no fire occurs, a stable Douglas-fir or spruce/fir stand eventually exists until a stand-replacing fire returns the stand to a herbaceous/shrub state (Bradley et al. 1992). Elevation is the primary factor determining whether lodgepole pine is seral to Douglas-fir or spruce/fir stands with Douglas-fir at lower elevations [<9500 ft (2896 m)] and spruce/fir at higher elevations. Topographic variables (aspect, slope, and topographic position) also influence successional pathways.

Lodgepole pine can also form stable stands similar to aspen, which evolved with stand replacing crown fires to promote seed establishment from serotinous cones, although open cones also exist. Stable lodgepole stands have been associated with thin, well-developed soils, cold microclimate, and in areas where shade tolerant species do not exist (Mehl 1992).

There is a paucity of site-specific research regarding lodgepole pine primarily because of its limited natural distribution and abundance at the Sub-Regional-scale. The majority of naturally occurring lodgepole regenerated during the late 1800's-early 1900's, similar to many aspen stands to the south, from drought initiated fire events. It appears that these stands are near the high end of their HRV for total landscape coverage in the White River N.F. and are dominated by older age classes (USDA 2002).

Mountain Shrubland

The mountain shrubland vegetation type compromises 222,042 acres (0.65%) of the Sub-Region and 13,566 acres within the GMUG and 3,032 acres within the SJ National Forests. This vegetation type is prominent along the southern and western slopes of the San Juan Mountains and on the Uncompahgre Plateau, which is similar to deciduous oak. This vegetation type is distinguished from deciduous oak by being dominated by a diversity of shrubs. Some of the dominant shrubs include mountain mahogany, bitterbrush, gooseberry (*Ribes cereum*), skunkbrush (*Rhus trilobata*), snowberry, Gambel oak and serviceberry. Shrubs that are unique to the mountain shrubland community include fendlerbush (*Fendlera rupicola*) and squaw-apple (*Peraphyllum ramossissimum*).

Fire is the major disturbance for this vegetation type and may be of frequent, low intensity or infrequent, high intensity but specifics on fire return intervals are unknown (Romme et al. 2003g). In addition, mountain shrub communities are commonly found on northern slopes where snow can accumulate and slide during warmer periods resulting in periodic disturbances that allow only flexible plant species to survive and therefore favoring shrubs over trees (Floyd-Hanna et al. 1996). The mountain shrub community has been hypothesized to represent a recovery stage from disturbance and that without frequent disturbances trees such as pinon pine, Utah juniper, and ponderosa pine would replace this vegetation type (Floyd-Hanna et al. 1996). Historically this vegetation type, similar to deciduous oak, would be in different stages of recovery depending on the timing, frequency, and intensity of the disturbance and its relation to climatic fluctuations. The variability, proportion, and distribution of these different successional stages historically are unknown and therefore assessing whether this vegetation type is outside its HRV within the Sub-Region is difficult to assess.

Woody Riparian/Wetlands

The woody riparian/wetland vegetation type compromises 118,599 acres (0.35%) of the Sub-Region and 8,911 acres within the GMUG and 709 acres within the SJ National Forests. Riparian woodlands and wetlands are defined as the transition between the aquatic environment and the upland terrestrial environment where the water table is generally at or near the surface or the land is covered with water (Cowardin et al 1979). This vegetation type occurs at all elevations throughout the Sub-Region as diverse vegetation types. Riparian woodlands and wetlands in the arid southwest are recognized as one of the most limited and vulnerable plant communities while at the same time the most biologically diverse. In the past century in Arizona and New Mexico alone, over 90 percent of riparian woodlands and wetlands have been lost (Johnson 1989). Riparian vegetation provides stream bank stabilization, water quality protection, fish and wildlife habitat, and flood control.

Four broad categories of wetlands have been identified within the Sub-Region: peatlands, marshes, wet meadows, and riparian. Peatlands are areas that accumulate decayed plant material, and the only peatland type within our Sub-Region is a fen. Fens are located generally above 8,000 ft (2438 m) in areas where groundwater intercepts the soil surface in low points of the landscape and inflow is maintained year round. These areas are rare within the Sub-Region and are confined to a few small areas on the Grand Mesa. Marshes are located adjacent to bodies of water that don't flow, such as lakes or ponds, or near slow-flowing bodies of water. Bullrushes, sedges, and cattails characterize this vegetation type. Wet meadows are the most abundant wetland type within Colorado and are grassland areas that are waterlogged year-round without standing water for the majority of the year. These wetlands are found at higher elevations within the Sub-Region. Riparian areas are the most easily recognized wetland type and are associated with moving water that is occasionally flooded. Riparian areas are found in desert to alpine vegetation types and are associated with cottonwood (*Populus* spp.) at low to middle elevations, grading into aspen, boxelder (*Acer negundo*), alder (*Alnus* *incana*), willows (*Salix* spp.) and a variety of conifer species (*Pinus* spp. *Picea* spp. and *Abies* spp.) at mid to high elevations.

Riparian woodlands and wetlands have experienced dramatic changes since Euro-American settlement due to livestock grazing, mining, logging, road building, water diversions, and beaver extirpation. These changes include changes to native species composition and abundance and introduction of non-native invasives such as tamarisk (*Tamarisk ramosissima*) and Russian olive (*Elaeagnus angustifolia*) that have altered community function. Specific quantitative information for how these communities have changed however is poorly documented; therefore, identifying the HRV for these types is difficult (Redders, 2003a). Within the Sub-Region there are dozens of specific riparian woodland and wetland plant associations, which is beyond the scope of this discussion (see the Forest-Scale Riparian Woodland/Wetland section for descriptive riparian woodland/wetland vegetation types). In addition, Redders (2003a) provides an in-depth discussion of these community types for the San Juan and Johnston (year/ref?) for the GMUG National Forests.

Greasewood

The greasewood vegetation type compromises 63,033 acres (0.18%) of the Sub-Region and is not present within the GMUG or SJ National Forests. This vegetation type is highly restricted in its distribution by hydrology, soil salinity, and soil texture. The dominant plant is black greasewood (*Sarcobatus vermiculatus)* and can be found in association with occasional salt tolerant grasses. Black greasewood is a phreatophyte that generally dominates as a monoculture over large stretches of valleys where salts and water accumulate every year (West and Young 2000). Livestock grazing in greasewood has decreased grass cover and increased shrub cover. Fires historically burned within greasewood communities but fire intervals, along with general HRV conditions, are poorly known (Romme et al. 2003f). Areas that have recently burned generally have higher grass cover than unburned areas and greasewood resprouts through belowground surviving structures. More studies are needed at the Sub-Region level to establish HRV for this vegetation type.

The seventeen vegetation classes described above cover about 95% percent of the subregion. The remaining 5% of the subregion are comprised of crops, urban areas and water bodies Figure 4-12.

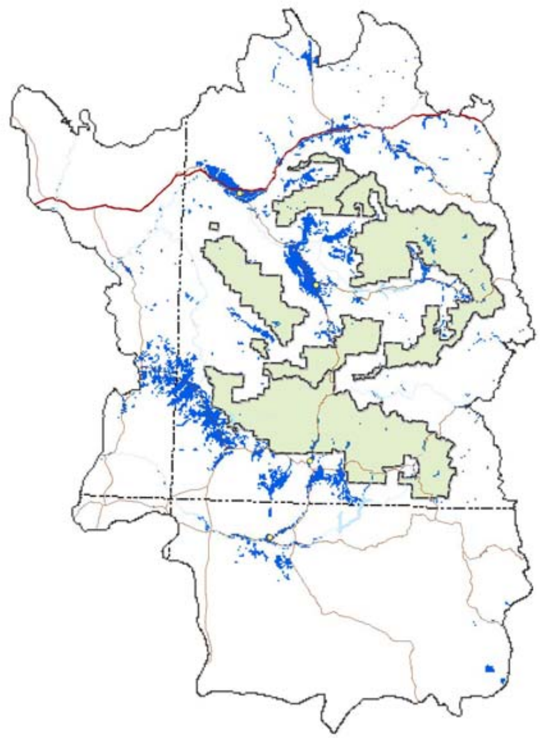


Figure 4-12 Three GAP non-vegetation classes in the subregion combined. Three GAP non-vegetation classes in the subregion combined cover 1,526,301 acres (4.47%) of the subregion.

General HRV Models

Community structure is the result of multiple interactions among species, environmental variability, perturbations and chance events (Samuels and Drake 1997). Spatial and temporal scales are crucial to understanding the dynamic nature of communities and whether communities converge towards a single state or multiple states. Two well-known generalized models can be used to assess how communities respond to perturbations over time and are suitable for HRV assessment of numerous vegetation types: the single stable state model and the alternative community state model (Figure 1).

The concept of alternative community states originated from the work of Lewontin (1969) who was investigating the concept of stability. Lewontin (1969) explained community structure through two opposing ways. His first explanation focused on the importance of history in explaining patterns of species, populations and communities, which is associated with multiple stable points. His second explanation emphasized the importance of fixed forces without any reference to historical events in explaining patterns of species, populations and communities (single stable point) (Lewontin 1969). A community is considered to have one unique stable point if it returns to its original state from all initial conditions following any disturbances, "like a marble seeking the bottom of a cup" (May 1977). In contrast, a community is considered to have alternative community states if a system's trajectory is influenced by the initial conditions. In this scenario, a system may return to its original state following a small perturbation or it may change to a new state following a large perturbation "like the ball in a pin-ball machine" (May 1977). At advanced stages of deterioration, removing anthropogenic stressors is usually inadequate to stop or slow down continuing degradation and therefore early detection of changes in ecosystem structure and function are necessary (de Soyza et al. 2000). For example, the recovery of native perennial grasses in regions of the Great Basin desert that have been invaded and dominated by cheatgrass (Bromus tectorum) appears unlikely even without the presence of domestic grazing (Knapp 1996).

A system's history matters under the concept of alternative community states and is unimportant under the concept of a single community state. History is important because it is needed to interpret observed differences in the structure of communities in the same locality. The disturbance history of an area will often interact with competitive interactions among species, and the intensity and timing of the disturbance will affect community dynamics (Hughes 1989; Berlow 1997). In addition to the knowledge of historical perturbations, knowledge of a system's historical and current species pool (the number of species potentially able to exist in a particular community) is important in explaining variation in communities (Partel and Zobel 1998). Law and Morton (1993) suggested that the ability for a system to have alternative states was more likely with a large species pool and therefore suggested that species-rich systems with strong interactions were more likely to have alternative states than species poor systems. In addition, the order of species arrival plays an important role in the community structure of alternative states (Drake 1991). As stated by Dudgeon and Petraitis (2001), "in many ecosystems, the distributions and abundances of organisms carry the signatures of historical events", and the history of species' relationships that no longer exist may be the invisible keys to understanding extant community structure (Samuel and Drake 1997).

Understanding if a system has alternative community states is important for land managers trying to reverse degraded systems because of the potential implications this knowledge can have on determining if systems are within their HRV and if they are not how the systems can be restored to be within their HRV. If all communities converged to a single stable point, returning systems within their HRV would be much easier because all degradation in systems would be along a linear gradient to the inevitable solution with time being the only preventative factor (Samuels and Drake 1997). In contrast, if all systems were represented by divergence, the communities would be idiosyncratic and structured by chance making the return of systems to their HRV impossible and entirely due to chance (Samuels and Drake 1997). A crucial component to understanding the ability for multiple stable points in a community is the critical threshold at which change occurs in a system. Wissel (1984) warns that dramatic changes at thresholds in systems with alternative states may be irreversible if caused by man. He cites examples of increased eutrophication in lakes and introduction of noxious weeds as changes that may be irreversible (Wissel 1984). As a result, being able to predict critical thresholds for irreversible events is crucial to preventing permanent changes (Knowlton 1992). A critical threshold has two characteristics: 1) it is a boundary in space and time between two or more states; and 2) the shift across the boundary is not reversible within a reasonable timeframe without intervention by humans (Friedel 1991).

Understanding whether major vegetation types at our Sub-Regional-scale respond to the single stable state model or the alternative community state model is important in the context of understanding HRV assessments and how HRV can be incorporated into management decisions. For example, the importance of understanding these two opposing models can be illustrated in ponderosa pine forests. The effects of anthropogenic impacts such as fire suppression have resulted in numerous young, small trees; increased forest fuel loads; and lower herbaceous production and diversity (Covington et al. 1997). Many studies that have reintegrating fire into ponderosa pine ecosystems have not been able to return the system to a single stable community that consists of trees intermixed with shrubs and herbaceous plants. Instead, other mechanisms such as tree removal through thinning have been necessary along with reintegrating fire, which suggests that ponderosa pine systems fall under the alternative community state model.

Ecosystem Function

Understanding ecosystem function of terrestrial vegetation is important for managing global nutrient cycling. Terrestrial carbon sequestration is a major concern for land managers because of its ability to offset anthropogenic carbon emissions and climate change, which have direct and indirect effects on carbon fluxes from ecological and physiological processes (Cao and Woodward 1998; Breshears and Allen 2002). The majority of research on carbon sequestration and net primary production (NPP) is conducted at a global or large regional scale. As a result, only broad generalizations can be made at the Sub-Regional-scale regarding ecosystem function and its relation to variations in climate and disturbance regimes.

Anthropogenic disturbances to terrestrial ecosystems impact global terrestrial function, including carbon balances and net biological productivity (Law et al. 2003). When carbon sequestration is being estimated at any scale, it is crucial that not only anthropogenic (e.g., land-use change, fire suppression) induced losses and gains be taken into consideration but also large-scale climate driven losses, such as forest fires, drought induced tree mortality, and soil erosion (Bachelet et al. 2001; Breshears and Allen 2002). These large-scale climate driven carbon losses are especially important at our Sub-Regional-scale because of the current drought we are experiencing in the Southwest that is resulting in significant changes to forest stand dynamics (e.g., pinon dieback in pinon-juniper forests due to drought stress and *Ips confusus* infestations, increased wildfire frequency and intensity in ponderosa pine and mixed-conifer forests) in both species composition, age, and size distributions. When anthropogenic and climate-driven changes to vegetation communities are compiled, large losses may occur in both plant and soil carbon pools that could result in a net carbon loss rather than a gain, triggering a positive feedback that could intensify global warming (Houghton et al. 1999).

Forested vegetation can provide a major carbon sink, depending on its size, age, and density (Birdsey et al. 1993). Young and middle aged forests are thought to be better carbon sinks than old forests because of active growth and higher densities in these stands leading to an overall increase in NPP and decreased nutrient availability and increased stomatal limitation as stands age, which decreases overall NPP (Cao and Woodward 1998). A study in ponderosa pine forests in Oregon illustrated that young (56-89 years) and mature (95-105 years) stands had higher NPP and NBP than old stands (109-316) and that the majority of stands were in the mature stage and thus reaching maximum carbon storage and uptake (Law et al. 2003). Numerous stands within the Sub-Region, especially within ponderosa pine forests, have increased in density and are middle aged (mature) as a result of anthropogenic changes (fire suppression and old-growth logging) and a good seed year in 1919, which has resulted in an increase in carbon sequestration for this forest type. Nationally, fire suppression has led to an increased rate of C sequestration in U.S. ecosystems by increasing storage in woody biomass, soils, litter, and coarse woody debris and a decrease in the rate of CO^2 release through burning Sohngren and Haynes 1997). A study in Minnesota oak savanna showed that C stores average ~ 272 Mg/acre (110 Mg/ha) with pre-settlement fire frequencies and ~220 mg/ha in stands with fire suppression (Tilman et al. 2000). If comparable alterations to C storage have occurred in western forested communities, than current C storage values within our Sub-Region are outside their HRV due to anthropogenic disturbances. Large-scale crown fires over the past decade in this forest type within our Sub-Region however have negated carbon sequestration benefits and are now a source of carbon loss. Fire regimes outside their HRV can also result in severe soil erosion, and thus a significant carbon loss, as evident in the Cerro Grande fire in 2000 in the Jemez Mountains, New Mexico and the 2002 Missionary Ridge fire in the San Juan Mountains, Colorado. In terrestrial ecosystems, the amount of carbon stored in soil (plant, animal and microbial residues in all stages of decomposition) is usually greater than the amount in living vegetation (Post and Kwon 2000). A complete forest demographics inventory is necessary in order to estimate carbon sinks caused by forest growth at the Sub-Regional-scale, which is beyond the focus of this assessment (Birdsey et al. 1993).

Finally, climate change may exasperate anthropogenic influences on ecosystem function. Warming events since the mid 1970's in the U.S. and increased precipitation until the past decade decreased net biological productivity (NBP) due to increased plant and soil respiration from warmer temperatures (Bachelet et al. 2001). In contrast, future climate change models indicate that Leaf Area Index (LAI), which is related to NPP, will increase dramatically in the Southwest particularly in desert environments due to increased precipitation and only moderate temperature increases. In addition, fires increase in the West because of increased fuel loads and precipitation couple with several wet-dry (El Nino/La Nina) cycles (Bachelet et al. 2001). Understand historical biogeographical shifts in species composition and abundance in relation to climate change may be extremely useful to predict future changes in both community structure and function.

Possible management recommendations to mitigate anthropogenic and climatically induced carbon losses include preventive thinning and prescribed burning in forest types outside their HRV and improved soil conservation techniques (Breshears and Allen 2002). Thinning in forest types not outside their HRV and the conversion of old-growth forests to younger stands to mitigate C inputs is not recommended.

Research in the Pacific Northwest illustrated that harvesting old-growth forests and converting them to younger stands actually increase C inputs to the atmosphere and these authors hypothesized similar patterns in other systems where the age of harvest is less than the age required to reach the old-growth stage (Harmon et al. 1990). More research is needed regarding individual species' function for even the most common forest species in the western U.S. While large-scale changes to communities have immense impacts on global and regional terrestrial function, understanding finer scale seasonal and annual variation of ecosystem function in relation to climate variation is also important (Law et al. 2001; Monson et al. 2002; Huxman et al. 2003).

Dominant forces affecting vegetative patterns at the Sub-Regional Scale

Natural influences

The Sub-Region is compromised of two dominant geographic features: the Colorado Plateau and the southern Rocky Mountains, which represents a diverse range of climatic environments influenced primarily by physiographic features including elevation and topography. For a more in-depth description of these two features see Module II: Biogeography Significance. Within these larger geographic features are heterogeneous landscapes that are influenced by both living (biotic) and physical (abiotic) factors that determine individual species ranges and thus biotic communities. One factor rarely limits the distribution of a species; rather it is more often an aggregate effect of many interacting factors that sets species distribution limits. Some biotic factors that influence species distribution patterns include species dispersal mechanisms and species interactions (mutualism, competition, parasitism).

Abiotic factors are often the primary drivers that influence species distribution patterns on the landscape. Species are found generally in environments where limiting factors (topography, geology, soil, climate) are conducive to their survival. The two main limiting physical factors within our Sub-Region are temperature and moisture, which are influenced by elevation and topography. Generally, precipitation is the limiting factor at lower elevations and temperature is the limiting factor at higher elevations. In the San Juan Mountains, Colorado the threshold between these two limiting factors is between 7,600-8,200 ft (2316-2500 m) (Spencer and Romme 1996). This threshold would be similar for other mountainous areas within the Sub-Region.

Elevation is the single most important gradient influencing vegetation patterns within the Sub-Region because of its direct influence on climate (temperature, relative humidity, solar radiation, precipitation, and wind) (Peet, 2000). Unique vegetation assemblages along this elevational gradient are the result of the interaction between elevation and topography (aspect and slope). Northern aspects with higher moisture availability allow vegetation communities typically found higher in elevation to establish at lower elevations and southern aspects with lower moisture availability allow vegetation communities to be located at higher elevations than normally situated. Slope position also alters general elevational patterns with ridges having higher solar radiation and lower soil moisture than valley bottoms that have higher soil moisture, lower solar radiation, and cold air drainage. Geological parent substrate and soils also interact with elevation. In the San Juan Mountains, adjacent areas with similar elevation and topography can have different vegetation communities based on soil types due to the amount of moisture availability to plants. For instance, shale derived soils have lower moisture availability than sandstone derived soils die to soil texture and depth (Spencer and Romme 1996). At higher elevations in the San Juan Mountains, differences in parent material (e.g., granite vs. limestone) have strong impacts on vegetation types due to influences on soil parameters, particularly soil pH. Numerous endemic species and unique species assemblages are found on limestone-derived soils in higher elevations of the Sub-Region.

Disturbance

Overlaying and interacting with biotic and abiotic factors that determine individual species ranges, and thus biotic communities, are natural disturbances. These disturbances play a fundamental role in the diverse patch mosaic of vegetation communities across the landscape. Within and between each vegetation type, patches of different structural characteristics are the result natural disturbance processes such as fire, insect outbreaks, wind, and avalanches. It is important to recognize that disturbances interact in complex ways and that when assessing vegetation communities all types of disturbances, both natural and anthropogenic, need to be incorporated. In addition, it is important to recognize synergism among disturbances (that the combined effects are greater than the sum of independently occurring events) and the timing or sequence of disturbances, which can have immense long-term impacts on vegetation communities. Examples include fires occurring in drought years in mixed conifer forests and ungulate browsing following fire in aspen. A few of the major natural disturbances that influence landscape heterogeneity are discussed below. See Module V: Landscape Disturbances for a more in-depth discussion.

Forested communities in the western U.S. are considered fire dependent because of their close relationship with fire. Looking at evolutionary adaptations of individual tree species can provide clues to the types of fires that they evolved with. For example, evidence of ponderosa pine evolution to frequent, low intensity surface fires is seen in selected survival adaptations: thick bark, self pruning lower branches (crown height), highly flammable needles (litter), and long needles to protect buds. There are three main types of fire regimes that occur within our Sub-Region: understory fire regime, mixed fire regime, and stand replacement fire regime (Arno and Allison-Bunnell 2002). The understory fire regime burns every 1-30 years and generally has thick barked, fire-resistant trees growing at medium or wide spacing and open understories. An example forest type in the Sub-Region would be ponderosa pine. The mixed fire regime has fires that alternate between light underburns and stand replacement. They are of intermediate intensity, killing most of the fire-susceptible trees while fire-resistant trees survive. An example forest type in the Sub-Region would be mixed conifer. Finally, infrequent fires at long intervals between 100-400 years characterize the stand replacement fire regime.

Each fire kills the majority of trees, which allows for the development of a new forest. Burning is often not uniform and can occur in large, irregular patches (Arno and Allison-Bunnell 2002). An example forest type in our Sub-Region would be Spruce-fir.

Within these three fire regimes, fire behavior can vary greatly based on fuels, topography, and climate. The type, size, quantity, arrangement, and moisture content of fuels are critical to how fires burn. For example, ponderosa pine forests historically were dominated by fine fuels (needles and dry grasses) along with shrubs within our Sub-Region. Ponderosa pine trees produce on average 1 ton/acre of dry pine needles every fall providing appropriate conditions for understory surface fires. In contrast, higher elevational forests generally lack fine fuels and are dominated by rotten and sound coarse woody debris. These larger fuels dry out more slowly than fine fuels and therefore have a large impact on fire behavior. In prolonged drought conditions, these large fuels become quite dry enabling stand replacing fires to occur. In addition, topography and its interaction with climate have a strong influence on the fire environment. Elevation, slope position, aspect, slope steepness, and natural and artificial barriers all influence fire behavior. Elevation affects the length of the growing season, type of vegetation, and weather patterns. Slope position affects temperature and relative humidity. Aspect influences solar radiation, soil moisture content, and wind patterns. Slope steepness has a direct effect on flame length and rate of spread of a surface fire, and finally barriers can influence the extent of fire spread. When assessing the role of fire on landscape patch dynamics and identifying HRV it is crucial that variability within a given fire regime for a specific forest type is recognized and that site specific fuel, topographic, and climatic differences are used to differentiate fire regime variability.

Insect outbreaks have a large influence on landscape heterogeneity. A current example within our Sub-Region is the large outbreak of pinon bark beetle (*Ips confusus)* killing pinon pine. This outbreak is currently coupled with drought, and therefore is having a larger impact on the landscape patch mosaic of our Sub-Region than if the outbreak were not accompanied by drought. Other insects that have a significant impact on forest stands over large geographic areas within our Sub-Region include mountain pine beetle (*Dendroctonus ponderosae*), Douglas-fir beetle (*Dendroctonus pseudotsugae*), spruce beetle (*Dendroctonus rufipennis*), and western spruce budworm (*Choristoneura occidentalis*). It is important to recognize that all of these insects are native to our Sub-Region and are generally present as low-density, endemic populations, which have little impact on forest structure. However, periodic outbreaks occur where large contiguous areas of mature trees are killed. The mechanisms that create the shift from endemic to outbreak conditions are poorly understood but are generally associated with other disturbance events (Romme et al. 2003b, e).

Windstorm disturbances are generally more pronounced within higher elevation forest types than lower elevation types. The majority of tree species at lower elevations have extensive root systems that often include deep taproots. In addition, windstorms that do occur are generally less intense than windstorms at higher elevations. In contrast, the two most common higher elevational trees species, Engelmann spruce and subalpine fir are shallow rooted and not windfirm (Kulakowski and Veblen 2004). Winds at this elevation are stronger and can cause extensive damage such as the 1997 Routt blowndown in northwestern Colorado (Baker et al. 2002). Interactions of wind intensity with previous site disturbances, shallow or poorly drained soils, steep slopes and dense stands can all increase the vulnerability of stands to windthrow (Kulakowski and Veblen 2004). Studies within our Sub-Region indicate that windthrow most likely does not play as an important disturbance role as in other adjacent areas to our Sub-Region (Romme et al. 2003e; Kulakowski and Veblen 2004). Kulakowski and Veblen (2004) suggest however that the high elevation of Grand Mesa, Colorado makes it likely that blowdowns play a role in the development of higher elevation forests. Finally, snow avalanches also impact higher elevation forest development. These disturbances are localized events that remove significant forest cover from certain topographic locations.

Anthropogenic influences

Landscapes are heterogeneous in both space and time even without anthropogenic influences (Pickett and Cadenasso 1995). Investigating and understanding the historical biogeography of individual species and biotic communities can illustrate how natural disturbances influence biotic communities structure and function and and the shifting mosaic of these communities over time and space (see Module II: Biogeographic Significance for examples at the Sub-Region-scale).

Fire suppression

Since the large conflagrations of 1910 in the western U.S., western landscapes have been impacted by fire suppression. Natural fire regimes create landscape heterogeneity while fire suppression often leads to homogenous landscapes (Keane et al. 2002). Fire suppression is different from other anthropogenic disturbances in that it occurs gradually over time and has the potential to alter future fire disturbances in both severity and aerial extent as illustrated in ponderosa pine stands within in the Sub-Region (Swetnam and Baisan 1996). See Module III: Historical range of variability for a more detailed discussion on how fire suppression, if at all, has altered the structure and function of dominant forested vegetation communities at the Sub-Regional scale.

Development

The two major types of development that are impacting biotic communities in our Sub-Region are resort development and low-density development. Ski resorts represent a type of micro-urban development often surrounded by natural landscapes (Travis et al. 2002). Ski resorts increase public access to higher elevation forest and alpine environments for year-round recreation activities including skiing, mountain biking, and hiking. Resorts are the fastest expanding residential and commercial land-use in the Rocky Mountain region; however when compared to metropolitan areas, resorts have a relatively small footprint on the landscape (Travis et al. 2002). The second major development, low-density residential, in the

Southern Rockies is growing faster in physical expansion across the landscape than actual population growth due to the large increase in "ranchette" or exurban development (SREP 2000). Exurban development results in high fragmentation primarily in low to middle elevation communities and along riparian systems. Exurban development in these areas also affects natural disturbance processes such as fire and flooding because efforts are generally taken to inhibit these processes around developed areas. Fire suppression and mitigation efforts around developed areas that interface with public land, often referred to as the wildland-urban interface, are becoming a top management priority for public land managers. The wildland-urban interface zone is forecasted to increase within the Southern Rockies because 75% of the region's forested communities are within 1.5 miles of private land (SREP 2000). These lands will eventually be developed, and therefore management options for fire on public lands will be restricted, often preventing ecologically beneficial fires from burning.

Roads

The presence of roads has a large impact on landscape patterns and processes by creating sharp edges in otherwise intact (interior) habitats. Some deleterious effects of roads include creating barriers to species mobility, acting as corridors for non-native and edge adapted species, and increasing human access to interior habitats (Baker and Knight 2000). Road density is more important than the actual presence of a road. Approximately 94% of all land located within the Southern Rockies is within 2 miles of a road (SREP 2000). Higher road densities can significantly affect the presence of large mammals such as elk, mountain lions, and black bear and also alters natural disturbance processes and biotic interactions with communities. Vegetation communities within the Sub-Region that are most impacted by roads are areas at low and middle elevation, which corresponds to communities most impacted by human development (SREP 2000). See Module V: Roads for a more detailed discussion within the Sub-Region.

Timber Harvest

Large areas of interior forest have been lost in Southern Rockies due to timber harvesting and subsequent road construction (Reed et al. 1996). While forestry practices do not result in land use change such as development and roads, timber harvesting does alter stand structure and function with the degree of impact based upon the size, intensity, and type of harvest, pre-existing harvest conditions (past management activities), biotic/abiotic factors (soil type, slope, aspect, and vegetation type), and how harvesting practices are distributed across the landscape. The majority of low and middle elevation forested landscapes within the Sub-Region are between 70-130 years old and are considered even-aged, mature stands due to logging and fires set intentionally by humans in the late 1800's to early 1900's (Smith 2000). This legacy of prior anthropogenic disturbances on the landscape, intertwined with natural landscape heterogeneity, has a large impact on the current mosaic of forested patches across the landscape and the diversity of structural classes for forested ecosystems. The majority of the old-growth logging occurred within the Sub-Region prior to the 1950's, however some old-growth logging still occurs today at low levels. For example, in the Intermountain Region (Four corner states plus Wyoming, Idaho, and Montana), the total volume of softwood timber from large trees decreased by 31.4% between 1952 and 1992 from 43,648 mbf to 30,067 mbf (Stohlgren et al. 2002). In addition, total overall harvest has also decreased as federal agency management policies have switched from resource-based management to ecosystem-based management. The types of harvests today also vary with many timber harvests focused on reducing dense stands compromised of small diameter trees. This is change in timber harvest is occurring within the Sub-Region in ponderosa pine and warm-dry mixed conifer forests to restore forest structure to within its HRV. See Module V: Timber Management for a more detailed discussion within the Sub-Region.

Grazing

Grazing impacts on biotic communities have changed over the past 150 years in the Southwest fro light grazing pressure in the 1800's to intense pressure in the early-mid 1900's and recently to current levels that in some instances attempt to balance the number of grazers with biotic community capacities (Dahms and Geils 1997). Nearly 70% of Forest Service land in the Southern Rockes has active grazing allotments and 93% of BLM land in Colorado (SREP 2000). Domestic livestock have significantly different impacts on natural communities than native free roaming herbivores. Domestic livestock graze in communities historically un-grazed by native herbivores (desert environments) and tend to congregate in certain areas, especially riparian systems (Belsky et al. 1999). Potential ecological impacts of livestock grazing include decreasing overall biomass of grasses and forbs, changing vegetation composition (shift towards non-palatable species), increasing the spread of non-native weeds, increasing soil erosion, and altering nutrient cycling and natural fire regimes (Fleischer 1994). Intense grazing by sheep in ponderosa pine forests that began in the late 1800's resulted in reduced grass fuel loads, which decreased the competition of grasses with pine seedlings for soil moisture, reduced low-intensity grass fires, and increased the exposure of mineral seedbeds due to excessive livestock trampling. The combination of intense grazing, good ponderosa pine seed years, and favorable climatic conditions permitted the germination and successful establishment of large numbers of ponderosa pine trees in the Southwest during the early 1900’s. See Module V: Livestock for a more detailed discussion within the Sub-Region.

Mining/Oil & Gas

Mining has played a large role historically and continues today within the Sub-Region. In Colorado alone there are over 7,000 abandoned mines and approximately 1,615 miles (2,600 km) of streams affected by mine drainage (Rueth et al. 2002). The major ecological impacts of ore mining include leaching of heavy and toxic metals into natural waterways and groundwater which impacts water quality and aquatic biotic communities and air pollution from smelters. While drilling for mineral extraction still occurs within the Sub-Region, drilling for energy minerals (methane) has increased dramatically in southwestern Colorado and the San Juan Basin, New Mexico. In the San Juan Basin alone, 18,000 oil and gas wells already exist and 10,000 new wells are currently in review for approval. Coalbed methane wells extract methane gas from coal 2,500 to 5,500 ft (762-1676 m) below the surface. Pumpjacks are used to release the methane gas by removing groundwater and therefore decreasing pressure, which allows the methane gas to release into the wells. Some ecological impacts associated with coalbed methane include gas seeps into the ground and rivers, ozone air pollution, landscape fragmentation due to increased roads, soil and vegetation disturbance and erosion due to well pads, underground coal fires, and decreased aquifer levels and drying springs. See Module V: Mineral Extraction for a more detailed discussion within the Sub-Region.

Recreation

Outdoor recreation is a growing industry due to the high percentage of public land within the Sub-Region. Recreational activities include fishing, hunting, camping, hiking, mountain biking, skiing, off road vehicle use, snowmobiling, rafting, and backpacking. The number of individuals recreating on public land has increased dramatically over the past few years. For example, in Colorado, the number of registered ATV users tripled from 1991-1998, up to 36,855 individuals (SREP 2000). While many recreational activities seem harmless to vegetation communities such as hiking and backpacking, these activities can have localized impacts on landscapes such as increasing erosion, increasing spread of non-native plants, disrupting wildlife, creating water and soil pollution and damaging vegetation similar to other more intensive recreation activities such as ATV and snowmobile use. Each vegetation community will respond differently to similar recreational activities depending on the community's natural ability to respond to these disturbances. For example, walking on a trail in a desert shrub community with biological cryptogramic crusts is more destructive to vegetation and soil communities than walking across a wet subalpine meadow. See Module V: Recreation for a more detailed discussion within the Sub-Region.

Literature Cited

Allen, C. D. 1989. Changes in the landscape of the Jemez Mountains, New Mexico. PhD thesis. University of California, Berkeley, Berkeley, CA.

Allen, C. D., J. L. Betancourt, and T. W. Swetnam. 1998. Landscape changes in the southwestern United States: techniques, long-term data sets, and trends. Pp. 71-84 In: Sisk, T. (editor) Perspectives on the land use history of North America: a context for understanding our changing environment. U.S. Geological Survey, Biological Science Report USGS/BRD/BSR-1998-0003.

Allen, R. B., R. K. Peet, and W. L. Baker 1991. Gradient analysis of latitudinal variation in southern Rocky Mountain forests. Journal of Biogeography 18: 123-129.

Arno, S. F. and S. Allison-Bunnell. 2002. Flames in our forest: disaster or renewal? Island Press, Washington, D.C.

Bachelet, D., R. P. Neilson, J. M. Lenihan, and R. J. Drapek. 2001. Climate change effects on vegetation distribution and carbon budget in the United States. Ecosystems 4: 164-185.

Baker, W.L., and D. J. Shinneman. 2004. Fire and restoration of pinon-juniper woodlands in the western United States: a review. Forest Ecology and Management 189: 1-21.

Baker, W. L., P. H. Flaherty, J. D. Lindemann, T. T. Veblen, K. S. Eisenhart, and D. Kulakowski. 2002. Effect of vegetation on the impact of a severe blowdown in the Southern Rocky Mountains, USA. Forest Ecology and Management

Baker, W. L, and R. L. Knight. 2000. Rods and forest fragmentation in the Southern Rocky Mountains. Chapter 5 Pp. 97-122 In: Knight, R. L., F. W. Smith, S. W. Buskirk, W. H. Romme, and W. L. Baker (editors), Forest fragmentation in the southern Rocky Mountains. University Press of Colorado. Boulder, Colorado.

Baker, W. L., and T. T. Veblen. 1990. Spruce beetles and fires in the nineteenth-century subalpine forest of western Colorado, U.S.A. Arctic and Alpine Research 22: 65-80.

Bartos, D. L., and R. B. Campbell, Jr. 1998. Decline of quaking aspen in the Interior West-Examples form Utah. Rangelands 20: 17-24.

Bartos, D. L. J. K. Brown, and G. D. Booth 1994. 12 years biomass response in aspen communities following fire. Journal of Range Management 47: 79-83.

Berlow, E. L. 1997. From canalization to contingency: historical effects in a successional rocky intertidal community. Ecological Monographs 67: 435-460

Birdsey, R. A., A. J. Plantinga, and L. S. Heath 1993. Past and prospective carbon storage in United States forests. Forest Ecology and Management 58: 33-40.

Belsky, A. J., A. Matzke, and S. Uselman. 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States Journal of Soils and Water Conservation 54: 419-431.

Billings, W. D. 2000. Alpine Vegetation. Chapter 14. Pp. 537-572 Barbour, M. G., and W. D. Billings (editors) In: Northern American terrestrial vegetation. Second Edition. Cambridge University Press, Cambridge, England.

Bowman, W. D., D. M. Cairns, J. S. Baron, and T. R. Seastedt 2002. Islands in the Sky: Alpine and Treeline Ecosystems of the Rockies. Pp. 183-202 In: Baron, J. S. (editor) Rocky Mountain Futures: an ecological perspective. Island Press, Washington, D.C.

Bowman, W. D., T. A. Theodose, J. C. Schardt, and R. T. Conant. 1993. Constraints of nutrient availability on primary production in two alpine communities. Ecology 74: 2085-2098.

Bradley, A. F., N. V. Noste, and W. C. Fischer. 1992. Fire ecology of forests and woodlands in Utah. General Technical Report INT-287. USDA Forest Service, Intermountain Research Station, Ogden, Utah.

Breshears, D. D. and C. A. Allen 2002. The importance of rapid, disturbance-induced losses in carbon management and sequestration. Global Ecology and Biogeography 11: 1-5.

Brown, P. M., and W. D. Shepperd. 2003. Preliminary fire history in ponderosa pine forest of the Uncompahgre Plateau. Final Report to GMUG National Forest, Colorado. Unpublished report.

Brown, P. M., M. R. Kaufmann, and W. D. Shepperd. 1999. Long-term landscape patterns of past fire events in a montane ponderosa pine forest of central Colorado. Landscape Ecology 14: 513-532.

Buell, M. F. and H. F. Buell 1959. Aspen invasion of prairie. Bulletin of Torrey Botanical Club 86: 264-269.

Cao, M. and F. I. Woodward. 1998. Dynamic responses of terrestrial ecosystem carbon cycling to global climate change. Nature 393: 249-252.

Cawker, K. B. 1980. Evidence of climatic control from population age structure of Artemisia tridentata Nutt. In southern British Columbia. Journal of Biogeography 7: 237-248.

Christensen, E. M. 1955. Ecological notes on the mountain brush in Utah. Utah Academy Proceedings 32: 107-111.

Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe. 1979. Classification of wetlands and deep water habitats of the United States. Technical Reort FWS/OBS/79/31. U.S. Fish and Wildlife Service, Official Bulletin Service.

Covington W. W. and S. S. Sackett 1984. The effect of a prescribed burn in southwestern ponderosa pine on organic matter and nutrients in woody debris and forest floor. Forest Science 30: 183-192.

Covington W. W. and M. M. Moore 1994. Postsettlement changes in natural fire regimes: ecological restoration of old-growth ponderosa pine forests. Journal of Sustainable Forestry 2: 153-181.

Covington W. W., P. Z., Fulé, M. M. Moore, S. C. Hart, T. E., Kolb, J. N. Mast, S. S. Sackett, and M. R. Wagner. 1997. Restoring ecosystem health in ponderosa pine forests of the southwest. Journal of Forestry 95: 23-29.

Crawford, J. A., C.-H.A., Wahren, S. Kyle, and W. H. Moir. 2001. Responses of exotic plant species to fires in Pinus ponderosa forests in northern Arizona. Journal of Vegetation Science 12: 261-268.

Daddy, F., M. J. Trlica, and C. D. Bonham. 1988 Vegetation and soil water differences among big sagebrush communities with different grazing histories. Southwestern Naturalist 33: 413-424.

Dahms, C. W., and B. W. Geils, tech. eds. 1997. An assessment of forest ecosystem health in the Southwest. General Technical Report RM-GTR-295. Fort Collins, CO: U.S.D.A., Forest Service, Rocky Mountain Forest and Range Experiment Station.

Debinski, D. M., M. E. Jakubauskas, and K. Kindscher 2000. Montane meadows as indicators of environmental change. Environmental Monitoring and Assessmentk 64: 213-225.

de Soyza, A. G., J. W. Van Zee, W. G. Whitford, A. Neale, N. Tallent-Hallsel, J. E. Herrick and K. M. Havstad. 2000. Indicators of Great Basin rangeland health. Journal of Arid Environments 45: 289-304.

Dieterich, J.H. 1983. Fire history of southwestern mixed conifer: a case study. Forest Ecology and Management 6: 13-31.

Drake, J. A. 1991. Community-assembly mechanics and the structure of an experimental species ensemble. The American Naturalist 137: 1-26.

Dudgeon, S. and P. S. Petraitis 2001. Scale-dependent recruitment and divergence of intertidal communities. Ecology 82: 991-1006.

Dunwiddle, P. W. 1977. Recent tree invasion of subalpine meadows in the Wind River Mountains. Wyoming. AAR 9: 393-399.

Eisenhart, K. S. 2004. Historical range of variability of pinon-juniper woodlands on the Uncompahgre Plateau, Western Colorado. Final report to GMUG National Forest. Unpublished document.

Elliott, G. P., and W. L. Baker 2004. Quaking aspen (Populus tremuloides Michx.) at treeline: a century of change in the San Juan Mountains, Colorado, USA. Journal of Biogeography 31: 733-745.

Feeney, S.R., T.E. Kolb, W.W. Covington, and M.R. Wagner. 1998. Influence of thinning and burning restoration treatments on presettlement ponderosa pines at the Gus Pearson Natural Area. Canadian Journal of Forest Research 28:1295-1306.

Fleischner, T. L. 1994. Ecoogical costs of livestock grazing in western north America. Conservation Biology 8: 629-644.

Floyd, L. M., D. D. Hanna, and W. H. Romme, in press. Historical and recent fire regimes in Pinon-Juniper woodlands on Mesa Verde, Colorado, USA. Forest Ecology and Management.

Floyd, L. M., W. H. Romme, and D. D. Hanna. 2003. Fire History. Pp. 261-278 In: Floyd, L. M. (editor). Ancient Pinon-Juniper woodlands: a natural history of Mesa Verde country. University Press of Colorado, Boulder, Colorado

Floyd-Hanna, L., A. Spencer, A.W., and W. H. Romme 1996. Biotic communities of the semiarid foothills and valleys. Pp. 143-158 In: Blair, R (editor). The Western San Juan Mountains: their geology, ecology and human history. University of Press Colorado, Boulder, Colorado.

Friedel, M. H. 1991. Range condition assessment and the concept of thresholds: A Viewpoint. Journal of Range Management 44: 422-433

Fulé P.Z., W. W. Covington, and M. M. Moore 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. Ecological Applications 7: 895-908.

Gray, S. T., J. L. Beatancourt, C. L. Fastie, and S. T. Jackson 2003. Patterns and sources of multidecadal oscillations in drought-sensitive tree-ring records from the central and southern Rocky Mountains. Geophysical Research Letters 30: 1316-1320.

Harmon, M. E., W. K. Ferrell, and J. F. Franklin 1990. Effects on carbon storage conversion of old-growth forest to young forests. Science 247: 699-702.

Harte, J. and R. Shaw. 1995. Shifting dominance within a montane vegetation community-results of a climate-woarming experiment. Science 267: 876-880.

Holling, C. S., and G. K. Meffe 1996. Command and control and the pathology of natural resource management. Conservation Biology 10: 328-337.

Houghton, R. A., Hackler, J. L., and K. T. Lawrence. 1999. The US carbon budget: contributions form land-use change. Science 285: 574-578.

Hughes, T. P. 1989. Community structure and diversity of coral reefs: the role of history. Ecology 70: 275-279.

Huxman, T. E., A. A. Turnipseed, J. P. Sparks, P. C. Harley, and R. K. Monson. 2003. Temperature as a control over ecosystem CO2 fluxes in a high-elevation, subalpine forest. Ecosystems Ecology

Jamieson, D. W., W. H. Romme, and P. Somers 1996. Biotic communities of the cool mountains. Pp. 159-174 In: Blair, R (editor). The Western San Juan Mountains: their geology, ecology and human history. University of Press Colorado, Boulder, Colorado.

Johnsen, T. N. 1962. One-seed juniper invasion of northern Arizona grasslands. Ecological Monographs 32: 187-206.

Johnson, A. F. 1989. The thin-green line: Riparian corridors and endangered species in Arizona and New Mexico. Pp. 35-46 In: G. Mackintosh (editor) In defense of wildlife, preserving communities and corridors. Washington D.C.: Defenders of Wildlife.

Johnston, B. C. 1997. Ecological types of the Upper Gunnison Basin. Review draft. USDA, Forest Service, Gunnison, CO.

Kaufmann, M. R., C. M. Regan, and P. M. Brown. 2000. Heterogeneity in ponderosa pine/Douglas-fir forests: Age and size structure in unlogged and logged landscapes of central Colorado. Canadian Journal of Forest Research 30: 698-711.

Kay, C. E. 1997. Is aspen doomed? Journal of Forestry 95: 4-11.

Keane, R. E., K. C. Ryan, T. T. Veblen, C. D. Allen, J. A. Logan, and B. Hawkes 2002. The cascading effects of fire exclusion in Rocky Mountain Ecoystems. Pp. 133-152 In: Baron, J. S. (editor) Rocky Mountain Futures: an ecological perspective. Island Press, Washington, D.C.

Knapp, P. A. 1996. Cheatgrass (Bromus tectorum L) dominance in the Great Basin Desert. Global Environmental Chagne 6: 37-52.

Knowlton, N. 1992. Thresholds and multiple stable states in coral reef community dynamics. American Zoologist 32: 674-682.

Kulakowski, D., T. T. Veblen, and P. Bebi. 2003. Effects of fire and spruce beetle outbreak legacies on the disturbance regime of a subalpine forest in Colorado. Journal of Biogeography 30: 1445-1456.

Kulakowski, D., and T. T. Veblen 2004. Historical Range of Variability Assessment for forest vegetation of the Grand Mesa National Forest, Colorado. USFS, Rocky Mountain Region, Lakewood, CO. Unpublished Final Draft Report.

Kulakowski, D., T. T. Veblen, and S. Drinkwater, in press. The persistence of quaking aspen (Populus tremuloides) in the Grand Mesa area, Colorado. Ecological Applications.

Landres, P., P. Morgan, and F. Swanson. 1999. Overview of the use of natural variability in managing ecological systems. Ecological Applications 9: 1279-1288.

Law, B. E., O. J. Sun, J. Campbell, S. Van Tuyl, and P. E. Thornton. 2003. Changes in carbon storage and fluxes in a chronosequence of ponderosa pine. Global Change Biology 9: 510-524.

Law, B. E., P. E. Thornton, J. Irvine, P. M., Anthoni, and S. Van Tuyl. 2001. Carbon storage and fluxes in ponderosa pine forests at different developmental stages. Global Change Biology 7: 755-777.

Law, R. and R. D. Morton 1993. Alternative permanent states in ecological communities. Ecology 74: 1347-1361.

Lewontin, R. C. 1969. The meaning of stability. Brookhaven Symposium of Biology 22: 13-24.

Loik, M. E., and S. P. Redar 2003. Microclimate, freezing, and cold acclimation along an elevation gradient for seedlings of the Great Basin Desert shrub, Artemisia tridentata. Journal of Arid Environments 54: 769-782.

Manier, D. J., and R. D. Laven 2002. Changes in landscape patterns associated with the persistence of aspen (Populus tremuloides Michx.) on the western slope of the Rocky Mountains, Colorado. Forest Ecology and Management 167: 263-284.

Mast, J.N., and J.J. Wolf. 2004. Ecotonal changes and altered tree spatial patterns in lower mixed-conifer forests, Grand Canyon National Park, Arizona, U.S.A. Landscape Ecology 19: 167-180.

Mast, J. N., T. T. Veblen, and M. E. Hodgson 1997. Tree invasion within a pine/grassland ecotone: an approach with historic aerial photography and GIS modeling. Forest Ecology and Management 93: 181-194.

May R. M. 1977. Thresholds and breakpoints in ecosytems with multiplicity of stable states. Nature 269: 471-477.

Mehl, M. S. 1992. Old growth descriptions for the major forest cover types in the Rocky Mountain region. Pp. 106-120 In: Kaufmann, M. R., Moir, W. H., and Bassett, R. L, (technical coordinators). Old-growth forests in the southwest and Rocky Mountain regions. Proceedings of a workshop. USDA Forest Service, General Technical Report RM-213. Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.

Monson, R. K., A. A. Turnipseed, J. P. Sparks, P. C. Harley, L. E. Scott-Denton, K. Sparks, and T. E. Huxman. 2002. Carbon sequestration in a high-elevation, subalpine forest. Global Change Biology 8: 459-478.

Mueggler, W. F. 1985. Vegetation associations. DeByle, N.V., Winokur, R. P. (editors) Pp. 45-55 In: Aspen: ecology and management in the western United States. General Technical Report, RM-119, USDA Forest Service.

Naveh, Z. 1998. Ecological and cultural landscape restoration towards a post-industrial symbiosis between human society and nature. Restoration Ecology 6: 135-143.

Norton, J. B., T. A. Monaco, J. M. Norton, D. A. Johnson, and T. A. Jones. 2004. Soil morphology and organic matter dynamics under cheatgrass and sagebrush-steppe plant communities. Journal of Arid Environments 57: 445-466.

Parker, A. J. and K. C. Parker 1983. Comparative successional roles of trembling aspen and lodgepole in the southern Rocky Mountains. Great Basin Naturalist 43: 447-455.

Partel, M. and M. Zobel 1998. Small-scale plant species richness in calcareous grasslands determined by the species richness in calcareous grasslands determined by the species pool, community age and shoot density. Ecography 22: 153-159.

Peet, R. K., 2000. Forests of the Rocky Mountains. Chapter 3. Pp. 75-122 Barbour, M. G., and W. D. Billings (editors) In: Northern American terrestrial vegetation. Second Edition. Cambridge University Press, Cambridge, England.

Pickett, S. T. A., and M. L. Cadenasso. 1995. Landscape Ecology: spatial heterogeneity in ecological systems. Science: 269: 331-334.

Post, W. M., and K. C. Kwon 2000. Soil carbon sequestration and land-use change: processes and potential. Global Change Biology 6: 317-327.

Redders, J. S. 2003. Landscape condition analysis for the South Central Highlands section, southwestern Colorado and northwestern New Mexico. Chapter 6-G: Alpine ecosystems. Final Report to the San Juan National Forest, Durango, Colorado Uunpublished report.

Redders, J. S. 2004. Vegetation of the San Juan National Forest Working White Paper. San Juan National Forest. Unpublished document.

Redders, J. S. 2003a. A classification of riparian area and wetland vegetation of the San Juan National Forest. San Juan National Forest, unpublished document.

Redders, J. S. 2003b. Landscape condition analysis for the South Central Highlands section, southwestern Colorado and northwestern New Mexico. Chapter 6-E: Mountain Grasslands. Final Report to the San Juan National Forest, Durango, Colorado Uunpublished report.

Reed, R. A., J. Johnson-Barnard, and W. L. Baker. 1996. Fragmentation of a forested Rocky Mountain landscape, 1950-1993. Biological Conservation 75: 267-277.

Rogers, P. 2002. Using Forest Health Monitoring to assess aspen forest cover change in the southern Rockies ecoregion. Forest Ecology and Management 155: 223-236.

Romme, W. H. 2003. Landscape Condition Analysis for the South Central Highlands Section, Southwestern Colorado and Northwestern New Mexico. Chapter 6: Other Vegetation Types. Final Report to the San Juan National Forest, Durango, Colorado Unpublished report.

Romme, W. H., M. L. Floyd-Hanna, and D. D. Hanna 2003a. Ancient Pinon-Juniper forests of Mesa Verde and the West: a cautionary note for forest restoration programs. In: Proceedings of the Conference on Fire, Fuel Treatments, and Ecological Restoration: Proper Place, Appropriate Time, Colorado State University, April 2002.

Romme, W. H., M. L. Floyd, D. Hanna, H. D. Grissino-Mayer, and D. Green 2003b. Landscape Condition Analysis for the South Central Highlands Section, Southwestern Colorado and Northwestern New Mexico. Chapter 2: Ponderosa Pine Forests. Final Report to the San Juan National Forest, Durango, Colorado. Unpublished report.

Romme, W.H., M.L. Floyd, D. Hanna, and J.S. Redders 2003c. Landscape condition analysis for the South Central Highlands section, southwestern Colorado and northwestern New Mexico. Chapter 3: Mixed Conifer Forests. Final Report to the San Juan National Forest, Durango, Colorado Uunpublished report.

Romme, W. H., M. L. Floyd, D. Hanna and E. J. Bartlett 2003d. Landscape condition analysis for the South Central Highlands section, southwestern Colorado and northwestern New Mexico. Chapter 4: Aspen Forests. Final Report to the San Juan National Forest, Durango, Colorado. Unpublished report.

Romme, W. H., M. L. Floyd, D. Hanna, J. S. Redders, M. McGarigal and M. Crist 2003e. Landscape condition analysis for the South Central Highlands section, southwestern Colorado and northwestern New Mexico. Chapter 5: Subalpine Forests. Final Report to the San Juan National Forest, Durango, Colorado. Unpublished report.

Romme, W. H., K. McGarigal, and D. Goodwin. 2003f. Vegetation dynamics of the Uncompahgre Plateau landscape, southwestern Colorado. Draft Report to Uncompahgre National Forest, Colorado. Unpublished report.

Romme, W. H., M. L. Floyd, D. Hanna, H. D. Grissino-Mayer, and D. Green 2003g. Landscape Condition Analysis for the South Central Highlands Section, Southwestern Colorado and Northwestern New Mexico. Chapter 6B: Mountain Shrublands. Final Report to the San Juan National Forest, Durango, Colorado. Unpublished report.

Romme, W. H., L. Floyd-Hanna, D. D. Hanna, and E. J. Bartlett 2001. Aspen's ecological role in the West. Sustaining aspen in western landscapes: symposium proceedings RMRS-P-18, USDA Forest Service, Fort Collins, CO.

Rueth, H. M., J. S. Baron, L. A. Joyce. 2002. Natural resource extraction: past, present, and future. Pp. 85-112 In: Baron, J. S. (editor) Rocky Mountain Futures: an ecological perspective. Island Press, Washington, D.C.

Samuels, C. L. and J. A. Drake 1997. Divergent perspectives on community convergence. Trends in Ecology and Evolution 12: 427-432.

Sauer, L.J. 1998. The once and future forest. Island Press, Washington, D.C.

Smith, F. W. 2000. Forestry practices and forest fragmentation in the Southern Rocky Mountains. Chapter 6 Pp. 123-134 In: Knight, R. L., F. W. Smith, S. W. Buskirk, W. H. Romme, and W. L. Baker (editors), Forest fragmentation in the southern Rocky Mountains. University Press of Colorado. Boulder, Colorado.

Smith, F. W., and A. Smith 2004. Aspen persistence on the Uncompahgre Plateau, Colorado. Final Report. GMUG National Forest. Unpublished report.

Sohngren, B. L., and R. W. Haynes 1997. The potential for increasing carbon storage in United States unreserved timberlands by reducing forest fire frequency: an economic and ecological analysis. Climate Change 35: 179-197.

Spencer, A.W., and W. H. Romme 1996. Ecological Patterns. Pp. 129-142 In: Blair, R (editor). The Western San Juan Mountains: their geology, ecology and human history. University of Press Colorado, Boulder, Colorado.

Stohlgren, T. J., T. T. Veblen, K. C. Kendell, W. L. Baker, C. D. Allen, J. A. Logan, and K. C. Ryan. 2002. The heart of the Rockies: montane and subalpine ecosystems. Pp. 203-218 In: Baron, J. S. (editor) Rocky Mountain Futures: an ecological perspective. Island Press, Washington, D.C.

Sudworth, G. B. 1900. White River Plateau and Battlement Mesa, Forest Reserves. Extract from the Twentieth Annual Report of the Survey, Part V, Forest Reserves. Department of the Interior, USGS. Washington, D.C.

Sutherland, J. P. 1990. Perturbations, resistance, and alternative views of the existence of multiple stable points in nature. American Naturalist 136: 270-275.

Swetnam, T. W. and C. H. Baisan. 1996. Historical fire regime patterns in southwestern United States since AD 1700. Pp. 11-32 In: C. D. Allen (editor). Fire effects in southwestern forests, Proceedings of the Second La Mesa Fire Symposium, March 29-31, 1994, Los Alamos, New Mexico. U.S. Department of Agriculture, U. S. Forest Service General Technical Report RM-GTR-286.

Swetnam, T. W. and J. L. Betancourt. 1998. Mesoscale disturbance and ecological response to decadal climatic variability in the American Southwest. Journal of Climate 11: 3128-3147.

SREP (Southern Rockies Ecosystem Project). 2000. The state of the southern Rockies ecoregion. Denver, Colorado: Southern Rockies Ecosystem Project. On-line at <http://csf.colorado.edu/srep>

Tiedemann, A. R., W. P. Clary, and R. J. Barbour. 1987. Underground system of Gambel oak (Quercus gambelii) in central Utah. American Journal of Botany 74: 1065-1071.

Tilman, D., P. Reich, H. Philliops, M. Menton, A. Patel, E. Vos, and D. Peterson, and J. Knops 2000. Fire suppression and ecosystem carbon storage. Ecology 81: 2680-2685.

Tisdale, E. W. and M. Hironaka. 1981. The sagebrush-grass region: a review of the ecological literature. Forest, Wildlife and Range Experiment Station Bulletin 33, University of Idaho, Moscow.

Travis, W. R., D. M. Theobald, and D. B. Fagre. 2002. Transforming the Rockies: Human forces, settlement patterns, and ecosystem effects. Pp. 1-24 In: Baron, J. S. (editor) Rocky Mountain Futures: an ecological perspective. Island Press, Washington, D.C.

Touchan, R., C. D. Allen, and T. W. Swetnam. 1996. Fire history and climatic patterns in ponderosa pine and mixed conifer forests of the Jemez Mountains, Northern New Mexico. Pp.33-46 In: C. D. Allen (editor). Fire effects in southwestern forests, Proceedings of the Second La Mesa Fire Symposium, March 29-31, 1994, Los Alamos, New Mexico. U.S. Department of Agriculture, U. S. Forest Service General Technical Report RM-GTR-286.

Turner, G. T. 1975. Mountain grassland ecosystem. USDA Forest Service Research Paper RM-161. Rocky Mountain Forest and Range Experimental Station, Fort Collins, Colorado.

Turner, M. G., W. H. Romme, R. A. Reed, and G. A. Tuskan 2003. Post-fire aspen seedling recruitment across the Yellowstone (USA) Landscape. Landscape Ecology 18: 127-140.

USDA 2002. White River National Forest, Colorado. Land and Resource Management Plan-2002 Revision. Final Environmental Impact Statement Volume 3. Appendix D. Historical Range of Natural Variability. Unpublished document.

Veblen, T. T. 1986. Age and size structure of subalpine forests in the Colorado Front Range. Bulletin of the Torrey Botanical Club 113: 225-240.

Veblen, T. T., K. S. Hadley, and M. S. Reid. 1991. Disturbance and stand development of a Colorado subalpine forest. Journal of Biogeography 18:707-716.

Veblen, T. T., K. S. Hadley, E. M. Nel, T. Kitzberger, M. Reid, and R. Villalba. 1994. Disturbance regime and disturbance interactions in a Rocky Mountain subalpine forest. Journal of Ecology 82:125-135.

Veblen, T. T. 2000. Disturbance patterns in southern Rocky Mountain forests. Chapter 3 Pp. 31-54 In: Knight, R. L., F. W. Smith, S. W. Buskirk, W. H. Romme, and W. L. Baker (editors), Forest fragmentation in the southern Rocky Mountains. University Press of Colorado. Boulder, Colorado.

Walker, D. A., J. C. Halfpenny, M. D. Walker, and C. A. Wessman. 1993. Long-term studies of snow-vegetatin interactions. Bioscience 43: 287-301.

Welch, B. L., and C. Criddle. 2003. Countering misinformation concerning big sagebrush. USDA Forest Service Research Paper RMRS-RP-40.

West, N. E. and J. A. Young 2000. Intermountain Valleys and Lower Mountain Slopes. Chapter 7. Pp. 255-284 Barbour, M. G., and W. D. Billings (editors) In: Northern American terrestrial vegetation. Second Edition. Cambridge University Press, Cambridge, England.

West, N. E. 1983. Intermountain salt-desert shrubland. Pp. 375-397. N. E. West (editor) In: Temperate deserts and semi-deserts, vol. 5, Ecosystems of the world. Elsevier: Amsterdam, The Nederlands.

White, A. S. 1985. Presettlement regeneration patterns in a southwestern ponderosa pine stand. Ecology 66: 589-594.

White, M.A., and J.L. Vankat 1993. Middle and high elevation coniferous forest communities of the North Rim region of Grand Canyon National Park, Arizona, USA. Vegetatio 109: 161-174.

White, P. S., J. L. Walker 1997. Approximating nature's variation: selecting and using reference information in restoration ecology. Restoration Ecology 5: 338-349.

Wissel, C. 1984. A universal law of the characteristic return time near thresholds. Oecologia 65: 101-107.

Wright, H. A., L. F. Neuenschwander, and C. M. Britton. 1979. The role and use of fire in sagebrush-grass and pinyon-juniper plant communities a state-of the art review. General Technical Report INT-58. Ogden, UT: USDA, Forest Service, Intermountain Forest and Range Experiment Station.

Wright R. J. 1996. Fire effects on fine roots, mycorrhizae, and nutrient dynamics in a southwestern ponderosa pine forest. M. S. Thesis, Northern Arizona University, Flagstaff, Arizona.

Wu, R. 1999. Fire history and forest structure in the mixed conifer forests of southwest Colorado. M.S. Thesis, Colorado State University, Fort Collins, Colorado.

Chapter 5. Chapter Forest-scale Vegetation description (Modules 3A and 3B)

LTA, PNV, Community Descriptions

Landtype Associations (LTAs) are broad landscape-scale map units defined (Table 5-1) by geomorphic process, geologic formations, landforms, potential natural vegetation (PNV), and soils (Forman and Godron 1986, Bailey and Avers 1993, Cleland et al. 1997, TEUI Guide). They typically range in area from one thousand to hundreds of thousands of acres.

Riparian areas, wetlands, rock outcrop, talus slopes, small-scale landforms, geology inclusions, and minor PNV types occur within the generalized LTAs, which were developed from the bottom up using landtype ecological units.

Table 5-1 Land Type Association Legend

LTA	LANDFORM	PNV (series)	GEOLOGY
20SXU	hills, mesas	pinyon pine, big sagebrush	sediments
25SXC	canyons	pinyon pine, ponderosa pine, Douglas fir, narrowleaf cottonwood	sediments
30SXM	mountains	ponderosa pine	sediments
30SXP	plateaus, mesas	ponderosa pine	sediments
35SXM	mountains	ponderosa pine, white fir, Douglas fir	sediments
35SXU	mesas, valleys, hills, mountains	ponderosa pine, white fir, Douglas fir	sediments
50SXM	mountains	white fir, Douglas fir	sediments
70IGM	mountains	Engelmann spruce, subalpine fir	igneous
70SXM	mountains	Engelmann spruce, subalpine fir	sediments
70VXM	mountains	Engelmann spruce, subalpine fir	volcanics
80XXM	mountains	kobresia, alpine avens, snow willow, planeleaf willow, tufted hairgrass, marsh marigold	mixed

LTA DESCRIPTIONS

LTA Name: 20SXU - Pinyon-Juniper Woodlands on Hills and Mesas

This land type association occurs at the lowest elevations of the SJNF, on the warmest and driest sites. Elevations range from about 6000 to 7500 feet. Slopes range from about 0 to 40%. Mean annual precipitation ranges from about 13-18 inches. The Ryman Creek area near Dissappointment Valley occurs in this LTA. There are about XX acres of this LTA on the SJNF.

- Landform: hills, mesas
- Climate Zone: semi-arid
- PNV: pinyon pine and big sagebrush series
- Existing Vegetation: pinyon-juniper woodland, mountain shrubland, sagebrush shrubland
- Geologic Formation: Dakota Sandstone, Burro Canyon, Mancos Shale, Animas
- Geomorphic Process: fluvial
- Lithology: sedimentary rock; sandstone, shale, conglomerate
- Soils: Haplustalfs, moderately-deep to deep, well drained
- Use & Management: livestock, winter range for wildlife, oil and gas
- Interpretations: moderate to high erosion hazard on shale soils

LTA Name: 25SXC – Mixed Vegetation in Canyons

This landtype association occurs at low to middle elevations mostly on the west side of the SJNF, at elevations ranging from about 7000 to 9000 feet. Slopes range from about 0 to 10% on the valley floors to 15 to 60% on the escarpments. Mean annual precipitation ranges from about 13-30 inches. The Animas River Valley, Dolores River Valley, West Dolores River Valley, Lost Canyon, and Narraguinnep Canyon occur in this LTA. There are about XX acres of this LTA on the Forest.

- Landform: canyons (escarpments and valley floors)
- Climate Zone: semi-arid, lower montane, and montane
- PNV: pinyon pine, ponderosa pine, Douglas fir, and narrowleaf cottonwood series
- Existing Vegetation: pinyon-juniper woodland, ponderosa pine and warm-dry mixed conifer, cool-moist mixed conifer forests, mountain shrubland, riparian area
- Geologic Formation: Morrison, Dolores, Hermosa
- Geomorphic Process: fluvial
- Lithology: sedimentary rock and surficial deposits; mudstone, sandstone, shale, limestone, alluvium
- Soils: Haplustalfs, Hapludalfs, Fluvents, Rock Outcrop, shallow to deep, well to poorly drained
- Use & Management: livestock, wildlife habitat, riparian areas
- Interpretations: moderate to high erosion hazard on shale soils

LTA Name: 30SXM - Ponderosa Pine Forests on Mountain Slopes

This landtype association occurs throughout the low to middle elevations of the SJNF, at elevations ranging from about 7000 to 8500 feet. Slopes range from about 15 to 60%. Mean annual precipitation is about 16-25 inches. The Cherry Creek, Junction Creek, Devil Mountain, and Corral Mountain areas occur in this LTA. There are about XX acres of this LTA on the Forest.

- Landform: mountains
- Climate Zone: lower montane
- PNV: ponderosa pine series
- Existing Vegetation: ponderosa pine forest, mountain shrubland
- Geologic Formation: Dakota Sandstone, Burro Canyon, Mancos Shale, Morrison
- Geomorphic Process: fluvial
- Lithology: sedimentary rock; sandstone, shale, mudstone
- Soils: Haplustalfs, moderately-deep to deep, well drained
- Use & Management: timber, livestock, wildlife habitat
- Interpretations: moderate to high erosion hazard on shale soils

LTA Name: 30SXP – Ponderosa Pine Forests on Plateaus and Mesas

This landtype association occurs throughout the low to middle elevations of the SJNF, at elevations ranging from about 7000 to 8500 feet. Slopes range from about 0 to 15%. Mean annual precipitation is about 16-25 inches. Haycamp Mesa, the Boggy Draw area, and the Glade area on the west side of the Forest, and Brockover Mesa on the east side occur in this LTA. There are about xx acres of this LTA on the Forest.

- Landform: plateaus and mesas
- Climate Zone: lower montane
- PNV: ponderosa pine and Arizona fescue series
- Existing Vegetation: ponderosa pine forest, mountain shrubland, mountain grassland
- Geologic Formation: Dakota Sandstone, Burro Canyon, Mancos Shale
- Geomorphic Process: fluvial
- Lithology: sedimentary rock; sandstone and shale
- Soils: Haplustalfs, shallow to deep, well drained
- Use & Management: timber, livestock, OHV recreation, wildlife habitat
- Interpretations: moderate erosion hazard on shale soils

LTA Name: 35SXM - Ponderosa pine and Mixed Conifer Forests on Mountain Slopes

This landtype association occurs throughout the middle elevations of the SJNF, at elevations ranging from about 7000 to 9500 feet. Slopes range from about 15 to 60%. Mean annual precipitation is about 16-32 inches. The Hermosa Creek, Yellowjacket

Creek, and Piedra roadless areas occur in this LTA. There are about XX acres of this LTA on the Forest.

- Landform: mountains
- Climate Zone: lower montane and montane
- PNV: ponderosa pine, white fir, and Douglas fir series
- Existing Vegetation: ponderosa pine, warm-dry mixed conifer, cool-moist mixed conifer, and aspen forests, mountain shrubland
- Geologic Formation: Cutler, Rico, Hermosa, Molas, Dakota Sandstone, Burro Canyon, Mancos Shale, Morrison
- Geomorphic Process: fluvial
- Lithology: sedimentary rock; sandstone, shale, siltstone, mudstone
- Soils: Haplustalfs and Hapludalfs, moderately-deep to deep, well drained
- Use & Management: timber, livestock, wildlife habitat
- Interpretations: moderate to high erosion hazard on shale soils

LTA Name: 35SXU – Ponderosa pine and Mixed Conifer Forests on Mixed Landforms

This landtype association occurs throughout the middle elevations of the SJNF, at elevations ranging from about 7000 to 9500 feet. Slopes range from about 0 to 60%. Mean annual precipitation is about 16-32 inches. The HD Mountains, Chimney Rock, Burns Canyon, Kenny Flats, and Jackson Mountain areas occur in this LTA. There are about XX acres of this LTA on the Forest.

- Landform: mesas, hills, mountains, valleys
- Climate Zone: lower montane and montane
- PNV: ponderosa pine, white fir, and Douglas fir series
- Existing Vegetation: ponderosa pine, warm-dry mixed conifer, cool-moist mixed conifer, and aspen forests, mountain shrubland
- Geologic Formation: San Jose, Blanco Basin, Telluride, Animas, Mancos Shale, Lewis Shale
- Geomorphic Process: fluvial
- Lithology: sedimentary rock; sandstone, conglomerate, shale, claystone
- Soils: Haplustalfs and Hapludalfs, moderately-deep to deep, well drained
- Use & Management: timber, livestock, wildlife habitat, oil and gas
- Interpretations: moderate to high erosion hazard on shale soils

LTA Name: 50SXM – Mixed Conifer Forests on Mountain Slopes

This landtype association occurs throughout the middle elevations of the SJNF, at elevations ranging from about 7500 to 9500 feet. Slopes range from about 15 to 60%. Mean annual precipitation is about 20-32 inches. The Stoner Mesa, Taylor Mesa, Haycamp Mesa, Missionary Ridge, Vallecito Reservoir, Piedra roadless, East Fork San Juan River, and Buckles/Harris Lakes areas occur in this LTA. There are about XX acres of this LTA on the Forest.

- Landform: mountains and mesas

- Climate Zone: montane
- PNV: white fir and Douglas fir series
- Existing Vegetation: warm-dry mixed conifer, cool-moist mixed conifer, and aspen forests
- Geologic Formation: Dakota Sandstone, Burro Canyon, Mancos Shale, Cutler, Rico, Hermosa, Molas
- Geomorphic Process: fluvial
- Lithology: mostly sedimentary rock, some volcanic rock and surficial deposits; sandstone, shale, siltstone, andesite, landslide debris
- Soils: Haplustalfs and Hapludalfs, moderately-deep to deep, well drained
- Use & Management: timber, livestock, wildlife habitat
- Interpretations: moderate to high erosion hazard on shale soils

LTA Name: 70IGM – Spruce-Fir Forests on Igneous Mountain Slopes

This landtype association occurs at high elevations of the SJNF, at elevations ranging from about 9000 to 11,800 feet. Slopes range from about 15 to 80%. Mean annual precipitation is about 30-40 inches. Rock outcrop, talus slopes, and glaciated sites are common in this LTA. The Electra Lake, Needles Mountains, West Needles Mountains, Emerald Lake, and LaPlata Mountains areas occur in this LTA. There are about XX acres of this LTA on the Forest.

- Landform: mountains
- Climate Zone: subalpine
- PNV: mostly subalpine fir series, some Engelmann spruce and Thurber fescue series
- Existing Vegetation: spruce-fir and aspen forests
- Geologic Formation: Eolus Granite, Irving, Uncompahgre, Rocks of LaPlata Mountains
- Geomorphic Process: tectonic
- Lithology: igneous rock; monzonite, granite, gneiss, quartzite, amphibolite, diorite, syenite, porphyry
- Soils: Dystrocryepts, Rock Outcrop, shallow to moderately deep, well drained
- Use & Management: wildlife habitat, recreation, wilderness
- Interpretations: high erosion hazard on steep slopes

LTA Name: 70SXM – Spruce-Fir Forests on Sedimentary Mountain Slopes

This landtype association occurs at high elevations of the SJNF, at elevations ranging from about 9000 to 11,800 feet. Slopes range from about 15 to 80%. Mean annual precipitation is about 30-40 inches. Rock outcrop, talus slopes, and some glaciated sites are included in this LTA. The Taylor Mesa, Rico, Dunton, Kennebec Pass, Orphan Butte, Upper Hermosa Creek, Henderson Lake, and Mosca areas occur in this LTA. There are about XX acres of this LTA on the Forest.

- Landform: mostly mountains
- Climate Zone: subalpine

- PNV: mostly subalpine fir series, some Engelmann spruce and Thurber fescue series
- Existing Vegetation: spruce-fir and aspen forests, mountain grassland
- Geologic Formation: Cutler, Rico, Hermosa, Molas, Dakota Sandstone, Burro Canyon, Mancos Shale, Morrison
- Geomorphic Process: fluvial
- Lithology: sedimentary rock; sandstone, shale, siltstone, mudstone, limestone
- Soils: Haplocryalfs, Rock Outcrop, shallow to deep, well drained
- Use & Management: timber, livestock, wildlife habitat, recreation
- Interpretations: moderate to high erosion hazard on shale soils

LTA Name: 70VXM – Spruce-Fir Forests on Volcanic Mountain Slopes

This landtype association occurs at high elevations of the SJNF, at elevations ranging from about 9000 to 11,800 feet. Slopes range from about 15 to 80%. Mean annual precipitation is about 30-40 inches. Rock outcrop, talus slopes, and glaciated sites are included in this LTA. The Red Mountain Pass area, and much of the Weminuche and South San Juan Wilderness Areas occur in this LTA. There are about XX acres of this LTA on the Forest.

- Landform: mountains
- Climate Zone: subalpine
- PNV: mostly subalpine fir series, some Engelmann spruce and Thurber fescue series
- Existing Vegetation: spruce-fir and aspen forests, mountain grassland
- Geologic Formation: Volcaniclastic Facies, Near-Source Facies
- Geomorphic Process: volcanic
- Lithology: volcanic rock; andesite, rhyodacite, tuff, landslide debris
- Soils: Dystrocryepts, Haplocryalfs, Rock Outcrop, shallow to deep, well drained
- Use & Management: wildlife habitat, recreation, Wilderness
- Interpretations: high erosion hazard on steep slopes

LTA Name: 80XXM – Alpine Vegetation on Mountain Slopes

This landtype association occurs at high elevations of the SJNF, at elevations ranging from about 11,800 to 14,250 feet. Slopes range from about 0 to 80%. Mean annual precipitation is about 30-50 inches. Rock outcrop, talus slopes, and glaciated sites are common in this LTA. Vegetation is highly diverse. The LaPlata Mountains, Wilson Peak, Needles Mountains, Ice Lake Basin, Highland Mary Lakes, Engineer Mountain, Granite Peak areas, and the high peaks of the Weminuche and South San Juan Wilderness Areas occur in this LTA. There are about XX acres of this LTA on the Forest.

- Landform: mountains, glacial features
- Climate Zone: alpine
- PNV: kobresia, alpine avens, snow willow, planeleaf willow, tufted hairgrass, marsh marigold series

- Existing Vegetation: turf, fellfield, dwarf willow, and wetland types
- Geologic Formation: Eolus Granite, Irving, Uncompahgre, Volcaniclastic Facies, Near-Source Facies, Rocks of LaPlata Mountains, Intrusive Rocks, Cutler
- Geomorphic Process: fluvial, volcanic, glacial, tectonic
- Lithology: volcanic, sedimentary, and igneous rock; monzonite, granite, gneiss, quartzite, amphibolite, andesite, rhyodacite, tuff, landslide debris, diorite, syenite, porphyry, shale, siltstone, mudstone
- Soils: Humic Dystrocryepts, Cryaquolls, Rock Outcrop, shallow to moderately deep, well to poorly drained
- Use & Management: recreation, Wilderness, wildlife habitat,
- Interpretations: high erosion hazard on steep slopes, fragile soils, short growing season, revegetation problems

PNV, Community Descriptions

POTENTIAL NATURAL VEGETATION

This section describes the potential natural vegetation (PNV) within the San Juan Planning Area (SJPA) (Fig. 5-1). PNV is defined as the biotic community that would be established under present environmental conditions, if all successional sequences were completed without additional large-scale disturbance, both human-caused and natural (FSH 2090.11). PNV is described here mostly at the series level of classification, named for the species showing the strongest evidence of self-perpetuation (climax species).

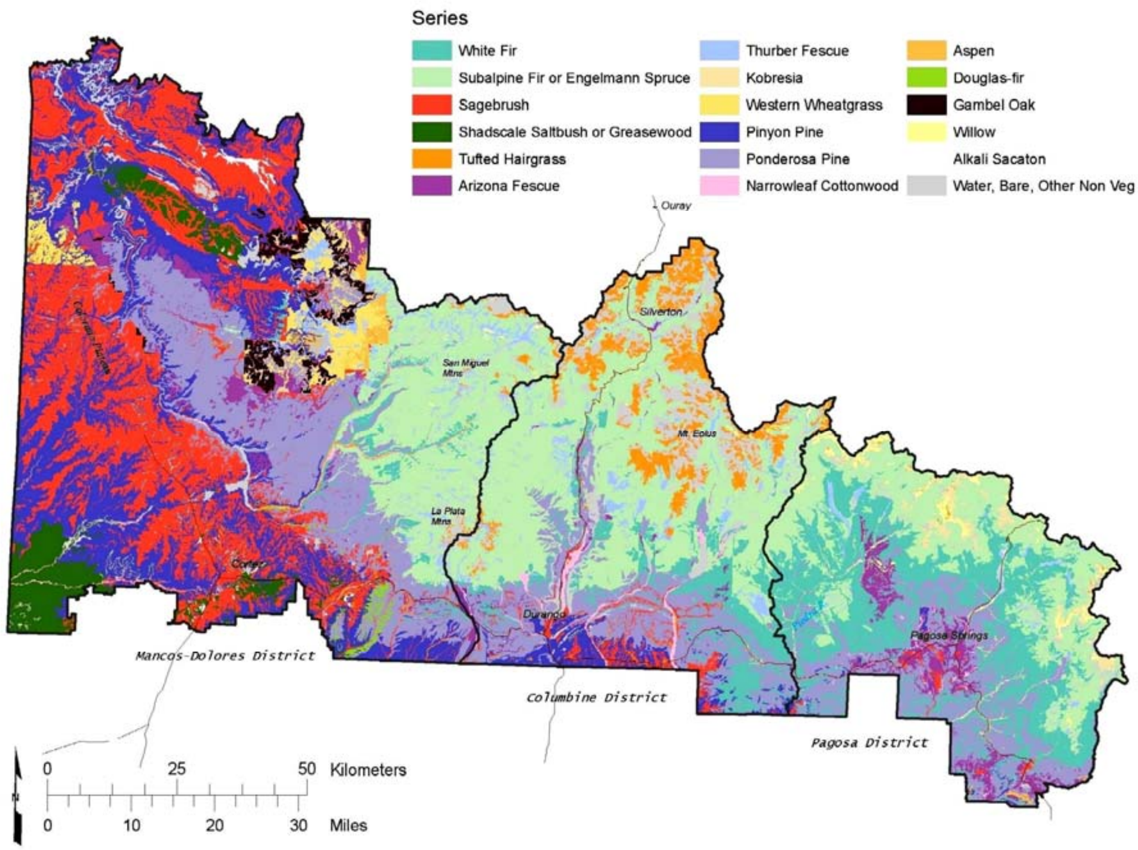


Figure 5-1 Potential Natural Vegetation- Forest Scale

Seral stages are plant communities representing stages along a successional pathway ranging from early-seral, which becomes established following a disturbance, to late-seral which represents the old growth or climax stage.

ENGELMANN SPRUCE SERIES - The Engelmann spruce series (DeVelice et al. 1986, Stuever and Hayden 1997, Johnston 1987) occurs throughout the highest elevations of the SJPA. Engelmann spruce (*Picea engelmannii*) trees dominate the overstory at climax, with subalpine fir trees occurring as minor components at the lowest elevations of this series. Successful regeneration of Engelmann spruce is characteristic of this series. Direct succession to a conifer-dominated forest occurs within this series, as this type is above the elevational limits of aspen (too cold). The major large-scale disturbance associated with these forests is fire. Small-scale disturbances include insect outbreaks, disease, wind events, and small fires. There are about 841,658 acres of this and the Subalpine fir PNV types within the SJPA.

Plant associations associated with this PNV type and found within the SJPA include Engelmann spruce/ blueberry-jacobs ladder (DeVelice et al. 1986).

SUBALPINE FIR SERIES - The Subalpine fir series (DeVelice et al. 1986, Stuever and Hayden 1997, Johnston 1987, Moir 1983) occurs throughout the high elevations of the SJPA. Subalpine fir (*Abies lasiocarpa or bifolia*) and Engelmann spruce (*Picea* *engelmannii*) trees dominate the overstory at climax. Successful regeneration of subalpine fir and Engelmann spruce is characteristic of this series. Aspen is a major seral species following disturbance on many sites within this series. Direct succession to a conifer-dominated forest occurs on some sites. The major large-scale disturbance associated with these forests is fire. Small-scale disturbances include insect outbreaks, disease, wind events, and small fires.

Plant associations associated with this PNV type and found within the SJPA include Subalpine fir/blueberry, Subalpine fir/forest fleabane, and Subalpine fir/thimbleberry (DeVelice et al. 1986).

The late-seral stage (includes old growth) of the Subalpine fir and Engelmann spruce series displays an uneven-aged, closed-canopy forest (¡Ý 70% cover) dominated by very large (¡Ý 16 inch), old (¡Ý 200 years) conifer trees (Engelmann spruce, subalpine fir, Douglas-fir or white fir for the Subalpine fir series and just Engelmann spruce for the Engelmann spruce series), many of which have dead or broken tops. Small, mid-sized, and large (< 16 inch), and young and middle-aged (< 200 years) conifer trees are present. Large (¡Ý 9 inch), old (¡Ý 80 years) aspen trees may be present for the Subalpine fir series. Structural diversity is high consisting of multiple canopy layers, many snags, and high amounts of large woody material on the forest floor. Herbs display low to moderate cover. Litter layer is thick.

WHITE FIR SERIES - The White fir series (DeVelice et al. 1986, Stuever and Hayden 1997, Johnston 1987, Moir 1983) occurs throughout the middle elevations of the SJPA. White fir (*Abies concolor),* Douglas-fir *(Pseudotsuga menziesii),* and ponderosa pine (*Pinus ponderosa*) trees or just the two firs dominate the overstory of the White fir series at climax. Successful regeneration of white fir is characteristic of the White fir series. Ponderosa pine can regenerate successfully on warmer, drier sites particularly when sites are opened up by disturbance, but it may decrease in abundance and have difficulty regenerating as stands age and become dominated by the more shade-tolerant white and Douglas-firs. Inclusions of the blue spruce series and limber pine series (DeVelice et al. 1986, Stuever and Hayden 1997, Johnston 1987) are found within this series. There are about 391,709 acres of this PNV type within the SJPA.

Gambel oak is a major seral species following disturbance on warm, dry sites. Aspen is a major seral species following disturbance on cool, moist sites. Subalpine fir and Engelmann spruce may be present as minor components on cool, moist sites, but they are not reproducing abundantly, and succession to the Subalpine fir series is unlikely within the next 100 years. The major large-scale disturbance associated with these forests is fire. Small-scale disturbances include insect outbreaks, disease, wind events, and small fires.

Plant associations associated with this PNV type and found within the SJPA include White fir/Gambel oak, White fir/blueberry, White fir/forest fleabane, and White fir/Rocky Mountain maple (DeVelice et al. 1986).

DOUGLAS-FIR SERIES - The Douglas-fir series (DeVelice et al. 1986, Stuever and Hayden 1997, Johnston 1987, Moir 1983) occurs throughout the middle elevations of the SJPA. Douglas-fir (*Pseudotsuga menziesii)* and ponderosa pine *(Pinus ponderosa)* trees dominant the overstory at climax. Successful regeneration of Douglas-fir is characteristic of this series. Ponderosa pine can regenerate successfully on warmer, drier sites particularly when sites are opened up by disturbance, but it may decrease in abundance and have difficulty regenerating as stands age and become dominated by the more shade-tolerant Douglas-fir. Gambel oak is a major seral species following disturbance on warm, dry sites. Aspen is a major seral species following disturbance on cool, moist sites. The major large-scale disturbance associated with these forests is fire. Small-scale disturbances include insect outbreaks, disease, wind events, and small fires. There are about 8,904 acres of this PNV type within the SJPA.

Plant associations associated with this PNV type and found within the SJPA include Douglas-fir/Gambel oak (DeVelice et al. 1986).

The late-seral stage (includes old growth) of the White fir and Douglas-fir series displays an uneven-aged, closed-canopy forest (¡Ý 70% cover) dominated by very large (¡Ý 16 inch), old (¡Ý 150 years) conifer trees (white fir, Douglas-fir, and ponderosa pine for the White fir series and Douglas-fir and ponderosa pine for the Douglas-fir series) many of which have dead or broken tops. Small, mid-sized, and large (< 16 inch), and young and middle-aged (< 150 years) conifer trees are present. Large (¡Ý 9 inch) and old (¡Ý 80 years) aspen trees may be present in the cool-moist phase. Structural diversity is high consisting of multiple canopy layers, many snags, and high amounts of large woody material on the forest floor. Tall shrubs display low to high cover. Herbs display low to moderate cover. Litter layer is moderate to thick.

PONDEROSA PINE SERIES - The Ponderosa pine series (DeVelice et al. 1986, Stuever and Hayden 1997, Johnston 1987, Moir 1983) occurs throughout the low to middle elevations of the SJPA. Ponderosa pine (*Pinus ponderosa)* trees dominate the overstory of these forests at climax. Successful regeneration of ponderosa pine is characteristic of this series. Gambel oak is a major seral species following disturbance. Douglas-fir and white fir may be present as minor components on cool, moist sites, but they are not reproducing abundantly, and succession to the White fir or Douglas-fir series is unlikely within the next 100 years. The major large-scale disturbance associated with these forests is fire. Small-scale disturbances include insect outbreaks, disease, wind events, and small fires. There are about 637,173 acres of this PNV type within the SJPA.

Plant associations associated with this PNV type and found within the SJPA include Ponderosa pine/Gambel oak and Ponderosa pine/Arizona fescue (DeVelice et al. 1986).

The late-seral stage (includes old growth) of the Ponderosa pine series displays an uneven-aged, moderately closed to closed-canopy forest (≥ 70% cover) dominated by very large (≥ 16 inch), old (≥150 years) ponderosa pine trees, many of which have dead or broken tops. Small, mid-sized, and large (< 16 inch), and young and middle-aged (< 150 years) ponderosa pine trees are present. Structural diversity is high consisting of multiple canopy layers, common to many snags, and moderate amounts of large woody material on the forest floor. Tall shrubs (including Gambel oak) display low to high cover. Herbs display low to moderate cover. Litter layer (mostly needles) is thick in many places, particularly under the larger trees.

ASPEN SERIES - The Aspen series (Hess and Alexander 1986, Johnston and Hendzel 1985, Johnston 1987, Hess and Wasser 1982, Hoffman and Alexander 1983, Johnston 2001) occurs throughout the low, middle, and high elevations of the SJPA. Aspen (*Populus tremuloides)* trees dominate the overstory of these forests at climax, with conifer species absent or very minor components. These stable aspen forests are unlikely to succeed to conifer forests over the next 100 years or longer. These forests, mostly occurring on the west side of the Forest, were likely established by frequent fire during the reference period, which eliminated the conifer seed source. There are about 14,559 acres of this PNV type within the SJPA.

Plant associations associated with this PNV type and found within the SJPA include Aspen/Elk sedge, Aspen/Common juniper, Aspen/Brackenfern, and Aspen/Snowberry (Johnston 1987).

The late-seral stage (includes old growth) of the Aspen series displays an uneven-aged, closed-canopy forest (≥ 70% cover) dominated by large (≥ 9 inch), old (≥ 80 years) aspen trees, many of which have dead or broken tops. Small and mid-sized (< 9 inch), and young and mid-aged (25-79 years) aspen trees may be present. Small, mid-sized, and large (< 16 inch), and young and middle-aged (< 150 years) conifer trees (Engelmann spruce, subalpine fir, Douglas-fir, white fir) may be present. Structural diversity is high consisting of multiple canopy layers, many snags, and high amounts of large woody material on the forest floor. Tall shrubs display low to high cover. Herbs display moderate to high cover. Litter layer is thick.

PINYON PINE SERIES - The Pinyon pine series (Stuever and Hayden 1997, Moir and Carleton 1986, Johnston 1987, Moir 1983) occurs throughout the lowest elevations of the SJPA. Pinyon pine (*Pinus edulis)* and Utah juniper *(Juniperus osteosperma)* trees dominant the overstory of this series at climax. Successful regeneration of pinyon pine is characteristic of these forests. Ponderosa pine may be present as a minor component, but it is not reproducing abundantly, and succession to a Ponderosa Pine forest type is unlikely within the next 100 years. Gambel oak, a major seral species following disturbance within this type, is a common component and very abundant on many sites. There are about 536,452 acres of this PNV type within the SJPA.

Plant associations associated with this PNV type and found within the SJPA include Pinyon pine/Big sagebrush, Pinyon pine/Gambel oak, Pinyon pine/blue gramma, and Pinyon pine/muttongrass (Stuever and Hayden 1997).

NARROWLEAF COTTONWOOD SERIES - The narrowleaf cottonwood series (Johnston 1987, Dick-Peddie 1993, Stuever and Hayden 1997, Moir 1983, Johnston

2001, Hess and Wasser 1982) occurs throughout the low and middle elevations of the SJPA. It occurs on terraces and floodplains of valley floors usually adjacent to streams. Narrowleaf cottonwood (*Populus angustifolia)* trees dominate the overstory of these forests at climax. The major large-scale disturbance associated with these forests is flooding. Small-scale disturbances include disease and wind events. There are about 28,182 acres of this PNV type within the SJPA.

Plant associations associated with this PNV type and found within the SJPA include Narrowleaf Cottonwood/Alder, Narrowleaf Cottonwood/Red-osier Dogwood, Narrowleaf Cottonwood/Hawthorn, Narrowleaf Cottonwood/Willow, and Narrowleaf Cottonwood/ Coyote Willow (Redders 2003).

GAMBEL OAK SERIES – The Gambel oak series (Johnston 1987, Dick-Peddie 1993) occurs throughout the low and middle elevations of the SJPA. Gambel oak (*Quercus gambelii)* shrubs dominate the overstory of these shrublands at climax. There are about 35,542 acres of this PNV type within the SJPA.

SAGEBRUSH SERIES - The big sagebrush series (Johnston 1987, Moir 1983, Hess and Wasser 1982) occurs throughout the low and middle elevations of the SJPA. Big sagebrush (*Seriphidium tridentatum ssp tridentatum)* shrubs dominate the overstory of these shrublands at climax. There are about 537,789 acres of this PNV type within the SJPA.

Inclusions of the black sagebrush series (Johnston 1987) and silver sagebrush series (Johnston 1987) are found within this series. Plant associations associated with this PNV type and found within the SJPA include big sagebrush/blue grama and big sagebrush/western wheatgrass (Johnston 1987).

GREASEWOOD SERIES - The greasewood series (Johnston 1987) occurs mostly on BLM lands on the west side and lowest elevations of the planning area. Greasewood (*Sarcobatus vermiculatus)* shrubs dominate these shrublands at climax. There are about 100,529 acres of this and the shadscale saltbush PNV types within the SJPA.

SHADSCALE SALTBUSH SERIES - The shadscale saltbush series (Johnston 1987, Dick-Peddie 1993) occurs mostly on BLM lands on the west side and lowest elevations of the planning area. Shadscale saltbush (*Atriplex confertifolia)* shrubs dominate these shrublands at climax. The plant association associated with this PNV type within the SJPA is Atriplex confertifolia-Atriplex canescens/Sporobolus airoides (Johnston 1987).

TUFTED HAIRGRASS SERIES - The tufted hairgrass series (Johnston 1987, Dick-Peddie 1993, Moir 1983, Hess and Wasser 1982) occurs throughout the middle and high elevations of the SJPA. Tufted hairgrass (*Deschampsia cespitosa),* a perennial bunchgrass, dominates the overstory of these grasslands at climax. Inclusions of the sedge species series (Johnston 1987), mixed sedge series (Dick Peddie 1993), and water sedge and beaked sedge series (Redders 2003, Johnston

2001) are found within this series. There are about 112,364 acres of this PNV type within the SJPA.

This series is characterized by dense stands of tufted hairgrass, which is generally 0.3-0.8 meters in height. In the spaces between the bunchgrass rosettes, small forbs or other graminoids often occur. This series occurs on sites with moist and wet soils, which are seasonally flooded by snowmelt and retain moisture throughout the growing season. The vegetation often occurs adjacent to perennially saturated sedge wetlands dominated by beaked and water sedge. Commonly associated graminoids include water sedge*, Baltic rush, and alpine timothy.*

THURBER FESCUE SERIES - The Thurber fescue series (Moir 1983, Johnston 1987, Hess and Wasser 1982, Dick-Peddie 1993, Johnston 2001) occurs throughout the middle and high elevations of the SJPA. It occurs as openings in forest-dominated planning areas. Thurber fescue (*Festuca thurberi),* a medium-tall perennial bunchgrass, dominates the overstory of these grasslands at climax. It occurs on upland sites with well-drained soils. It occurs at elevations from about 8500 to 11600 feet, and is primarily associated with spruce-fir and cool-moist mixed-conifer forests. There are about 86,198 acres of this PNV type within the SJPA.

This series is characterized by dense stands of Thurber fescue, which is generally 0.5-1.5 meters in height. In the spaces between the bunchgrass rosettes, forbs or other graminoids occur including Arizona fescue, timber oatgrass, mountain brome, needlegrass, wild rye, slender wheatgrass, and American vetch. Bare soil is typically in the 10 to 20% range.

ARIZONA FESCUE SERIES - The Arizona fescue series (Moir 1983, Johnston 1987, Dick-Peddie 1993, Johnston 2001) occurs throughout the low and middle elevations of the SJPA. It occurs as openings in forest-dominated planning areas. Arizona fescue (*Festuca arizonica),* a medium-tall perennial bunchgrass, dominates the overstory of these grasslands at climax. It occurs on upland sites with well-drained soils. It occurs at elevations from about 7500 to 9000 feet, and is primarily associated with ponderosa pine and warm-dry mixed-conifer forests. There are about 115,824 acres of this PNV type within the SJPA.

This series is characterized by dense stands of Arizona fescue, which is generally 0.5-1 meter in height. In the spaces between the bunchgrass rosettes, forbs or other graminoids occur including mountain muhly, Parry oatgrass, blue grama, junegrass, and American vetch. Bare soil is typically in the 10 to 20% range.

KOBRESIA SERIES - The kobresia series (Johnston 1987, Dick-Peddie 1993, Hess and Wasser 1982) occurs on mostly well-drained soils throughout the high elevations of the SJPA. It occurs in the alpine zone above timberline at elevations above about 11,500 feet. Kobresia (*Kobresia myosuroides),* a graminoid in the sedge family, dominates the overstory of these grasslands at climax. Inclusions of the alpine avens and fellfield series (Dick-Peddie 1993, Johnston 1987) are found within this series. There are about 25,273 acres of this PNV type within the SJPA.

This series is characterized by moderately dense to dense stands of Kobresia. Other common graminoids can include Carex rupestris var. drummondiana, tufted hairgrass, and Carex elynoides. Alpine avens, a forb, is a common associate.

WESTERN WHEATGRASS SERIES – The western wheatgrass series occurs throughout the low elevations of the SJPA. Western wheatgrass (*Pascopyrum smithii)* dominates the overstory of these grasslands at climax. There are about 61,299 acres of this PNV type within the SJPA.

ALKALI SACATON SERIES – The alkali sacaton series (Dick-Peddie 1993) occurs mostly on BLM lands on the west side and lowest elevations of the planning area. Alkali sacaton *(Sporobolus airoides)* dominates the overstory of these grasslands at climax. There are about 11,550 acres of this PNV type within the SJPA.

WILLOW SUBCLASS - The willow subclass (Carsey et al. 2003) or series (Dick-Peddie 1993, Johnston 1987) is dominated by deciduous willows at climax. Willows (salix) are found along the stream channel and usually display high cover in the specific series named after them. The Planeleaf Willow, Wolf Willow, Shortfruit Willow, and Geyer Willow series are wetland types that occur at elevations above about 9500 feet. Drummond Willow and Mountain Willow series occur at mid to high elevations above about 8000 feet. The Bebb, Strapleaf, and Coyote Willow series occur at elevations from about 7000 to 9000 feet. Graminoids and forbs are found in patches between the shrubs. There are about 26,294 acres of this PNV type within the SJPA.

Plant associations associated with this PNV type and found on the SJNF include Drummond Willow/Mesic Forb, Mountain Willow/Mesic Forb, Coyote Willow/Mesic Graminoid, Wolf Willow/Mesic Forb, Coyote willow/Bare Ground, Geyer willow/Mesic Forb, Shortfruit Willow/Mesic Forb, Planeleaf Willow/Marsh Marigold, Planeleaf Willow/Water Sedge, Wolf Willow/Water Sedge, and Wolf Willow/Beaked Sedge (Redders 2003).

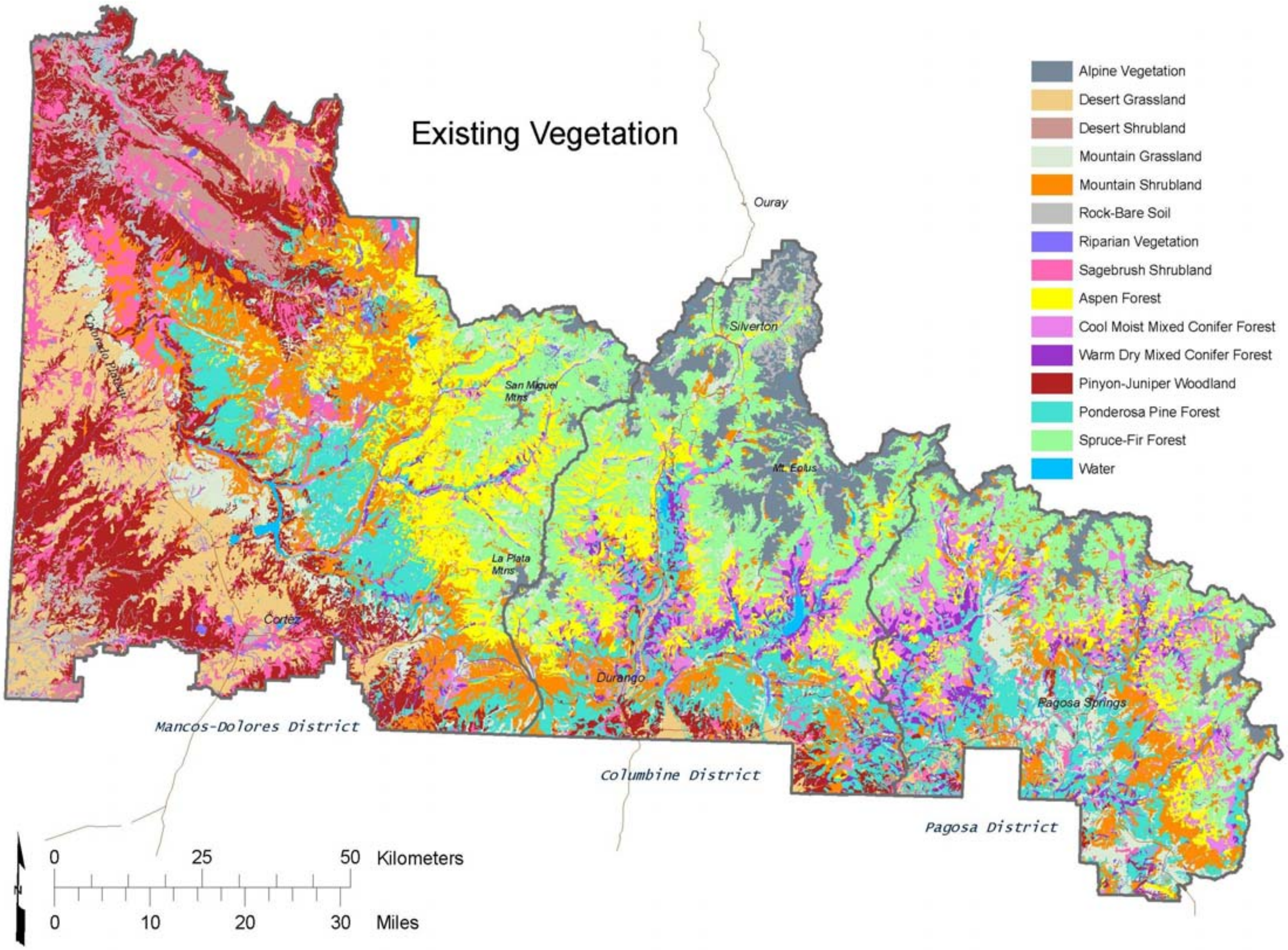


Figure 5-2 Forest Scale- Existing Vegetation

Existing Vegetation Types - Forest Scale

Spruce-Fir Forest Type

Composition and Distribution of Spruce-Fir Forests

The Spruce-Fir forest type within the San Juan Planning Area (SJPA) is dominated by Engelmann spruce and subalpine fir trees. It occurs throughout the SJPA on mountain and plateau landscapes at elevations ranging from about 9000 to 11,800 feet. Spruce-Fir forests are the coldest and wettest forest type within the SJPA. They are associated with the subalpine climatic zone, the cryic soil temperature regime, and the udic soil moisture regime. Mean annual precipitation ranges from about 30-40 inches. There are about 510,220 acres of this type within the SJPA. More than one-third of the spruce-fir forest type within the SJPA occurs within designated wilderness areas (South San Juan Wilderness, Weminuche Wilderness, Lizard Head Wilderness, and the Piedra Area).

Engelmann spruce and subalpine fir trees combined, make up the largest component within a stand (comprise a stocking plurality in association with other species) and occur together in most stands. Engelmann spruce occurs alone at the highest elevations of this type. Douglas fir and white fir occur in association with Engelmann spruce and subalpine fir on the warmest and driest sites (usually at the lowest elevations) within the Spruce-Fir type, (although white fir is absent or only a minor component on the west side of the Forest, west of the high peaks of the La Plata Mountains). Aspen trees are a common component in spruce-fir forests, and often display high cover. Blue spruce and limber pine may be present, usually as minor components (< 10% cover), the latter commonly found on steep slopes with southerly aspects. Bristlecone pine is a rare component within the SJPA, occurring near timberline on steep slopes with southerly aspects on only a few sites on the Pagosa District. Lodgepole pine, a species not native to the SJPA, occurs to a minor extent (1227 acres) in small patches within the spruce-fir type. The combined canopy cover of trees for is ¡Ý 25%, or trees make up the most canopy cover of any life form (CVU, Eyre 1980).

Due to the variability in the composition of the spruce-fir forest type at the stand level, it can be separated into cold-wet and warm-dry phases, the former occurring on the coldest and wettest sites within the type (usually at the highest elevations) where Douglas-fir and white fir are absent or minor components (< 15% combined overstory cover), and the latter occurring on the warmest and driest sites within the type (usually at the lowest elevations) where Douglas-fir and white fir display a combined overstory canopy cover ¡Ý 15%. There are about 460,313 acres of the cold-wet phase and about 50,000 acres of the warm-dry phase within the SJPA.

This spruce-fir forest type is associated with the Engelmann spruce-Subalpine fir (SAF #206) and the IRI-CVU Spruce-Fir cover types. Potential natural vegetation communities associated with this type at the series level of classification are Engelmann spruce and Subalpine fir, and at the plant association level are Subalpine fir/blueberry, Subalpine fir/forest fleabane, Subalpine fir/thimbleberry, and Engelmann spruce/blueberry-jacobs ladder (DeVelice et al. 1986).

Some spruce-fir forests succeeded from aspen-dominated forests that were established following stand-replacing fires. Others formed through the establishment of Engelmann spruce, subalpine fir, white fir, or Douglas-fir trees which initially colonized sites following disturbance events. Some may have succeeded from cold-wet mixed conifer forests where the less shade-tolerant Douglas-fir component decreased, as the more shade-tolerant Engelmann spruce and subalpine fir components increased in abundance. Many warm-dry spruce-fir forests are likely to maintain their Douglas-fir component over time due to natural disturbances (fire, insects, and disease) and environmental factors (microclimate,slope, aspect, soil type) that perpetuate a more open canopy and give the Douglas-fir component a competitive advantage to persist and regenerate.

The spruce-fir forest type dominates some high elevation landscapes displaying large, relatively uniform, closed-canopy stands of mid to late seral spruce-fir forest, or displaying smaller more diverse spruce-fir stands that vary in size, shape, age, and canopy cover. Some landscapes display spruce-fir stands intermixed with aspen forests, cool-moist mixed conifer forests, alpine tundra, mountain grasslands, wetlands, riparian areas, rock outcrop, or talus slopes. Diverse geology and mountain topography add to the complexity and diversity of these landscapes. Many spruce-fir-dominated landscapes are unroaded and mostly undisturbed by human activities. Many others are roaded and have been extensively harvested.

Common shrubs and herbs found in the spruce-fir forest type include blueberry, common juniper, current, elderberry, honeysuckle, thimbleberry, mountain ash, maple, raspberry, buffaloberry, mountain lover, arnica, one-sided wintergreen, lousewort, strawberry, sweet cicely, violet, peavine, death camas, baneberry, sagewort, false Solomon's seal, forest fleabane, jacobs ladder, goldenweed, pipsissewa, columbine, bluebells, rattlesnake plantain, Richardson geranium, avalanche lily, fairy slipper, pussytoes, fireweed, wood nymph, wintergreen, fringed bromegrass, trisetum, ryegrass, mountain ricegrass, and elk sedge.

Stand Structure and Habitat Structural Stages of Spruce-Fir Forests

Stand Structure and Habitat Structural Stages of Spruce-Fir Forests (Table 5-2)

The Sapling-Pole, Mature, and Old Growth stages are extensions of the Wildlife Habitat Structural Stages (HSS) found in the "Standard Specifications Stand Exam" guide. In addition to tree size and crown cover (the 2 components used for HSS) these stage descriptions include other structural and compositional components important to ecosystems and wildlife species.

The Sapling-Pole Stage (HSS 3a and 3b) of the spruce-fir forest type displays an even or uneven-aged, open to moderately closed-canopy forest (25-69% cover) dominated by small to mid-sized (< 9 inch), young (< 50 years) conifer trees (Engelmann spruce, subalpine fir, Douglas-fir, white fir). Small (< = 5 inch) and young (< = 25 years) aspen trees may be present. Structural diversity is low to moderate consisting of one or two canopy layers, few to many snags, and low to high amounts of large woody material on the forest floor. More snags and large woody material would be expected if the disturbance event that initiated this stage left abundant standing and down woody material, which is common following a fire or wind event. Tall shrubs and herbs display moderate to high cover. Litter layer is thin to moderately thick.

The Mature Stage (HSS 4a, 4b, 4c) of the spruce-fir forest type displays an uneven-aged, moderately closed to closed-canopy forest (> = 40% cover) dominated by large (9-15 inch), mid-aged (50-199 years) conifer trees (Engelmann spruce, subalpine fir, Douglas-fir, white fir), some of which have dead or broken tops. Small to mid-sized (< 9 inch) and young (< 50 years) conifer trees are present. Mid-sized to large (> =5 inch) and mid-aged to old (> = 25 years) aspen trees may be present. Structural diversity is high consisting of multiple canopy layers, common to many snags, and moderate to high amounts of large woody material on the forest floor. More snags and large woody material would be expected if the stand succeeded from an aspen forest. Tall shrubs display low to moderate cover. Herbs display low to moderate cover. Litter layer is thick.

The Old Growth Stage (HSS 5) of the spruce-fir forest type displays an uneven-aged, closed-canopy forest (> = 70% cover) dominated by very large (> = 16 inch), old (> = 200 years) conifer trees (Engelmann spruce, subalpine fir, Douglas-fir, white fir), many of which have dead or broken tops. Small, mid-sized, and large (< 16 inch), and young and middle-aged (< 150 years) conifer trees are present. Large (> = 9 inch), old (> = 80 years) aspen trees may be present. Structural diversity is high consisting of multiple canopy layers, many snags, and high amounts of large woody material on the forest floor. Tall shrubs and herbs display low to moderate cover. Litter layer is thick.

Table 5-2 Current Acres of Spruce-Fir Forest by Habitat Structural Stage

HSS	HSS Description	Acres
3A	Sapling-pole (3-8" dbh), < 40% canopy closure	16,972
3B	Sapling-pole (3-8") dbh, 40-69% canopy closure	13,424
3C	Sapling-pole (3-8" dbh), ≥ 70% canopy closure	2,269
4A	Mature (9-15" dbh), < 40% canopy closure	61,749
4B	Mature (9-15" dbh), 40-69% canopy closure	248,282
4C	Mature (9-15" dbh), ≥ 70% canopy closure	47,671
5	Old Growth (≥ 16" dbh), ≥ 70% canopy closure	117,482

Historic Range of Variability (HRV) of Spruce-Fir Forests (McGarigal and Romme 2005)

The pre-1900 range of variability (hereafter referred to as the "historic range of variability" (HRV) disturbance regimes and vegetation dynamics is described from about 1300 to the late 1800s, and represents the period from Ancestral Puebloan abandonment to EuroAmerican settlement (the period of indigenous settlement). This period represents a time when broad-scale climatic conditions were generally similar to those of today, but Euro-American settlers had not yet introduced the sweeping ecological changes that now have greatly altered many Rocky Mountain landscapes. Moreover, it was a time of relatively consistent (though not static) environmental and cultural conditions in the region, and a time for which we have a reasonable amount of specific information to enable us to model the system.

The two most important and ubiquitous kinds disturbance agents in high-elevation spruce-fir forest landscapes during the HRV period were stand-destroying fire and insect outbreaks (Baker and Veblen 1990, Veblen et al. 1994, Veblen,

Spruce-fir forests are subject to two different insect disturbance processes in RMLANDS: spruce beetle and western spruce budworm. Because spruce-fir stands are assumed to contain a mixture of spruce and fir, neither insect alone is sufficient to cause a stand-replacing event. Rather, an epidemic of one insect will simply shift the tree species composition of the forest in favor of the non-host species. Both insects working in concert, however, can result in near or complete overstory mortality and therefore invoke stand-replacement (although this is very uncommon).

The spruce beetle (*Dendroctonus rufipennis)* is a native insect whose larvae feed on the phloem of large, living or dead, Engelmann spruce trees *(Furniss and Carolin 1977, Schmid and Mata 1996).* Most of the time the beetles persist as low-density, endemic populations that have little impact on forest structure. Periodically, however, populations explode into an outbreak, and the beetles may kill millions of mature spruce trees over thousands of hectares.

The frequency and extent of simulated wildfires in spruce-fir forests (includes RMLANDS spruce-fir and spruce-fir-aspen forest types) varied markedly among decades. In almost half of the decades, less than 1% of the spruce-fir forest burned, inclusive of both high and low-mortality affected areas. However, on average, almost once per 100 years (70 for spruce-fir-aspen), > 10% of the area burned, and roughly once every 300-400 years (200-300 for spruce-fir-aspen), > 20% of the spruce-fir forest burned. Under this wildfire regime, the return interval between fires (of any mortality level) varied widely from 38 years to > 800 years, with a mean and median of 267 years (220 and 267 for spruce-fir-aspen), although very little eligible area (1%) escaped disturbance altogether over the course of an 800-year simulation. As expected, return intervals varied spatially across the forest. In general, return intervals increased with elevation, reflecting the moister, cooler conditions at higher elevations. In addition, spruce-fir stands embedded in a neighborhood containing cover types with shorter return intervals (aspen) exhibited shorter return intervals, reflecting the importance of landscape context on fire regimes.

The frequency and extent of simulated spruce beetle epidemics in spruce-fir forest varied markedly among decades. In almost half of the decades, less than 1% of the spruce-fir forest was disturbed, inclusive of both high and low-mortality affected areas. However, on average, roughly twice per 100 years, > 10% of the area was disturbed, and perhaps once every 400 years, > 40% of the spruce-fir forest was disturbed. The return interval between epidemics (of any mortality level) at a single location varied widely from 47 years to > 800 years, with a mean and median of about 160 years, although very little eligible area (< 2%) escaped disturbance altogether over the course of an 800-year simulation. Not surprisingly, the return interval varied spatially across the forest and exhibited a highly contagious or clumped distribution. In general, the areas of most extensive and connected spruce-fir forest exhibited the shortest return intervals, but there was considerable variation among runs owing to the stochastic nature of spruce beetle outbreaks.

The frequency and extent of simulated spruce budworm epidemics in spruce-fir forest varied markedly among decades. In almost two-thirds of the decades, spruce budworm populations *were* at endemic levels and none of the spruce-fir forest was disturbed. However, roughly 3 times per 100 years, epidemics occurred affecting > 10% of the area, and approximately once every 400 years, 50% of the spruce-fir forest was disturbed. The return interval between epidemics (of any mortality level) at a single location varied from 36 years to > 800 years, with a mean and median of 114 years, although almost no eligible area (< 1%) escaped disturbance altogether over the course of an 800-year simulation. The return interval varied spatially across the forest, but at a very fine grain and in a seemingly random pattern.

The age structure and dynamics of spruce-fir forest reflected the interplay between disturbance and succession processes. The survivorship distribution represents the percent of stands that survived to any age, where age represents time since stand origin, not necessarily the age of the oldest trees in the stand. On average (over time), roughly 50% of the spruce-fir forest was > 160 years since stand origin, although at any point in time this varied from < 27% to > 77%. On average, about 33% of the spruce-fir forest survived to > 300 years, 5% survived to > 560 years, and a very small percentage (~1%) survived a stand-replacing disturbance for > 800 years. This highlights the stochastic nature of disturbances, in which some areas by chance alone escaped catastrophic disturbance for very long periods.

The distribution of area among stand conditions within spruce-fir forest fluctuated markedly over time, as expected. For example, the percentage of spruce-fir forest in late-seral stages (i.e., understory reinitiation and shifting mosaic conditions) varied from 32% to 93%, reflecting the extremely dynamic nature of this high elevation forest cover type when considered over century-long periods. Due to the large fluctuations, it was not clear whether the seral-stage distribution achieved dynamic equilibrium over the 800-year simulation period (the percentage of spruce-fir forest in each stand condition did not reach a perfectly stable mean). This likely reflects the relatively long rotation period of 300-350 years (250-300 for spruce-fir-aspen) of the major stand-replacing disturbance process (wildfire) relative to the simulation length (800 years). The spatial configuration of stand conditions fluctuated markedly over time as well, although there was considerable variation in the magnitude of variability among configuration metrics. Patch area, core area and the proximity index (a measure of patch isolation) exhibited the greatest variability over time.

For the RMLANDS spruce-fir type, our estimate of the current seral-stage distribution was almost never observed under the simulated HRV. Specifically, the current landscape contains only 6% of this cover type in the stand initiation condition, yet this situation was almost never observed under the simulated HRV (~0^{th} percentile of the HRV distribution), and 32% in the stem exclusion condition, yet 93% of the time there was less than this amount under the simulated HRV. However, the current landscape contains a preponderance (62%) of spruce-fir forest in the late-seral stages (i.e, understory reinitiation and shifting mosaic conditions), and this situation was quite common under the simulated HRV (59th percentile of the HRV distribution). Overall, based on the four separate stand conditions (without aggregating the late-seral stages), the seral-stage departure index was 93%. When the late-seral stages were aggregated, the seral-stage departure index declined dramatically to 57%. The current seral-stage configuration deviated substantially (86%) from the simulated HRV, although there was considerable variation (50-100% deviation) among metrics. Unfortunately, the nature of the configuration departure varied depending on whether the late-seral stages were treated separately or combined into a single class. Thus, the only reliable conclusion we can reach is that, in general, the current landscape contains fewer, smaller and geometrically less complex and more isolated patches of early-seral forest than existed under the simulated HRV.

For the RMLANDS spruce-fir-aspen type, our estimate of the current seral-stage distribution was almost never observed under the simulated HRV. Specifically, the current landscape contains only 1% and 8% of this cover type in the stand initiation and stem exclusion conditions, respectively, yet this situation was uncommon under the simulated HRV (5th and 3rd percentiles, respectively, of the HRV distribution). The paucity of spruce-fir-aspen forest in the stem exclusion condition compared to the reverse situation in spruce-fir forest likely reflects differences in succession between these two cover types. In the spruce-fir-aspen type, the stem exclusion stage is dominated by aspen, and therefore succession to the understory reinitiation stage of development is dictated by the rapid rate of maturity and senescence of the relatively short-lived aspen. Thus, spruce-fir-aspen stands succeed to the understory reinitiation condition much sooner than spruce-fir stands. Due to the lack of large disturbances over the past century, most of the spruce-fir-aspen forest has already succeeded to the understory reinitiation or shifting mosaic stages, whereas much of the spruce-fir forest still remains in the stem exclusion stage. Consequently, the current landscape contains a preponderance (92%) of spruce-fir-aspen forest in the late-seral stages (understory reinitiation and shifting mosaic conditions), a situation almost never observed under the simulated HRV.

Overall, based on the four separate stand conditions (without aggregating the late-seral stages), the seral-stage departure index was 92%, almost identical to that observed for spruce-fir forest. When the late-seral stages were aggregated, the seral-stage departure index declined slightly to 89%. The current seral-stage configuration deviated substantially (70%) from the simulated HRV, although there was considerable variation (40-98% deviation) among metrics and, in general, the departure was less than observed for spruce-fir forest. Unfortunately, the nature of the configuration departure varied depending on whether the late-seral stages were treated separately or combined into a single class. Thus, we were unable to reach any reliable conclusions regarding configuration departure in the late-seral stages. In addition, there does not appear to be any consistent and strong pattern of configuration departure in the early- or mid-seral stages for this cover type, although there is some indication that the current landscape contains fewer, smaller and geometrically less complex and more isolated patches of early- and mid-seral forest than existed under the simulated HRV.

Management Implications for Spruce-Fir Forests (Romme et al. 2003)

The biotas of the SJPA are well adapted to the wide range of natural disturbances that have affected them during their evolutionary history (fires, insect outbreaks, wind storms, and climatic variability). However, some recent anthropogenic disturbances may be unprecedented (different in kind and/or intensity from any of the natural disturbances to which the organisms are adapted). Therefore, a coarse-filter strategy for reducing the impacts of anthropogenic disturbance is to design the human-caused disturbances to mimic natural disturbances to the greatest extent possible (Romme et al., *in press).* Strategies for making anthropogenic disturbances mimic more closely the natural disturbance regime in high-elevation landscapes involve three primary considerations: (1) revising the size, shape, and spatial arrangement of logging units, (2) providing biological legacies of the former stand in logged areas, and (3) minimizing roads.

Design of logging units since the 1970s has emphasized small cutting units widely dispersed across the landscape. The rationale has been reduction of aesthetic impacts of clearcuts, enhancement of tree regeneration, and buffering of undesirable effects such as soil erosion and stream sedimentation. However, recent empirical and theoretical research has demonstrated clearly that this strategy maximizes fragmentation of mature and old growth forest (Franklin and Forman 1987, Li et al. 1993, Forman 1995, Franklin et al. 1997). Moreover, the natural fire regime was one of infrequent but large disturbances rather than frequent, small, spatially dispersed burns and native biota have numerous adaptations enabling them to become reestablished readily in large as well as small burned patches (Turner et al. 1997). Many of the problems of poor regeneration and erosion in large clearcuts are related to the lack of biological legacies in clearcuts rather than clearcut size per se (see below). Therefore, many ecologists are now recommending that logging units should be larger in size, and should be aggregated within a small portion of the landscape with patch sizes and shapes that resemble fire-created patches (Li et al. 1993, Crow and Gustafson 1997, Franklin et al. 1997). This design minimizes fragmentation of mature forest and assures at least some large patches of mature forest, and it mimics the natural disturbance regime with respect to spatial patterns of disturbance. This recommendation does not imply that a greater total area should be logged; rather it deals with the optimal spatial patterning of cutting units given the target for timber production that has been established through the broader planning process.

Recently logged areas in subalpine forests of the central Rocky Mountains usually bear little resemblance to areas of recent fires or insect outbreaks. Most conspicuous is the paucity of large standing and fallen dead trees in logged areas (Spies et al. 1988, Hutto 1995, Wei et al. 1997). These and other "biological legacies" of the previous stand are critical structural elements for numerous species and ecological processes involved in recovery from the disturbance (Franklin et al. 1997, Kaila et al. 1997, Spies 1997). Therefore, Franklin et al. (1997) and others are now recommending variable retention harvest systems, based on retaining structural elements of a harvested stand (large trees and snags) for at least the next rotation, to achieve three objectives: (i) "lifeboating" species and processes immediately after logging, (ii) "enriching" the new forest stand with structures that otherwise would be lacking, and (iii) "enhancing connectivity" between patches of unlogged forest habitat (Franklin et al. 1997).

Finally, attention should be given to reducing the extent and impact of roads, which have little or no precedent in the evolutionary history of the native biota, and which strongly facilitate the fragmentation of interior forest habitats, dispersal of alien species, and human-caused disturbance of wildlife. Another advantage of logging large, aggregated patches rather than small, dispersed patches is that the former spatial pattern requires a less extensive road system. Necessary roads also can be placed in areas that minimize their fragmenting effects, like running around a large patch of mature forest rather than bisecting it (Reed et al. 1996a), and unnecessary roads can be permanently closed. Finally, attention should be given to implementing logging techniques that do not require roads and skid trails, like removing logs with horses in winter. Economic considerations may preclude such techniques in many areas, but they should be explored and used where possible to reduce the heavy impact of roads and mechanized equipment in forest ecosystems.

In addition to modifying timber production strategies to minimize the adverse effects of anthropogenic fragmentation, managers should re-introduce fire as a natural process in high-elevation forests of the central Rocky Mountains. Some of the ecological effects of fire can never be entirely simulated by logging, even with variable retention techniques, and some native species like the black-backed woodpecker are almost completely dependent upon recently burned subalpine forests (Hutto 1995). Manager-ignited fire probably is not appropriate in most high-elevation forests, but many lightning-ignited fires can be allowed to burn without interference to provide unique early successional habitats that cannot be produced in any other way.

Aspen Forest Type

Composition and Distribution of Aspen Forests

The Aspen forest type within the SJPA is dominated by aspen trees. It occurs throughout the SJPA on mountain and plateau landscapes at elevations ranging from about 8000 to 11,200 feet. It is most abundant in the west end of the SJPA on the Mancos-Dolores Ranger District. It is associated with the subalpine and montane climate zones, the cryic and frigid soil temperature regimes, and the udic soil moisture regime. Mean annual precipitation ranges from about 22-35 inches. There are about 346,384 acres of this type within the SJPA.

This type occurs as relatively pure stands of aspen trees comprising **> =** 80 % of the canopy cover, or as a mix of aspen and conifer trees where aspen makes up the largest component within a stand (comprises a stocking plurality in association with other species). Conifer trees, displayed as understory or overstory components, include Engelmann spruce and subalpine fir at higher elevations, and white fir, Douglas-fir, blue spruce, and ponderosa pine at lower elevations. The combined canopy cover of trees is > =25%, or trees make up the most canopy cover of any life form (CVU, Eyre 1980).

Due to the variability in the composition and structure of the aspen forest type at the stand level, it can be separated into conifer and aspen phases, with the former having conifer species that display a combined canopy cover > = 20%, and the latter with conifer species absent or displaying a combined canopy cover < 20 %. There are about 182,592 acres of the conifer phase and 163,967 acres of the aspen phase within the SJPA.

This aspen forest type is associated with the Aspen (SAF #217) and IRI-CVU Aspen cover types. Potential natural vegetation (PNV) communities associated with the aspen forest type at the series level of classification are subalpine fir, white fir, Douglas-fir (DeVelice et al. 1986), and Aspen (Hess and Alexander 1986, Johnston and Hendzel 1985).

Most aspen forests within the SJPA are earlier seral stages of conifer PNV types (DeVelice et al. 1986). Stable aspen forests, which currently display little or no conifer species and are unlikely to succeed to conifer forests over the next 100 years or longer, are considered a PNV type as described for the aspen series. Stable aspen forests likely became established by frequent fire during the reference period, which eliminated the conifer seed source.

Some aspen stands on the west side of the SJPA (Dolores District) are adjacent to stands of the ponderosa pine forest type, and display ponderosa pine trees. Ponderosa pine forests succeeding from an early seral aspen stage is not the normal successional pathway for the ponderosa pine forest type, since ponderosa pine trees are shade-intolerant, but some sites appear to be evolving this way. In some cases, aspen trees have moved into valley floor positions that were previously occupied by narrowleaf cottonwood trees and willows (Redders observation), and appear to be expanding from valley floors into upland sites that display an open canopy of ponderosa pine trees.

The aspen forest type dominates some landscapes displaying large, relatively uniform, closed-canopy stands of mature aspen forest, or displaying smaller more diverse aspen stands that vary in size, shape, age, species composition, and canopy cover. Some landscapes display aspen stands intermixed with spruce-fir and mixed conifer forests, mountain grasslands, wetlands, and riparian areas. Diverse geology and mountain topography add to the complexity and diversity of these landscapes. Many aspen-dominated landscapes are unroaded and mostly undisturbed by human activities. Others are roaded and have been heavily disturbed by timber harvest and livestock grazing.

Common shrubs and herbs found in the aspen forest type include snowberry, thimbleberry, common juniper, Woods rose, mountain lover, sweet cicely, meadowrue, arnica, brackenfern, peavine, American vetch, bluebell, violet, yarrow, false Solomon's seal, forest fleabane, columbine, bluebells, rattlesnake plantain, avalanche lily, baneberry, dandelion, Richardson geranium, Gray lousewort, osha, mountain parsley, northern bedstraw, strawberry, fringed bromegrass, ryegrass, trisetum, mountain ricegrass, Kentucky bluegrass, and elk sedge.

Stand Structure and Habitat Structural Stages of Aspen Forests (Table 5-3)

The Sapling-Pole Stage (HSS 3a, 3b, 3c) of the aspen forest type displays a mostly even-aged, open to closed-canopy forest (> = 25% cover) dominated by small (< 5 inch), young (< 25 years) aspen trees. Structural diversity is low to moderate consisting of one canopy layer, few to many snags, and low to high amounts of large woody material on the forest floor. More snags and large woody material would be expected if the disturbance event that initiated this stage left abundant standing and down woody material. Tall shrubs display low to high cover. Herbs display high cover. Litter layer is thin to moderately thick.

The Mature Stage (HSS 4a, 4b, 4c) of the aspen forest type displays an even or uneven-aged, moderately closed to closed-canopy forest (> =40% cover) dominated by mid-sized (5-8 inch), mid-aged (25-80 years) aspen trees. Small (< = 5 inch) and young (< =25 years) aspen trees may be present. Small to mid-sized (< 9 inch) and young (< 50 years) conifer trees (Engelmann spruce, subalpine fir, Douglas-fir, white fir) may be present. Structural diversity is low to moderate consisting of one or two canopy layers, common snags, and low to moderate amounts of large woody material on the forest floor. Tall shrubs display low to high cover. Herbs display moderate to high cover. Litter layer is thick.

The Old Growth (HSS 5) of the aspen forest type displays an uneven-aged, closed-canopy forest (> = 70% cover) dominated by large (> = 9 inch), old (> = 80 years) aspen trees, many of which have dead or broken tops. Small and mid-sized (< 9 inch), and young and mid-aged (25-79 years) aspen trees may be present. Small, mid-sized, and large (< 16 inch), and young and middle-aged (< 150 years) conifer trees (Engelmann spruce, subalpine fir, Douglas-fir, white fir) may be present. Structural diversity is high consisting of multiple canopy layers, many snags, and high amounts of large woody material on the forest floor.

Tall shrubs display low to high cover. Herbs display moderate to high cover. Litter layer is thick.

Table 5-3 Current Acres of Aspen Forest by Habitat Structural Stage

HSS	HSS Description	Acres
3A	Sapling-pole ≤ 8” dbh, < 40% canopy closure	14,123
3B	Sapling-pole ≤ 8” dbh, 40-69% canopy closure	58,319
3C	Sapling-pole ≤ 8” dbh, ≥ 70% canopy closure	29,505
4A	Mature ≥ 9” dbh, < 40% canopy closure	23,368
4B	Mature ≥ 9” dbh, 40-69% canopy closure	142,800
4C	Mature ≥ 9” dbh, ≥ 70% canopy closure	52,615
5	Old Growth ≥ 14” dbh, ≥ 70% canopy closure	25,195

Historic Range of Variability (HRV) of Aspen Forests (McGarigal and Romme 2005)

The frequency and extent of simulated wildfires in the RMLANDS pure aspen forest type (it is assumed that the vegetation dynamics of the other aspen forests that contain conifers, as described by Redders, are similar to those of the pure aspen forest type) varied markedly among decades. In almost half of the decades, less than 5% of the aspen forest burned, inclusive of both high and low-mortality affected areas. However, on average, roughly 2-3 times per 100 years, > 10% of the area burned, and roughly once every 200 years, > 30% of the pure aspen forest burned. Under this wildfire regime, the return interval between fires (of any mortality level) varied widely from 26 years to > 800 years, with a mean and median of 110 years and 114 years, respectively, and no eligible area escaped disturbance altogether over the course of an 800-year simulation. As expected, return interval varied spatially across the forest. In general, return intervals increased with elevation, reflecting the moister, cooler conditions at higher elevations. In addition, pure aspen stands embedded in a neighborhood containing cover types with shorter return intervals (mountain shrubland) exhibited shorter return intervals, reflecting the importance of landscape context on fire regimes.

The age structure and dynamics of pure aspen forest reflected the interplay between disturbance and succession processes. The survivorship distribution represents the percent of stands that survived to any age, where age represents time since stand origin, not necessarily the age of the oldest trees in the stand. On average (over time), roughly 50% of the pure aspen forest was > 90 years since stand origin, although at any point in time this varied from < 27% to > 80%. On average, 10% of the pure aspen forest survived to > 300 years, 5% survived to > 360 years, and a very small percentage (< 1%) survived a stand-replacing disturbance for > 530 years. This highlights the stochastic nature of disturbances, in which some areas by chance alone escaped catastrophic disturbance for very long periods.

The distribution of area among stand conditions within pure aspen forest fluctuated markedly over time, as expected. For example, the percentage of pure aspen forest in the late-seral stages (understory reinitiation and shifting mosaic conditions) varied from 35% to 86%, reflecting the extremely dynamic nature of this cover type when considered over century-long periods. The seral-stage distribution appeared to be in dynamic equilibrium (the percentage in each stand condition varied about a stable mean), despite the fact that the proportion in each condition during any snapshot (time step) varied considerably over time. The spatial configuration of stand conditions fluctuated markedly over time as well, although there was considerable variation in the magnitude of variability among configuration metrics. Patch area, core area and the proximity index (a measure of patch isolation) exhibited the greatest variability over time.

Our estimate of the current seral-stage distribution was never observed under the simulated HRV. Specifically, the current landscape contains < 1% of this cover type in the stand initiation condition, a situation never observed under the simulated HRV, and 82% in the late-seral stages (understory reinitiation and shifting mosaic conditions), a situation almost never observed under the simulated HRV. The current landscape does, however, contain a pulse (17%) of pure aspen forest in the stem exclusion stage, which is well within the simulated HRV (46th percentile of the HRV distribution). Overall, based on the four separate stand conditions (i.e., without aggregating the late-seral stages), the seral-stage departure index was 67%. When the late-seral stages were aggregated, the seral-stage departure index declined slightly to 65%. The current seral-stage configuration deviated similarly (60%) from the simulated HRV, although there was considerable variability (16-100%) among metrics. In general, the current landscape contains fewer, larger and more clumped (less isolated) late-seral patches and fewer, smaller and geometrically less complex and less clumped (more isolated) early-seral patches of pure aspen forest than existed under the simulated HRV.

Management Implications for Aspen Forests (Romme et al. 2003)

The broad-scale geographic and elevational distribution of aspen during the period of indigenous settlement probably was similar to what we see today. Floristic composition probably also was generally similar. However, landscape-scale patterns of stand age, structure, and composition varied through time in response to disturbance and successional processes. There undoubtedly has been a decrease in aspen abundance within the SJPA during the latter half of the twentieth century because of fire exclusion. A decrease in aspen is potentially serious because of aspen's beneficial effects on biodiversity and aesthetics.

We can hypothesize that aspen was unusually abundant in the early twentieth century because of extensive fires in the previous century, and that the reduced abundance we see today is still within the historic range of variation. An alternative hypothesis is that aspen abundance was never as low in previous centuries as is it today. Unfortunately, we simply lack the necessary empirical data from the reference period that would enable us to confirm or reject either of these landscape-level hypotheses.

The two most important changes in aspen forests during the twentieth century are related to (i) the landscape mosaic of aspen age classes and (ii) the structure and composition of young aspen stands developing after disturbance. In some areas, the proportion of older aspen stands has increased during this century as a result of low fire frequency and relatively low levels of logging. Young stands in the 1880s were mostly of fire origin, and they contained abundant snags and fallen logs from the forest that had been burned. This "biological legacy" of the previous forest provided important habitat for birds and other fauna, and played a role in nutrient cycling and soil development. In the young stands developing after logging today, these biological legacies are generally lacking. Consequently, species that evolved to thrive in young aspen stands of fire origin (many cavity nesting species; Winternitz 1980, Scott et al. 1980, Hutto 1995) may be suffering for lack of quality habitat -- not only because of the paucity of recent fires, but also from the fact that young stands of logging origin are not adequate substitutes for postfire aspen stands in terms of key habitat characteristics.

Comparisons of burned vs logged aspen stands also have shown that fire stimulates increased herbaceous production and species diversity in the understory, but logging either does not enhance the herbaceous stratum or does so minimally (Anderson and Bailey 1980, Bartos and Mueggler 1982, Crouch 1983a, b). Fire also rapidly decomposes some of the litter and duff on the forest floor, provides a pulse of nutrients, and blackens the soil surface sufficiently to increase soil temperatures. The postfire increase in nutrients and soil temperatures is transient, lasting only a few years at most, but it may be important in stimulating herbaceous growth as well as aspen suckers. Logging really is not an ecological equivalent of fire. Logging resembles fire in that both kinds of disturbance kill the canopy and stimulate root sprouting, but logging is very different from fire in terms of post-disturbance environmental conditions, nutrient dynamics, and responses by animals and herbaceous plants.

In order to maintain aspen in the SJPA, it seems reasonable to increase prescribed burning and continue timber harvest in aspen forests of the SJPA. However, we offer two important caveats along with this recommendation. First, we note that the decrease in aspen abundance is occurring slowly, and so we should not regard aspen decline as a crisis situation in the SJPA. Aspen will not disappear from the canopy of most seral stands for several decades; though aspen is regarded as a short-lived species compared with some conifers. And even if the canopy trees die out completely, the root systems can persist for at least a few more decades during which time a disturbance may occur to stimulate new sprouting and stem regeneration (Peterson and Peterson 1992). Secondly, it must be emphasized that loss of aspen due to lack of disturbance may occur only in the seral stands, where conifers are slowly replacing the aspen. Stable aspen stands exhibit continued recruitment of new stems into the canopy and maintain an all-aged canopy structure even in the absence of intense stand-level disturbance. If we choose to embark on programs of logging and/or prescribed fire with the goal of restoring or maintaining our declining aspen forests, these efforts should be focused on seral aspen stands, not on the stable aspen communities.

Neither is there any strong ecological rationale for eliminating logging altogether from aspen forests of the SJPA. Aspen is a valuable timber species, and logging is important to the economy and social fabric of many communities in the region. Moreover, aspen is resilient to disturbances of many kinds, and it regenerates readily after clearcutting in particular. These ecological characteristics make aspen more suitable than some other forest types (spruce-fir forests) for sustainable timber production. There are a number of specific strategies that managers should consider for reducing the undesirable impacts of logging in aspen and other forest types in the region. For example, the size and shape of timber harvest patches can be modified to more closely resemble patches created by fires during the reference period. By creating fewer larger patches, anthropogenic fragmentation of the aspen landscape can be minimized. In addition, techniques for leaving a greater biological legacy (snags, coarse woody material) in harvested areas, as well as coupling harvest with prescribed fire, can provide young stands developing after timber harvest with some of the ecological structure and processes that were found in young post-fire stands but are conspicuously lacking in post-harvest stands today.

Status : Draft

Cool-Moist Mixed Conifer Forest Type

Composition and Distribution of Cool-Moist Mixed Conifer Forests

The Cool-Moist Mixed Conifer forest type within the SJPA is dominated by white fir and Douglas fir trees. It is associated with the montane and subalpine climate zones, the frigid and cryic soil temperature regimes, and the udic soil moisture regime. It occurs on mountain and plateau landscapes at elevations ranging from about 8500 to 10000 feet. Mean annual precipitation ranges from about 25-32 inches. The cool-moist mixed conifer forest type occurs on cooler and wetter sites, usually at higher elevations, compared to the warm-dry mixed conifer type. There are about 199,412 acres of this type within the SJPA.

Douglas fir and white fir trees are present in most stands. Engelmann spruce and subalpine fir occurs on the coolest and wettest sites of this type, while aspen may be found throughout. Blue spruce and limber pine may be present, usually as minor components (< 10% cover). Ponderosa pine and Gambel oak are absent or very minor components (< 10% cover). The combined canopy cover of trees is > = 25%, or trees make up the most canopy cover of any life form (CVU, Eyre 1980).

Due to the variability in the composition of the cool-moist mixed conifer forest type at the stand level, it can be separated into cold-wet and cool-moist phases, the former occurring on the coldest and wettest sites within the type (usually at the highest elevations) with Engelmann spruce and subalpine fir displaying a combined overstory canopy cover > = 15%, and the latter occurring on the warmest and driest sites within the type (usually at the lowest elevations) with Engelmann spruce and subalpine fir absent or minor components (< 15% combined overstory cover).

The cool-moist mixed conifer forest type is associated with the Interior Douglas-fir (SAF #210), and the IRI-CVU Douglas-fir and White Fir cover types. Potential natural vegetation communities associated with this type are White fir and Subalpine fir at the series level of classification, and White fir/maple, White fir/forest fleabane, Subalpine fir/blueberry, and Subalpine fir/forest fleabane at the plant association level (DeVelice et al. 1986).

Some cool-moist mixed conifer forests succeeded from aspen-dominated forests that were established following stand-replacing fires. Others formed when white fir and Douglas fir trees initially colonized a site following a disturbance event. Some may have succeeded from the warm-dry mixed conifer type, where the less shade-tolerant ponderosa pine component decreased as the more shade-tolerant Douglas fir and white fir components increased in abundance. This latter successional pathway would likely take centuries to complete due to the persistence of the long-lived ponderosa pine component. Some cool-moist mixed conifer forests were created when timber harvest in the warm-dry mixed conifer type removed the ponderosa pine component.

The cool-moist mixed conifer forest type is usually intermixed with spruce-fir, warm-dry mixed conifer, and aspen forests, mountain grasslands, and riparian areas. It usually does not display large contiguous stands that dominate landscapes. Diverse geology and mountain topography add to the complexity and diversity of these landscapes. Many landscapes that contain the cool-moist mixed conifer type are roaded and have been extensively harvested.

Common shrubs and herbs found in the cool-moist mixed conifer type include snowberry, maple, current, serviceberry, Woods rose, blueberry, thimbleberry, common juniper, elderberry, raspberry, buffaloberry, honeysuckle, mountain ash, mountain lover, violet, Richardson geranium, strawberry, forest fleabane, fairy slipper, baneberry, clematis, rattlesnake plantain, false Solomon's seal, columbine, pipsissewa, meadowrue, osha, northern bedstraw, avalanche lily, pussytoes, wood nymph, wintergreen, elk sedge, ryegrass, mountain ricegrass, and fringed bromegrass.

Stand Structure and Habitat Structural Stages for the cool-moist and warm-dry mixed conifer forests (Table 5-4)

The Sapling-Pole Stage (HSS 3a and 3b) of the mixed conifer forest type displays an even or uneven-aged, open to moderately closed-canopy forest (25-69% cover) dominated by small to mid-sized (< 9 inch), young (< 50 years) trees (white fir and Douglas-fir for the cool-moist phase, and ponderosa pine, white fir and Douglas-fir for the warm-dry phase). Small (< = 5 inch) and young (< = 25 years) aspen trees may be present. Structural diversity is low to moderate consisting of one or two canopy layers, few to many snags, and low to high amounts of large woody material on the forest floor. More snags and large woody material would be expected if the disturbance event that initiated this stage left abundant standing and down woody material. Tall shrubs and herbs display moderate to high cover. Litter layer is thin to moderately thick.

The Mature Stage (HSS 4a, 4b, 4c) of the mixed conifer forest type displays an uneven-aged, moderately closed to closed-canopy forest (> = 40% cover) dominated by large (9-15 inch), mid-aged (50-150 years) conifer trees (white fir, Douglas-fir, Engelmann spruce, subalpine fir for the cool-moist phase, and ponderosa pine, white fir, Douglas-fir for the warm-dry phase), some of which have dead or broken tops. Small to mid-sized (< 9 inch) and young (< 50 years) conifer trees are present. Mid-sized to large (> = 5 inch), mid-aged to old (> = 25 years) aspen trees may be present for the cool-moist phase. Structural diversity is high consisting of multiple canopy layers, common to many snags, and moderate to high amounts of large woody material on the forest floor. More snags and large woody material would be expected if the stand succeeded from an aspen forest. Tall shrubs display low to high cover for the warm-dry phase, and low to moderate cover for the cool-moist phase. Herbs display low to moderate cover. Litter layer is thick for the cool-moist type and moderate for the warm-dry type.

The Old Growth Stage (HSS 5) of the mixed conifer forest type displays an uneven-aged, closed-canopy forest (> = 70% cover) dominated by very large (> = 16 inch), old (> = 150 years) conifer trees (white fir, Douglas-fir, Engelmann spruce, subalpine fir for the cool-moist phase, and ponderosa pine, white fir, Douglas-fir for the warm-dry phase), many of which have dead or broken tops. Small, mid-sized, and large (< 16 inch), and young and middle-aged (< 150 years) conifer trees are present. Large (> = 9 inch) and old (> = 80 years) aspen trees may be present in the cool-moist phase. Structural diversity is high consisting of multiple canopy layers, many snags, and high amounts of large woody material on the forest floor. Tall shrubs display low to high cover for the warm-dry type, and low to moderate cover for the cool-moist type. Herbs display low to moderate cover. Litter layer is thick for the cool-moist type and moderate for the warm-dry type.

Table 5-4 Current Acres of Cool-Moist Mixed Conifer Forest by HSS

HSS	HSS Description	Acres
3A	Sapling-pole (3-8” dbh), < 40% canopy closure	4,582
3B	Sapling-pole (3-8”) dbh, 40-69% canopy closure	11,432
3C	Sapling-pole (3-8” dbh), ≥ 70% canopy closure	4,385
4A	Mature (9-15” dbh), < 40% canopy closure	15,261
4B	Mature (9-15” dbh), 40-69% canopy closure	94,688
4C	Mature (9-15” dbh), ≥ 70% canopy closure	45,172
5	Old Growth (≥ 16” dbh), ≥ 70% canopy closure	23,740

Disturbance History of Cool-Moist Mixed Conifer Forests

With respect to disturbance history and landscape-scale dynamics, the cool-moist mixed conifer forests probably are the least well understood of all of the forest types in the SJPA. One reason is their inherent compositional variability and complexity. Another is the fact that there have been no recent fires in this forest type to provide us with direct information on fire effects and biotic responses.

The cool-moist mixed conifer forests have a very different stand structure and disturbance history than the warm-dry mixed conifer forests. Cool-moist mixed conifer forests often grow in areas that are too wet for the frequent fires that characterize the lower-elevation forest types. Late-lying snowpacks and more frequent summer rains keep fuels moist throughout most of the fire season in most years. Consequently, stands regularly may persist for many decades or centuries without fire. When fires finally do occur during prolonged dry periods, the fuel bed that has developed during the long fire-free period tends to support a high-intensity fire that kills most of the aboveground vegetation in the stand.

Thus, in the cool-moist mixed conifer zone we see a mosaic of stands that have developed following lethal fires at various times in the past. However, it is important to recognize that low-severity fires also occasionally occurred in the high elevation stands.

In contrast with the striking impacts of Euro-American settlement in lower elevation forests, the fire regimes of cool-moist mixed conifer and other high elevation forests, were not so strongly affected by early grazing and fire suppression. Fire frequency and behavior in high elevation forests were controlled primarily by weather conditions not fuels, and most fires that covered large areas were severe, stand-replacing fires. Thus, removal of fine herbaceous fuels by livestock grazing did not have as much impact on fire spread in these crown-fire ecosystems as it did in lower elevation forests. There have been few large fires during the twentieth century in cool-moist mixed conifer and other high elevation forests of the SJPA, but this may be due as much to wet weather conditions and limited ignitions as to active fire suppression efforts. However, fire exclusion at lower elevations (ponderosa pine forests) may partially explain the lack of fires at higher elevations during the last century (in previous centuries fires may have commonly ignited at lower elevations and then spread into higher elevations when weather conditions were suitable).

Historic Range of Variability (HRV) of Cool-Moist Mixed Conifer Forests (McGarigal and Romme 2005)

The two most important and ubiquitous kinds of disturbance in cool-moist mixed conifer forests during the period of indigenous settlement were stand-destroying fire and insect outbreaks. Wildfires tend to be high-mortality, stand-replacing fires that initiate a process of post-fire forest succession. High-mortality fires kill large as well as small trees, and may kill many of the shrubs and herbs as well, although below-ground organs of at least some individual shrubs and herbs survive and re-sprout.

Cool-moist mixed conifer forest is subject to three different insect disturbance process in RMLANDS: Douglas-fir beetle, spruce beetle, and spruce budworm. Douglas-fir beetle and spruce beetle kill Douglas-fir and Engelmann spruce trees, respectively, especially in the larger size classes (> 9 inches dbh). Western spruce budworm affects

The frequency and extent of simulated wildfires in cool-moist mixed conifer forests (includes RMLANDS cool-moist mixed conifer and cool-moist mixed conifer-aspen forest types) varied markedly among decades. In about one third of the decades, less than 1% of the cool-moist mixed conifer forest burned, inclusive of both high and low-mortality affected areas. However, on average, 2-3 times per 100 years, > 10% of the area burned, and roughly once every 150-200 years, > 20% of the cool-moist mixed conifer forest burned. Under this wildfire regime, the return interval between fires (of any mortality level) varied widely from 26 years to > 800 years, with a mean and median of 144 (136 for cool-moist mixed conifer-aspen) and 160 years, respectively, although very little eligible area (< 1%) escaped disturbance altogether over the course of an 800-year simulation. As expected, return intervals varied spatially across the forest. In general, return intervals increased with elevation, reflecting the moister, cooler conditions at higher elevations. In addition, cool-moist mixed conifer stands embedded in a neighborhood containing cover types with shorter return intervals (aspen) exhibited shorter return intervals, reflecting the importance of landscape context on fire regimes.

The frequency and extent of simulated spruce beetle epidemics in cool-moist mixed conifer forest varied markedly among decades. In about one third of the decades, less than 1% of the cool-moist mixed conifer forest was disturbed, inclusive of both high and low-mortality affected areas. However, on average, roughly once per 100 years, > 10% of the area was disturbed, and at least once every 400 years, > 20% of the cool-moist mixed conifer forest was disturbed. The return interval between epidemics (of any mortality level) at a single location varied widely from 53 years to > 800 years, with a mean and median of 370 (392 for cool-moist mixed conifer-aspen) and 400 years, respectively, although > 20% of the eligible area escaped disturbance altogether over the course of an 800-year simulation. Not surprisingly, the return interval varied spatially across the forest and exhibited a somewhat contagious or clumped distribution. In general, areas of extensive and connected cool-moist mixed conifer forest at the highest elevations where it is juxtaposed to spruce-fir forest (the most susceptible cover type) exhibited the shortest return intervals, but there was considerable variation among runs owing to the stochastic nature of spruce beetle outbreaks.

The frequency and extent of simulated spruce budworm epidemics in cool-moist mixed conifer forest varied markedly among decades. In almost two-thirds of the decades, spruce budworm populations were at *endemic levels and none of the cool-moist mixed conifer forest was disturbed. However, roughly 3 times per 100 years, epidemics occurred affecting > 10% of the area, inclusive of both high- and low-mortality affected areas, and approximately once every 150 years, > 50% of the cool-moist mixed conifer forest was disturbed. The return interval between epidemics (of any mortality level) at a single location varied from 33 years to > 800 years, with a mean and median of roughly 90 years, although almost no eligible area (< 1%) escaped disturbance altogether over the course of an 800-year simulation. The return interval varied spatially across the forest, but at a very fine grain and in a seemingly random pattern.*

The frequency and extent of simulated Douglas-fir beetle epidemics in cool-moist mixed conifer forest varied markedly among decades. In most decades < 1% of the cool-moist mixed conifer forest area was affected by an outbreak; however, at least once every 400 years an epidemic affecting > 2% of the host area would occur. The return interval between epidemics (of any mortality level) at a single location varied from 114 years to > 800 years, with a mean and median of > 800 years; in fact most of the area (> 70%) escaped disturbance altogether over the course of an 800-year simulation. The return interval varied spatially across the forest, but at a very fine grain and in a seemingly random pattern.

The age structure and dynamics of cool-moist mixed conifer forest reflected the interplay between disturbance and succession processes. The survivorship distribution represents the percent of stands that survived to any age, where age represents time since stand origin, not necessarily the age of the oldest trees in the stand. On average (over time), roughly 50% of the cool-moist mixed conifer forest was > 150 years since stand origin, although at any point in time this varied from < 33% to > 69%. On average, 28% (23% for cool-moist mixed conifer-aspen) of the cool-moist mixed conifer forest survived to > 300 years, 5% survived to

> 530 years, and a very small percentage (< 1%) survived a stand-replacing disturbance for > 800 years. This highlights the stochastic nature of disturbances, in which some areas by chance alone escaped catastrophic disturbance for very long periods.

The distribution of area among stand conditions within cool-moist mixed conifer forest fluctuated over time, as expected. For example, the percentage of cool-moist mixed conifer forest in the late-seral stages (i.e., understory reinitiation and shifting mosaic conditions) varied from 38% to 85%, reflecting the extremely dynamic nature of this high-elevation forest cover type when considered over century-long periods. The seral-stage distribution appeared to be in dynamic equilibrium (i.e., the percentage in each stand condition varied about a stable mean), despite the fact that the proportion in each condition during any snapshot (i.e., time step) varied considerably over time. The spatial configuration of stand conditions fluctuated markedly over time as well, although there was considerable variation in the magnitude of variability among configuration metrics. Patch area, core area and the proximity index (a measure of patch isolation) exhibited the greatest variability over time.

For the RMLANDS cool-moist mixed conifer type, our estimate of the current seral-stage distribution was never observed under the simulated HRV. The most notable departure was in the early (i.e., stand initiation) and late-seral stages (understory reinitiation and shifting mosaic conditions). The current landscape contains only 2% of the cool-moist mixed conifer forest in the stand initiation condition, yet this condition was always better represented (3-36%) under the simulated HRV. Conversely, the current landscape contains 72% of the cover type in the late-seral stages, yet this high a percentage was the upper extreme of the range observed under the simulated HRV. The current landscape does, however, contain a pulse (26%) of cool-moist mixed conifer forest in the stem exclusion stage, which is well within the simulated HRV (15^{th} percentile of the HRV distribution), perhaps representing a legacy of naturally occurring wildfires during the 19th century.

For the RMLANDS cool-moist mixed conifer-aspen type, our estimate of the current seral-stage distribution was never observed under the simulated HRV. The most notable departure was in the late-seral stages (understory reinitiation and shifting mosaic conditions). Specifically, the current landscape contains 88% of this cover type in the late-seral stages, yet this situation was never observed under the simulated HRV. In addition, the current landscape contains only 4% of this cover type in the stand initiation condition, an uncommon situation under the simulated HRV (9th percentile of the HRV distribution), and 8% in the stem exclusion condition, a situation almost never observed under the simulated HRV. Notably, the current landscape contains much less of this cover type in the early- and mid-seral stages (i.e., stand initiation and stem exclusion conditions) than in cool-moist mixed conifer forest (12% versus 28%). This is likely due to differences between cover types in the rate of succession from the stem exclusion stage to the understory reinitiation stage. In the mixed conifer-aspen forest, the stem exclusion stage is dominated by aspen, which transitions to the understory reinitiation stage between 80-120 years after stand origin - owing to the relatively short-lived aspen and early break-up of the aspen-dominated canopy. In contrast, the conifer canopy in the pure conifer forest takes 150-250 years to break up in the absence of other major disturbance processes. Hence, in the absence of large wildfires over the past century, most mixed conifer-aspen stands have already transitioned to the understory reinitiation stage, whereas a larger proportion of the pure conifer stands have not yet transitioned.

Overall, based on the four separate stand conditions (without aggregating the late-seral stages), the seral-stage departure index was 85% (91% for cool-moist mixed conifer-aspen). When the late-seral stages were aggregated, the seral-stage departure index declined slightly. The current seral-stage configuration deviated 87% (66% for cool-moist mixed conifer-aspen) from the simulated HRV. Unfortunately, the nature of the configuration departure varied depending on whether the late-seral stages were treated separately or combined into a single class. Thus, the only reliable conclusion we can reach is that, in general, the current landscape contains fewer, smaller and geometrically less complex and more isolated patches of early-seral forest than existed under the simulated HRV.

Pseudotsuga menziesii and Abies lasiocarpa trees of all sizes, often weakening trees and making them more susceptible to beetle attack. Because cool-moist mixed conifer stands are assumed to contain a mixture of host species, no insect alone is sufficient to cause a stand-replacing event. Rather, an epidemic of one insect will simply shift the tree species composition of the forest in favor of the non-host species. Two or three of the insects working in concert, however, can result in near or complete overstory mortality and therefore invoke stand-replacement (although this is very uncommon).

Management Implications for Cool-Moist Mixed Conifer Forests (Romme et al. 2003)

Cool-moist mixed conifer stands that have not been logged in the past generally do not need any kind of restoration treatment, even though such stands may be dense and may contain abundant white fir and other shade-tolerant species. Because many old-growth cool-moist mixed conifer forests in the SJPA have not been extensively disturbed and because they harbor unique species that require a dense canopy and abundant coarse woody material (Romme et al. 1992), old stands with no logging history generally should be protected from anthropogenic disturbance.

Other cool-moist mixed conifer stands that have been altered by past timber harvests that selectively removed the large trees (Douglas-fir) and left a dense stand of more shade-tolerant white fir trees may benefit from silvicultural treatments designed to restore lost components of structure and composition. Moreover, because understory conditions are now unsuitable for regeneration of less shade-tolerant species like Douglas fir, the stands are not likely to return to a natural structure and composition for many centuries, even if they are protected from further human disturbance. However, just how to effect restoration of these kinds of stands is not well understood, and all restoration treatments should be regarded as experimental.

Warm-Dry Mixed Conifer Forest Type

Composition and Distribution of Warm-Dry Mixed Conifer Forests

The Warm-Dry Mixed Conifer forest type within the SJPA is dominated by ponderosa pine, white fir and Douglas fir trees. It occurs on mountain and plateau landscapes at elevations ranging from about 7500 to 9000 feet. It is associated with the montane climate zone, the frigid soil temperature regime, and the ustic soil moisture regime. Mean annual precipitation ranges from about 20-25 inches. The warm-dry mixed conifer forest type occurs on warmer and drier sites, usually at lower elevations, compared to the cool-moist mixed conifer type, and often occurs on sites with southerly aspects. The warmer, drier condition accounts in part for the more open-canopy structure, and the more frequent fire disturbance regime of this type, as compared to the other mixed conifer type. There are about 95,392 acres of this type within the SJPA.

Douglas fir and white fir trees (combined), or ponderosa pine trees make up the largest component within a stand (comprise a plurality in association with other species). All three of these species are present in most stands, but white fir or Douglas fir may be absent on some sites. Ponderosa pine is the key indicator species of this type, displaying a canopy cover >= 10%. Gambel oak is a significant component in these forests, occurring in most stands. Blue spruce, southwestern white pine, aspen, pinyon pine, and Rocky Mountain juniper may be present, usually as minor components (< 10% cover). Subalpine fir and

Engelmann spruce are absent. The combined canopy cover of trees is > = 25%, or trees make up the most canopy cover of any life form (CVU, Eyre 1980).

The warm-dry mixed conifer forest type is associated with the Interior Douglas-fir (SAF #210) and Interior Ponderosa Pine (SAF #237) cover types, and the IRI-CVU Douglas-fir, White Fir, and Ponderosa Pine cover types. Potential natural vegetation communities associated with the warm-dry mixed conifer type at the series level of classification are White fir and Douglas-fir, and at the plant association level are White fir/Gambel oak and Douglas-fir/Gambel oak (DeVelice et al. 1986).

Some warm-dry mixed conifer forests succeeded from Gambel oak-dominated shrublands that were established following stand-replacing fires. Some formed when white fir, Douglas fir, and ponderosa pine trees initially colonized a site following a disturbance event. Others may have succeeded from the ponderosa pine type due to succession and fire suppression, which increased the abundance of white fir and Douglas fir. Warm-dry mixed conifer forests are likely to maintain their ponderosa pine component over time due to natural disturbances (fire, insects, and mistletoe) and environmental factors (microclimate, slope, aspect, soil type) that perpetuate an open canopy and give the ponderosa pine component a competitive advantage to persist and regenerate.

At a landscape scale, the warm-dry mixed conifer forest type is usually intermixed with ponderosa pine, cool-moist mixed conifer, and aspen forests, mountain grasslands, mountain shrublands, wetlands, and riparian areas. On the SJNF, it usually does not display large contiguous stands that dominate landscapes. Diverse geology and mountain topography add to the complexity and diversity of these landscapes. Most landscapes that contain the warm-dry mixed conifer type are roaded and have been extensively harvested.

Common shrubs and herbs found in the warm-dry mixed conifer type include Gambel oak, snowberry, serviceberry, Oregon grape, Woods rose, kinnikinnik, geranium, peavine, strawberry, clematis, meadowrue, pussytoes, goldenrod, mountain parsley, daisy, lupine, American vetch, skyrocket, yarrow, Arizona fescue, elk sedge, bottlebrush squirreltail, junegrass, Kentucky bluegrass, and muttongrass.

Stand Structure and Habitat Structural Stages of Warm-Dry Mixed Conifer Forests (Table 5-5)

See cool-moist mixed conifer forest type for description of the Sapling-Pole Stage (HSS 3a and 3b), the Mature Stage (HSS 4a, 4b, 4c), and the Old Growth Stage (HSS 5) of the warm-dry mixed conifer forest type.

Table 5-5 Current Acres of Warm-Dry Mixed Conifer Forest by Habitat Structural Stage

HSS	HSS Description	Acres
3A	Sapling-pole (3-8" dbh), < 40% canopy closure	984
3B	Sapling-pole (3-8") dbh, 40-69% canopy closure	5,234
3C	Sapling-pole (3-8" dbh), ≥ 70% canopy closure	1,861
4A	Mature (9-15" dbh), < 40% canopy closure	4763
4B	Mature (9-15" dbh), 40-69% canopy closure	55,851
4C	Mature (9-15" dbh), ≥ 70% canopy closure	17,345
5	Old Growth (≥ 16" dbh), ≥ 70% canopy closure	9,354

Disturbance History of Warm-Dry Mixed Conifer Forests

The major Euro-American impacts on warm-dry mixed conifer forests are similar to impacts in ponderosa pine forests. Because warm-dry mixed conifer forests often occur on moderate terrain with good access, they are more roaded than cool-moist mixed conifer forests.

The introduction of large herds of cattle and sheep, beginning in the 1870s, led to profound changes in the herbaceous component of warm-dry mixed conifer forests (Fleischner 1994). Livestock grazing also removed fine herbaceous fuels that formerly carried low-intensity surface fires over extensive areas, and resulted in a relatively sudden and dramatic drop in fire frequency in the late nineteenth century (Swetnam and Baisan 1996). Livestock use on the National Forests has been regulated since the early 1930s, but many areas at lower elevations (including ponderosa pine and warm-dry mixed conifer forests) still show the effects of excessive grazing at the turn of the twentieth century. These effects include altered herbaceous communities and unnaturally long fire intervals. The Forest Service also actively suppressed fires throughout most of the twentieth century. As a result of heavy grazing and active fire control, many mixed conifer forest stands that formerly burned every 5 - 40 years, now have had no fire for over 100 years.

The combination of fire exclusion, livestock grazing, selective logging that removed the large trees, and favorable climatic conditions for young tree establishment in the early twentieth century, has created an unusual stand structure in many warm-dry mixed conifer forests today. The large, old ponderosa pine and Douglas fir trees that formerly dominated the canopy are now gone, and the stands are dominated by smaller, young individuals of ponderosa pine, Douglas-fir, and white fir. White fir especially has increased in density during the long fire-free period of the twentieth century, and establishment of new ponderosa pine and Douglas fir individuals has tapered off or stopped in many stands because of the dense stand conditions.

Historic Range of Variability (HRV) of Warm-Dry Mixed Conifer Forests (McGarigal and Romme 2005)

The two most important and ubiquitous kinds of disturbance in high-elevation landscapes during the period of indigenous settlement were stand-destroying fire and insect outbreaks. Wildfires are common and frequent with mortality depending on vegetation vulnerability and wildfire intensity. Low-severity fires kill small trees and consume above-ground portions of shrubs and herbs, but do not kill large trees or below-ground organs of most shrubs and herbs which promptly re-sprout. High-severity fires kill large as well as small trees, and may kill many of the shrubs and herbs as well. Fire kills the above-ground portions of the shrubs and herbs, but most shrubs and herbs promptly re-sprout from surviving below-ground organs.

Warm-dry mixed conifer forest is subject to four different insect disturbance process in RMLANDS: Pine beetle, Douglas-fir beetle, spruce beetle, and spruce budworm. Spruce beetle outbreaks are of minor concern in this cover type due to the paucity of spruce trees, so while they are allowed to occur, they have no affect on transitions. Pine beetle and Douglas-fir beetle kill

The frequency and extent of simulated wildfires in warm-dry mixed conifer forests (includes RMLANDS warm-dry mixed conifer and warm-dry mixed conifer-aspen forest types) varied markedly among decades. Wildfire was quite prevalent in this cover type. In most decades, > 10% of the warm-dry mixed conifer forest burned, inclusive of both high and low-mortality affected areas, and roughly 1-2 times per 100 years, > 30% of the area burned. Under this wildfire regime, the return interval between fires (of any mortality level) varied widely from 24 years to > 800 years, with a mean and median of 53 and 50 years (62 years for warm-dry mixed conifer-aspen), and almost no eligible area escaped disturbance altogether over the course of an 800-year simulation. As expected, return intervals varied spatially across the forest. In general, return intervals increased with elevation, reflecting the moister, cooler conditions at higher elevations. In addition, warm-dry mixed conifer stands embedded in a neighborhood containing cover types with longer return intervals (aspen, cool-moist mixed conifer forest) exhibited longer return intervals, reflecting the importance of landscape context on fire regimes.

Spruce beetles can venture into warm-dry mixed conifer forest, but due to the scarce and patchy distribution of suitable host trees in this cover type, epidemics are relatively insignificant and have negligible impact on this cover type. Only occasionally (once every 400-800 years) did spruce beetles disturb > 1% of the warm-dry mixed conifer forest, inclusive of both high- and low-mortality affected areas. Consequently, return intervals between epidemics (of any mortality level) at a single location typically exceeded 800 years.

The frequency and extent of simulated spruce budworm epidemics in warm-dry mixed conifer forest varied markedly among decades. In almost two-thirds of the decades, spruce budworm populations were at endemic levels and none of the warm-dry mixed conifer forest was disturbed. However, 2-3 times per 100 years, epidemics occurred affecting > 10% of the area, inclusive of both high- and low-mortality affected areas, and approximately once every 200 years, > 30% of the warm-dry mixed conifer forest was disturbed. The return interval between epidemics (of any mortality level) at a single location varied from 36 years to > 800 years, with a mean and median of 133 years, although very little eligible area (~1-2%) escaped disturbance altogether over the course of an 800-year simulation. The return interval varied spatially across the forest, but at a very fine grain and in a seemingly random pattern.

The frequency and extent of simulated Douglas-fir beetle epidemics in warm-dry mixed conifer forest varied markedly among decades. In most decades < 1% of the warm-dry mixed conifer forest area was affected by an outbreak; however, 2-3 times per 100 years an epidemic affecting > 2% of the host area would occur, and once every 300 years an epidemic would affect > 6% of the host area, inclusive of both high and low-mortality affected areas. The return interval between epidemics (of any mortality level) at a single location varied from 80 years to > 800 years, with a mean and median of 695 (751 for warm-dry mixed conifer-aspen) and 800 years, respectively, although a large portion of the area (> 30%) escaped disturbance altogether over the course of an 800-year simulation. The return interval varied spatially across the forest, but at a very fine grain and in a seemingly random pattern.

The frequency and extent of simulated pine beetle epidemics in warm-dry mixed conifer forest varied markedly among decades. In most decades, pine beetles were at endemic levels and none of the warm-dry mixed conifer forest was disturbed. However, 1-3 times per 100 years an epidemic occurred. On average, once every 50 years an epidemic affecting > 5% of the host area would occur, and roughly once every 400 years a major epidemic affecting > 15% of the host area would occur, inclusive of both high- and low-mortality affected areas. The return interval between epidemics (of any mortality level) at a single location varied from 62 years to > 800 years, with a mean and median of 395 (425 for warm-dry mixed conifer-aspen) and 400 years, respectively, although > 10% of the eligible area escaped disturbance altogether over the course of an 800-year simulation. The return interval varied spatially across the forest, but at a very fine grain and in a seemingly random pattern.

The age structure and dynamics of warm-dry mixed conifer forest reflected the interplay between disturbance and succession processes. The survivorship distribution represents the percent of stands that survived to any age, where age represents time since stand origin, not necessarily the age of the oldest trees in the stand. On average (over time), roughly 50% of the warm-dry mixed conifer forest was > 350 years since stand origin, although at any point in time this varied from < 9% to > 61%. On average, 25% of the warm-dry mixed conifer forest survived to > 550 years, and roughly 8% survived a stand-replacing disturbance for > 800 years. The relatively "old" age structure of this cover type may seem surprising at first; however, most wildfires in this cover type were low-mortality fires that did not result in stand reinitiation.

Pinus ponderosa and *Pseudotsuga menziesii* trees, respectively, especially in the larger size classes (> ca. 8 inches dbh). Western spruce budworm affects Pseudotsuga menziesii and Abies concolor trees of all sizes, often weakening trees and making them more susceptible to beetle attack. Because warm-dry mixed conifer stands are assumed to contain a mixture of host species, no insect alone is sufficient to cause a stand-replacing event. Rather, an epidemic of one insect will simply shift the tree species composition of the forest in favor of the non-host species. Two or three of the insects working in concert, however, can result in near or complete overstory mortality and therefore invoke stand-replacement (although this is very uncommon).

The distribution of area among stand conditions within warm-dry mixed conifer forest fluctuated over time, as expected. For example, the percentage of warm-dry mixed conifer forest in the late-seral stages (understory reinitiation and shifting mosaic conditions) varied from 21% to 62%, reflecting the dynamic nature of this cover type when considered over century-long periods. However, the fluctuations in the seral-stage distribution were much less pronounced in this cover type than in most others, with the exception of the ponderosa pine forest types. The seral-stage distribution appeared to be in dynamic equilibrium (the percentage in each stand condition varied about a stable mean), and appeared to reach equilibrium relatively quickly compared to other cover types. The spatial configuration of stand conditions fluctuated markedly over time as well, although there was considerable variation in the magnitude of variability among configuration metrics. Patch area and extent (radius of gyration) and the proximity index (a measure of patch isolation) exhibited the greatest variability and the stand initiation and understory reinitiation conditions were particularly dynamic relative to the other stages.

Our estimate of the current seral-stage distribution was never observed under the simulated HRV. The most notable departure was in the fire-maintained open canopy (FMO) condition. The current landscape contains no warm-dry mixed conifer forest in the FMO condition, yet this condition was always well represented (24-64%) under the simulated HRV.

For the RMLANDS warm-dry mixed conifer type, the current landscape contains 27% of the cover type in the stem exclusion condition and 69% in the late-seral stages (understory reinitiation and shifting mosaic conditions), yet these high percentages were never observed under the simulated HRV. The only stand condition not deviating from the simulated HRV was the stand initiation condition.

For the RMLANDS warm-dry mixed conifer-aspen type, the current landscape contains 89% of the cover type in the late-seral stages (understory reinitiation and shifting mosaic conditions), yet this high a percentage was never observed under the simulated HRV. There was a notable difference in seral-stage departure between warm-dry mixed conifer forest with and without aspen. The current landscape contains much less mixed conifer with aspen in the stem exclusion stage than in the pure conifer cover type (7% versus 27%). This is likely due to differences between cover types in the rate of succession from the stem exclusion stage to the understory reinitiation stage. In the mixed conifer with aspen type, the stem exclusion stage is dominated by aspen, which transitions to the understory reinitiation stage between 80-120 years after stand origin - owing to the relatively short-lived aspen and early break-up of the aspen-dominated canopy. In contrast, the conifer canopy in the pure conifer forest takes 150-250 years to break up in the absence of other major disturbance processes. Hence, in the absence of wildfires over the past century, most mixed conifer-aspen stands have already transitioned to the understory reinitiation stage, whereas a larger proportion of the pure conifer stands have not yet transitioned. The only stand condition not deviating from the simulated HRV was the stand initiation condition.

Overall, based on the five separate stand conditions (i.e., without aggregating the late-seral stages), the seral-stage departure index was 80% (77% for warm-dry mixed conifer-aspen). When the late-seral stages were aggregated, the seral-stage departure index declined slightly to 75% (71% for warm-dry mixed conifer-aspen). The current seral-stage configuration deviated more dramatically (95%) from the simulated HRV and was relatively consistent among metrics, with the lowest departure index equal to 66%. Despite the difficulties in classifying the late-seral stages in the current landscape, there is consistent evidence that, in general, the current landscape contains fewer, larger and more geometrically complex and clumped (less isolated) patches of all seral stages than existed under the simulated HRV.

Management Implications for Warm-Dry Mixed Conifer Forests (Romme et al. 2003)

In warm-dry mixed conifer forests that have already been subjected to removal of the canopy dominants in previous logging operations, and to fire exclusion throughout the twentieth century, managers may wish to emphasize restoration of the ecological components and processes that have been lost. Specific guidelines for restoration, by means of selective timber harvest and prescribed fire, that have been developed and partially tested in ponderosa pine forests in southwestern Colorado and elsewhere can be reasonably applied to many warm-dry mixed conifer forests. However, restoration efforts should still be regarded as experimental and should be approached in a framework of adaptive management.

Selective harvest focuses on removal of small trees and retention of any large canopy pine and Douglas fir that may still remain. In this way, the effects of the timber harvest resemble the effects of a natural, low-severity fire with respect to tree mortality among the different size classes. Selective harvest is followed by prescribed fire, to reduce the fuel loads created by slash, and to consume some of the litter and duff on the forest floor. In many cases, a second prescribed fire will be needed within a few years to reduce litter and duff below what they were before the harvest. However, even a single fire may stimulate renewed growth and diversity in the suppressed herbaceous layer (Romme et al. 1999). In restoration treatments of this kind, it is not necessary to remove all of the white fir, aspen, and oak from the stand, because these species have always been natural components of warm-dry mixed conifer forests. In fact, large fir, aspen, and oak generally should be retained, because they provide valuable structural features for the stand. Even if large stems are retained but subsequently killed in the prescribed fire, they become snags, which are a critical structural component that is missing in many lower-elevation forests of the SJPA.

Ponderosa Pine Forest Type

Composition and Distribution of Ponderosa Pine Forests

The Ponderosa Pine forest type within the SJPA is dominated by ponderosa pine trees. It occurs on mountains, hills, and mesas at elevations ranging from about 7000 to 8500 feet. It is associated with the lower montane climate zone, the frigid soil temperature regime, and the ustic soil moisture regime. Mean annual precipitation ranges from about 16-25 inches. There are about 411,790 acres of this type within the SJPA. Ponderosa pine forests are most abundant on the Dolores Ranger District.

This type occurs as relatively pure stands of ponderosa pine trees comprising > = 80% of the canopy cover. Rocky Mountain juniper, white fir, Douglas fir, southwestern white pine, blue spruce, aspen, and pinyon pine trees may be present usually as minor components (< 10% canopy cover). Gambel oak is a significant component in these forests, occurring in most stands, but it is a minor component in the ponderosa pine/Arizona fescue plant association on the mesas in the Turkey Springs area west of Pagosa Springs. There is a significant amount of variability in the composition and abundance of understory species, and in the size, age, and density of live and dead trees at the stand level. The combined canopy cover of trees is > = 25%, or trees make up the most canopy cover of any life form (CVU, Eyre 1980).

This ponderosa pine forest type is associated with the Interior Ponderosa Pine (SAF #237) and the IRI-CVU Ponderosa Pine cover types. Potential natural vegetation communities associated with the ponderosa pine forest type at the series level of classification are Ponderosa pine, and Ponderosa pine/Gambel oak and Ponderosa pine/Arizona fescue at the plant association level (DeVelice et al. 1986).

Some ponderosa pine forests succeeded from Gambel oak-dominated shrublands that were established following stand-replacing fires. Others formed when ponderosa pine trees initially colonized a site following a disturbance event.

The ponderosa pine forest type dominates some landscapes displaying large, relatively uniform stands of ponderosa pine forest, such as occur on the plateaus on the west side of the SJNF. Some landscapes display ponderosa pine stands intermixed with pinyon-juniper woodlands, warm-dry mixed conifer forests, aspen forests, mountain grasslands, mountain shrublands, and riparian areas. Diverse geology and mountain topography add to the complexity and diversity of these landscapes. Most ponderosa pine-dominated landscapes are roaded, and have been heavily disturbed by timber harvest and livestock grazing.

Common understory species shrubs found in the ponderosa pine forest type include Oregon grape, buckbrush, Gambel oak, bitterbrush, skunkbush, snowberry, mountain mahogany, serviceberry, mountain parsley, lupine, butterweed, pussytoes, American vetch, peavine, strawberry, cinquefoil, goldenrod, beardstongue, geranium, paintbrush, puccoon, deervetch, buckwheat, skyrocket, pasqueflower, daisy, yarrow, Kentucky bluegrass, bottlebrush squirreltail, junegrass, mountain muhly, Parry oatgrass, Arizona fescue, pine dropseed, little bluestem, muttongrass, blue grama, and elk sedge.

Stand Structure and Habitat Structural Stages of Ponderosa Pine Forests (Table 5-6)

The Sapling-Pole Stage (HSS 3a and 3b) of the ponderosa pine forest type displays a mostly even-aged, open to moderately closed-canopy forest (25-69% cover) dominated by small to mid-sized (< 9 inch), young (< 50 years) ponderosa pine trees. Structural diversity is low to moderate consisting of one or two canopy layers, few snags, and low amounts of large woody material on the forest floor. More snags and large woody material could be expected if the disturbance event that initiated this stage left an abundance of standing and down woody material. Tall shrubs and herbs display moderate to high cover. Litter layer (mostly needles) is thin following fire.

The Mature Stage (HSS 4a, 4b, 4c) of the ponderosa pine forest type displays an uneven-aged, moderately closed to closed-canopy forest (> = 40% cover) dominated by large (9-15 inch), mid-aged (50-149 years) ponderosa pine trees, some of which have dead or broken tops. Small to mid-sized (< 9 inch) and young (< 50 years) pine trees are present. Structural diversity is moderate consisting of one or multiple canopy layers, few to common snags, and low to moderate amounts of large woody material on the forest floor. Less snags and large woody material would be expected if firewood gatherers easily access the stand. Tall shrubs (including Gambel oak) display low to high cover. Herbs display low to moderate cover. Litter layer (mostly needles) is thick in many places, particularly under the large trees.

The Old Growth (HSS 5) of the ponderosa pine forest type displays an uneven-aged, moderately closed to closed-canopy forest (> = 40% cover) dominated by very large (> = 16 inch), old (> = 150 years) ponderosa pine trees, many of which have dead or broken tops. Small, mid-sized, and large (< 16 inch), and young and middle-aged (< 150 years) ponderosa pine trees are present. Structural diversity is high consisting of multiple canopy layers, common to many snags, and moderate amounts of large woody material on the forest floor. Less snags and large woody material would be expected if firewood gatherers easily access the stand. Tall shrubs (including Gambel oak) display low to high cover. Herbs display relatively low cover. Litter layer (mostly needles) is thick in many places, particularly under the large trees.

Table 5-6 Current Acres of Ponderosa Pine Forest by Habitat Structural Stage

HSS	HSS Description	Acres
3A	Sapling-pole (3-8" dbh), < 40% canopy closure	4,511
3B	Sapling-pole (3-8") dbh, 40-69% canopy closure	4,828
3C	Sapling-pole (3-8" dbh), ≥ 70% canopy closure	310
4A	Mature (9-15" dbh), < 40% canopy closure	109,161
4B	Mature (9-15" dbh), 40-69% canopy closure	104,539
4C	Mature (9-15" dbh), ≥ 70% canopy closure	2,131
5	Old Growth (≥ 16" dbh), ≥ 70% canopy closure	15,047

Many or even most of the ponderosa pine forest stands present today in the SJPA are dominated by relatively small, young trees. Eighty percent of stems greater than breast height are 12 inches or smaller in diameter; 95% are 16 inches or smaller. Very large trees (> 24 inches dbh) comprise less than 0.5% of the population. The current forests have a relatively uniform stand structure: they tend to be relatively dense, most of the trees are small to medium sized, and most are 70 - 100 years. The strong clumping pattern of the forests during the reference period also has been largely lost in today's dense stands. Trees that established after 1900 have filled in many of the former open spaces between clumps, and the trees are thicker in the areas of the former clumps as well.

Disturbance History of Ponderosa Pine Forests

Many of the ponderosa pine forests of the SJPA have been dramatically altered during the last century. The major agents of change have been logging, livestock grazing, and fire exclusion introduced by Euro-American settlers in the late 1800s, although climatic variability also has been involved.

It appears that the logging operations of the late 1800s and the first half of the twentieth century affected nearly all of the accessible ponderosa pine forests. This early logging usually involved "high-grading" (selectively removing the highest quality trees; generally the large, old-growth trees) and leaving the smaller trees or species of lesser monetary value. Consequently, unlogged, old growth ponderosa pine stands today are very few in number and nearly all are in locations where it was physically impossible or uneconomic to log them. The most obvious effect of the early logging of ponderosa pine forests is the general lack of large, old trees and snags in these forests today.

Heavy livestock grazing, which occurred in the late 1800s and early 1900s, has left a long-lasting legacy of altered plant community composition in ponderosa pine forests in the SJPA. Some highly palatable species, such as mountain muhly and Arizona fescue, have been locally extirpated from many stands, and the relative abundance of surviving species has been drastically altered. Grazing, along with fire exclusion and some climatic events led to a shift from dominance or co-dominance of herbs in the ground layer of ponderosa pine forests to an overwhelming dominance of woody vegetation (shrubs and tree saplings) in many areas. We may never know the pre-grazing composition of many ponderosa pine forests, though we can make some reasonable guesses. One more key legacy of early, heavy livestock grazing is the virtual elimination of fire as a dominant ecological process in ponderosa pine forests of the southwest, as described below.

Fire suppression by the Forest Service was a factor in the abrupt cessation of formerly frequent and extensive fires in ponderosa pine forests of the Southwest, but organized and effective fire control programs were not instituted over large areas until well into the twentieth century (Pyne 1982). The fire regimes in the Southwest was the introduction of large herds of livestock by Euro-American settlers. Throughout most of the Southwest, the cessation of frequent, extensive fires coincided with the arrival of large herds of cattle and sheep in the late 1800s (Swetnam and Baisan 1996, Covington and Moore 1994). The animals grazed off the grasses and herbs that formerly carried light fires through the forest. Lightning and humans still started fires, but they could not spread across bare ground. Grazing began to be regulated more effectively on federal lands beginning in the 1930s but by this time the Forest Service was developing an effective fire suppression capability. We now know that by suppressing all fires we simply created the potential for even more severe fires in the future (Covington and Moore 1994).

Small fires continued well into the twentieth century in many areas, indicating that lightning and humans were still igniting fires. The major change, however, was that these fires were nearly all very small. The extensive fires of the previous centuries were no longer a part of the landscape. Romme documented a few wildfires in the twentieth century in the San Juan National Forest, as well as a prescribed fire within the last decade at the Hermosa site. None of these fires were extensive, and the intervals between even these small fires have been significantly longer than the intervals between extensive fires during the reference period.

Due to the changes described above, the following changes have been noticed in ponderosa pine stands. A large portion of ponderosa pine stands are experiencing increasing shade-tolerant species, such as white fir. The number of small-diameter ponderosa pine trees (less than 9-inch DBH) in mid and late-successional ponderosa pine forests has also increased significantly. Additionally, shrub densities (primarily Gambel oak) in some locations have increased probably a result, in part, of past human activities such as timber harvest that opened up the stands. Collectively, these species composition changes are making stands more susceptible to insects, disease, and unnaturally intense wildfire.

Historic Range of Variability (HRV) of Ponderosa Pine Forests (McGarigal and Romme 2005)

The two most important and ubiquitous kinds of disturbance in high-elevation landscapes during the period of indigenous settlement were stand-destroying fire and insect outbreaks (Baker and Veblen 1990, Veblen et al. 1994, Veblen, *in press).* Wildfires are common and frequent, with mortality depending on vegetation vulnerability and wildfire intensity. Low-severity fires kill small trees and consume above-ground portions of shrubs and herbs, but do not kill large trees or below-ground organs of most shrubs and herbs which promptly re-sprout. High-severity fires kill large as well as small trees, and may kill many of the shrubs and herbs as well. Fire kills the above-ground portions of the shrubs and herbs, but most shrubs and herbs promptly re-sprout from surviving below-ground organs.

The frequency and extent of simulated wildfires in the ponderosa pine forest type (includes RMLANDS ponderosa pine-oak and ponderosa pine-oak-aspen forest types) varied markedly among decades. Wildfire was quite prevalent in this cover type. In most decades, > 10% of the ponderosa pine forest burned, inclusive of both high- and low-mortality affected areas, and roughly 2-4 times per 100 years, > 30% of the area burned. Under this wildfire regime, the return interval between fires (of any mortality level) varied widely from 19 years to > 800 years, with a mean and median of 38 years for the ponderosa pine-oak type (46 for ponderosa pine-oak-aspen type), and almost no eligible area escaped disturbance altogether over the course of an 800-year simulation. Note, the return interval between low-mortality fires as measured by the sample-based approach (in which each recorded interval between low-mortality fires in a cell was treated as an independent observation, in order to approximate the method of most dendrochronological fire history studies) was somewhat shorter (30 years for ponderosa pine-oak, 43 for ponderosa pine-oak-aspen). As expected, return intervals varied spatially across the forest. In general, return intervals increased with elevation, reflecting the moister, cooler conditions at higher elevations. In addition, ponderosa pine stands embedded in a neighborhood containing cover types with longer return intervals (e.g., aspen, cool-moist mixed conifer forest) exhibited longer return intervals, reflecting the importance of landscape context on fire regimes.

The frequency and extent of simulated pine beetle epidemics in ponderosa pine forests varied markedly among decades. Pine beetles kills ponderosa pine trees, especially in the larger size classes (> ca. 8 inches dbh). In most decades, pine beetles were *at endemic levels and none of the ponderosa pine forest was disturbed. However, 1-3 times per 100 years an epidemic occurred. On average, once every 70 years an epidemic affecting > 10% of the host area would occur, and roughly once every 300 years a major epidemic affecting > 20% of the host area would occur, inclusive of both high- and low-mortality affected areas. The return interval between epidemics (of any mortality level) at a single location varied from 50 years to > 800 years, with a mean and median of 251 (230 for ponderosa pine-oak-aspen) and 267 years, respectively, although ~5% of the eligible area escaped disturbance altogether over the course of an 800-year simulation. The return interval varied spatially across the forest, but at a very fine grain and in a seemingly random pattern.*

The age structure and dynamics of ponderosa pine-oak forest reflected the interplay between disturbance and succession processes. The survivorship distribution represents the percent of stands that survived to any age, where age represents time since stand origin, not necessarily the age of the oldest trees in the stand. On average (over time), roughly 50% of the ponderosa pine-oak forest was > 380 years since stand origin, although at any point in time this varied from 2% to 66%. On average, 25% of the ponderosa pine-oak forest survived to > 580 years, and roughly 8% survived a stand-replacing disturbance for > 800 years. The relatively "old" age structure of this cover type may seem surprising at first; however, most wildfires in this cover type were low-mortality fires that did not result in stand reinitiation.

The distribution of area among stand conditions within ponderosa pine forests fluctuated over time, as expected. For example, the percentage of ponderosa pine-oak forest in the late-seral stages (i.e., understory reinitiation and shifting mosaic conditions) varied from 8% to 42% (15% to 50% for ponderosa pine-oak-aspen), reflecting the dynamic nature of this cover type when considered over century-long periods. The seral-stage distribution appeared to be in dynamic equilibrium (i.e., the percentage in each stand condition varied about a stable mean), and appeared to reach equilibrium relatively quickly compared to other cover types. The spatial configuration of stand conditions fluctuated markedly over time as well, although there was considerable variation in the magnitude of variability among configuration metrics. Patch area and extent (radius of gyration) and the proximity index (a measure of patch isolation) exhibited the greatest variability and the understory reinitiation condition was particularly dynamic relative to the earlier and later stages.

Our estimate of the current seral-stage distribution was never observed under the simulated HRV. The most notable departure was in the fire-maintained open canopy (FMO) condition. The current landscape contains no ponderosa pine forests in the FMO condition, yet this condition was always well represented (39-77%) under the simulated HRV. Similarly, for ponderosa pine-oak the current landscape contains 67% of the cover type in the stem exclusion condition, yet this large proportion was never observed under the simulated HRV, and for ponderosa pine-oak-aspen the current landscape contains 88% of the cover type in the late-seral stages (understory reinitiation and shifting mosaic conditions), yet this high a percentage was never observed under the simulated HRV.

Overall, based on the five separate stand conditions (i.e., without aggregating the late-seral stages), the seral-stage departure index was 80% for ponderosa pine-oak (91% for ponderosa pine-oak-aspen). The current seral-stage configuration for all ponderosa pine forests deviated more dramatically (94%) from the simulated HRV and was relatively consistent among metrics, with the lowest departure index equal to 75%. Despite the difficulties in classifying the late-seral stages in the current landscape, there is consistent evidence that, in general, the current landscape contains fewer, larger and more geometrically complex and clumped (less isolated) patches of ponderosa pine forests of all seral stages than existed under the simulated HRV.

Management Implications for Ponderosa Pine Forests (Romme et al. 2003)

The ponderosa pine forests within the SJPA today are substantially different from the forests that existed during the reference period. The fire regime of ponderosa pine forests in the SJPA currently exists outside the historical range of variability for fire frequency and extent (Romme 2003, Grissino-mayer et al. 2004). The changes have come about primarily through human activities during the last century -- notably logging, grazing, and fire exclusion -- although some key climatic events also have played a role.

When fires occur today in ponderosa pine forests, they sometimes are much hotter and more destructive than they were before 1880, because of the abnormal fuel conditions associated with great quantities of dead pine needles, branches, and other heavy fuels that have accumulated during 100+ years without fire (Covington and Moore 1994). There is a greater probability of the fire burning the canopy and inflicting significant tree damage and mortality than was usual in pre-1870 fires. Another consequence of the high density and small tree size of many ponderosa pine stands is that the forests are vulnerable to outbreaks of insects and disease.

If past management approaches are continued into the future, it is unlikely that conditions in ponderosa pine forests will improve, since these past methods are in part responsible for the present state of the forest. Similarly, if no action is taken, conditions probably will not improve for a long time. The dominant trees are all near the midpoint of their life spans and will not experience much natural thinning or mortality for several decades, and the deep organic litter layers and fuels will decompose only very slowly. With no action, or a continuation of current management, there is a real possibility that an uncontrollable fire or bark beetle outbreak will produce a disturbance event far outside the prehistoric range of natural variability, thereby introducing additional undesirable ecological changes.

There are opportunities now for new, innovative approaches to silviculture and ecological restoration in ponderosa pine forests. These approaches are based on mimicking the kinds of natural disturbances that shaped these forests for hundreds of years.

To restore a more open canopy containing greater diversity of tree sizes (including large trees), we can employ selective harvesting to remove predominantly smaller trees from a stand that is overly dense and homogeneous. Begin by identifying and marking clumps of the larger trees in the stand that resemble the clumps that existed in the forest prior to high-grade logging. Stumps of the pre-1900 trees, if present, may give some clues to the former clumping patterns, or it may be necessary to rely on general familiarity with the pre-1900 forest structure. After marking the clumps, the loggers then remove most of the smaller trees plus a few larger trees from the spaces between clumps, and leave the trees of all sizes within the clumps.

Selective logging is followed by prescribed fire, to reduce slash and litter, kill the aboveground portions of the shrubs, expose mineral soil for pine seedling establishment, and stimulate sprouting and growth of suppressed herbaceous plants. Once the stand has been thinned and the fuel loads reduced in this way, it will be necessary to continue prescribed burning into the indefinite future. The average intervals between fires, the variability in fire intervals, and the seasonality of burning, should mimic the pre-1870 fire regime of the area. Maintaining a natural range of variability in fire intervals, intensities, and seasons is critical. In most stands, a single prescribed fire will not be sufficient to reduce abnormal fuel loads and litter layers or to re-invigorate the long suppressed herbaceous community. In conjunction with the prescribed burning program, it is necessary to protect recently treated stands from excessive livestock grazing. Both domestic and native ungulates will be attracted by the increased quantity and quality of forage in thinned and burned forests, and potentially may wipe out the gains in the herbaceous vegetation that have been obtained through treatment.

There are numerous potential pitfalls associated with a prescribed burning program of the magnitude called for above. For one thing, prescribed fire has the potential to actually hasten the demise of the few old-growth ponderosa pines that still remain in dense stands. Prescribed fire also may cause significant nutrient loss, especially of nitrogen. The immediate, short-term effect of burning usually is an increase in available soil nitrogen (Kovacic et al. 1986, Monleon et al. 1997). However, over the next several years or decades, there may be changes in mineralization rate, quality of soil organic matter, and soil microbial activity, that result in reduced amounts of available nitrogen in the soil (Klemmedson 1976, Vance and Henderson 1984, Monleon et al. 1997). Loss of nitrogen, due to volatilization during the fire and then reduced substrate quality and quantity for several years after the fire, may lead to long-term decreases in site productivity and tree growth.

Another potential and undesirable effect of prescribed burning relates to alien weeds. Fire stimulates prolific sprouting of rhizomatous species such as Canada thistle, and creates mineral seedbeds where widely dispersed, highly invasive weeds like musk thistle can become established. In a recent prescribed fire in the San Juan National Forest (Hidden Valley area) and in the Mesa Verde wildfire of 1996, we observed that burning in locations where Canada thistle was already present resulted in dense stands of thistle that appeared to exclude most of the desirable native herbs. How long such a dense patch of thistle will persist is unknown, but it may dominate a local area for decades. A combination of thinning the canopy and burning the forest floor will create conditions that are extremely conducive to invasion by new populations of alien weeds and to increases in populations already present. Therefore, prior to prescribed burning, a thorough weed survey should be conducted and local populations of rhizomatous weeds controlled by chemical or other means.

Despite these potential pitfalls, the prescribed fire program is a critical component of the overall restoration effort. Logging alone will not reduce fuel loads, create mineral soil and reduce shrub competition for new pine seedlings, or stimulate the growth of suppressed herbaceous plants.

Restoration of an open stand structure should be applied only to some, not all, ponderosa pine forests throughout the SJPA. During the reference period, there were some dense stands of small ponderosa pine trees, similar to the stands that dominate the landscape today (Shinneman and Baker 1997). These dense stands provide excellent habitat for many highly desirable species (elk, deer, and turkey), and so they should not be eliminated altogether. The main problem to be addressed through restoration is not that the dense, homogeneous stands we have today are undesirable in all respects, but that an excessive proportion of the landscape appears to be covered by stands of this kind -- a far higher proportion than occurred during the reference period.

Restoration efforts should be concentrated, at least initially, in areas that already have extensive road systems and a long history of logging. The few remaining roadless areas and tracts of unlogged ponderosa pine forest are not "pristine" -- they have the same legacies of early grazing and fire exclusion that characterize nearly all ponderosa pine forests in the region, and they may benefit in some ways from programs of thinning and prescribed fire. However, by virtue of their lack of roads and lack of logging history, they still retain certain characteristics of the reference period forests -- characteristics that are now gone from more heavily disturbed stands (complete age structures in the tree populations and absence or very low abundance of alien plant species).

Pinyon-Juniper Woodland Type

Composition, Distribution, and Stand Structure of Pinyon-Juniper Woodlands

The Pinyon-Juniper woodland type within the SJPA is dominated by pinyon pine and Utah juniper trees. It occurs on mountains, hills, and mesas at elevations ranging from about 6000 to 7500 feet. It is associated with the semi-arid climate zone, the mesic soil temperature regime, and the ustic soil moisture regime. Mean annual precipitation ranges from about 13-18 inches. It is woodland because most trees are shorter than 20 feet and their crowns rarely touch (Eyre 1980). There are about 444,147 acres of this type within the SJPA.

Pinyon pine and Utah juniper trees, combined, make up the largest component within a stand (comprise a plurality in association with other species). They both are present in most stands, the former usually more abundant on cooler, moister sites, the latter more abundant on warmer, drier sites. Rocky mountain juniper, ponderosa pine, or Douglas fir may be present in some stands, usually as minor components (< 10% canopy cover). Shrub cover tends to be high, with Gambel oak or big sagebrush common in many stands. Stands vary from closed-canopy with dense trees and shrubs and little herbaceous vegetation, to open-canopy with widely scattered trees and shrubs, and low to moderate herbaceous cover. The combined canopy cover of trees is > = 25%, or trees make up the most canopy cover of any life form (CVU, Eyre 1980).

This pinyon-juniper woodland type is associated with the Pinyon-Juniper (SAF #239) and the IRI-CVU Pinyon-Juniper cover types. Potential natural community types associated with the pinyon-juniper type include Pinyon pine at the series level, and Pinyon pine/Gambel oak and Pinyon pine/big sagebrush at the plant association level (Stuever and Hayden 1997, Moir and Carleton 1986).

Some pinyon-juniper woodlands succeeded from Gambel oak-dominated shrublands that were established following stand-replacing fires (Daubenmire 1966, Floyd et al. 1998). Others formed when pinyon pine and Utah juniper trees initially colonized a site following a disturbance event.

The pinyon-juniper woodland type is usually intermixed with ponderosa pine forests, warm-dry mixed conifer forests, mountain shrublands, sagebrush shrublands, and riparian areas. On the SJNF, it usually does not display large contiguous stands that dominate landscapes. Most landscapes that contain the pinyon-juniper type are roaded and have been heavily disturbed by livestock grazing.

Terrestrial cryptogams composed of cyanobacteria, algae, lichen, moss, fungi, or liverwort are often significant components in pinyon-juniper woodlands forming crusts on the ground surface that contribute to soil stability, nutrient supply (organic matter and nitrogen), and biodiversity (Ladyman and Muldavin 1996).

Shrubs associated with this type include Gambel oak, big sagebrush, bitterbrush, buckbrush, skunkbush, mountain mahogany, serviceberry, snakeweed, Oregon grape, prickly pear, and banana yucca. Common herbs found in this type include butterweed, pussytoes, cinquefoil, goldenrod, sand aster, beardstongue, geranium, paintbrush, puccoon, deervetch, buckwheat, redroot buckwheat, lupine, lousewort, skyrocket, daisy, sagewort, goldeneye, yarrow, hairy goldenaster, Kentucky bluegrass, bottlebrush squirreltail, needle-and-thread, junegrass, pine dropseed, little bluestem, blue grama, muttongrass, side-oats grama, western wheatgrass, spike muhly, sand dropseed, and Indian ricegrass.

Historic Range of Variability (HRV) of Pinyon-Juniper Woodlands (McGarigal and Romme 2005)

The major agents of natural disturbance in pinyon-juniper woodlands included fire, insects, and fungal diseases. Bark beetles (primarily *Dendroctonus ponderosae and Ips confusus)* and the black stain root fungus *(Ophiostoma wagereri)* often interacted to produce a phenomenon of "pinon decline." The effects of fire, beetles, and fungus were manifest at multiple scales, from large burns or regional insect outbreaks to small fires or localized outbreaks that created canopy gaps within otherwise continuous woodland.

The Pre-1900 Fire Regime in the Pinyon-juniper shrub woodland type consisted of infrequent, high-severity, stand-replacing crown fires carried by shrubs & trees and only occurring under conditions of extreme drought and wind (Romme 2003). Prior to Euro-American settlement in the mid-1800s, trees would become established during the fire-free intervals that lasted from several years to a few decades, but the next fire would kill nearly all of the trees. Fire exclusion during the last century has allowed this normal successional process to proceed to the point of tree dominance across large areas where trees were formerly sparse. Thus, this kind of vegetation may escape fire for many centuries, and develop striking old-growth characteristics, including a dense, multi-storied canopy with ancient living and dead trees. These characteristics are documented in the old forests of Mesa Verde (Floyd et al. 2004), where some individual stands have not burned since abandonment of the area 700 years ago (Floyd 2000). Pinyon pine and Utah juniper are very fire-sensitive and are easily killed even by low-intensity burns (Jameson 1966, Floyd 1982 and 1986).

Bark beetles (primarily *Dendroctonus ponderosae and Ips confusus)* and the black stain root fungus *(Ophiostoma wagereri)* attack only pinyon trees, which may be one reason why junipers often live longer than pinyon. The two most important bark beetles in pinyon trees of the SJPA are the pine engraver (Ips pini) and the Pinyon Ips (Ips confusus). Bark beetle outbreaks commonly occur during drought conditions, when water stress reduces the host trees' ability to resist the beetle attack. Outbreaks also occur following any disturbance that creates large quantities dead and down Pinyon trees like windthrow or mechanical thinning operations (Furniss and Carolin 1977). Indeed, a major bark beetle outbreak is now underway in southwestern Colorado and elsewhere in the southwest, fueled in large part by the severe regional drought of the last few years. In contrast, black stain root fungus produces the greatest Pinyon mortality following years of well-distributed, heavy summer rain (J. Worrall, personal communication).

The frequency and extent of simulated wildfires in pinyon-juniper woodland (includes RMLANDS pinyon-juniper woodland, pinyon-juniper-sagebrush woodland, and pinyon-juniper-oak-serviceberry woodland types) varied markedly among decades. In most decades,

> 5% of the pinyon-juniper woodland burned, inclusive of both high- and low-mortality affected areas, and roughly once per 100 years, > 20% of the area burned.Under this wildfire regime, the return interval between fires (of any mortality level) varied widely from 23 years to > 800 years, with a mean and median of 100 years (123 and 133 years for pinyon-juniper-sagebrush woodland, 108 and 114 years for pinyon-juniper-oak-serviceberry woodland), and roughly 2% of the area escaped disturbance altogether over the course of an 800-year simulation.

The frequency and extent of simulated pinyon decline epidemics in pinyon-juniper woodland varied among decades in an episodic fashion. In most decades, pinyon decline was at endemic levels and < 1% of the pinyon-juniper woodland was disturbed. However, roughly once every 200 years an epidemic affecting > 15% of the host area would occur, and roughly once every 300-400 years a major epidemic affecting > 30% of the host area would occur, inclusive of both high- and low-mortality affected areas. The return interval between epidemics (of any mortality level) at a single location varied from 80 years to > 800 years, with a mean and median of 400 years (337 and 400 years for pinyon-juniper-sagebrush, 378 and 400 years for pinyon-juniper-oak-serviceberry), although ~9-15% of the eligible area escaped disturbance altogether over the course of an 800-year simulation.

The age structure and dynamics of pinyon-juniper woodland reflected the interplay between disturbance and succession processes. The survivorship distribution represents the percent of stands that survived to any age, where age represents time since stand origin, not necessarily the age of the oldest trees in the stand. On average (over time), roughly 50% of the pinyon-juniper-sagebrush woodland was > 70 years since stand origin, although at any point in time this varied from 22% to 87%. On average, 25% of the pinyon-juniper woodland survived to > 160 years, and < 1% survived a stand-replacing disturbance for > 800 years. Note, the relatively "young" age distribution of this cover type (compared to expectations for this cover type) was due to its landscape position in the project area, where it exists as small disjunct patches at the higher range of its elevational distribution and is juxtaposed to ponderosa pine forests and mountain shrublands - which burned frequently in the simulations. Lower elevation, more extensive pinyon-juniper-sagebrush woodlands just outside the project boundary burned less frequently in the simulations and facilitated the development of an older age structure.

The distribution of area among stand conditions within pinyon-juniper woodland fluctuated markedly over time, as expected. For example, the percentage of pinyon-juniper woodland in the tree-dominated stage varied from 1% to 50%, reflecting the dynamic nature of this cover type when considered over century-long periods. However, given the scarcity of this cover type in the project area, it was not surprising that the range of variation was so wide. The seral-stage distribution appeared to be in dynamic equilibrium (i.e., the percentage in each stand condition varied about a stable mean), and appeared to reach equilibrium relatively quickly compared to other cover types. The spatial configuration of stand conditions fluctuated markedly over time as well, although there was considerable variation in the magnitude of variability among configuration metrics. The herb-dominated condition was particularly dynamic, with coefficients of variation typically two to several times greater than the later seral stages.

Our estimate of the current seral-stage distribution was infrequently observed under the simulated HRV. In particular, the current landscape contains much less area in the shrub-tree stage and much more in the tree-dominated stage than was observed under the simulated HRV, which, in combination with the other stand conditions, resulted in an overall seral-stage departure index of 62% (50% for pinyon-juniper-sagebrush, 24% for pinyon-juniper-oak-serviceberry). The current seral-stage configuration deviated 61-66% from the simulated HRV, although the magnitude of departure varied considerably among metrics. In general, the current landscape contains fewer, larger and more clumped (less isolated) patches of the same seral stage than existed under the simulated HRV. **WARNING,** due to the scarcity of this cover type within the project area and the lack of reliable field data on current stand age and condition in this cover type, these findings must be viewed with extreme caution. Until more complete data are obtained, we are unable to reach firm conclusions regarding HRV departure for this cover type.

Management Implications for Pinyon-Juniper Woodlands (Romme et al. 2003)

The impacts of 20th century land use are greater in Pinyon-juniper woodland vegetation than in many of the higher-elevation forest types, in large part because they occur at lower elevations in proximity to towns and other developments. However, the ecological effects of grazing, fire suppression, and other land uses vary greatly throughout the Pinyon-juniper zone. Therefore, it is important not to apply a "one-size-fits-all" approach to managing or restoring Pinyon-juniper woodland vegetation in the SJPA.

The need for better knowledge about prehistoric fire regimes in Pinyon–Juniper woodlands is becoming increasingly urgent, as land managers begin to implement ambitious policies intended to mitigate wildland fire hazards across the western US by means of fuel reduction (Healthy Forests, 2002).

To help identify priorities for fuels treatments, public lands managers are classifying major vegetation types into “condition classes” which reflect the nature of the prehistoric fire regime and the degree to which that fire regime has been altered during the last century (Schmidt et al., 2002). These classifications typically regard Pinyon– Juniper vegetation as having burned frequently in the past and as having “missed” several fires that would have occurred in the last 100 years had fire suppression not occurred (e.g. Schmidt et al., 2002). The occurrence of several large, severe fires during the last decade in Pinyon–Juniper vegetation often is taken as further evidence that these ecosystems have become degraded and are in immediate need of intensive treatment (usually involving mechanical tree removal and/or prescribed low-severity fire) to restore them to a more natural condition. However, without an adequate understanding of the historical range of variability of Pinyon–Juniper woodland vegetation, it is impossible to assess the current state of these ecosystems or to determine whether recent fire sizes and behaviors have been normal or abnormal. Indeed, a systematic review of the literature (Baker and Shinneman, 2004) uncovered no empirical support for any sweeping generalization that Pinyon Juniper vegetation now lays outside the historical range of variability. On the contrary, infrequent severe fires appear to be the norm for many Pinyon–Juniper systems. Therefore, the current aggressive effort to “restore” Pinyon–Juniper woodlands may be misguided or even damaging to ecological integrity (Landres et al., 1999; Swetnam et al., 1999; Cole, 2000; Romme et al., 2003; Baker and Shinneman, 2004).

The Pinyon-juniper woodland type probably has not been substantially altered by fire exclusion in the last century. Thus, these forests generally should not be subjected to extensive mechanical thinning or prescribed burning, although fuel reduction may be appropriate in localized areas to protect human lives, property, or other sensitive resources. Such localized fire mitigation treatments should be called just that – they should not be called "restoration" and they should acknowledge that relatively rare and ecologically significant old-growth characteristics may be sacrificed to protect other resources and values.

In places, there is an urgent need to restore fire as an ecosystem process in Pinyon-juniper vegetation. In some places, ecological restoration may not be the goal. For example, fuel reduction and wildfire mitigation may be called for in the vicinity of homes and other developments, regardless of whether such actions actually "restore" the vegetation or further alter it from historical conditions. However, the discussion below focuses on opportunities and challenges where the primary goal is restoration of key structural and functional conditions that existed before EuroAmerican settlement.

The most urgent restoration need generally is in the Pinyon-juniper woodland.type with a high shrub component. This is the type in which fire was most frequent during the reference period, and the type that has changed the most since EuroAmerican settlement. Fuel loads have become very high and continuous, and some recent fires probably have been larger and more severe than would have occurred before the late 1800s (Miller and Tausch 2001). Fire hazard mitigation via mechanical thinning and prescribed burning (some of it designed to be stand-replacing) can be linked to a broader goal of ecological restoration and aggressive treatment of this kind is an urgent need in many places. Restoration tools might include mechanical tree removal, with an emphasis on the young trees that have accumulated since the onset of fire exclusion, and prescribed fire, ignited either by lightning or by managers. In contrast, most of the Pinyon-juniper woodland type probably requires no active restoration of fire or stand structure, though they may need treatments to mitigate invasive species or excessive grazing.

There may be little that managers can do to reduce the threat of future wildfires in many parts of the SJPA where Pinyon-juniper woodland is a dominant vegetation type. In Mesa Verde National Park, for example, despite a policy of complete fire suppression since 1906, the total area that burned within the park from 1951 – 2000 was equal to or greater than what burned from 1851 – 1900 when there was no attempt at fire control (Floyd et al. 2000). Pinyon-juniper and mountain shrubland communities burn relatively infrequently, but they burn ferociously under certain weather conditions. Even with modern fire fighting technologies, it appears that, in these vegetation types, we mainly put out fires that would have been relatively small anyway. One thing that managers can do is be very judicious in their use of prescribed fire. Prescribed fires, ignited by managers under low-severity weather conditions, have become an important tool for reducing fuel loads and restoring desirable ecological conditions in other vegetation types, notably ponderosa pine forests and grasslands. However, prescribed burning in Pinyon-juniper forests of the Colorado Plateau probably should be avoided – because of the slow regeneration of forests, and also because of the risk of invasion by non-native species (Romme et al. 2003, Floyd *in press).*

At this time, we have no compelling evidence that fire frequency or severity actually have increased significantly in southwestern Colorado. Large, severe fires did occur in and around Mesa Verde National Park in 1996, 2000, and 2002, and burned more total area in the park than had burned in the previous half century. However, similar large fires occurred during the second half of the nineteenth century (Floyd et al. 2000, Floyd in press). Thus, the fire regime in Pinyon-juniper woodlands of southwestern Colorado does not yet appear to be outside the historic range of variability in fire frequency, extent, and severity that characterized previous centuries. However, the fire activity since about 1980 probably has been near the upper limit of that historic range of variability.

After severe fire, Pinyons and junipers can be very slow to re-establish (Erdman 1970). They do not re-sprout, so seeds must be transported by birds and mammals into the burned area and buried in suitable growing locations. The young seedlings are vulnerable to spring drought, winter freezing and thawing, and herbivory by birds and small mammals. A new Pinyon or juniper tree takes many decades to grow to maturity, and a stand of Pinyon-juniper woodland requires centuries to develop the old-growth structural characteristics

What is often missing in plans for restoration of Pinyon-juniper vegetation is recognition of the fact that throughout the West there are also ancient Pinyon-juniper stands with trees > 400 years old. These stands were already well developed even before the late 1800s, and should not be regarded as abnormal consequences of grazing, fire exclusion, and climate change. Unfortunately, thinning and burning programs that would be appropriate and effective in Pinyon-juniper stands that have developed abnormal tree density during the last century, are being proposed or implemented in old-growth Pinyon-juniper stands that probably need no restoration.

Mountain Shrubland Type

Composition, Distribution, and Stand Structure of Mountain Shrublands

The Mountain Shrubland type within the SJPA is a diverse, shrub-dominated type that occurs on mountains, hills, and canyon slopes at elevations ranging from about 6000 to 9000 feet. It occurs on upland sites with well-drained soils, and is often found on steep slopes with southerly aspects. It occurs as small patches in forest-dominated landscapes, but sometimes occupies extensive areas. It is found in association with pinyon-juniper, ponderosa pine, and warm-dry mixed conifer vegetation types, in the semi-arid, lower montane, and montane climate zones. Mean annual precipitation ranges from about 14-25 inches. There are about 450,190 acres of this type within the SJPA.

This tall shrubland type occurs as relatively pure stands of Gambel oak, or as a mix of Gambel oak and other deciduous shrubs. It also includes stands dominated by mountain mahogany, serviceberry, or other shrubs where Gambel oak is absent or a minor component. Mature stands tend to be moderate to closed-canopy with low to moderate herbaceous cover. Structure is highly variable depending on the time since the last disturbance (tall if no recent disturbance and short if recent disturbance. Other shrubs associated with this type include bitterbrush, Woods rose, chokecherry, fendlerbush, snowberry, skunkbrush, squawapple, big sagebrush, Oregon grape, and prickly pear. Many of these shrub species (all except the last three which reproduce solely by seed) re-sprout prolifically from roots, rhizomes, or lignotubers after fire kills aboveground portions of the plants (Floyd et al. 2000). Although usually found in shrub form, Gambel oak also occurs in tree form with a large diameter trunk and a height of 10 feet or more. The combined canopy cover of shrubs is ¡Ý 25%, or shrubs make up the most canopy cover of any life form (IRI-CVU).

Pinyon pine, Utah juniper, Rocky mountain juniper, ponderosa pine, white fir, or Douglas-fir trees may be present in the mountain shrubland type, usually occurring as scattered individuals (< 25% total canopy). Common herbs associated with this type include goldenrod, locoweed, milkvetch, lupine, geranium, yarrow, dandelion, meadowrue, American vetch, mulesears, daisy, trailing fleabane, cinquefoil, muttongrass, junegrass, blue grama, bottlebrush squirreltail, needle and thread, Indian ricegrass, Kentucky bluegrass, elk sedge, and mountain muhly.

This type is described as Gambel oak cover type (CVU), Mountain Shrubland (Mutel and Emerick 1984, Blair et al. 1996, Harrington 1964), Foothill Shrubland (Andrews and Righter 1992), Foothills Shrubland Ecosystem (Kingery 1998), Petran Chaparral (Keeley and Keeley 1988, Floyd et al. 2000), Montane Scrub and Successional-Disturbance Scrub (Dick-Peddie 1993), Montane Shrubland (Fitzgerald et al. 1994, Spence et al. 1995), Oak Woodland (Spence et al. 1995), Southwest Chaparral Ecosystem (Paulsen Jr. 1975), Rocky Mountain Montane Deciduous Scrub (NM GAP), Deciduous Oak (Colorado GAP), Brushy Loam (SCS ecological site), Gambel Oak series (Johnston 1987), Oak-Serviceberry Community Type (Johnston 2001), and Gambel Oak Habitat Type (Crane 1982).

Disturbance and Succession of Mountain Shrublands

The gambel oak-dominated mountain shrubland type is an early seral stage of the pinyon pine/gambel oak, ponderosa pine/gambel oak, Douglas fir/gambel oak, and white fir/gambel oak PNV plant associations (DeVelice et al. 1986, Stuever and Hayden 1997, Floyd-Hanna et al. 1996), and is an early seral stage of some pinyon-juniper, ponderosa pine, and warm-dry mixed conifer existing vegetation types (Brown 1958, Crane 1982, Floyd et al. 2000, Harrington 1964, Engle et al. 1983, Dixon 1935, DeVelice et al. 1986, and Stuever and Hayden 1997). Following stand-replacing fire in these conifer types, the mountain shrubland type is established and may persist for long periods as a fire disclimax community (Daubenmire 1966), maintained by recurrent fires that eliminate the conifer seed source and the naturally slow rate of succession back to conifer dominance. The often dense, closed-canopy structure of this type can make it hard for conifer seedlings to become established. Over time (sometimes hundreds of years) without additional fire disturbance, these shrublands will likely succeed to the conifer types mentioned above.

Other disturbance agents for the mountain shrubland type include frost damage, browsing by livestock and wildlife, hydromowing, firewood cutting, prescribed fire, and fire suppression.

Historic Range of Variability (HRV) of Mountain Shrublands (McGarigal and Romme 2005)

The frequency and extent of simulated wildfires in mountain shrubland varied markedly among decades. Wildfire was fairly common in this cover type. In most decades, > 5% of the mountain shrubland burned, inclusive of both high- and low-mortality affected areas, and roughly 2 times per 100 years, > 20% of the area burned. Under this wildfire regime, the return interval between fires (of any mortality level) varied widely from 23 years to > 800 years, with a mean and median of 73 years, and almost no eligible area escaped disturbance altogether over the course of an 800-year simulation. Interestingly, the return interval distribution was strongly bimodal, suggesting that there was in fact two separate disturbance regimes in this cover type. This largely reflected the bimodal elevation distribution of this cover type. Higher-elevation shrubland (3000-4000 m) exhibited a much longer average return interval (~160 years) than lower-elevation shrubland (2000-3000 m; 60 years), reflecting the moister, cooler conditions at higher elevations and the lower frequency of wildfire in general at higher elevations.

The age structure and dynamics of mountain shrubland reflected the interplay between disturbance and succession processes. The survivorship distribution represents the percent of stands that survived to any age, where age represents time since stand origin, not necessarily the age of the oldest individuals in the stand. On average (over time), roughly 50% of the mountain shrubland was > 50 years since stand origin, although at any point in time this varied from 25% to 69%. On average, 25% of the mountain shrubland survived to > 100 years, and < 1% survived a stand-replacing disturbance for > 400 years.

The distribution of area among stand conditions within mountain shrubland fluctuated over time, as expected. For example, the percentage of mountain shrubland in the late shrub-dominated stage varied from 25% to 69%, reflecting the dynamic nature of this cover type when considered over century-long periods. The seral-stage distribution appeared to be in dynamic equilibrium (i.e., the percentage in each stand condition varied about a stable mean), and appeared to reach equilibrium relatively quickly compared to other cover types. The spatial configuration of stand conditions fluctuated markedly over time as well, although there was considerable variation in the magnitude of variability among configuration metrics. Patch area and the proximity index (a measure of patch isolation) exhibited the greatest variability and the herb-shrub condition was particularly dynamic relative to the early and late shrub-dominated stages.

Our estimate of the current seral-stage distribution was almost never observed under the simulated HRV. In particular, the current landscape contains more area in the herb-shrub and early shrub-dominated stage and less in the late shrub-dominated stage than was observed under the simulated HRV, which resulted in an overall seral-stage departure index of 97%. The current seral-stage configuration deviated much less dramatically but still substantially (71%) from the simulated HRV. The magnitude of departure varied dramatically among metrics. In general, the current landscape contains fewer, larger and more geometrically complex and clumped (less isolated) patches of the same seral stage than existed under the simulated HRV, although the nature and magnitude of departure differed considerably among stand conditions. WARNING, due to the lack of reliable field data on current stand age and condition in this cover type, these findings must be viewed with extreme caution. Until more complete data are obtained, we are unable to reach firm conclusions regarding HRV departure for this cover type.

Management Implications for Mountain Shrublands (Romme et al. 2003)

Mountain shrublands are used primarily for grazing, woodcutting, and recreation. Opportunities and interest in restoring, improving, or otherwise modifying this type of vegetation appear limited. Prescribed burning or mechanical removal of biomass is sometimes employed to improve forage for elk, deer, and livestock, but the effectiveness of such treatment is usually short-lived because of the rapid re-sprouting and growth of the shrubs. Dense stands of Gambel oak are also recognized as a fire hazard in the wildland-urban interface because they can burn intensely under extreme fire weather conditions (Romme et al. 2001). At least some component of tall, dense mountain shrublands should be retained in the landscape because they provide valuable nesting habitat for several bird species, including green-tailed towhee and orange-crowned warbler.

Sagebrush Shrubland Type

Composition, Stand Structure, and Distribution of Sagebrush Shrublands

The Sagebrush Shrubland type within the SJPA is a sagebrush-dominated type that occurs on hills, mesas, and valley floors at elevations ranging from about 5000 to 9000 feet. The combined canopy cover of shrubs is ¡Ý 25%, or shrubs make up the most canopy cover of any life form (IRI-CVU). It occurs in association with pinyon-juniper woodland, ponderosa pine forest, semi-desert grassland, mountain grassland, and semi-desert shrubland types. It occurs on well-drained soils in the semi-arid and lower montane climate zones. Soils are variable ranging from deep and non-saline to shallow and alkaline. Mean annual precipitation ranges from about 10-18 inches. There are about 210,030 acres of this type within the SJPA, most of which is located in the west portion of the SJPA.

These shrublands are dominated by basin big sagebrush (*Artemisia tridentata ssp. Tridentata),* black sagebrush *(Artemisia nova),* and Wyoming big sagebrush *(Artemisia tridentata ssp. wyomingensis).* Basin big sagebrush tends to occur on valley floors with deeper soils. Black sagebrush tends to occur on shallow, alkaline soils with high clay contents. Wyoming big sagebrush tends to occur on more upland landscape positions at lower elevations within this type. Mountain big sagebrush (Artemisia tridentata ssp. vaseyana), silver sagebrush (Artemisia cana), and Bigelow sagebrush (Artemisia bigelovii) may also occur within this type, the former at higher elevations, the middle on moister sites, and the latter on dry rocky sites. Basin big sagebrush is a tall species (over 4 feet), while the other types are shorter (less than 4 feet tall). Sagebrush species hybridize, which can make identification difficult (Beetle et al. 1982).

Stands are characterized by moderate to high cover (20-75%) of shrubs, mostly sagebrush. Herbaceous cover, which is highly variable, and bare soil occupy the space between shrubs. Occasionally scattered trees may be present that may include ponderosa pine, pinyon pine, or Utah juniper. Terrestrial cryptogams composed of cyanobacteria, algae, lichen, moss, fungi, or liverwort are often significant components in sagebrush shrublands forming crusts on the ground surface that contribute to soil stability, nutrient supply (organic matter and nitrogen), and biodiversity (Ladyman & Muldavin 1996).

This type is described as sagebrush cover type (CVU), Sagebrush Shrublands (Mutel and Emerick 1984, Spence et al. 1995, Andrews and Righter 1992, Kingery 1998), Semidesert Shrublands (Fitzgerald et al. 1994), Shrub-Steppe community (Blair et al. 1996), Great Basin Desert Scrub (Dick-Peddie 1993), Loamy Foothills and Semi-desert Loam (SCS range sites), Big Sagebrush Shrublands (Colorado GAP, Johnston 2001), Big Sagebrush series (Johnston 1987), Inter-Mountain Basins Big Sagebrush Shrubland (Nature Serve), Sagebrush and Sagebrush/Perennial Grass (Utah GAP), and Colorado Plateau Mixed Low Sagebrush Shrubland (Nature Serve).

Other native shrubs associated with this type may include snakeweed, rubber rabbitbrush, longflower rabbitbrush, bitterbrush, Green¡¯s rabbitbrush, greasewood, fourwing saltbush, winterfat, plains pricklypear, and snowberry. Common native herbs include Indian ricegrass, junegrass, Arizona fescue, mountain muhly, blue grama, galleta, western wheatgrass, needle and thread, muttongrass, bottlebrush squirreltail, lupine, fringed sage, paintbrush, yarrow, trailing fleabane, buckwheat, redroot buckwheat, daisy, owlclover, hairy goldenaster, and globemallow. Common non-native species include Kentucky bluegrass, smooth brome, and cheatgrass.

Historic Range of Variability (HRV) of Sagebrush Shrublands (McGarigal and Romme 2005)

The frequency and extent of simulated wildfires in sagebrush shrublands (includes RMLANDS mesic sagebrush type) varied markedly among decades. In roughly three out of every four decades, > 10% of the mesic sagebrush burned, inclusive of both high- and low-mortality affected areas, and roughly once per 100 years, > 40% of the area burned. Under this wildfire regime, the return interval between fires (of any mortality level) varied from 24 to 800 years, with a mean and median of 55 years and 53 years, respectively, and essentially no mesic sagebrush escaped disturbance altogether over the course of an 800-year simulation. As expected, return intervals varied spatially across the forest, however the patterns were not explainable by any discernable factors, with the possible exception that mesic sagebrush patches embedded in a neighborhood containing cover types with shorter return intervals (ponderosa pine and mountain shrubland) exhibited shorter return intervals, whereas those patches surrounded by cover types with longer return interval (pinyon-juniper woodlands and cool-moist mixed conifer forest) exhibited longer return intervals. Thus, landscape context appeared to have an important influence on wildfire return intervals in mesic sagebrush.

The age structure and dynamics of mesic sagebrush reflected the interplay between disturbance and succession processes. The survivorship distribution represents the percent of stands that survived to any age, where age represents time since stand origin, not necessarily the age of the oldest individuals in the stand. On average (over time), roughly 50% of the mesic sagebrush was > 30 years since stand origin, although at any point in time this varied from < 15% to > 90%. On average, 14% of the mesic sagebrush survived to > 100 years and < 3% survived a stand-replacing disturbance for > 200 years. This highlights the stochastic nature of disturbances, in which some areas by chance alone escaped catastrophic disturbance for relatively long periods.

The distribution of area among stand conditions within mesic sagebrush fluctuated markedly over time, as expected. For example, the percentage in the shrubs-herbs condition varied from 11% to 68%, reflecting the extremely dynamic nature of this cover type when considered over century-long periods. The seral-stage distribution appeared to be in dynamic equilibrium (i.e., the percentage in each stand condition varied about a stable mean), despite the fact that the proportion in herbs-shrubs versus shrubs-herbs during any snapshot (i.e., time step) varied considerably over time. The spatial configuration of stand conditions fluctuated markedly over time as well, although there was considerable variation in the magnitude of variability among configuration metrics. Patch area, core area and the proximity index (a measure of patch isolation) exhibited the greatest variability over time and, in general, the shrub-herb stage was more dynamic than the herb-shrub stage.

Our estimate of the current seral-stage distribution (83% in herb-shrub condition, 17% in shrub-herb condition) was infrequently observed under the simulated HRV. In most decades there was greater equity between stand conditions. Overall, the seral-stage departure index was 83%. The current seral-stage configuration deviated substantially (58%) from the simulated HRV, although there was considerable variation (0-100% deviation) among metrics. **WARNING**, due to the lack of reliable field data on current stand age and condition in this cover type, these findings must be viewed with extreme caution. Until more complete data are obtained, we are unable to reach firm conclusions regarding HRV departure for this cover type.

Landscape structure

Landscape composition, measured as the proportion of the landscape in each of 57 distinct and dynamic patch types (defined by unique combinations of cover type and stand condition), fluctuated markedly over time. The average coefficient of variation (representing the 90 percentile range of variation about the median) across patch types was 131%, although it ranged broadly from 32% to 746%. There were no obvious discernable factors (cover type, seral stage, elevation) explaining the variability among patch types; however, the less common patch types (i.e., those comprising a smaller proportion of the landscape on average over time) were more likely to exhibit the greatest variability over time. Interestingly, despite the high degree of dynamism exhibited by most patch types, the overall diversity of the mosaic, as represented by the Simpson's diversity index (which is a function of the number of patch types and the equatability in the distribution of area among patch types), was surprisingly stable with a coefficient of variation of only 1%.

Landscape configuration, measured by 19 different metrics that characterize the spatial character and arrangement, position, or orientation of patches within the landscape, was much less dynamic than landscape composition. The average coefficient of variation across metrics was only 25% (range 1-73%). There were a couple of notable patterns of variation. First, metrics associated with patch size (area, core area, density) and isolation (proximity index) were more dynamic than metrics associated with patch geometry (e.g., shape, core area index) and edge contrast. Thus, the principal dynamics were associated with the grain of the patch mosaic. Second, in general, the area-weighted metrics were more dynamic than their unweighted counterparts. The area-weighted metrics give more weight to the larger patches when calculating the mean and are therefore not influenced greatly by variations affecting small patches. Thus, these metrics are less affected by the fine-grained heterogeneity created by the disturbance processes (e.g., abundant, small scattered patches) and provide a better overall measure of changes in the coarse-grain structure of the landscape. Taken in aggregate, these results suggest that the principal dynamics were associated with changes in the size and continuity of the large patches in the landscape; i.e., that large contiguous patches were periodically broken up (fragmented) by disturbance events but eventually coalesced to reform a coarse-grain mosaic, only to be broken up again.

The current landscape structure deviates substantially (84% departure index) from the simulated HRV, and departure is much greater for the spatial configuration of the landscape (91% departure index) than the composition of the landscape (76% departure index). With respect to the composition of the landscape, more than half of the patch types (29/57) are completely outside their HRV (100% departure index), while roughly one-sixth of the patch types (9/57) are within their 25-75th percentile range of variation (0% departure index). When the late-seral stages (i.e., understory reinitiation and shifting mosaic) are combined into a single, aggregated "late-seral" stage within each forest cover type to reflect the inadequate discrimination between these conditions in the current landscape database, the proportion of patch types completely outside their HRV declines to one-third (16/48) and the landscape composition departure index declines to 68%. The principal cause of the decline is the change in departure for ponderosa pine-oak forest and spruce-fir forest. In both cases, our estimates of the current percentage of the landscape (or cover type) in the separate understory reinitiation and shifting mosaic conditions are completely outside their simulated HRV (100% departure index); whereas, the combined late-seral stages are both well within their respective 25-75th percentile ranges of variation (i.e., 0% departure index).

With respect to the configuration of the landscape, most of the metrics (15/19) are completely outside their HRV (i.e., 100% departure index) and two others are nearly so, while only one metric (interspersion and juxtaposition index) is completely within its 25-75th percentile range of variation (0% departure index). In general, the current landscape contains fewer, larger, more extensive and less isolated patches than existed under the simulated HRV. The larger patches in the current landscape also tend to be geometrically less complex and contain proportionately more core area than existed under the simulated HRV. As a result, the current landscape contains much less edge than existed under the simulated HRV, but the edge that exists has higher average contrast than observed under the HRV. This latter finding is likely due to the fact that most of the edge in the current landscape is derived from abutting patches of different cover types and often involves high-contrast edges between forest and nonforest cover types (e.g., water, barren); whereas, under the simulated HRV, proportionately more of the edge is due to abutting patches of different seral stages of the same or similar cover types - which have lower contrast. Overall, the current landscape is more contagious and less structurally diverse than ever existed under the simulated HRV.

Management Implications for Sagebrush Shrublands

Management activities that occur in sagebrush shrublands within the SJPA include livestock grazing, gas development, fuels management, and wildlife habitat enhancement for big game winter range and Gunnison sage grouse.

Throughout the West, sagebrush stands are being invaded by exotic annual species such as cheatgrass (McArthur 1992). Fire frequency increases as cheatgrass increases. This is evident in Disappointment Valley on the SJNF and pronounced in many areas on the adjacent BLM lands.

Excessive wildlife and livestock grazing removes grasses and forbs within the sagebrush type. Re-establishment of the grass-forb understory is difficult.
The encroachment of pinyon-juniper woodlands can have significant impacts on sagebrush communities. Without fire or other management activities, pinyon pine and juniper trees can become established within a sagebrush habitat with eventual conversion to woodlands.

Following fire most sagebrush species, including basin big, black, and Wyoming big must reestablish themselves by seeds, so growth and recovery are slow (Bunting et al. 1997 as cited in Johnston 1997).

Much of the sagebrush system in western Colorado is in poor condition (Winward 2004). Major impacts to lower elevation sagebrush include: loss of understory grass and forb production and diversity; reduction of surface litter and the resulting loss of organic matter being incorporated into the upper soil horizons (necessary for holding moisture needed for recruitment of new plant seedlings); increase in bare ground and soil erosion; and out of balance sagebrush densities, canopy cover values and age class ratios (Winward 2004).

Mountain Grassland Type

Composition, Structure, and Distribution of Mountain Grasslands

The Mountain Grassland type within the SJPA occurs as openings in forest-dominated landscapes. It occurs on upland sites with well-drained soils in mountain and plateau landscapes. It is associated with the lower montane, montane, and subalpine climate zones at elevations ranging from about 7500 to 11,600 feet. The combined canopy cover of grasses is >= 25%, or grasses make up the most canopy cover of any life form (IRI-CVU). Grasslands are devoid of trees and shrubs due to many factors including competition from herbaceous plants, temperatures, grazing, soil heaving, and fire (Paulsen 1975). There are about 304,314 acres of this type within the SJPA.

Within the general mountain grassland type, there are three more specific grassland types within the SJPA. The Arizona fescue type occurs at elevations from about 7500 to 9000 feet, and is primarily associated with ponderosa pine and warm-dry mixed conifer forests. The Thurber fescue type occurs at higher elevations from about 8500 to 11600 feet, and is primarily associated with spruce-fir and cool-moist mixed conifer forests. The Kentucky bluegrass type occurs at elevations from about 7500 to 11600 feet, and is associated with ponderosa pine, all mixed conifer, and spruce-fir forests.

The Arizona and Thurber fescue types are tall, bunchgrass communities dominated by Arizona and Thurber fescue respectively. Forbs, sod-forming grasses, and sedges occupy the spaces between bunchgrasses. Sites in good ecological condition display high canopy-cover of Arizona and Thurber fescue, abundant litter, minimal bare soil, and soils with thick, organic-matter-rich surface horizons. Sites where ecological conditions are not as good display lower canopy-cover of Arizona and Thurber fescue, less litter, more bare soil, and more Kentucky bluegrass, noxious weeds, dandelion, annual species, and native increaser species including yarrow, strawberry, cinquefoil, sneezeweed, trailing fleabane, pussytoes, and mulesears.

The Kentucky bluegrass type is a short, sod-forming type dominated by Kentucky bluegrass and the native increaser species mentioned above. Noxious weeds, dandelion, and annual species are common. Arizona and Thurber fescue are absent or minor components. This type commonly displays high amounts of bare soil, minimal litter, and soil erosion and compaction problems. These grasslands are a zootic disclimax type (Daubenmire 1966) due to sustained livestock grazing since the reference period that has significantly changed their composition and structure, from that of an Arizona or Thurber fescue bunchgrass type during the reference period, to the current bluegrass type (Fleischner 1994, Wuerthner 1992, Belsky et al. 1997, Savage 1991, Turner 1976).

The mountain grassland type is described as a Mountain Grassland PNV Type (Kuchler 1964), Mountain Grassland Type (Paulsen Jr. 1975, Turner and Paulsen 1976), Mountain Grassland Ecosystem (Kingery 1998, Garrison et al. 1977), Grass/Forb Cover Type (CVU), Mountain Loam, Shallow Loam and Subalpine Loam (SCS Range Sites), Arizona fescue, Thurber fescue, and Kentucky bluegrass series (Moir 1983, Johnston 1987, Hess and Wasser 1982, Dick-Peddie 1993, Johnston 2001), and Arizona fescue and Thurber fescue type (Turner and Paulsen 1976, Currie 1975, Costello and Schwan 1946, Johnston 2001).

Common species of the Arizona fescue type include Arizona fescue, mountain muhly, Parry oatgrass, junegrass, bottlebrush squirreltail, mountain brome, blue grama, needle-and-thread, needlegrass, pine dropseed, muttongrass, elk sedge, Kentucky bluegrass, American vetch, daisy, trailing fleabane, yarrow, pussytoes, dandelion, mulesears, lupine, fringed sage, cinquefoil, buckwheat, peavine, geranium, and goldenrod. Common species of the Thurber fescue type include Thurber fescue, Arizona fescue, timber oatgrass, junegrass, mountain brome, needlegrass, wild rye, sedge, Kentucky bluegrass, cinquefoil, American vetch, trailing fleabane, daisy, yarrow, sneezeweed, dandelion, harebell, meadowrue, and geranium. Shrubs, including big sagebrush and black sagebrush for the Arizona fescue type, and shrubby cinquefoil and snowberry for the Thurber fescue type may be present usually as minor components or small patches.

Some mountain grasslands within the SJPA are dominated by smooth brome, intermediate wheatgrass, timothy, crested wheatgrass, and orchardgrass. These exotic species were introduced through the seeding of roads, trails, and disturbed sites, and have flourished in some grassland sites. Mountain grasslands that have lost native species often have high amounts of bare soil, which makes them susceptible to the establishment of these exotic plants. Some grasslands were purposely seeded with these exotic species for erosion control and to improve forage conditions for livestock.

Disturbance and Succession of Mountain Grasslands

Since the reference period, livestock grazing has had profound impacts on the composition, structure, and function of grassland ecosystems in the western U.S. (Fleischner 1994, Wuerthner 1992, Belsky and Blumenthal 1997), including those of the SJPA. This is not surprising, since cattle prefer mountain grasslands because they tend to be easily accessible, forage is more abundant compared to forests, and grassland species are highly palatable (Clary 1975, Paulsen 1975). These impacts include 1) alteration of species composition, including decreases in density and biomass of individual species, and reduction of species richness; 2) disruption of ecosystem functioning, including interference in nutrient cycling and ecological succession; 3) alteration of ecosystem structure, including changes in vegetation stratification and community organization; 4) a decrease in biological diversity; and 5) an increase in soil erosion.
Cattle and sheep livestock grazing occurred in the SJPA before the reference period (Touchen et al. 1996, Savage 1991), but extensive livestock grazing and the associated adverse impacts, began around the 1870s, when Anglo settlers arrived in increasing numbers (Savage 1991). Heavy grazing throughout the Area continued into the 20th century, and more of this unregulated grazing occurred in upland forests and meadows in the national forests (DuBois 1903). This early livestock grazing became excessive in many areas (Dishman 1982). Range conditions began deteriorating in the 1890s, and accelerated over extensive areas between 1904 and 1920 (Forest History, Volume 1). As grassland sites became overgrazed and unable to support cattle, cattlemen found new, unexploited grasslands for their animals (Dishman 1982). Since the 1930s, aided by the passage of the Taylor Grazing Act in 1934, livestock grazing has been regulated on public lands, which has undoubtedly helped rangeland conditions, but the legacies of uncontrolled livestock grazing that occurred before that time are still with us.

The extensive livestock grazing since the reference period has resulted in significant ecological changes to the mountain grasslands within the SJPA. Some of the grasslands of the Thurber fescue type have experienced limited livestock grazing and display characteristics similar to those of the reference period. Other grasslands of the Thurber fescue type, and most of the grasslands of the Arizona fescue type, have experienced intensive livestock grazing since the reference period, and have changed significantly in composition, structure, and function. Altered plant-community composition is the most obvious change. The dominant bunchgrasses that are so critical to the ecological integrity of these ecosystems have been extirpated in many areas, and the abundance and distribution of remaining bunchgrasses have been drastically reduced in others. Exotic species (including Kentucky bluegrass, dandelion, and noxious weeds), and native increaser species (including yarrow, cinquefoil, mulesears, sneezeweed, and annual forbs) have taken the place of bunchgrasses in many places. This change in species composition has altered the structure of many of these grasslands from tall, bunchgrass-dominated community types to short, sod-dominated or forb-dominated types. Additional livestock grazing-related changes to the mountain grasslands within this Section include the reduction of litter, an increase in bare soil, increases in soil erosion and compaction resulting in a loss of soil productivity, and a decrease in native species richness.

In addition to changes in the composition and structure of mountain grasslands within the SJPA, the function of these ecosystems has also been affected since the reference period, due to changes in the process of grazing associated with the large numbers of livestock that were introduced. Unlike the grazing process associated with wild ungulates, domestic livestock grazing maintains consistently high concentrations of animals, and incorporates fences and cattleguards that restrict movement, resulting in higher frequencies and intensities of defoliation on forage species (Archer and Smeins 1991). The resulting elimination and decrease in the abundance and distribution of bunchgrasses and the litter they produce not only changed the composition and structure of the grasslands, but also decreased the amount of organic matter available for nutrient cycling to the soil, changed successional pathways, and lessened the likelihood of low-intensity fires carrying through these grasslands. During the reference period, the dense and uniformly distributed bunchgrasses of the mountain grasslands provided the fine fuels necessary for fire that ignited in the adjacent forests to burn through the grasslands (Touchen et al. 1993). Fire exclusion over the last half-century has also lessened the chances for fire to spread from adjacent forests into the mountain grasslands.

Land development and land management activities have introduced exotic plants and noxious weeds into the mountain grasslands of the SJPA. Roads and trails needed for home sites, recreation, timber sales, livestock grazing, mining, pipelines, communication towers, and fire suppression have been conduits for the introduction and spread of exotic plants into the forested landscapes where the grasslands occur. Mountain grasslands that have lost native species often have high amounts of bare soil, which makes them susceptible to the establishment of exotic plants. Many roads and grasslands were purposely seeded with exotic species for erosion control and to improve forage conditions for livestock. Common exotic species found within this Section include smooth brome, intermediate wheatgrass, timothy, orchardgrass, Kentucky bluegrass, and yellow sweet clover.

Historic Range of Variability (HRV) of Mountain Grasslands

Descriptions such as the following give us insight and help paint a picture of what the mountain grasslands in the SJPA looked like during the reference period.
J.T.Rothrock in 1878 (cited in Cooper 1960) wrote as he traveled through southwest Colorado, "In the beautiful valley of the Conejos River, we found luxuriant bunch-grass covering the ground as thickly as it could stand".

E.F.Beale in 1858 (cited in Cooper 1960) wrote as he traveled through northern Arizona, "We came to a glorious forest of lofty pines, through which we have traveled ten miles. The country was beautifully undulating, every foot being covered with the finest grass, and beautiful broad grassy vales extending in every direction."
Aside from historic anecdotal descriptions, there is little information or scientific data pertaining to the condition of mountain grasslands within the SJPA during the reference period. Furthermore, intensive livestock grazing since the reference period has eliminated many undisturbed reference sites that could have been used to represent reference-period conditions. It is therefore difficult to accurately reconstruct the composition and structure of these plant communities prior to the arrival of domestic livestock in the late 1800s. That considered, the following description is based on the best reference sites available, current and historic range analysis data, classification and inventory sample plots, and extensive field reconnaissance within the SJPA.

During the reference period many of the mountain grasslands within the SJPA displayed high diversity and cover of herbaceous species, particularly the native bunchgrass species Arizona and Thurber fescue that displayed a combined canopy cover greater than 50%. Forbs, sod-forming grasses, and sedges occupied the spaces between bunchgrasses. Litter amounts were high due to the abundance of plant material associated with the robust bunchgrasses. Total vegetation cover (including litter) was about 80-90%. Bare soil occupied 10-20% of the ground surface. Exotic species (including Kentucky bluegrass and dandelion) and noxious weeds were absent. Annual forbs and native increaser species such as yarrow, strawberry, cinquefoil, sneezeweed, and mulesears occurred to a minor extent. Soil erosion and compaction were absent or minimal. Soils had thick, organic matter-rich surface horizons formed from the consistent supply of plant material associated with the abundant and well-distributed vegetation. Structural conditions displayed a relatively closed canopy, reflecting the high density and well-distributed arrangement of the robust bunchgrasses. The tall, thick foliage of these bunchgrasses provided an abundance of litter and organic matter for energy flow, nutrient cycling, and soil development processes, and influenced hydrologic processes by protecting the soil surface from raindrop impacts, runoff, compaction, and erosion.

The function of mountain grasslands within the SJPA during the reference period was associated with the ecological processes of grazing by wild ungulates and fire. Intermountain grasslands of North America evolved with light grazing by wild herbivores, including elk and deer (Mack and Thompson 1982). The numbers of wild ungulates and patterns of their grazing likely varied from year to year. The animals were free to move to new areas when the abundance or palatability of desired forage species decreased. Since the plant species associated with the grasslands of this Section presumably were well adapted to the frequency, intensity, extent, and magnitude of historic grazing by wild ungulates, the composition, structure, function, and productivity of these ecosystems were maintained during the reference period.

Periodic fire was a part of the natural-disturbance regime of North American grasslands (Sims 1988, Wright and Bailey 1982). The grasslands of the Arizona fescue type were part of the overall forest-dominated landscape associated with the fire-adapted ponderosa pine and warm-dry, mixed conifer forests of this Section (Currie 1975, Clary 1975). Since these forests were characterized by frequent, low-intensity fires throughout the reference period (Romme et al. 1998, Touchen et al. 1996), we assume that frequent fires also played a significant role in maintaining the composition, structure, and function of the adjacent mountain grasslands during the reference period. The grasslands had plenty of biomass serving as fine fuel to carry a fire through them. Fire functions to reduce litter, recycle nutrients to the soil, stimulate new herbaceous growth, and restrict woody-plant establishment in grasslands (McPherson 1995). The ecological significance of fire in the Thurber fescue type was and is probably less significant, compared to the Arizona fescue series, since fire frequencies are much longer in the higher-elevation spruce-fir, cold-wet, and cool-moist mixed conifer forests that these grasslands are associated with (Romme et al. 1998).

Management Implications for Mountain Grasslands

Ecologically, the changes that have occurred to many of the mountain grasslands within the SJPA since the reference period can be regarded as undesirable, since they have altered the composition, structure, and function of the grasslands, and because the changes have resulted in a decrease in the biological diversity of these ecosystems (Mullen 1992). Livestock grazing has had a significant effect on the grasslands since the reference period, and it appears that grazing will continue to be a major, long-term land use within the SJPA. If unsustainable grazing practices (continuous grazing, overstocking, overutilization) occur, the ecological condition of the mountain grasslands will decline, resulting in further loss of ecosystem integrity and biological diversity. In addition to livestock grazing, the increasing effects of people associated with land development, recreation, timber harvest, and roads, and the effects associated with fire suppression and foraging wildlife species must be considered in the management of the mountain grasslands.

Updated resource information, new management practices, and new livestock grazing methods are available to help us manage these grasslands for long-term viability and sustainability. New classification, inventory, and analysis information will help us to better recognize and understand the ecological components and relationships associated with these ecosystems. New research and monitoring efforts will help us to implement appropriate grazing systems, grazing times and durations, stocking levels, utilization standards, and distribution patterns. Prescribed fire (both natural and management-ignited) and selective logging are being implemented in ponderosa pine and warm-dry, mixed conifer forests that will help to restore ecological conditions more similar to those of the reference period. This will influence the adjacent mountain grasslands by reestablishing a more frequent fire regime, and by creating structural conditions more conducive to the establishment and spread of native herbaceous species.

Mountain grasslands within the SJPA that are not in good ecological condition (Kentucky bluegrass type where keystone native bunchgrasses are absent or minor and soil productivity is reduced) must be recognized, but may not be worth the expenditure of energy and resources, since they pose such difficult management questions. Plant communities are rather stable and resistant to change up to certain biotic and abiotic thresholds. Beyond that, changes can be rapid, dramatic, and potentially irreversible over reasonable time frames. Once a threshold is exceeded, it may not be possible for the ecosystem to return to the previous condition, even with large inputs of energy such as seeding, fertilization, and mechanical manipulation (Archer & Smeins 1991). In some mountain grassland sites within the SJPA, the biotic thresholds have been exceeded due to the extirpation of key native species. The potential for natural reproduction and recruitment of new individuals appears to have been lost, since the plants needed for asexual and sexual reproduction are gone. It may not be worthwhile to direct energy and resources into the restoration of these sites, due to economic constraints and the difficulties and uncertainties associated with this type of restoration.

Simply removing livestock from grasslands that are in poor ecological condition may not improve those conditions. When grazing pressure is reduced or removed from a site, the rate of succession or change back to a previous condition depends on the extent to which soils, seedbank, and vegetative regeneration potential of the previous vegetation has been modified. It should not be assumed that the pathway of succession following reduction in grazing intensity would be a simple reversal of the pathway of retrogression. Rates of succession on sites released from grazing generally proceed much more slowly than desired for management purposes. The goal of sustainable management is to recognize and anticipate the critical thresholds described above, and manage so as not to exceed them.

The importance and ecological values associated with the mountain grasslands of the SJPA that currently are in relatively good ecological condition (Arizona and Thurber fescue types where keystone native bunchgrasses are common and well distributed) must be recognized and properly managed for. Energy and resources devoted to these grasslands are essential when management activities (particularly livestock grazing) are being considered, in order to develop appropriate management and monitoring plans that will insure the ecological integrity and sustainability of these ecosystems. Important management objectives for these grasslands must be to perpetuate all native species, and to maintain or increase the abundance and distribution of the keystone species, Arizona and Thurber fescue. This must occur in order to ensure the productivity, reproductive capability, competitiveness, and long-term survival of these species. Other objectives for these grasslands should include keeping exotic plants and noxious weeds out of these ecosystems, and maintaining soil productivity. Management activities that cannot meet these objectives should not be considered.

Cooperrider (1991) noted that "the principal purpose of most rangeland rehabilitation projects has been restoration of livestock forage," which often involves the introduction or maintenance of exotic forage species at the expense of native species. If we decide to expend energy and resources to restore our highly altered mountain grasslands, the principal purpose should instead be to restore the ecological integrity of these native ecosystems through the elimination of exotic species, the reduction of native species that have increased to levels beyond those which existed during the reference period, and the reintroduction of native bunchgrass species.

Alpine Type

Composition and Distribution of the Alpine Type

Alpine ecosystems within the SJPA occur on mountain landscapes above timberline at elevations above about 11,500 feet. Mean annual precipitation ranges from about 30-50 inches. The alpine climate zone is characterized by short-cool growing seasons, long-cold winters, snow, high wind, and intense light. The alpine landscape was shaped by geologic events and glaciation, and is both steep and rugged, and gentle and smooth. Rock outcrop and talus slopes are common. Soils are shallow and rocky on steep slopes and exposed ridges, and deeper and more productive on other sites. Climate, geomorphologic processes, and on-going disturbances including nivation, solifluction, and frost action are major factors influencing the distribution of biota in alpine ecosystems. Small-scale topographic changes exert a significant influence on snow and moisture conditions, and the associated vegetation. There are about 186,494 acres of this type within the SJPA.

There is tremendous diversity of species and vegetation communities within the alpine type, including mosses and lichens, which constitute a significant contribution to the total flora (Johnson and Brown). Vegetation types often change quickly and abruptly over short distances. Within the general alpine type, there are four more specific vegetation types on the SJNF: turf, fellfield, dwarf willow, and wetland (Baker 1983, Paulsen 1960, Dick-Peddie 1993, Thilenius 1975).

The dwarf willow type is composed of alpine and net willows, and commonly occurs on sites with a heavy snowpack that extends into the summer.

The fellfield type occurs on harsh, wind-swept sites with shallow, rocky, excessively-drained, and undeveloped soils. It is dominated by a relatively low canopy cover of short cushion plants (forbs) including nailwort, desert sandwort, dwarf clover, shaggyleaf clover, alpine phlox, alpine forget-me-not, alpine campion, and alpine sandwort.

The turf type occurs on protected sites away from excessive wind, and tends to have deeper, moister, and better developed soils. It is dominated by forbs and graminoids, and usually displays a relatively high canopy cover. Within the general turf type, a number of community types can be recognized including elynoid sedge, kobresia, alpine avens, and Thurber fescue. *A combination turf/fellfield community (curly sedge/cushion plants) is also recognized (Baker 1983, Dick-Peddie 1993).*

The wetland type primarily occurs on poorly-drained, low-lying sites where water accumulates. It tends to display high cover and includes the following community types: tufted hairgrass, shrubby cinquefoil, herbaceous wetland, tall willow, and Drummond rush. The tall willow type is composed of planeleaf and gray willows, and occurs along riparian corridors that are often connected to lower elevation subalpine streams. The herbaceous wetland type is variable and displays a great diversity of species including marsh marigold, elephanthead, kings crown, rose crown, Parry primrose, and sedges.

Krummholz, composed of dwarfed conifers (mostly Engelmann spruce) and alpine herbaceous species, is a transition type that occurs between spruce-fir forests of the subalpine climate zone and true alpine. A rockland or talus type is recognized and dominated by scree groundsel, alpine nodding groundsel*, and lichens (Dick-Peddie 1993).*

At a landscape scale, the alpine type dominates many high elevation landscapes. It also occurs as a component in other landscapes intermixed with spruce-fir forests, aspen forests, and mountain grasslands. Diverse geology and glacial topography add to the complexity and diversity of these landscapes. Most alpine-dominated landscapes are unroaded and mostly undisturbed by human activities.

Past Disturbance of the Alpine Type

Relative to the reference period, the current state of alpine ecosystems in the SJPA reflects impacts associated with livestock grazing, mining, and recreation. Possible effects of warming climates and acid deposition are also concerns (Dick-Peddie 1993). The effects of cadmium found in ore-rich areas of the SJPA (Silverton area) may be affecting white-tailed ptermigan and other herbivores in alpine ecosystems throughout much of Colorado (Larison 2001).

Sheep are the principle domestic livestock using alpine lands. Cattle and sheep livestock grazing occurred in the SJPA before the reference period (Touchen et al. 1996, Savage 1991), but extensive livestock grazing and the associated adverse impacts, began around the 1870s, when Anglo settlers arrived in increasing numbers (Savage 1991). Heavy grazing throughout the Section continued into the 20th century, and more of this unregulated grazing occurred in upland forests and meadows in the national forests (DuBois 1903). This early livestock grazing became excessive in many areas (Dishman 1982). Range conditions began deteriorating in the 1890s, and accelerated over extensive areas between 1904 and 1920 (Forest History, Volume 1).

Sheep numbers peaked in the western U.S. about 1910, and have declined since (Stoddart and Smith 1955). The early sheepmen grazed their animals yearlong, and moved into the alpine zone for summer grazing. Historically, sheep in the alpine are grazed in tightly grouped bands and continuously bedded in the same location for several nights in a row. These practices resulted in large losses of forage through trampling and in soil damage from excessive trailing to and from the bedding ground to water (Thilenius 1975). Sheep driveways in the alpine (including the SJPA) are also noteworthy for the extent of range deterioration present on them, including a lack forage plants, and an increase in erosion and runoff. Concern for the condition of alpine ranges is reflected in reduction in grazing by sheep. From 1939 to 1959, sheep numbers on alpine ranges were reduced about 50 percent (Wasser and Retzer 1966). Since the 1930s, aided by the passage of the Taylor Grazing Act in 1934, livestock grazing has been regulated on public lands, which has undoubtedly helped rangeland conditions, but the legacies of uncontrolled grazing that occurred before that time are still with us.

In addition to loss of forage for livestock and wildlife, impacts to alpine ecosystems from livestock include 1) alteration of species composition of communities, including decreases in density and biomass of individual species, and reduction of species richness; 2) disruption of ecosystem functioning, including interference in nutrient cycling and ecological succession; and 3) alteration of ecosystem structure, including changing vegetation stratification and community organization. These impacts have contributed to soil erosion and to a decreased availability of water to biotic communities (Fleischner 1994).

Mining was a common activity in alpine ecosystems. Old supply roads, deserted structures, settling ponds, and mine tailings are common mining features in many areas. When many of these mines were abandoned, steps to limit pollution from them were stopped or never initiated, causing major pollution problems (the upper Animas River within the SJPA is an example). Abandoned mine tailings often contain soils that are highly acidic and laden with toxic concentrations of heavy metals. Erosion, acid-water runoff, and sedimentation from mines often result in adverse impacts to vegetation, streams, and aquatic ecosystems (Johnston and Brown 1979).

Recreation in alpine ecosystems of the SJPA and elsewhere is increasing. Recreation activities in the alpine include hunting, fishing, hiking, camping, mountain climbing, photography, skiing, off-road vehicle use, horseback riding, and nature watching. The 14,000-foot peaks within the Weminuche Wilderness are receiving an increase in use as the popularity of climbing Colorado's "fourteeners" increases (SJNF Wilderness Management Direction). As recreation increases, adverse impacts associated with recreation will also increase including human and recreational livestock trampling of vegetation, reduction in vegetation cover and species richness, trail deterioration, soil compaction, soil SJNF Wilderness Management Direction). Increased recreation in alpine ecosystems can increase the risk of introducing exotic plants and noxious weeds to the alpine of this Section. Disturbance from off-road vehicles in places like the "Alpine Loop" which traverses a lot of alpine can destroy vegetation leading to severe erosion and sedimentation. Damage can result from a few or even one vehicle pass in the alpine, and often becomes worse each year even after the use has stopped (Johnston and Brown 1979). These high visibility vehicle scars and erosion channels are very difficult to revegetate and rehabilitate.

Air pollution impacts on alpine ecosystems, particular aquatic systems are important. High elevation lakes found in alpine zones appear to be the most sensitive to air pollution (Haddow et al. 1998). Potential air pollution-caused changes from commercial industrial processes include growth and mortality effects to lichens, zooplankton, and mosses, changes in soil PH and soil metal concentrations, and changes in water PH, alkalinity, metals, and dissolved oxygen. Long-term monitoring is needed to detect trends from human-caused disturbance, and to determine how significant the effects of acidic deposition, air toxic compounds, airborne nutrients, and mercury are to alpine ecosystems. In addition to monitoring high elevation aquatic systems, monitoring of lichens, mosses, and some alpine vascular plants, which often grow in nitrogen-deficient environments, is possible (Haddow et al. 1998).

Historic Range of Variability (HRV) of the Alpine Type

Historic information and inventories of alpine ecosystems during the reference period in the SJPA are limited. The reference period is defined as the period of indigenous settlement, from about 1500 until the mid to late 1800s, prior to major impacts of Euroamerican settlement (Romme, this document). Use of alpine by native peoples during the reference period was apparently mainly associated with campsites during game drives or as stopover points on travel across the range (Wendorf and Miller 1959, Husted 1974). Many alpine ranges have been used for summer grazing continuously since the reference period (mid to late 1800s), so it is difficult to determine what the historic condition of those alpine ranges was (Thilenius 1975).

Since historic information is lacking and livestock grazing since the reference period has affected some sites, it is difficult to accurately reconstruct the vegetation composition and structure of some alpine ecosystems during the reference period. During the reference period some alpine ecosystems looked much like they do today, except where known impacts have occurred such as livestock impacts along stock driveways, or sheep bedding grounds.

More studies are needed within the SJPA to establish HRV for this vegetation type.

Management Implications for the Alpine Type

Updated resource information, new management practices, and new livestock grazing methods are available to help us manage alpine ecosystems for long-term viability and sustainability. New classification, inventory, and analysis information will help us to better recognize and understand the ecological components and relationships associated with these ecosystems. New research and monitoring efforts will help us to implement appropriate grazing systems, grazing times and durations, stocking levels, utilization standards, and distribution patterns. Additional alpine ecological classification work would greatly help to better understand vegetation/soil/ geomorphic relationships and the normal variability associated with them. This would help to better determine the effects of livestock grazing on alpine ecosystems and the community types within them.

It appears that grazing will continue to be a long-term land-use in the alpine within the SJPA. If unsustainable grazing practices (continuous grazing, overstocking, overutilization) occur, the ecological condition of alpine ecosystems will decline, resulting in a loss of ecosystem integrity and biological diversity. Livestock grazing in the alpine zone poses some difficult challenges. A short growing season is one. A lack of fenced pastures is another. A given area should be grazed only one time during the grazing season, and the sheep should not be allowed to remain in a given area long enough to cause excessive utilization. One-night bedding at one location is recommended to avoid trailing to and from water during multiple nights at one bedding site. Herding is still the main practice on alpine sheep ranges on federal land. Whether or not good grazing systems in the alpine are used depends to a great extent upon the desire and experience of the herder.

Certain alpine vegetation types are better suited to livestock grazing than others. Wetland types may be a concern because of the susceptibility of their vegetation to trampling damage. Fellfield types may be a concern because of their low herbage production and their high erosion potential. Lands with steep slopes (greater than 40%) are a concern because of their high erosion potential.

New guidance and direction, such as that found in SJNF Wilderness Management Direction will be important to minimize recreational impacts and protect alpine ecosystems in the SJPA including those found in the Weminuche, South San Juan, Lizard Head, Wheeler Peak, and Pecos Wilderness Areas. Ways to manage and protect alpine areas from recreation use include reducing or controlling use in problem areas, changing visitor behavior through information and education, and restoring damaged sites (Cole et al. 1997).

The importance and ecological values associated with alpine ecosystems of the SJPA must be recognized and properly managed for. Energy and resources devoted to these ecosystems are essential when management activities (particularly livestock grazing) are being considered, in order to develop appropriate management and monitoring plans that will insure the ecological integrity and sustainability of these ecosystems. An important management objective must be to perpetuate all native species, and to maintain, perpetuate, or increase the abundance and distribution of key species. This must occur in order to ensure the productivity, reproductive capability, competitiveness, and long-term survival of these species. Other objectives include keeping exotic plants and noxious weeds out of these ecosystems, and maintaining soil productivity. Management activities that cannot meet these objectives should not be considered.

Rehabilitation of disturbed alpine ecosystems is a challenge. Research data indicates that recovery of alpine disturbances by natural successional processes alone often takes centuries (Brown et al. 1978b, Griggs 1956). The harsh nature of the alpine environment often precludes the use of traditional rehabilitation methods developed for more moderate climates. The Wilderness status of most alpine lands also precludes the use of motorized equipment usually used for rehabilitation efforts. Short growing seasons, cool summer temperatures, unique physiological features of alpine plants, strong winds, drought, high evaporation rates, and frost action complicate rehabilitation efforts (Johnston et al. 1975). Soils that are shallow, rocky, undeveloped, acidic, low in organic matter and nutrients, and have low water-holding capacity and availability also complicate rehabilitation efforts.
A major problem associated with rehabilitating alpine disturbances is the selection of adapted plant species (Brown et al. 1976). Native species offer the most advantages regarding natural selection of adaptive mechanisms, since they often show increased vigor and rates of spread compared to exotic species (Brown et al. 1976, Brown and Johnson 1976). Species found successfully occupying disturbed sites are ones to be included in a rehab program, since they provide direct evidence of adaptability and survival on disturbed sites. Unfortunately, adapted native species are usually commercially unavailable, and if available are very expensive. Seed collection by hand in alpine ecosystems is difficult and is compounded by the fact that many alpine species are low seed producers and their seed has low viability.

Riparian Area And Wetland Type

Riparian areas and wetlands occur at all elevations within the SJPA. They occur on valley floors and in other low-lying landscape positions, and are primarily associated with perennial streams. Although they are small in extent, they represent a very important ecological component of SJNF landscapes and of the Section's biological diversity. There are about 77,964 acres of this type within the SJPA.

Riparian areas and wetlands are defined here together, as the interface between the riverine aquatic ecosystem and the adjacent upland ecosystem, where the water table is usually at or near the land surface (Gregory et al. 1991, Risser 1990, Knoft et al. 1988, Brinson et al. 1981, Cowardin et al. 1979). They are frequently flooded, or at least seasonally saturated by a fluctuating water table, and have plant species, soils, and topography that differ considerably from those of the adjacent uplands (Elmore and Beschta 1987, Jones 1990). Riparian areas and wetlands are inundated or saturated with water at a frequency and duration sufficient to support a prevalence of hydrophytic vegetation adapted for life in saturated soil conditions (Corps of Engineers 1987).

The variability within the general riparian area and wetland type is described here at the Subclass level (Grossman et al. 1998, Muldavin et al. 2000, Redders 2003). Subclasses are aggregations of alliances, and based on the predominant leaf phenology of the life form in the upper canopy layer. Alliances are aggregations of community types named by species that dominate the uppermost canopy layer.

Evergreen Riparian Forests: This subclass is dominated by evergreen trees at elevations of about 8000 to 10000 feet. Subalpine fir, Engelmann spruce, and blue spruce trees occur along the stream channel and throughout the valley floor, and commonly display a relatively open and often patchy canopy cover. Common shrubs and herbs associated with the Evergreen Forest type include alder, Drummond willow, honeysuckle, red-osier dogwood, currents, bittercress, cow parsnip.

Deciduous Riparian Forests: This subclass is dominated by broad-leaved deciduous trees at elevations of about 6500 to 9500 feet. Cottonwood and box elder trees occur along the stream channel and throughout the valley floor, and commonly display a relatively open and often patchy canopy cover. Common shrubs and herbs associated with the Deciduous Forest type include alder, honeysuckle, red-osier dogwood, maple, rose, serviceberry, gooseberry, skunkbrush, hawthorn, coyote willow, bluebells, Fendler cowbane, brook saxifrage, twisted stalk, horsetail, arrowleaf groundsel, goldenglow, geranium, and bluejoint reedgrass. , Bebb willow, strapleaf willow, mountain willow, whiplash willow, Drummond willow, Geyer willow, false Solomon seal, bittercress, cow parsnip, bluebells, Fendler cowbane, brook saxifrage, horsetail, goldenglow, geranium, and Kentucky bluegrass.

Mixed Evergreen-Deciduous Riparian Forests: This subclass is dominated by evergreen and broad-leaved deciduous trees at elevations from about 7500 to 9500 feet. Narrowleaf cottonwood, subalpine fir, Engelmann spruce, and blue spruce trees occur along the stream channel and throughout the valley floor, and commonly display a relatively open and often patchy canopy cover. Douglas-fir usually occurs in the valley floor away from the channel. Common shrubs and herbs associated with the Mixed Evergreen-Deciduous Forest type are described above for the Deciduous and Evergreen Forests.

Deciduous Riparian Shrublands: This subclass is dominated by deciduous shrubs. Alder, river birch, saltcedar, and the willows are found along the stream channel and usually display high cover in the alliances named after them. The other shrubs and Bebb willow occur throughout the valley floor (including along the channel) and display low to high cover in the alliances named after them. The Bog Birch, Planeleaf Willow, Wolf Willow, Shortfruit Willow, and Geyer Willow alliances are wetland types that occur at elevations above about 9500 feet. Drummond Willow, Mountain Willow, and Alder alliances occur at mid to high elevations above about 8000 feet. The Bebb, Strapleaf, and Coyote Willow alliances, and Red osier Dogwood, Hawthorn, Saltcedar, and River Birch alliances occur at elevations from about 7000 to 9000 feet. Graminoids and Forbs tend to be found in herb-dominated patches between shrubs in the community types named after them.

Common herbs associated with the Deciduous Shrubland type include marsh marigold, goldenglow, geranium, cow parsnip, bluebells, bedstraw, bittercress, Fendler cowbane, arrowleaf groundsel, brook saxifrage, pink stonecrop, strawberry, osha, elephant head, king's crown, water sedge, beaked sedge, tufted hairgrass, mannagrass, Kentucky bluegrass, bluejoint reedgrass, and horsetail.

Perennial Forbs: This subclass is dominated by forbs at elevations above about 9000 feet. Common herbs associated with this type include marsh marigold, bluebells, bittercress, Fendler cowbane, arrowleaf groundsel, brook saxifrage, pink stonecrop, tall larkspur, elephant head, star gentian, willowherb, water sedge, and tufted hairgrass.

Perennial Graminoids: This subclass is dominated by grasses, sedges and rushes. Common herbs associated with this type include water sedge, beaked sedge, smallwing sedge, tufted hairgrass, reed canarygrass, threesquare bulrush, softstem bulrush, broadleaf cattail, common spikerush, baltic rush, wooly sedge, elephant head, brook saxifrage, willowherb, arrowleaf groundsel, and bluejoint reedgrass.

Past Disturbance of the Riparian areas and Wetland Type

Riparian areas and wetlands have changed dramatically during the last century and a half in the southwestern United States due to human impacts (Blair et al. 1996). In New Mexico, riparian vegetation has probably been impacted more by human activities than any other type of vegetation (Dick-Peddie 1993). Human impacts that have occurred in the SJPA include urbanization, agriculture, logging, livestock grazing, mining, recreation, roads, dams, diversions, and the introduction of exotic species.

Riparian areas and wetlands in the San Juan Mountains are lands that were first settled and developed for townsites, agriculture, and roads, resulting in vast lands being cleared of vegetation and modified beyond recognition (Blair et al. 1996). Riparian area trees in the Southwestern U.S. were initially cut for fuel and shelter, and also cut to clear land for agriculture and settlement (Dahms and Geils 1997). Settlement and agriculture occurred along riparian areas near towns throughout the SJPA.

The construction of dams, reservoirs, and diversions not only cleared the vegetation and modified the topography, but also decreased and regulated water flow, blocked movements of aquatic organisms, and changed the natural geomorphic stream processes of channel formation and erosion/deposition. The associated drop in water tables and lack of flooding have resulted in significant changes to the abundance, distribution, and reproductive mechanisms (germination and seedling survival) of native riparian area/wetland plant species, particularly willows and cottonwoods (Rood and Mahoney 1993, Glinski 1977, Brady et al. 1985). Regulated stream flow is thought to be the most important factor contributing to the decline of cottonwood and willow riparian ecosystems (Rood and Heinze-Milne 1989, Fenner et al. 1985, Rood and Mahoney 1990, Brown et al. 1977, Crawford et al. 1993). Dams and reservoirs are present throughout the SJPA and include Vallecito, Lemon, Williams, and Electra.

The effects of livestock grazing on riparian areas/wetlands are well documented in the literature (Gifford 1981, Knopf and Cannon 1982, Kauffman and Krueger 1984, Platts and Raleigh 1984, Skovlin 1984, Clary and Webster 1989, Clary and Medin 1990, Schulz and Leininger 1990, Kovalchik and Elmore 1992). Platts summarized this body of information as follows, "It is clear from the literature that improper livestock grazing can affect the riparian-stream habitat by eliminating riparian vegetation, widening stream channels, causing channel aggradation through increased sediment transport, changing stream bank morphology, and lowering surrounding water tables". The reduction or elimination of woody riparian species by livestock is particularly detrimental to riparian areas/wetlands that are dependant on those species to stabilize banks and hold those systems together. Even when environmental conditions are conducive to cottonwood and willow regeneration, stands of cottonwood and willow can be destroyed by excessive grazing by livestock (Glinski 1977, Carothers 1977, Kauffman et al. 1983). Extensive cattle and sheep livestock grazing, and the associated adverse impacts, began in the SJPA around the 1870s, when Anglo settlers arrived in increasing numbers (Savage 1991). Heavy grazing throughout the Section continued into the 20th century, and much of this unregulated grazing occurred in forests and meadows in the national forests (DuBois 1903).

Logging affects riparian areas/wetlands by removing vegetation, which decreases the interception and infiltration of water, and increases runoff. With increased surface runoff, erosion increases and more sediment is dumped into stream systems. Roads, built and maintained for logging activities, are the dominant source of soil erosion and stream sediment in forest environments (Swank and Crossley 1988, Reid 1981). Sediment fills gravel and cobble beds that are sites of attachment for insect eggs, larvae, macroinvertebrates, and fish eggs (Noss and Cooperrider 1994). This greatly diminishes critical fish habitat. Logging and the associated road building have been extensive on public and private lands throughout the SJPA.

The reduction or elimination of native riparian area/wetland plant species in the SJPA due to human impacts described above, has allowed exotic species such as Russian olive and salt cedar (tamarisk) to become established and highly competitive in these ecosystems. The diversity of Southwestern riparian ecosystems has been greatly reduced by these two plants (Dick-Peddie 1993). Salt cedar and Russian olive apparently outcompete native cottonwoods and willows, limiting the regeneration success of native woody plants (Finch et al. 1995).

Historic Range of Variability (HRV) of Riparian areas and Wetlands

Historic information of riparian areas and wetlands during the reference period in the SJPA is limited. The reference period is defined as the period of indigenous settlement, from about 1500 until the mid to late 1800s, prior to major impacts of Euroamerican settlement (Romme et al. 1998). Reference sites at low to middle elevations that could have been used to represent reference-period conditions are rare due to human impacts to these ecosystems since the reference period. The following brief descriptions are based on the best reference sites available, classification and inventory plots, extensive field reconnaissance within the SJPA, and knowledge of human impacts since the reference period.

Evergreen Forests - These forests have probably been the least affected by human impacts due to their more remote locations and the limited access to them. During the reference period they looked much like they do today, with subalpine fir, Engelmann spruce, and blue spruce displaying a relatively open and patchy canopy cover along the stream channel and throughout the valley floor.

Deciduous Forests and Mixed Evergreen-Deciduous Forests - These forests have probably been the most affected by human impacts due to their easy access and their desirability for settlement, agriculture, livestock grazing, and roads (as described below). During the reference period narrowleaf and Rio Grande cottonwood trees, and willows were abundant and well distributed along perennial and intermittent streams, much more so than what is displayed in most places today. Palatable herbs, particularly graminoids, were also more abundant and well distributed than what is displayed in most places today.

Deciduous Shrublands - These shrublands have been significantly affected by human impacts due to their easy access and their desirability for settlement, agriculture, livestock grazing, and roads. During the reference period willows were abundant and well distributed along perennial and intermittent streams, much more so than what is displayed in most places today. Palatable herbs, particularly graminoids, were also more abundant and well distributed than what is displayed in most places today.

Perennial Forbs and Perennial Graminoids - During the reference period these wetlands looked much like they do today, but there are probably less acres of them due to human impacts associated with their desirability for settlement, agriculture, livestock grazing, and roads.

Management Implications for Riparian areas and Wetlands

The management of riparian areas/wetlands needs to be considered at the subregion and landscape scales, as well as the more site-specific land unit scale. Managers must consider the dynamics, activities, and ecological characteristics of entire watersheds since activities and actions that occur throughout them can have significant affects on the associated riparian areas/wetlands.

New resource information and new technologies in data collection, modeling, information transfer (internet), and Geographic Information Systems (GIS) are available to help manage riparian areas/wetlands for long-term viability and sustainability. Continued efforts to develop and update ecosystem classification systems, soil and ecological inventories, and GIS applications will help people to better recognize and understand the ecological components and relationships associated with these ecosystems. New research efforts are needed to understand the complexity of riparian areas/wetlands, and to answer questions about historic conditions, range of natural variability, disturbance agents, wildlife habitat requirements, and restoration. The development and use of broad-scale assessments, including watershed and terrestrial ecological assessments, can provide an understanding of the interrelationships between human land-use, species and ecosystem diversity, ecosystem health, and the inherent disturbance processes and biophysical capabilities of the land.

From an ecological perspective, some of the changes that have occurred to many of the riparian area/wetlands in the SJPA since the reference period can be regarded as undesirable, since they have altered the composition, structure, and function of these ecosystems, and because the changes have resulted in a decrease in native biological diversity (Mullen 1992).

Livestock grazing has had a significant adverse effect on the riparian areas/wetlands since the reference period, and since it appears that grazing will continue to be a major, long-term land use within the SJPA, riparian areas/wetlands will continue to be affected. Cattle are attracted to wet sites, and have a tendency to stay in them even when not actively feeding (Myers 1989). There is no single or simple solution on how to graze livestock in riparian areas/wetlands in ecologically and economically feasible ways. What is required is a carefully considered prescription drawn up to address the conditions at a specific site with its unique circumstances and desired objectives (Anderson 1993, Buckhouse and Elmore

1993). New and improved livestock grazing management practices and methods, continued research, and a new commitment to monitoring are needed to implement appropriate grazing systems, grazing times and durations, stocking levels, utilization standards, and distribution patterns. If unsustainable grazing practices (continuous grazing, overstocking, overutilization) occur, the ecological condition of the riparian areas/wetlands will decline, resulting in further loss of ecosystem integrity and biological diversity.

Restoration needs and priorities for degraded riparian areas/wetlands in the SJPA should be identified. Social and ecological concerns need to be considered, as well as economic constraints and the uncertainties associated with restoration efforts. It may be more appropriate and cost-effective to focus restoration efforts on sites that are at risk but only moderately degraded, rather than targeting the most severely deteriorated ones. The Proper Functioning Condition (PFC) method for assessing the condition of riparian areas/wetlands is a good tool for identifying restoration needs and priorities (Prichard 1993). Restoration needs include the elimination of exotic species (Russian olive and salt cedar/tamarisk), streambank stabilization, and the revegetation of native riparian area/wetland species, particularly species like cottonwoods, willows, and sedges that are so important to the functioning of these ecosystems.

The solution to riparian area problems is the development of informed people, through education and the formation of partnerships (Elmore 2000). This is important due to the interconnectedness of riparian areas/wetlands since the water associated with them moves in continuous corridors through landscapes that contain multiple land ownerships. Impacts at a riparian area site can have a major affect on downstream riparian ecosystems and landowners. Partners, including private and federal landowners, academia, conservation groups, and concerned citizens, should be included in the development of goals and objectives, and should be involved in the decision-making process (Elmore 2000).

Adaptive management is necessary to protect and preserve riparian areas/wetlands in the SJPA. It encompasses both deliberate experimentation to gain new knowledge, and the ongoing process of using monitoring and inventory information to assess the effects of management actions (Dahms and Geils 1997). Adaptive management involves being explicit about expected outcomes, designing methods to measure responses, collecting and analyzing information to compare expectations and actual outcomes, learning from comparisons, and changing actions and plans accordingly (USDA Forest Service 1995).
The importance and ecological values of the riparian areas/wetlands in the SJPA that are currently in good ecological condition must be recognized and properly managed for. When activities (livestock grazing, recreation, land development) are considered for these lands, appropriate management and monitoring plans need to be developed that will ensure the ecological integrity and sustainability of these ecosystems. An important management objective for riparian areas/wetlands must be to perpetuate all native species, and to maintain or increase the abundance and distribution of key species, including cottonwoods, willows, and sedges. Other objectives include keeping exotic plants out of these ecosystems and maintaining soil productivity. Management activities that cannot meet these objectives should not be considered.

Reserves and protected areas that display relatively undisturbed riparian area/wetland sites need to be established for research, monitoring, biodiversity protection, and as reference areas to assess the affects of management activities.

Semi-Desert Grassland Type

Composition, Structure, and Distribution of Semi-Desert Grasslands

The Semi-Desert Grassland type within the SJPA is an herbaceous type that occurs on hills, mesas, alluvial flats, terraces, and valley floors at elevations ranging from about 4500 to 7000 feet. It is a grass-dominated type where grasses make up 25% or more of the canopy, or grasses make up the most canopy cover of any life form (IRI-CVU). Shrubs make up less than 25% of the canopy. It occurs in association with semi-desert shrubland, pinyon-juniper woodland, and sagebrush shrubland types. It occurs mostly on well-drained soils in the semi-arid climate zone. Soils are often sandy or loam-textured and derived from sedimentary parent materials, but are quite variable and may include fine-textured soils derived from other sources. Mean annual precipitation ranges from about 7-16 inches. There are about 301,538 acres of this type within the SJPA, which is about 8%. Most of this type is located on BLM lands in the west portion of the SJPA.

In good condition, these grasslands are characterized by moderate to high cover (20-50%) of grasses, with forbs and bare soil occupying the space between grasses. Terrestrial cryptogams composed of cyanobacteria, algae, lichen, moss, fungi, or liverwort are often significant components in semi-desert grasslands forming crusts on the ground surface that contribute to soil stability, nutrient supply (organic matter and nitrogen), and biodiversity (Ladyman & Muldavin 1996).

This type is described as Plains and Great Basin grasslands (Brown 1982), Plains-Mesa grasslands (Dick-Peddie 1993), Great Basin grasslands (Robbie 2004), Colorado Plateau grasslands (Robbie 2004), semi-arid grassland (Abruzzi et. al), semidesert loam and semidesert sandy loam (NRCS), loamy saltdesert (NRCS), Inter-Mountain Basins Semi-Desert Grassland (Nature Serve), Great Basin foothill-Piedmont grasslands (NM GAP), and Desert Grassland (Utah GAP).

Common native grass species include needle and thread, Indian ricegrass, galleta, bottlebrush squirreltail, blue grama, purple threeawn, sand dropseed, and alkali sacaton. Native shrubs may include basin big sagebrush, shadscale saltbush, winterfat, plains pricklypear, broom snakeweed, and fourwing saltbush. Common native forbs include scarlet globemallow, woolly plantain, sego lily, tall mountain larkspur, western tansymustard, longleaf phlox, woolly locoweed, Wright's bird's beak, cleftleaf wildheliotrope, and tufted evening-primrose. Common non-native species include cheatgrass. The dominant perennial grasses and shrubs within this type are all very drought-resistant.

Disturbance in Semi-Desert Grasslands

Changes in precipitation, grazing intensity, and fire suppression are considered the most important factors influencing vegetation changes in semi-desert grasslands (Rodriguez Iglesias and Kothmann 1997).

The invasion of semi-desert grasslands by shrubs and trees since Anglo settlement is at least in part due to livestock grazing (Martin 1975, Brown 1982).
Areas that have been affected by historical grazing have experienced a shift in species composition towards grazing increasers and non-native species similar to the semi-desert shrubland vegetation type. Some of these sites have been converted to semi-desert shrublands, as livestock grazing has decreased grass cover and increased shrub cover.

Historic Range of Variability (HRV) of Semi-Desert Grasslands

Southwestern grasslands today (including semi-desert grasslands) share general differences from their pre-Euro-American settlement conditions, as historically they were more diverse in plant and animal species composition, more productive, more resilient, and better able to absorb the impact of disturbances (Fletcher and Robbie 2004).
Fire in the Colorado Plateau grasslands (semi-desert grasslands) was of low intensity but adequate to keep shrubs from expanding (Fletcher and Robbie 2004). Fire most often occurred in these sites, when adjacent shrublands burned. Fires were typically mixed with an average fire return interval of 37 years and stand replacement with an average fire return interval of 75 years (Landfire Model, 2005).

Management Implications for Semi-Desert Grasslands

The semi-desert grassland system is very vulnerable to invasion by exotic species, particularly cheatgrass. Although frequent fires in grasslands may have been common historically, the introduction of cheatgrass has altered the dynamics of the system, and fire often results in cheatgrass dominance. Once overtaken by cheatgrass, more frequent fires are encouraged by the dry flammable material, leading to further domination by cheatgrass. Even a few cheatgrass plants in a stand will produce enough seed to dominate the stand within a few years after fire (Monsen, pers. comm.)

Change in the composition of native grasslands is a concern to land managers and others. Heavy grazing of grasslands that occurred in the late 1800's and early in the twentieth century (and continues today in many areas), favors the increase of shrubs over grasses. Since the native bunchgrasses of the west, unlike the rhizomatous grasses of the Great Plains, did not evolve with large ungulate grazers like bison, they are not adapted to survive heavy grazing. Grasses produce organic matter which provides the habitat for forb, grass and shrub seedling survival. Without the perennial grass understory, the organic matter and nutrients in the soil oxidize and blow away (Dreman 2005). Determining the pre-settlement extent of grasslands and changes in the relative cover of grasses and shrubs has been attempted by comparing historic written accounts and photographs with present vegetation. It is thought that the perennial grasses (*Hesperostipa comata, Bouteloua* *gracilis* and *Pleuraphis jamesii)* were generally dominant in most areas (Romme et al. 2003). In heavily disturbed areas, grazing tolerant species such as snakeweed *(Gutierrezia sarothrae)*, winterfat *(Krascheninnikovia lanata)* and prickly pear *(Opuntia polyacantha* and *O. fragilis)* have replaced much of the grass.

Shrubs will be killed by fire, maintaining an earlier successional stage. Most shrubs (sagebrush and saltbushes) must reestablish from seed, while herbaceous species resprout. Prairie dogs may be also be a key disturbance agent in these ecosystems (Romme et al. 2003). These rodents typically remove all vegetation from the vicinity of their extensive burrows or "towns" in order to detect predators.

Additional threats in the SJPA are increased oil and gas exploration and drilling, especially in Big Gypsum Valley, where some of the most extensive semi-desert grasslands of the SJPA are located. In addition to direct disturbance and replacement, the risks of introducing more exotic species are high.

Semi-Desert Shrubland Type

Composition, Structure, and Distribution of Semi-Desert Shrublands

The semi-desert shrubland type within the SJPA occurs on hills, mesas, alluvial flats, terraces, and valley floors at elevations ranging from about 4500 to 7000 feet. It is a shrub-dominated type where shrubs make up 25% or more of the canopy, or shrubs make up the most canopy cover of any life form (IRI-CVU). It occurs in the semi-arid climate zone in association with semi-desert grassland, pinyon-juniper woodland, and sagebrush shrubland types. Soils are mostly well-drained, but some sites near drainages have a higher water table and flood intermittently. Soils may be calcareous or sodic, and fine or coarse-textured. The soil surface often has a white salt crust. Mean annual precipitation ranges from about 7 to16 inches. There are about 95,380 acres of this type within the SJPA, most of which is located on BLM lands in the west portion of the SJPA.

Stands are characterized by moderate to high cover (20-50%) of shrubs. Herbaceous cover, which is highly variable, and bare soil occupy the space between shrubs. Terrestrial cryptogams composed of cyanobacteria, algae, lichen, moss, fungi, or liverwort are often significant components in semi-desert shrublands forming crusts on the ground surface that contribute to soil stability, nutrient supply (organic matter and nitrogen), and biodiversity (Ladyman & Muldavin 1996).

This type is described as Great Basin Desert Scrub (Brown 1982, Dick-Peddie 1993), Inter-Mountain Basins Semi-Desert Shrub-Steppe (Nature Serve), Inter-Mountain Basins Mixed Salt Desert Scrub (Nature Serve), Inter-Mountain Basins Greasewood Flat (Nature Serve), Greasewood Fans & Flats (Colorado GAP), alkali bottom (NRCS), alkali flat and slopes (NRCS), salt flat (NRCS), clayey and loamy saltdesert (NRCS), and saltdesert breaks (NRCS).

Common native shrubs in the semi-desert shrubland type include shadscale saltbush, winterfat, fourwing saltbush, plains pricklypear, broom snakeweed, rubber rabbitbrush, spiny hopsage, and basin big sagebrush. These shrublands are sometimes dominated by greasewood, particularly on valley floors and other low-lying landscape positions. Common native grass species include needle and thread, Indian ricegrass, galleta, bottlebrush squirreltail, western wheatgrass, blue grama, purple threeawn, sand dropseed, alkali muhly, and alkali sacaton. Common native forbs include scarlet globemallow, woolly plantain, sego lily, tall mountain larkspur, western tansymustard, longleaf phlox, woolly locoweed, Wright's bird's beak, cleftleaf wildheliotrope, and tufted evening-primrose. Common non-native species include cheatgrass. The dominant perennial grasses and shrubs within this type are all very drought-resistant.

Disturbance in Semi-Desert Shrublands

Livestock grazing has had the most impact on this vegetation type since Euro-American settlement, altering the vegetation type towards non-palatable species.

There is little evidence of fire in semi-desert shrublands during pre-settlement times. Fires historically burned within these communities but fire intervals, along with general HRV conditions, are poorly known (Romme et al. 2003). Fires in recent years have emerged in this type because of the establishment of non-native annuals, primarily cheatgrass (*Bromus tectorum).*

Research from the USFS Desert Experimental Range indicates shifting mosaics of semi-desert shrubland communities based on drought, flooding, and insect outbreaks.

Historic Range of Variability (HRV) of Semi-Desert Shrublands

More studies are needed within the SJPA to establish HRV for this vegetation type.

Management Implications for Semi-Desert Shrublands

Naturally sparse plant cover and fine-grained saline soils make this type especially vulnerable to water and wind erosion (Blaisdell and Holmgren 1984). When trampling or other disturbances weaken the soil crust, soil particles are easily moved by wind or water. In addition to livestock grazing, disturbances such as construction of transportation corridors, mining, seismic exploration and drilling for oil and gas, and recreation have caused the deterioration of much of this type. Depleted salt-desert shrub communities (semi-desert shrublands) are very slow to recover, even under good management. Revegetation is difficult (Blaisdell and Holmgren 1984).

Invasion by non-native annual plants, especially cheatgrass *(Bromus tectorum),* has replaced many desirable native species in the west. Cheatgrass has invaded millions of acres throughout the United States in sagebrush and salt desert shrub communities. It can grow from seed following fire, and because it dries out early in the summer, encourages fires that destroy perennial shrubs and grasses.

A great threat to these shrublands is oil and gas extraction, which has increased dramatically in the SJPA in 2005. In addition to direct disturbance and replacement, the risks of introducing more exotic species are high.

Scientific Names Of Plants

(mostly Weber 2001) in alphabetical order by common name, sorted by life form

Trees -

- alder (*Alnus incana ssp tenufolia),*
- aspen (*Populus tremuloides),*
- blue spruce *(Picea pungens),*
- box elder *(Acer negundo),*
- bristlecone pine *(Pinus aristata),*
- Douglas-fir *(Pseudotsuga menziesii),*
- Engelmann spruce *(Picea engelmannii),*
- limber pine *(Pinus flexilis),*
- lodgepole pine *(Pinus contorta),*
- narrowleaf cottonwood *(Populus angustifolia),*
- pinyon pine *(Pinus edulis),*
- ponderosa pine *(Pinus ponderosa),*
- Rio Grande cottonwood *(Populus deltoids ssp wislizeni),*
- Rocky Mountain juniper *(Juniperus scopulorum),*
- Southwestern white pine *(Pinus strobiformis),*
- *subalpine fir *(Abies lasiocarpa or bifolia),*
- Utah juniper *(Juniperus osteosperma),* and
- white fir *(Abies concolor).*

*Due to their ecological similarities and the difficulty of field identification subalpine fir (*Abies lasiocarpa or bifolia)* and corkbark fir *(Abies arizonica),* both of which occur on the SJNF, are grouped together in this document and referred to as subalpine fir.

Shrubs

- alpine willow *(Salix arctica),*
- banana yucca *(Yucca baccata),*
- basin big sagebrush *(Seriphidium tridentatum ssp tridentatum),*
- Bebb willow *(Salix bebbiana),*
- bitterbrush *(Purshia tridentata),*
- black sagebrush *(Seriphidium novum),*
- blueberry *(Vaccinium myrtillus and scoparium),*
- bluestem willow *(Salix irrorata),*
- buckbrush *(Ceanothus fendleri),*
- bog birch *(Betula glandulosa),*
- buffaloberry *(Shepherdia canadensis),*
- chokecherry *(Prunus virginiana),*
- common juniper *(Juniperus communis),*
- coyote willow *(Salix exigua),*
- currant *(Ribes montigenum),*
- Drummond willow *(Salix drummondiana),*
- elderberry *(Sambucus microbotrys),*
- fendlerbush *(Fendlera rupicola),*
- fourwing saltbush *(Atriplex canescens),*
- Gambel oak *(Quercus gambelii),*
- Geyer willow *(Salix geyeriana),*
- gray willow *(Salix glauca),*
- greasewood *(Sarcobatus vermiculatus),*
- Green's rabbitbrush *(Chrysothamnus greenei),*
- hawthorn *(Crataegus rivularis),*
- honeysuckle *(Distegia involucrata),*
- kinnikinnik *(Arctostaphylos uva-ursi),*
- longflower rabbitbr*ush (Chrysothamnus depressus),*
- maple *(Acer glabrum),*
- Mormon tea *(Ephedra viridis),*
- mountain mahogany *(Cercocarpus montanus),*
- mountain ash *(Sorbus scopulina),*
- mountain lover *(Paxistima myrsinites),*
- mountain sagebrush *(Seriphidium vaseyanum),*
- mountain willow *(Salix monticola),*
- net willow *(Salix reticulata),*
- Oregon grape *(Mahonia repens),*
- plains prickly pear *(Opuntia polyacantha),*
- planeleaf willow *(Salix planifolia),*
- raspberry *(Rubus idaeus),*
- rubber rabbitbrush *(Chrysothamnus nauseous),*
- red osier dogwood *(Swida sericea),*
- river birch *(Betula fontinalis),*

- rose *(Rosa woodsii),*
- Russian olive *(Elaeagnus angustifolia),*
- saltcedar *(Tamarix ramosissima),*
- shadscale saltbush *(Atriplex confertifolia),*
- silver sagebrush *(Seriphidium canum),*
- snakeweed *(Gutierrezia sarothrae),*
- serviceberry *(Amelanchier alnifolia),*
- shortfruit willow *(Salix brachycarpa),*
- shrubby cinquefoil *(Pentaphylloides floribunda),*
- silver sagebrush *(Artemisia cana ssp viscidula),*
- silverberry *(Shepherdia argentea),*
- skunkbrush *(Rhus trilobata),*
- snakeweed *(Gutierrezia sarothrae),*
- snowberry *(Symphoricarpos rotundifolius),*
- spiny hopsage *(Atriplex grayi),*
- squawapple *(Peraphyllum ramosissimun),*
- strapleaf willow *(Salix eriocephala var. ligulifolia),*
- thimbleberry *(Rubacer parviflorum),*
- winterfat *(Krascheninnikovia lanata),*
- wolf willow *(Salix wolfii),* and
- Wyoming big sagebrush *(Seriphidium tridentatum ssp wyomingensis).*

Forbs

- alpine avens (*Geum rossii),*
- alpine campion *(Silene acaulis),*
- alpine nodding groun*dsel (Ligularia soldanella),*
- alpine phlox *(Phlox caespitosa),*
- alpine sandwort *(Lidia obtusiloba),*
- American vetch (*Vicia americana),*
- arnica *(Arnica latifolia and cordifolia),*
- arrowleaf groundsel *(Senecio triangularis),*
- avalanche lily *(Erythronium grandiflorum),*
- baneberry *(Actaea rubra),*
- beardstongue *(Penstemon barbatus),*
- bedstraw *(Galium triflorum),*
- bittercress *(Cardamine cordifolia),*
- bluebells *(Mertensia ciliata and fransiscana),*
- brackenfern *(Pteridium aquilinum),*
- broadleaf cattail *(Typha latifolia),*
- brook saxifrage *(Micranthes odontoloma),*
- buckwheat *(Eriogonum umbellatum),*
- butterweed *(Packera neomexicana),*
- cinquefoil *(Potentilla hippiana),*
- cleftleaf wildheliotrope *(Phacelia crenulata),*
- clematis *(Atragene columbiana),*
- columbine *(Aquilegia caerulea and elegantula),*
- cow parsnip *(Heracleum lanatum),*

- daisy *(Erigeron formosissimus),*
- dandelion *(Taraxacum officinale),*
- death camas *(Anticlea elegans),*
- deervetch *(Lotus wrightii),*
- desert sandwort *(Eremogone fendleri),*
- dwarf clover *(Trifolium nanum),*
- elephanthead *(Pedicularis groenlandica),*
- fairy slipper *(Calypso bulbosa),*
- false Solomon's seal *(Maianthemum amplexicaule and stellatum),*
- Fendler cowbane *(Oxypolis fendleri),*
- fireweed *(Chamerion danielsii),*
- fleabane *(Erigeron eximius),*
- forget-me-not *(Myosotis asiatica),*
- fringed sage *(Artemisia frigida),*
- geranium *(Geranium caespitosum),*
- globemallow *(Spharalcea coccinea),*
- goldeneye *(Heliomeris multiflora),*
- goldenglow *(Rudbeckia ampla),*
- goldenrod *(Solidago simplex),*
- goldenweed *(Oreochrysum parryi),*
- Gray lousewort *(Pedicularis grayi),*
- hairy goldenaster *(Heterotheca villosa),*
- harebell *(Campanula rotundifolia),*
- Hornemann's willowherb *(Epilobium hornemannii),*
- jacobs ladder *(Polemonium pulcherrimum),*
- kings crown *(Rhodiola integrifolia),*
- locoweed *(Oxytropis),*
- longleaf phlox *(Phlox longifolia),*
- lousewort *(Pedicularis racemosa and centranthera),*
- lupine *(Lupinus argenteus),* marsh marigold *(Caltha leptosepala),*
- meadowrue *(Thalictrum fendleri),*
- milkvetch *(Astragalus),*
- mountain parsley *(Pseudocymopterus montanus),*
- mulesears *(Wyethia arizonica),*
- nailwort *(Paronychia pulvinata),*
- northern bedstraw *(Galium septentrionale),*
- one-sided wintergreen *(Orthilia secunda)*
- osha *(Ligusticum porteri),*
- owlclover *(Orthocarpus luteus),*
- paintbrush *(Castilleja integra),*
- Parry primrose *(Primula parryi),*
- pasqueflower *(Pulsatilla patens),*
- peavine *(Lathyrus leucanthus),*
- pipsissewa *(Chimaphila umbellata),*
- puccoon *(Lithospermum multiflorum),*
- pussytoes *(Antennaria rosea),*
- rattlesnake plantain *(Goodyera oblongifolia),*
- redroot buckwheat *(Eriogonum racemosum),*

- Richardson geranium *(Geranium richardsonii),*
- rose crown *(Clementsia rhodantha),*
- sagewort *(Artemisia ludoviciana and franseriodes),*
- sand aster *(Leucelene ericoides),*
- scree groundsel *(Senecio atratus),*
- sego lily *(Calochortus nuttallii),*
- shaggyleaf clover *(Trifolium dasyphyllum),*
- skyrocket *(Ipomopsis aggregata),*
- sneezeweed *(Dugaldia hoopesii),*
- star gentian *(Swertia perennis),*
- strawberry *(Fragaria vesca and virginiana),*
- sweet cicely *(Osmorhiza depauperata),*
- tall larkspur *(Delphinium barbeyi),*
- trailing fleabane *(Erigeron flagellaris),*
- tufted evening-primrose *(Oenothera caespitosa),*
- twisted stalk *(Streptopus amplexifolius),*
- violet *(Viola canadensis),*
- western tansymustard *(Descurainia pinnata),*
- wintergreen *(Pyrola minor),*
- wood nymph *(Moneses uniflora),*
- woolly locoweed *(Astragalus mollissimus),*
- woolly plantain *(Plantago patagonica),*
- Wright's bird's beak *(Cordylanthus wrightii),*
- and yarrow *(Achillea lanulosa).*

Graminoids

- alkali muhly (*muhlenbergia asperifolia),*
- alkali sacaton *(Sporobolus airoides),*
- Arizona fescue *(Festuca arizonica),*
- baltic rush *(Juncus balticus),*
- beaked sedge *(Carex utriculata),*
- bluejoint reedgrass *(Calamagrostis Canadensis),*
- blue grama *(Chondrosum gracile),*
- bottlebrush squirreltail *(Elymus elymoides),*
- cheatgrass *(Anisantha tectorum),*
- common spikerush *(Eleocharis palustris),*
- crested wheatgrass *(Agropyron cristatum),*
- curly sedge *(Carex rupestris),*
- elk sedge *(Carex geyeri),*
- elynoid sedge *(Carex elynoides),*
- *fringed* bromegrass *(Bromopsis canadensis),*
- galleta *(Hilaria jamesii),*
- horsetail *(Equisetum arvense and pratense),*
- Indian ricegrass *(Achnatherum hymenoides),*
- intermediate wheatgrass *(Thinopyrum intermedium),*
- junegrass *(Koeleria macrantha),*

- Kentucky bluegrass *(Poa pratensis),*
- kobresia *(Kobresia myosuroides),*
- little bluestem *(Schizachyrium scoparium),*
- mannagrass *(Glyceria striata),*
- mountain sedge *(Carex scopulorum),*
- mountain muhly *(Muhlenbergia montana),*
- mountain brome *(Bromopsis pumpelliana),*
- mountain ricegrass *(Oryzopsis asperifolia),*
- muttongrass *(Poa fendleriana),*
- needle-and-thread *(Hesperostipa comata),*
- needlegrass *(Achnatherum nelsonii),*
- orchardgrass *(Dactylis glomrata),*
- Parry oatgrass *(Danthonia parryi),*
- pine dropseed *(Blepharoneuron tricholepis),*
- purple threeawn *(aristida purpurea),*
- reed canarygrass *(Phalaris arundinacea),*
- rush *(Juncus),*
- ryegrass *(Elymus glauca),*
- sand dropseed *(Sporobolus cryptandrus),*
- sedge *(Carex),*
- side-oats grama *(Bouteloua curtipendula),*
- smallwing sedge *(Carex microptera),*
- smooth brome *(Bromopsis inermis),*
- softstem b*ulrush (Scirpus tabernaemontani),*
- spike muhly *(Muhlenbergia wrightii),*
- threesquare bulrush *(Scirpus pungens),*
- Thurber fescue *(Festuca thurberi),*
- timber oatgrass *(Danthonia intermedia),*
- *timothy (Phleum pratense),*
- trisetum *(Trisetum montanum),*
- tufted hairgrass *(Deschampsia cespitosa),*
- water sedge *(Carex aquatilis),*
- western wheatgrass *(Pascopyrum smithii),*
- wild rye *(Elymus canadensis),* and
- wooly sedge *(Carex lanuginosa).*

Chapter 6. Landscape Disturbances

Natural Disturbance Patterns (Module 4a)

Insects and Disease

Introduction

Most insects and diseases in forest cover types are natural components of the ecosystem and play important ecological roles. Tree mortality and other impacts of insects and diseases regulate forest vegetation composition, influence stand density and structure; provide wildlife habitat in dead and dying trees; and contribute nutrients to soils. Insects are also food for birds and other wildlife.

At low levels of infestation individual trees are weakened and killed, resulting in small scale changes affecting limited areas. Under certain conditions such as stand maturity, overcrowding, drought, blowdown, and poor site conditions, populations of forest insects and pathogens can increase, resulting in widespread mortality.

This assessment provides:
1. General descriptions of the organisms that impact forest cover types in and around the Grand Mesa, Uncomphagre, Gunnison (GMUG), and San Juan (SJ) National Forests (NF);

2. Current insect and disease conditions at the Subregional, Landscape, and Geographic Area (only for GMUG) scales; and 3. Vulnerability to future

insect and disease infestations at the Landscape and Geographic Area (only for GMUG) scales.

Dominant Insect/Pathogen Organisms by Cover Type

Aspen

Black target canker (caused by *Ceratocystis fimbriata* Ellis & Halst.) Sacc.), Cryptosphaeria canker (*Cryptosphaeria populina* Ces. & De Not), Cytospora canker (*Cytospora chrysosperma* (Pers.) Fr.), sooty bark canker (*Encoelia pruinosa* (Ellis & Everth.) Torkelson & Eckblad), aspen trunk rot (*Phellinus tremulae* (Bondartsev) Bondartsev & Borisov in Bondartsev) and white mottled rot (*Ganoderma applanatum* (Pers.) Pat.) are present in mature stands (R. Mask, personal communication). Cankers infect susceptible trees, enlarge, and eventually girdle and kill the infected tree (DeByle and Winokur 1985). Aspen forests over 110 years old have a high susceptibility to decline from pathogen-related mortality (USDA Forest Service 2001b). Root disease and stem decay can lead to root failure and windthrow. Root disease can dramatically reduce suckering following disturbance (DeByle and Winokur 1985; USDA Forest Service 2000e), and therefore stands with root disease may deteriorate to the point that regeneration is poor following disturbance. Cankers and root diseases can cause a stand to decline to the extent aspen clone regeneration is also inhibited. Two aspen leaf blights can be an aesthetic problem, especially during wet springs (e.g., 1995): ink spot (*Ciborinia whetzelii* Seaver) and *Marssonina populi*. Shepherd's crook (*Venturia macularis* (Fr.) Müll. & Arx.) (T. Eager, personal communication) affects aspen regeneration. Also occurring in southeast Colorado are aspen leaf rust (*Melampsora medusae* Thuem), aspen borer (*Saperda calcarata* Say), bronze aspen borer (*Agrilus liragus* Barter & Brown) and defoliators such as western tent caterpillar (*Malacosoma californicum* Packard) and large aspen tortrix (*Choristoneura conflictana* (Walker)). Defoliations do not usually result in tree mortality.

Douglas-fir/Mixed Conifer

Three insect species can affect Douglas-fir: Douglas-fir beetle (*Dendroctonus pseudotsugae* Hopk.), western spruce budworm (*Choristoneura occidentalis* Freeman), and fir engraver beetle (*Scolytus ventralis* LeConte)). Douglas-fir beetle infests and kills Douglas-fir. Typically beetles attack highly stressed trees, such as windfall, defoliated or fire-scorched trees. Douglas-fir beetle will also attack trees and stands previously defoliated by western spruce budworm (Harris et al 2001). If enough suitable host material is present, beetles can increase in stressed trees and infest nearby healthy trees (McMillin and Allen 2000).

The multi-layered stand structure of the Douglas-fir forests make them vulnerable to continued western spruce budworm attacks. Western spruce budworm feeds on new foliage. When an outbreak persists over years, complete defoliation results in decreased growth, tree deformity, top-kill, and death. A long-term attack of budworm can shift the tree species composition of the affected stand to non-host or less susceptible species (Dahms and Geils 1997).

On the San Juan NF, Douglas-fir is often part of a mixed conifer cover type in association with ponderosa pine and white fir in warm, dry environments (D. Dallison, personal communication). In this mixture, the Douglas-fir and white fir are susceptible to fir engraver beetle. Drought, root disease (see below), and severe defoliation by western spruce budworm exacerbate infestations of the fir engraver beetle. Fir engraver attacks are correlated with logging; when engraver populations are high, harvesting activities should be confined to mid-August to December (Edmonds et al. 2000).

Several diseases are found in mixed conifer stands: Armillaria (*Armillaria ostoyae* (Romagnesi) Herink), Annosus (*Heterobasidion annosum* (Fr.) Bref.), Douglas-fir dwarf mistletoe (*Arceuthobium douglasii* Englem.;see dwarf mistletoe description in Lodgepole Pine section below). Armillaria and Annosus are the primary root diseases found in the true firs of the mixed conifer stands. Both diseases cause structural root damage by decaying large, woody roots. Armillaria root disease is distributed worldwide with a huge host range (Edmonds et al. 2003). In western North American forests, *Armillaria ostoyae* is the dominant species (Shaw and Kile 1991), and, in the drier inland forests, *A. ostoyae* is an aggressive pathogen causing tree mortality, especially under drought conditions. Symptoms of Armillaria include reduced leader growth, tree decline, thinning foliage, and excess cone crops (Edmonds et al. 2000). Armillaria frequently forms complexes with bark beetles, impacting thousands of hectares of Douglas-fir, true fir and pine forests in the Subregion (see subalpine fir decline discussion).

Annosus root disease is widely distributed in the northern hemisphere and has many symptoms in common with Armillaria, such as reduced height growth, thinning foliage, distress cones, tree decline, etc (Edmonds et al. 2000). In resinous trees such as pines, Annosus tends to kill trees outright by girdling, whereas in spruces and firs, it causes root and butt rot leading to stem breakage or windthrow. Weakened trees in either case can become susceptible to bark beetle kill (Edmonds et al. 2000).

Red ring rot is a trunk rot most often occurring on Engelmann spruce in southern CO, though other conifer hosts are common as well. Sporophores are produced on living trees and may provide a general guide to the amount and distribution of internal decay within one tree species and region. Sporophores often develop adjacent to branch stubs and, in cases of extensive infection, may form more or less along the length of the bole. Decay can progress from the heartwood to the sapwood and cause tree death (source: Canadian Forest Service website search for Phellinus pini: http://www.cfl.scf.rncan.gc.ca/imfoc-idwcf/fichemaladie_e.asp?id=1000004l).
Indian paint fungus (*Echinodontium tinctorium* Ellis & Everh.) is less extensive but present in southwest Colorado (R. Mask personal communication).

Gambel Oak

Oak leaf roller (*Tortrix* spp.) and oak looper (*Lambdina fiscellaria somniaria* Hulst) are defoliators. Western oak looper larvae are about 1" long and light brown with black spots when mature. They move in a characteristic "looping" or measuring-worm fashion. The adult moth is yellowish to dark brown. Severe infestations are rare, but can result in almost total defoliation of trees. No permanent damage is done by occasional outbreaks (source: Washington State University Cooperative Extension website http://pep.wsu.edu/hortsense/scripts/query/displayProblem.asp?problemID=479&tableName=plant).

Lodgepole Pine

Three insect and disease organisms are common and destructive to lodgepole pine: dwarf mistletoe (*Arceuthobium americanum* Nutt.), mountain pine beetle (*Dendroctonus ponderosae* Hopkins), and pine engraver beetles (*Ips pini*). Dwarf mistletoes are parasitic plants without true leaves. They take up nutrients, water, and carbohydrates from their hosts, in this case lodgepole pine, which stresses the tree leading to reduced growth and sometimes mortality (Edmonds et al. 2000). Wildfire is the most important factor affecting the presence and distribution of dwarf mistletoe, particularly in lodgepole pine in the Rocky Mountains where wildfires tend to reestablish seral lodgepole pine stands, but fire behavior is patchy, allowing live infected trees to reinfect regenerating stands (Edmonds et al. 2000). See ponderosa pine section below for descriptions of mountain pine beetle and pine engraver beetles.

Piñon/Juniper

Five major insect and disease agents operate in the piñon-juniper woodlands cover type in Colorado: black stain root disease (caused by *Leptographium wagneri* var. *wageneri* (Kendrick) M.J. Wingfield), Armillaria root disease (see description in the Douglas-fir/Mixed Conifer section above), piñon pine dwarf mistletoe (*Arceuthobium divaricatum* Englem.; see dwarf mistletoe description in the Lodgepole Pine section above), and piñon Ips beetle (*Ips confuses* LeConte). Black stain root disease infects and kills piñon trees of all sizes and age classes except seedlings. Mortality centers usually begin with a single tree or small patch of trees and spread outwards, creating patches of dead trees of less than 0.5 acre. These outbreaks are usually self-limiting. The disease tends to be most severe in dense stands at higher elevations on good sites; relatively cool, moist and with deep soils (Landis and Helburg 1976). The pathogen is likely to persist in an area as long as susceptible hosts remain. Junipers are not affected. Black stain creates patchy mosaics of various successional stages of piñon-juniper and juniper woodlands.

The piñon Ips beetle is a bark beetle that kills predominantly large or medium sized piñon trees. At endemic levels infestations begin with a single tree or small patch of trees and spread outward creating semi-circular patches of dead trees. Dense stands are more likely to have endemic populations. This species can produce up to four broods a year and is capable of spreading quickly. As a result of ips beetle activities, a pattern of mixed successional stages of woodlands develops on the landscape.

Black stain and *Ips* often act in concert, resulting in piñon decline. Drought is a stress factor that greatly increases trees susceptibility to piñon decline, and large areas can be affected (Harris et al 2001). Clearing trees for development has also compounded this situation, with the dead and dying trees providing suitable brood material for the beetles and adding additional stresses on native forests (Harris et al. 2002). Vegetation management activities designed to reduce fuels under the National Fire Plan are also creating increased amounts of slash that provides suitable sites to Ips beetle colonization (Harris et al. 2002).

Less extensive but present in Colorado are juniper true mistletoe (*Phoradendron juniperinum* Viscaceae), Gymnosporangium stem rust of juniper (*Gymnosporangium* spp.), piñon pitch moth (*Vespamima* spp.), and piñon twig beetle (*Pityophthorus* spp.).

Ponderosa Pine

Three insects are common or particularly damaging to ponderosa pine: mountain pine beetle (*Dendroctonus ponderosae* Hopkins), western pine beetle (*D. brevicomis* LeConte), and pine engraver beetle (*Ips pini* Say). At endemic levels, mountain pine beetle kills occasional ponderosa pine trees or small patches, preferring trees greater than 8 inches in diameter, in more dense stands. These conditions increase the competition for light, nutrients, and moisture, stressing the trees. The stress increases the tree's susceptibility to successful insect attacks. Endemic level outbreaks usually affect areas less than 2.5 acres for about 10 years (Romme 2003). Similar outbreaks could occur every 20 to 25 years (Roy Mask personal communication). When stand conditions are favorable for increases in the beetle population, epidemic levels may result and tree mortality can be extensive. Epidemic outbreaks occur when many simultaneous smaller outbreaks grow together. Drought conditions further stress trees making them more susceptible to beetle attacks. As beetle populations increase, outbreaks can spread across the landscape affecting stands of any age and density. Schmid and Amman (1992) estimated that the intervals between successive epidemic mountain pine bark beetle outbreaks might range from 50 to 100 years. Western pine beetles kill overmature ponderosa pine, particularly those that are senescing, water-stressed, or infected with root rot. Drought increases susceptibility, and persistent drought can increase pine beetle populations to epidemic levels (Edmonds et al. 2000).

Pine engraver beetles breed in downed pine under latent population levels, but when populations increase they also colonize trees and often create outbreaks that are exacerbated by drought. *Ips pini* (hosts are ponderosa and lodgepole pine in this area) is the most widespread pine engraver beetle in North America. During latent population levels, scattered single trees are killed. "Drought-stressed ponderosa pine in the 60- to 80-year age class, averaging 15 to 20 cm diameter at breast height, are most susceptible to attack" (Edmonds 2000).

Other important disturbance factors in ponderosa pine include: Armillaria root disease, Annosus root disease, ponderosa pine dwarf mistletoe (*Arceuthobium vaginatum* (Willd.) J. Presl ssp. *cryptopodium*), red ring rot (see Douglas-fir/Mixed conifer section above), ponderosa pine needlecast (*Davisomycella ponderosae*), and comandra blister rust (*Cronartium comandrae* Peck). Armillaria (see description in Douglas-fir section above) is the most common root disease in the Subregion. Past forest management practices such as fire suppression, which has increased stem densities and therefore water stress, and partial cutting of ponderosa pine, which has encouraged growth of more susceptible tree species such as Douglas-fir and true firs, has increased the incidence of Armillaria (Edmonds et al. 2000). Annosus root disease (see description in Douglas-fir/Mixed conifer section above) is common throughout Region 3 (USDA Forest Service 2003) and on the San Juan NF (D. Dallison personal communication), functioning as both a pathogen and a saprophyte, with mortality rates highest in young regeneration (USDA Forest Service 2003). Parasitic dwarf mistletoe plants reduce growth, deform stems and branches, and sometimes cause the death of individual trees. Dwarf mistletoe is found throughout the ponderosa pine cover type, generally occurring in patches of several acres. Thinning and selective timber harvest of affected trees in the ponderosa pine stands keep infestations in check. Little specific information exists about the historic intensity and extent of these factors (Romme et al. 2003). Dwarf mistletoe also increases susceptibility of individual trees to beetles (Romme et al. 2003).

In 2000 and 2001 foliage discoloration in ponderosa pine was caused by ponderosa pine needlecast (*Davisomycella ponderosae*) (Worrall and Sullivan 2002). Trees of all sizes seem to be affected, with no more effects in any part of the crown over another. Discoloration was mostly in needles two years old or older. Discolored needles are eventually lost, leading to thin crowns. This fungal disturbance was related to warm, wet spring conditions in previous years followed by hot, dry conditions, and there has not been a long-term effect to most affected trees.

Comandra blister rust is a fungus that grows in the inner bark of lodgepole and ponderosa pines in the western U.S. The fungus causes growth reduction, stem deformity, and mortality. In addition, pines with stem cankers produce significantly fewer cones and seeds than healthy trees. The fungus has a complex life cycle. It infects hard pines but needs an alternate host, an unrelated plant, to spread from one pine to another. Epidemics in the Rocky Mountains are thought to occur after slow, moist warm fronts pass during late summer. This type of weather provides optimum conditions for the dispersal of basidiospores. Generally, however, outbreaks are localized. The rust attacks pines of all sizes and ages. Seedlings are the most susceptible and are usually killed within a few years by stem cankers. It usually takes much longer to kill older pines: 50-year-old cankers have been found on recendy killed trees (Johnson 1997).

Also present but less extensive are: roundheaded pine beetle (*Dendroctunus adjunctus* Blandford), red turpentine beetle (*Dendroctunus valens* LeConte), roundheaded borer, long horn beetle (*Cerambycidae*), and flatheaded borer, metallic wood borer (*Buprestidae*).

Spruce-fir

Spruce beetle (*Dendroctonus rufipennis* Kirby) is host specific to Engelmann and blue spruce on the GMUG NFs. At low population levels, spruce beetles are restricted to scattered trees weakened by disease, old age, or competition. Spruce beetle also feeds on windthrown trees. Large disturbances like fire or blowdown can allow beetle population buildups, which can then spread into unaffected areas, resulting in large scale mortality (Romme 2002). Spruce beetle epidemics can last approximately 15 years. Spruce beetle mortality is not easily seen in aerial surveys.

Western spruce budworm (*Choristoneura occidentalis* Freeman) is a defoliating insect, with the caterpillar stage eating new needles as they form. This insect affects Engelmann and blue spruce, subalpine fir and Douglas-fir on this landscape. Budworm outbreaks occur at intervals of 20 to 35 years, lasting about 10 to 12 years. Defoliations result in deformities, and increase susceptibility to other insects and pathogens. Repeated defoliations over 4 or 5 years can kill host trees.

Subalpine fir decline is a combination of western balsam bark beetle (*Dryocoetes confuses* Swaine) and Armillaria root disease (Johnson 2002). The root disease contributes to increases in the western balsam bark beetle population by weakening and killing trees through disease or associated windthrow, thus providing a susceptible host for the western balsam bark beetle. Western balsam bark beetle adults and larvae feed on the phloem layer of inner bark and can lead to tree death upon complete girdling (Edmonds et al. 2000).

True firs are host to the fir engraver beetle (*Scolytus ventralis* LeConte). Attacking pole-sized to fully mature trees, outbreaks are associated with root pathogens (i.e., *Armillaria* and *Annosum*), prolonged droughts, and logging (Edmonds et al. 2000). After drastic thinning from *S. ventralis*, residual fir stands are healthier with an enhanced ability to repel future attacks (Knight and Heikkenen 1980; Edmonds et al. 2000). Where engraver populations are high, harvesting activities should be confined between mid-August and December to prevent a buildup in the slash piles, and when poorer sites are harvested, reforestation should be done with pine seedlingss (Edmonds et al. 2000).
Armillaria root disease (see description in Douglas-fir/Mixed conifer section above) itself is a significant disturbance agent in both spruce and fir. Small patches of mortality are common in spruce/fir, especially in more mature stands. Gap-phase regeneration patches are created, establishing structural heterogeneity in the forest and increasing fuel loading. Since subalpine fir appears to be killed more quickly, especially in concert with western balsam bark beetle, the spruce component of the overstory becomes more dominant. Over many years, this patch-phase mortality increases the likelihood of catastrophic disturbance by spruce beetle and fire.

True firs are also vulnerable to Annosus root disease (see description in Douglas-fir/Mixed conifer section above).

Spruce broom rust (*Chrysomyxa arctosaphyli* Dietel), fir broom rust (*Melampsorella caryophyllacearum* (DC.) J. Schröt), and red ring rot (see description in Douglas-fir/Mixed Conifer section above) are also present in spruce/fir stands.

Subregional Scale

The information presented in this assessment relies on Forest Health Management documents (Johnson 1996, 1997; Harris et al. 2001, 2002; Harris 2003), personal communication with NF silviculturists, and aerial detection surveys for recent occurrences of insects and diseases, and computer modeling for predictions of future infestations.

Recent Insect and Disease Occurrences

Ground and aerial detection surveys are used to document damage and mortality in forests caused by insects and diseases (Rogers et al. 2001). Aerial surveys do not cover the entire subregion every year. Not all damage from forest insect and pathogen agents is visible during aerial surveys, so their results do not include all ongoing activity. For example, damage from spruce beetle is not easily visible from the air, because many affected trees are in the understory. Root disease effects are not seen from the air and on-the-ground surveys may be used to confirm the causal agent of suspected root disease mortality. Mistletoe infestations are impossible to see in aerial surveys.

Aggregated aerial survey data for 1994-2003 are displayed in Figure 6-1 for the Colorado and Mexico portion of the Subregional Scale. The most obvious and widespread insect and disease occurrences are of subalpine fir decline in the central to northern portion of the subregion, western spruce budworm in the mountainous areas of the central and southern portion of the subregion in Colorado and in the southeast portion of the subregion in New Mexico, and piñon decline from the Uncompahgre Plateau in southwest Colorado southward into Utah and New Mexico (Figure 6-1). Many other insects and disease occurrences are documented, but they are more limited in extent.

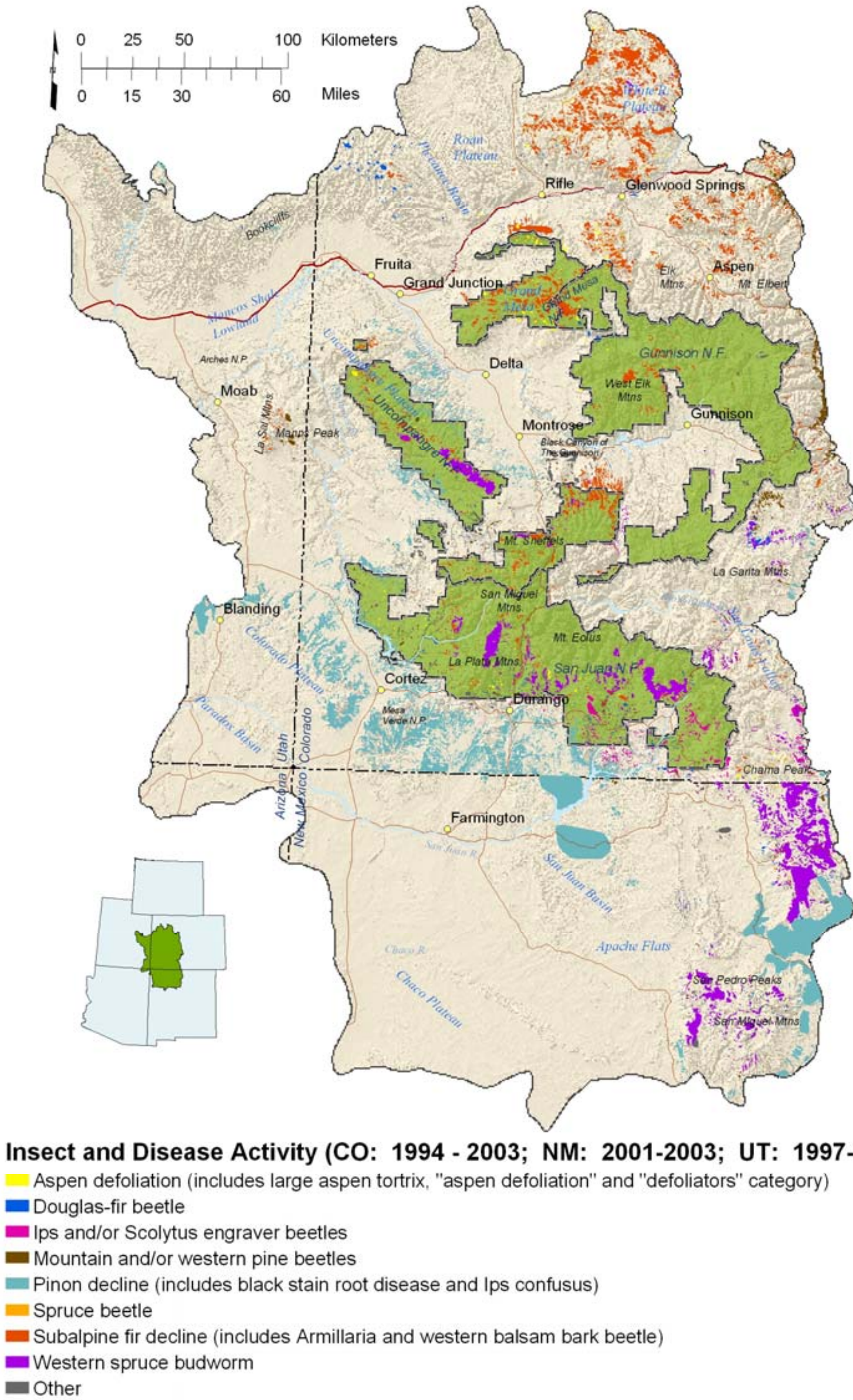


Figure 6-1 Recent insect and disease activity.
Recent insect and disease activity for Colorado and New Mexico portions of the Subregional Scale ascertained using aerial detection surveys from 1994-2003.

(Other includes: dead top, Diplodia blight, Douglas-fir needlecast, five-needle pine decline, foliage discoloration, forest tent caterpillar, Lophoderma needlecast, mistletoe, miscellaneous bark beetles, miscellaneous blight, miscellaneous mortality, oak leafroller, ponderosa pine needleminer, twig beetles, unknown, western tent caterpillar, and white pine blister rust.)

Within the Subregional Scale boundary, the most recent activity (since 1995) of several insect and disease organisms is noted below.

- Annosus root disease is common in Region 3 causing scattered mortality in spruce/fir, mixed conifer, and ponderosa pine (USDA Forest Service 2003). Scattered distribution of this disease in white fir of the mixed conifer cover type has been noted throughout southern Colorado (Johnson 1996, 1997; Harris et al. 2001; Harris et al. 2002).

- Armillaria root disease is the most common root disease in Regions 2 and 3 and is found throughout Region 4 as well (Johnson 1996, 1997; Harris et al. 2001; Harris et al. 2002; USDA Forest Service 2003). Armillaria is one of the key agents contributing to subalpine fir decline (along with western balsam bark beetle and other biotic and abiotic agents; see below), which accounts for the most tree mortality in the Rocky Mountain Region (Harris et al. 2001; USDA Forest Service 2003). In 1996, subalpine fir damage from Armillaria was particularly high at Aspen Mountain, Crested Butte, and Powderhorn Ski Areas and on the Uncompahgre Plateau at Ouray Springs, and white fir damage was high in the San Juan NF (Johnson 1997).

- The following aspen diseases are noted: Shephard's crook was common on aspen regeneration in many areas of Colorado in 1995 (Johnson 1996); black target canker, Cytospora, Cryptosphaeria, and sooty bark cankers and white mottled and aspen trunk rots were common throughout Colorado from 1995 through 2001 (Johnson 1996, 1997; Harris et al. 2001, 2002). Additionally in 1995, due to the cool, wet conditions in the late spring of 1995, aspen leaf blights created great aesthetic concern throughout Colorado (Johnson 1996) but declined significantly by 1997 (Harris et al. 2001).

- Black stain root disease continues as a problem for piñon in the southwestern corner of Colorado (Johnson 1996, 1997; Harris et al. 2001). In 1995, widespread mortality was detected on BLM land just south of Redvale, Colorado (Johnson 1996). In 1996, private land near Colona, Colorado and recreation areas near McPhee Reservoir and at Mesa Verde National Park (also areas of concern in 1995) were reported to have mortality (Johnson 1996, 1997). As of 2002, this disease is affecting piñon in Utah (1,350 acres on Dixie and Manti LaSal NFs), in northern New Mexico in two isolated areas, contributing to piñon decline (see below) (USDA Forest Service 2003).

- Brown cubical root and butt rot (*Phaeolus schweinitzii*) is common on old Douglas-fir in many parts of Region 3, causing defects and contributing to blowdowns (USDA Forest Service 2003).

- Comandra blister rust continues as one of the most common diseases of lodgepole pine in northern Colorado (Harris et al. 2001).

- An extensive outbreak of Douglas-fir beetle occurred in western Colorado (from Rifle west to the Douglas Pass area to the Utah border) and eastern Utah for several years prior to and including 1996, but the outbreak was presumed to be declining (Johnson 1997). In southwestern Colorado, Douglas-fir beetles impacted stands previously defoliated by western spruce budworm (e.g., on the Saguache RD, Rio Grande NF) and areas scorched during spring prescribed burn events (Harris et al. 2001). Light activity continues in the San Juans, north of Durango in approximately 1,700 trees; activity was reported in the Dallas Divide area and just west of Vallecito Reservoir (Harris et al. 2001). Douglas-fir mortality caused by Douglas-fir beetles decreased dramatically in Saguache County, Colorado in 2001 (Harris et al. 2002). Activity is anticipated to increase in 2003 for many areas of the San Juan and White River NFs from the 2002 wildfires that burned in the Douglas-fir forests (Harris 2003; USDA Forest Service 2003). In northern New Mexico, Douglas-fir beetle mortality was detected on the Carson (90 acres) and Sante Fe (175 acres) NFs, but decreased overall in the Southwest Region from 3,125 acres to 2,500 acres between 2001 and 2002 (USDA Forest Service 2003).

- Dwarf mistletoes caused the "greatest disease losses" in Region 2 (Johnson 1996, 1997 and Harris et al. 2001) and continue to be the most widespread and damaging pathogens in Regions 2, 3 and 4 (USDA Forest Service 2003). In Colorado, approximately 50% of the lodgepole pine cover type is infested with *A. americanum* (Johnson 1996, 1997). In 1995, sanitation thinning was completed in 11 campgrounds on the Gunnison and San Isabel NFs (Johnson 1996). Areas of concern in Colorado in 1996 for Douglas-fir dwarf mistletoe (*A. doglasii*) include: Columbine Ranger District (RD), San Juan NF; Mountain Lion Lookout area, Saguache RD, Rio Grande NF; and Long Creek on the Uncompahgre Plateau, Uncompahgre NF (Johnson 1997). For *A. vaginatum* subsp. *crytopodium* (ponderosa pine dwarf mistletoe), silvicultural control projects in 1995 and 1996 for Colorado are funded on the Southern Ute Reservation and the Salida RD, San Isabel NF (Johnson 1996, 1997). As a result of mild winters and periods of drought, dwarf mistletoes are contributing to mortality of lodgepole, ponderosa, limber, and piñon pines and Douglas-fir throughout the Rocky Mountain Region (Harris et al. 2002; Harris 2003).

- Fall webworm defoliated cottonwoods and boxelder trees along the Arkansas River near Salida, Colorado and in Grand Junction, Colorado during the summer of 2001 (Harris et al. 2002).

- Fir broom rust was reported as common throughout the spruce/fir cover type in Colorado from 1995 through 2001, causing only minor damage (Johnson 1996, 1997; Harris et al. 2001, 2002).

- Fir engraver beetle has almost doubled tree mortality in Region 3 to 13,725 acres in 2002 from 7,385 acres in 2001, with 2,110 acres of fir mortality on the Carson and

Sante Fe NFs in northern New Mexico. Fir mortality in Region 4 has also increased due to this beetle, with the Monti LaSal NF in Utah one of the most affected areas (3,150 trees on 1,000 acres) (USDA Forest Service 2003).

- Gambel oak discoloration is reported in the aerial survey data for most years, with no causal agent identified. The defoliation associated with this discoloration may have several causes. Oak has been affected by late frosts during several years. Oak looper is also known to have caused some of the recorded defoliation. None of this defoliation resulted in mortality.

- *Ips confuses* caused increasing mortality, in combination with black stain root disease, of piñon pine to the east and south of the Uncompahgre Plateau of CO in 1995 (Johnson 1996). *I. confuses* was reported in piñon pine on the western slope in 1996, particularly west of Durango and south of Montrose, Colorado (Johnson 1997). In 2000-2001, major piñon pine mortality was occurring in the southwest corner of Colorado centered around Mancos and Montezuma Counties (lower Dolores Canyon to the north, McElmo Canyon to the west, Mesa Verde to the south, and Hesperus to the east) due to drought/*Ips* combination (Harris et al. 2002). The 2002 Forest Insect and Disease Conditions in the Rocky Mountain Region report stated, "the amount of tree mortality due to Ips continues to skyrocket, in combination with dwarf mistletoe infection and drought." Piñon decline in the southwestern part of the state extends from Pagosa Springs west to the Four Corners area and north to Norwood. Surveys indicate roughly 50% of the piñon forests are dead (Harris 2003; USDA Forest Service 2003). There are also signs of black stain root disease in some piñon in these areas (USDA Forest Service 2003). In Region 3, Ips-caused ponderosa pine mortality was detected in the Sante Fe NF (825 acres) of New Mexico, and Ips-caused piñon mortality was detected on the Carson NF (16,240 acres) of New Mexico. Tree mortality from Ips has remained low in Region 4 (USDA Forest Service 2003).

- Mountain pine beetle mortality continued to increase in ponderosa and lodgepole pine, particularly on the western foothills of the Arkansas Valley in Colorado in 1996 (Johnson 1997). The 1996 aerial detection survey showed that, of the 86% of Colorado flown,12,891 trees on 10,879 acres were dead (Johnson 1997). For 1997 through 1999, mountain pine beetle populations and associated pine mortality increased and expanded in Colorado for each year, particularly in the Upper Arkansas Valley ponderosa pine (Harris et al. 2001). Large proportions of ponderosa pine at the wildland-urban interface between Buena Vista and Salida have had mortality levels as high as 80% in some stands (Harris et al. 2001). Mountain pine beetle caused mortality in ponderosa and lodgepole pine stands continued to increase in 2000-2001 in the Upper Arkansas Valley (Harris et al. 2002). During 2000-2001 beetle activity was increasing, but not yet at outbreak stage, for ponderosa pine stands due west of Monarch Pass of the GMUG NFs, and lodgepole pine infestations were taking place in Eagle and Chaffee Counties (Harris et al. 2002). During 2002, outbreaks in epidemic proportions were reported for the Upper Arkansas Valley (Chaffee County) in Colorado with 67,300 trees killed. Additionally, mountain pine beetle populations are dramatically increasing for Eagle (23,200 trees killed) and Saguche (15,600 trees killed) Counties of Colorado in 2002

(Harris 2003). Mountain pine beetle-caused mortality of ponderosa pine has also increased in Region 3, with Carson and Sante Fe NFs reporting 3,265 and 230 affected acres, respectively (USDA Forest Service 2003).

- Oak leaf roller defoliated large areas of Gambel oak in Colorado on both sides of the Continental Divide from 1997 to 1999 (Harris et al. 2001). Most of the defoliation occurred in the late spring and most affected oaks were able to produce a second flush of leaves, which escaped defoliation. This phenomenon was previously unrecorded. (Harris et al. 2001).

- Piñon needle scale (*Matsucoccus acalyptus*) affecting piñons in areas near Nathrop and Buena Vista were observed in 2002 (Harris 2003).

- Piñon decline of undetermined cause continued along the Arkansas River near Buena Vista, Colorado; neither black stain root disease or *Ips* spp. were contributing to the mortality (Harris et al. 2001).

- Ponderosa pine needlecast started as widespread in the Target Tree Campground, San Juan NF in 1996 (Johnson 1997). In 2000 and 2001, foliage discoloration of ponderosa pine has increased in severity each year since it was noted in 1999 on the San Juan NF (Harris et al. 2002). By 2002, the needlecast was discoloring foliage of ponderosa pines across the San Juan, Grand Mesa, Uncompahgre, and Gunnison NFs (USDA Forest Service 2003).

- Spruce beetle populations were building in Colorado in Conejos, Garfield, and Eagle counties in 1999. In 2001, the Flat Tops of Rio Blanco, Garfield, and Moffat Counties in Colorado were experiencing a rise in spruce beetle killed trees (3500 trees in 2001, double that of 2000). Additionally, scattered blowdowns of spruce stands in several areas of the Grand Mesa (i.e., Land's End area), Rio Grande (i.e., near Creede), and White River (i.e., near Triangle Park and the Buford-Newcastle road corridor) NFs were being attacked, with sanitation efforts underway to prevent large-scale outbreaks (Harris et al. 2002). Population increases continued in 2002 in Conejos, Mineral, Rio Blanco, and Garfield Counties in Colorado, specifically, in the spruce/fir forest type on the Uncompahgre Plateau and Grand Mesa NF portion of the GMUG and in the southern San Juan Mountains (Harris 2003). Weather conditions in 2002 (i.e., mild winter and warm dry summers) throughout Colorado and southwestern United states, in general, were highly conducive to increases in spruce beetle (Harris 2003). Region 3 saw significant increases in spruce beetle-caused mortality in 2002, with the Carson (1,675 acres) and Sante Fe (2,440 acres) NFs affected in northern New Mexico. Mortality from spruce beetles decreased in Region 4, however the Manti LaSal NF of Utah near the Colorado border contained the largest portion of killed trees in Region 4 (77,000 acres) (USDA Forest Service 2003).

- Spruce broom rust was reported as common throughout the spruce/fir cover type in Colorado from 1995 through 2001, generally causing only minor damage (Johnson 1996, 1997; Harris et al. 2001, 2002).

- Tiger moth (*Lophocampa ingens*) defoliated leaders of piñon at Ridgway State Park in an unusual case (Harris et al. 2002).

- Twig beetles (*Pityophthorus* spp. and *Pityogenes* spp.) in large numbers were evident in piñon stands south of Montrose, Colorado between 1997 and 1999.

- Western balsam bark beetle along with Armillaria root disease are causing mortality throughout the spruce/fir cover type of the Colorado Rocky Mountains in 1995 and 1996, particularly from Wyoming to Grand Junction and Leadville, Colorado (Johnson 1996, 1997). Subalpine fir decline has been the most widespread damage agent detected in the region since 1999 (Harris et al 2001, Harris et al 2002). The condition can be found in nearly all stands of subalpine fir and can be a significant challenge for forest managers (Harris et al 2002, Harris 2003). Ten Colorado counties within the Subregional Scale boundary had significant subalpine fir decline with > 10,000 trees per county killed (i.e., Delta, Eagle, Garfield, Gunnison, Mesa, Moffat, Pitkin, Rio Blanco, Routt, and San Miguel Counties) (Harris 2003).

- Significant activity of western pine beetle in New Mexico was detected on the Sante Fe NF (2,970 acres). Aerial surveys from 2003 also detected western pine beetle-caused mortality on the Uncompahgre NF in Colorado (372 dead trees on 966 acres).

- Western spruce budworm continued to cause widespread defoliation throughout forests of southern Colorado in 1995 and 1996 (Johnson 1996 and 1997). Many of the same areas have been repeatedly defoliated for almost a decade, leading to Douglas-fir and true fir mortality (Johnson 1996). In 1996 for Colorado, areas between South Fork and Wagon Wheel Gap and north of Lake City had reported activity of western spruce budworm (Johnson 1997). In 1998, heavy Douglas-fir defoliation was evident along the Lake Fork of the Gunnison River and immediately surrounding Lake City (Harris et al. 2001). Counties affected in southern Colorado in 2001 included: Conejos, Hinsdale, La Plata, Mineral, Montrose, Ouray, Rio Grande, and Sagauche Counties (Harris et al. 2002). Moderate defoliation of Douglas-fir was evident throughout portions of the Rio Grande NF in southern Colorado, and defoliation (over 4,000 acres) of subalpine fir and Engelmann spruce was detected in the northern San Juan Mountains near Ouray (Harris et al. 2001). As of 2002, this insect has lightly defoliated Engelmann spruce in areas of the Sangre de Cristo Mountains in Colorado (Harris 2003). Defoliation was also detected on the Carson (114,680 acres) and Sante Fe (32,075 acres) NFs in northern New Mexico within the Subregional Scale (USDA Forest Service 2003).

- White pine blister rust (*Cronartium ribicola*) was found in Colorado in 1996 on one western white pine nursery tree shipped from Idaho (Johnson 1997). By 2000/2001, limber pine stands in Colorado were experiencing low to moderate infection levels (Harris et al. 2002).

Landscape Scale

Page intentionally blank.

Grand Mesa, Uncompahgre and Gunnison National Forests

Recent Insect and Disease Occurrences

The five geographic areas (Grand Mesa, Gunnison Basin, North Fork Valley, San Juan, and Uncomphagre Plateau) of the GMUG NF administrative unit will be discussed individually in greater detail in the following "Geographic Area Scale" section. Here, it is useful to make comparisons between the geographic areas to depict the spatial pattern of aerial detected insect and disease occurrences from 1995-2003 across the entire Forest administrative area (Figure 6-2).

Subalpine fir decline in all five GAs and western spruce budworm on the Uncompahgre Plateau and San Juan GAs are the most densely distributed aerial documentation. Smatterings of aspen defoliation mainly in the northwest GAs and piñon decline on the Uncompahgre Plateau GA are also relatively prominent in the recent aerial documentation of insect and disease occurrences.

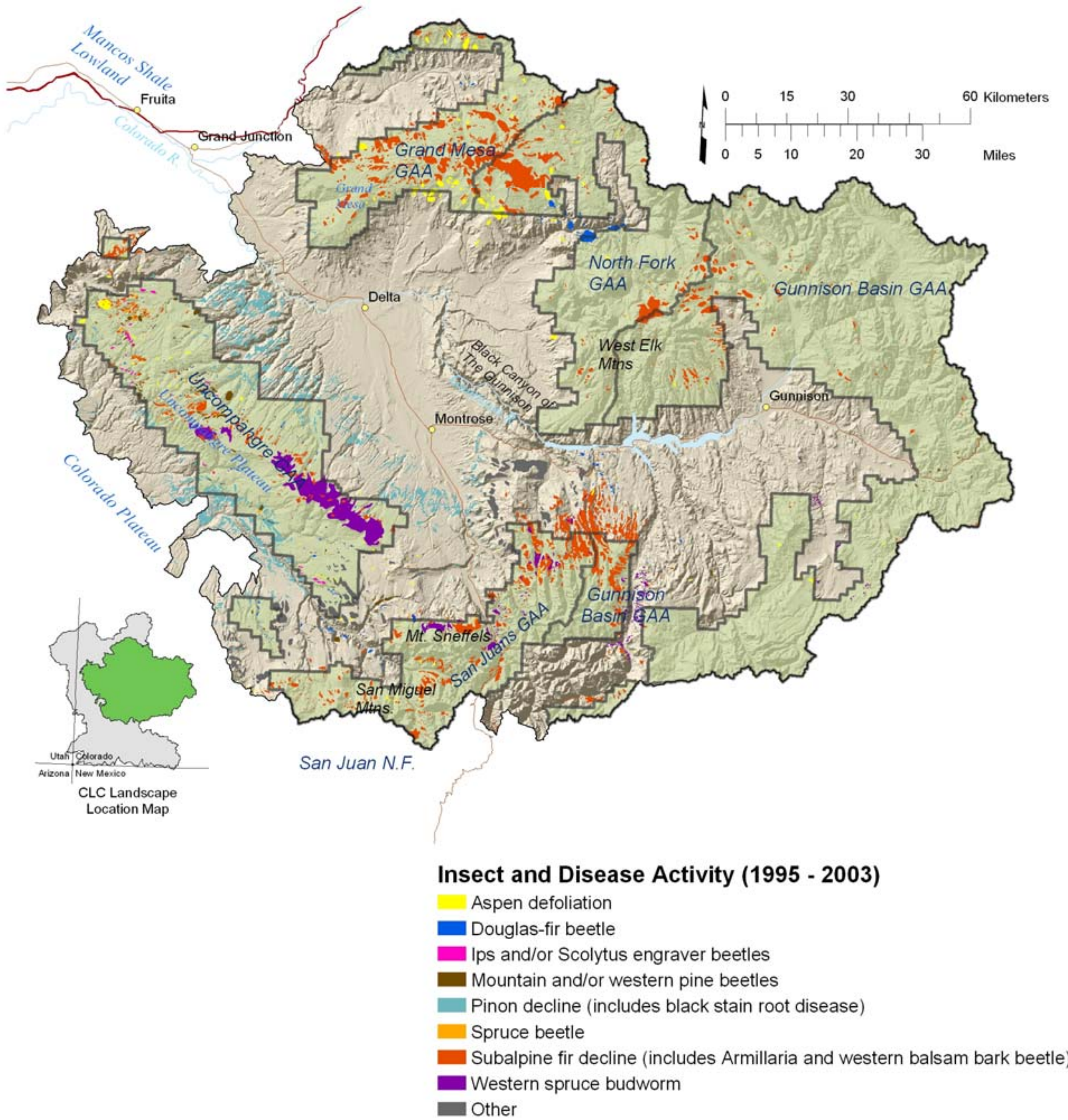


Figure 6-2 Recent insect and disease activity on the Grand Mesa, Uncompahgre, and Gunnison National Forests.
Recent insect and disease activity on the Grand Mesa, Uncompahgre, and Gunnison National Forestsascertained using aerial detection surveys from 1995-2003. Other includes: foliage discoloration, miscellaneous mortality, oak leafroller, ponderosa pine needleminer, and unknown.

Vulnerability to Future Infestations

Vulnerabilities to five insects and two diseases were modeled using a modification of a process described by Hessburg et al 1999. The areas shown in Table 6-1 and similar tables in subsequent Geographic Area discussions are not exclusive for the different organisms because a tree species can be a host for more than one organism listed.

For the entire GMUG administrative unit, western spruce budworm is predicted to impact the greatest area of host tree species (1,293,999 ac; 523,665 ha), with 41% of the total GMUG area in the high and moderate vulnerability rankings (Table 6-1). Spruce beetle is predicted to have the second largest impact on the GMUG land area (1,167,056 ac; 472,293 ha), with 33% of the land area in the high and moderate vulnerability rankings. Douglas-fir beetle is predicted to have the least impact on GMUG forest cover types (153,193 ac; 61,995 ha), which is only about 5% of the total land area in the high and moderate vulnerability rankings.

Table 6-1 Vulnerability ranking for future insect/pathogen infestation for the entire Grand Mesa, Uncompahgre, and Gunnison National Forests administrative unit.

Organism	Units	Vulnerability Ranking High	Moderate	Low	None	Host Tree Species
Western spruce budworm	acres	1,020,634	272,331	1,034	1,892,651	Douglas-fir, Engelmann spruce, Subalpine fir
	hectares	413,038	110,209	419	765,932	
	%	32	9	<1	59	
Spruce beetle	acres	701,094	352,963	112,999	2,019,594	Engelmann spruce
	hectares	283,724	142,840	45,729	817,304	
	%	22	11	4	63	
Douglas-fir beetle	acres	37,954	115,238	8,338	3,025,119	Douglas-fir
	hectares	15,360	46,635	3,374	1,224,227	
	%	1	4	<1	95	
Mountain pine beetle 1	acres	87,716	89,222	23,874	2,985,839	Ponderosa pine
	hectares	35,497	36,107	9,662	1,208,331	
	%	3	3	1	94	
Mountain pine beetle 2	acres	318,109	84,494	5,118	2,778,929	Lodgepole pine
	hectares	128,735	34,194	2,071	1,124,597	
	%	10	3	<1	87	
Lodgepole pine dwarf mistletoe	acres	378,181	29,461	79	2,778,929	Lodgepole pine
	hectares	153,045	11,923	32	1,124,597	
	%	12	1	<1	87	
Armillaria	acres	355,554	351,439	131,512	2,348,145	Blue spruce, Douglas-fir, Engelmann spruce, Subalpine fir
	hectares	143,888	142,223	53,221	950,264	
	%	11	11	4	74	

Figures 6-3 to figure 6-9 display the spatial distribution of the areas vulnerable to future infestations of the seven modeled insect and disease organisms. High and moderate vulnerability rankings for western spruce budworm (Figure 6-3), spruce beetle (Figure 6-4), and, to a slightly lesser degree, Armillaria (Figure 6-5) are predicted to be relatively evenly distributed across the GMUG NFs. The Uncompahgre Plateau to the west would be the exception with limited distribution of these organisms, because the host tree species for these three insect and disease organisms are a much smaller component of the landscape.

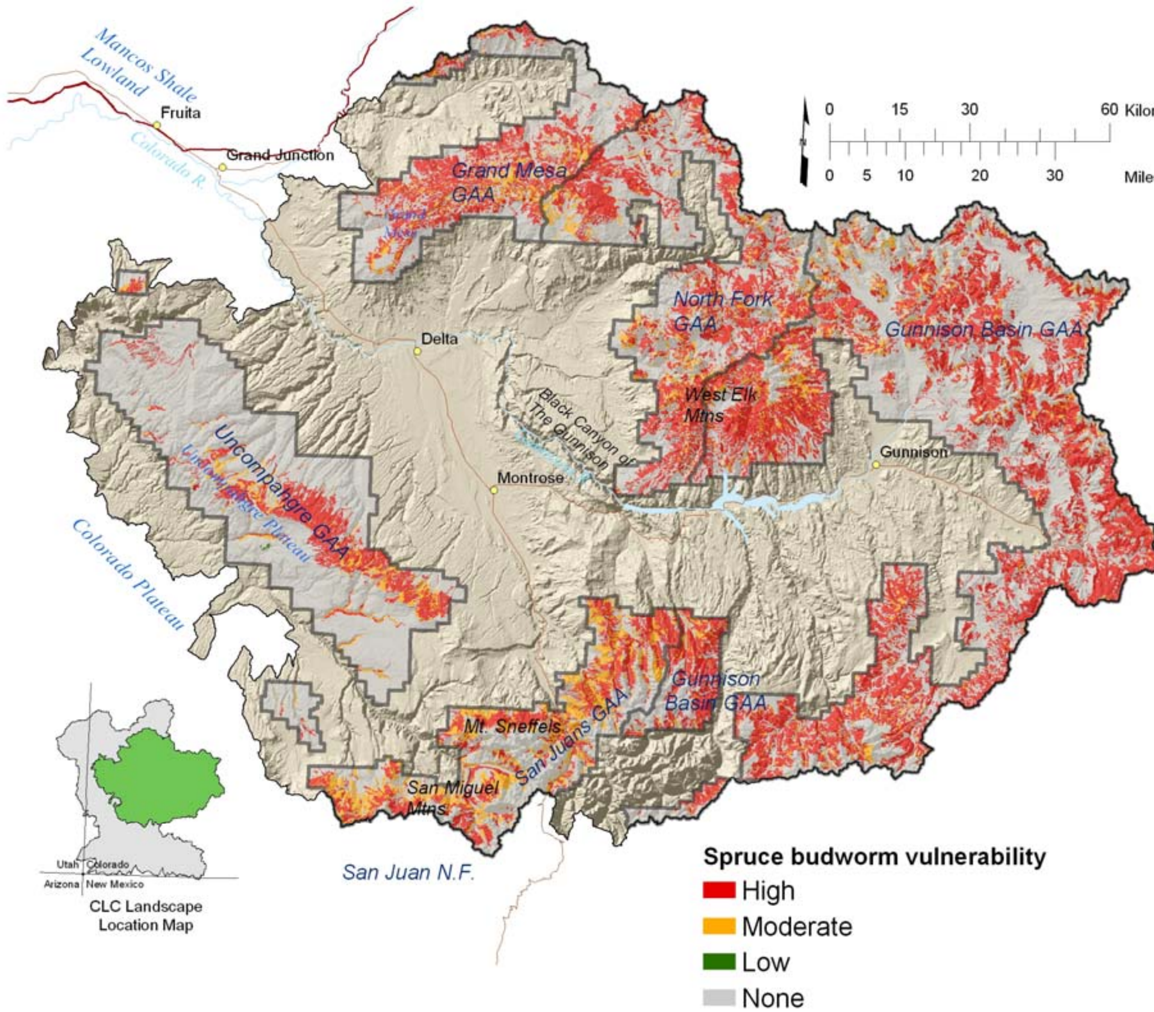


Figure 6-3 Western spruce budworm (Douglas-fir, Engelmann spruce, and subalpine fir hosts) vulnerability rankings across the Grand Mesa, Uncompahgre, and Gunnison National Forests.

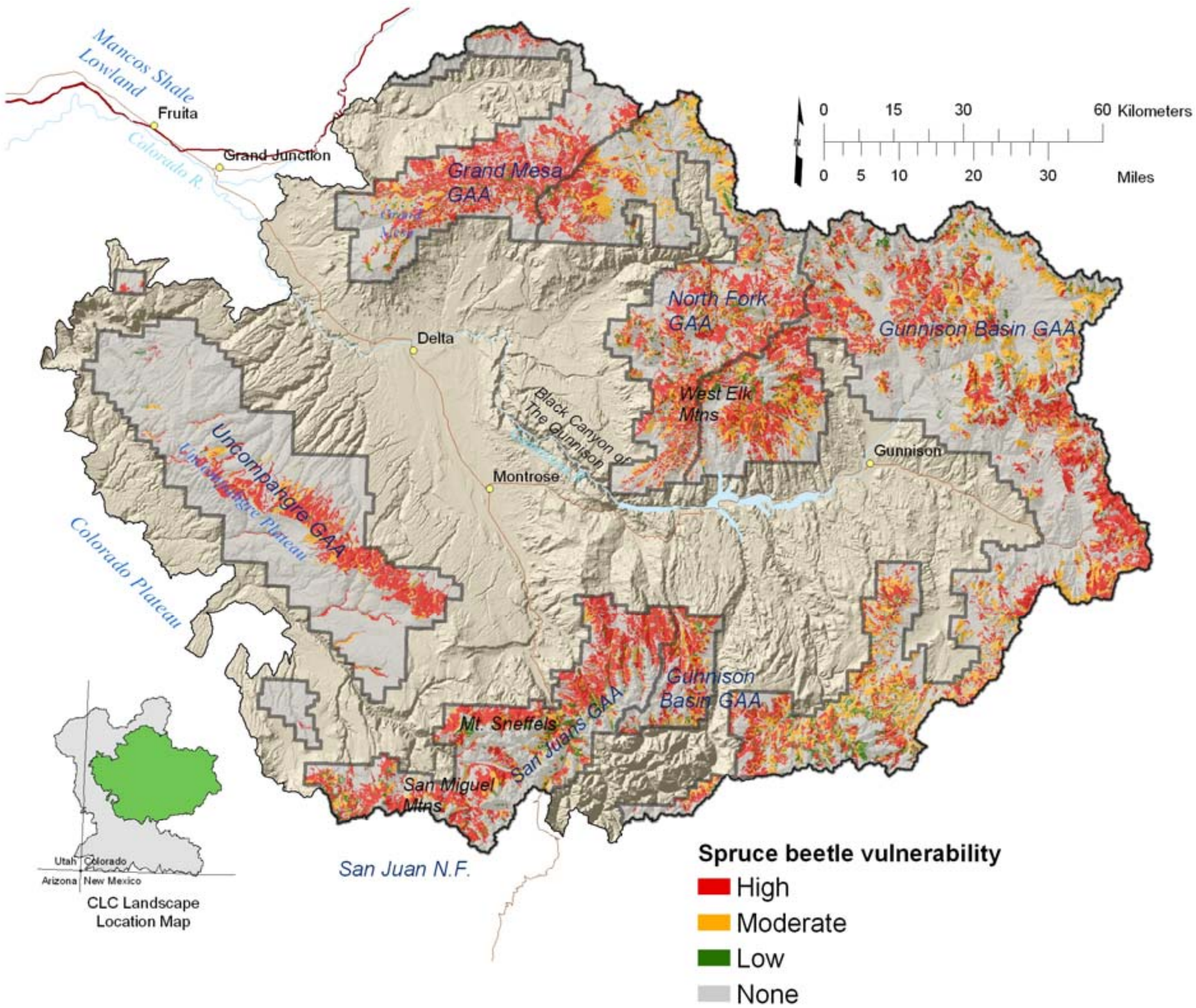


Figure 6-4 Spruce beetle (Engelmann spruce host) vulnerability rankings across the Grand Mesa, Uncompahgre, and Gunnison National Forests.

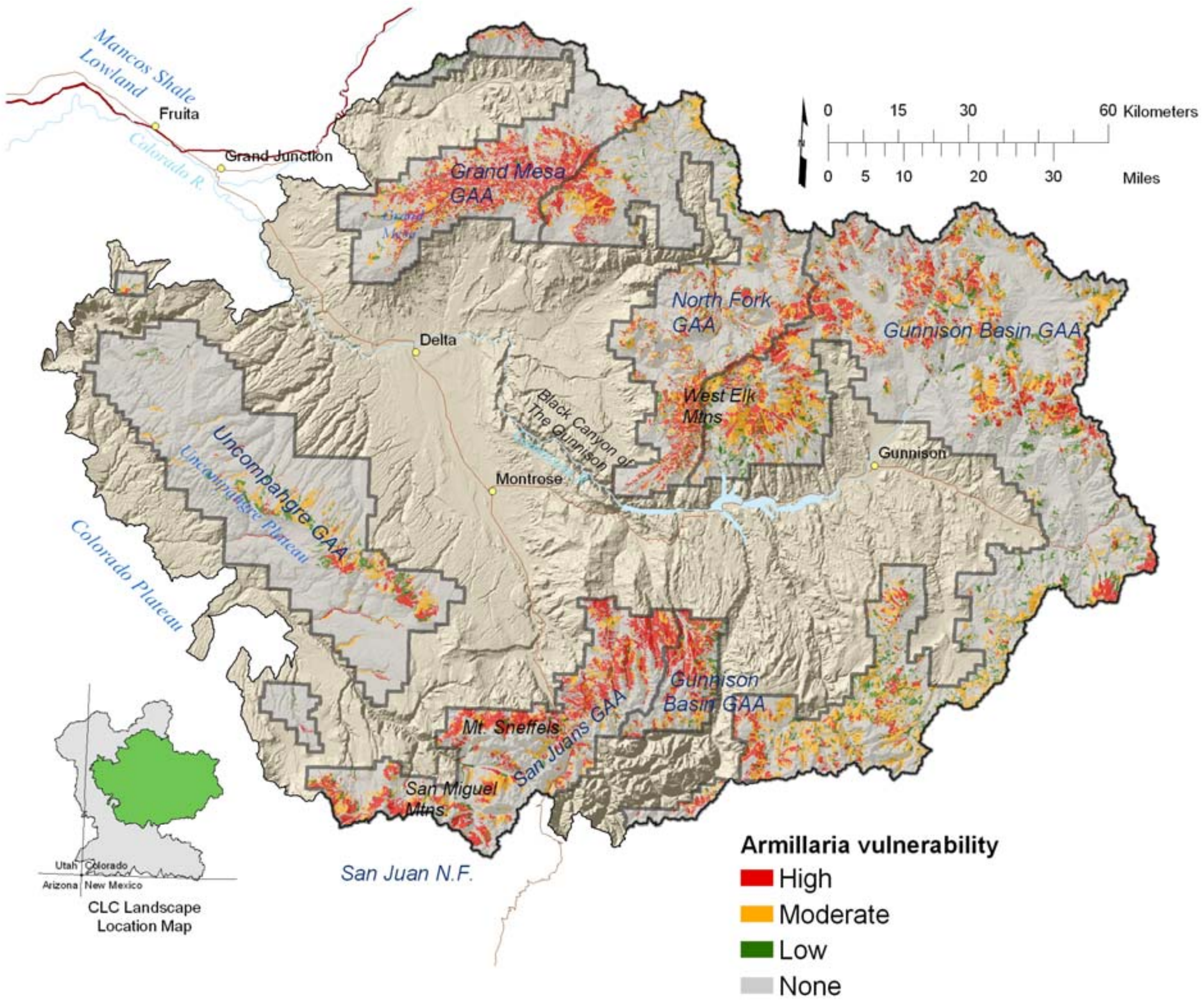


Figure 6-5 Armillaria (blue spruce, Douglas-fir, Engelmann spruce, and subalpine fir hosts) vulnerability rankings across the Grand Mesa, Uncompahgre, and Gunnison National Forests.

The high and moderate vulnerability rankings for Douglas-fir beetle are located in sparsely distributed pockets of Douglas-fir, mostly in the central to southern components of the GMUG NFs (Figure 6-6).

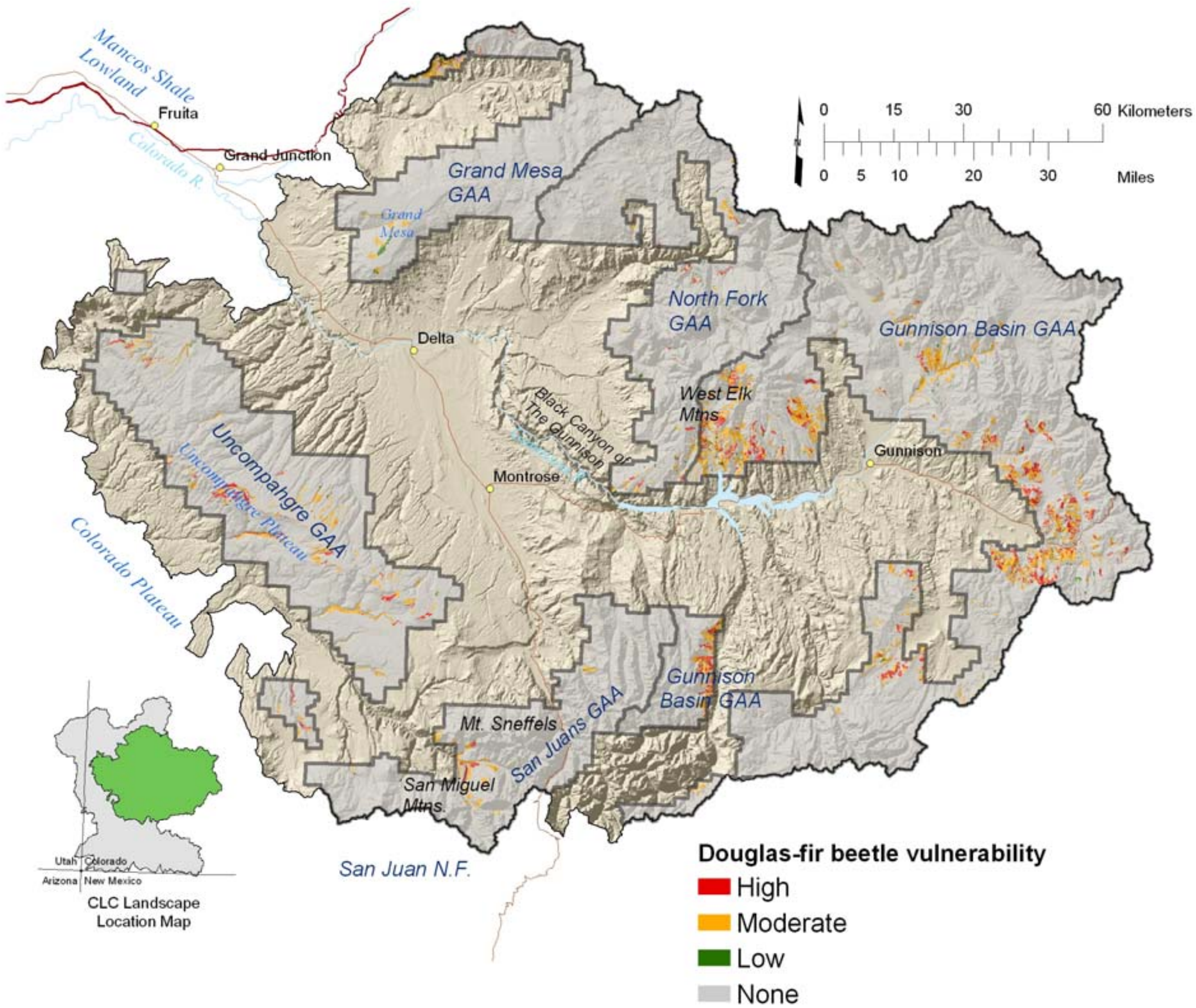


Figure 6-6 Douglas-fir beetle (Douglas-fir host) vulnerability rankings across the Grand Mesa, Uncompahgre, and Gunnison National Forests.

High and moderate vulnerability rankings for mountain pine beetle 1 are concentrated on the Uncompahgre Plateau to the west where ponderosa pine, the host species, is a larger component of the landscape (Figure 6-7).

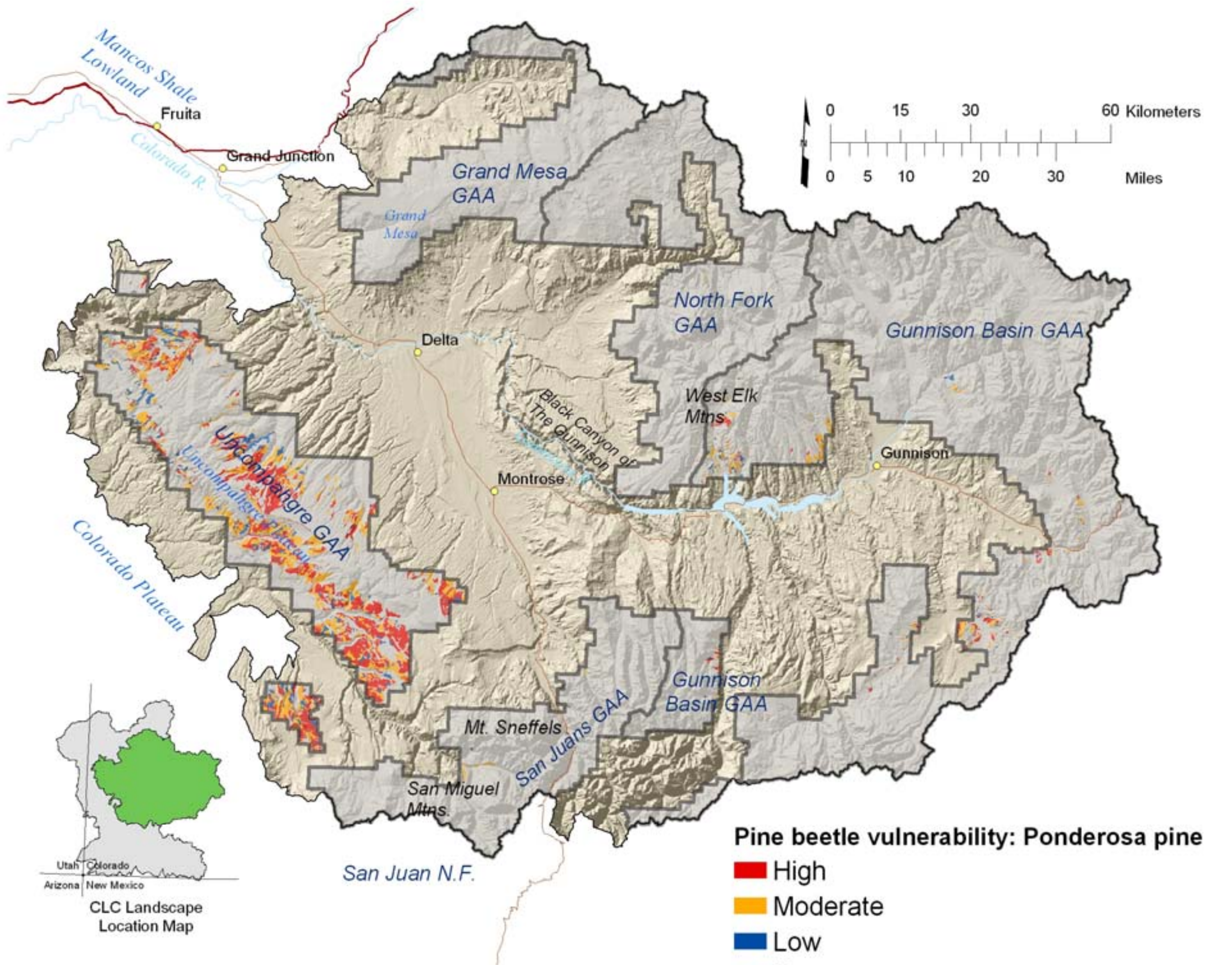


Figure 6-7 Mountain pine beetle 1 (ponderosa pine host) vulnerability rankings across the Grand Mesa, Uncompahgre, and Gunnison National Forests.

In direct contrast to the predicted distribution of mountain pine beetle 1, high and moderate vulnerabilities for mountain pine beetle 2 and lodgepole pine dwarf mistletoe are both concentrated in the eastern portion of the GMUG NFs, all within the Gunnison Basin GA, where lodgepole pine, the host species, is a larger component of the forest cover types.

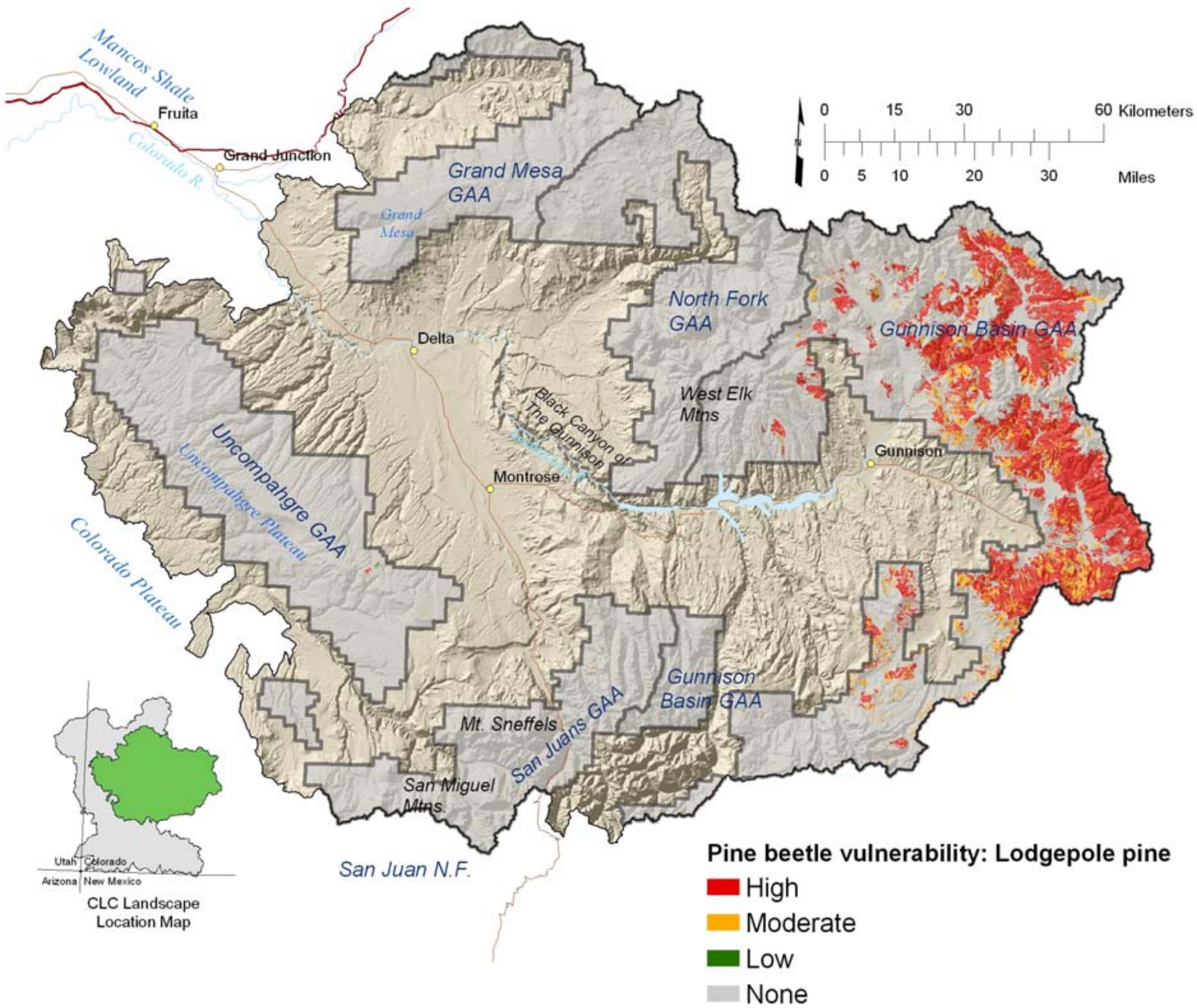


Figure 6-8 Mountain pine beetle 2 (lodgepole pine host) vulnerability rankings across the Grand Mesa, Uncompahgre, and Gunnison National Forests.

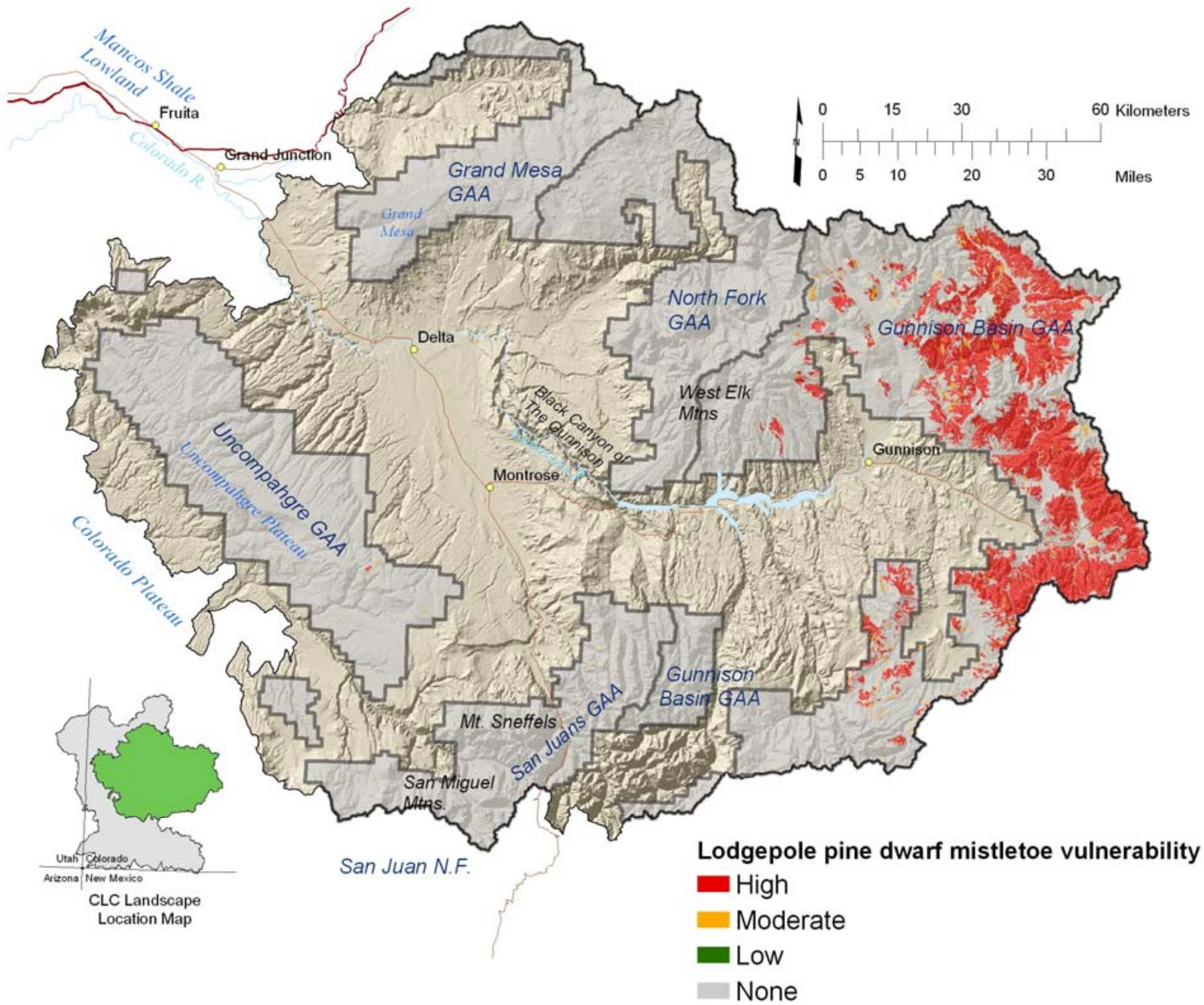


Figure 6-9 Lodgepole pine dwarf mistletoe (lodgepole pine host) vulnerability rankings across the Grand Mesa, Uncompahgre, and Gunnison National Forests.

San Juan National Forest

Recent Insect and Disease Occurrences

The acreage and percent of the San Juan NF covered by the aerial detection surveys since 1994 are provided in Table 6-2.

Table 6-2 Annual area of aerial detection surveys for the San Juan National Forest.

National Forest	Total acreage	1994		1995		1996		1997		1998		2000		2001		2002		2003	
		acres	%	acres	%	acres	%	acres	%	acres	%	acres	%	acres	%	acres	%	acres	%
San Juan	3,765,528	764,985	20	478,692	13	2,026,684	54	2,071,093	55	624,782	17	93,319	2	1,210,394	32	1,316,520	35	2,632,370	70

Table 6-3 Forest health aerial survey results for the San Juan National Forest from 1994 to 2003.
(note: there were no aerial survey data for 1999)

Damage causing agent	1994		1995		1996		1997		1998		2000		2001		2002		2003
	# of dead trees	acres affected	# of dead trees	acres affected	# of dead trees	acres affected	# of dead trees	acres affected	# of dead trees	acres affected	# of dead trees	acres affected	# of dead trees	acres affected	# of dead trees	acres affected	# of dead trees
Armillaria	--	--	--	--	--	--	21	8	--	--	--	--	--	--	--	--	--
Aspen Defoliation	--	--	8	76	na	166	na	1,570	na	64	na	20	na	238	na	3,440	na
Douglas-fir beetle	1,017	1,734	188	368	1,076	1,879	2,267	2,163	1,707	1,576	13	7	979	661	4,943	2,600	33,078
Douglas-fir engraver (Scolytus unispinosus)	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	2
Fir engraver (Scolytus ventralis)	--	--	976	1,757	638	1,254	946	1,831	5	1	--	--	--	--	--	--	84,015
Ips spp.	--	--	40	31	--	--	--	--	--	--	--	--	--	--	230	35	2,415
Ips pini (pine engraver)	--	--	--	--	--	--	2	0	--	--	--	--	--	--	--	--	21,849
Mountain pine beetle	1,780	3,480	772	1,680	535	1,617	471	599	97	65	--	--	647	1,360	3,546	3,884	missing data
Oak leafroller	--	--	--	--	--	--	--	--	--	--	na	1,567	--	--	--	--	--
Pinon decline (includes black stain root disease)	--	--	--	--	--	--	--	--	na	10,812	na	164	--	--	2,043	1,811	1,241,985
Spruce beetle	--	--	14	6	1	0	79	53	74	33	--	--	64	153	610	369	359
Subalpine fir decline	--	--	160	106	4,490	2,532	19,241	12,553	2,452	1,018	--	--	6,962	1,916	5,234	4,317	6,472
Western pine beetle	--	--	--	--	--	--	--	--	--	--	--	--	--	--	1,944	5,025	--
Western balsam bark beetle	40	34	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Western spruce budworm	na	4	--	--	--	--	--	--	--	--	--	--	na	16,489	na	72,817	na
Other	--	--	26	271	--	1,400	22	1,058	6	5,264	--	--	191	12,098	--	--	8,427

na = not applicable—mortality was not caused by the specific agent.
Other = foliage discoloration, mortality, ponderosa pine needleminer, and/or unknown

There were no flights over the San Juan NF in 1999, with very limited coverage in 1995, 1998, and 2000 and limited coverage in 1994, 2001, and 2002. Based on aerial survey detections, western spruce budworm consists of the largest contiguous areas of insect activity across the central portion of the San Juan NF; fir engraver and pine engraver are both very prominent across the southern and eastern portions of the Forest, and pinon decline is prominent across the southern and western portions of the Forest, becoming even more prominent to the southwest of the Forest (Figure 6-10). Smaller patches of aspen defoliation, subalpine fir decline, and Douglas-fir beetle are also evident from the aerial survey map (Figure 6-10).

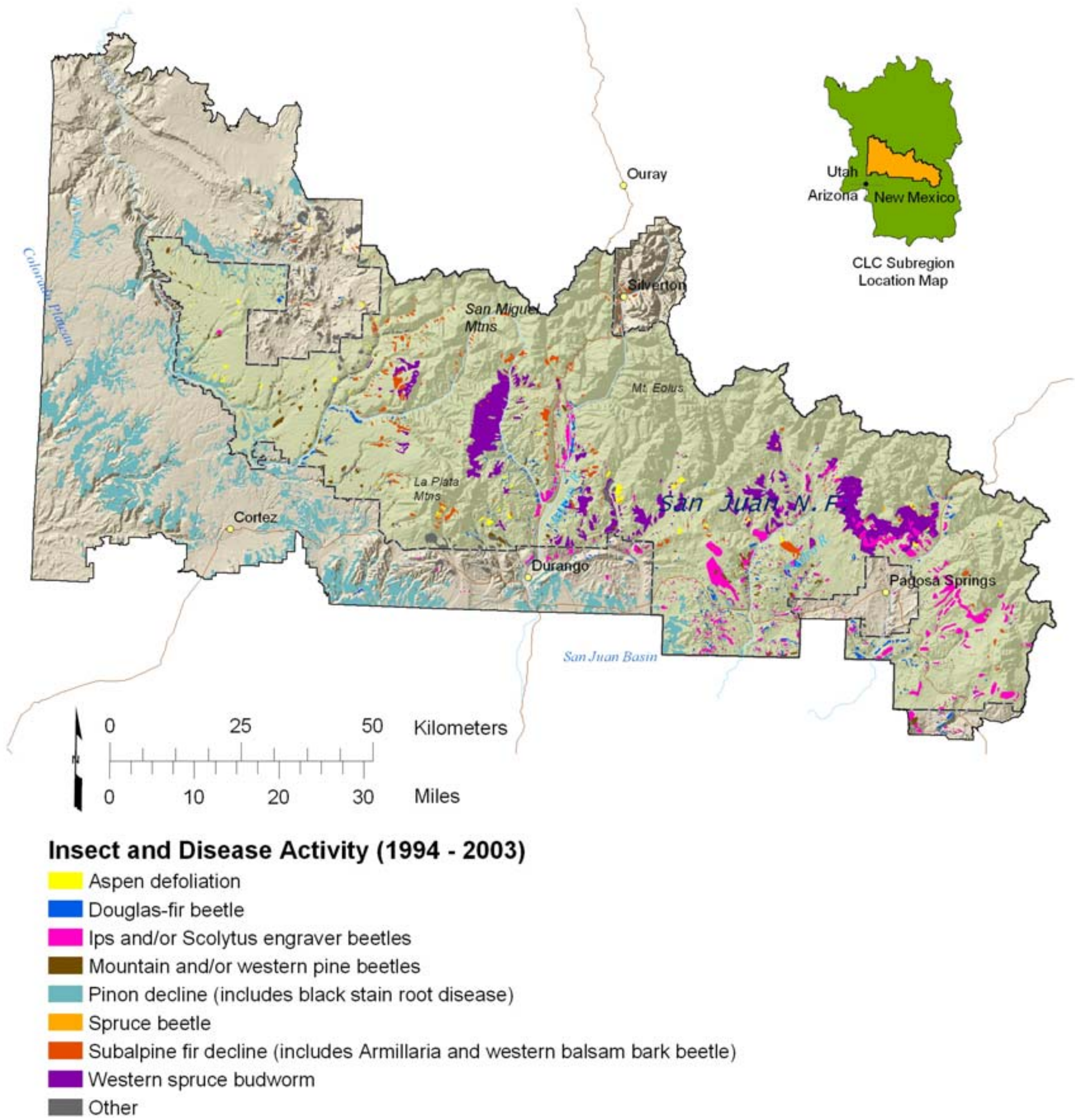


Figure 6-10 Recent insect and disease activity for the San Juan National Forest.

(ascertained using aerial detection surveys from 1994-2003). Other includes: foliage discoloration, miscellaneous mortality, oak leafroller, ponderosa pine needleminer, and unknown.

Piñon decline has shown the most drastic increase over the past decade, with almost 6% of the Forest documented to be affected (Table 6-3). Douglas-fir beetle, fir engraver, pine engraver and western spruce budworm are also on the rise over the past decade, though not to the degree of piñon decline. Subalpine fir decline has remained relatively constant since 1997 (Table 6-3).

Insect and diseases that currently occur or are/were a problem on the San Juan NF include:

- Aspen—Armillaria root disease and especially *Phellinus tremulae* are common, causing economic loss and mortality in some cases (D. Dallison, personal communication). Stem cankers (e.g., sooty bark, black, Cytospora, etc.) are also common, causing deformity and entrance wounds for other pathogens but seldom causing mortality (D. Dallison, personal communication). Additionally, defoliators (i.e., great basin tent caterpillar, large aspen tortrix, aspen leaf miner, and forest tent caterpillar) are common at times, however there are no large scale outbreaks currently (D. Dallison, personal communication).

- Douglas-fir/Mixed conifer--Douglas-fir beetle caused notable, scattered mortality from Bayfield to Pagosa Springs on the San Juan NF in 1995 and again in 1996 with mortality expected to continue (Johnson 1996, 1997). Currently Douglas-fir beetle activity is high with scattered mortality primarily on the eastern-half of the Forest, especially on the dry mixed conifer cover type (D. Dallison, personal communication). Fir engraver beetle is widespread on white fir and Douglas-fir primarily on the east side of the Forest due to favorable stand conditions (i.e., overly dense stands, drought conditions, chronic western spruce budworm defoliation, and root disease) (D. Dallison, personal communication). Armillaria and Annosus root diseases are prevalent in the San Juan NF, killing primary true firs and Douglas-fir (D. Dallison, personal communication). In 1996, Armillaria caused high white fir damage in the San Juan NF (Johnson 1997). Douglas-fir dwarf mistletoe was a concern in 1996 for the Columbine RD, San Juan NF (Johnson 1997). Western spruce budworm infestations have been chronic for the past decade due to the drought; most mixed conifer stands show some defoliation with new activity beginning in spruce/fir stands as well (D. Dallison, personal communication; Harris et al. 2001).

- Ponderosa pine— Ponderosa pine needlecast was widespread in the Target Tree Campground, San Juan NF in 1996 (Johnson 1997). In 2000 and 2001, foliage discoloration of ponderosa pine has increased in severity each year since it was noted in 1999 on the San Juan NF (specifically, from the West Fork of the Dolores River to the eastern and southern extent of the Pagosa RD) (Harris et al. 2002). By 2002 the needlecast was discoloring foliage of ponderosa pines across the San Juan, Grand Mesa, Uncompahgre, and Gunnison NFs (USDA Forest Service 2003). Trees have been left in a weakened condition, and mortality due to the effects of drought and needlecast has been observed, and western pine beetle has been detected in the needlecast-affected areas (Harris 2003). Mountain pine beetle is present on the Forest at endemic levels, with some increased activity due to the drought (D.

Dallison, personal communication). Significant mortality caused by western pine beetle is occurring in the Animas Valley and Hermosa Creek drainage, with scattered activity throughout the Forest (D. Dallison, personal communication). Ips pini is beginning to effect lower elevation ponderosa pine stands (D. Dallison, personal communication). Dwarf mistletoe, comandra rust, and Armillaria diseases are causing deformities and reductions in growth (estimated around 20-30% reductions in net growth) (D. Dallison, personal communication).

- Spruce/Fir—In 1996, Annosus root disease was identified in the Vallecito Campground, Columbine RD, San Juan NF (Johnson 1997). There is an upswing in spruce beetle populations, which have caused large scale mortality in the past following wind throw events (D. Dallison, personal communication). Western spruce budworm has also begun to defoliate understory trees in many spruce/fir stands (D. Dallison, personal communication).

- Piñon/Juniper—*Ips* and black stain root disease continue to be widespread in the southwest (D. Dallison, personal communication).

Vulnerability to Future Infestations

Vulnerabilities to four insects and one disease were modeled using a modification of a process described by Hessburg et al 1999. The areas shown in Table 3 are not exclusive for the different organisms because a tree species can be a host for more than one organism listed.

For the San Juan NF, western spruce budworm is predicted to impact the greatest area of host tree species (977,319 ac; 395,509 ha), with 26% of the total San Juan NF area in the high and moderate vulnerability rankings (Table 6-4). Armillaria (635,396 ac; 257,136 ha), mountain pine beetle (586,142 ac; 237,205 ha), and spruce beetle (580,868 ac; 235,070 ha) are predicted to have the next largest impacts on the San Juan land area, with 17%, 16%, and 16%, respectively, of the land area in the high and moderate vulnerability rankings.

Table 6-4 Vulnerability ranking for future insect/pathogen infestation for the entire San Juan National Forest.

Organism	Units	Vulnerability Ranking				Host Tree Species
		High	Moderate	Low	None	
Western spruce budworm	acres	785,966	191,353	892	2,786,761	Douglas-fir, Engelmann spruce, Subalpine fir, White fir
	hectares	318,071	77,438	361	1,127,767	
	%	21	5	<1	74	
Spruce beetle	acres	444,360	136,508	88,625	3,095,480	Engelmann spruce
	hectares	179,827	55,243	35,865	1,252,701	
	%	12	4	2	82	
Douglas-fir beetle	acres	146,796	235,943	7,969	3,374,265	Douglas-fir
	hectares	59,406	95,483	3,225	1,365,522	
	%	4	6	<1	90	
Mountain pine beetle	acres	437,490	148,652	141,140	3,037,691	Ponderosa pine
	hectares	177,047	60,158	57,117	1,229,315	
	%	12	4	4	81	
Armillaria	acres	303,373	332,023	139,781	2,989,797	Blue spruce, Douglas-fir, Engelmann spruce, Subalpine fir
	hectares	122,771	134,365	56,568	1,209,933	
	%	8	9	4	79	

Figures 6-11 through 6-15 display the spatial distribution of the areas vulnerable to future infestations of the five modeled insect and disease organisms. High and moderate vulnerability rankings for western spruce budworm (Figure 6-11) and, to a slightly lesser extreme, for spruce beetle (Figure 6-12) and Armillaria (Figure 6-13) are concentrated in the mountainous areas across the entire northeast portion of the Forest.

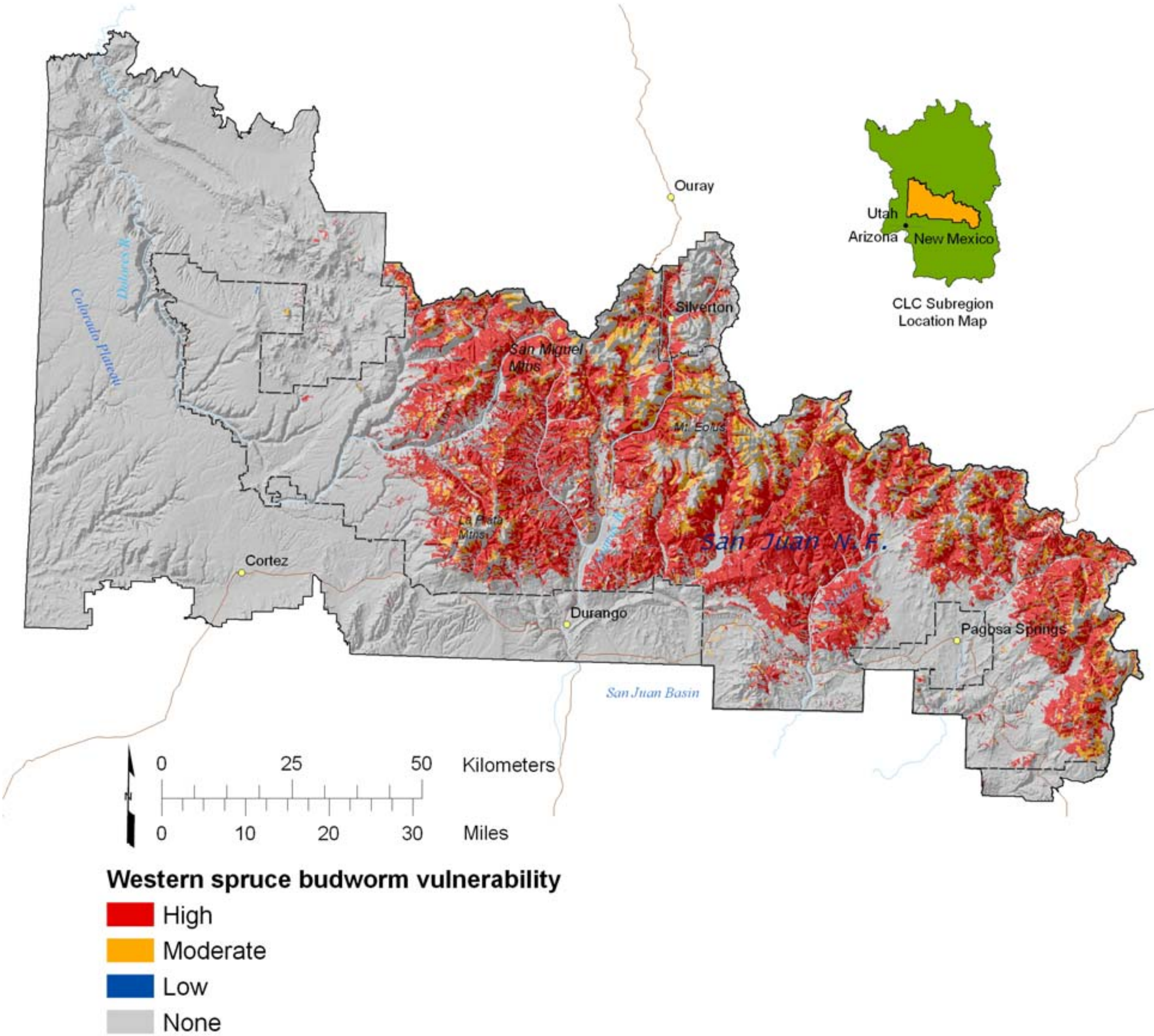


Figure 6-11 Western spruce budworm (Douglas-fir, Engelmann spruce, subalpine fir, and white fir hosts) vulnerability rankings across the San Juan National Forest.

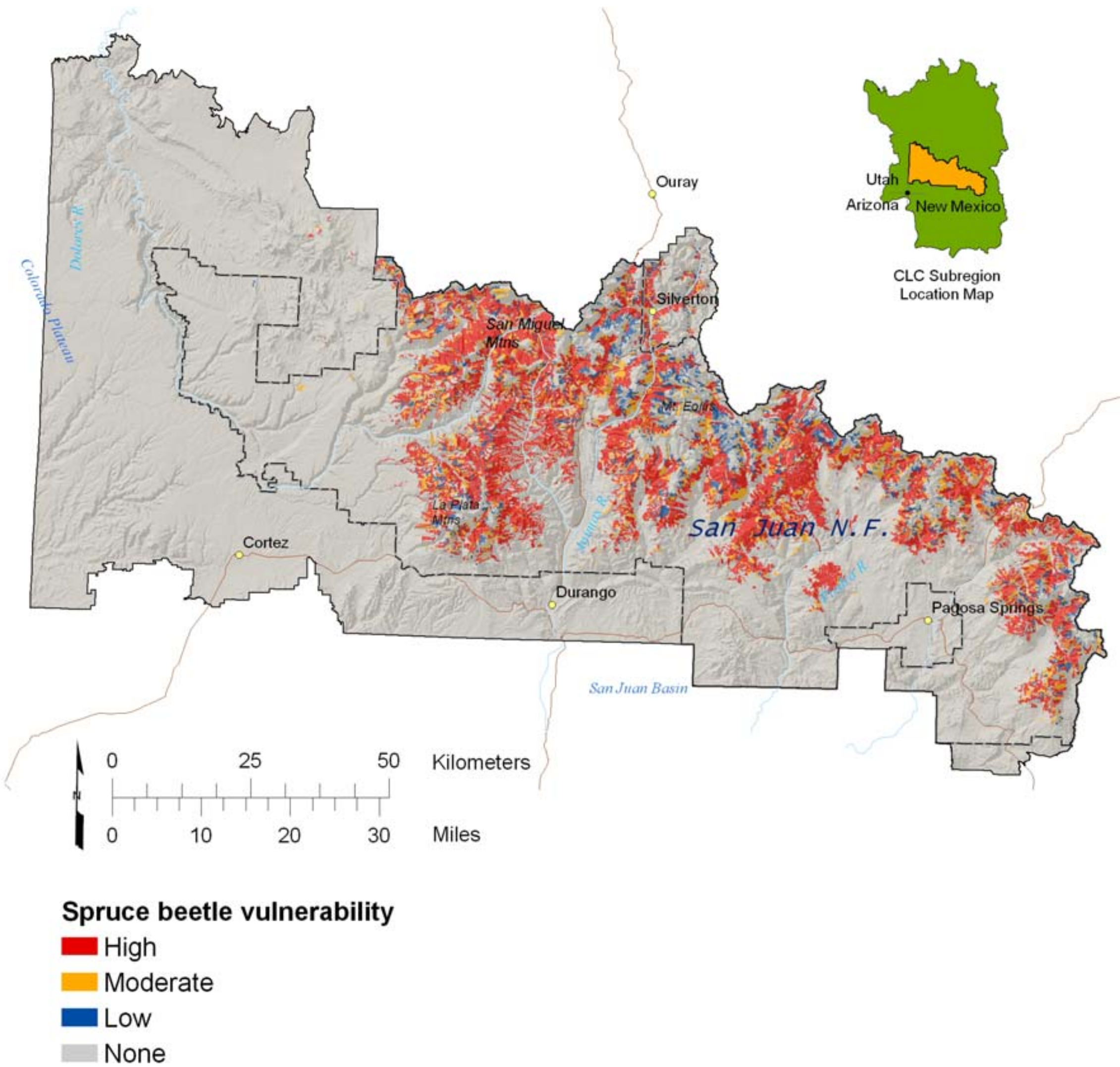


Figure 6-12 Spruce beetle (Engelmann spruce host) vulnerability rankings across the San Juan National Forest.

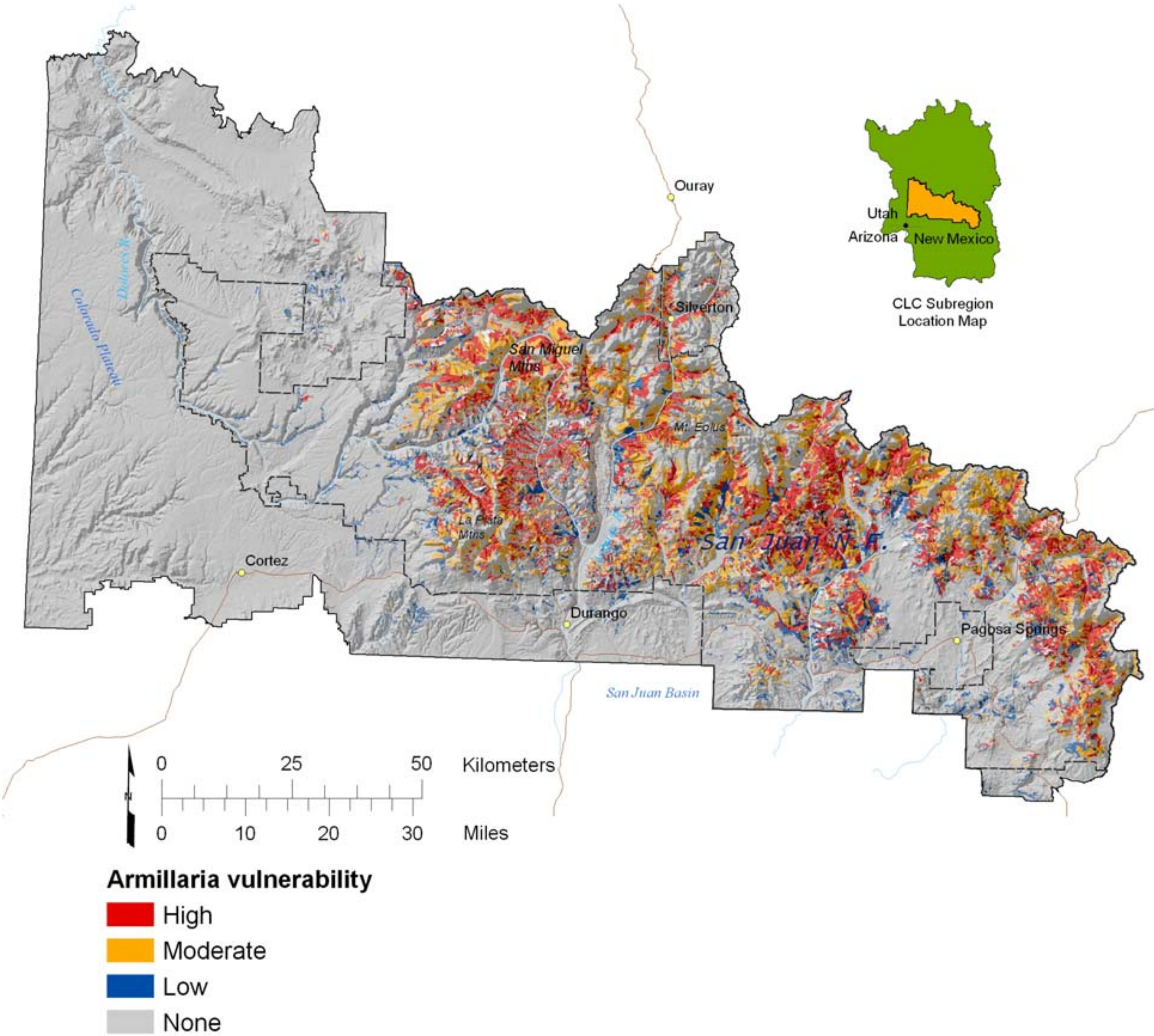


Figure 6-13 Armillaria (blue spruce, Douglas-fir, Engelmann spruce, and subalpine fir hosts) vulnerability rankings across the San Juan National Forest.

Mountain pine beetle (Figure 6-14) high and moderate vulnerabilities extend farther west and across the entire southern portion of the Forest compared with those of western spruce budworm.

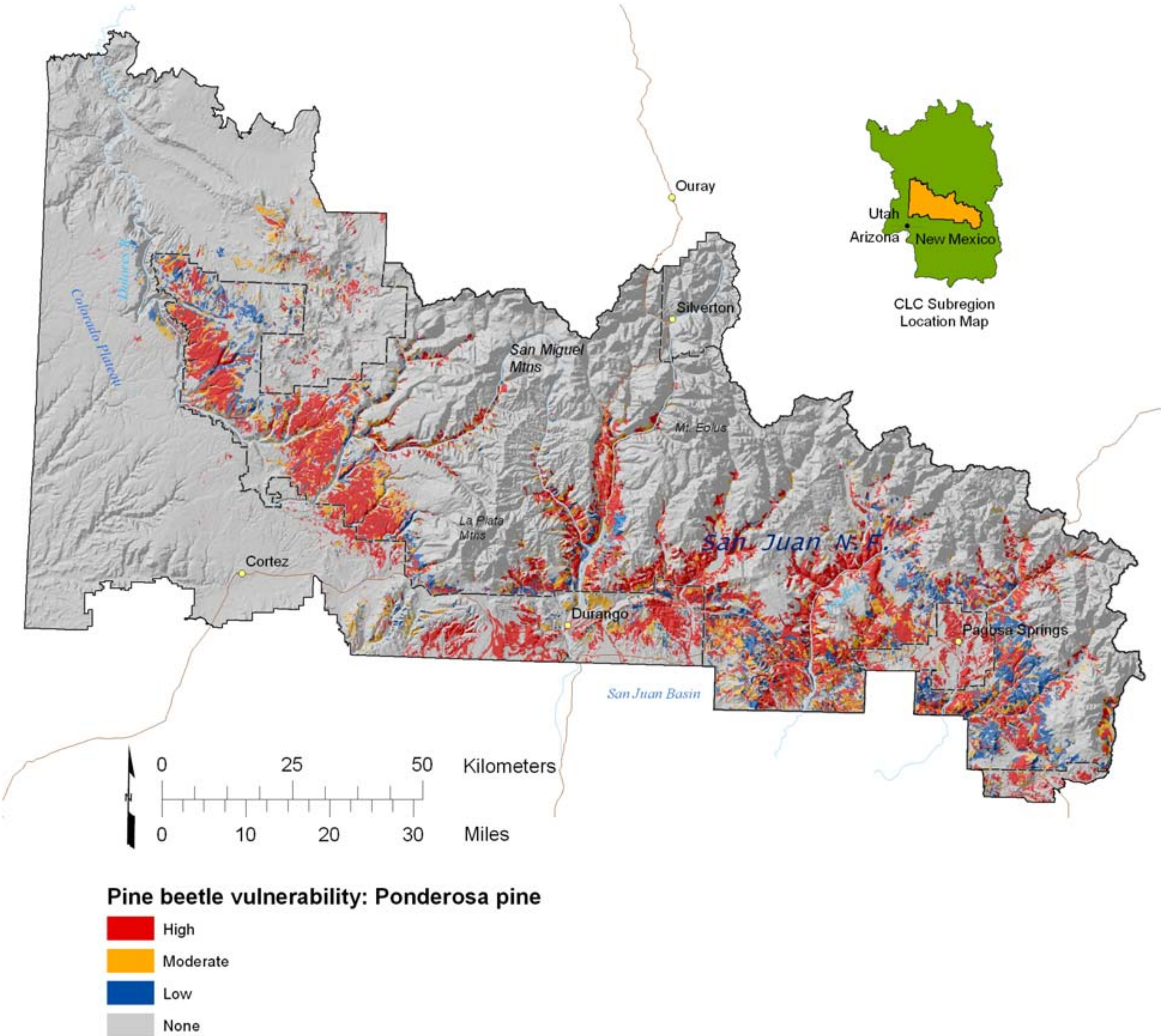


Figure 6-14 Mountain pine beetle (ponderosa pine host) vulnerability rankings across the San Juan National Forest.

Douglas-fir beetle (Figure 6-15) high and moderate vulnerability rankings are scattered across the entire Forest, but are much less concentrated compared with the other insect and disease vulnerabilities.

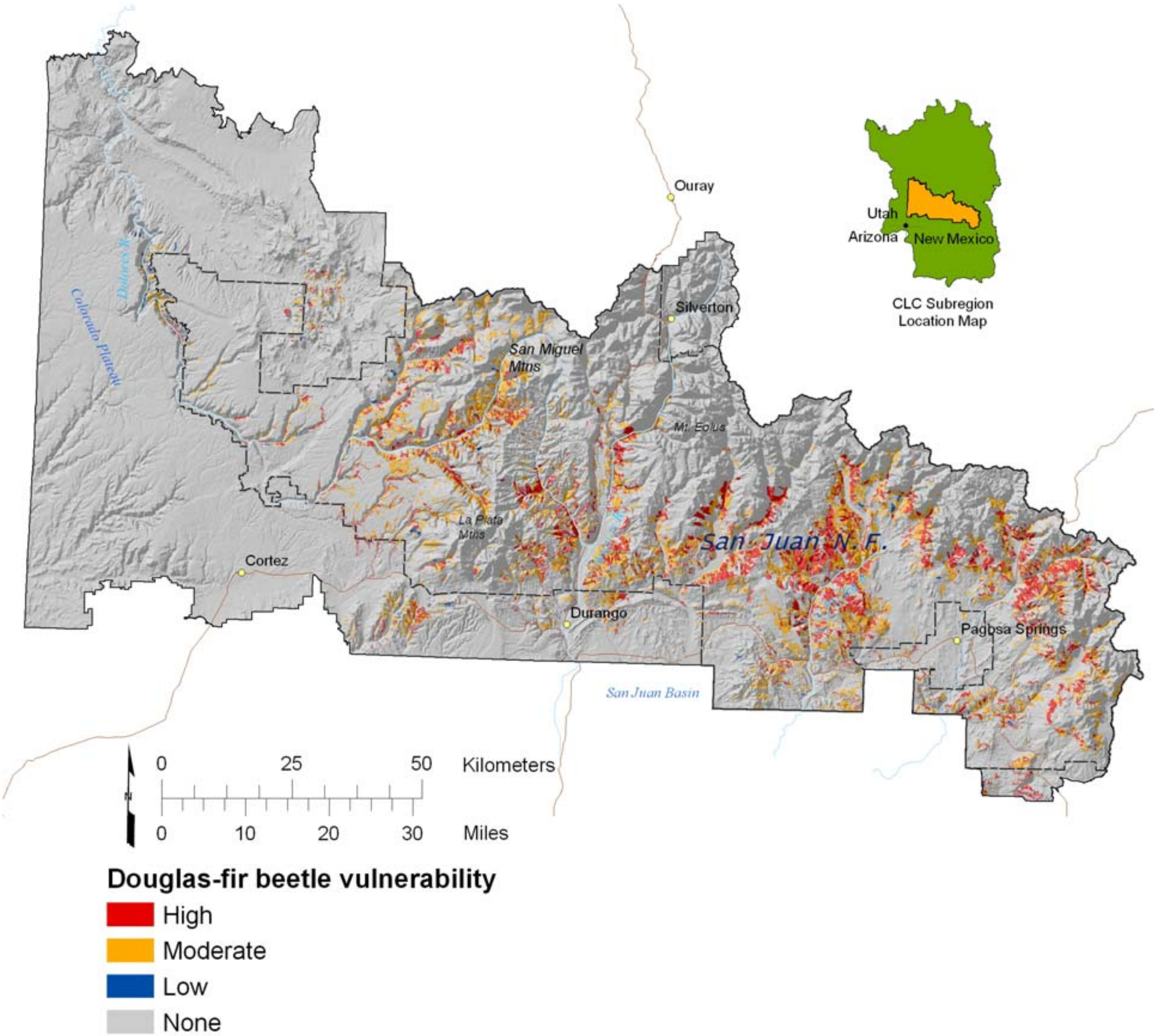


Figure 6-15 Douglas-fir beetle (Douglas-fir host) vulnerability rankings across the San Juan National Forest.

For a given host species, there are several cover types that may contain some proportion of that host species either in the canopy or understory, which may be vulnerable to insect and/or disease infestation. Table 6-5 summarizes the areas of each cover type that are vulnerable to the five insect/disease organisms listed in Table 6-4 for the San Juan NF. Douglas-fir (vulnerable to western spruce budworm, Douglas-fir beetle, mountain pine beetle, and Armillaria), spruce/fir (vulnerable to western spruce budworm, spruce beetle, and Armillaria), white fir (vulnerable to western spruce budworm, mountain pine beetle and Armillaria), ponderosa pine (vulnerable to mountain pine beetle), and aspen (vulnerable to western spruce budworm) are the main cover types with substantive area in the high and moderate vulnerability rankings.

Table 6-5 Vulnerability ranking for five modeled insect and disease organisms by forest cover types found within the San Juan National Forest.

Cover Type*	Units	Western spruce budworm vulnerability				Spruce beetle vulnerability				Douglas-fir beetle vulnerability				Mountain pine beetle vulnerability				Armillaria v	
		High	Moderate	Low	None	High	Moderate	Low	None	High	Moderate	Low	None	High	Moderate	Low	None	High	Moderate
Aspen	acres	125,261	11,780		210,642	38,154	19,614	640	289,274	22,992	48,205	599	275,887	35,810	18,413	99	293,362		427
	hectares	50,692	4,767	<1	85,244	15,441	7,938	259	117,066	9,305	19,508	242	111,648	14,492	7,451	40	118,720		173
	%	36	3	<1	61	11	6	<1	83	7	14	<1	79	10	5	<1	84		<1
Blue spruce	acres	547	35		4,130	169	35		4,509	352	616		3,745	702	251		3,760	223	359
	hectares	221	14		1,672	68	14		1,825	142	249		1,516	284	102		1,522	90	145
	%	12	1		88	4	1		96	7	13		79	15	5		80	5	8
Douglas-fir	acres	167,139	15,698	12	53,745	26,022	7,810	267	202,497	107,668	116,640	1,571	10,716	56,845	10,762	83	168,905	75,189	79,371
	hectares	67,639	6,353	5	21,750	10,531	3,161	108	81,948	43,572	47,203	636	4,337	23,005	4,355	33	68,354	30,428	32,121
	%	71	7	<1	23	11	3	<1	86	46	49	1	5	24	5	<1	71	32	34
Gambel oak	acres	71	1,958		355,343			102	357,271		5s50	2,020	354,803		4,571	101,685	251,117		
	hectares	29	793		143,803			41	144,583		222	817	143,584		1,850	41,151	101,624		
	%	<1	1		99			<1	100		<1	1	99		1	28	70		
Piñon/ Juniper	acres		16		444,514		16		444,514		706	169	443,654	1,769	8,383	303	434,074		
	hectares	<1	6		179,889		6		179,889		286	68	179,541	716	3,393	122	175,664		
	%	<1	<1		100		<1		100		<1	<1	100	<1	2	<1	98		
Ponderosa pine	acres	35,965	7,672		395,204	593	103		438,147	2,883	33,613	390	401,956	331,790	103,825	1,723	1,504		251
	hectares	14,555	3,105		159,934	240	41		177,313	1,167	13,603	158	162,667	134,271	42,017	697	609		102
	%	8	2		90	<1	<1		100	1	8	<1	92	76	24	<1	<1		<1
Spruce/Fir	acres	431,644	71,935	105	6,694	376,759	107,067	15,792	10,760	12,869	35,361	44	462,105	1,691	356		508,331	250,213	236,266
	hectares	174,681	29,111	43	2,709	152,470	43,328	6,391	4,355	5,208	14,310	18	187,008	684	144		205,715	101,258	95,614
	%	85	14	<1	1	74	21	3	2	3	7	<1	91	<1	<1		100	49	46
White fir	acres	25,111	1,963		23	2,664	1,237		23,195				27,097	8,661	1,969		16,467	11,007	15,348
	hectares	10,162	794		9	1,078	501		9,387				10,966	3,505	797		6,664	4,454	6,211
	%	93	7		<1	10	5		86				100	32	7		61	41	57

* Cover types with host tree species present in mix

Intersection of Fire with Insect and Disease

No data yet

Geographic Area Scale (for GMUG administrative unit only)

Grand Mesa

The majority of Grand Mesa Geographic Area (GA) consists of four cover types: spruce/fir (26% of the total GA), aspen (26%), Gambel oak (14%), and grass/forb (11%). Existing conditions for all forest types are dominated by trees classified as mature, dense stands with canopy closures greater than 40 percent.

Recent Insect and Disease Occurrences

The acreage and percent of each GA covered by the aerial detection surveys since 1995 are provided in Table 6-6. There were no flights over the Grand Mesa GA in 1995, 1996, 1998, 1999, or 2000, with limited coverage (27% of GA) in 2001. Based on aerial survey detections, subalpine fir decline continues to affect the most area compared with all other detected organisms on the Grand Mesa GA since 1997 (Figure 6-16 and Table 6-7). Aspen defoliation though prominent in Figure 6-16, appears to have declined since 1997, while Douglas-fir beetle, piñon decline, and spruce beetle have affected increasingly greater area since 1997. However, given the paucity of flights over this GA, trends are difficult to generalize with great reliability.

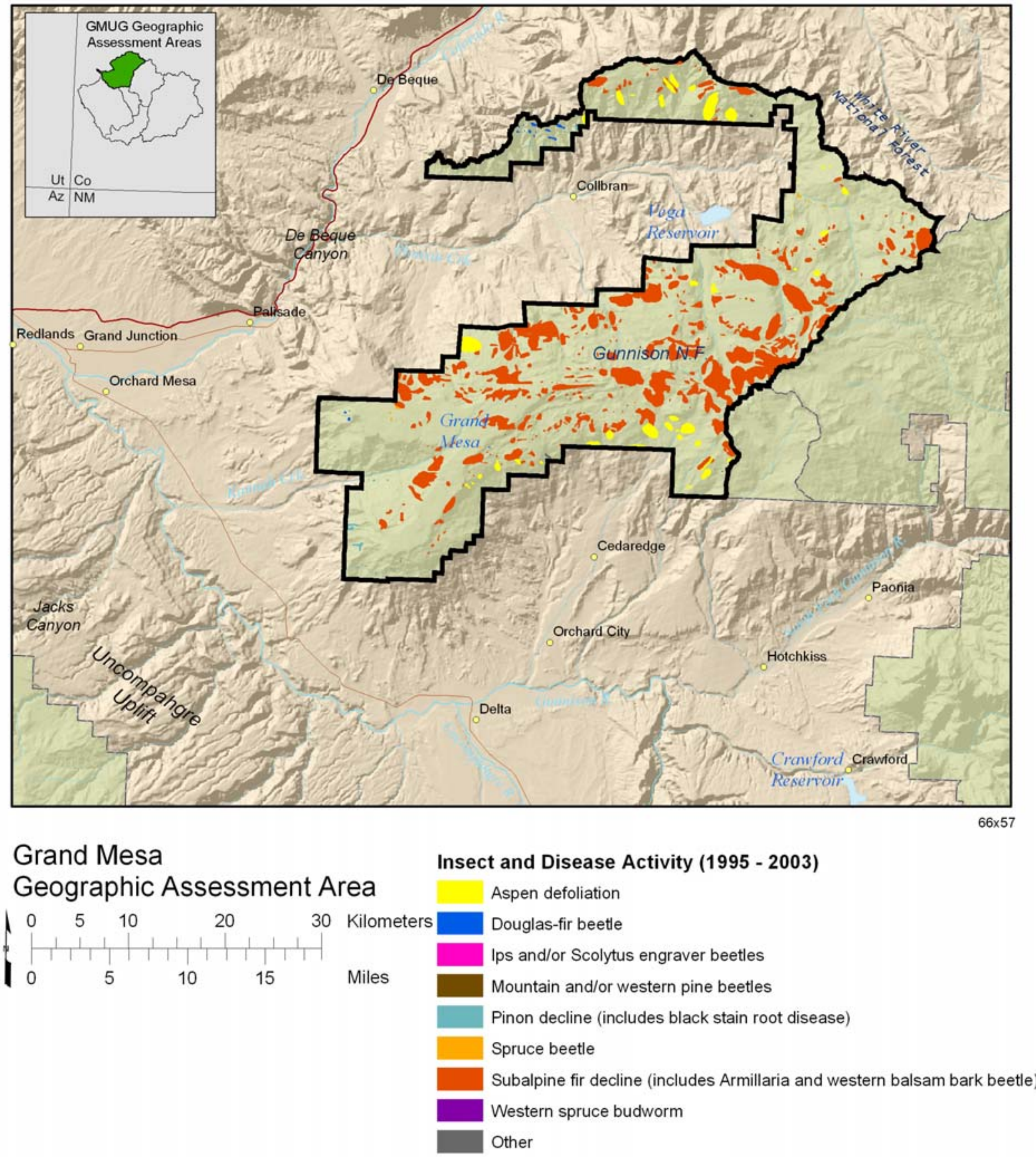


Figure 6-16 Recent insect and disease activity for Grand Mesa Geographic Area. Recent insect and disease activity for Grand Mesa Geographic Area ascertained using aerial detection surveys from 1995-2003. Other includes: foliage discoloration, miscellaneous mortality, oak leafroller, ponderosa pine needleminer, and unknown.

Table 6-6 Annual areal coverage of aerial detection surveys for each Geographic Area (GA) of the Grand Mesa, Uncompahgre, and Gunnison National Forests.

Geographic Area	Total acreage of GA	1995		1996		1997		1998		1999		2000		2001		2002		2003	
		acres	%	acres	%	acres	%	acres	%	acres	%	acres	%	acres	%	acres	%	acres	%
Grand Mesa	320,670	0	0	0	0	320,670	100	0	0	0	0	0	0	87,558	27	278,699	87	320,670	100
Gunnison Basin	1,389,690	0	0	1,336,167	96	0	0	164,585	12	696,264	50	580,068	42	59,323	4	59,790	4	0	0
North Fork Valley	504,810	0	0	481,910	95	43,866	9	232	0	22,175	4	0	0	73,031	14	182,543	36	214,279	42
San Juan	356,690	0	0	29,595	8	1,761	0	357,890	100	0	0	342,362	96	0	0	272,936	77	59,909	17
Uncompahgre Plateau	614,795	366,257	60	599,609	98	297,178	48	86,787	14	0	0	139,724	23	586,660	95	603,364	98	586,660	95

Table 6-7 Forest health aerial survey results for the National Forest portion of the Grand Mesa Geographic Area.
(note: there were no aerial survey data for 1995, 1996, 1998, 1999, and 2000 for this GA).

Damage Causing Agent	1997		2001		2002		2003	
	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected
Aspen Defoliation	na	5,225	--	--	na	608	na	117
Douglas-fir beetle	16	49	6	0	21	8	271	220
Mountain pine beetle	--	--	--	--	2	4	--	--
Piňon decline	--	--	--	--	--	--	2,241	460
Spruce beetle	10	14	25	14	115	206	263	480
Subalpine fir decline	22,169	19,368	1,544	1,124	34,362	19,994	25,122	16,772
Western spruce budworm	--	--	--	--	--	--	na	3
Other	na	70	8	9	--	--	--	--
na = not applicable—mortality was not caused by the specific agent. Other = Diplodia blight, Douglas-fir needle cast, foliage discoloration, unidentified bark beetles, and/or unidentified mortality								

Insect and diseases that currently occur or are/were a problem on the Grand Mesa GA include:

- Aspen—The 2001 GMUG MIS Assessment reported that aspen mortality in mature stands has been increasing, due to the natural aging of these aspen stands (USDA Forest Service 2001). Various cankers and rots are common and active, but they generally do not show up as "epidemics" (C. McKenzie, personal communication). Aspen defoliating and boring insects are not present in any large scale (C. McKenzie, personal communication).

- Piñon/Juniper—*Ips confusus* is effecting the area, and twig beetle activity is starting to dramatically increase (C. McKenzie, personal communication).

- Ponderosa pine—Starting in the San Juan NF in the late 1990s, by 2002 ponderosa pine needlecast was discoloring foliage of ponderosa pines across the San Juan, Grand Mesa, Uncompahgre, and Gunnison NFs (USDA Forest Service 2003).

- Spruce/Fir—Spruce beetle activity is high (C. McKenzie, personal communication). Spruce beetle population increases continued in 2002 in the spruce/fir forest type on the Grand Mesa NF portion of the GMUG (Harris 2003). Subalpine fir decline from western balsam bark beetle in combination with Armillaria root disease is present, and Armillaria root disease is showing up quite frequently in the Engelmann spruce (C. McKenzie, personal communication). Red ring rot and conifer broom rusts are common throughout but not in large specific areas (C. McKenzie, personal communication).

Vulnerability to Future Infestations

Western spruce budworm (41% of the total GA land area in the high and moderate vulnerability ranking), spruce beetle (35%), and Armillaria (29%) are predicted to be of concern (Table 6-8).

Table 6-8 Vulnerability ranking for future insect/pathogen infestation for the Grand Mesa Geographic Area.

Organism	Units	Vulnerability Ranking High	Moderate	Low	None	Host Tree Species
Western spruce budworm	acres	103,563	29,506	34	187,567	Engelmann spruce, Douglas-fir, Subalpine fir
	hectares	41,911	11,941	14	75,906	
	%	32	9	<1	58	
Spruce beetle	acres	92,901	19,463	8,350	199,955	Engelmann spruce
	hectares	37,596	7,877	3,379	80,919	
	%	29	6	3	62	
Douglas-fir beetle	acres	654	6,002	1,996	312,017	Douglas-fir
	hectares	265	2,429	808	126,269	
	%	<1	2	1	97	
Mountain pine beetle 1	acres				320,669	Ponderosa pine
	hectares				129,771	
	%				100	
Mountain pine beetle 2	acres				320,669	Lodgepole pine
	hectares				129,771	
	%				100	
Lodgepole pine dwarf mistletoe	acres				320,669	Lodgepole pine
	hectares				129,771	
	%				100	
Armillaria	acres	67,613	24,474	7,652	220,931	Blue spruce, Douglas-fir, Engelmann spruce, Subalpine fir
	hectares	27,362	9,904	3,097	89,408	
	%	21	8	2	69	

For a given host species, there are several cover types that may contain some proportion of that host species either in the canopy or understory, which may be vulnerable to insect and/or disease infestation. Table 6-9 summarizes the areas of each cover type that are vulnerable to the seven insect/disease organisms listed in Table 6-8 for the Grand Mesa GA (note: note only four of the seven organisms had any vulnerable area). Spruce/fir and aspen are the only cover types with substantive area in the high and moderate vulnerability rankings. Notably, all the blue spruce and Douglas-fir are also in the high and moderate vulnerability rankings for western spruce budworm and/or spruce beetle; however these cover types together account for only approximately 3,500 ac (1,400 ha) or 1% of the Grand Mesa land area.

Table 6-9 Vulnerability ranking for seven modeled insect and disease organisms by forest cover types found within the Grand Mesa Geographic Area.

Cover Type*	Units	Western spruce budworm vulnerability				Spruce beetle vulnerability				Douglas-fir beetle vulnerability				Armillaria vulnerability			
		High	Moderate	Low	None	High	Moderate	Low	None	High	Moderate	Low	None	High	Moderate	Low	None
Spruce/fir	acres	75,287	17,790		396	76,991	14,352	1,196	934	81	882		92,513	67,613	24,226	1,611	27
	hectares	30,468	7,200		160	31,157	5,808	484	378	33	357		37,439	27,362	9,804	652	11
	%	81	19		<1	82	15	1	1	<1	1		99	72	26	2	0
Douglas-fir	acres	2,240	728			28			2,940	573	2,365	29			28	1,690	1,249
	hectares	907	295			11			1,190	232	957	12			11	684	506
	%	75	25			1			99	19	80	1			1	57	42
Blue spruce	acres	55				55							55		55		
	hectares	22				22							22				
	%	100				100							100				
Aspen	acres	23,762	2,181		63,498	15,826	5,082	156	68,377		948	26	88,468		164	3,827	85,450
	hectares	9,616	883		25,697	6,404	2,057	63	27,671		383	10	35,802		66	1,549	34,581
	%	27	2		71	18	6	<1	76		1	<1	99		<1	4	96
Cottonwood	acres	82			317				399		82		317				399
	hectares	33			128				162		33		128				162
	%	21			79				100		21		79				100
Piñon/juniper	acres	663	838		17,499				19,000		1,502		17,499				19,000
	hectares	268	339		7,081				7,689		608		7,081				7,689
	%	3	4		92				100		8		92				100
Gambel oak	acres	461	57	1	35,904		27	19	36,379		223	231	35,971				36,424
	hectares	187	23	1	14,530		11	8	14,722		90	93	14,557				14,740
	%	1	<1	<1	99		<1	<1	100		1	1	99				100

* Cover types with host tree species present in mix

Gunnison Basin

The majority of Gunnison Basin GA consists of four cover types: spruce/fir (25% of the total GA), lodgepole pine (20%), grass/forb (19%), and aspen (14%). Existing conditions for all forest types are divided between sapling/pole and mature classification in dense stands with canopy closures greater than 40 percent.

Recent Insect and Disease Occurrences

The acreage and percent of each GA covered by the aerial detection surveys since 1995 are provided in Table 2. There were no flights over the Gunnison Basin GA in 1995, 1997, or 2003, with very limited coverage (<12% of GA area) in 1998, 2001, and 2002. Aerial survey detections illustrate patches of subalpine fir decline, aspen defoliation, and "other" (Figure 6-19). Number of trees killed and acres affected by various insects and diseases on the Gunnison Basin GA as detected by aerial survey are provided in Table 6-10. Subalpine fir decline has affected the largest number of trees and area since 1996, with number of trees killed and the affected area appearing to decrease greatly after 2000, though this may only be an artifact of the limited areal coverage of the survey data for 2001 and 2002.

Insect and diseases that currently occur or are/were a problem on the Gunnison Basin GA include:

- Aspen—The 2001 GMUG MIS Assessment reported that aspen mortality in mature stands has been increasing, due to the natural aging of these aspen stands (USDA Forest Service 2001). Various cankers and rots are common and active, with *Phellinus tremulae* being the most damaging in this area (R. Vermillion, personal communication).

- Douglas-fir—Fairly extensive areas of western spruce budworm and Douglas-fir beetle are along the Lake Fork of the Gunnison River, and Douglas-fir beetle is currently very active in the southeastern portion of the Gunnison RD between Green Mountain and Lion's Head, west of the town of Sargents (R. Vermillion, personal communication).

- Lodgepole—Ips beetle is active in Taylor Canyon, lodgepole dwarf mistletoe is very prominent in Taylor Park and Pitkin areas, and mountain pine beetle activity is reported (R. Vermillion, personal communication). Lodgepole pine dwarf mistletoe was the cause of sanitation thinning in 11 campgrounds on the Gunnison NF in 1995 (Johnson 1996).

- Ponderosa pine—Mountain pine beetle is reported (R. Vermillion, personal communication). Starting in the San Juan NF in the late 1990s, by 2002 ponderosa pine needlecast was discoloring foliage of ponderosa pines across the San Juan, Grand Mesa, Uncompahgre, and Gunnison NFs (USDA Forest Service 2003).

- Spruce/Fir—Spruce beetle is active in Red Creek and Alpine Plateau portions of the Gunnison RD, and western spruce budworm, Armillaria root disease, subalpine fir decline, western balsam bark beetle, conifer broom rusts are found throughout (R. Vermillion, personal communication).

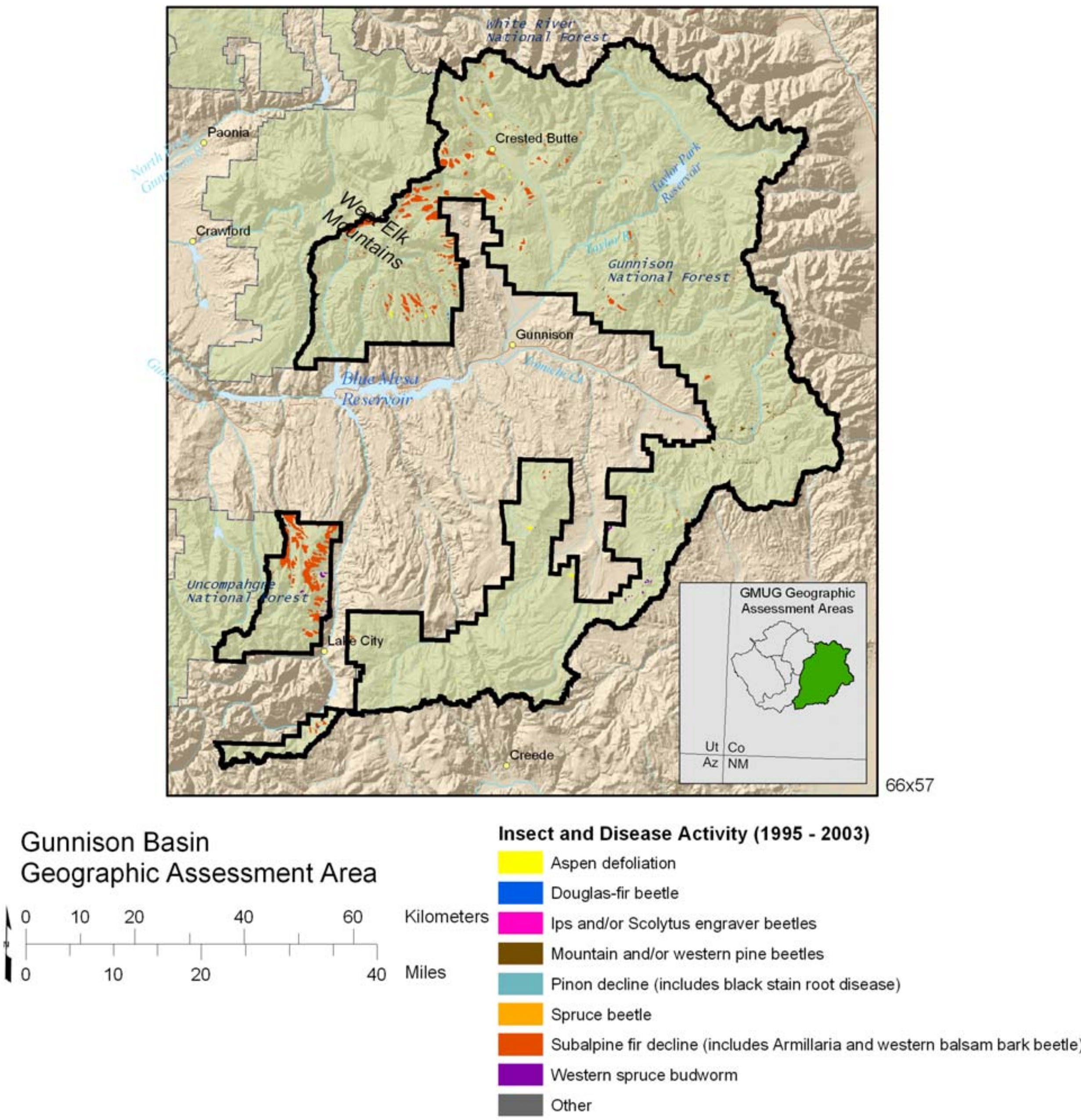


Figure 6-17 Recent insect and disease activity for Gunnison Basin Geographic Area. Recent insect and disease activity for Gunnison Basin Geographic Area ascertained using aerial detection surveys from 1995-2003. Other includes: foliage discoloration, miscellaneous mortality, oak leafroller, ponderosa pine needleminer, and unknown.

Table 6-10 . Forest health aerial survey results for the National Forest portion of the Gunnison Basin Geographic Area.
(note: there were no aerial survey data for 1995, 1997 and 2003 for this GA)

Damage Causing Agent	1996		1998		1999		2000		2001		2002	
	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected
Armillaria	30	17	--	--	--	--	--	--	--	--	--	--
Aspen Defoliation	na	754	--	--	na	287	na	41	--	--	--	--
Douglas-fir beetle	69	109	2	1	9	7	19	8	--	--	90	88
Ips spp.	--	--	--	--	145	30	136	20	20	2	--	--
Mistletoe	na	7,265	--	--	--	--	--	--	--	--	--	--
Mountain pine beetle	231	103	134	100	500	256	578	571	361	61	5	2
Piňon decline	--	--	--	--	--	--	--	--	--	--	--	--
Spruce beetle	15	13	--	--	--	--	--	--	--	--	20	6
Subalpine fir decline	64,857	15,452	15,173	6,908	8,302	1,947	30,728	5,154	357	96	12,718	5,043
Western spruce budworm	na	160	na	345	na	102	na	368	--	--	--	--
Other	na	203	--	--	--	--	na	5	--	--	--	--

na = not applicable—mortality was not caused by the specific agent.
Other = Diplodia blight, Douglas-fir needle cast, foliage discoloration, unidentified bark beetles, and/or unidentified mortality

Vulnerability to Future Infestations

Western spruce budworm (47% of the total GA land area in the high and moderate vulnerability ranking), spruce beetle (36%), mountain pine beetle 2 (29%), lodgepole pine dwarf mistletoe (29%), and Armillaria (24%) are all predicted to be some concern (Table 6-11).

Table 6-11 Vulnerability ranking for future insect/pathogen infestation for Gunnison Basin Geographic Area.

Organism	Units	Vulnerability Ranking High	Moderate	Low	None	Host Tree Species
Western spruce budworm	acres	564,006	83,010	353	742,318	Engelmann spruce, Douglas-fir, Subalpine fir
	hectares	228,246	33,593	143	300,407	
	%	41	6	<1	53	
Spruce beetle	acres	298,126	209,810	60,427	821,324	Engelmann spruce
	hectares	120,648	84,908	24,454	332,380	
	%	21	15	4	59	
Douglas-fir beetle	acres	28,450	77,510	3,847	1,279,881	Douglas-fir
	hectares	11,513	31,367	1,557	517,952	
	%	2	6	<1	92	
Mountain pine beetle 1	acres	5,419	11,595	2,840	1,369,833	Ponderosa pine
	hectares	2,193	4,692	1,149	554,354	
	%	<1	1	<1	99	
Mountain pine beetle 2	acres	317,896	84,241	4,830	982,720	Lodgepole pine
	hectares	128,649	34,091	1,955	397,694	
	%	23	6	<1	71	
Lodgepole pine dwarf mistletoe	acres	377,947	28,942	79	982,720	Lodgepole pine
	hectares	152,950	11,712	32	397,694	
	%	27	2	<1	71	
Armillaria	acres	133,457	195,945	82,951	977,335	Blue spruce, Douglas-fir, Engelmann spruce, Subalpine fir
	hectares	54,008	79,296	33,569	395,515	
	%	10	14	6	70	

Table 6-12 summarizes the areas of each cover type that are vulnerable to the seven insect/disease organisms listed in Table 6-11 for the Gunnison Basin GA. Spruce/fir, lodgepole pine, and aspen are the only cover types with substantive area in the high and moderate vulnerability rankings. Notably, almost all the Douglas-fir and over half the blue spruce and limber pine cover types (with host tree species in the mix) of the GA are also in the high and moderate vulnerability rankings for western spruce budworm, spruce beetle, Douglas-fir beetle, mountain pine beetle 2, and/or lodgepole pine dwarf mistletoe; however these cover types together account for only approximately 38,000 ac (15,000 ha) or 3% of the Gunnison Basin land area.

Table 6-12 Vulnerability ranking for seven modeled insect and disease organisms by forest cover types found within the Gunnison Basin Geographic Area.

Cover Type*	Units	Western spruce budworm vulnerability				Spruce beetle vulnerability				Douglas-fir beetle vulnerability				Mountain pine beetles 1 (ponderosa pine) vulnerability			
		High	Moderate	Low	None	High	Moderate	Low	None	High	Moderate	Low	None	High	Moderate	Low	None
Spruce/fir	acres	307,062	42,398	8	1,773	208,696	113,855	26,497	2,192	3,994	11,377	102	335,768	132	320		350,789
	hectares	124,264	17,158	3	717	84,457	46,076	10,723	887	1,616	4,604	41	135,881	53	129		141,960
	%	87	12	<1	1	59	32	8	1	1	3	<1	96	<1	<1		100
Bristlecone pine	acres	1,004	278		6,054	98	631	519	6,088		45		7,291	171	269	73	6,823
	hectares	406	112		2,450	40	255	210	2,464		18		2,950	69	109	29	2,761
	%	14	4		83	1	9	7	83		1		99	2	4	1	93
Douglas-fir	acres	29,508	3,331		361	4,656	3,641	242	24,662	10,956	21,569	314	361	1,665	1,755	9	29,772
	hectares	11,942	1,348		146	1,884	1,474	98	9,980	4,434	8,729	127	146	674	710	4	12,048
	%	89	10		1	14	11	1	74	33	65	1	1	5	5	<1	90
Blue spruce	acres	1,518			753	274	261		1,737	372	839		1,061		11		2,261
	hectares	614			305	111	106		703	150	340		429		4		915
	%	67			33	12	11		76	16	37		47		<1		100
Aspen	acres	84,077	6,518		109,766	32,032	35,587	1,653	131,035	6,350	19,124	564	174,270	558	1,324	89	198,390
	hectares	34,025	2,638		44,421	12,963	14,401	669	53,028	2,570	7,739	228	70,525	226	536	36	80,286
	%	42	3		55	16	18	1	65	3	10	<1	87	<1	1	<1	99
Lodgepole pine	acres	134,700	1,506		145,919	51,617	54,888	2,698	172,922	6,627	22,864	583	252,052	77	848	7	281,192
	hectares	54,511	610		59,051	20,889	22,212	1,092	69,979	2,682	9,253	236	102,002	31	343	3	113,795
	%	48	1		52	18	19	1	61	2	8	<1	89	<1	<1	<1	100
Limber pine	acres	293			292	278	15		292				585				585
	hectares	119			118	113	6		118				237				237
	%	50			50	48	3		50				100				100
Cottonwood	acres		14		715	14			715				729				729
	hectares		6		290	6			290				295				295
	%		2		98	2			98				100				100
Ponderosa	acres	2,024	349		8,329	461	159	45	10,036	152	1,587	13	8,950	2,816	7,068	401	416
	hectares	819	141		3,371	187	64	18	4,061	61	642	5	3,622	1,140	2,860	162	169
	%	19	3		78	4	1	<1	94	1	15	<1	84	26	66	4	4
Piñon/ juniper	acres	101			268				369		101		268				369
	hectares	41			109				149		41		109				149

	%	27			73				100		27		73				100

Table 6-12 continued.

Cover Type*	Units	Mountain pine beetles 2 (lodgepole pine) vulnerability				Lodgepole pine dwarf mistletoe vulnerability				Armillaria vulnerability			
		High	Moderate	Low	None	High	Moderate	Low	None	High	Moderate	Low	None
Spruce/fir	acres	43,144	23,453		284,644	59,980	6,617		284,644	129,968	185,863	34,743	667
	hectares	17,460	9,491		115,192	24,273	2,678		115,192	52,596	75,216	14,060	270
	%	12	7		81	17	2		81	37	53	10	0
Bristlecone pine	acres	49	142		7,144	152	40		7,144			148	7,188
	hectares	20	58		2,891	62	16		2,891			60	2,909
	%	1	2		97	2	1		97			2	98
Douglas-fir	acres	4,074	1,919		27,208	5,437	555		27,208	3,158	5,683	17,322	7,037
	hectares	1,648	777		11,011	2,200	225		11,011	1,278	2,300	7,010	2,848
	%	12	6		82	16	2		82	10	17	52	21
Blue spruce	acres	170	217		1,884	183	205		1,884	331	168	941	831
	hectares	69	88		762	74	83		762	134	68	381	336
	%	8	10		83	8	9		83	15	7	41	37
Aspen	acres	22,574	23,558	8	154,221	42,944	3,195		154,221		48	8,573	191,740
	hectares	9,135	9,534	3	62,411	17,379	1,293		62,411		19	3,470	77,595
	%	11	12	<1	77	21	2		77		0	4	96
Lodgepole pine	acres	247,566	33,314	838	408	269,010	12,685	22	408		4,183	20,378	257,564
	hectares	100,187	13,482	339	165	108,865	5,133	9	165		1,693	8,247	104,233
	%	88	12	<1	<1	95	4	<1	<1		1	7	91
Limber pine	acres	272	12		300	220	65		300				585
	hectares	110	5		122	89	26		122				237
	%	47	2		51	38	11		51				100
Cottonwood	acres				729				729				729
	hectares				295				295				295
	%				100				100				100
Ponderosa	acres		143		10,559	21	121		10,559			250	10,452
	hectares		58		4,273	9	49		4,273			101	4,230
	%		1		99	<1	1		99			2	98
Piñon/ juniper	acres				369				369				369
	hectares				149				149				149
	%				100				100				100

* Cover types with host tree species present in mix

North Fork Valley

The majority of the North Fork Valley GA consists of three cover types: aspen (40% of the total GA), spruce/fir (23%), and Gambel oak (14%). Existing conditions for all forest types are mature, dense stands with canopy closures greater than 40 percent.

Recent Insect and Disease Occurrences

The acreage and percent of each GA covered by the aerial detection surveys since 1995 are provided in Table 6-6. There were no flights over the North Fork Valley GA in 1995 or 2000, with very limited coverage (<14% of GA area) in 1997, 1998, 1999, and 2001. Aerial survey detections illustrate some larger patches of subalpine fir decline, and small patches of aspen defoliation and Douglas-fir beetle as the only real concern for the North Fork Valley GA between 1995-2003 (Figure 6-18). Subalpine fir mortality appears to have increased in recent years, as does Douglas-fir and spruce beetle, though in smaller numbers compared with subalpine fir decline since 1996 on the North Fork Valley GA (Table 6-13). Again, the limited coverage of the survey data for this GA, may not allow reliable conclusions about trends.

Insect and diseases that currently occur or are a problem on the North Fork Valley GA include:

- Aspen—The 2001 GMUG MIS Assessment reported that aspen mortality in mature stands has been increasing, due to the natural aging of these aspen stands (USDA Forest Service 2001). Various cankers and rots are common and active, but they generally do not show up as "epidemics" (C. McKenzie, personal communication). Aspen defoliating and boring insects are not present in any large scale (C. McKenzie, personal communication).

- Ponderosa pine—Starting in the San Juan NF in the late 1990s, by 2002 ponderosa pine needlecast was discoloring foliage of ponderosa pines across the San Juan, Grand Mesa, Uncompahgre, and Gunnison NFs (USDA Forest Service 2003).

- Spruce/Fir—spruce beetle activity is high at Stephens Gulch (C. McKenzie, personal communication). Subalpine fir decline from western balsam bark beetle in combination with Armillaria root disease is present (C. McKenzie, personal communication). Red ring rot and conifer broom rusts are common throughout but not in large specific areas (C. McKenzie, personal communication).

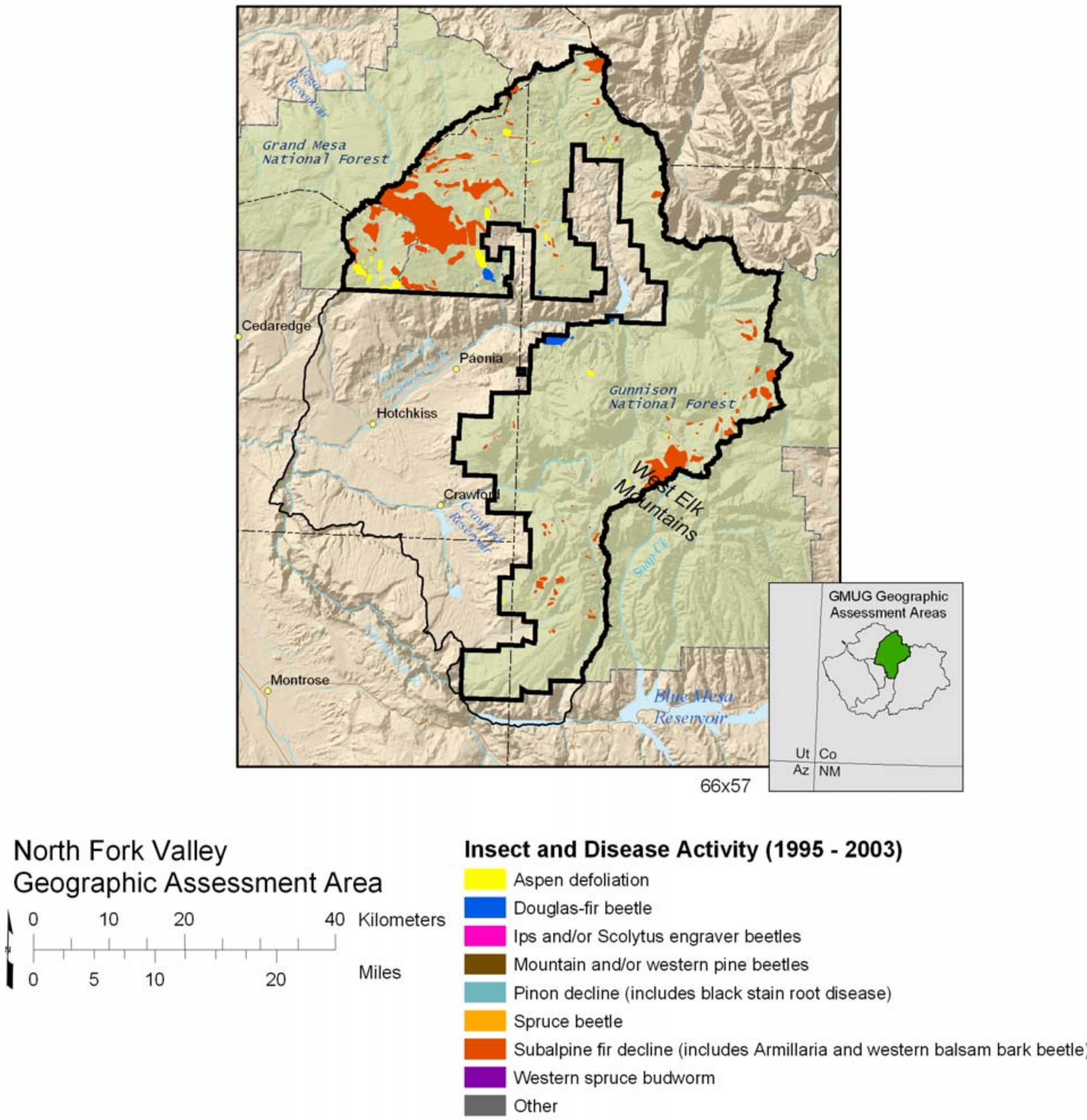


Figure 6-18 Recent insect and disease activity for North Fork Valley Geographic Area.

Recent insect and disease activity for North Fork Valley Geographic Area ascertained using aerial detection surveys from 1995-2003. Other includes: foliage discoloration, miscellaneous mortality, oak leafroller, ponderosa pine needleminer, and unknown.

Table 6-13 Forest health aerial survey results for the National Forest portion of the North Fork Valley Geographic Area. (note: there were no aerial survey data for 1995 and 2000 for this GA)

Damage Causing Agent	1996		1997		1998		1999		2001		2002		2003	
	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected
Aspen Defoliation	na	886	na	1,338	--	--	--	--	na	42	na	1,197	na	83
Douglas-fir beetle	91	1,139	--	--	--	--	6	4	--	--	1,450	640	1,748	361
Mountain pine beetle	2	0	--	--	--	--	--	--	--	--	--	--	2	9
Spruce beetle	2	0	--	--	--	--	--	--	47	10	29	46	139	113
Subalpine fir decline	44,021	18,198	3,834	4,940	63	0	914	408	351	136	14,357	11,195	21,574	10,344
Other	--	--	--	--	--	--	--	--	--	--	--	--	na	7

na = not applicable—mortality was not caused by the specific agent.
Other = Diplodia blight, Douglas-fir needle cast, foliage discoloration, mistletoe, unidentified bark beetles, and/or unidentified mortality

Vulnerability to Future Infestations

Western spruce budworm (43% of the total GA land area in the high and moderate vulnerability ranking), spruce beetle (38%), and Armillaria (23%) are all of some concern for their respective host species (Table 6-14).

Table 6-14 Vulnerability ranking for future insect/pathogen infestation for North Fork Valley Geographic Area.

Organism	Units	Vulnerability Ranking				Host Tree Species
		High	Moderate	Low	None	
Western spruce budworm	acres	174,446	41,695	86	288,582	Engelmann spruce, Douglas-fir, Subalpine fir
	hectares	70,596	16,873	35	116,785	
	%	35	8	<1	57	
Spruce beetle	acres	114,912	75,027	20,669	294,201	Engelmann spruce
	hectares	46,503	30,363	8,364	119,059	
	%	23	15	4	58	
Douglas-fir beetle	acres	1,748	5,710	1,034	496,317	Douglas-fir
	hectares	707	2,311	418	200,853	
	%	<1	1	<1	98	
Mountain pine beetle 1	acres	53	230	35	504,490	Ponderosa pine
	hectares	22	93	14	204,161	
	%	<1	<1	<1	100	
Mountain pine beetle 2	acres				504,809	Lodgepole pine
	hectares				204,290	
	%				100	
Lodgepole pine dwarf mistletoe	acres				504,809	Lodgepole pine
	hectares				204,290	
	%				100	
Armillaria	acres	54,685	60,717	14,974	374,434	Blue spruce, Douglas-fir, Engelmann spruce, Subalpine fir
	hectares	22,130	24,571	6,060	151,529	
	%	11	12	3	74	

Table 6-15 summarizes the areas of each cover type that are vulnerable to the seven insect/disease organisms listed in Table 6-14 for the North Fork Valley Geographic Area (note: only five of the seven organism had any vulnerable area). Spruce/fir and aspen are the only cover types with substantive area in the high and moderate vulnerability rankings for western spruce budworm, spruce beetle, and Armillaria. Notably, almost all the Douglas-fir and a substantial portion of ponderosa pine cover types (with host species present in mix) in the GA are also in the high and moderate vulnerability rankings for western spruce budworm, Douglas-fir beetle, mountain pine beetle 1, and/or Armillaria; however these cover types together account for only approximately 1,500 ac (600 ha) or <1% of the North Fork Valley GA land area.

Table 6-15 Vulnerability ranking for seven modeled insect and disease organisms by forest cover types found within the North Fork Valley Geographic Area.

Cover Type*	Units	Western spruce budworm vulnerability				Spruce beetle vulnerability				Douglas-fir beetle vulnerability				Mountain pine beetles 1 (ponderosa pine) vulnerability			
		High	Moderate	Low	None	High	Moderate	Low	None	High	Moderate	Low	None	High	Moderate	Low	None
Spruce/fir	acres	99,745	22,467		1,469	74,466	40,415	6,876	1,883	829	2,082	149	120,581		7		123,674
	hectares	40,366	9,092		594	30,136	16,356	2,783	762	335	842	60	48,798		3		50,049
	%	81	18		1	60	33	6	2	1	2	<1	97		<1		100
Douglas-fir	acres	792	275		9	163	200		714	284	762	20	9		20		1,056
	hectares	320	111		4	66	81		289	115	309	8	4		8		427
	%	74	26		1	15	19		66	26	71	2	1		2		98
Aspen	acres	70,422	8,005		128,610	40,172	32,981	1,884	131,993	635	2,602	249	203,545	2	53	13	206,969
	hectares	28,499	3,240		52,047	16,257	13,347	763	53,416	257	1,053	101	82,372	1	22	5	83,758
	%	34	4		62	19	16	1	64	<1	1	<1	98	<1	<1	<1	100
Cottonwood	acres	83	27		707	110			707				818				818
	hectares	34	11		286	45			286				331				331
	%	10	3		86	14			86				100				100
Ponderosa	acres	41			212				253		41		212	51	150	17	34
	hectares	16			86				102		16		86	21	61	7	14
	%	16			84				100		16		84	20	59	7	14
Piñon/juniper	acres	230			2,562				2,792		195		2,597				2,792
	hectares	93			1,037				1,130		79		1,051				1,130
	%	8			92				100		7		93				100
Gambel oak	acres	482	757		58,802		236	849	58,957		20	134	59,887				60,041
	hectares	195	306		23,797		95	344	23,859		8	54	24,236				24,298
	%	1	1		98		<1	1	98		<1	<1	100				100

Table 6-15 continued.					
Cover Type*	Units	Armillaria vulnerability			
		High	Moderate	Low	None
Spruce/fir	acres	54,549	60,490	8,134	508
	hectares	22,075	24,479	3,292	205
	%	44	49	7	0
Douglas-fir	acres	136	227	505	209
	hectares	55	92	204	84
	%	13	21	47	19
Aspen	acres			6,039	200,998
	hectares			2,444	81,341
	%			3	97
Cottonwood	acres				818
	hectares				331
	%				100
Ponderosa	acres				253
	hectares				102
	%				100
Piñon/juniper	acres				2,792
	hectares				1,130
	%				100
Gambel oak	acres				60,041
	hectares				24,298
	%				100
* Cover types with host tree species present in mix					

San Juan

The majority of the San Juan GA consists of four cover types: spruce/fir (36% of the total GA), aspen (20%), grass/forb (19%), and bare/rock (19%). Existing conditions for all forest types are mature, dense stands with canopy closures greater than 40 percent.

Recent Insect and Disease Occurrences

The acreage and percent of each GA covered by the aerial detection surveys since 1995 are provided in Table 6-6. There were no flights over the San Juan GA in 1995, 1999, or 2001, with very limited coverage (<17% of GA area) in 1996, 1997, and 2003. Based on aerial survey detections, subalpine fir decline and western spruce budworm appear to be the largest concern, with small patches of aspen defoliation and spruce beetle evident from 1996-2003 for the San Juan GA (Figure 6-19). Aerial detection of subalpine fir decline (number of trees killed and area affected) increased dramatically between 1998 and 2002, but appears to decline in 2003, which is likely an artifact of the limited flight coverage in 2003 (Table 6-6; Table 6-16). Other insect and disease organisms have remained at relatively low, stable levels over the past decade.

Insect and diseases that currently occur or are/were a problem on the San Juan GA include:

- Aspen—The 2001 GMUG MIS Assessment reported that aspen mortality in mature stands has been increasing, due to the natural aging of these aspen stands (USDA Forest Service 2001).
    - Ponderosa pine—Moderate bark beetle activity in the Love Mesa area (T49N, R14W, sections 7, 8, 17, and 18) has been going on for years but has not intensified (Tim Garvey, personal communication). Starting in the San Juan NF in the late 1990s, by 2002 ponderosa pine needlecast was discoloring foliage of ponderosa pines across the San Juan, Grand Mesa, Uncompahgre, and Gunnison NFs (USDA Forest Service 2003).
- Spruce/Fir—Spruce beetle activity is heavy around Ouray Spring (T47N, R13W, section 1) affecting about 200 acres, especially south of the Divide Road, and moderate-heavy mortality is reported around T48N, R15W, section 6 (Tim Garvey, personal communication). Subalpine fir mortality is the most significant problem (Tim Garvey, personal communication). In 1995, the Annosus root disease was responsible for tree failure (spruce/fir and/or ponderosa pine) in the Amphitheater Campground, Ouray RD, Uncompahgre NF (Johnson 1996). In 1996, a sanitation project was implemented at the Amphitheater Campground, Ouray RD, Uncompahgre NF (Johnson 1997). In 1995, western spruce budworm defoliation of white fir continued at the Amphitheater Campground, Ouray RD, Uncompahgre NF (Johnson 1996). Moderate defoliation of Engelmann spruce was detected in the northern San Juan Mountains near Ouray (Harris et al. 2001; Harris et al. 2002).

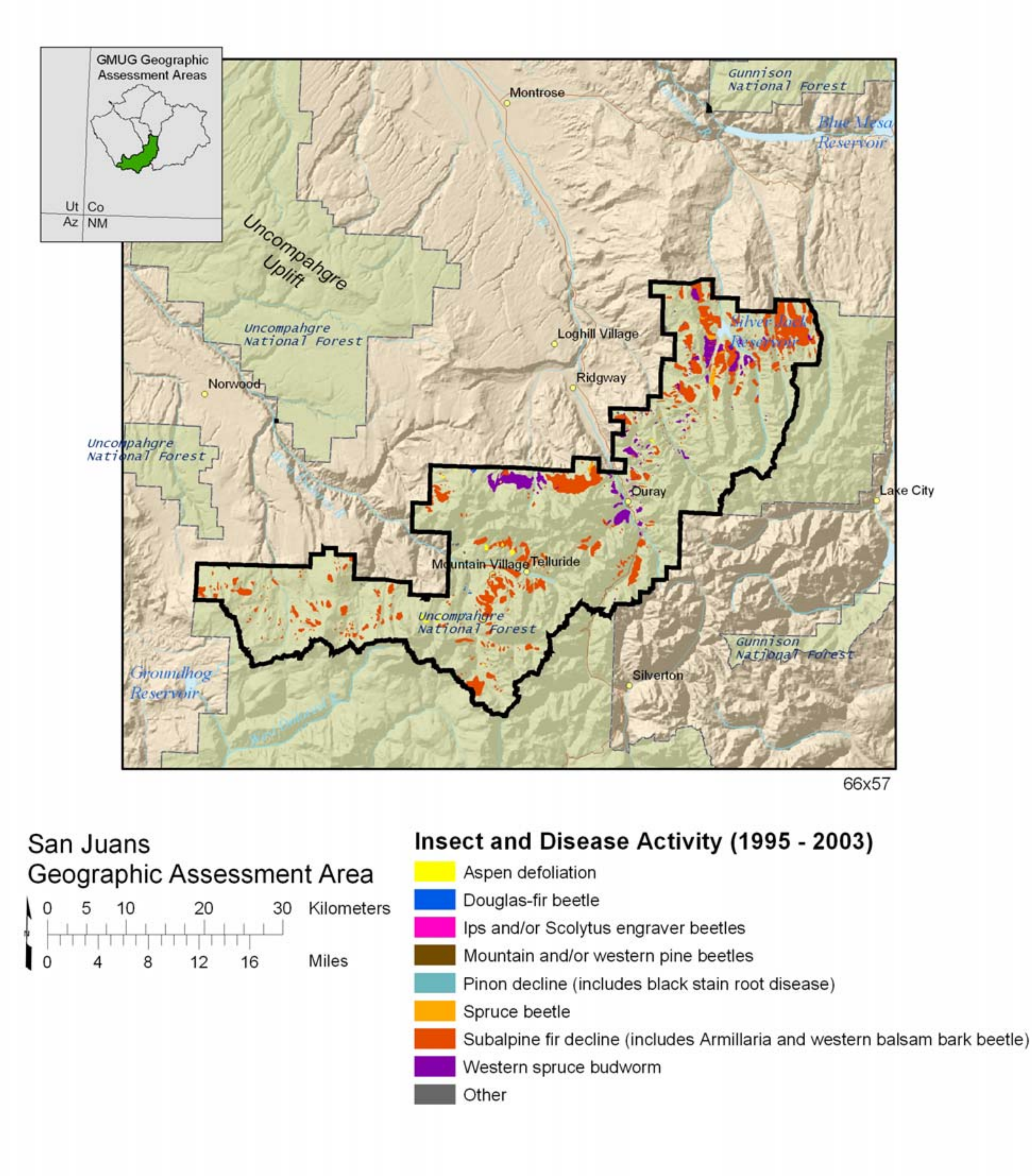


Figure 6-19 Recent insect and disease activity for San Juan Geographic Area.
Recent insect and disease activity for San Juan Geographic Area ascertained using aerial detection surveys from 1995-2003. Other includes: foliage discoloration, miscellaneous mortality, oak leafroller, ponderosa pine needleminer, and unknown.

Table 6-16 Forest health aerial survey results for the National Forest portion of the San Juan Geographic Area.
(note: there were no aerial survey data for 1995, 1999, and 2001 for this GA)

Damage Causing Agent	1996		1997		1998		2000		2002		2003	
	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected
Aspen Defoliation	--	--	--	--	na	547	na	78	--	--	--	--
Douglas-fir beetle	--	--	--	--	46	122	20	21	42	34	--	--
Ips spp.	--	--	--	--	--	--	--	--	--	--	12	10
Mountain pine beetle	2	0					4	8	24	38	--	--
Piňon decline	--	--	--	--	--	--	--	--	--	--	10	5
Spruce beetle	--	--	--	--	353	186	48	38	859	566	--	--
Subalpine fir decline	244	155	150	119	22,226	12,104	40,517	10,234	32,424	23,400	58	17
Western spruce budworm	--	--	--	--	na	3,480	na	1,167	na	4,081	--	--
Other	--	--	--	--	na	357	na	78	--	--	--	--

na = not applicable—mortality was not caused by the specific agent.
Other = Diplodia blight, Douglas-fir needle cast, foliage discoloration, mistletoe, unidentified bark beetles, and/or unidentified mortality

Vulnerability to Future Infestations

Western spruce budworm (52% of the total GA land area in the high and moderate vulnerability ranking), spruce beetle (42%), and armillaria (36%) are all of concern for their respective host species (Table 6-17).

Table 6-17 Vulnerability ranking for future insect/pathogen infestation for San Juan Geographic Area.

Organism	Units	Vulnerability Ranking High	Moderate	Low	None	Host Tree Species
Western spruce budworm	acres	96,959	89,618	199	169,914	Engelmann spruce, Douglas-fir, Subalpine fir
	hectares	39,238	36,267	80	68,762	
	%	27	25	<1	48	
Spruce beetle	acres	122,416	29,818	21,492	182,963	Engelmann spruce
	hectares	49,540	12,067	8,698	74,043	
	%	34	8	6	51	
Douglas-fir beetle	acres	1,085	6,240	281	349,084	Douglas-fir
	hectares	439	2,525	114	141,270	
	%	<1	2	<1	98	
Mountain pine beetle 1	acres	30	865	168	355,627	Ponderosa pine
	hectares	12	350	68	143,918	
	%	<1	<1	<1	100	
Mountain pine beetle 2	acres		99	142	356,449	Lodgepole pine
	hectares		40	57	144,250	
	%		<1	<1	100	
Lodgepole pine dwarf mistletoe	acres		241	356,449		Lodgepole pine
	hectares		97	144,250		
	%		<1	100		
Armillaria	acres	85,477	43,187	9,936	218,089	Blue spruce, Douglas-fir, Engelmann spruce, Subalpine fir
	hectares	34,592	17,477	4,021	88,258	
	%	24	12	3	61	

Table 6-18 summarizes the areas of each cover type that are vulnerable to the seven insect/disease organisms listed in Table 6-17 for the San Juan GA. Spruce/fir and aspen are the only cover types with substantive area in the high and moderate vulnerability rankings for western spruce budworm, spruce beetle, and/or Armillaria. Notably, a substantial portion of Douglas-fir, lodgepole pine, cottonwood, and ponderosa pine cover types (with host tree species present in mix) in the GA are also in the high and moderate vulnerability rankings for western spruce budworm, spruce beetle, Douglas-fir beetle, mountain pine beetle 1 and 2, lodgepole pine dwarf mistletoe and/or Armillaria; however these cover types together account for only approximately 4,500 ac (1,800 ha) or 1% of the San Juan GA land area.

Table 6-18 Vulnerability ranking for seven modeled insect and disease organisms by forest cover types found within the San Juan Geographic Area.

Cover Type*	Units	Western spruce budworm vulnerability				Spruce beetle vulnerability				Douglas-fir beetle vulnerability				Mountain pine beetles 1 (ponderosa pine) vulnerability			
		High	Moderate	Low	None	High	Moderate	Low	None	High	Moderate	Low	None	High	Moderate	Low	None
Spruce/fir	acres	88,853	44,294	21	14	103,184	23,579	5,556	863	170	2,350	20	130,642			27	133,155
	hectares	35,958	17,925	9	6	41,757	9,542	2,249	349	69	951	8	52,869			11	53,886
	%	67	33	<1	<1	77	18	4	1	<1	2	<1	98			<1	100
Douglas-fir	acres	1,404	931	20		571	488		1,295	626	1,697	31		13	314	83	1,944
	hectares	568	377	8		231	198		524	254	687	12		5	127	33	787
	%	60	40	1		24	21		55	27	72	1		1	13	4	83
Blue spruce	acres				32				32				32				32
	hectares				13				13				13				13
	%				100				100				100				100
Aspen	acres	6,545	25,684		37,882	18,570	5,462	1,617	44,461	289	1,752	61	68,009		110		70,001
	hectares	2,649	10,394		15,330	7,515	2,210	655	17,993	117	709	25	27,522		45		28,328
	%	9	37		54	26	8	2	63	<1	2	<1	97		<1		100
Lodgepole pine	acres		203					203					203				203
	hectares		82					82					82				82
	%		100					100					100				100
Cottonwood	acres	125	32		190	91		65	190				346				346
	hectares	50	13		77	37		26	77				140				140
	%	36	9		55	26		19	55				100				100
Ponderosa	acres		441		17				457		441		17	17	441		
	hectares		178		7				185		178		7	7	178		
	%		96		4				100		96		4	4	96		
Piñon/ juniper	acres				616				616				616				616
	hectares				249				249				249				249
	%				100				100				100				100

Table 6-18 continued.													
Cover Type*	Units	Mountain pine beetles 2 (lodgepole pine) vulnerability				Lodgepole pine dwarf mistletoe vulnerability				Armillaria vulnerability			
		High	Moderate	Low	None	High	Moderate	Low	None	High	Moderate	Low	None
Spruce/fir	acres			38	133,144		38		133,144	85,249	42,356	5,576	
	hectares			15	53,882		15		53,882	34,499	17,141	2,257	
	%			<1	100		<1		100	64	32	4	
Douglas-fir	acres				2,354				2,354	228	831	1,099	196
	hectares				953				953	92	336	445	79
	%				100				100	10	35	47	8
Blue spruce	acres				32				32			32	
	hectares				13				13			13	
	%				100				100			100	
Aspen	acres				70,111				70,111			2,874	67,237
	hectares				28,373				28,373			1,163	27,210
	%				100				100			4	96
Lodgepole pine	acres		99	104			203						203
	hectares		40	42			82						82
	%		49	51			100						100
Cottonwood	acres				346				346				346
	hectares				140				140				140
	%				100				100				100
Ponderosa	acres				457				457				457
	hectares				185				185				185
	%				100				100				100
Piñon/ juniper	acres				616				616				616
	hectares				249				249				249
	%				100				100				100
* Cover types with host tree species present in mix													

Uncompahgre Plateau

The majority of the Uncompahgre Plateau GA consists of four cover types: aspen (25% of the total GA), Gambel oak (25%), pinon/juniper (16%), and ponderosa pine (15%). Existing conditions in all forest types are dominated by trees classified as mature, in dense stands with canopy closures greater than 40 percent with average ages greater than 100 years old.

Recent Insect and Disease Occurrences

The acreage and percent of each GA covered by the aerial detection surveys since 1995 are provided in Table 6-6. There were no flights over the Uncompahgre Plateau GA in 1999, with limited coverage (<23% of GA area) in 1998 and 2000. The 1996 flight covered most of the Geographic Area, with the exception of the northern half of the Naturita Division. The 1997 flight covered the Fruita Division and the north half of the Uncompahgre Plateau. The 1998 and 2000 flights covered the southern 1/3 of the Geographic Area. There was no overflight of the Geographic Area in 1999. The 2001 flight covered all but the Naturita Division, and the 2002 flight covered essentially the entire Geographic Area.

Aerial detection surveys illustrate patches of subalpine fir decline (many small patches), western spruce budworm (very large swath), and piñon decline as the most prominent insect and disease organisms, with small isolated patches of aspen defoliation, *Ips* spp., and mountain pine beetle over the past 10 years (Figure 6-20).

Table 6-19 summarizes information from aerial surveys from 1995 through 2003. Douglas-fir beetle, Ips spp., piñon decline, subalpine fir decline, and western pine beetle all appear to be increasing in recent years (Table 6-19).

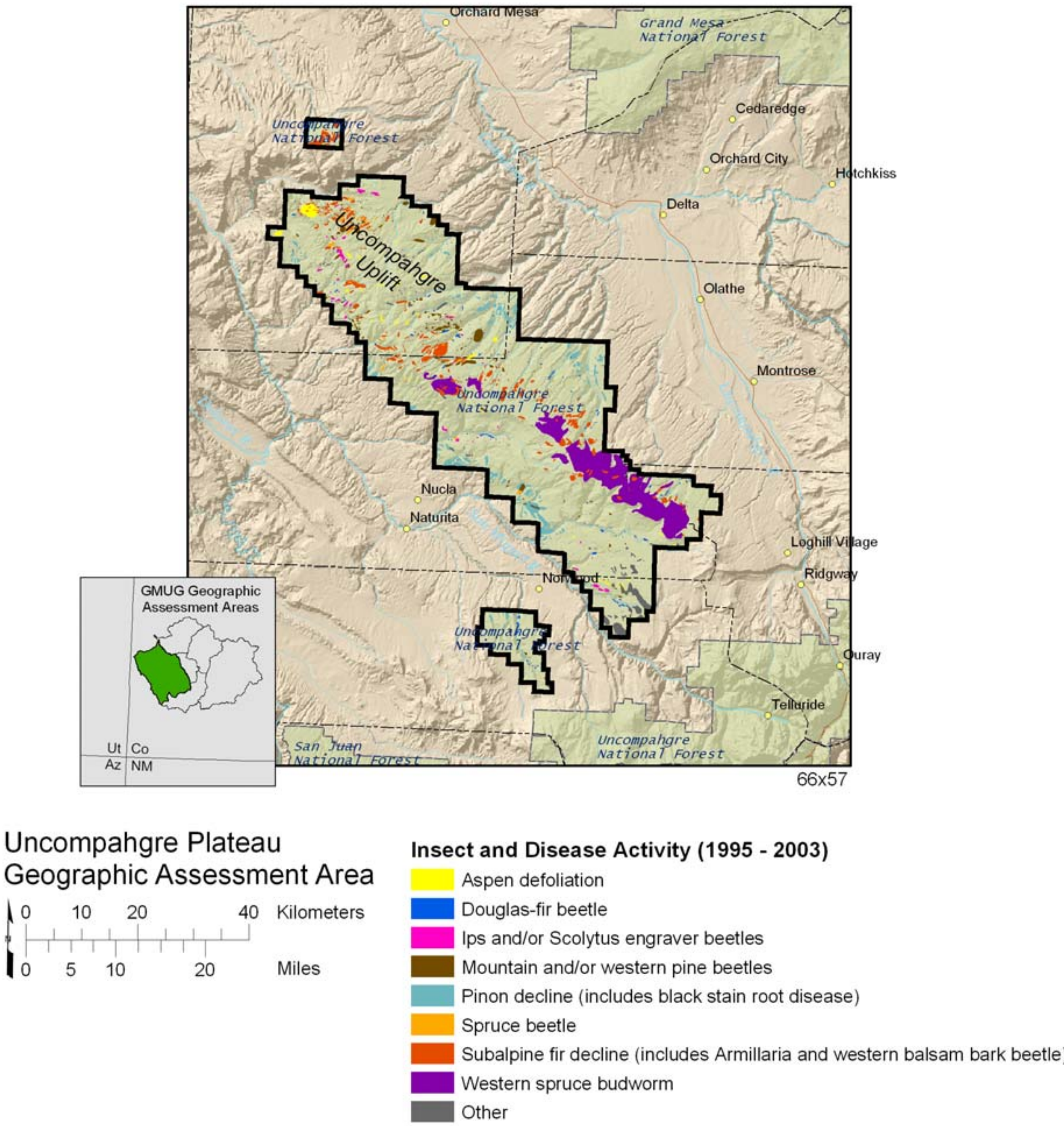


Figure 6-20 Recent insect and disease activity for Uncompahgre Geographic Area.

Recent insect and disease activity for Uncompahgre Geographic Area ascertained using aerial detection surveys from 1995-2003. Other includes: foliage discoloration, miscellaneous mortality, oak leafroller, ponderosa pine needleminer, and unknown.

Table 6-19 Forest health aerial survey results for the National Forest portion of the Uncompahgre Plateau Geographic Area.
(note: there were no aerial survey data for 1999 for this GA)

Damage Causing Agent	1995		1996		1997		1998		2000		2001		2002		2003	
	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected	# Dead Trees	Acres Affected
Aspen Defoliation	--	--	na	517	na	1,722	--	--	na	3	na	764	na	192	na	790
Douglas-fir beetle	19	35	24	48	139	187	121	223	233	98	271	265	773	447	2,647	1,468
Ips spp.	--	--	--	--	--	--	13	5	--	--	10	16	7	20	2,400	2,472
Mountain pine beetle	862	700	920	1,267	256	502	22	75	30	66	3,002	2,780	718	1,343	--	--
Oak leafroller	--	--	--	--	--	--	--	--	na	47	--	--	--	--	--	--
Piňon decline	--	--	--	--	--	--	na	456	na	1,499	--	--	na	71	196,869	14,692
Spruce beetle	2,407	1,750	8	5	--	--	--	--	111	36	283	409	43	28	68	61
Subalpine fir decline	--	--	3,520	2,454	6,376	3,431	26	52	--	--	18,448	7,916	7,595	3,014	20,424	8,979
Western pine beetle	--	--	--	--	--	--	--	--	--	--	--	--	--	--	372	966
Western spruce budworm	--	--	--	--	--	--	--	--	na	6,827	na	12,619	na	35,342	na	43
Other	--	--	na	255	157	765	na	6309	na	6	10	532	na	92	5	525

na = not applicable—mortality was not caused by the specific agent.
Other = Diplodia blight, Douglas-fir needle cast, foliage discoloration, mistletoe, unidentified bark beetles, and/or unidentified mortality

Insect and diseases that currently occur or are/were a problem on the Uncompahgre Plateau GA include:

- Aspen—The 2001 GMUG MIS Assessment reported that aspen mortality in mature stands has been increasing, due to the natural aging of these aspen stands (USDA Forest Service 2001). Various cankers and rots are common and active, but they generally do not show up as "epidemics" (C. McKenzie, personal communication). Aspen defoliating and boring insects are not present in any large scale (C. McKenzie, personal communication). Fungal pathogens of aspen are reported throughout the aspen cover type on the Plateau (USDA Forest Service 2000d). Cytospora (*Cytospora* spp.), Cryptosphaeria (*Cryptosphaeria populina*), and sooty bark (*Encoelia pruinosa*) cankers and aspen trunk rot (*Phellinus tremulae*) were common throughout Colorado in 1995 and were noted as affecting management in the Clear Creek Timber Sale area, Norwood RD of the Uncompahgre NF (Johnson 1996).

- Douglas-fir—Douglas-fir dwarf mistletoe was a concern in 1996 for Long Creek area on the Uncompahgre Plateau, Uncompahgre NF (Johnson 1997). The southern portion of the Uncompahgre Plateau has seen significant levels of western spruce budworm defoliation Douglas-fir cover type (Johnson 2002, Harris et al 2002, Harris 2003).

- Piñon/Juniper—*Ips confusus* is effecting the area, and twig beetle activity is starting to dramatically increase (C. McKenzie, personal communication). In 1995, dieback on piñon/juniper was observed in the Dry Park and Hoarsefly Canyon areas of the Norwood RD, Uncompahgre NF (Johnson 1996). As of 2002, black stain root disease is affecting piñon in isolated areas of the Uncompahgre Plateau in Colorado, contributing to piñon decline (USDA Forest Service 2003).

- Ponderosa pine—Starting in the San Juan NF in the late 1990s, by 2002 ponderosa pine needlecast was discoloring foliage of ponderosa pines across the San Juan, Grand Mesa, Uncompahgre, and Gunnison NFs (USDA Forest Service 2003), and is currently fairly heavy in the Pine Mountain and Campbell Point areas (C. McKenzie, personal communication). Mountain pine beetle was a problem in the past for the Campbell Point and Kelso areas (C. McKenzie, personal communication). In the 1980s, a significant bark beetle outbreak occurred in the Uncompahgre Plateau Geographic Area of the GMUG NFs. Three bark beetle species were attacking ponderosa pine in the outbreak—mountain pine beetle, roundheaded pine beetle (*Dendroctonus adjunctus*), and western pine beetle. Over 67,000 acres were affected and over 214,000 trees killed, mostly on the Norwood District. Some areas had extensive mortality. The most severely impacted region was located in the Ute area, encompassing 1,660 acres. Many of these areas were subsequently planted with ponderosa pine in part to reduce the risk that these sites would convert to Gambel oak. (See Timber Management section for discussion of planting success) This outbreak of bark beetles on the Uncompahgre Plateau was followed by salvage and sanitation logging and prescribed burning. Approximately 8,200 acres of salvage and sanitation harvests were done. The prescribed burning reduced fuels buildup and reintroduced fire to the forest ecosystem in localized areas. In 1992, a risk assessment study was completed to determine the susceptibility of stands on the Uncompahgre Plateau to bark beetle attack (USDA Forest Service 1992). This study modeled growth to predict future forest stand conditions and associated risk factors to forest insects and diseases. Single-storied stands with trees averaging greater than 10 inches in diameter (diameter at breast height) and stand density greater than 150 square feet basal area were considered at the highest risk for beetle outbreak. Since this risk assessment was made, approximately 3,700 acres of these ponderosa pine stands have been commercially thinned and 1,600 acres have been precommercially thinned (USDA Forest Service, RMACT database). Thinning was designed to reduce stand densities and improve tree age diversity to lessen the potential risk of bark beetle outbreaks. Two small outbreaks of mountain pine beetle were noted on the Uncompahgre Plateau GA of the GMUG NFs in 1995 (Johnson 1996). Mortality continued to increase in ponderosa and lodgepole pine, particularly on the Uncompahgre Plateau (Johnson 1997). During 2000-2001 beetle activity was increasing, but not yet at outbreak stage, for ponderosa pine stands on the northern portion of the Uncompahgre Plateau (Harris et al. 2002).

- Spruce/Fir— Spruce beetle population increases continued in 2002 in the spruce/fir forest type on the Uncompahgre Plateau NF portion of the GMUG (Harris 2003). Subalpine fir decline from western balsam bark beetle in combination with Armillaria root disease is present (C. McKenzie, personal communication). Subalpine fir mortality is especially significant in the eastern portion of the Uncompahgre Plateau between Dave Wood and Transfer Roads (Tim Garvey, personal communication). In 1996, subalpine fir damage from Armillaria was particularly high on the Uncompahgre Plateau at Ouray Springs (Johnson 1997). Red ring rot and conifer broom rusts are common throughout but not in large specific areas (C. McKenzie, personal communication). In 1995, western spruce budworm defoliation increased on the Uncompahgre Plateau, indicating expanding populations, (Johnson 1996). In 2000-2001, the southern portion of the Uncompahgre Plateau had significant levels of Engelmann spruce defoliation (Harris et al. 2002). As of 2002, this insect has lightly defoliated Engelmann spruce in areas of the Uncompahgre Plateau (Harris 2003).

Vulnerability to Future Infestations

Mountain pine beetle 1 (25% of the total GA land area in the high and moderate vulnerability ranking), western spruce budworm (18%), and spruce beetle (15%) are the organisms affecting substantive area (Table 6-20).

Table 6-20 Vulnerability ranking for future insect/pathogen infestation for Uncompahgre Plateau Geographic Area.

Organism	Units	Vulnerability Ranking				Host Tree Species
		High	Moderate	Low	None	
Western spruce budworm	acres	81,660	28,502	362	504,271	Engelmann spruce, Douglas-fir, Subalpine fir
	hectares	33,047	11,534	146	204,072	
	%	13	5	<1	82	
Spruce beetle	acres	72,739	18,844	2,060	521,151	Engelmann spruce
	hectares	29,437	7,626	834	210,903	
	%	12	3	<1	85	
Douglas-fir beetle	acres	6,018	19,777	1,181	587,820	Douglas-fir
	hectares	2,435	8,004	478	237,883	
	%	1	3	<1	96	
Mountain pine beetle 1	acres	82,213	76,531	20,832	435,219	Ponderosa pine
	hectares	33,271	30,971	8,430	176,128	
	%	13	12	3	71	
Mountain pine beetle 2	acres	213	154	145	614,282	Lodgepole pine
	hectares	86	62	59	248,592	
	%	<1	<1	<1	100	
Lodgepole pine dwarf mistletoe	acres	234	279		614,282	Lodgepole pine
	hectares	95	113		248,592	
	%	<1	<1		100	
Armillaria	acres	14,323	27,117	15,999	557,357	Blue spruce, Douglas-fir, Engelmann spruce, Subalpine fir
	hectares	5,796	10,974	6,475	225,555	
	%	2	4	3	91	

Table 6-21 summarizes the areas of each cover type that are vulnerable to the seven insect/disease organisms listed in Table 6-20 for the Uncompahgre Plateau Geographic Area. Aspen, spruce/fir, and ponderosa pine are the only cover types with substantive area in the high and moderate vulnerability rankings for western spruce budworm, spruce beetle, mountain pine beetle 1, and/or Armillaria. Notably, a substantial portion of Douglas-fir and lodgepole pine cover types (with host tree species in mix) in the GA are also in the high and moderate vulnerability rankings for western spruce budworm, spruce beetle, Douglas-fir beetle, mountain pine beetle 1 and 2, lodgepole pine dwarf mistletoe and/or Armillaria; however these cover types together account for only approximately 2,800 ac (1,130 ha) or <1% of the Uncompahgre Plateau GA land area.

Table 6-21 Vulnerability ranking for seven modeled insect and disease organisms by forest cover types found within the Uncompahgre Plateau Geographic Area.

Cover Type*	Units	Western spruce budworm vulnerability				Spruce beetle vulnerability				Douglas-fir beetle vulnerability				Mountain pine beetles 1 (ponderosa pine) vulnerability			
		High	Moderate	Low	None	High	Moderate	Low	None	High	Moderate	Low	None	High	Moderate	Low	None
Spruce/fir	acres	30,430	11,487	44	62	34,052	6,595	565	811	1,168	5,457	32	35,366	2,200	418		39,405
	hectares	12,315	4,649	18	25	13,780	2,669	229	328	473	2,208	13	14,312	890	169		15,947
	%	72	27	<1	<1	81	16	1	2	3	13	<1	84	5	1		94
Douglas-fir	acres	1,917	794			1,088	56	6	1,561	1,247	1,348	116		863	329		1,519
	hectares	776	321			440	23	2	632	505	545	47		349	133		615
	%	71	29			40	2	<1	58	46	50	4		32	12		56
Blue spruce	acres	28			147				175		10		165	28			147
	hectares	11			60				71		4		67	11			60
	%	16			84				100		6		94	16			84
Aspen	acres	44,231	12,193		100,872	36,717	11,435	690	108,454	3,523	7,237	265	146,270	19,839	14,716	249	122,491
	hectares	17,900	4,934		40,822	14,859	4,628	279	43,890	1,426	2,929	107	59,193	8,029	5,955	101	49,570
	%	28	8		64	23	7	<1	69	2	5	<1	93	13	9	<1	78
Lodgepole pine	acres	11	141		27		152		27				178				178
	hectares	4	57		11		61		11				72				72
	%	6	79		15		85		15				100				100
Cottonwood	acres		173		1,146	173			1,146				1,319		225		1,093
	hectares		70		464	70			464				534		91		442
	%		13		87	13			87				100		17		83
Ponderosa	acres	3,027	1,081		90,661	710	165	243	93,651	80	2,873	228	91,589	57,622	36,049	962	137
	hectares	1,225	438		36,689	287	67	99	37,899	32	1,163	92	37,065	23,319	14,588	389	55
	%	3	1		96	1	<1	<1	99	<1	3	<1	97	61	38	1	<1
Piñon/juniper	acres	180	110		89,643		37		89,896		57	142	89,734	1,661	3,849	22	84,401
	hectares	73	45		36,277		15		36,380		23	58	36,314	672	1,558	9	34,156
	%	<1	<1		100		<1		100		<1	<1	100	2	4	<1	94
Gambel oak	acres	1,838	2,173		143,286		405	412	146,480		2,795	223	144,279		20,756	12,856	113,685
	hectares	744	879		57,986		164	167	59,279		1,131	90	58,388		8,400	5,203	46,007
	%	1	1		97		<1	<1	99		2	<1	98		14	9	77

Table 6-21 continued.

Cover Type*	Units	Mountain pine beetles 2 (lodgepole pine) vulnerability				Lodgepole pine dwarf mistletoe vulnerability				Armillaria vulnerability			
		High	Moderate	Low	None	High	Moderate	Low	None	High	Moderate	Low	None
Spruce/fir	acres				42,023				42,023	13,613	26,596	1,784	30
	hectares				17,006				17,006	5,509	10,763	722	12
	%				100				100	32	63	4	0
Douglas-fir	acres				2,710				2,710	691	521	1,067	431
	hectares				1,097				1,097	280	211	432	174
	%				100				100	26	19	39	16
Blue spruce	acres				175				175	18		150	7
	hectares				71				71	7		61	3
	%				100				100	10		86	4
Aspen	acres	213	117	4	156,961	217	117		156,961			12,359	144,936
	hectares	86	47	2	63,520	88	47		63,520			5,002	58,654
	%	<1	<1	<1	100	<1	<1		100			8	92
Lodgepole pine	acres		37	141		16	162						178
	hectares		15	57		7	66						72
	%		21	79		9	91						100
Cottonwood	acres				1,319				1,319				1,319
	hectares				534				534				534
	%				100				100				100
Ponderosa	acres				94,770				94,770			622	94,147
	hectares				38,352				38,352			252	38,100
	%				100				100			1	99
Piñon/juniper	acres				89,933				89,933				89,933
	hectares				36,395				36,395				36,395
	%				100				100				100
Gambel oak	acres				147,297				147,297				147,297
	hectares				59,609				59,609				59,609
	%				100				100				100

* Cover types with host tree species present in mix

Literature Cited

Dahms, C.W., B.W. Geils. [tech. eds.]. 1997. An Assessment of Forest Ecosystem Health in the Southwest. Gen. Tech. Rep. RM-GTR-295. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. 97 p.

DeByle, N.V., R.P. Winokur [eds]. 1985. Aspen: Ecology and Management in the western United States. Gen. Tech. Rep. RM-119. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. 283 p.

Eager T. 2001. Draft Needs for GMUG Forest Plan Revision – Forest Health.
Eager, T. 2003. Personal communication.

Edmonds, R.L., J.K. Agee, and R.I. Gara. 2000. Forest health and protection. McGraw Hill, New York, NY.

Harris, J.L., M. Frank, and S. Johnson. 2001. Forest Insect and Disease Conditions in the Rocky Mountain Region 1997-1999. USDA Forest Service, Forest Health Management, Lakewood, CO.

Harris, J.L., R. Mask and J. Witcosky. 2002. Forest Insect and disease Conditions in the Rocky Mountain Region, 2000-2001. USDA Forest Service. Forest Health Management. Lakewood, CO.

Harris, J.L. 2003. Forest Insect and Disease Conditions in the Rocky Mountain Region, 2002. USDA Forest Service. Forest Health Management. Lakewood, CO.

Hessburg, P.F., B.G. Smith, C.A. Miller, S.D. Kreiter, and R.B. Salter. 1999. Modeling Change in Potential Landscape Vulnerability to Forest Insect and Pathogen Disturbances: Methods for Forested Subwatersheds Sampled in the Midscale Interior Columbia River Basin Assessment. USDA Forest Service. Pacific Northwest Research Station. General Technical Report PNW-GTR-454. 56 pp.

D.W. Johnson. 1997. Comandra Blister Rust. Forest Insect & Disease Leaflet 62, U.S. Department of Agriculture Forest Service.

Johnson, E. 2002. 2001 Aerial Detection survey of the Rocky Mountain Region. USDA Forest Service, Forest Health Management. Lakewood, CO.

Johnson, S.J. 1996. Forest insect and disease conditions in the Rocky Mountain Region 1995. USDA Forest Service. Forest Health Management. Lakewood, CO.

Johnson, S.J. 1997. Forest insect and disease conditions in the Rocky Mountain Region 1996. USDA Forest Service. Forest Health Management. Lakewood, CO.

Knight F.B. and H.J. Heikkenen. 1980. Principles of forest entomology, 5th Edition. McGraw-Hill Book Company, New York, NY, USA.

Landis, T.D. and L.B. Helburg. Black stain root disease of pinyon pine in Colorado. Plant Disease Reporter 60:713-717.

Mask,R. 2003. Personal communication.

McMillin, J.D.; and K.K. Allen. 2000. Impacts of Douglas-fir beetle on overstory and understory conditions of Douglas-fir stands, Shoshone National Forest, Wyoming. USDA For. Serv., Rocky Mountain Region, Renewable Resources. Tech. Rep. R2-64. 17 pp.

Rogers, P., D. Atkins, M. Frank, and D. Parker. 2001. Forest Health Monitoring in the Interior West: A baseline summary of forest issues, 1996-1999. USDA Forest Service RMRS-GTR-75. 40 p.

Romme, W.H., M.L. Floyd, D. Hanna. 2003. Draft Landscape Condition Analysis for the South Central Highlands Section, Southwestern Colorado and Northwestern New Mexico.

Schmid, J.M. and G.D. Amman. 1992. Dendroctonus beetles and old-growth forests in the Rockies. In: Kaufmann, M. R., W.H. Moir, and R.L. Bassett (tech. coord.). Old-growth Forests in the Southwest and Rocky Mountain Regions, Proceedings of a Workshop. USDA For. Serv., Rocky Mountain For. And Range Exp. Stn. GTR RM-213, Fort Collins, CO. 201p.

Shaw, C.G., III, and G.A. Kile. 1991. Armillaria root disease. USDA Forest Service Agricultural Handbook 691, Washington, D.C.

USDA Forest Service. 1992. Analysis of Bark Beetle Infestation in Ponderosa Pine and Management Practices to Treat Epidemic Populations and Vegetative Management to Reduce Risk. Delta, CO: USDA Forest Service, Grand Mesa, Uncompahgre and Gunnison National Forests.

USDA Forest Service. 2000d

USDA Forest Service 2000e

USDA Forest Service. 2001. Management Indicator Species Assessment, Grand Mesa, Uncompahgre and Gunnison National Forests, Version 1.0. Delta, CO: USDA Forest Service.

USDA Forest Service. 2003. Forest insect and disease conditions in the United States 2002. USDA Forest Service. Forest Health Protection. 124 pp.

Worrall, J.J. and K.F. Sullivan. 2002. Discoloration of ponderosa pine on the San Juan National Forest, 1999-2001. USDA Forest Service. Renewable Resources, Rocky Mountain Region. Biol. Eval. R2-02-06. 20 pp.

Fire and Fire Management

Summary from RMLANDS HRV

http://www.umass.edu/landeco/research/rmlands/applications/hrv_sjnf/documents/summary.htm

Objectives

Provide a spatial representation of expected historic fire regimes. Display the current condition as a measure of the departure from historic conditions. Map areas at high risk for fire. Identify and map ecological risks associated with the interaction of wildfire with old growth/older forests, insects and disease. To the extent possible given information limitations, discuss the ecological implications of the current condition.

Introduction

Disturbances have been identified as important drivers of heterogeneity in landscapes for several decades (Watt 1947), and have more recently been recognized as essential elements in the dominant, natural processes in ecological communities (Wu and Loucks 1995). Understanding ecological systems, particularly at broad scales, demands that we examine the patterns, consequences, and importance of disturbances (Baker 1989; Pickett and White 1985; Turner and Dale 1998). Although the concept of landscapes as mosaics of patches generated by disturbance (Pickett and White 1985) is relatively recent, ecologists have since noted the importance of natural disturbances in determining the spatial configuration for ecological processes (Paine and Levin 1981; Pickett and White 1985; Wiens 1995). In any case, understanding the nature of the disturbance mosaic and the factors controlling landscape patterns is crucial for predicting ecosystem dynamics and vegetation development in disturbance-prone landscapes (Turner et al. 2001).

Numerous types of natural disturbance occurred in the study area during the reference period: fire, snow avalanches, windthrow, and a variety of tree-killing insects, fungi and other pathogens (Veblen et al. 1989, Lertzman and Krebs 1991, Veblen et al. 1991a, b, Roovers and Rebertus 1993, Veblen 2000, Romme et al. 2003). However, the most important and coarsest in scale of these natural disturbances was fire. Based on a number of fire history studies and relatively extensive local empirical data, Romme et al. (2003) concluded that the median fire return interval varied dramatically across the landscape along an elevational gradient in relation to fuels and moisture conditions, ranging from 10-30 years in the lower elevation ponderosa pine type, 20-50 years in the dry mixed-conifer type, 60-120 years in the aspen type, and 200-350 years in the spruce-fir type. They also noted that many individual stands escaped fire for far longer than the median return interval and some burned at shorter intervals, creating a complex vegetation mosaic at the landscape scale. They further hypothesized that under these reference period conditions, stand replacement fires initiated stand development and maintained a coarse-grain mosaic of successional stages and cover types across the landscape, while non-replacement fires functioned to maintain communities in a particular condition (e.g., open canopy ponderosa pine forest) or accelerate the successional process of stand development.

The objective of this module is to show and describe the current landscape condition associated with natural disturbances on the San Juan National Forest. Fire and insects are the major disturbances affecting the San Juans and are the primary focus here. A discussion of historical fire regimes is followed by a comparison with the current condition of the landscape as a means to measure departure from historic conditions, probable hazard of severe fire, and wildfire occurrence probability. For insects, recent insect activity is summarized, as well as identification of areas of high risk of insect activity. Other natural disturbances, including wind throw and common forest pathogens, are discussed briefly.

Rocky Mountain Landscape Simulator (RMLANDS) **Applications**
This page contains links to all RMLANDS applications. Documents are in a mixture of formats, including html (viewable by any web browser), Microsoft Word, and pdf (which requires Adobe Acrobat Reader to view).
San Juan National Forest - Historic Range of Variability --This application involves using a suite of models (RMLANDS, FRAGSTATS, and HABIT@) to simulate and quantify the range of variation in landscape structure and habitat capability for selected indicator species on the San Juan National Forest under a disturbance regime characteristic of a historic reference period (the pre-1900 period of indigenous settlement).

Fire on the San Juan National Forest Landscape

Wildfire.--Numerous types of natural disturbance occurred in the study area during the reference period: fire, snow avalanches, windthrow, and a variety of tree-killing insects, fungi and other pathogens (Veblen et al. 1989, Lertzman and Krebs 1991, Veblen et al. 1991a, b, Roovers and Rebertus 1993, Veblen 2000, Romme et al. 2003). However, the most important and coarsest in scale of these natural disturbances was fire. Based on a number of fire history studies and relatively extensive local empirical data, Romme et al. (2003)

concluded that the median fire return interval varied dramatically across the landscape along an elevational gradient in relation to fuels and moisture conditions, ranging from 10-30 years in the lower elevation ponderosa pine type, 20-50 years in the dry mixed-conifer type, 60-120 years in the aspen type, and 200-350 years in the spruce-fir type. They also noted that many individual stands escaped fire for far longer than the median return interval and some burned at shorter intervals, creating a complex vegetation mosaic at the landscape scale. They further hypothesized that under these reference period conditions, stand replacement fires initiated stand development and maintained a coarse-grain mosaic of successional stages and cover types across the landscape, while non-replacement fires functioned to maintain communities in a particular condition (e.g., open canopy ponderosa pine forest) or accelerate the successional process of stand development.

RMLANDS simulations, McGargil 2005, largely confirmed these observations and provided a detailed quantitative summary of the wildfire disturbance regime. In particular, the simulations identified a similar elevational gradient in return intervals (or rotation periods), ranging from 40-46 years in the lower elevation ponderosa pine (without and with aspen) type, 53-63 years in the warm dry mixed-conifer (without and with aspen) type, 109 years in the pure aspen type, 138-146 years in the cool moist mixed-conifer (with and without aspen) type, and 218-267 years in the spruce-fir (with and without aspen) type (Table-rotation). The minor differences between our findings and Romme et al. (2003) largely reflect biases associated with the approaches used to estimate return intervals in each study. For example, RMLANDS estimates of a 40-year mean return interval in ponderosa pine forests was inclusive of high- and low-mortality fires and was averaged over every cell classified as ponderosa pine, whereas the estimate of Romme et al. (2003) was based on intervals between recorded fire scars (and therefore limited to low-mortality fires only) from a sample of trees in large homogeneous stands of ponderosa pine (which generally have the shortest return intervals). RMLANDS simulations computed the return interval between low-mortality fires using a comparable approach, in which each recorded interval between low-mortality fires in a cell was treated as an independent observation in order to approximate the method of Romme et al. (2003), the mean return interval was somewhat shorter (30 years) and within the range reported by Romme et al (2002). The difference in these estimates reflects the fact that we averaged over all ponderosa pine cells instead of restricting our estimate to only large homogeneous stands (with shorter return intervals). Similarly, the RMLANDS method of computing the mean return interval in the higher elevation spruce-fir forest included both low- and high-mortality fires, whereas the estimate of Romme et al. (2003) was based on stand-replacement (i.e., high mortality) fires only. When RMLANDS restricted computations to high-mortality fires, the mean return interval increased to 266-329 years and was consistent with the previous study.
Overall, simulation results were remarkably consistent with those determined by Romme et al. (2003) based on empirical field studies.

McGargil also noted the distinct variability in return intervals among locations within a single cover type. For example, even in ponderosa pine-oak forest - the cover type with the shortest mean return interval (~40 years), the mean return interval between fires (of any mortality level) varied widely from 19 years to > 800 years (Figure-return) and varied spatially across the forest (Figure-map). In general, return intervals increased with elevation, reflecting the moister, cooler conditions at higher elevations, and increased for stands embedded in a neighborhood containing cover types with longer return intervals (e.g., aspen, cool moist mixed-conifer forest). These patterns of variation were remarkably consistent among all cover types, highlighting the importance that landscape context has on fire regimes and demonstrating that no single statistic, such as mean fire interval (MFI), is adequate to characterize historical fire regimes, and that the widely used MFI actually may be quite misleading if taken literally - it may connote homogeneity and/or consistency when in fact spatial and temporal variability is the trademark of these regimes.

Perhaps the single greatest insight gained from our simulations with regards to wildfire stems from the shear magnitude of wildfire disturbance that is required to produce the widely accepted return intervals for the reference period. On average, once every two decades, > 10% of the area (~80,000 ha) was burned, and roughly once every 120 years, > 20% (~160,000 ha) was burned (Figure-recurrence), inclusive of both high- and low-mortality affected areas. This is a tremendous amount of burning and perhaps a magnitude of burning that is poorly appreciated by land managers and the general public based on public reactions to recent "large" fires in the west. Note the largest recorded fire in the project area was the Missionary Ridge fire of 2002, which burned a mere 20,000 ha, yet prompted significant reaction among managers and the public.

Fire Regime Condition Class

Fire Regimes

Fire regimes describe historical fire conditions that influenced how vegetation communities evolved and were maintained over time (Hardy and other 1998 as cited in Schmidt et al 2002). Fire regimes are generally characterized by fire frequency and severity. Fire frequency is the average number of years between fires, and fire severity is the effect fire has on the dominant overstory vegetation. (Schmidt et al 2002).). A low-severity fire, or surface fire, burns less than 25 percent of the overstory vegetation. A high-severity fire, or a stand replacement fire, burns more than 75 percent of the overstory vegetation. Mixed-severity fires have areas of both low and high-severity fires that result in a mosaic, or patchwork of burned and unburned conditions.

Five fire regimes have been defined by fire frequency and severity as part of national-level fire planning (Schmidt et al 2002, Interagency Fire Regime Condition Class Guidebook Version 1.1; updated VERSION 1.2 May 2005).
http://www.frcc.gov/docs/1.2.2.2/Complete_Guidebook_V1.2.pdf

Table 6-22 includes brief descriptions of each fire regime.

Table 6-22 also lists the potential natural vegetation (PNV) types that have a given fire regime. As described in the Vegetation section, a PNV type is defined as the area where a particular climax plant community could potentially develop based on soils, slope, aspect, climate and elevation. Each PNV type experienced a characteristic pattern of succession and natural disturbances (i.e. fires, insect outbreaks) that occurred at certain intervals with expected intensities. All five fire regimes occur on the San Juan National Forest, based on the PNV types that occur in this Geographic Area. Figure ? displays the distribution of fire regimes on the Forest.

Table 6-22 Historic Fire Regimes for the San Juan National Forest

Table 6-22 Historic Fire Regimes for the San Juan Public Lands Geographic Area

Fire Regime Class	Frequency (Fire Return Interval)	Severity	Existing Vegetation Types	Acres of SJPL	% of SJPL Land
I	0 - 35 + years, frequent	Predominantly Low	Ponderosa Pine	411,790	11
			Warm-Dry Mixed Conifer	95,392	3
			Mtn Grasslands	304,314	8
II	0-35 + years, frequent	Replacement	Semi-Desert Shrubland	95,380	3
			Sagebrush Shrublands	210,030	6
III	35 – 100 + years, less infrequent	Mixed and Low	Cool-Moist Mixed Conifer	199,412	6
			Aspen	346,384	10
			Mtn Shrubland	450,190	12
			Sagebrush	**	
IV	35 – 100+ years, less infrequent	Replacement	Piñon-Juniper Shrubland	**	
			Semi-Desert Grassland	301,538	8
			Semi-Desert Shrubland	95,380	
			Riparian & Wetland	77,964	
			Spruce-Fir	510,220	14
V	200+ years	Replacement and Other Fires Occurring Within this Frequency Range	Piñon-Juniper Woodland	444,147	12
			Alpine	186,494	5

Departures from Historical Fire Regimes

HRV Departure

One of the principal purposes of gaining a better quantitative understanding of the historic reference period is to know whether recent human activities have caused landscapes to move outside their historic range of variability. To this end, we modified the approach for FRCC determination (see methods above) and made the following key findings.

- The current landscape structure appears to deviate substantially from the simulated HRV (Table-hrv-summary-combined), although the level of "departure" varies spatially across the forest in relation to differences among cover types (Figure-hrv-departure-map). Many characteristics appear far outside that range of variability. Indeed, one-third of the landscape composition metrics (16/48) and most of the landscape configuration metrics (15/19) are completely outside their HRV's (i.e., 100% departure index)(Table-hrv-combined).

- In general, the current landscape has fewer, larger, more extensive and less isolated patches with less edge habitat than existed under the simulated HRV. The larger patches tend to be geometrically less complex and contain proportionately more core area than existed under the simulated HRV. Overall, the current landscape is more contagious and less structurally diverse than ever existed under the simulated HRV. This can be interpreted as a more homogenous landscape, where the lack of any extensive disturbance during the past 100 years has led to large, mostly late-seral patches, with low contrast due to the paucity of younger seral stages. This landscape condition appears to be largely a legacy of the last century of land management practices, in particular fire exclusion. However, the generally benign climate of the twentieth century also was a significant reason for the lack of large, stand-replacing disturbances, either by fire or spruce beetle.

- The patterns of departure were generally similar at the class level for each of the cover types with reliable data on current conditions (i.e., forest types). In particular, the current high-elevation landscape is dominated by large patches of late-seral conifer forest and an almost total absence of early-seral forest due to the lack of extensive disturbance. The story is similar for pure aspen forests. Aspen-dominated forest, which includes the early- and mid-seral stages of the mixed conifer-aspen forest types, exhibits a notable deviation from this general pattern. While the current landscape has less aspen in the early- and mid-seral stages, and more in the late-seral stage - similar to the other high-elevation forest types - the coarse spatial configuration of aspen-dominated forest appears to be generally within the simulated range of variation. The low-elevation forests, especially ponderosa pine, also contain an overabundance of stands in the late-seral stages, but the most notable departure is the complete absence of stands in the fire-maintained open canopy condition.

- Due to the above vegetation conditions and patterns, the current landscape appears to deviate substantially from the HRV in susceptibility to at least four of the simulated insects/pathogens disturbances (Table-hrv-susceptibility). The current landscape appears especially vulnerable to pine beetle, spruce budworm and spruce beetle outbreaks.

- Given the direct link between vegetation patterns and wildlife habitat, it is not surprising that the wildlife indicator species we analyzed also exhibit substantial departure from their simulated HRVs (Table-LC index). Of the species considered, the pine marten is the principal beneficiary of the current landscape departure. Extensive late-seral conifer forest in the higher elevations is likely providing ideal habitat conditions for this species. Three-toed woodpeckers also benefit from these conditions, even though this species is better adapted to exploit post-disturbance environments. The three-toed woodpecker is the only species not exhibiting significant departure. Elk and olive-sided flycatchers are both disadvantaged in the current landscape. Both species benefit from edges between early- and late-seral vegetation patches. The paucity of disturbances over the past century has left the current landscape rather deprived of edge habitat and has reduced the overall interspersion and juxtaposition of the vegetation mosaic, with negative consequences on habitat capability for these two indicator species.

- Managers need to be cognizant of two important considerations when interpreting these HRV departure results. First, although it is clear that the current landscape structure is not within the modeled range of variability, the magnitude of departure is less clear due to inconsistencies in the spatial resolution of the initial cover type map. Specifically, the fine-grained heterogeneity in vegetation created by the disturbance processes in RMLANDS was probably not comparably represented in the Forest Service map of current vegetation. We took precautions to safeguard against reaching spurious conclusions in this regard. First, we eliminated the finest heterogeneity by rescaling the vegetation maps to a 0.5-ha minimum mapping unit and then evaluated HRV departure on these rescaled vegetation maps in addition to the original high-resolution maps. Second, in our interpretation of HRV departure, we emphasized several area-weighted landscape metrics that are insensitive to variations affecting very small patches. Nevertheless, several landscape configuration metrics sensitive to fine-grained heterogeneity were incorporated into the overall configuration departure index. Consequently, while we feel confident in concluding that the current landscape structure is well outside the modeled range of variability, it is important to be aware that our reported HRV departure indices, except for the seral-stage departure index and landscape composition departure index, are probably biased high (i.e., inflated).

- Second, any conclusions regarding HRV departure depend on an accurate mapping of stand conditions in the current landscape. In particular, we lack reliable age and stand condition data for most non-forested types (e.g., mountain shrublands, mesic sagebrush, pinyon-juniper woodlands). Consequently, our initial assignment of stands to condition classes (seral stages) was based on interpolation from sparse data or on a random assignment based on seral-stage distributions estimated by local experts. In either case, we are not confident that our current condition estimates are accurate. Hence, until more complete data on current stand age and condition are available, the HRV departure results for these cover types must be viewed with extreme caution.

- Our simulations indicate that returning the landscape structure to a condition that falls within the simulated HRV would likely be a difficult and long-term undertaking if it were deemed desirable. We deduced this from the time it took the current landscape to equilibrate to the reference-period disturbance regime. We can infer that if management activities were designed to emulate natural disturbance processes, then it would take a length of time equal to the equilibration period to return the landscape to its HRV. In our simulations, most landscape structure metrics equilibrated within 100-300 years. It must be emphasized, however, that this does not imply that it should be our goal in management to recreate all of the ecological conditions and dynamics of the reference period. Complete achievement of such a goal would be impossible, given the climatic, cultural, and ecological changes that have occurred in the last century. Moreover, the extent and intensity of disturbance required to emulate the natural disturbance regime would be unacceptable socially, economically, and politically.

20th Century Fire Activity on the San Juan National Forest

FIRE AND FUELS MANAGEMENT

National emphasis on fire and fuels management has increased over the past five years as a result of large fires, droughts, increasing forest health concerns and effects to communities. New policies and laws provide direction to manage wildfires more effectively, reduce hazardous fuels especially in wildland urban interface areas, restore and maintain fire-dependant ecosystems, and promote collaboration with local communities to address wildfire related issues.

The planning process for the San Juan Public Lands (SJPL) will consider the historic role of fire as an ecological disturbance agent. This knowledge will be used to determine how fire can be used to achieve desired conditions on the landscape.
Environmental, social and economic concerns will be used in developing future fire and fuels management strategies for the SJPL.

Fire Regimes

Fire regimes describe the historical ecological role fire played in creating and maintaining vegetation communities before European American settlement activities and active fire suppression began. Fire regimes, or more generally, disturbance regimes are a key component of the historical range of variability (HRV) characterization for vegetation types.

HRV characterizations are important for providing ecological context for management decisions and guidance for developing desired future conditions (DFC).

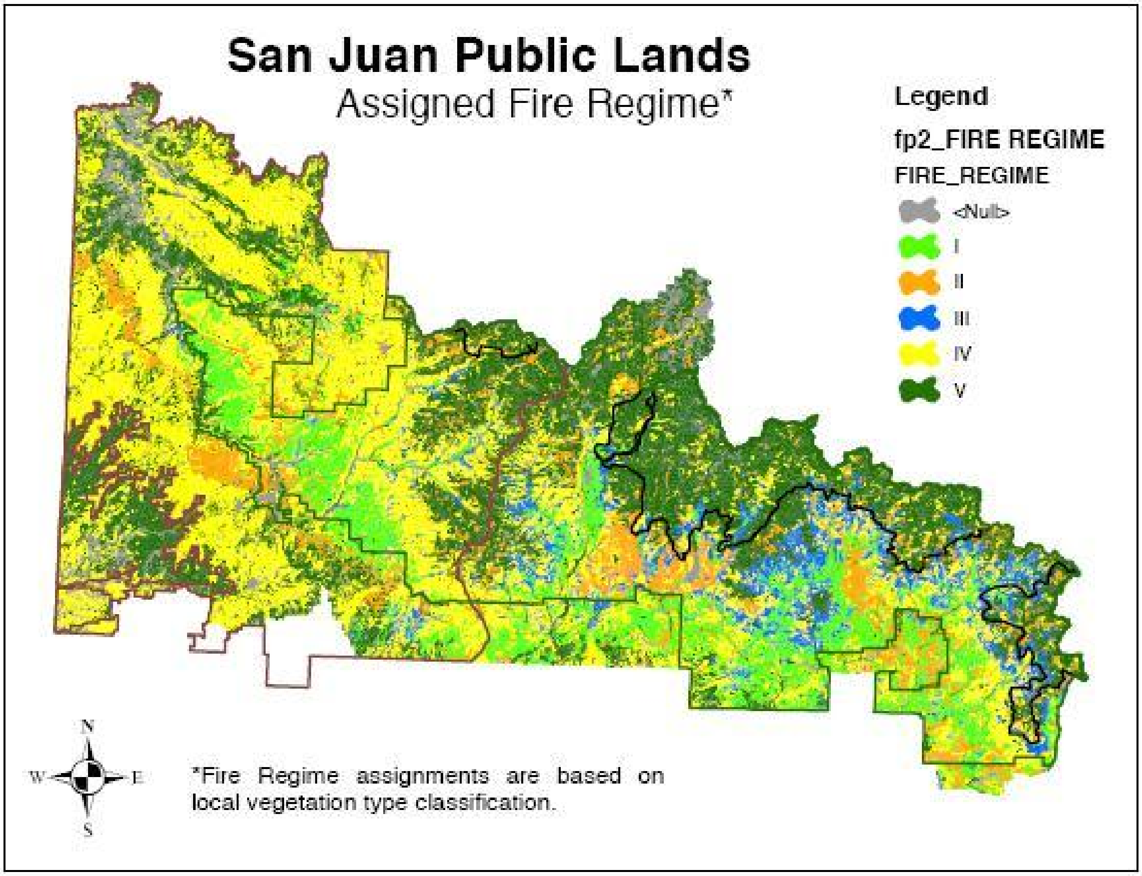
Fire regimes are typically described by fire frequency, intensity, size, and vegetation type (Heinselman, 1981; Kilgore, 1981; Sando, 1978):

- Fire frequency is measured by average number of years between fires, often referred to as the mean fire interval (MFI).
- Fire intensity is a general description of surface or crown fire. Severity describes fire effects on overstory vegetation (Hann et al 2003). A low-severity fire burns less than 25 percent of the overstory vegetation. A high-severity fire burns more than 75 percent of the overstory vegetation. Mixed-severity fires exhibit surface and crown fire behavior to create a mosaic, or patchwork of low, high, and unburned severity. Mixed severity fires should be interpreted cautiously because no standard exists for surface to crown fire ratios and interpretations are scale dependent.
- Historic fire size is a difficult parameter of the fire regime to reconstruct. Size estimates of surface fire are especially ambiguous because they do not leave distinct boundaries on the landscape. However mapping dated fire scarred samples can be used to gauge the relative extents of past surface fires. Crown fire systems create a more readable footprint across the landscape. Distinct stand structure and age structure can be used to estimate past fire sizes and dates. However this method is limited to the most recent burns because more recent burn will obscure older footprints, thereby limiting long temporal assessments.
- Vegetation types by fire regime are listed in Table 6-23 based on Fire Regime Condition Class definitions (Hann et al 2003). Existing vegetation types are found in the in the Vegetation section [ref] which describes current vegetation and the potential natural vegetation community that would develop in the absence of disturbance. However, each existing vegetation type has experienced a characteristic pattern of succession and natural disturbances (i.e. fires, insect outbreaks) that occurred at varying intervals and characteristic intensities. These natural disturbances were key to maintaining a diversity of seral communities, and therefore a variety of plant and wildlife habitat across the landscape.

HRV characterizations are needed to determine fire regimes for different vegetation types. SJPL is developing a local fire history database for its major fire dependent vegetation types. The most extensive data is for the ponderosa pine and warm-dry mixed conifer types. Local data for aspen, cool-moist mixed conifer, and piñon-juniper is also available(Brown and Wu 2005, Floyd 2004, Grissino-Mayer et al 2004, Romme et al 2003, Floyd et al 2000). Where local data was sparse, relevant literature and expert input were used create the following map and fire regime descriptions.

Figure 6-21 Fire Regimes, San Juan Public Lands Geographic Area

Figure 6-21 Fire Regimes, San Juan Public Lands Geographic Area



MAP NOTE: Official Fire Regime and Condition Class (FRCC) maps are currently in draft form. FRCC maps will be available after regional biophysical site descriptions are finalized. Therefore, the FRCC maps shown in this document were not developed with the nationally standardized FRCC mapping protocols (Hann et al 2003).

Fire Regime I (Low Severity, 0–35+ Years)

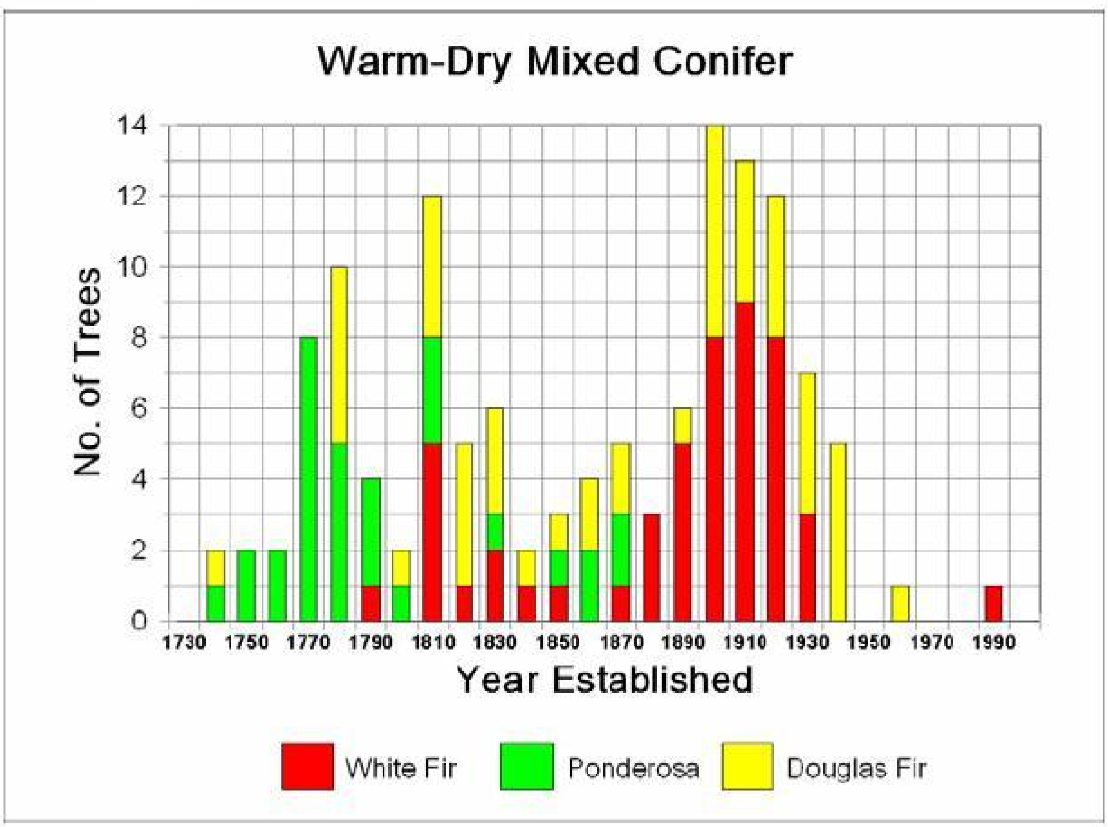
This regime applies to ponderosa pine and warm/dry mixed-conifer forests. Natural History Studies have shown these forests were dominated by frequent surface fires (Brown and Wu 2005, Grissino-Mayer et al. 2004, Wu 1999 and unpublished data). Smaller, spatially localized fires occurred on the order of one or more times a decade (6–10 yrs.). Landscape-scale fires that burn larger occur less frequently, but still on a ‘short’ decadal time scale (13–30 yrs). Climatically forced, long fire-free intervals that were spatially extensive of up to 40 years have also been documented.

Fire exclusion across the lower forest elevations started in the late 1800s to early 1900 (typically 1880), first due to heavy livestock grazing and later to fire suppression (Covington and Moore 1994, Swetnam and Betancourt 1998, Grissino-Mayer et al. 2004).

The general pattern observed in stand structure is an overall densification of the forests, as more regeneration survives without frequent fires to thin them. Distinctive patches or groups of large ponderosa pine trees exist, but second growth has in-filled them or they have been harvested by past logging. The grass cover associated with open stands has greatly declined or is lost entirely under dense tree canopies. The Gambel oak (Quercus gambelii) shrub component has also been increasing in cover and height in the understory. Gambel oak and small trees growing underneath larger ones, and the dense canopy cover found across the great majority of ponderosa pine are the key conditions making way for the uncharacteristic crown fire threatens ponderosa pine forests. This same pattern in the warm-dry mixed conifer has added complexity, due to the presence of competing tree species. Fire exclusion is causing a shift in tree species dominance from ponderosa pine toward white fir and Douglasfir. Historically, fire gave ponderosa pine a competitive edge among the other species found in the warm-dry mixed conifer. Stand level age structure and fire chronology illustrates this point .
(Figure 6-22 , Wu 1999).

Figure 6-22 Warm Dry Mixed Conifer Age and Fire History

Figure 6-22 Warm Dry Mixed Conifer Age and Fire History



Frequent fires thinned excessive tree regeneration of all species, but more ponderosa pine survived than white and Douglas-fir. Current Conditions after over a century of fire exclusion, it is still a component of the dominant overstory species, but more white fir has joined the overstory, filled in the subcanopy, and now dominates regeneration. Little to no ponderosa pine regeneration is found in most warm-dry mixed conifer, because of the lack of fire and closed-forest conditions.

Ponderosa pines were selectively logged from some warm-dry mixed conifer sites converting them to cool-moist mixed conifer stands. Over time, white fir could replace the ponderosa pine. Douglas-fir, not as prolific as white fir, has integrated with the white fir. The exclusion of fire from 20th century management practices has shifted ponderosa pine-Gambel oak and warm-dry mixed conifer from surface fire regimes to crown fire regimes. Although some crown fires probably occurred in these forests in the past, they were very limited in scale to small groups or patches of trees. The landscape-scale crown fires that occur today on the order of hundreds and thousands of acres and the lack of understory grasses put these two forest types outside the range of historic (natural) variability in terms of stand structures, understory species health, and fire behavior that occurred in pre-1900 times.

Fire Regime II (Replacement, 0–35 Years)

Mountain grasslands are grouped under Fire Regime II.

Natural History

Reconstructing historic fire intervals for grasslands is problematic because no evidence for empirical research has lasted, such as basal fire scars. However, it is reasonable to assume that the fire frequency in grassy meadows is largely linked to the forests surrounding it. If the forest type burned often, as ponderosa pine forests did, then the meadows probably burned relatively often as well. Meadows at higher elevations in forest types that have longer fire intervals would burn less often. In addition, Native Americans may have burned meadow areas regularly to maintain pasture for their animals or game, or to promote specific plants, creating areas of locally increased frequency.

Mountain grasslands were probably always associated with forests even during the reference period and fire appears to have played a maintenance role. Periodic fires would have prevented encroachment from woody species and renew bunch grasses by removing the buildup of dead plant material.

Current Conditions

Today, mountain grasslands occur as openings in forested dominated landscapes (see Existing Vegetation) [ref]. During the historic reference period (pre-1900), ponderosa pine stands were more open and park-like having an abundant herbaceous understory.

Forest densification in ponderosa pine has decreased grass cover to the point of exclusion in the most dense stands and the pattern may be similar in warm-dry mixed conifer. No formal assessment of meadow encroachment has been conducted in the planning area, but the pattern of forest densification in ponderosa pine strongly suggests that grassy understories have greatly diminished compared to the reference period.

Fire Regime III (Mixed Severity, 35–100+ Years)

This regime applies to cool-moist mixed conifer. Cool-moist mixed conifer forests are generally found between the warm-dry mixed conifer and spruce-fir forests with aspen intermixed throughout. It is a complex, heterogeneous band of forest where trees and understory of the wet and dry extremes mix, depending on specific site conditions. Drier sites can have small amounts of ponderosa pine, while white fir and Douglas-fir dominate wetter sites.

Natural History

Forest studies have shown that fire intervals range in the multi-decadal to century time scale in the cool-moist mixed conifer, with evidence of both surface and crown fire (Grissino-Mayer et al. 2004, Wu 1999). Fires in cool-moist mixed conifer forests were not fuel limited systems because these productive mesic sites always have enough live and dead fuels to carry fire. Fuel moisture, linked to climate, was the most limiting factor. Intra annual drying (i.e. seasons) creates a potential for fire in most years.

During typical seasonal climate and weather patterns, fires were probably predominantly surface fires with limited high intensity patches. The sizes of those fires ranged from small to large. Larger fires with a greater ratio of crown fire to surface fire area would burn in significantly dry years. Overall size, intensity, severity and of any fire would depend on fuel moisture, weather conditions, and climate context.

This fire regime promotes a complex and heterogeneous forest.

Current Conditions

Forest fuel loading in terms of woody debris and live biomass have increased over the period of fire exclusion. The increases however are not necessarily out of HRV on stand level. HRV concerns are for diversity in tree species and seral stages across the coolmoist mixed conifer on a landscape level. The landscape structure was probably more diverse during the pre-1900 reference period with early to late seral stands in various proportions, and more variety in stand densities than what is observed on landscape currently. Periodic surface and crown fires are particularly important to the persistence of ponderosa pine and aspen stands in this landscape. Ponderosa pine is a small yet important component of the cool-moist mixed conifer, occurring where fires burned more frequently due to topography, aspect, or warm climatic periods that supported more fire. The individuals or small groups likely are likely to disappear over time from old age, insects and disease, or crown fire under current conditions.

Aspen stands are aging uniformly within the cool-moist mixed conifer landscape. In the past, periodic fires would have renewed these stands. If the aspen is to persist over the long term, they will need to burn again. Current aspen distribution is probably not outside HRV conditions (Romme et al. 2003), and mixed-severity fires under normal fire weather conditions could adequately rejuvenate the patch mosaic over time.

Fire Regime IV (Stand Replacement, 35–100+ Years)

Mountain and Sage Shrublands, Semi-Desert Shrub and Grasslands

Mountain shrublands, semi-desert shrub and grasslands, and sage shrublands are classified under Fire Regime IV, however shorter intervals than 35yrs can probably occur. Climate conditions and the time needed for an adequate fuel complex to develop are likely factors that control fire frequency in these ecosystems. Therefore, in the driest and least productive systems, such as the semi-desert shrub and grasslands, fuel load is the more limiting factor. In these systems, vegetation develops very slowly under ons of scant rainfall and poor soils. Bare ground is prevalent even in the more productive of these sites. There is a lack of information about fire regimes for semidesert shrub and grasslands. Fire may not be a primary disturbance in these ecosystems.

Mountain shrubland ecosystems occur at higher elevations and moister climates making them more productive and giving them a greater potential to burn more often than semi-desert systems.

Natural History

The only pre-1900 fire history data available for any mountain shrublands in the study area comes from a study conducted in Mesa Verde National Park (Floyd et al. 2000). Floyd calculated a 100-year fire rotation period for Gambel oak (the time to burn an area equal to the total extent of shrubland in the study area). One hundred years may pass and all of the Gambel oak in the Mesa Verde study area could burn at once, or patches of various sizes could burn in the study area until an area equal to the study area has burned. In this latter scenario, some stands may burn more than once, while others may completely escape burning during that 100-year period. This scenario is reasonable, given the frequency that favorable fuel and weather conditions for fire occur.

Current Conditions

Further study on Gambel oak fire frequency and landscape structure is warranted because it occurs across a large elevational gradient across the SJPL. Furthermore, the fire rotation found at Mesa Verde National Park may not hold true across the SJPL. The Gambel oak in Mesa Verde National Park is associated with piñon-juniper woodlands, which burn less often than Gambel oak. Gambel oak is also associated with ponderosa pine, warm-dry and cool-moist mixed conifer, and also occurs in extensive stands of Gambel oak alone.

Sage shrublands are assumed to function similarly to Gambel oak shrublands. Regional studies from the Great Basin, Montana, Oregon have reported a range of fire intervals, mostly in the multi decadal time scale. There is limited localized information for the planning area, however there is an adequate basis for assuming that fire played a large role in structuring sagebrush ecosystems much as fire does in other major vegetation types.

In summary, more productive shrublands such as Gambel oak are not seriously outside their HRV in terms of fire intervals, however landscape diversity of patch sizes and seral stages should be concern. Sage shrublands and the semi-desert grass and shrublands are also not outside HRV. However, these drier ecosystems are currently experiencing an invasion of cheatgrass (Bromus tectorum). Cheatgrass is not only crowding out native vegetation, but will also alter this low frequency fire regime to a high frequency one by creating a fine fuel bed able to carry fire more frequently and extensively than during the reference period.

Stable and Seral Aspen

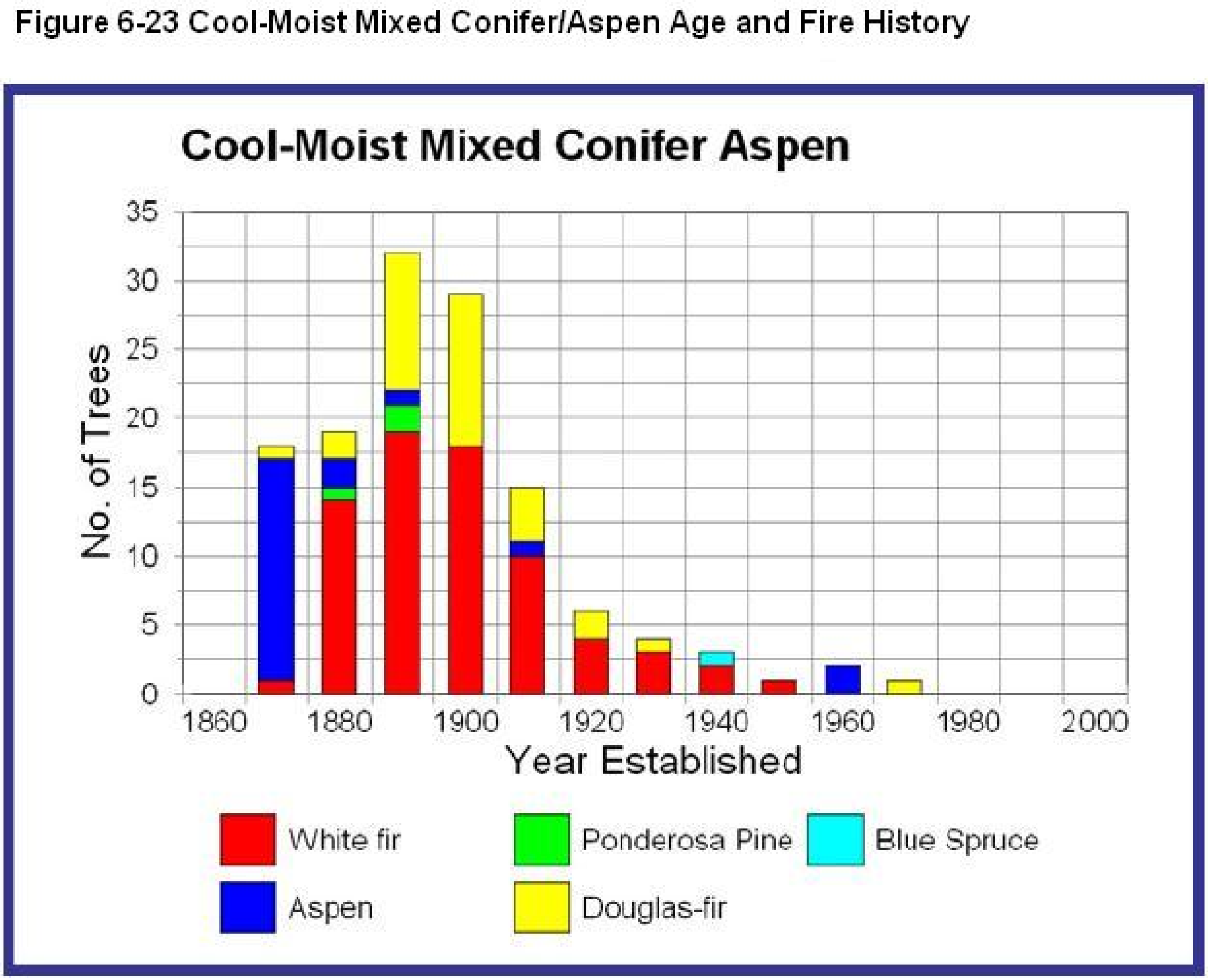
Fire Regime IV also includes stable and seral aspen.

Natural History

The importance of fire to aspen regeneration is well documented (Figure 6-23). Romme et al. (2003) conducted a landscape assessment of aspen on the west side of the SJNF. They found a fire occurred in every decade from 1740 through 1910 within their 77 km2 study area, with stands establishing in every decade from the 1760s through the 1870s. They calculated the median age of aspen stands to be 70 years in 1880 and interpreted the fire rotation to be 140 years for an area equal to the study area to burn. An individual stand may go more than a century before burning, whereas another could reburn at a much shorter interval.

These data show that fires occurred frequently across an aspen landscape. Significant climatic events (drought) could synchronize large areas of aspen, but stands would diversify again over time, with subsequent random fire occurrences and burn patterns.

Figure 6-23 Cool-Moist with Aspen Age and Fire History



Existing Conditions

Current aspen distribution does not differ greatly from the 1880 landscape (Romme et al. 2003). Continued fire exclusion could cause a decrease in aspen on the landscape if clones become so decadent over time that their underground root systems die with no overstory stems to maintain them. However, no conclusive information is available about the duration for which aspen roots remain viable. Small stems that go undetected may be able to sustain root systems for, many years.

Fire Regime V (Stand Replacement, 200+ Years)

This regime applies to forest or vegetation types that rarely burn, due to fuel or moisture limitations. On the SJPL, spruce-fir and piñon-juniper forests seldom burn.

Spruce-Fir

Cold/wet mixed conifer and spruce-fir are assumed to have the same fire regimes. Romme et al. (2003) estimate centuries passed between fires in a spruce-fir stand on the landscape. They could not determine exact fire intervals, because too many of their sampled stands were too old to estimate the time since the last fire. A summary of their age structure data does show that mature (>150 yrs.) spruce-fir stands were common on the landscape during the reference period (pre-1900). Young stands (<100 yrs) made up just less than 40% of the landscape and a small amount of the landscape was of intermediate ages (100–150 yrs). No significant fires have burned in their study area since their reference period, therefore their entire study landscape has aged over 100 years.

Spruce-fir forests with such long fire intervals do not have quantifiable landscape equilibrium in stand structure or age over practical management time scales.

Spruce-fir forests with such long fire intervals do not have quantifiable landscape equilibrium in stand structure or age over practical management time scales. Old or young stands will dominate a landscape depending on when the last major fires burned. Aspen stands within the spruce-fir are a concern: they must burn or be succeeded by conifer. Questions remain about how long an aspen clone can remain viable without fire. Overall, the spruce-fir landscape is likely within HRV limits in its current condition.

Piñon-Juniper

Piñon-juniper woodlands also have very long fire intervals. The fire rotation period for stands in Mesa Verde National Park is about 400 years while some have not burned for 700 years (Floyd et al. 2000). Another study from the Uncompahgre Plateau estimates piñon-juniper fire intervals to be 200–1000 years (Eisenhardt 2004).

Long fire intervals seem unlikely for forests that experience hot, dry summers with ample lightning strikes. Unlike the spruce-fir that is typically too wet to burn, piñon juniper forests often do not have the fuel continuity for fires to spread easily. These stands have scant herbaceous understory to lay the fine-fuel bed fires would need to spread. Only wind-driven fires burn significant numbers of acres.

These piñon-juniper forests have not been affected by fire suppression, and they appear to be within their HRV. However, noxious weeds pose a threat to their ecological integrity. Cheatgrass has proliferated throughout the planning area piñon-juniper, especially in beetle-killed stands, introducing surface fires to this forest type. The long term ecological consequences are serious, because cheatgrass is such an aggressive colonizer, and fire appears to promote it. Its presence will increase fire frequency, and its persistence may delay or completely alter typical post-fire successional pathways in piñon-juniper woodlands.

Piñon-juniper shrublands are common throughout the SJPL. Sagebrush (Artemisia tridentate and other spp.) is found on dry, low-elevation sites and Gambel oak on moister, higher-elevation ones. Fires may burn more often in the shrubland type than the forest type, because the shrubs and trees would develop a continuous fuel complex more quickly. The shrubland type may have been moderately affected by fire suppression, decreasing the diversity the over- and midstory structure across the landscape. Like the piñon-juniper woodland, noxious weeds are a serious problem. Cheatgrass, especially, will promote more frequent fire. If fires do burn, the post-fire successional trajectory will be altered, with cheatgrass or thistle dominating the stand.

Other Fire Regimes (Special Vegetation Types)

Important vegetation types that do not appear in Table 6-22 are riparian areas and wetlands. Information about disturbance is scarce for these vegetation types. Fire frequency in these vegetation types is influenced by the vegetation type surrounding it. Thus riparian and wetland areas in the subalpine will rarely burn compared to those found in ponderosa pine forests. Riparian and wetlands in lower elevations probably did not burn as often as its surrounding vegetation either because of persistent mesic conditions. Therefore the occurrence of fires is most likely controlled by drought.

Native burning may have played a role in these areas, but that topic must be researched.

Condition Class

Fire Regime Condition Class (FRCC) is a standardized interagency tool for assessing a current landscape's departure from historical (natural) conditions (Hann et al 2003). Historical or reference period is defined as the time period when ecosystems along with their natural disturbance regimes were still intact and functioning in sustainable landscapes before Euro-American settlement activities. Current condition departure assessments are based on fire frequency and intensity, current species composition, structural stage, age and canopy closure, and fuel accumulations compared to conditions under the historic disturbance regimes.

Departures between current and historic fire regimes may be due to past management activities like fire suppression, timber harvest, grazing, and presence of exotic or invasive species. The SJPL historical reference period spans the late 17th century to late 19th century (1600s – 1800s). Tree-ring fire chronologies identify 1880 as the last widespread fire year and that date represents the beginning of fire exclusion (Wu 1999).

Table 6-23 defines the three FRCC conditions classes. Low departure (CC1) is considered to be within HRV. Moderate departure (CC2) indicates that components of the fire regime, such as fire frequency, have been altered resulting in changes in vegetation and landscape patterns. These areas may require varying levels of management actions before fire can be restored and allowed to plays its historical natural role.

Table 6-23 Fire Regime Condition Class Descriptions

Table 6-23 Fire Regime Condition Class Descritons

Condition Class	Descriptions
CC1	Fire regimes are within the historical range and the risk of losing key ecosystem components is low. Vegetation attributes (species composition and structure) are intact and functioning within their historical range.
CC2	Fire regimes have been moderately altered from their historical range. The risk of losing key ecosystem components is moderate. Fire frequencies have departed from historical frequencies by one or more return intervals (either increased or decreased). This may result in moderate changes to one or more of the following: fire size, intensity and severity and landscape patterns. Vegetation attributes have been moderately altered from their historical range.
CC3	Fire regimes have been significantly altered from their historical range. The risk of losing key ecosystem components is high. Fire frequencies have departed from historical frequencies by multiple return intervals. This may result in dramatic changes to one or more of the following: fire size, intensity, and severity and landscape patterns. Vegetation attributes have been significantly altered from their historical range.

High departure (CC3) means that fire regimes and vegetation are significantly altered from historical conditions. Uncharacteristic fire behavior and fire effects will occur under certain conditions resulting in vegetation composition and assemblages not known to exist during reference conditions. Condition class is a calculated number and the protocols are outlined in the FRCC Guidebook v1.2 (Hann et al 2003). However, the SJPL's condition class map (Figure 6-21) presents assigned CC values based on a vegetation polygon's type and fire regime and will be updated with the new FRCC map when available.

The original FRCC assessments and map outputs did not accurately reflect true public land vegetation conditions across many National Forests and BLM lands. These results prompted an effort on the national and regional level to refine and adjust the biophysical site (BpS) descriptions[3]. Once BpS's have been finalized, a new FRCC assessment and mapping effort will be conducted following the nationally standardized protocols.

Table 6-24 shows each major vegetation type by its assigned CC. More detailed HRV discussion is found in the Fire Regime section above. In general, ecosystems with longest return fire intervals, such as spruce-fir and piñon-juniper, have not missed fire intervals and therefore on a stand level their structure and species composition is well within estimated HRV conditions. From an ecological perspective, fires can be allowed to

[3] BpS's are descriptions of physical site and potential natural vegetation incorporating the effects of natural disturbance processes that serves as reference conditions for a vegetation type.

burn in these forests under any conditions and it will burn with characteristic intensity with characteristic effects. Some concerns about landscape structure and mosaic exist and need to be assessed, but because of the very long fire intervals the landscape is probably still within HRV. Thus spruce-fir is in CC1. Piñon-juniper is in CC2. Even though its fire regime and macro woody structure is intact, piñon-juniper is only considered CC2 because grazing, chainings, and degraded herbaceous composition.

The current cheatgrass invasion may push piñon-juniper further out to CC3 over time. Cheatgrass cover will introduce frequent surface fire to this low frequency-high intensity fire regime and alter post fire successional pathways.

Table 6-24 Fire Regime Condition Class by Existing Vegetation Type

Table 6- 24 Fire Regime Condition Class by Existing Vegetation Type

Condition Class (assigned)	Existing Vegetation Type	Acres of Public Lands	Percent of Public Lands
1	Spruce-fir	510,220	14
	Alpine	186,494	5
	Aspen	346,384	10
2	Cool-Moist Mixed Conifer	199,412	6
	Mtn. Grasslands	304,314	8
	Mtn. Shrubland	450,190	12
	Piñon-Juniper Woodland	444,147	12
	Piñon-Juniper Shrubland	**	
	Semi-Desert Grassland	301,538	8
	Semi-Desert Shrubland	95,380	3
	Sage Shrublands	210,030	6
3	Ponderosa Pine	411,790	11
	Warm-Dry Mixed Conifer	95,392	3
0	Riparian & Wetland	77,964	2

Aspen is currently assigned to CC1, but is trending towards CC2. The current distribution and age structure across the landscape is an assemblage within HRV, however it appears to be on the longer extreme of its HRV. Many stands succeeding to conifer would benefit from burning to regenerate the stands. An important question to aspen persistence on the landscape is how long a clone can remain viable after its last stand replacement fire.

Cool-moist mixed conifer is assigned to CC2 because, although it has missed some fire intervals, its vegetation composition and landscape mosaics are still within HRV with fires still behaving characteristically and producing characteristic effects. This is true for the other vegetation types listed in CC2, however the semi-desert vegetation types and sagebrush type are threatened by cheatgrass and other noxious weeds and have the same situation described for piñon-juniper.

Ponderosa pine and warm-dry mixed conifer are both frequent surface fire regimes and have been the most affected by fire suppression, logging, and grazing since Euro- American settlement; therefore they are assigned to CC3. They have missed numerous fires relative to historic patterns, their stand structures are overly dense, understory herbaceous life is degraded, and white fir is overtaking ponderosa pine in the warm-dry mixed conifer. Forest fire regimes have shifted from high frequency-low intensity surface fire to low frequency-high intensity stand replacement fire.

Map 10. Condition Class, San Juan Public Lands Geographic Area
Dropped from fire regime section:
Aerial coverage of vegetation types indicates a fluctuation over time across the landscape. However, while structure and composition within vegetation types has been significantly altered in some types, it is assumed that the time period of European American settlement has not obscured major vegetation type boundaries significantly. The existing vegetation footprint will serve as an adequate pilot for mapping efforts and management purposes. All five fire regimes occur on the SJPL Geographic Area. Map 8 displays the distribution of fire regimes on the Geographic Area.
For the SJPL geographic area, the reference period for fire regimes spans the late 15th century to late 19th century (1600s – 1900s).

Current Wildfire Hazard on the San Juan National Forest

Fire hazard is directly related to vegetation or fuel conditions (type of vegetation, age, structure, density, amount of live and dead material), topography (slope, aspect, elevation), and weather conditions (wind speed and direction, fuel moisture). These elements all affect fire behavior, the intensity and rate of spread of fires. Fire hazard changes with changing conditions.

Fire hazard for the San Juan Public Lands was modeled for current vegetation conditions (topography is considered to be constant) under 97th percentile weather conditions, based on weather data taken from the Remote Automated Weather Station (RAWS) located throughout the unit. Data was summarized from 1990 to 2004. Modeled weather conditions included wind gusts of 23 mph, coming from the west and southwest. The resulting fire hazard displayed as areas of surface or crown fire activity is shown in Map 11.

Map 11. Fire Hazard (97th percentile weather), San Juan Public Lands Geographic Area

Past Fire Activity

Past fire activity records from 1980 thru 2004 were evaluated for the San Juan Public Lands. Table 6-25 summarizes the number of fires, cause, and acres burned for each year. Figure 6-24 Fire Occurences 1980-2004 displays fire start locations and cause for this same period. Only fires that burned on the San Juan Public Lands are included.

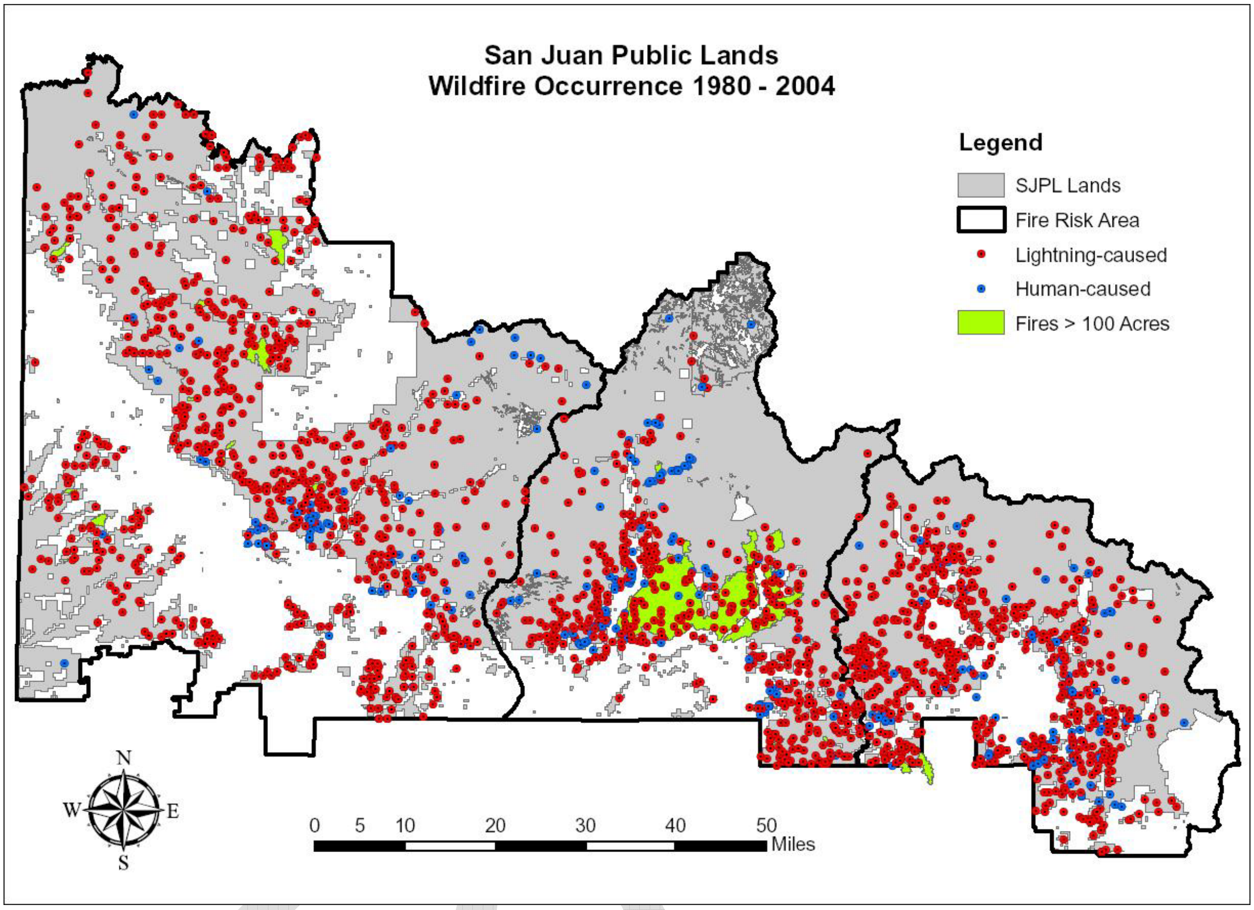
Table 6-25 Fire Activity 1980-2004 San Juan Public Lands

Table 6-25 Fire Activity 1980-2004 San Juan Public Lands Geographic Area

Year	Lightning Caused		Human Caused		Total Fires	
	# of Fires	Acres Burned	# of Fires	Acres Burned	# of Fires	Acres Burned
1980	55	143	13	20	68	163
1981	60	41	6	3	66	45
1982	31	44	4	0	35	45
1983	22	5	10	6	32	11
1984	30	13	3	0	33	13
1985	47	177	3	1	50	178
1986	23	11	4	16	27	27
1987	32	831	17	339	49	1,169
1988	54	87	7	18	61	104
1989	127	679	31	165	158	844
1990	72	1,358	8	8	80	1,365
1991	49	100	14	99	63	199
1992	39	27	13	12	52	40
1993	32	13	12	3	44	16
1994	163	2,079	11	280	174	2,359
1995	52	376	12	12	64	388
1996	159	4,662	14	700	173	5,362
1997	43	73	8	6	51	79
1998	64	1,776	13	52	77	1,829
1999	24	9	13	6	37	15
2000	181	4,854	19	122	200	4,975
2001	65	780	13	107	78	888
2002	119	80	23	74,121	142	74,202
2003	225	4,615	6	7	231	4,622
2004	124	673	14	62	138	734
Total	1,892	23,505	291	76,167	2,183	99,672
% Total	87%	24%	13%	76%	100%	100%

Fires are classified based on size (A = 0.1 to 0.25 acres, B = 0.26 to 9 acres, C = 10 to 99 acres, D = 100 to 299 acres, E = 300 to 999 acres, F = .1000 to 4999 acres, G = 5000+ acres.

Figure 6-24 Fire Occurences 1980-2004



Fire Risk - Wildfire Probability on the San JuanNational Forest

Fire risk is the likelihood that an area will be affected by fire in a given time period. Fire risk is determined from past fire activity. Fire risk is simply the number of fire starts on a per 1,000-acre basis over a ten-year period. Risk ratings are defined as (USFS 2004): Low: 0 to 0.49 – projects a fire every 20+ years per 1000 acres Moderate: 0.5 to 0.99 – projects a fire every 11-20 years per 1000 acres High: > 1.0 – project a fire every 0-10 years per 1000 acres.

The fire risk results for the three Forest Service Districts and combined BLM Field Offices in the San Juan Public Lands are shown in Table 6-26. Both lightning and human caused ignitions were considered.

Table 6-26 Fire Risk Analysis (1995-2004) San Juan Public Lands

Table 6-26 Fire Risk Analysis (1995-2004) San Juan Public Lands Geographic Areas

Analysis Area	Acres*	# of Ignitions	Lightning Ignitions	Human-Caused Ignitions	Fire Risk
Dolores District	1,198,300	521	475	46	0.4 LOW
Columbine District	744,900	261	218	43	0.4 LOW
Pagosa Springs District	586,400	409	363	46	0.7 MODERATE

* Acres rounded to nearest 100.

Prescribed Fire and Fuels Reduction Treatments

The San Juan Public Lands area uses prescribed fire and mechanical treatments to achieve multiple objectives including hazardous and natural fuels reduction, wildlife habitat improvement, ecosystem restoration, and range betterment. Smoke management has always been a challenge for the San Juan Public Lands, specifically in the fall when many days of poor smoke dispersion are forecasted by the National Weather Service (NWS). Fire managers work closely with the NWS and Colorado Department of Public Health and Environment to use a multitude of methods to reduce the amount of smoke in the various airsheds.

Prescribed fire and mechanical fuels treatments project areas tracked in a variety of databases. On National Forest lands, about 8,500-10,500 acres of hazardous fuels are treated annually, primarily through prescribed burning. Mechanical treatments have occurred only in the last several years and constitute about 30% of the overall fuels program. Yearly accomplishments vary depending on the length of the spring and fall burning windows, smoke management conditions, other wildfire activity throughout the West, availability of contingency resources and qualified burning staff, and other priorities. Historically, most of the burning (60% to 80%) has occurred close to the WUI but not within it. Recently, most of the acres receiving prescribed fire or mechanical treatments have occurred upon lands identified within the Community Fire Plans. All of the acres treated were in Condition Class 3 or 2. Since 1974, the Forest has treated approximately 141,000 acres of hazardous fuels, excluding wildfires and wildland fire use.

Since 1999 the BLM has treated about 2-3,000 acres of hazardous fuels annually, primarily by mechanical means (hydromowing, hydro-axe, some thinning) with small amounts of prescribed burning. All fuels reduction has occurred within the WUI and

Condition Class 3 vegetation. Since FY90, the BLM has treated roughly 20,000 acres of hazardous fuels, excluding wildfires and wildland fire use.

Map 13. Prescribed Burning and Mechanical Vegetation Treatments, San Juan Public Lands

Fire in the Wildland Urban Interface

Much of the focus to reduce fire risk is in the Wildland Urban Interface (WUI), areas where structures and other human development meet or intermingle with undeveloped wildland or vegetative fuels. The WUI has been identified on the San Juan Public Lands is displayed in Figure 6-25. This includes communities-at-risk (as identified in the federal register FR Vol. 66, No. 3, Pages 751-754, January 4, 2001) as well as other private lands where housing density is greater than one structure per 40 acres, buffered by 1.5 miles34. The relative risk to all these WUI areas varies depending on location, slope, aspect, surrounding vegetation (fuel conditions), type and density of the development. These risk factors must be considered in developing treatment strategies and priorities for these areas, in cooperation with other local partners (counties, communities, Colorado State Forest Service, NPS, BIA, and landowners).

In the San Juan Public Lands Geographic Area use patterns on private lands in and around the public lands have been changing from what was historically livestock ranching with little structural development, to subdivided residential areas. This is occurring throughout the area encompassed by the public lands. These changes increase the risk factors associated fire management in and around these WUI areas.

[4] The Wildland-Urban Interface in the U.S., SILVIS Lab, Forest Ecology & Management, University of Wisconsin - Madison

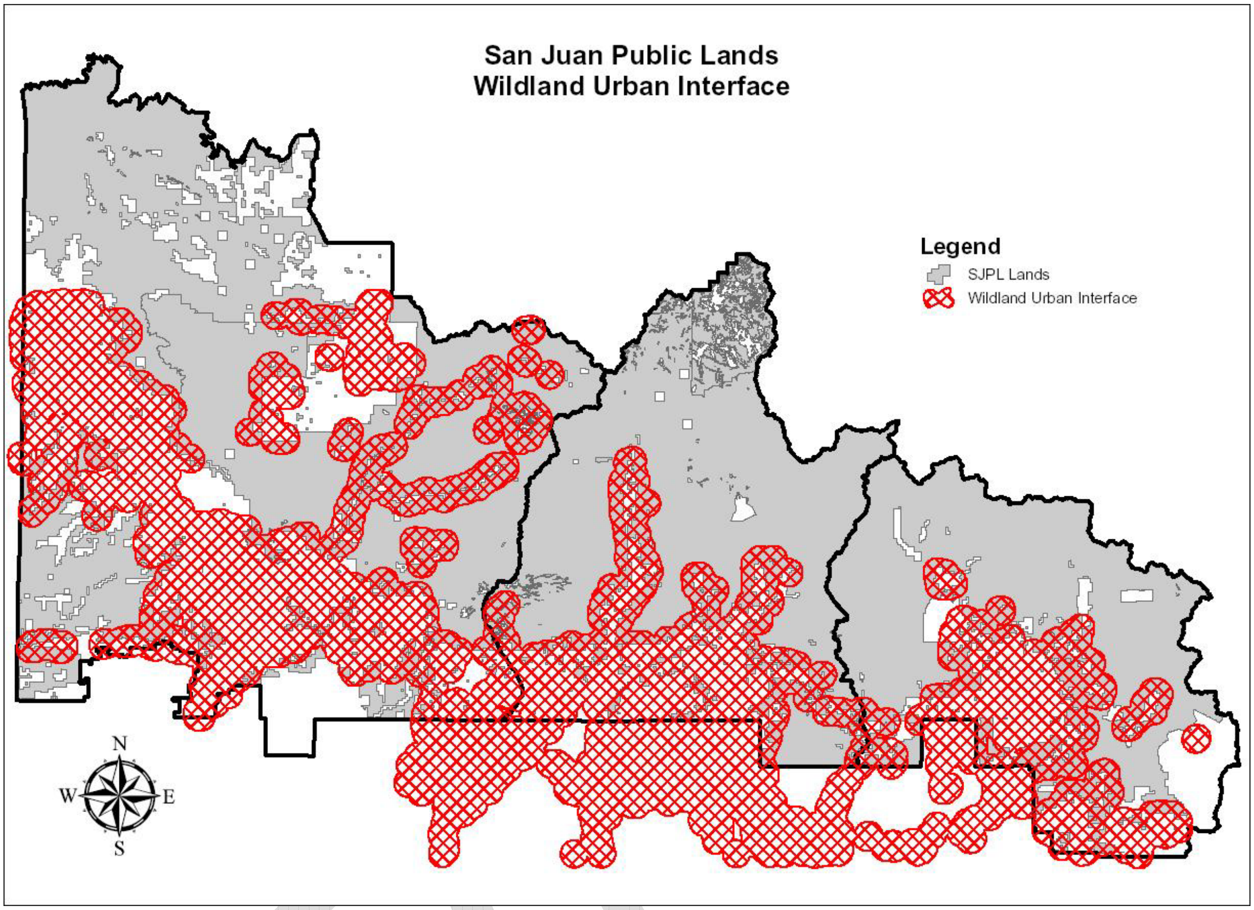


Figure 6-25 Wildland Urban Interface, San Juan Public Lands

Fire Management and Coordination Efforts

Fire management in the San Juan Public Lands Geographic Area is coordinated between multiple agencies through, the Durango Interagency Dispatch Center which serves the San Juan National Forest, the San Juan BLM Center, Mesa Verde National Park, Southern Ute Agency and Ute Mountain Ute Agency of the BIA, and the 12 surrounding counties. The dispatch center provides support for initial attack, large incidents and oversees air operations (air tankers, smokejumpers, helicopters).

Since the mid 1990s, fire management policy has evolved beyond just suppression actions to include a variety of management options, included in the National Fire Plan. The four primary goals include: improve fire prevention and suppression, reduce hazardous fuels, restore fire adapted ecosystems, and promote community assistance. Fire management policy differs between the various agencies and landowners (USFS, BLM, NPS, State of Colorado, and private property owners) that have jurisdiction in the Geographic Area. Wildfires on private lands are suppressed by rural and county fire departments; full suppression of fires is the goal of these agencies. For public lands managed by the SJPL's, four categories of treatment options guide fire management and fuels treatment (described in the San Juan Fire Plan, April 1997, amended to the San Juan Public Lands Fire Management Plan , 2004. This plan allows the use of prescribed fire and Wildland fire use under the following categories: Areas where fire is not desired. Fire never played a significant role in ecosystem function or suppression is required to prevent direct threats to life or property. All fires will be aggressively suppressed. Areas where the four categories described above would be applied have been identified and are displayed in Map 15, Fire Management Emphasis.

Table 6-27 Fire Management Emphasis Area Descriptions

Table 6 27 Fire Management Emphasis Area Descriptions

Areas where wildfire is not desired. Unplanned ignitions could have negative effects on resource values (i.e. wildland urban interface lands, cultural resources, areas with unnatural fuels build-up) without mitigation. Fire suppression will be aggressive. However, prescribed fires and/or mechanical treatments will be considered to reduce hazards when resource concerns can be mitigated.
Areas where fire is desirable but social, economic and ecological constraints must be considered (i.e. State air quality emission standards, wildlife species and habitats). A variety of suppression efforts may be used. Prescribed fire and mechanical fuel reduction treatments are acceptable tools for meeting resource objectives.
Areas where fire is desired and there are few resource constraints to its use. Fires may be managed under a Wildland Fire Use strategy which allows a full range of appropriate management responses. Prescribed fire and mechanical fuel reduction treatments are also acceptable tools for meeting resource objectives.
Areas where fire is desired and there are few resource constraints to its use. Fires may be managed under a Wildland Fire Use strategy which allows a full range of appropriate management responses. Prescribed fire and mechanical fuel reduction treatments are also acceptable tools for meeting resource objectives.

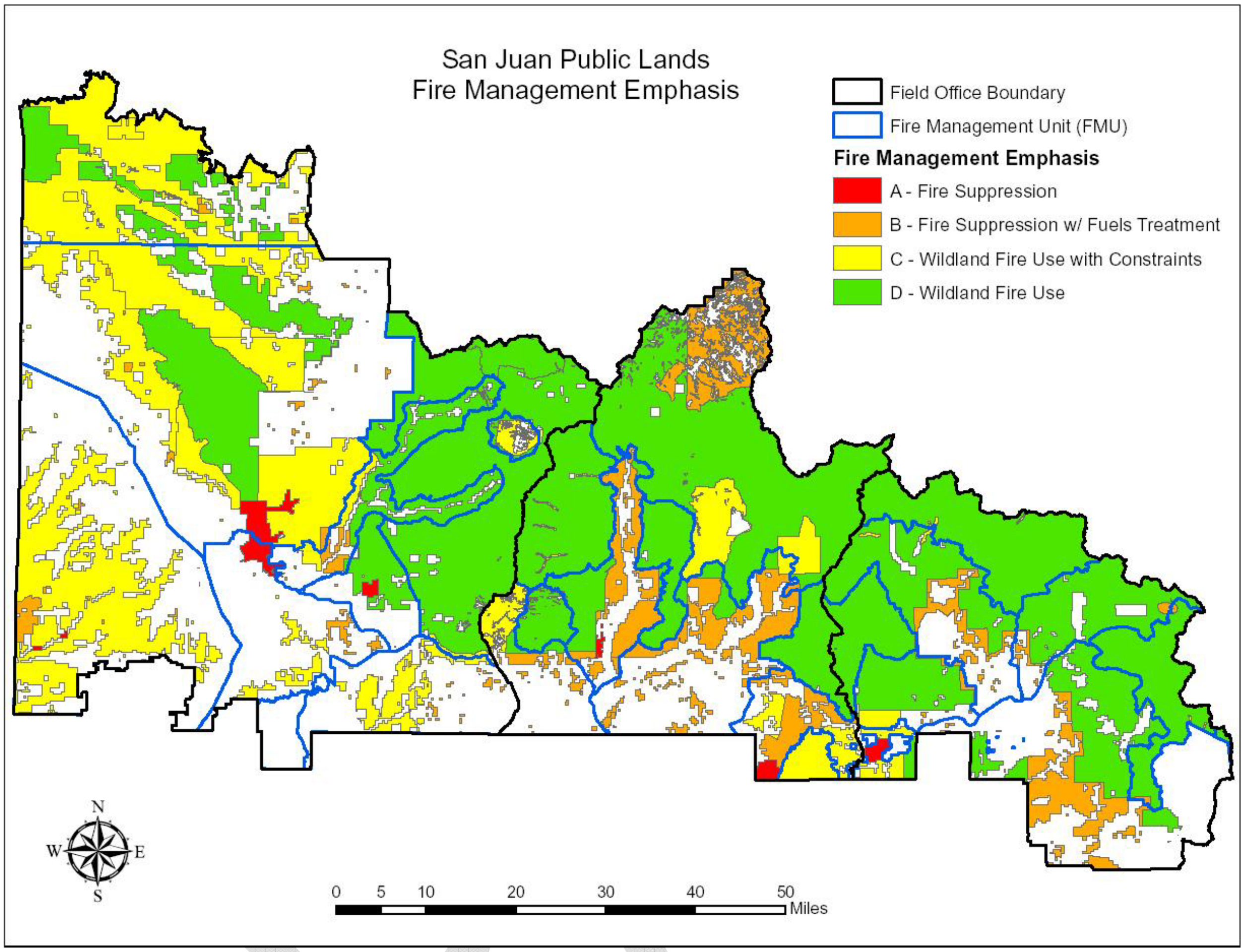


Figure 6-26 Fire Management Areas, San Juan Public Lands

Wildland Fire Use

Wildland Fire Use is defined as the application of the appropriate management response to naturally-ignited wildland fires to accomplish specific resource management objectives in predefined designated areas outlined in fire management plans. Wildland fires will be used to protect, maintain, and enhance resources and, as nearly as possible, be allowed to function in its natural ecological role. Use of fire will be based on approved fire management plans and will follow specific prescriptions contained in operational plans. Wildland fire use, based on Federal Fire Policy direction, is a direct component of wildland fire management. It is a management action equal to wildfire suppression and thus, constitutes an emergency action. It receives consideration, management attention, and management policies equal to wildfire suppression, except for specific differences related to ignition source and management action. In 1997 San Juan Public Lands unit was the first public lands office in the state to initiate a wildland fire use program via an environmental assessment, when wildland fire use was called “Prescribed Natural Fire”. Since that time, the program have evolved throughout most federal land management agencies and can be used if wildland fire use is a part of an approved fire management plan and is consistent with local unit land and resource management plans.

Table 6-28 illustrates the number of wildland fire use fires managed by the San Juan Public lands unit since 1997.

Table 6-28 San Juan Public Lands, Wildland Fire Use Fires

Table 6 28 San Juan Public Lands, Wildland Fire Use Fires

Year	Number of Wildland Fire Use Fires	Total acres
1997	8	50
1998	13	1184
1999	5	5
2000	2	330
2001	11	740
2002	0	0
2003	4	1873
2004	8	636
2005	7	2313

The San Juan Public Lands has developed an Accelerated Watershed/Vegetation Restoration Plan (AWRP), created under the auspices of the National Fire Plan and Healthy Forest Initiative. The main focus of AWRP is to implement fuels and vegetation treatments in WUI areas adjacent to communities-at-risk, and/or in important watersheds to improve safety to the public and firefighters, reduce the threat to real property, infrastructure and municipal watersheds, and in the long term restore or enhance ecological conditions.

Key Findings

- The five historic fire regimes distribution on the San Juan Public Lands is: I - < 1%, II – 22 %, III – 25%, IV – 5%, V – 41%. Water and bare/rock areas that do not burn make up the remaining 7%.
- Since 1980, 2,183 wildfires have occurred on Forest and BLM lands, burning 99,672 acres. Eighty seven percent (87 %) have been caused by lightning, thirteen percent (13 %) have been human-caused. Most of these fires were very small (74 % were < 0.25 acres, 21 % were between 0.25 and 10 acres), accounting for 1.2 percent of the total area burned.
- The Missionary Ridge fire accounts for 73% of the total burned area. Based on 25 years of fire history, the San Juan Public Lands a relatively moderate risk for fire occurrences.
- Wildland Urban Interface areas occur on approximately 25 % of the public lands.

Trends

- Between 1980 and 2004 there has not been a clear trend in human-caused fires with 12 years having a number of human-caused fires above the 13 % average. The period from 1995 to 2004 had 6 years with numbers above the average with an overall percentage of 11 % human-caused fires, while the period from 1985 to 1994 also had 6 years with numbers above the average with an overall percentage of 15 % human-caused fires.
- Lightning caused fires tend to be more weather related. There has been a short upward trend in the number of lightning caused fires over the past few years. The associated trend in lightning acres burned between 1980 and 2004 has been upward.
- The trend in acres affected by fire is related to trends in vegetation conditions – increasing age, density, and fuel loading in all woodland and forest cover types over the past 30 years. These conditions have the potential to allow fires to spread to larger areas and burn with higher intensities than would have occurred historically.
- Fire size trends are also related to weather conditions. Drought conditions have prevailed in western Colorado for the past five years. These weather conditions not only stress vegetation making it more susceptible to insect attack and mortality; mortality due to drought is also increase. This is resulting in increasing amounts of dead fuel building up on the San Juan Public Lands Geographic Area.
- Development of private land within public land boundaries is dramatically increasing on the as land use changes from livestock ranching to subdivisions.

Need for Change

The increased public and political awareness concerning the effects of fire or lack of fire in forest landscapes will be a driving force directing future management of public lands. The National Fire Plan, 10-year Comprehensive Strategy and Implementation Plan, Healthy Forest Initiative and most recently the Health Forest Restoration Act are directing immediate actions to reduce the risk of wildfire on landscapes that currently have high risk.

It is important to consider integrated ecological, social and economic perspectives when prioritizing and designing restoration and fuel treatment projects. Projects need to consider long-term effects as well as short-term consequences. Constraints such as budgets, public acceptance of different practices, resource conditions, habitat requirements, air quality standards, and contractor/skill availability must also be considered.

As mentioned throughout this section,, current dense stand conditions in most woodland and forest cover types increase the potential for high-intensity stand replacing fires when fire does occur.

Before fire can be reintroduced into many areas of the San Juan Public Lands Geographic Area the amount of vegetation, both in the overstory and understory, may first need to be reduced by some sort of mechanical treatment(s) (i.e. thinning, harvest, mastication, etc.).

Chapter 7. Anthropogenic Disturbances

Subregional Scale

Management of Forest and Woodland Ecosystems (Module 4B)
Page intentionally blank.

Management of Grassland and Shrubland Ecosystems (Module 4C)
Page intentionally blank.

Invasive Plant Species (Module 4D)
Page intentionally blank.

Roads and Trails (Module 4E)- Key Findings and Introduction

Key Findings

- There is an estimated 75,773 miles of road and trail in the subregion based on U.S. Geological Survey 100,000 scale Digital Line Graph (DLG) data. Of these, there are 63,111 miles of road excluding trails.
- There are an estimated 287,059 acres of disturbance result from 75,773 miles of roads and trails in the subregion.
- Class 3 gravel roads comprise 14.3% percent and Class 4, dirt/native surface roads comprise 60.8% of all roads in the subregion. These two classes sum to 56,957 miles for 75.2% of all roads and trails.
- Class 4, 4WD and Trails are dirt/native surfaced and disturb 177,169 acres, about 62% of the total disturbance from roads in the subregion.
- Slope may be correlated to road densities to generate a road density proxy layer. This slope proxy layer can be used to anticipate areas of high road activity and augment coarse scale data.
- Weighted mean road densities are higher for lowland, dry vegetation types and decrease in an orderly way from lowland to foothills to upland vegetation types. In the western reaches of the San Juan Forest and the Uncompahgre Uplift portion of the GMUG, Ponderosa Pine and Deciduous Oak vegetation communities have some of the highest road densities on these Forests.
- The highest road densities are found on private lands. Relatively high densities are found also on tribal, state and BLM lands. While road densities on Forest Service lands can be locally high, overall area weighted densities are low because of the relatively large areas of upland/high slope terrains on Forest Service lands.

Introduction

This section summarizes road and trail mileage, distribution and pattern and potential for the CLC subregion. We estimate the area of disturbance on the landscape due to roads. Roads are further summarized by their relation to major vegetation classes (GAP) and by land ownership.

The base data supporting the mapping and analysis of roads is U.S. Geological Survey (U.S.G.S.) 1:100,000 scale digital line graph (DLG) data (U.S.G.S., 2004a). The DLG data provides accurate representations of Interstate, U.S. Highway, State and County roads. While these DLG data also fairly represent the over all regional pattern and distribution of backcountry dirt roads and trails, finer scale analyses within the San Juan and GMUG forests bounds reveal higher road densities, especially for non-system roads and tracks. Within this analysis we develop a slope based model to partially overcome these data limitations.

Roads and Trails (Module 4E)- Road Class

Road Class

Using the U.S.G.S. DLG data, there are an estimated 75,773 miles of road, including trails, in the subregion. These roads are classified by the U.S. Geological Survey as primary, secondary, Class 3, Class 4, Four Wheel Drive (4WD) and trail (Table 7-1). A small proportion of the DLG data was not suitable for classification and is classified as “NotDefined”. Subtracting trail mileage, there are a total of 63,111 miles of road in the subregion.

Table 7-1 Road Classes in the subregion.
Percentage values are relative to the total subregion mileage.

Table 7-1 Road Classes in the subregion. Percentage values are relative to the total subregion mileage.

Road Class	Road Surface	Miles	Pct
Primary	Paved	1,849	2.4%
Secondary	Paved	2,710	3.6%
Class 3	Gravel	10,854	14.3%
Class 4	Dirt	46,103	60.8%
4WD	4WD	1,596	2.1%
Trail	Trail	12,459	16.4%
Not Defined	Not Defined	202.0	0.3%
		75,773	100.0%

Primary roads include Interstate 70 and roads in the U.S. Federal Highway system and form the backbone of the highway network in the subregion. These roads comprise about 2.4% percent of all roads and traverse 1,849 miles of principally low dry basin-land, foothills and follow major drainages and cross major mountain passes in the subregion. Primary roads are highly engineered and are fully paved, maintained and include erosion control structures and systems (Fig. 7-1).

Figure 7-1 Primary roads in the subregion.
These roads traverse 1,849 miles.

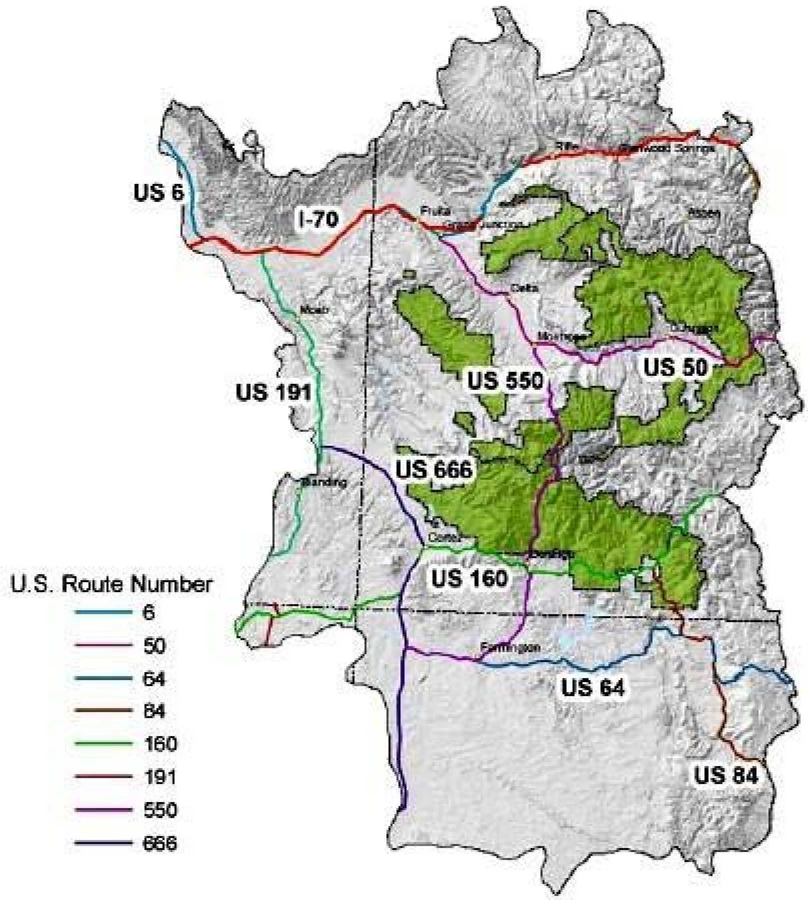


Figure 7-1 Primary roads in the subregion. These roads traverse 1,849 miles.

Secondary roads include paved state, county and agency roads. These roads link up communities, rural areas, agricultural areas and federal lands to the primary road network. Class 3 roads, include maintained gravel roads, maintained by local communities, counties and agencies form local community networks, interconnecting rural homes, farms, development areas and recreation areas. Combined, secondary and Class 3 roads traverse 13,564 miles and comprise about 18% percent of the total road and trail network in the subregion (Fig. 7-2).

Figure 7-2 Primary, secondary and Class 3 roads in the subregion.

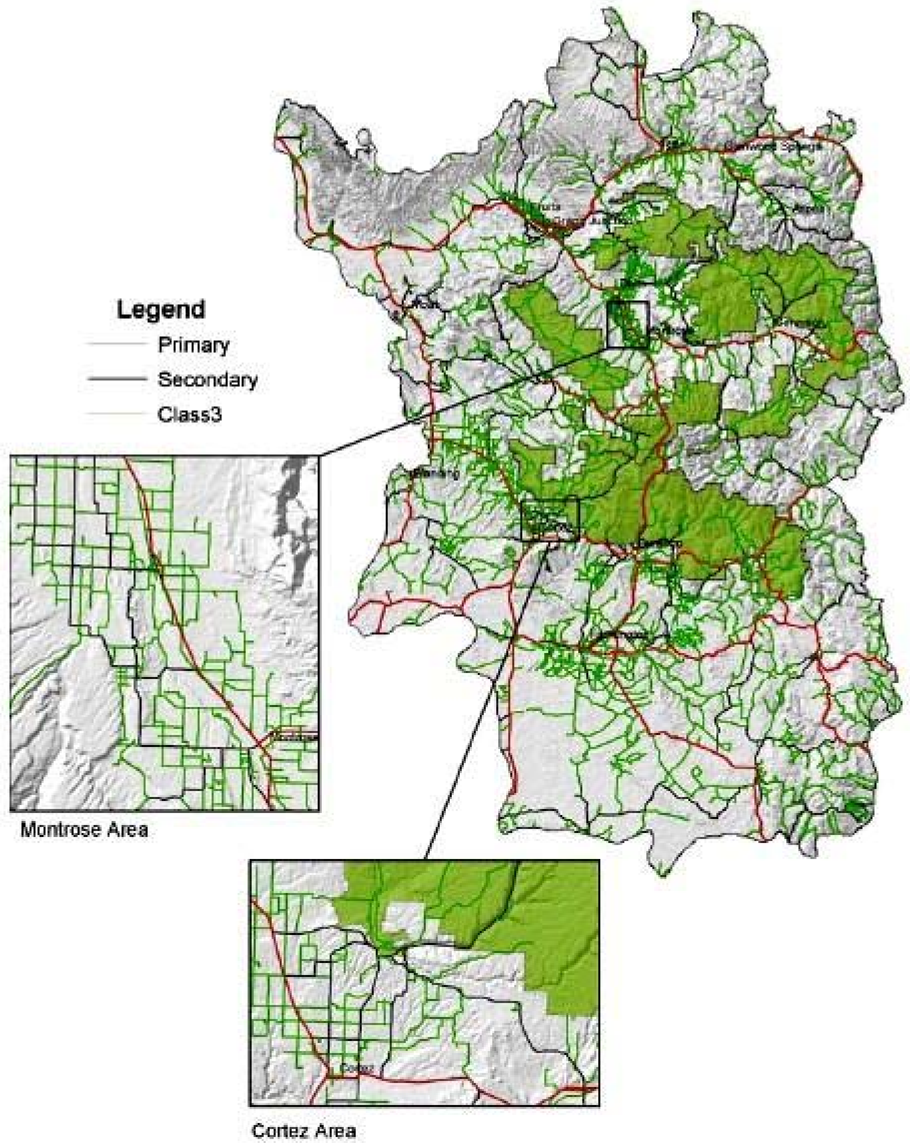


Figure 7-2 Primary, secondary and Class 3 roads in the subregion. Combined, secondary and Class 3 roads traverse 13,564 miles, about 18% percent of all roads and trails in the subregion.

Combined, secondary and Class 3 roads traverse 13,564 miles, about 18% percent of all roads and trails in the subregion.

Class 4 roads are typically dirt/native surfaced and form a dense network, weaving into all but the most rugged landscapes in the subregion. This network is comprised of both private, county and agency maintained roads. These roads typically invade native terrain to support recreation, hunting, agriculture, vegetation harvest and fire control and mineral development (Fig. 7-3). There are 46,103 miles of Class 4 roads in the subregion. On the basis of mileage, these are by far the dominant road class with 60.8% of the total road and trail mileage in the subregion. Significantly, these roads, having native surfaces and minimal erosion control structures significantly contribute sediment and contamination to local aquatic systems.

Figure 7-3 Class 4 roads traverse 46,103 miles of the subregion.
Class 4 roads traverse 46,103 miles of the subregion and comprise 60.8% percent of all roads and trails in the subregion.

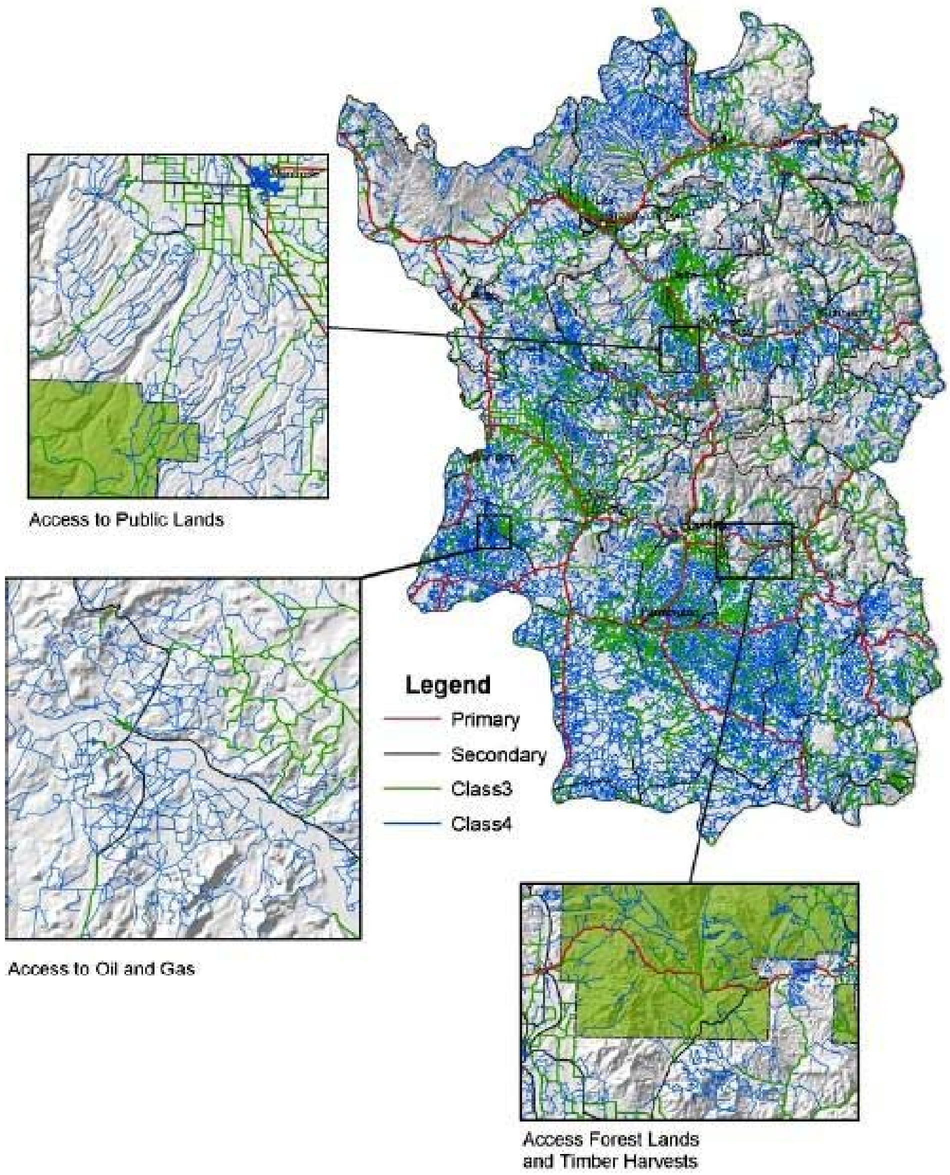


Figure 7-3 Class 4 roads traverse 46,103 miles of the subregion and comprise 60.8% percent of all roads and trails in the subregion. The highest densities of roads are evident in the subregion map as the blue road lines merge into fields of color.

The highest densities of roads are evident in the subregion map as the blue road lines merge into fields of color.

The most primitive elements of the subregion transportation network are naturally, four-wheel drive (4WD) and trails (Figure 7-4). In some areas, 4WD roads have emerged as the remnants of historic mining and are popular recreational attractions and important to some local economies. More recently, some of these roads occur in areas of mineral exploration, especially oil and gas. However, many come about as users of all terrain vehicles and sturdy 4WD vehicles push tracks further and further into frontier areas. This is especially problematic on public lands – principally BLM and Forest Service lands. Trails typically occur in areas otherwise inaccessible to large vehicles. Greater proportions of trails occur in the upland and alpine areas.

Figure 7-4 Four Wheel Drive (4WD) roads and Trails in the subregion.

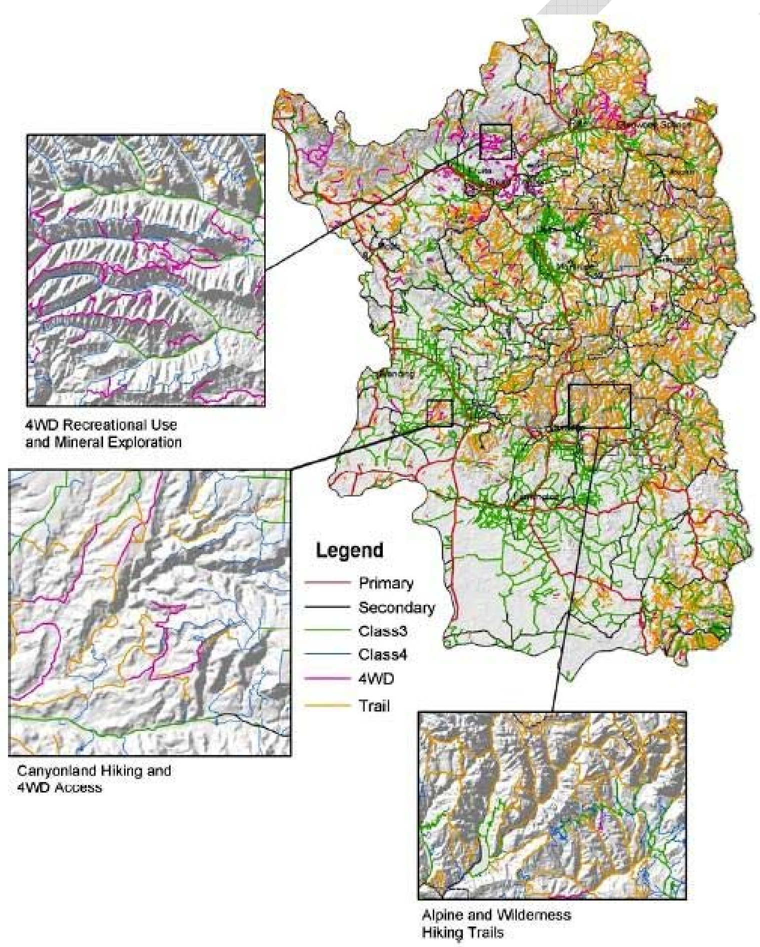


Figure 7-4 Four Wheel Drive (4WD) roads and Trails in the subregion. 4WD roads and trails traverse 14,055 miles in the subregion. (Class 4 roads are not shown in the subregion view). Actual road densities are higher in some Forest areas and public lands than indicated by the U.S.G.S. data shown here.

4WD roads and trails traverse 14,055 miles in the subregion. (Class 4 roads are not shown in the subregion view). Actual road densities are higher in some Forest areas and public lands than indicated by the U.S.G.S. data shown here.

Roads and Trails (Module 4E)- Disturbance Estimates

Disturbance Estimates

An overall estimate of disturbed area resulting directly from roads and trails may be estimated by multiplication of roadway lengths times estimated widths. Width estimates are based on measurement of highway widths outside of the subregion and therefore assumes that generally road width standards are common inside and outside of the subregion. Using these estimates there are about 287,059 acres of disturbed area in the subregion (Table 7-2 and Fig. 7-5).

Table 7-2 Estimated disturbance area in subregion.
Estimated disturbance area in subregion based on disturbance widths that include the road surface and beyond to the edge of the right-of-way.

Table 7-2 Estimated disturbance area in subregion based on disturbance widths that include the road surface and beyond to the edge of the right-of-way.

Road Class	Example Hwy.	Disturbance Width Feet	Road Length Miles	Disturbed Area Acres
Primary	U.S. 36	135	1,849	30,248
Secondary	Colo. Hwy. 117	80	2,710	26,282
Class 3	Broadway	40	10,854	52,624
Class 4	Saw Mill Rd	30	46,103	167,647
4WD	Two Track	18	1,596	3,482
Trail	Estimated	4	12,459	6,041
NotDefined	Estimated	30	202.0	735
			75,773	287,059

Figure 7-5 Relative proportions of disturbance area by road class.

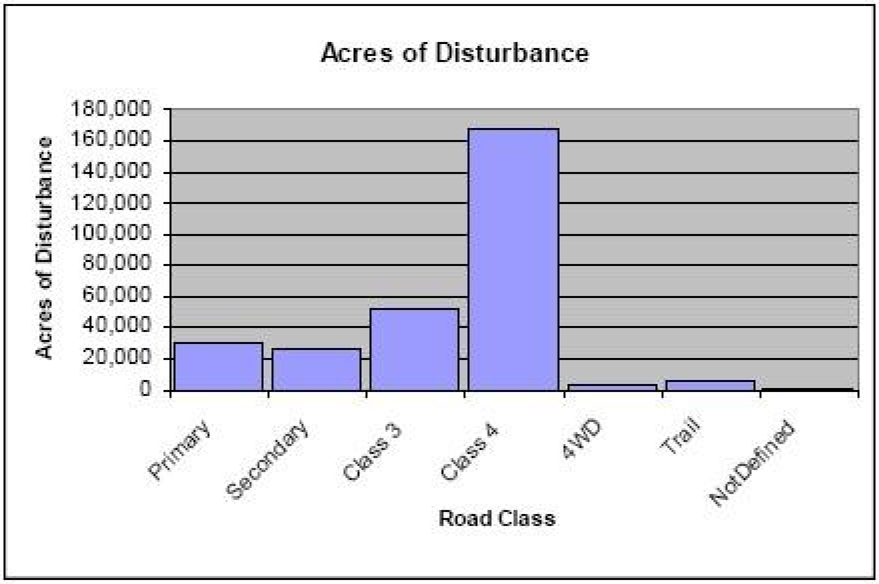


Figure 7-5 Relative proportions of disturbance area by road class.

About 20% percent of the disturbed areas are by hard surfaced roads. These roads are surfaced with asphalt and concrete and other road structures and right-of-way areas are designed to limit erosion and sedimentation in runoff. Class 3 roads constitute about 18% percent of roadway disturbed area. Sedimentation levels from these roads are higher than hard surface roads but somewhat mitigated by the gravel surface and road engineering.

Significantly, the remaining 62% of roads and trails have dirt/native surfaces and are most subject to erosion and the generation of sediment. Moreover, many of these are not maintained and continue to degrade over time, expanding road disturbance area and consequent sedimentation (Table 7-3).

Table 7-3 Estimated disturbance areas by road surface type.

Table 7-3 Estimated disturbance areas by road surface type.

Class	Surface	Area of Disturb	Pct
Primary,Secondary & NotDef.	Paved	57,265	20%
Class 3	Gravel	52,624	18%
Class 4, 4WD, Trail	Dirt/Native	177,169	62%
		287,059	100%

Roads and Trails (Module 4E)- Disturbance Estimates- Road Density

Road Density

Road density models of the subregion provide important aerial measures of distribution. These distribution measures are useful to highlight road density hotspots where road influences are the highest and where they are minimal or absent. Importantly, road density data may be overlain with vegetation and ownership to gain insight into the terrestrial communities most influenced by roads and those agencies with the greatest responsibility and opportunity to mitigate these influences. Moreover, in this analysis road density models can be strongly correlated to slope models to generate predictive models of road density. These predictive models allow managers to anticipate areas of greatest risk from road activity and encroachment.

Road density is expressed as unit length per unit area. The units expressed here are miles of road per square mile. In this analysis we excluded trails and examined the density of roads only. Road class and surface are all aggregated to yield gross road density estimates.

First and most generally, in the subregion, the gross road density is 63,111 road miles divided by 53,315 subregion square miles area for a gross measure of road density of 0.84 miles per square mile.

More specifically, by draping a 1 mile x 1 mile grid on the subregion we find that densities in the subregion range from extremes 0.0 to 24.6 miles per square mile. These are calculated by summing miles or road (excluding trails) per grid polygon. Aggregation of the resulting densities into seven classes shows that almost two-thirds of the subregion has road densities of zero. In the remaining lands, where densities are greater than zero, the average road density is 1.7 milesper mile (Table 7-4). Densities tend to be in lower elevations and basins in proximity of urban and agricultural areas. On public land, higher densities tend to be associated with mineral development, vegetative treatments and recreation areas (Fig. 7-6).

Table 7-4 Road Density classes in the subregion.
Almost two-thirds of the subregion has densities of zero.

Table 7-4 Road Density classes in the subregion. Almost two-thirds of the subregion has densities of zero.

Density Class	Density Mi/Mi	Acres	Pct. Of Subregion	Average Road Density
0	0	39,665,904	61.9%	0.0
1	0 to 1	6,938,245	10.8%	0.5
2	1 to 2	9,678,080	15.1%	1.4
3	2 to 3	5,105,274	8.0%	2.4
4	3 to 4	1,840,636	2.9%	3.4
5	4 to 5	540,163	0.8%	4.4
6	> 5	295,681	0.5%	6.7
		64,063,982	100.0%	1.7

Figure 7-6 Road density in the subregion.

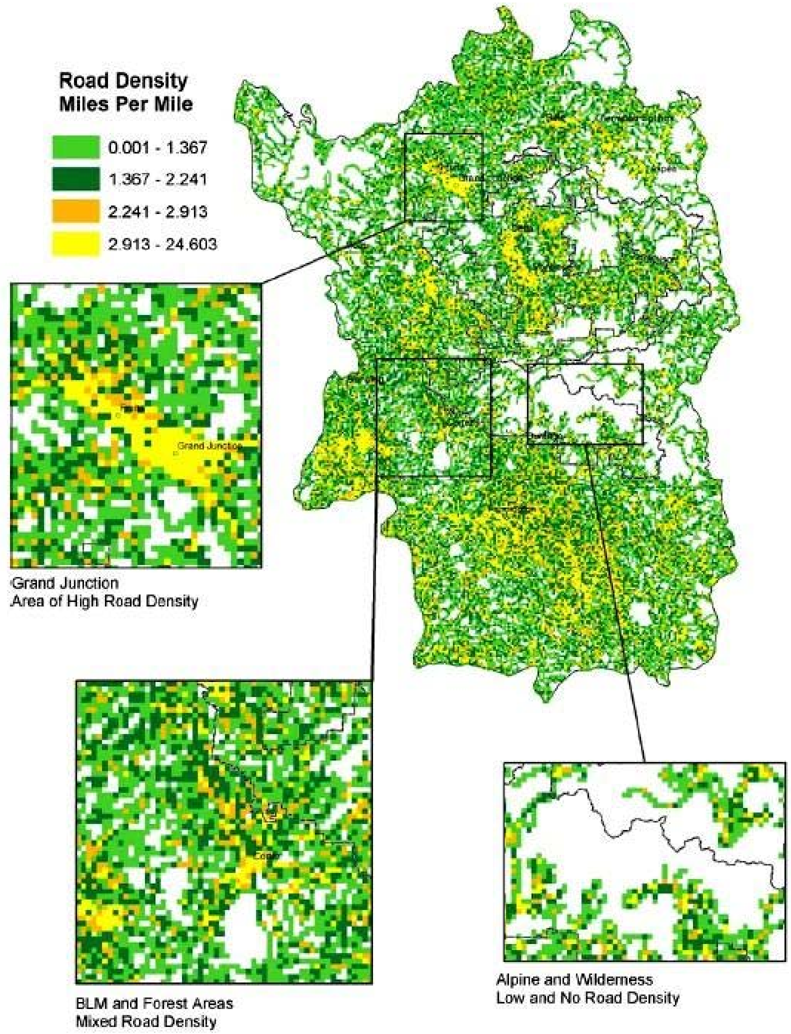


Figure 7-6 Road density in the subregion.

As noted above, the U.S. Geological Survey 100K DLG data does not always fully represent roads on the landscape. While these data fairly represent the primary and secondary and Class 3 roads, they fall short in representing Class 4, 4WD and trails. This is evident in where both San Juan and GMUG Forest data sets provide greater detail and insight. Consequently, road densities using road layers for these Forests are higher in some areas than is evident in the subregion wide densities calculated using the U.S.G.S. DLG data (Fig 7-7). A method to address this is discussed next.

Figure 7-7 Road densities calculated using San Juan Forest roads data.
Road densities calculated using San Juan Forest roads data yields densities greater than densities calculated using U.S.G.S. 100K DLG roads data.

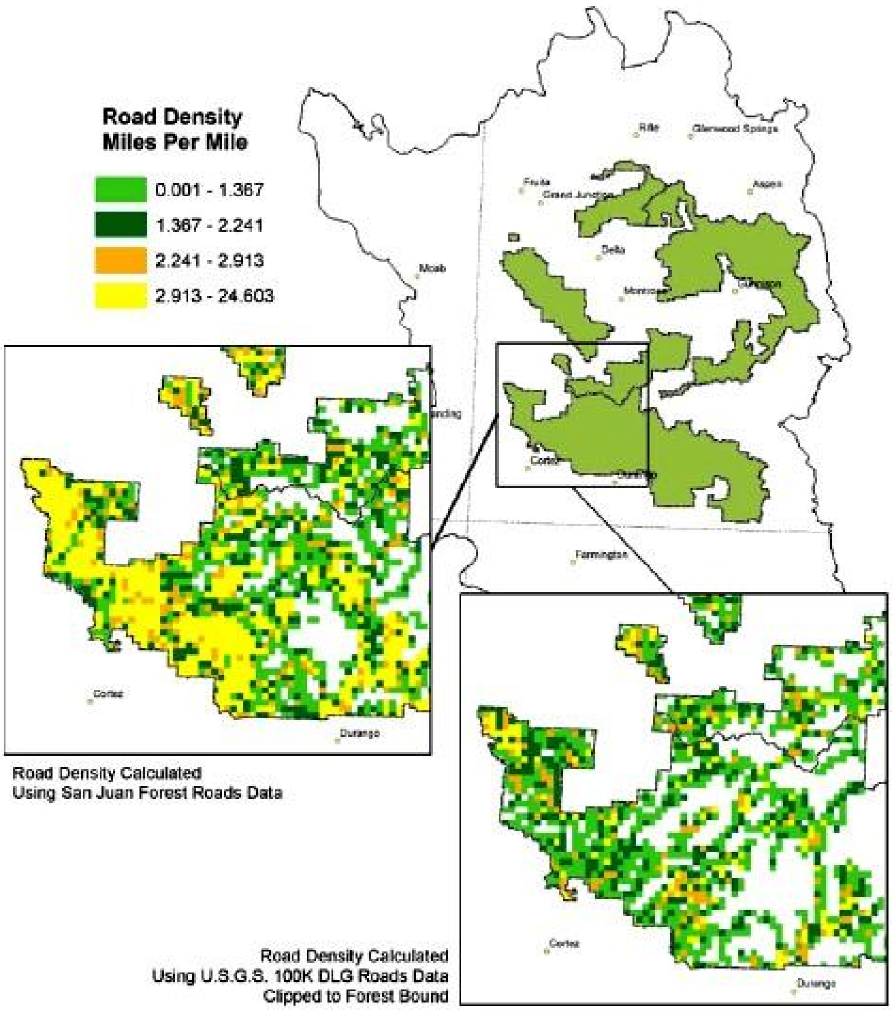


Figure 7-7 Road densities calculated using San Juan Forest roads data yields densities greater than densities calculated using U.S.G.S. 100K DLG roads data.

We have noted in this analysis that Class 4 roads account for the most significant levels of mileage and disturbance in the subregion. Yet, it is evident from the discussion above, road densities for Class 4 and 4WD roads are understated in some cases, where the U.S.G.S. DLG data is used as foundation data. To meet this shortcoming, in the absence of more comprehensive layers, we developed a proxy model to assess areas of highest potential road densities throughout the subregion.

Slope strongly correlates to the position of roads on the landscape. Qualitatively, this relationship is self evident – roads on steep terrain are difficult to build and navigate.

Correlating slope to road density we find that a strong linear relationship exists between slope and road density. The relation holds for road densities calculated using the U.S.G.S. DLG data as well as the San Juan/GMUG roads data (Figs. 7-8 and 7-9). The shape of the relation in both functions is notable.

Figure 7-8 Correlation of U.S.G.S. DLG roads layer based road density to subregion wide slope.

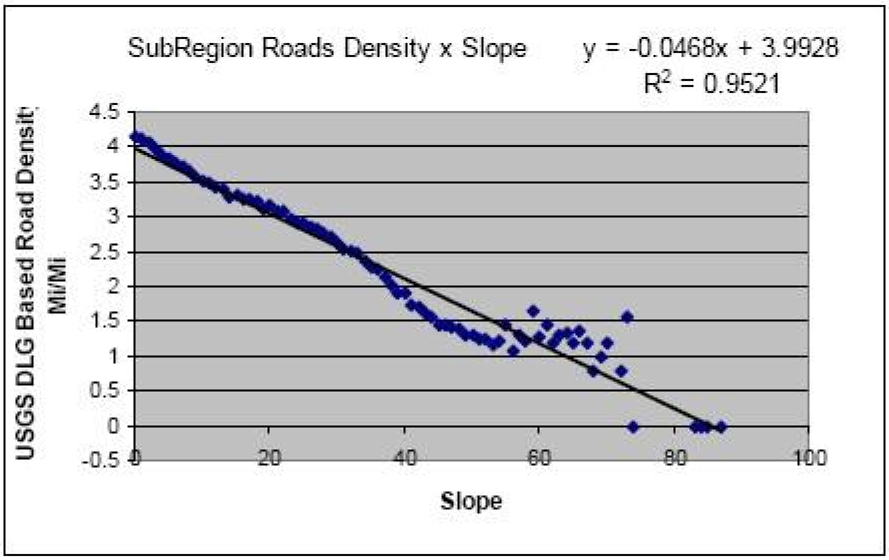


Figure 7-8 Correlation of U.S.G.S. DLG roads layer based road density to subregion wide slope.

Figure 7-9 Correlation of San Juan/GMUG roads layer based road density to subregion wide slope.

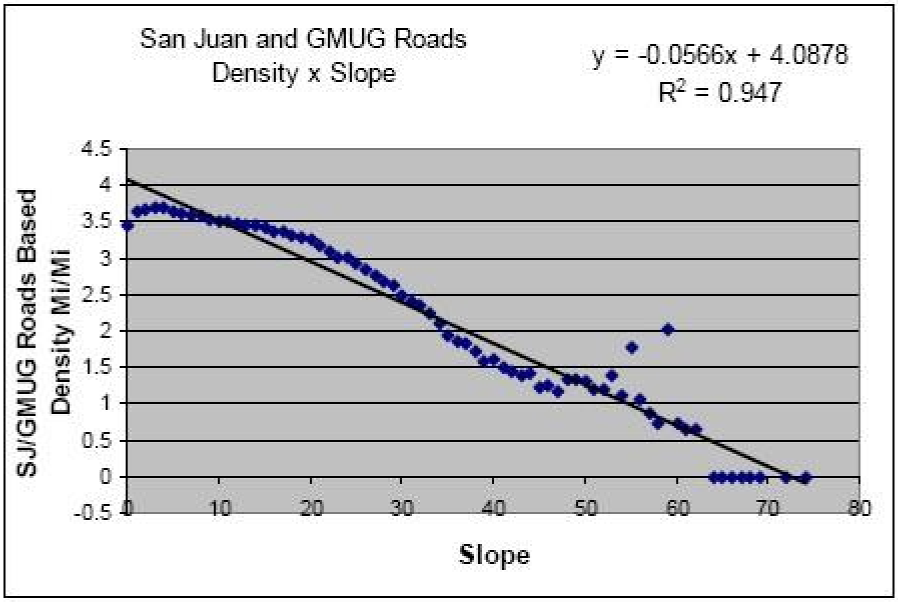


Figure 7-9 Correlation of San Juan/GMUG roads layer based road density to subregion wide slope.

Using GRID functions in Arc/Info it is possible generate a synthetic road density layer by applying the linear functions shown in Figure 7-9 to a slope layer. The slope layer used is a 100 meter ARC Grid covering the entire subregion. Arc/Info GRID algebra makes this operation very simple. The expression is:

RoDenGrid = = -0.0468 * SlopeGrid + 3.9928.

Patterns in the resulting synthetic road density grid are identifiable in the polygon based road density layer, calculated using existing road data (Fig. 7-10). In the figure, areas of high potential road density correspond to areas with high densities calculated from existing roads data. More significantly, areas with high slopes and the lowest densities are evident.

This discrimination allows managers to localize areas with both the highest and lowest potentials for roads. For example, it is clear, from existing data that areas of the western San Juan Forest are significantly influenced by high road densities. What is less clear from the road data and the road density models is where future road encroachment is likely to take place. This slope based potential road density model can be aimed at that problem. Moreover, it may be used to augment data, such as the U.S.G.S. DLG data, across the subregion.

Figure 7-10 The synthetic road density layer calculated

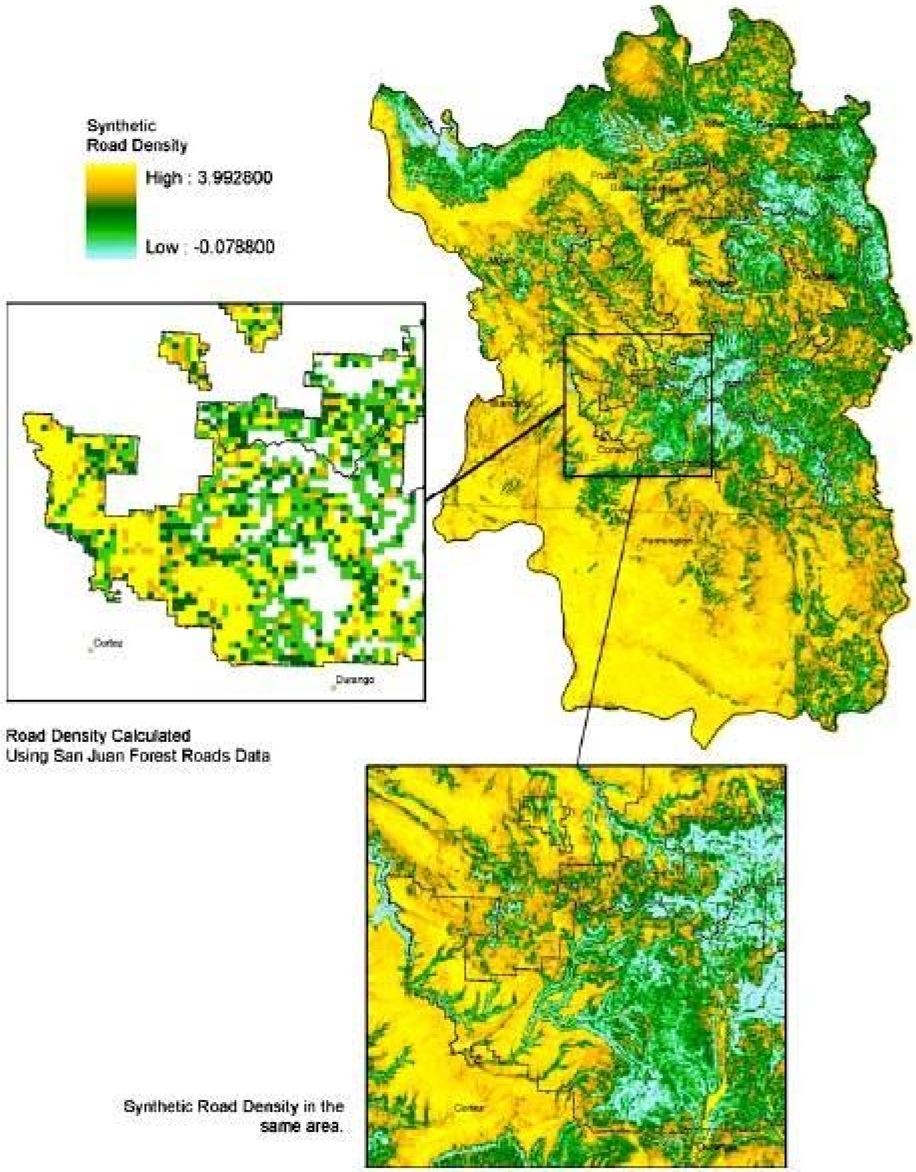


Figure 7 10 The synthetic road density layer calculated by applying the function in Figure 7-8 compared to high road densities in the western San Juan National Forest. The synthetic density layer allows managers the opportunity to better discriminate the likelihood of both high and low road density.

Roads and Trails (Module 4E)- Disturbance Estimates- Road Density and Gap Vegetation

Road Density and Gap Vegetation

Road density varies among the twenty GAP classes categorized for the subregion from a high of 6.59 miles per mile, for the urban category, to 0.29 miles per mile, for the alpine category (Table 7-5). On the landscape an ordered trend of decreasing road densities, by vegetation type follows the trend of lowland to foothill to upland vegetation types (Fig. 7-11). This reflects cultural preferences, agricultural activity and the higher relief of upland terrain. Within the San Juan and GMUG Forests the Ponderosa Pine and Deciduous Oak types (western San Juan and the Uncompahgre Uplift) are most influenced by high road densities.

Overall, the highest densities are in lowland/dry vegetations types including desert grassland, greasewood, desert shrub and sagebrush. These types also correspond to areas of highest potential road density as identified above.

Conversely, the lowest road densities area in upland areas and include the aspen, lodge pole, spruce-fir and alpine vegetation types. Vegetation types most at risk correspond to those displaced by both urban areas and crops along with valley bottom riparian types. And again, these areas correspond largely to upland areas of lowest potential based on the slope model discussed above.

Table 7-5 Ranked road densities.
Ranked road densities. Density values are the weighted mean for each GAP vegetation category in the subregion.

Table 7-5 **Ranked road densities. Density values are the weighted mean for each GAP vegetation category in the subregion.**

GAP Class	Weighted Mean Road Density
urban	6.59
crops	2.31
desert grassland	1.82
woody riparian/wetland	1.70
greasewood	1.45
water	1.40
mountain shrubland	1.31
desert shrub	1.31
sagebrush	1.31
ponderosa pine	1.23
mountain grassland	1.21
pinyon -juniper	1.20
herbaceous riparian/wetland	1.15
barren	1.06
deciduous oak	1.06
mixed conifer	0.91
aspen	0.73
lodgepole pine	0.65
spruce - fir	0.59
alpine	0.29

Figure 7-11 Ranked GAP vegetations classes, arranged increasing weighted road densities for each class from Alpine (0.28 miles/mile) to Urban (6.59 miles/mile).

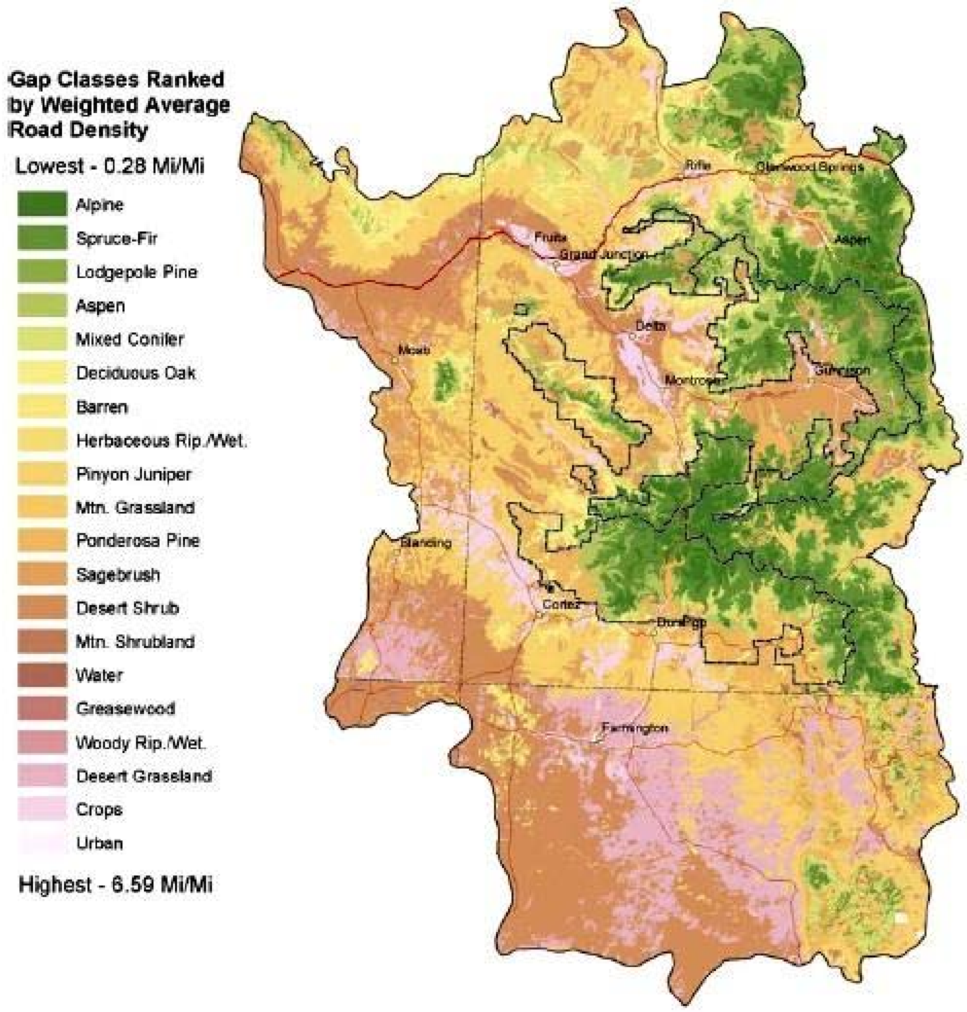


Figure 7-11 Ranked GAP vegetations classes, arranged increasing weighted road densities for each class from Alpine (0.28 miles/mile) to Urban (6.59 miles/mile).

Roads and Trails (Module 4E)- Disturbance Estimates- Road Density and Ownership

Road Density and Ownership

Road densities in the subregion are the highest on private lands. This, naturally, corresponds the highest densities also being found in urban areas and crop lands where the lands are almost exclusively private. Higher densities on tribal, state and BLM lands reflect both the lowland/low slope settings in these jurisdictions as well as elevated levels of development, especially for oil and gas. While local densities on National Forest lands can be high, the overall weighted mean is among the lowest, due to the amount of upland/high slope settings and there.

Because of the displacement of lowland dry vegetation types in Urban and agricultural areas is so high, it may be important that the BLM and Forests aim management at limiting further disturbance in these lowland/dry vegetation communities.

Table 7-6 Ranked road densities.
Ranked road densities. Density values are the weighted mean for each ownership category in the subregion.

Table 7-6 Ranked road densities. Density values are the weighted mean for each ownership category in the subregion.

Ownership	Weighted Mean Road Density
Private	1.72
Tribal	1.54
BOR	1.39
State	1.12
BLM	1.12
DOD	0.81
NPS	0.78
USFS	0.67
FWS	0.42

Roads and Trails (Module 4E)- Information Needs and Gaps

Information Needs and Gaps

A more comprehensive survey of roads and trails is needed for the Class 4, 4WD roads and trails. These types of roads are especially problematic because of the unauthorized and invasive modes of their creation and often without mitigation. Furthermore, because these roads are continually evolving and being added, on private, state and federal lands, it is important to conduct surveys with great frequency.

The estimates of disturbance used here should be verified. Furthermore, rates of sedimentation, introduced under the different road classes should be researched and added to this characterization. Also, the overall sphere of influence should be further explored. To what degree do roads influence communities beyond the area of physical disturbance?

Roads and Trails (Module 4E)- Bibliography

Bibliography

U.S.G.S., 2004a – Reference to USGS DLG roads data.

Timber and Wildlife Treatments (Module 4B)

Management of Forest and Woodland Ecosystems (Module 4B)

Timber Management

The objectives for timber management on NFS lands include timber production[1], sustaining healthy forest conditions, and creating forest conditions that benefit or are conducive to management of other resource values such as wildlife habitat, recreation, aesthetics, water yield, and livestock grazing. There are several components to timber management that the Forest Service considers in its planning process. They include: determination of capability and suitability of NFS lands for timber production, the type of silvicultural systems[2] that can be used, and the amount of timber that can be harvested in a sustainable manner.

Capability

Determining which areas are capable of producing commercial timber is done by evaluating physical, biological and administrative limitations of an area. Table 7-7 lists the evaluation steps taken to identify areas capable of producing commercial timber in this process and the preliminary results for the San Juan Basin GA. Tentatively suitable for timber production is another term used to describe areas capable of producing commercial timber. The final determination of areas suitable for timber production is discussed in the following Suitability section.

[1] Timber production is the purposeful growing, tending, harvesting, and regeneration of regulated crops of trees to be cut into logs, bolts, or other round sections for industrial or consumer use. FSH 2409.13 .05.26

[2] Silvicultural system - A combination of interrelated actions whereby forests are tended, harvested, and re-established in order to produce a distinctive form and character. Systems are classified as even-aged and uneven-aged. FSH 2409.26, R2 Amendment 2409.26-96-8

Table 7-7 Capable (Tentatively Suitable) Timber Determination.
Capable (Tentatively Suitable) Timber Determination for the San Juan National Forest 1992 amendment.

Table 7-7 Capable (Tentatively Suitable) Timber Determination for the San Juan National Forest 1992 amendment.

Questions to Answer	Classification	Acres* Withdrawn	Acres* Remaining
Is it National Forest System Land?	National Forest System Land		**1,867,961**
Has the area been designated as unavailable for timber harvest?		300,234	1,567,727
Is the land forested?		356,818	1,210,909
	Forested Land		
Is the area capable of producing commercial tree species?		172,450	1,038,459
Is there a potential for irreversible soil or watershed damage if harvest occurs?		0	
Are there other resource management concerns that would limit timber production?		121,219	917,240
	Capable (Tentatively Suitable)		**917,240**

Suitability

Suitability determinations are a further refinement of forested lands found to be capable of producing commercial timber. These determinations are based more on social and economic considerations. Capable (tentatively suitable) timberland within management theme areas that emphasize resources other than timber (i.e. Theme 3) will not be considered as part of the suitable timber base. Capable timberland within management theme areas that emphasize resource management (i.e. Theme 5) will be considered suitable for timber production. The final decision on which areas are suitable for timber production will be made in the revised Forest Plan.

A similar process was used to determine suitable timberland for the 1992 Amended Forest Plan. Table 7-8 lists the existing suitable timberland for the entire San Juan National Forest, as well as the BLM lands included. Figure 7-12 shows existing acres of suitable timberland in the 1992 Amended Forest Plan.

Table 7- 8 Existing Acres of Suitable Timberland in 1992 Amended Forest Plan

Category	Acres	Percentage of Total San Juan NF.
San Juan National Forest Suitable Timber	375,000	20%
Suitable Conifer Timber	285,784	15%
Suitable Aspen Timber	89,216	5%
Suitable BLM Conifer Timber	13,000	

Table 7-8 Existing Acres of Suitable Timberland in 1992 Amended Forest Plan

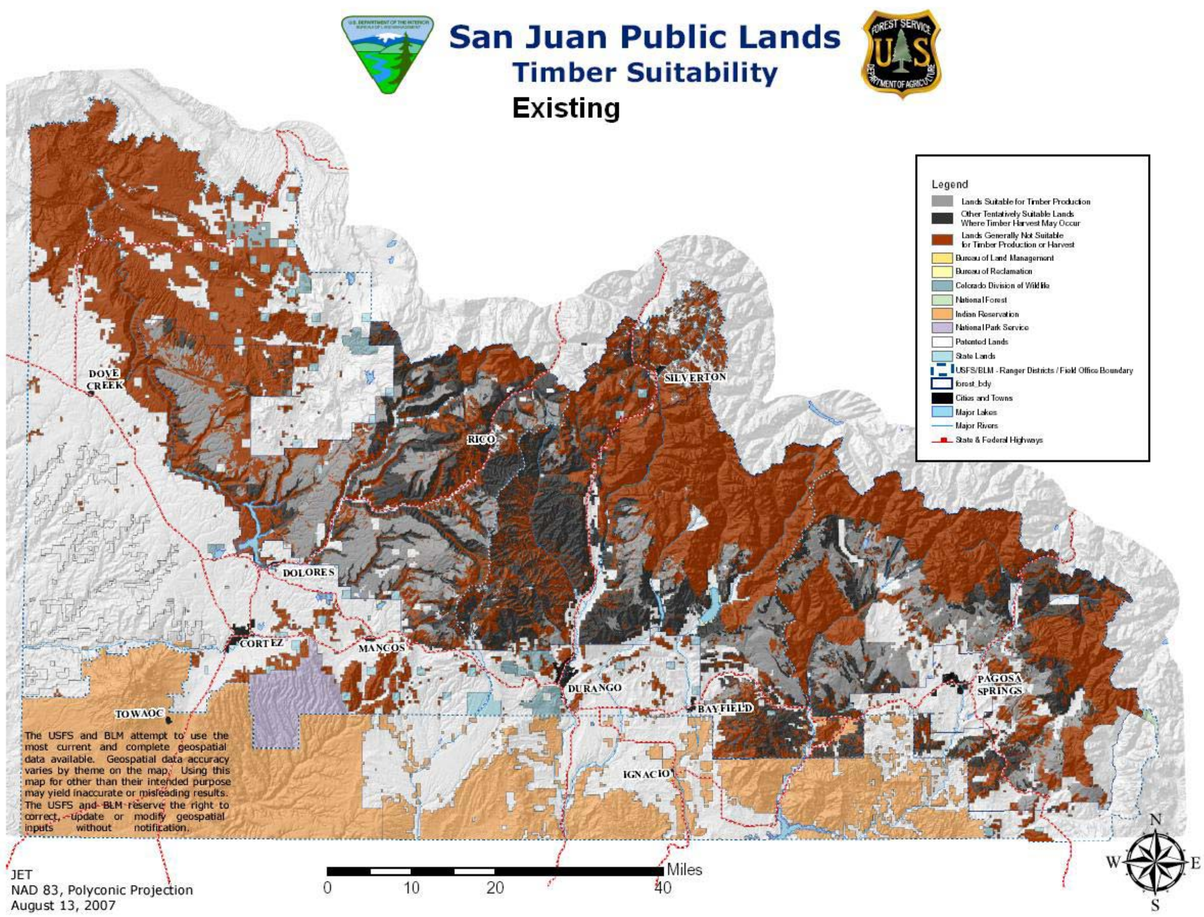


Figure 7-12 Existing suitable timberland on the San Juan NF

Timber Harvest Activity

Timber harvest activities began in the San Juan Basin in the early 1880's, approximately 20 years before the National Forest was established (1905). The Ponderosa pine forest type was most affected by these early, multiple-entry logging activities. Since then, harvest activities have occurred in all commercial forest types. The methods used to harvest trees to manage stands under a specific silvicultural system (even-aged and uneven-aged) have varied over time. Appendix A includes descriptions of the different methods that have been used on this Forest. Assessments- Appendix-A. Silvicultural Methods Descriptions

This appendix also lists the tree species that each method can be used on in the existing Forest Plan. Timber harvest activities over the past 50 years are summarized below (information from RMACT database). Figure 7-13 shows the acres harvested by silvicultural method. Table 7-9 displays the same data as acres harvested in the different cover types. Figure 7-14 shows where these harvest activities have occurred on the San Juan National Forest.

Figure 7-13 Timber Harvest by Silvicultural Method (1955-2004), San Juan NF

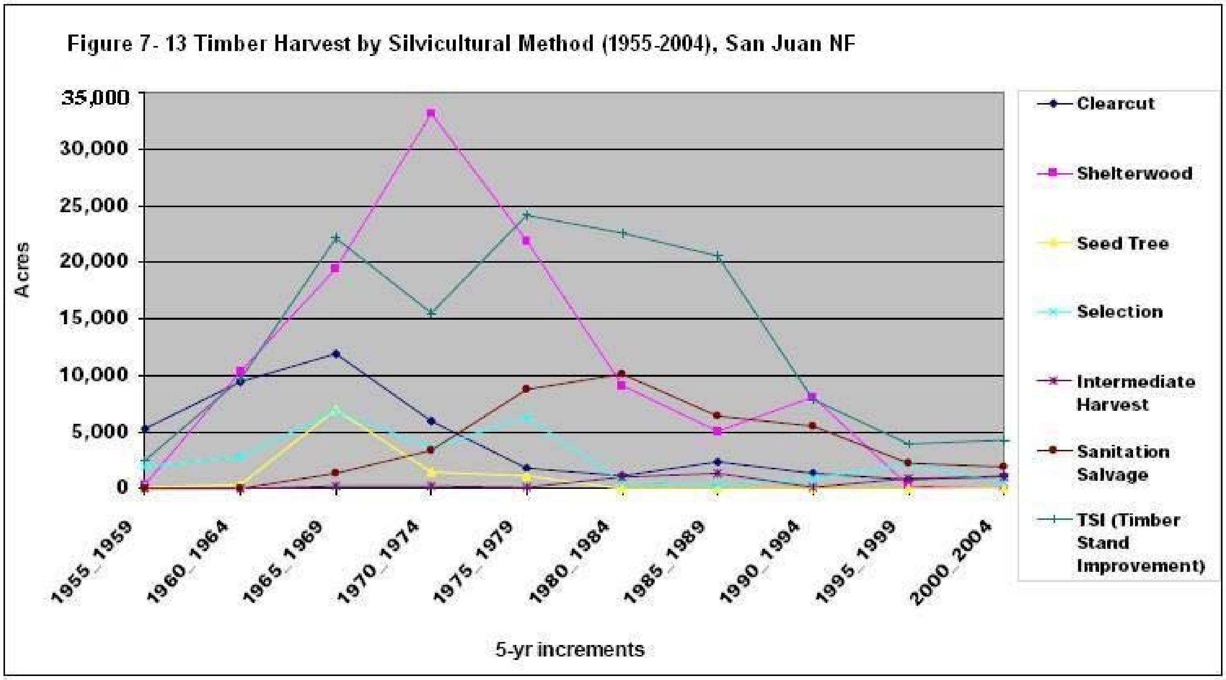


Table 7-9 Acres* harvested by Silvicultural Method on the San Juan NF, 1955 through 2004

Table 7- 9 Acres* harvested by Silvicultural Method on the San Juan NF, 1955 through 2004

ACRES BY 5 YEAR TIME PERIOD

HARVEST	1955_ 1959	1960_ 1964	1965_ 1969	1970_ 1974	1975_ 1979	1980_ 1984	1985_ 1989	1990_ 1994	1995_ 1999	2000_ 2004	SubTotal
Clearcut	5,333	9,413	11,920	5,940	1,794	1,142	2,404	1,397	741	1,179	41,263
Shelterwood	385	10,327	19,447	33,164	22,001	9,134	5,097	8,118	261	0	107,934
Seed Tree	167	392	7,030	1,516	1,073	0	0	32	0	0	10,210
Selection	1,869	2,791	6,821	3,647	6,255	612	373	953	2,175	571	26,067
Intermediate Harvest	0	0	231	200	90	1,013	1,360	130	908	969	4,901
Sanitation Salvage	0	0	1,406	3,424	8,790	10,125	6,407	5,464	2,235	1,891	39,742
TSI	2,513	9,694	22,150	15,512	24,141	22,617	20,603	7,881	3,888	4,247	133,246
SubTotal	10,267	32,617	69,005	63,403	64,144	44,643	36,244	23,975	10,208	8,857	363,363

Figure 7 -14 Timber Harvest Activity on the San Juan NF, 1955-2004

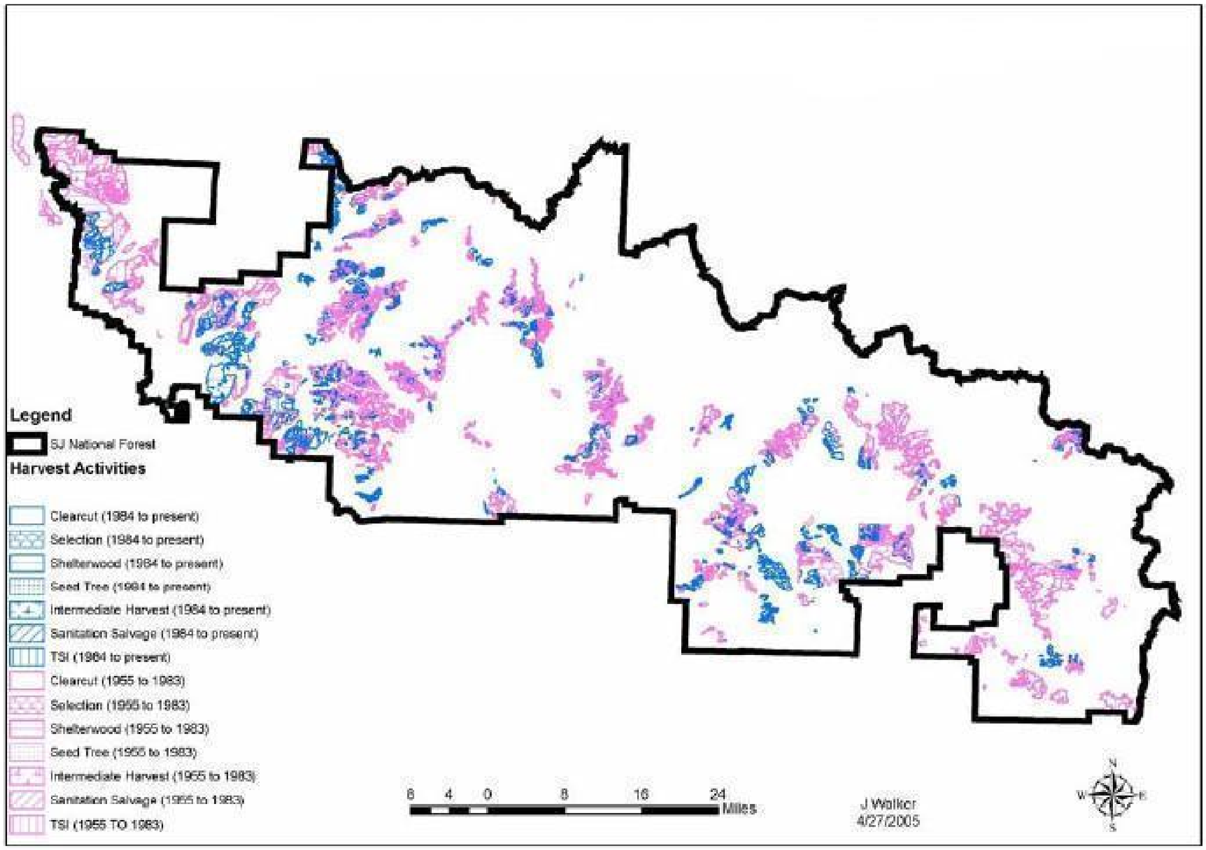


Figure 7-14 Timber Harvest Activity on the San Juan NF, 1955-2004

Between 1955 through 2004, approximately 230,117 acres have been harvested along with another 133,246 acres of TSI (timber stand improvement). This equates to nineteen percent (19%) including all treatments, and (12%) if TSI is excluded. These figures are for the San Juan National Forest. Harvest figures were not available for the rest of the geographic area.

The even-aged silvilcultural system of shelterwood harvests have been prescribed for the largest areas on the San Juan National Forest, mostly for spruce/fir and ponderosa pine cover types even aged silviculture was common in the past, but is seldom used today with the exception of aspen which is nearly always clearcut and regenerated by root suckering (coppice) . Uneven-aged silvicultural systems including group selection and individual tree selection are well suited to the spruce-fir and ponderosa pine stand conditions found in the San Juan Basin. These silvicultural treatments have been applied since the mid-1990s and are the most common today.

Timber stand improvement (TSI) is an intermediate treatment made to improve the composition, structure, condition, health and growth of even- or uneven-aged stands. This treatment may include thinning, release, cleaning, weeding and liberation. The use of this treatment in the San Juan Basin peaked in the 1980s and continues today, primarily for fuels reduction and restoration treatments in the dry mixed conifer and ponderosa pine cover types.

Sanitation and salvage cutting has occurred in ponderosa pine, mixed conifer and spruce-fir. In ponderosa pine these methods have been used to treat dwarf mistletoe infestations and recover sound mortality, with harvests occurring in the 1980's through the early 1990's and since 2000. Sanitation and salvage in spruce-fir was most prevalent in the late 1970's to treat spruce beetle mortality, but also continues today generally on a smaller scale to recover pockets of sound mortality and to reduce the spread of insects and disease.

Clearcutting was used to harvest spruce/fir during the 1960's and 1970's in the San Juan Basin . This harvest method was discontinued for spruce/fir in the late 1970's. Clearcutting is considered the optimum silvicultural treatment in aspen (coppice) with most of the activity occurring since the mid-1940's when aspen harvest for the Mancos match plant, (currently Western Excelsior) began.

While most harvest activity has occurred on suitable timberland, it is important to note that harvests have also occurred in areas that were not identified as suitable for timber production. This happened when the purpose for the harvest was to achieve some goal other than timber production, such as wildlife habitat improvement, or salvage and sanitation to remove fuels or other hazardous conditions.

Regeneration Success

Timber regeneration needs on the San Juan Basin GA have resulted from timber harvests wildfire and insect or disease caused mortality. Regeneration can occur through natural reseeding and/or suckering (as in aspen), or from artificial methods such as hand seeding or planting. Different methods have been used on the San Juan Basin for different reasons. These differences pertain to tree species, silvicultural system (if the need is generated by harvesting), or restoration (if the need is generated by natural causes).

On suitable timber lands, areas must be adequately stocked (have a minimum number of live trees per acre) within five years following a final regeneration harvest. Final regeneration harvests include: clearcuts, shelterwood removal cut, seed tree removal cut, or a selection harvest. Reforestation following natural disturbances such as wildfire or insect and disease is not required to meet the five year restocking timeframe. Areas that are not regenerated within the five year period are tracked in a reforestation backlog, and planting/seeding efforts are often continued until regeneration is accomplished.

If natural regeneration is inadequate, it may be supplemented with hand seeding or planting. Regeneration/survival surveys are normally done one, three and five years after treatment. Regeneration standards (the required number of live seedlings/saplings per acre) vary by species and site productivity.

Past regeneration failures are attributed to high elevation spruce fir harvests of the 1960s, where clearcutting was the silvicultural method used. This method, as stated, is no longer used on spruce-fir sites. Reforestation challenges resulted from this practice. Many of the initial planting efforts failed because there was no protection for planted seedlings on these sites. Natural regeneration has been very successful on aspen, and true fir sites, and all cover types where selection harvest methods have been used. Planting has been required on many spruce/fir and ponderosa pine sites where even aged silviculture has been used and has been successful about 75% of the time. Regeneration survey data is also tracked in the RMACT database. Table 7-10 summarizes the regeneration success for Englemann spruce, lodgepole pine, aspen, true fir, and ponderosa pine on the San Juan Basin GA from 1983 to present that is currently in the RMACT database.

Table 7-10 San Juan National Forest Regeneration Success by Tree Species, 1983-2004

Table 7- 10 San Juan National Forest Regeneration Success by Tree Species, 1983-2004

Tree Species	Natural Regeneration Certified as Stocked (3)	Planting Success (4)	Seeding Success (5)	Regen Surveys in Progress (6)
White Fir	100%	--	--	0%
SubAlpine Fir	82%	18%	--	0%
Aspen (1)	98%	--	--	2%
Lodgepole Pine (2)	8%	92%	--	0%
Engelmann Spruce	25%	72%	--	3%
Ponderosa Pine	24%	73%	--	3%
Douglas-Fir	9%	91%	--	0%
Unknown	28%	12%	--	60%

(1) 'Aspen' regeneration has been certified in areas where aspen was harvested (73%), and in areas where Spruce-Fir was harvested (25%).

(2) 'Lodgepole Pine' is not native to the San Juan National Forest but was seeded in 1911, and then planted from 1940-1983. Currently, it is also naturally regenerating from the planted seed source following the Bear Creek Fire
.

(3) 'Natural Regeneration Certified as Stocked' by tree species was calculated based on acres of natural regeneration certification compared to total regeneration harvests for that species.

(4) 'Planting Success' by tree species was calculated based on acres of certified planting compared to total regeneration harvests for that species.

(5) 'Seeding Success' for this time period is not applicable since seeding of tree species only occurred between 1911 and 1971.

(6) 'Regen Surveys in Progress' represents those areas where regen surveys are currently planned, and regeneration success is being monitored. The tree species regenerating is 'Unknown' at this time, until the survey is completed.

Fuelwood Harvest

One of the goals of the current Forest Plan is to provide a supply of fuelwood to local residents. Fuelwood harvest is accomplished both through commercial and personal use permits. Mostly dead timber is harvested with limited amounts of green wood also being provided in specified areas.

Fuelwood areas vary by year, and acres affected are not tracked. The volume of fuelwood harvested is monitored, based on the number of permits sold. Volumes are measured in board feet (a board one inch thick X one foot wide X one foot long). Total volumes are summarized in units of 1000 board feet, often referred to as MBF. Figure 7-15 displays the trend in fuelwood harvest for the past 20 years.

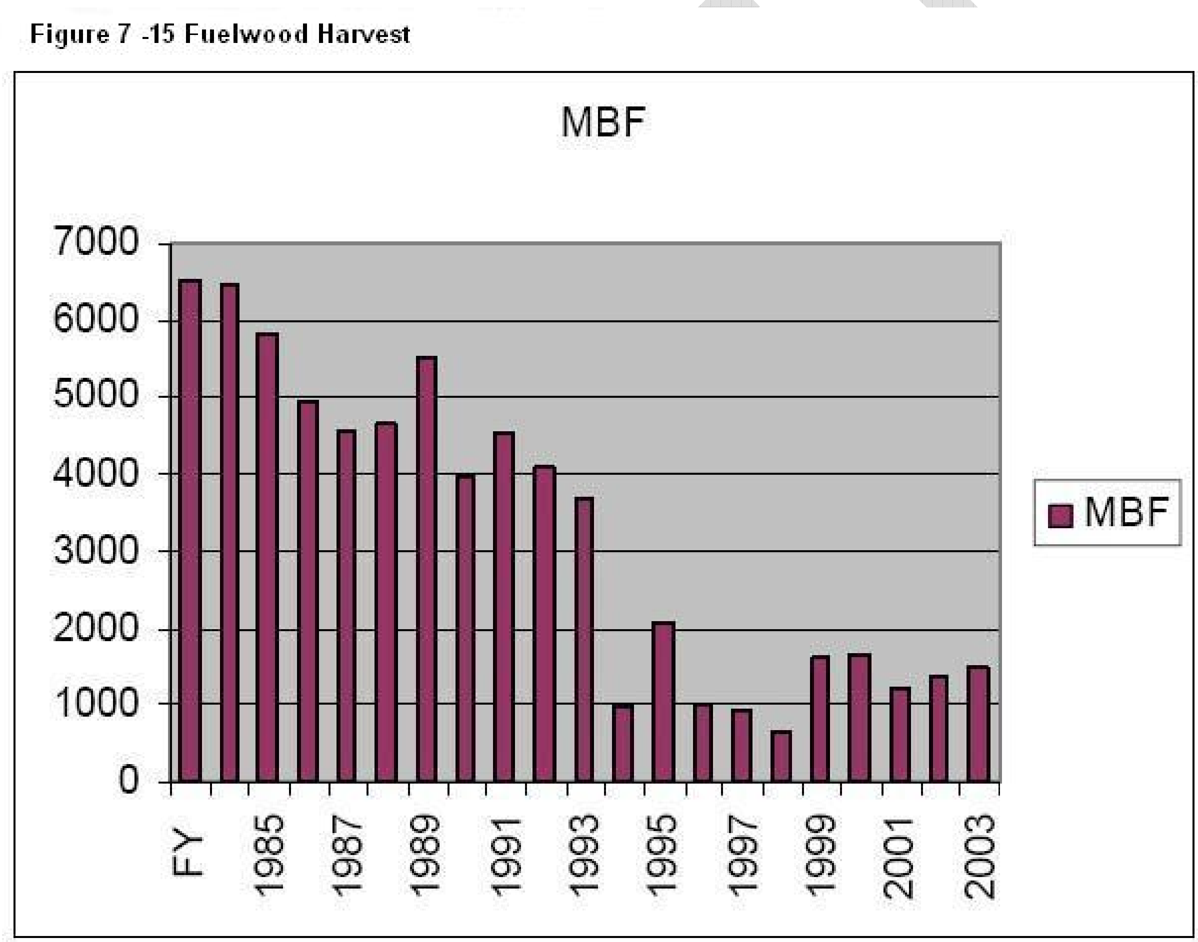


Figure 7-15 Fuelwood Harvest

Allowable Sale Quantity

The Forest Service is required[1] to determine the average annual allowable sale quantity (ASQ) in Forest Plans. This is the quantity (volume) of timber that may be sold from the suitable timberland identified by the Forest Plan. This annual harvest level must be sustainable over the long-term. Table 7-11 shows the annual ASQ determined for the San Juan National Forest in both the 1983 Forest Plan and the 1992 Amended Forest Plan. The 92 Amendment estimated that 25% or 6 MMBF of the ASQ would be aspen.

Table 7-11 San Juan National Forest, Allowable Sale Quantity.

Table 7 -11 San Juan National Forest, Allowable Sale Quantity.

Plan	Total Volume MMBF*
1983 Forest Plan	41
1992 Amended Forest Plan	24

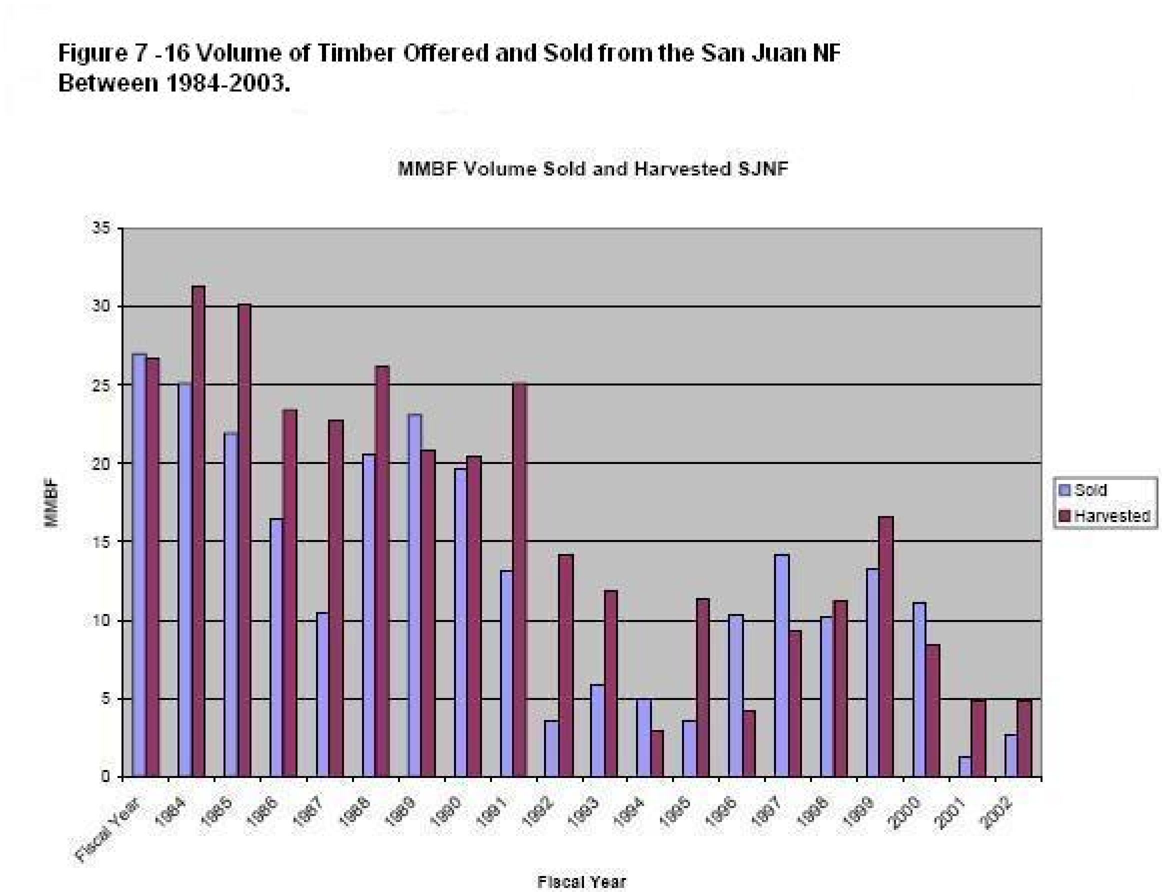
* Volume in million board feet (MMBF). 1 MMBF = 1000 MBF

Volumes Offered and Sold

As mentioned previously, timber may be harvested from both areas identified as suitable timberland, and areas that are not suitable timberland. The purposes for timber harvest on these two types of areas are different. Timber production is the objective on suitable timberlands. Timber is harvested to achieve another objective, such as habitat improvement, or hazard reduction, on areas that are not suitable timberlands. Only volumes harvested off suitable timberland are considered as part of the ASQ.

Figure 7-16 displays the volume of timber offered and sold from the San Juan National Forest between 1984 and 2003.

Figure 7-16 Volume of Timber Offered and Sold from the San Juan NF Between 1984-2003.



A timber sale sold in one year may have volume harvested for several years. The Forest Service tracks harvested timber volumes by species, and type of product or component. (Sawtimber is a log greater than 8 inches in diameter; Products other than logs [POL] include posts and poles with diameters less than 8 inches, and all aspen products.)

Figure 7-16 also displays the volume harvested off the San Juan National Forest between 1984 and 2003.

Key Findings

- Approximately 49 percent of the NFS lands on the San Juan NF are capable of growing commercial timber.

- Currently (under the 1992 Amended Forest Plan) 20 percent of NFS lands on the San Juan NF are identified as suitable timberland.
- Since 1955, less than nineteen percent of the San Juan Basin GA has been affected by timber harvest activities. This translates into less than thirty percent of the forested cover on the San Juan Basin GA.

- The most common harvest methods have been shelterwood, mostly in spruce-fir and ponderosa pine, followed by clearcut in spruce/fir in the past, and currently in aspen (coppice), then timber stand improvement, and sanitation/salvage. The most common treatment currently is selection and intermediate treatments for fuels or restoration purposes.
- Most fuelwood is harvested off the San Juan Basin Geographic Area through personal use permits.
- The ASQ for the San Juan was never fully offered and sold or harvested any year between 1984 and 2004.
- Past regeneration failures are attributed to high elevation spruce fir harvests of the 1960s, where clearcutting was the silvicultural method used.
- Natural regeneration has been very successful on aspen, and true fir sites (90-100%), and all cover types where selection harvest methods have been used. Planting has been required on many spruce/fir and ponderosa pine sites where even aged silviculture has been used and has been successful about 75% of the time.

Trends

- The trend in total acres harvested shows a peak in the late 1980s, with a steady decline over the past 20 years.
- The trend in volume offered and sold over the past 20 years shows peaks in 1983, 1988 and 1997. The trend in total volume harvested over the past 20 years show a peak in 1984 and 1991, and 1999. Annual harvest activities show less fluctuation between years than sale offerings.
- Trends in fuelwood demand show a peak in 1985. Demand has leveled out at approximately 1000 MBF for the last ten years.

Management Implications

- Final timber suitability determinations will be based on the management theme designations ultimately decided on in the Forest Plan revision, as well as other considerations, such as stand size, distance from existing roads, and terrain factors like slope. A final decision on the Roadless Area Conservation Rule will also influence where timber resources could be managed as part of the suitable timber base.
- The legacy of past timber harvest have resulted in current vegetation conditions that must be considered in planning for the future. If original silvicultural prescriptions are going to be followed, many areas are due for second shelterwood entries, especially in spruce-fir. Areas that have been treated more recently will not be available for subsequent harvest activities during the planning period covered by the Revised Forest Plan (approximately 15 years).
- The majority of forest cover types are in mature and dense stand conditions (see Vegetation section) particularly the ponderosa pine and dry mixed conifer type. These stand conditions are vulnerable to future insect and/or disease attack . Timber management activities can be used to alter stand

conditions to reduce ongoing insect and disease activity and to reduce the risk for future outbreaks.
- Timber stand improvement activities may also be used to reduce stand density and ladder fuel accumulations. These types of treatments may be used prior to reintroducing fire through prescribed fires or wildland fire use (natural ignitions) into forest cover types that historically had frequent fires.
- Timber management activities can be designed to improve wildlife habitat.
- Both timber demand and timber industry capacity has decreased.   These conditions may limit future opportunities to obtain desired conditions in forested cover types through any type of vegetation treatments that harvest wood products.

[1] 1982 Planning Rule (36 CFR §219.16)

Livestock Management (Module 4C)

Page intentionally blank.

Invasive Plant Species (Module 4D)

Page intentionally blank.

Roads and Trails (Module 4E)

Page intentionally blank.

Recreation and Exurban development (Module 4F)

Developed Recreation – Key Findings and Introduction[5]

Key Findings

- The CLC Subregion is located at the margin of the Colorado Plateau and Southern Rocky Mtns. This very scenic region is a "recreational hotspot".
- The CLC subregion is at the crossroads to many important regional attractions including the Grand Canyon, Mesa Verde, Canyonlands and Arches National Parks. The subregion is geographically central to important and growing population centers including Salt Lake City, Denver, Albuquerque, Sante Fe and Phoenix.
- More than half of the visitors to National Parks adjacent to the subregion come from California, Colorado, Utah and a few eastern states. Park visitors also utilize Forest recreation opportunities.
- On Forest lands, about half the users are from local communities, especially, Durango, Montrose, Grand Junction and Gunnison. This is important because levels of visitation will likely increase significantly with continued robust growth for these communities and others in the subregion.
- Recently, while visitation to National Parks seems to be leveling off or dropping, visitation to the National Forests and BLM lands in the subregion are increasing.
- As use of National Forests and BLM lands increases and user quality of experience may eventually drop as availability of recreational resources diminishes. At the same time, current and future increases in recreational use of public lands will lead to increased disturbance and potential conflicts and competition between recreation and other management programs.
- Using an estimated average of 101.6 acres per site, about 33,255 to 34,546 acres are currently directly disturbed and/or influenced by developed recreation.
- About two thirds of sites are found from 8,000 to 12,000 feet in upland vegetation communities.
- Vegetation communities most influenced by developed recreation disturbance are found in lands largely managed by the U.S. Forest Service.

[5] David Plume March 29, 2005
ChapterIV_DevelopedRecSubRegion032905.doc

Introduction

The striking and challenging geography, scenery and recreational opportunities of the Colorado Plateau and Southern Rocky Mountains attract growing numbers of visitors to public, state and private lands in the CLC subregion. Developed recreation opportunities include camping, and picnicking in developed sites, skiing at developed resorts along with sightseeing and visits to sites of cultural and historical importance.

Recreational opportunities in and adjacent to the subregion draw visitors from both local and distant communities. Scenery, attractions and recreational opportunities in the subregion are of such a quality as to provide destinations attracting significant numbers of visitors. At the same time, the subregion is at an important crossroads for recreational visitors to well known National Parks in the surrounding five-state region (Fig. 7-17). These Parks include the Grand Canyon, Arches, Canyonlands, Bridges, Dinosaur, Rocky Mountain and the Sand Dunes. Furthermore, the Mesa Verde National Park is in the subregion itself. Very often, these "travel-through" visitors visiting these Parks also take advantage of recreational opportunities in the subregion.

The Colorado Plateau and Rocky Mountains attract local, national and international visitors. As of 2003, 50% percent of visitors to Arches National Park were from six states. Of these, 33% percent were from California (16%), Colorado (9%) and Utah (8%) (Meldrum, et al. 2004). The remaining visitors are from three eastern states including Illinois (7%), New York (5%) and Virginia (4%). Similarly, over 50% of surveyed visitors to Canyonlands National Park are from three western states. These states include Colorado (29%), California (15%) and Utah (11%) (Canyonlands, 2005).

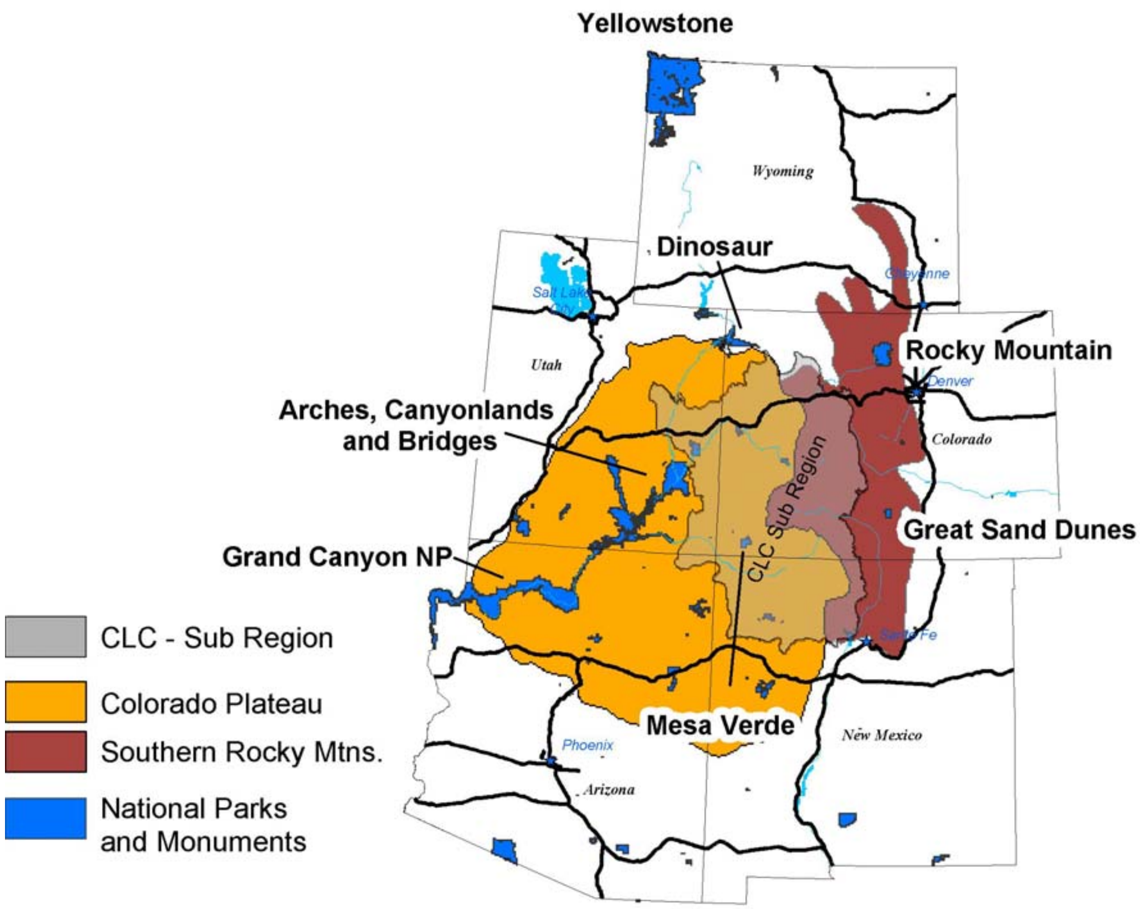


Figure 7-17 The CLC Sub Region straddles both the Colorado Plateau and Southern Rocky Mountains physiographic regions.

And the subregion falls at the cross-roads for travelers from large regional metropolitan centers including Salt Lake City, Phoenix, Albuquerque, Santa Fe and Denver.

Importantly, studies show that over recent years there have indeed been significant increases in public participation in recreation activities nationally. This trend includes a wide spectrum of recreation opportunities and settings. The most significant increases in recreation activities include snow skiing, canoeing/kayaking, cycling, camping, sailing, swimming, fishing, horseback riding and hunting. With the exception of hunting, these activities have shown significant increases since 1995 (Cordell and Super, 2000). World class opportunities for these activities are found in the subregion.

This trend of increasing public interest may be combined with changes in use patterns on National Parks to show that both BLM and National Forests in the subregion are of key importance to developed recreational activities in the subregion. As a result, levels of recreational activity, demand and influences on terrestrial and aquatic ecosystems are also increasing in character, degree and extent.

For example, visitation to Mesa Verde, Arches and Canyonlands National Parks steadily increased to maximums in the early 1990's. Then during the decade of the 1990's visitation leveled out and slumped (Fig. 7-18). While the slump in visitation may be at least partly attributed to a down turn in the national economy, or as a response to recent drought and fire, the overall trend may also attributed to a diminishing quality of user experience in these National Parks. This diminishing quality of experience may be an indication that National Parks are at or near their capacity to provide positive recreational experiences.

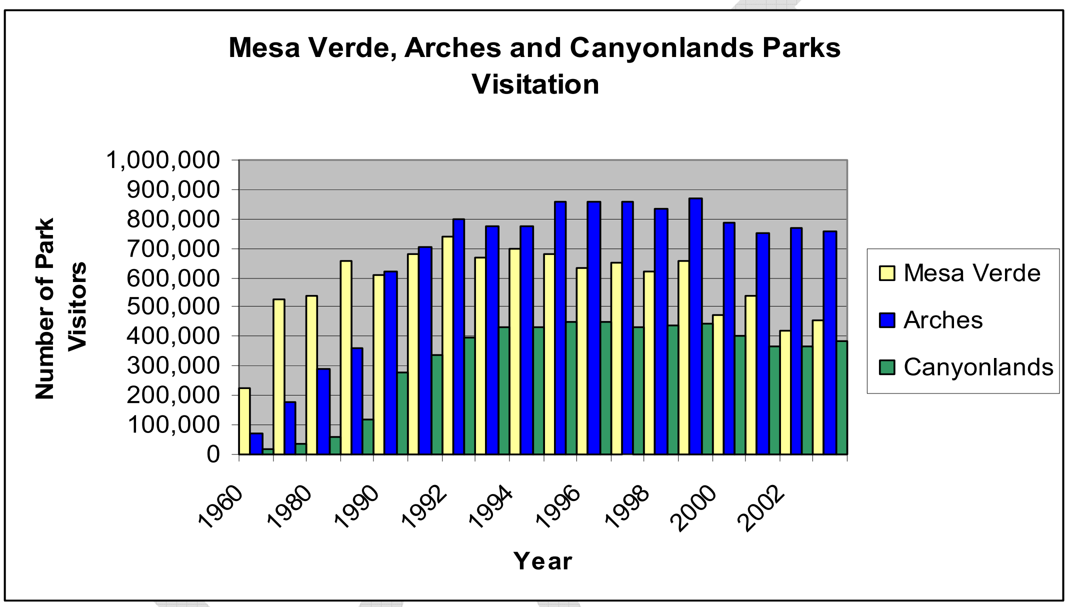


Figure 7-18 Visitation trends for the Mesa Verde, Arches and Canyonlands National Parks.

These trends show and overall trend of diminishing visitation in these three parks (NPS, 2005).

So, as overall demand for recreational resources continues to increase, and quality of experience diminishes in National Parks, recreational users are seeking out new opportunities elsewhere. From 1986 to 1996 visitation levels for the National Forests, BLM lands, Federal lands managed by the Army Corps of Engineers and the Bureau of Reclamation (BOR) all increased while visits to National Parks dropped (Fig. 7-19).

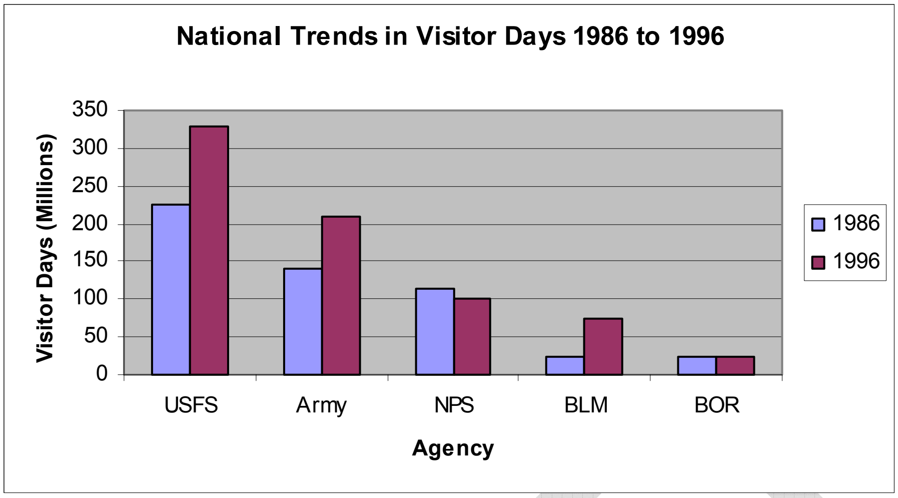


Figure 7-19 National trends in visitation.
National trends in visitation suggest a shift away from National Parks to National Forests, Army Corps of Engineers lands and BLM lands. (Adapted after Cordell and Super, 2000).

Large areas of these public lands, especially lands managed by the U.S. Forest Service (USFS) and Bureau of Land Management (BLM), are available in and adjacent to the CLC subregion (Table 7-12 and Fig. 7-20). Upland areas along the eastern margin of the subregion are principally in Forest Service jurisdiction (San Juan and GMUG) while the desert basins and foothills include significant tracts of BLM land. Both the Forest Service and BLM provide developed recreation sites on these lands.

Table 7-12 Land Ownership/Jurisdiction by agency in the Subregion

Ownership/ Jurisdiction	Acres	Hectares	Pct	Sum Pct
USFS	10,515,806	4,255,596	30.8%	30.8%
BLM	9,123,884	3,692,304	26.7%	57.6%
Private	7,338,073	2,969,613	21.5%	79.1%
Tribal	5,646,754	2,285,160	16.6%	95.6%
State	1,175,076	475,536	3.4%	99.1%
NPS	229,352	92,815	0.7%	99.8%
DOD	55,547	22,479	0.2%	99.9%
BOR	25,807	10,444	0.1%	100.0%
FWS	2,981	1,206		
Water	248	100		
Other	32	13		
	34,113,560	13,805,267	100.0%	

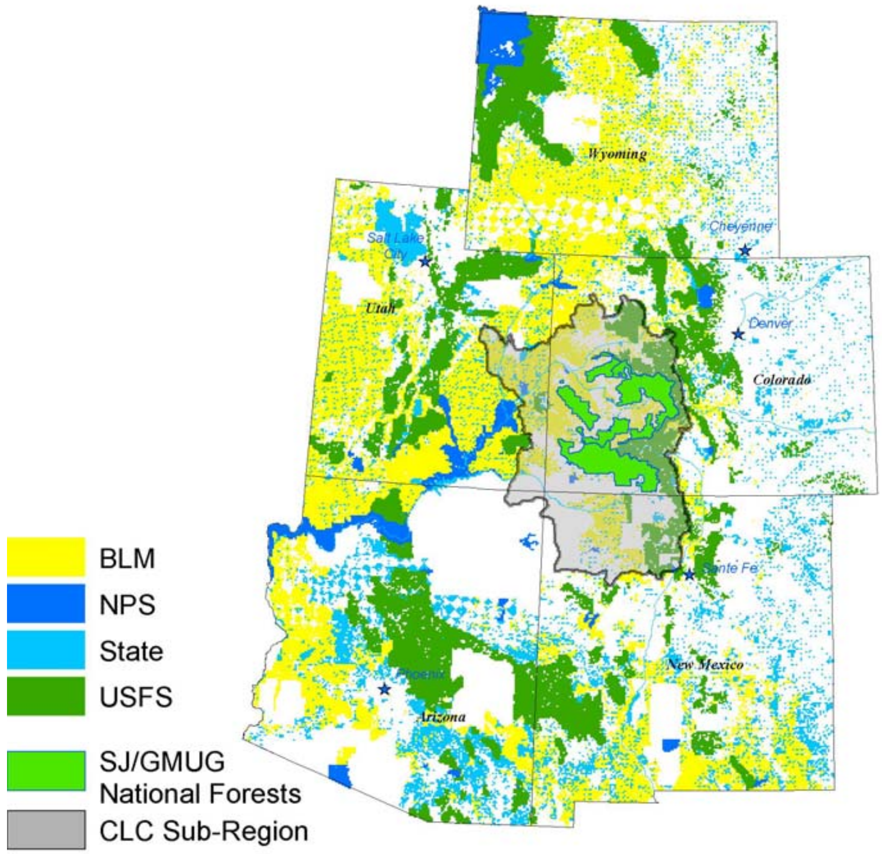


Figure 7-20 Significantly large tracts of public lands in and around the CLC subregion. Significantly large tracts of public lands in and around the CLC subregion provide important recreational opportunities both locally and nationally.

Local visitation is important too. U.S. Forest Service surveys for the San Juan and GMUG Forests show that about half the number of Forest visitors report home zip code locations that are beyond 50 miles of the Forests (Kocis, et al., 2004 and USDA, 2001). About 51% percent of visitors to the San Juan Forest (Fig. 7-21) and 44% of visitors to the GMUG Forest (Fig. 7-22) are from communities within 50 miles. The data show that the principal source communities for Forest visits include the Colorado communities of Grand Junction, Montrose, Durango and Gunnison.

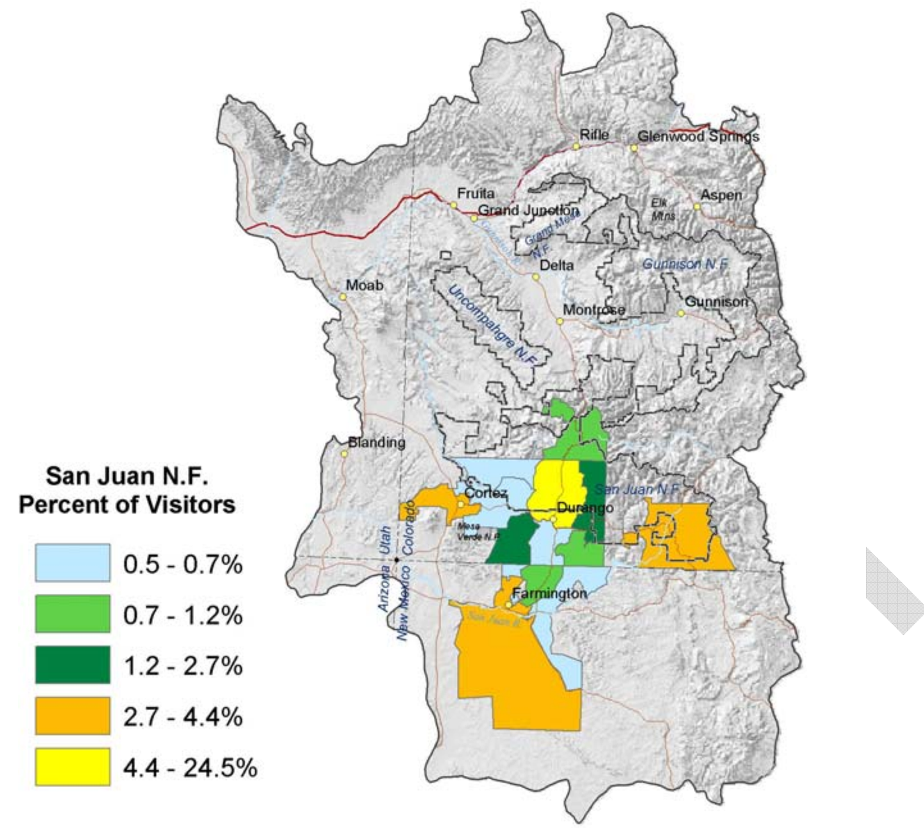


Figure 7-21 Home communities by zip code of local of visitors to the San Juan Forest.

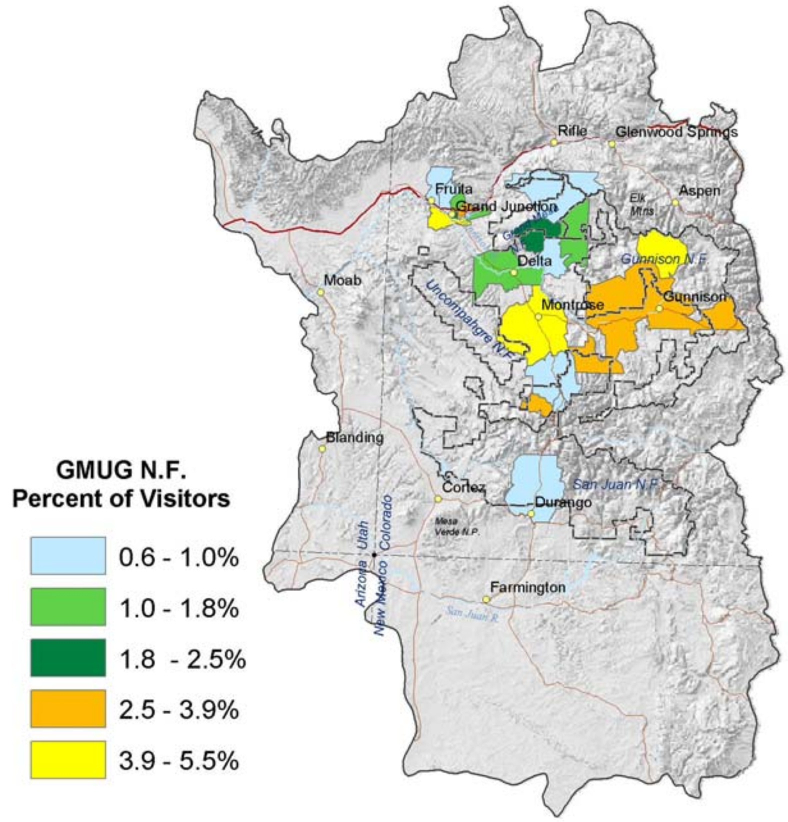


Figure 7-22 Home communities by zip code of local of visitors to the GMUG National Forest.

These western slope communities are expected to grow significantly over the coming years. By 2025 the population of Colorado is expected to grow by another 48% percent. Currently reported growth in the region for five years (1997 to 2001) averages just over 2% per year (Fig. 7-23). Much of this growth will take place in rural counties with access to public lands. (SCORP, 2003).

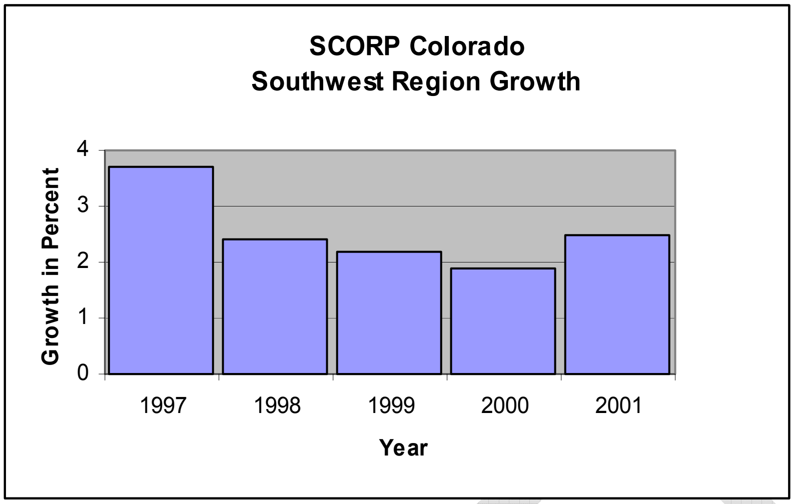


Figure 7-23 Graph showing percentage population growth in the SCORP southwest region.
Graph showing percentage population growth in the SCORP southwest region, including the CLC subregion subject lands. Adapted after SCORP, 2003

To conclude, we can expect the demand for developed recreation on Forests in the ecoregion to continue to grow. Growth is likely to continue along upward trends as interest in National Forests and BLM lands increases. The increases will come as local communities continue to grow and as people look beyond the National Parks for their recreational experiences.

CLC Subregion Analysis

In this analysis we have identified 340 developed recreation sites on public lands in the subregion (Fig.7-24). The precise number of sites is difficult to determine because not all Federal and State agencies have completed and published inventories to date. The problem is compounded by the fact that there is no universally accepted standard defining *developed recreation site* characteristics. Never the less, in this analysis we generally consider overnight campgrounds as developed recreation sites. These data provide a good estimate of disturbance and affected vegetation communities. Sites on private lands are not included.

Under this definition, developed recreation sites are comprised of familiar elements. They include campground features such as fire rings, parking areas, picnic tables, toilets and disturbed congregation and tent pitching areas. Some sites include day use areas and facilities for recreational vehicle parking and facilities. In surrounding areas, vegetation disturbance results from trampling, cutting and foraging for firewood.

The overall area of disturbance for a site is difficult to determine. No definite boundary exists for campgrounds and the resulting area of disturbance.

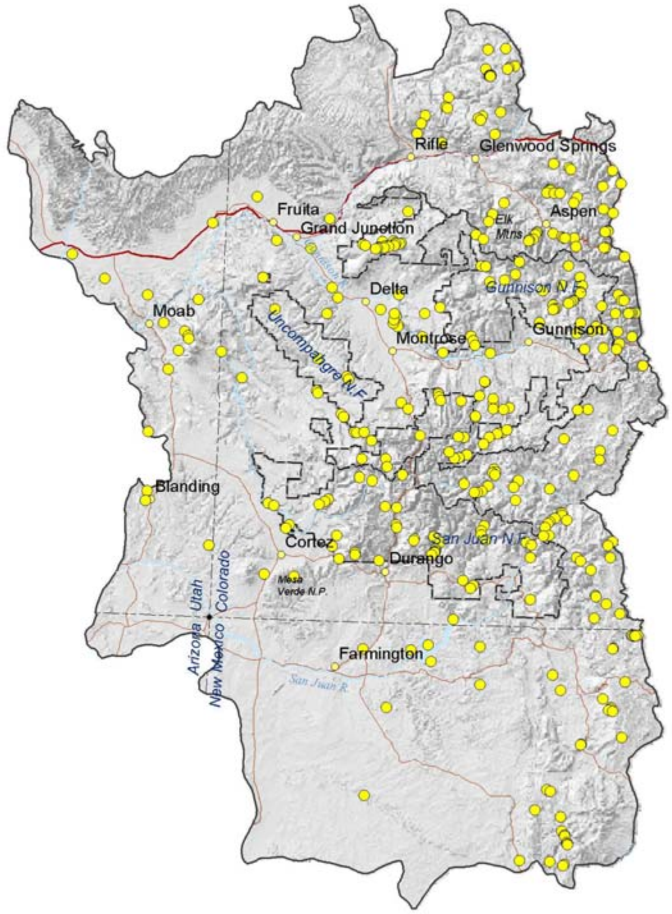


Figure 7-24 Developed recreation sites on public lands in the subregion.

National Forests in the Ecoregion

More than 75% percent of developed recreation sites in the subregion are found on National Forests (Table 7-13 and 7-14). An additional 13.5% percent of all sites are found on lands managed by the BLM. These two agencies combined cover about 58% percent of the land area in the subregion. The remaining sites are found on National Park Service lands and State lands. Bureau of Reclamation sites are included in the State sites because of co-management.

Table 7-13 Number of sites by agency.
State sites include Bureau of Reclamation sites in some cases. Almost 90% percent of sites are on either Forest or BLM lands.

Agency	Number of Sites	Pct
USFS	259	76.2%
BLM	46	13.5%
State	22	6.5%
NPS	11	3.2%
Other	2	0.6%
	340	100.0%

These sites are distributed among nine National Forests (Table 7-14). In the subregion, the GMUG Forest has the greatest number of sites. More than half of all sites are found in the GMUG, White River, Rio Grande and San Juan Forests.

Table 7-14 Number of sites by National Forest for the 9 Forests in the subregion.

National Forest	Number of Sites	Pct.	Sum Pct.
GMUG	71	20.9%	20.9%
White River	46	13.5%	34.4%
Rio Grande	41	12.1%	46.5%
San Juan	39	11.5%	57.9%
Pike-San Isabel	20	5.9%	63.8%
Carson	18	5.3%	69.1%
Santa Fe	14	4.1%	73.2%
Routt	5	1.5%	74.7%
Manti-La Sal	5	1.5%	76.2%
Subtotal:	259	76.2%	
Outside Forest Bounds	81	23.8%	23.8%
Total:	340	100.0%	

Developed Recreation Sites and GAP Vegetation Classes

Nearly two-thirds of all sites are found in elevations from 8,000 to 10,000 feet. More than 90% percent of all sites in the subregion are located in upland settings above 6,000 feet (Table 7-15). In general, the sites in upland settings are located on Forest Service lands while lower elevation sites are often associated with BLM and State lands. Sites in different settings correspond to variation climate, season of use and generally to vegetation community.

Table 7-15 Number of sites per elevation class.
1000 foot elevation interval.

Number of Sites	Elevation Feet	Pct. Of All Sites	Sum Pct.
2	Not Classified	0.59%	0.59%
8	4000 - 5000	2.35%	2.94%
15	5000 - 6000	4.41%	7.35%
34	6000 - 7000	10.00%	17.35%
54	7000 - 8000	15.88%	33.24%
78	8000 - 9000	22.94%	56.18%
109	9000 - 10000	32.06%	88.24%
36	10000 - 11000	10.59%	98.82%
4	11000 - 12000	1.18%	100.00%
340		100.00%	

In the subregion, developed recreation sites are found in sixteen different GAP vegetation communities (Fig. 7-25, Table 7-16). Nearly two-thirds of sites fall in upland settings comprised of either spruce-fir, aspen, mixed conifer, Ponderosa pine, Lodgepole pine or alpine vegetation (Table 7-17). The remaining nearly one-third of sites fall in foothills and desert vegetation communities comprised of Pinyon-juniper, sagebrush, mountain grassland, desert shrub, deciduous oak and desert grassland (Table 7-18).

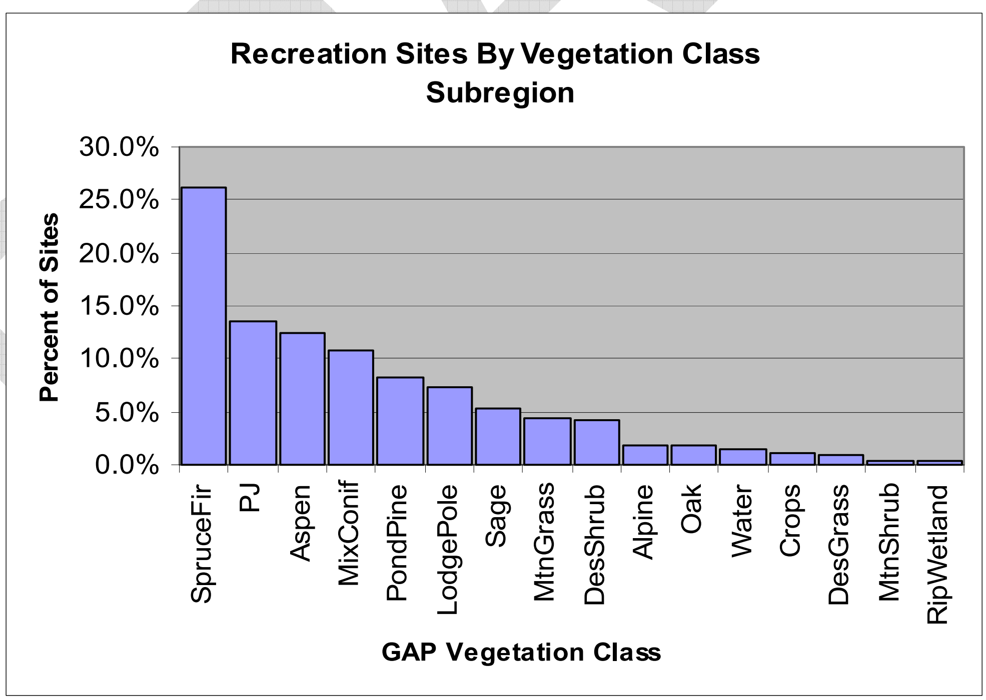


Figure 7-25 Percentage of all sites by GAP Vegetation Class.

Table 7-16 Summary table showing the number of developed recreation sites for all GAP vegetation classes.

GAP Vegetation Class	Number of Sites	Pct	Sum Pct
spruce - fir	89	26.2%	26.2%
pinyon -juniper	46	13.5%	39.7%
aspen	42	12.4%	52.1%
mixed conifer	37	10.9%	62.9%
ponderosa pine	28	8.2%	71.2%
Lodgepole pine	25	7.4%	78.5%
sagebrush	18	5.3%	83.8%
mountain grassland	15	4.4%	88.2%
desert shrub	14	4.1%	92.4%
alpine	6	1.8%	94.1%
deciduous oak	6	1.8%	95.9%
water	5	1.5%	97.4%
crops	4	1.2%	98.5%
desert grassland	3	0.9%	99.4%
mountain shrubland	1	0.3%	99.7%
woody riparian/wetland	1	0.3%	100.0%
	340	100.0%	

Table 7-17 Summary table showing the number of developed recreation sites by GAP vegetation class for upland/forested vegetation classes.

GAP Vegetation Class	Number of Sites	Pct. of All Sites
spruce - fir	89	26.2%
aspen	42	12.4%
mixed conifer	37	10.9%
ponderosa pine	28	8.2%
lodgepole pine	25	7.4%
alpine	6	1.8%
	227	66.8%

Table 7-18 Summary table showing the number of developed recreation sites by GAP vegetation class for foothills and desert vegetation classes.

GAP Vegetation Class	Number of Sites	Pct. Of All Sites
pinyon -juniper	46	13.5%
sagebrush	18	5.3%
mountain grassland	15	4.4%
desert shrub	14	4.1%
deciduous oak	6	1.8%
desert grassland	3	0.9%
	102	30.0%

Estimation of Disturbance

The total area of disturbance related to developed recreation is dependent upon average site. Average area has been estimated by examination of a subset of all sites in high resolution image data. This estimate is supported by reported recreation site area size in reports (e.g. Chelan, 2001 and Spangle, 2004).

Once an average area has been determined then simple estimates may be made. The most direct measure is generated by multiplication of average area times the number of sites. Here, that method is applied along with a slightly more sophisticated method using circular buffers around each site. The resulting buffer can then be overlain on GAP vegetation polygons to discover measures of plant communities most closely associated with developed recreation.

In this analysis the average site disturbance area is estimated at 101.6 acres. This area includes highly disturbed areas such as camp sites, eating and congregation areas, paths, parking and roads. Importantly, it also includes a zone of activity and foraging where vegetation communities are subject to some level of disturbance. The estimate of 101.6 acres results from averaging the area of 71 polygons for developed recreation sites on the San Juan and Rio Grande National Forests. Multiplying 101.6 acres times 340 sites yields an overall area influenced directly by activities at developed recreation sites of about 34,546 acres.

The actual area resulting from buffering yields a collection of areas whose area sum is about 1,300 acres less than the estimate of 34,546 acres obtained by simple multiplication. That is: using the buffer model we found a total area of disturbance of 33,255 acres. The difference between the two models comes from situations where sites closer than the buffer radius of 362 meters overlap reducing the total area.

Disturbance totals range from a maximum of 8,250 acres for the spruce-fir vegetation community down to a minimum of 31.5 acres for herbaceous/riparian wetland (Table 7-19). As a percentage of each GAP vegetation community, disturbance values range from a maximum proportion of 0.47% percent for lodgepole pine down to 0.02% for the desert grassland community.

Table 7-19 Summary table shows the sum of area by GAP vegetation class.
This summary table shows the sum of area by GAP vegetation class using the buffer model with 102 acre circular buffers with radii of 362 meters. The table is sorted by total disturbance area.

GAP Vegetation Class	Acres Sum in Site Buffers	% of Total Disturbance
spruce - fir	8,250.0	24.9%
aspen	4,383.9	13.2%
pinyon -juniper	4,356.6	13.1%
mixed conifer	3,430.3	10.3%
ponderosa pine	2,867.0	8.6%
lodgepole pine	2,169.9	6.5%
sagebrush	1,818.1	5.5%
mountain grassland	1,436.1	4.3%
desert shrub	1,211.5	3.7%
water	716.0	2.2%
deciduous oak	654.0	2.0%
alpine	583.3	1.8%
desert grassland	418.6	1.3%
crops	373.0	1.1%
mountain shrubland	253.6	0.8%
woody riparian/wetland	174.0	0.5%
barren	48.9	0.1%
herbaceous riparian/wetland	31.5	0.1%
Total Buffer Acres:	33,176.6	100.0%

Table 7-20 Percent of total GAP class area in the subregion.

GAP Vegetation Class	Buffer Acres	% of Subregion GAPClass
lodgepole pine	2,169.9	0.47%
mixed conifer	3,430.3	0.28%
mountain grassland	1,436.1	0.24%
spruce - fir	8,250.0	0.24%
aspen	4,383.9	0.19%
herbaceous riparian/wetland	31.5	0.18%
woody riparian/wetland	174.0	0.15%
ponderosa pine	2,867.0	0.14%
mountain shrubland	253.6	0.11%
sagebrush	1,818.1	0.06%
pinyon -juniper	4,356.6	0.06%
deciduous oak	654.0	0.04%
alpine	583.3	0.04%
crops	373.0	0.03%
desert shrub	1,211.5	0.02%
desert grassland	418.6	0.02%
water	716.0	1.12%
barren	48.9	0.02%
	33,176.6	

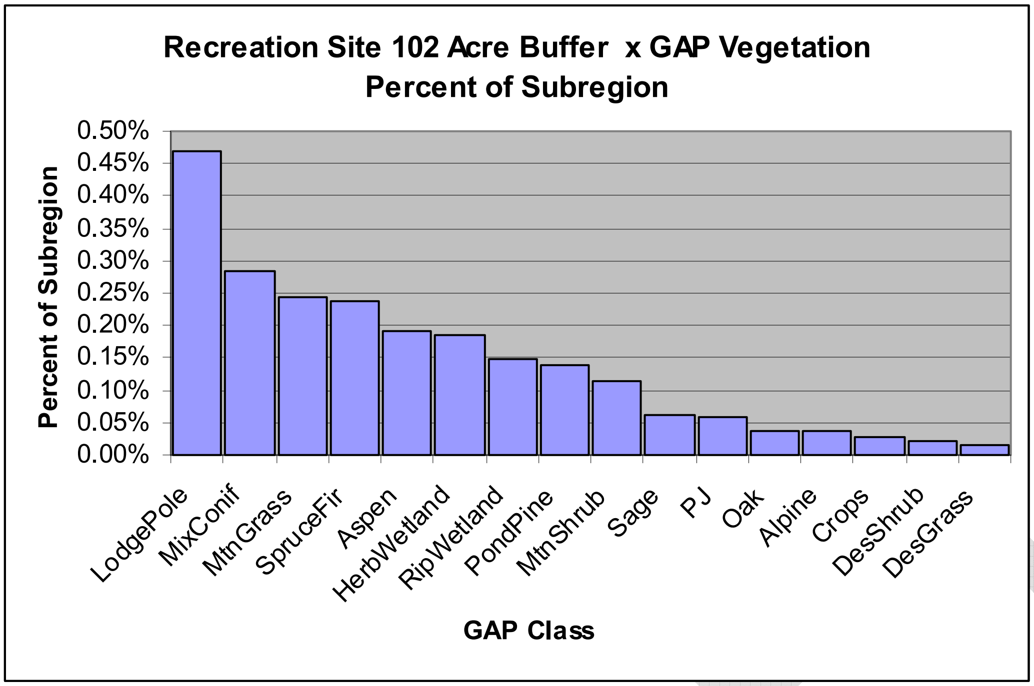


Figure 7-26 GAP Vegetation Classes organized by percentage of subregion class area. The top five classes are shown in Figure 7-27 below.

The five classes having the highest levels of disturbance, as a percentage of total GAP vegetation community area, are located in upland areas managed almost exclusively by the Forest Service. These GAP classes are Lodege Pole, Mixed Conifer, Mountain Grassland, Spruce Fir and Aspen. About half (172) of all sites are found in these vegetation communities (Fig. 7-27). Consequently, Forest Service management, development and use of these sites should take into consideration plans and actions aimed at maintaining these vegetation communities and related ecological components.

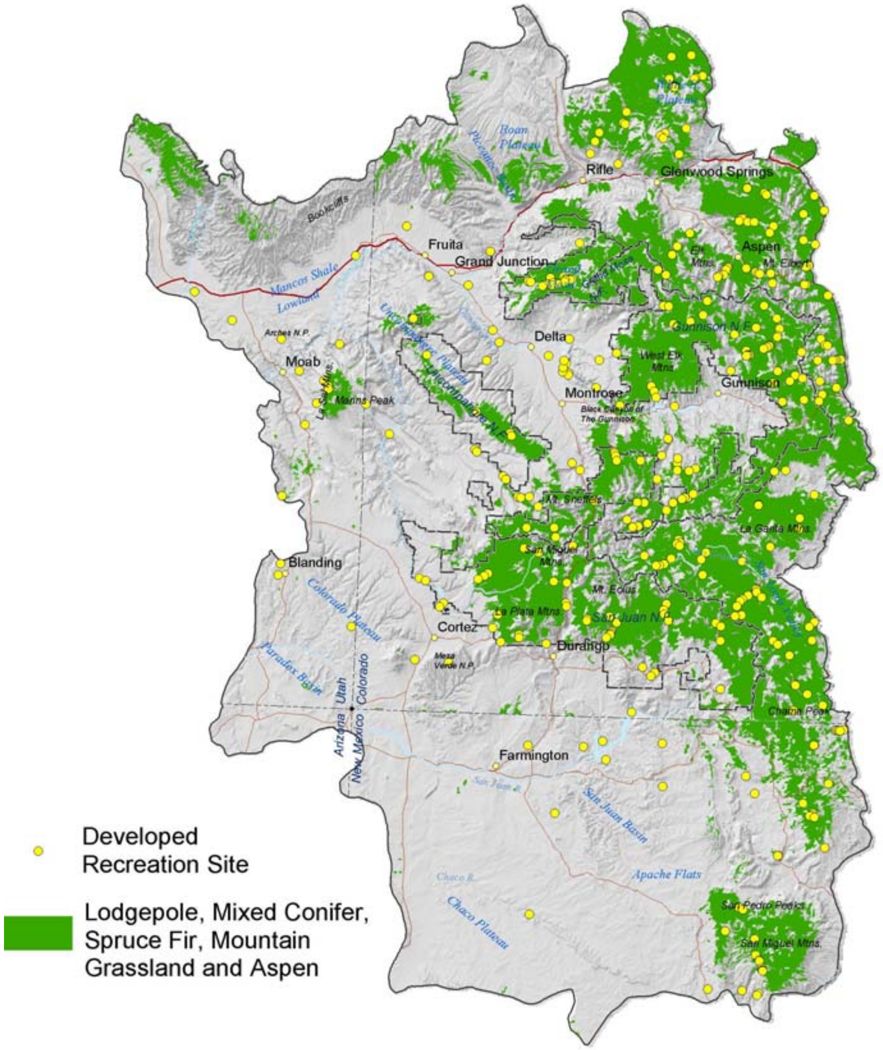


Figure 7-27 Distribution of the five GAP classes having a disturbance percentage over than 0.10%.
(Table 7-20 above). These lands are almost exclusively managed by the Forest Service.

Data Needs

Better estimates of developed recreation site disturbed areas are important. Current data are point location only and provide little insight into geometry and extent of use areas.

Developed recreation databases should be standardized between agencies. Inventories of privately owned and managed developed recreation sites would add to our understanding of recreation.

Future analyses would be more complete with measures of disturbance resulting from developed recreation such as boating ramps, picnic areas etc.

Future analyses would be enhanced by a comprehensive study of seasonal use patterns and duration.

Analyses should include recreational sites on private lands.

Bibliography

Canyonlands, 2005. Canyonlands National Park Visitor Study. Online: Address to be recovered.

Chelan County PUD. 2001. Recreation Resources Inventory Summary Report. Preliminary Discussion Draft. Rocky Reach Hydroelectric Project. FERC Project No. 2145. Online: http://www.chelanpud.org/rr_relicense/study/reports/2731_1.pdf

Cordell, Ken H. and Gregory R. Super. 2000. Trends in Americans' Outdoor Recreation. In: Trends in Outdoor Recreation and Leisure and Tourism. Eds. W.C. Gartner and D. W. Lime. CAB International.

Gramann, James H. 2003. Visitation Forecasting and Predicting Use of NPS Parks and Visitor Centers: Focus Group Report.

Kocis, Susan M., Donald B.K. English, Stanley J. Zarnoch, Ross Arnold, Larry Warren and Catherine Ruka. 2004. National Visitor Use Monitoring Results, GMUG National Forests, USDA Forest Service Region 2.

Meldrum, Bret H. Margaret A. Littlejohn and Steven J. Hollenhorst. 2004. Arches National Park Visitor Study, Spring 2003. Visitor Services Project, Report 150. July, 2004. Online: http://www.psu.uidaho.edu/files/vsp/reports/150ARCH_rept.pdf

NPS, 2005. Public Use Statistics Office, NPS Visitation Database Reports. U.S. Department of the Interior. Online: http://www2.nature.nps.gov/stats/

SCORP. 2003. Statewide Comprehensive Outdoor Recreation Plan, Colorado's Outdoor Recreation Future. Strategies for Colorado's Outdoors Heritage. State of Colorado State Parks. Online: http://parks.state.co.us/scorp/index.asp

Spangle, Steven L. 2004. Memo to E.J. Zieroth, Forest Supervisor, Apache-Sitgreaves National Forest. Online: http://arizonaes.fws.gov/Documents/BiologicalOpinions/2004/040107_BigLakeCampground.pdf

USDA. 2001. National Visitor Use Monitoring Results, August 2001. Rio Grande National Forest. USDA Forest Service, Region 2.

Anthropogenic Influences

Dispersed Recreation – CLC Subregion

San Juan and GMUG National Forests

Key Findings

- Correlation of **3,037** known dispersed recreation sites in the San Juan and GMUG Forests to roads, streams, slope, aspect, vegetation and ownership allows the creation of a predictive model of dispersed recreation across the CLC subregion.
- There are **2,373,907 acres (3,709 square miles)** of lands with high potential for dispersed recreation sites in the CLC subregion.
- Using a ratio of 7.4 sites per square mile of high potential site area, there is a potential for about **27,443** sites in the CLC Subregion, distributed across public lands, mostly in upland valleys.
- Dispersed recreation site average barren area is about **45** square feet per site (about 7 x 7 feet). Overall disturbed area is about **905.32** square feet per site (about 30 x 30 feet). Potential disturbed area across the subregion is about 24,845,090 square feet, or **570.3** acres.
- Research and validation is required to develop a more robust statistically valid model. The existing model is only qualitative and rests on some important assumptions.

Introduction

The influence of dispersed recreation on both terrestrial and aquatic systems is an emerging issue for public land managers as levels of recreational use increase on public lands. Dispersed recreation is defined here as: *the ad hoc location of short-term campsites by the public*. Typical sites include disturbance features such as fire rings, tent sites, areas of congregation, bare areas, parking and automobile tracks. Repeated use of sites and abusive practices increase can disturb important vegetation communities and lead to increased levels of sedimentation and contamination of aquatic systems. As a consequence, measures of disturbance are important to the CLC assessment.

Inventories of dispersed recreation sites are now being completed by forest staffs but **large areas of public lands remain to be examined**. In the meantime, an estimate of potential levels of dispersed recreation use is required to support assessments at the CLC sub-region scale. **This document describes a model aimed at meeting this requirement.** The resulting model may also prove useful in driving selection of lands for future inventories.

The model described here relates existing dispersed recreation sites from both San Juan and GMUG National Forests surveys to landscape and cultural features. These features include streams, roads, slope, aspect, vegetation and jurisdiction. We extrapolate from these known sites by selection of feature attributes that most strongly correlate to known sites. This spatial model yields a qualitative geographic model indicative of potential. More robust studies are required to generate a more robust statistically based quantitative model.

Here we first describe a model to extrapolate from known dispersed recreation sites outward across the subregion. The model may be then used to highlight areas most at risk from dispersed recreation activity and to measure potential levels of disturbance.

Description of the Model

Geographic data sets were obtained from both the San Juan and GMUG Forests. These sets include 3,037 point locations for dispersed recreation sites. Of these 3,037, 2,909 fall inside the CLC subregion. These 2,909 sites are well distributed among a variety of landscape settings. These include alpine, upland and lowland sites. Sites are well distributed among both forests (Fig. 7-28). Patterns in these data are strongly suggestive of a correlation to drainage and road networks.

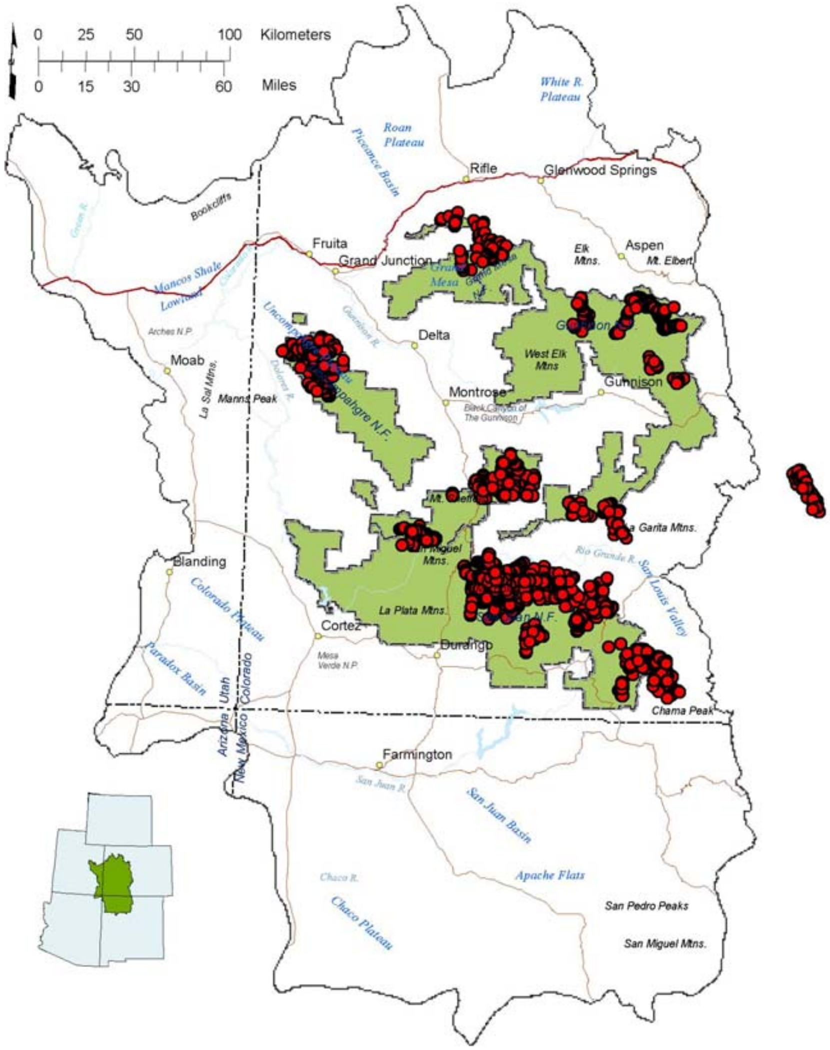


Figure 7-28 Dispersed recreation sites.
There are 3,037 sites shown in red. Note that 128 sites fall outside the subregion leaving 2909 inside. These 2909 known sites are the basis of the predictive model described here.

Furthermore, **correlation of the inventoried sites to roads, streams, slope, vegetation, aspect and ownership provides a method to extrapolate into un-inventoried areas**. Very strong relationships are evident in measures of point distances to roads and streams and these relationships are described here.

Ultimately, in the creation of the final predictive model we select from the source data sets (e.g. roads, streams etc.) areas that fall within a 90% threshold. In other words we assume a correlation to attributes of these data sets where we find that 90% or more of dispersed recreation sites occur in a given range. For example, a range might be distance to features, slopes values or vegetation classes. These are all described below.

The model has been developed across the CLC sub-region to support analysis at multiple scales for analysis of both terrestrial and aquatic systems.

First we describe the model in terms of the correlates and then we use the model to estimate levels of disturbance in the subregion.

Roads and Trails

U.S. Geological digital line graph (DLG) 100K scale data was used to represent roads and trails. According to these data, in the CLC subregion, there are about 75,000 miles of roads and trails (Table 7-21). This measure is likely to be highly accurate for primary, secondary and Class 3 roads. However, the number of miles of actual Class 4, 4WD and Trails (e.g. non-system roads) is likely to be somewhat higher than exists in the 100K DLG data. Never the less, the overall spatial distribution is consistent and accurate.

Table 7-21 Road Classes in USGS 100K DLG data for the sub-region.

ClassCode	Road Class	Miles	Pct
1	Primary	1,848.5	2.4%
2	Secondary	2,710.4	3.6%
3	Class 3	10,853.8	14.4%
4	Class 4	46,102.8	61.0%
5	4WD	1,595.8	2.1%
6	Trail	12,459.4	16.5%
		75,570.6	100.0%

In the model, roads and trails of all classes, including trails were used. **Over 90 percent of all sites in the sub-region fall within 800 meters of roads (Fig. 7-29)**. Sixty-three percent are found within 100 meters.

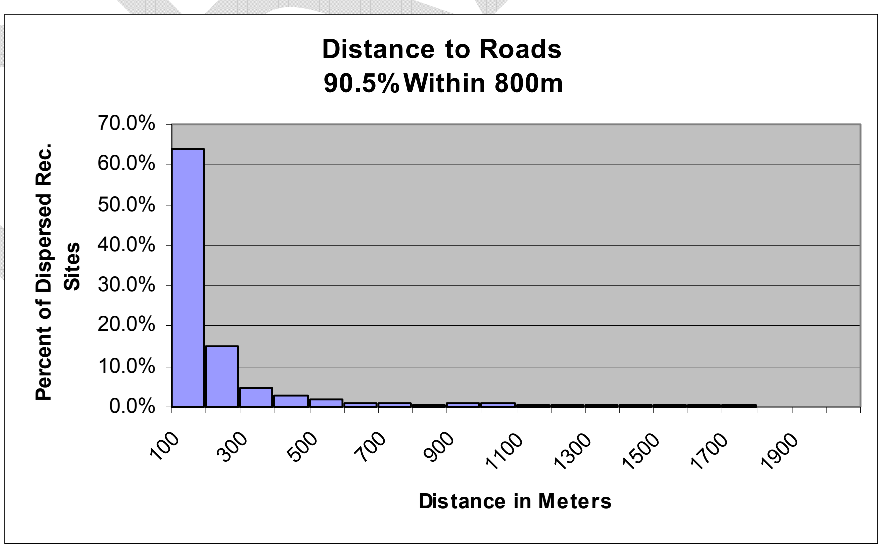


Figure 7-29 Just over 90 percent of sub-region sites fall within 800 meters of a road.

Streams

Like roads, U.S. Geological digital line graph (DLG) 100K scale data was used to represent streams. According to these data, **in the CLC subregion, there are about 60 thousand miles of stream by all types (Table 7-22)**. Almost two-thirds of these are intermittent. Just over thirty percent are either perennial streams or shoreline. Diversions, other, tunnel and washes comprise the remainder.

Table 7-22 Stream classes in USGS 100K DLG data for the sub-region.

StreamType	Length (Mi)	Pct
Diversion	2,661.8	4.5%
Intermittent Stream	35,793.5	60.1%
Other	955.5	1.6%
Shoreline	3,078.3	5.2%
Stream	15,686.6	26.4%
Tunnel	7.7	0.0%
Wash	1,337.5	2.2%
	59,521.0	100.0%

Almost 60 percent of all sites are found within 100 meters of a stream. Nearly 90% of sites are located within 700 meters of a stream (Fig. 7-30).

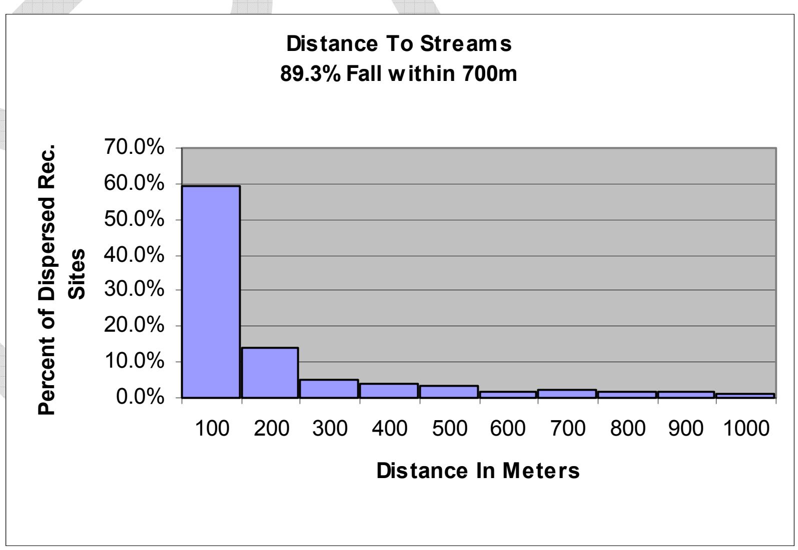


Figure 7-30 Just under 90 percent of sub-region sites fall within 700 meters of a stream.

Slope

Naturally, slope would seem to be a strong limiting factor in selection of dispersed recreation sites for camping and/or day use. A slope model based on subregion-wide 100 meter DEM was used. In the subregion, **ninety percent of dispersed recreation sites are found on slopes of 17 degrees or less (Fig. 7-31).**

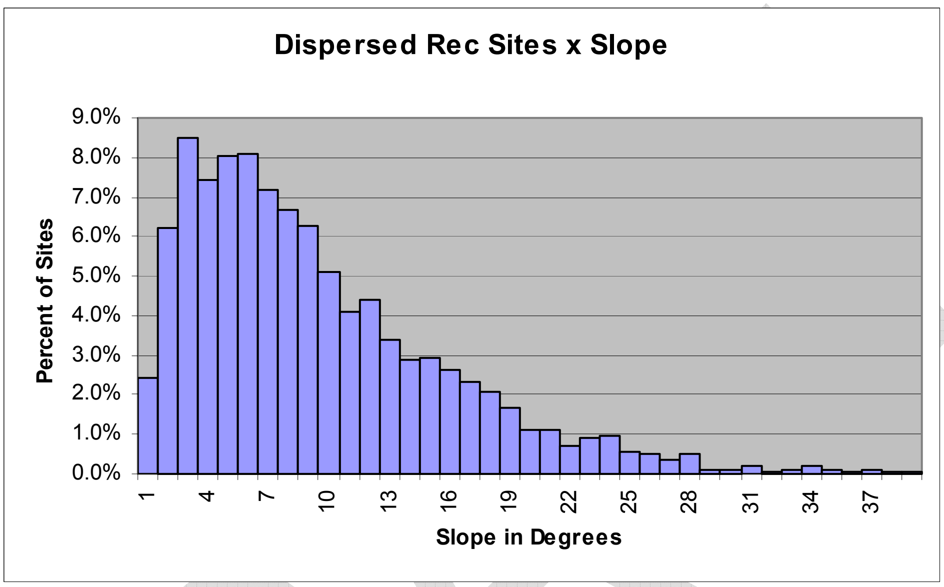


Figure 7-31 Ninety percent of sub-region sites fall in areas with slopes of 17 degrees or less.

Aspect

Similar to slope, it would seem that aspect might be a strong limiting factor in selection of dispersed recreation sites for camping and/or day use. It might be expected that users would tend to select sites with southerly, warm and sunny, sites. This assumption seems to be weakly so – i.e. it is less evident in the data than slope. **About 90% percent of all sites fall in an aspect range between 35 and 325 degrees.**

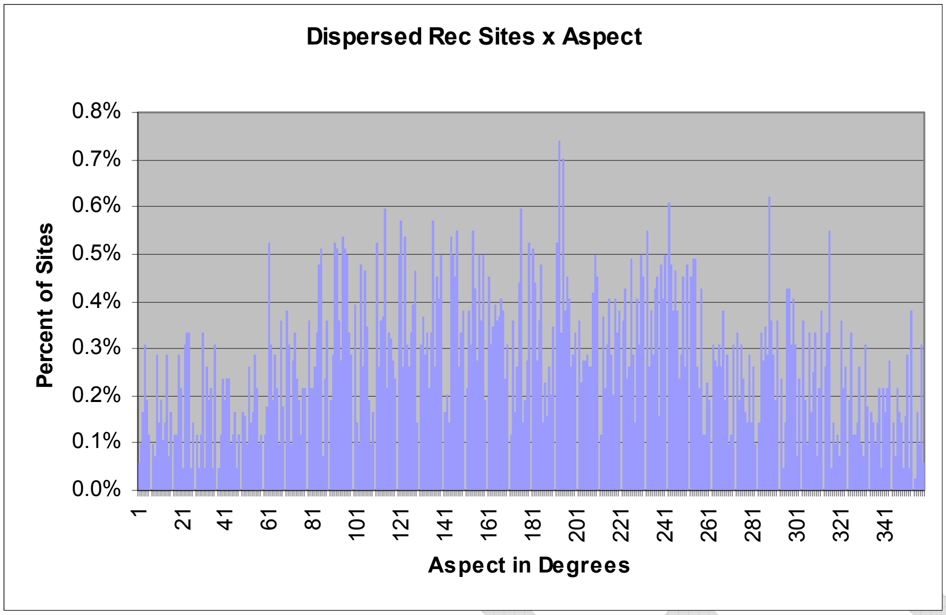


Figure 7-32 Aspect of Dispesed Recreation Sites.
Ninety percent of sub-region sites fall in areas with an aspect in the range from 35 to 325 degrees. A minor trend seems to be evident in the data, centered on 180 degrees.

Vegetation

About 82% percent of sites fall within three generalized GAP vegetation classes. These include Spruce-Fir, Alpine, Aspen, suggesting an upland preference. Just over 50% percent of sites are found in the Spruce-fir GAP class. Another 6.4% percent of sites fall in the Sagebrush vegetation class.

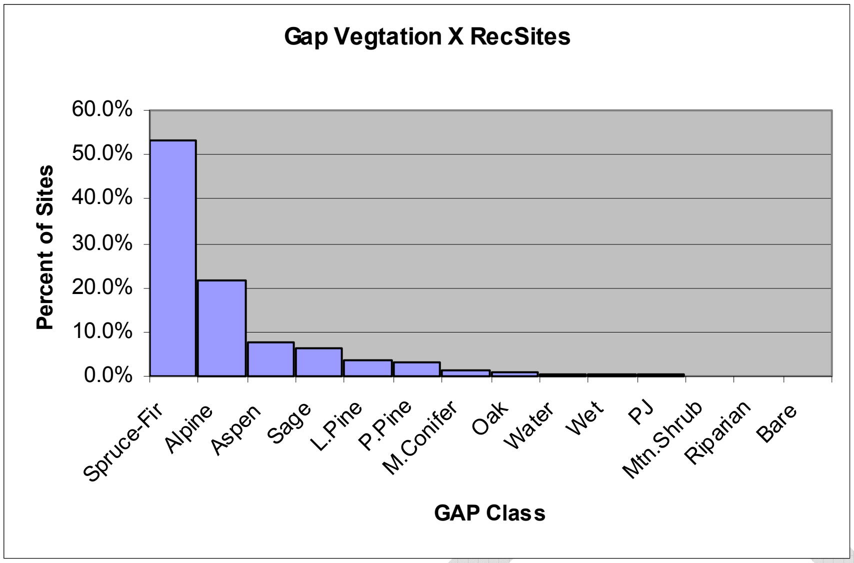


Figure 7-33 Dispersed Recreation Sites in GAP vegetation classes: Spruce-fir, Alpine and Aspen.
Just under 82% percent of sites occur in three GAP vegetation classes: Spruce-fir, Alpine and Aspen. Sagebrush constitutes another 6.4% percent for a sum of about 89% percent of all sites.

Ownership

Almost 97% percent of sites are found on Forest Service lands. This is no surprise as the surveys were deliberately aimed at Forest Service lands. Thus, a correlation to ownership is not directly evident in the sample data. **Here the model includes ownership as an a priori assumption**. While some level of dispersed recreation activity may occur on non-public lands, it is assumed to be actively discouraged on other lands while most public lands remain largely open to it. The model is applied within BLM and Forest Service lands only (Fig. 7-34).

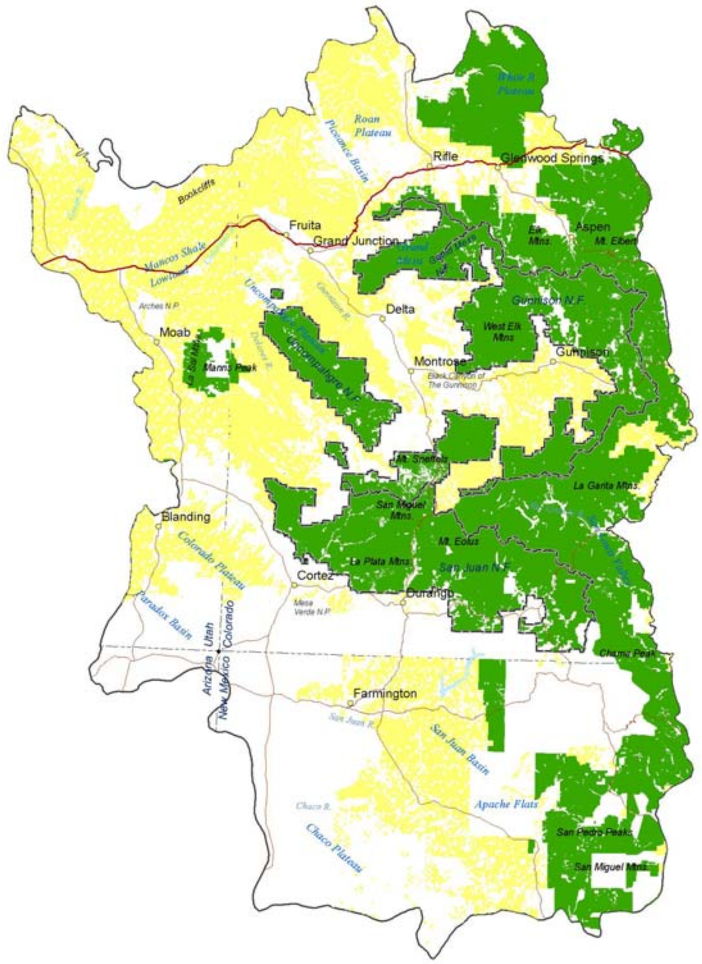


Figure 7-34 The dispersed recreation model is applied to BLM and Forest lands in the sub-region.

The Dispersed Recreation Model

The dispersed recreation model extrapolates away from the known sites by selecting from the base data sets, those areas where over 90% percent of the sites occur. These values are summarized in Table 7-23.

Table 7-23 The six layers used to define the dispersed recreation model.

Layer	Selection Attributes
Roads	Within 800 meters
Streams	Within 700 meters
Slope	0 to 17 degrees
Aspect	35 to 325 degrees
Vegetation	Spruce/Fir, Alpine, Aspen
Ownership	BLM and Forest Service

To prepare the model we converted vector data roads, streams, vegetation and ownership to 50 meter cell-size grids over the subregion. From these new grids, along with existing 100 meter slope and aspect grids, we selected areas matching the "Selection Attributes" shown above. The resulting subset grids were integrated using the Arc/Info GRID function "combine". The integrated grid was then re-classed into a dichotomous classification of 1 or NODATA where 1 is indicative of a high potential for dispersed recreation (Fig. 7-35). **The resulting model covers 2,373,907 acres (3,709 square miles)**. This is just under 7% percent of the area within the subregion.

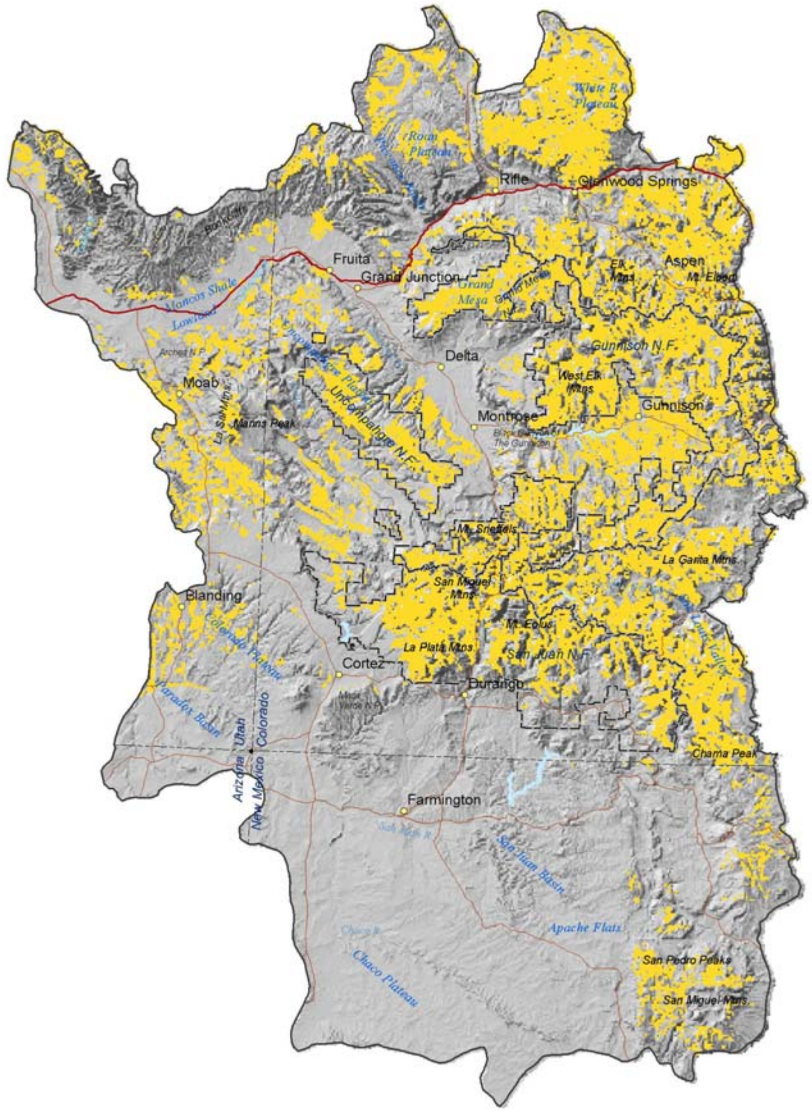


Figure 7-35 The dispersed recreation model shows areas having attributes that make these areas likely for dispersed recreation site selection and use.

Estimation of the Number of Potential Sites

With the model in hand, it is useful next to develop some estimate of the number of sites that might be found across the subregion. To do so, we calculated a ratio of number of sites to model area in localized regions. This ratio may then be applied throughout the model to estimate the cumulative use. Disturbance levels may then be calculated by multiplication of site estimates times average site disturbance metrics taken from a subset of the inventoried site data.

Definition of Localized Regions

Localized regions were defined as polygons containing aggregated groups of points from the actual dispersed recreation site data. These containing polygons were then used to “clip” areas of the dispersed recreation model. A ratio of the number of points, in the localized area, to the clipped model area is indicative of the potential number of sites per unit area, overall, in the model. Figure 7-36 illustrates the relationship between a localized region (window) and associated model areas and points.

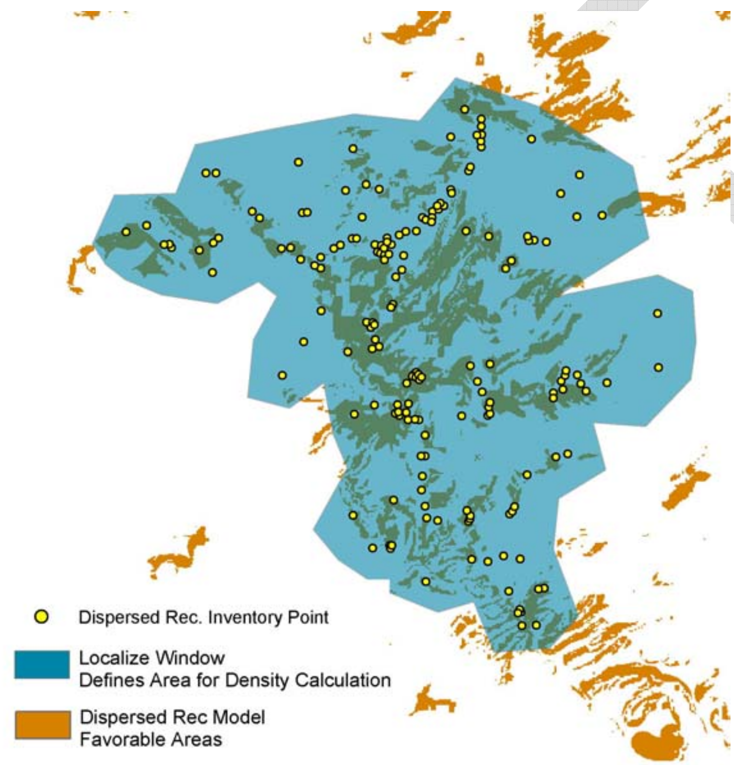


Figure 7-36 Recreation model used to define areas suitable for calculations of site density.
The “localize window” provides a container for groups of points and then is used to clip the recreation model to define areas suitable for calculations of site density.

For twenty-one localize windows there are 2,909 sites. The same windows contain model areas summing to 393.2 square miles. Dividing 2,909 by 393.2 we arrive at a ratio of 7.4 sites per model square mile.

Table 7-24 For all 21 localize windows, count the number of sites and model area in square miles.

Rank	Number of Sites	Model Area SqMiles
1	5	0.0
2	38	0.3
3	73	0.7
4	20	0.3
5	41	1.8
6	1	0.0
7	148	9.3
8	1337	101.4
9	19	1.8
10	356	34.6
11	234	25.8
12	205	34.7
13	90	17.2
14	21	4.2
15	21	4.5
16	172	53.8
17	19	6.2
18	16	6.4
19	2	1.4
20	1	0.7
21	90	87.8
	2909	393.2
	2909 / 393.2 =	7.40 Sites Per SqMi.

As shown above, the overall area of the dispersed recreation model is **3,709** square miles. At **7.4** sites per square mile, the potential number of sites in the sub-region is **27,443**.

Estimation of Disturbance

Of the sites provided by the GMUG National Forest, 353 sites include attribute information indicating **barren area per site**. And 357 sites include attribute information suitable to estimate **overall disturbed area**. These data allow us to calculate an overall estimate of potential disturbance resulting from dispersed recreation. Table 7-25 illustrates how an estimate of **45.04** square feet of barren area has been derived. Table 7-26 illustrates the same leading to an estimate of total area per site of **905.3** square feet.

Table 7-25 In the existing site data, 353 sites provide a way to estimate average barren area per site.

Number of Sites	Class	Barren Area SqFeet	NumSites x Area
240	1	0	0
68	2	50	3,400
30	3	100	3,000
11	4	500	5,500
4	5	1,000	4,000
353			15,900
Average Barren Area = 15900 / 353 = 45.04 SqFt			

Multiplying this estimate of **45.04** square feet per site times **27,443** sites yields an estimated 1,236,117 square feet, or **28.4** acres of potential barren area in the subregion.

Table 7-26 In the existing site data, 357 sites provide a way to estimate average total area per site.

Number of Sites	Class	Area Sq Feet	Approx Sum Area
27	1	500	13,500
26	2	2000	52,000
11	3	5000	55,000
134	a	50	6,700
45	b	100	4,500
61	c	500	30,500
26	d	1000	26,000
27	e	5000	135,000
357			323,200
Total Area = 323,200 / 357 = 905.32 sq. feet.			

Multiplying this estimate of **905.32** square feet per site times **27,443** sites yields an estimated 24,845,090 square feet, or **570.3** acres of total disturbed area in the subregion.

Vegetation Classes and Potential Dispersed Recreation

GAP vegetation data is a driving variable in the development of the dispersed recreation model. Namely, four classes comprise almost 90% percent of all inventoried dispersed recreation sites. These are Alpine, Aspen, Spruce-fir and Sagebrush. Alpine, Aspen and Spruce-fir communities are principally confined to upland settings with sage communities spanning a broader ecological gradient.

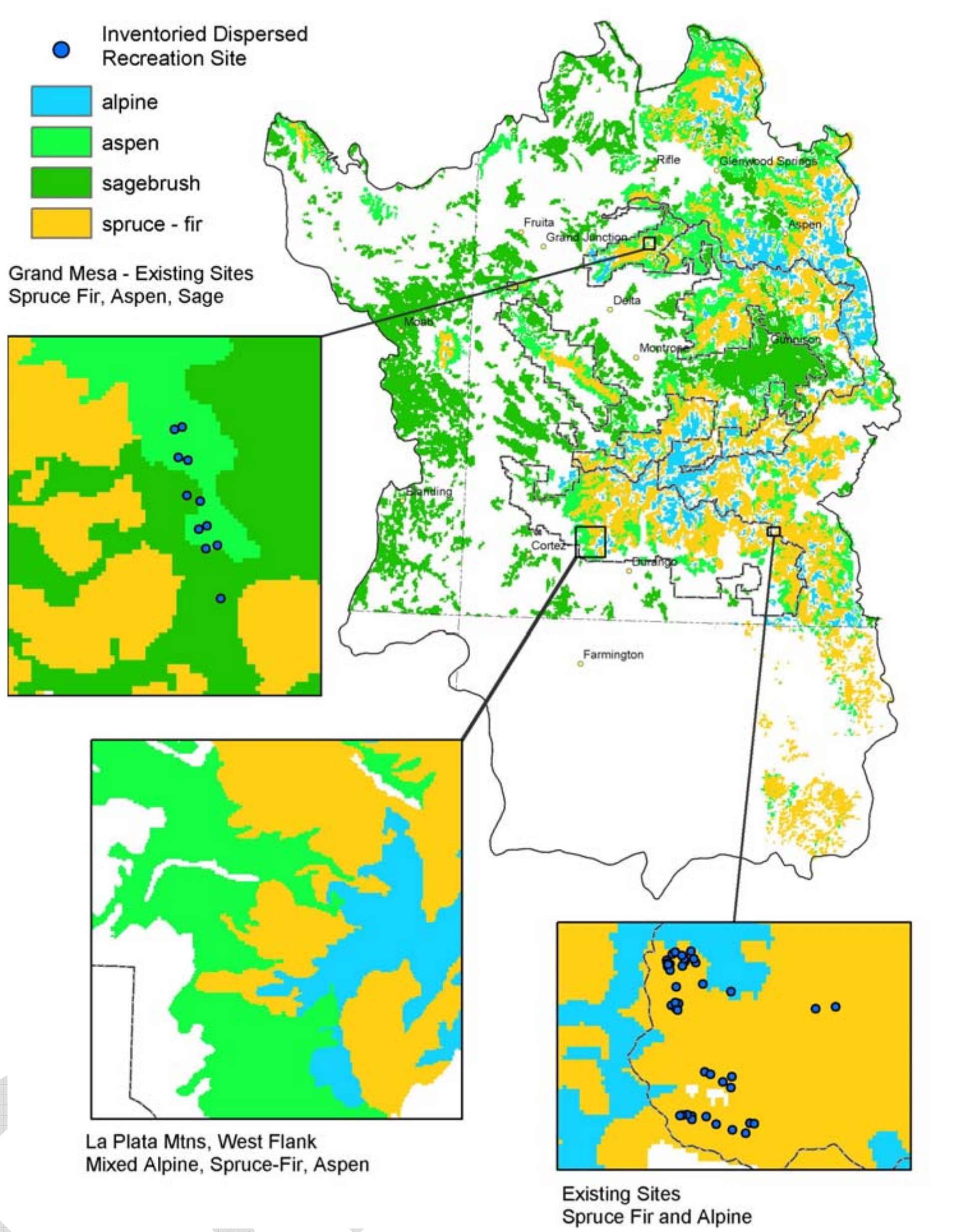


Figure 7-37 Distribution of GAP vegetation types most often associated with dispersed recreation sites.

In the subregion, about 23% percent of these four classes fall within areas classified as having a high probability of disturbance from dispersed recreation activities (Table 7-27). The distribution of these classes, intersected with the dispersed recreation model is shown in Figure 7-38.

Table 7-27 The relation of overall class area to area covered by the dispersed recreation model.

Vegetation Class	Overall Class Area Square Miles	Dispersed Rec. Class Area, Sq. Miles	Percent by Class
sagebrush	4,633	1,063	22.94%
aspen	3,552	910	25.62%
spruce - fir	5,445	1,323	24.29%
alpine	2,510	414	16.49%
	16,140	3,710	22.98%

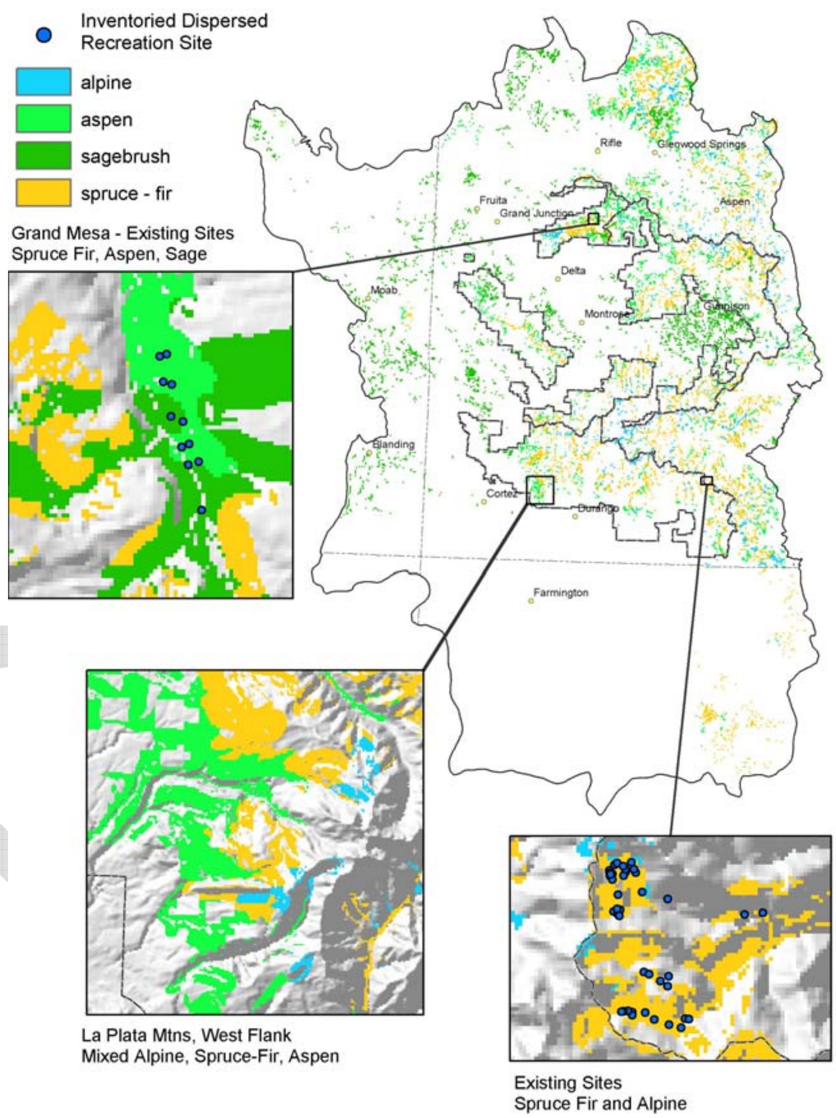


Figure 7-38 Distribution of principal GAP vegetation types that intersected with the dispersed recreation model.

Information Needs

Forest inventories developed to date have provided a baseline to extrapolate potential densities used here. Inventories should continue to refine estimates and ultimately directly characterize the actual setting in the field.

While methods to develop estimates continue to be used more systematic, statistically robust, sampling methods should be used to ensure that the observed relationship of existing sites to roads is a valid correlate and not a bias introduced because the primary method to access sites is by roadway. Also, the method used to calculate an overall site density of 7.4 sites per model square mile should be evaluated. At the same time, the existing model may also be useful in the development of study plans and target areas for sampling.

Sampling systems should also be developed that develop estimates in the rate of growth or expansion in site location along with trends in site selection.

Developed Recreation – Developed Recreation San Juan NF/BLM CLC Landscape 042005

Key Findings

- The CLC landscape is located at the margin of the Colorado Plateau and Southern Rocky Mountains. This very scenic region is a "recreational hotspot".

- The CLC landscape is at the crossroads to many important regional attractions including the Grand Canyon, Mesa Verde, Canyonlands and Arches National Parks. The landscape is geographically central to important and growing population centers including Salt Lake City, Denver, Albuquerque, Sante Fe and Phoenix.

- More than half of the visitors to National Parks adjacent to the landscape come from California, Colorado, Utah and a few eastern states. Park visitors also utilize Forest recreation opportunities.

- On Forest lands, about half the users are from local communities, especially, Durango, Montrose, Grand Junction and Gunnison. This is important because levels of visitation will likely increase significantly with continued robust growth for these communities and others in the landscape.

- Recently, while visitation to National Parks seems to be leveling off or dropping, visitation to the National Forests and BLM lands in the landscape are increasing.

- As use of National Forests and BLM lands increases and user quality of experience may eventually drop as availability of recreational resources diminishes. At the same time, current and future increases in recreational use of public lands will lead to increased disturbance and potential conflicts and competition between recreation and other management programs.

- Using estimated averages by site type for 84 developed recreation sites in the CLC Landscape influence about 4,700 acres.

- Almost half of all developed recreation sites are found in the four vegetation classes: Warm Dry Mixed Conifer, Riparian, Cool Moist Mixed Conifer and Ponderosa Pine

Introduction

The striking and challenging geography, scenery and recreational opportunities of the Colorado Plateau and Southern Rocky Mountains attract growing numbers of visitors to public, state and private lands in the San Juan landscape and surrounding areas. Developed recreation opportunities include camping, and picnicking in developed sites, skiing at developed resorts along with sightseeing and visits to sites of cultural and historical importance.

Recreational opportunities in and adjacent to the landscape draw visitors from both local and distant communities. Scenery, attractions and recreational opportunities in the landscape are of such a quality as to provide destinations attracting significant numbers of visitors. At the same time, the landscape is at an important crossroads for recreational visitors to well known National Parks in the surrounding five-state region (Fig. 7-39). These Parks include the Grand Canyon, Arches, Canyonlands, Bridges, Dinosaur, Rocky Mountain and the Sand Dunes. Furthermore, the Mesa Verde National Park is in the landscape itself. Very often, these "travel-through" visitors visiting these Parks also take advantage of recreational opportunities in the landscape.

The Colorado Plateau and Rocky Mountains attract local, national and international visitors. As of 2003, 50% percent of visitors to Arches National Park were from six states. Of these, 33% percent were from California (16%), Colorado (9%) and Utah (8%) (Meldrum, et al. 2004). The remaining visitors are from three eastern states including Illinois (7%), New York (5%) and Virginia (4%). Similarly, over 50% of surveyed visitors to Canyonlands National Park are from three western states. These states include Colorado (29%), California (15%) and Utah (11%) (Canyonlands, 2005).

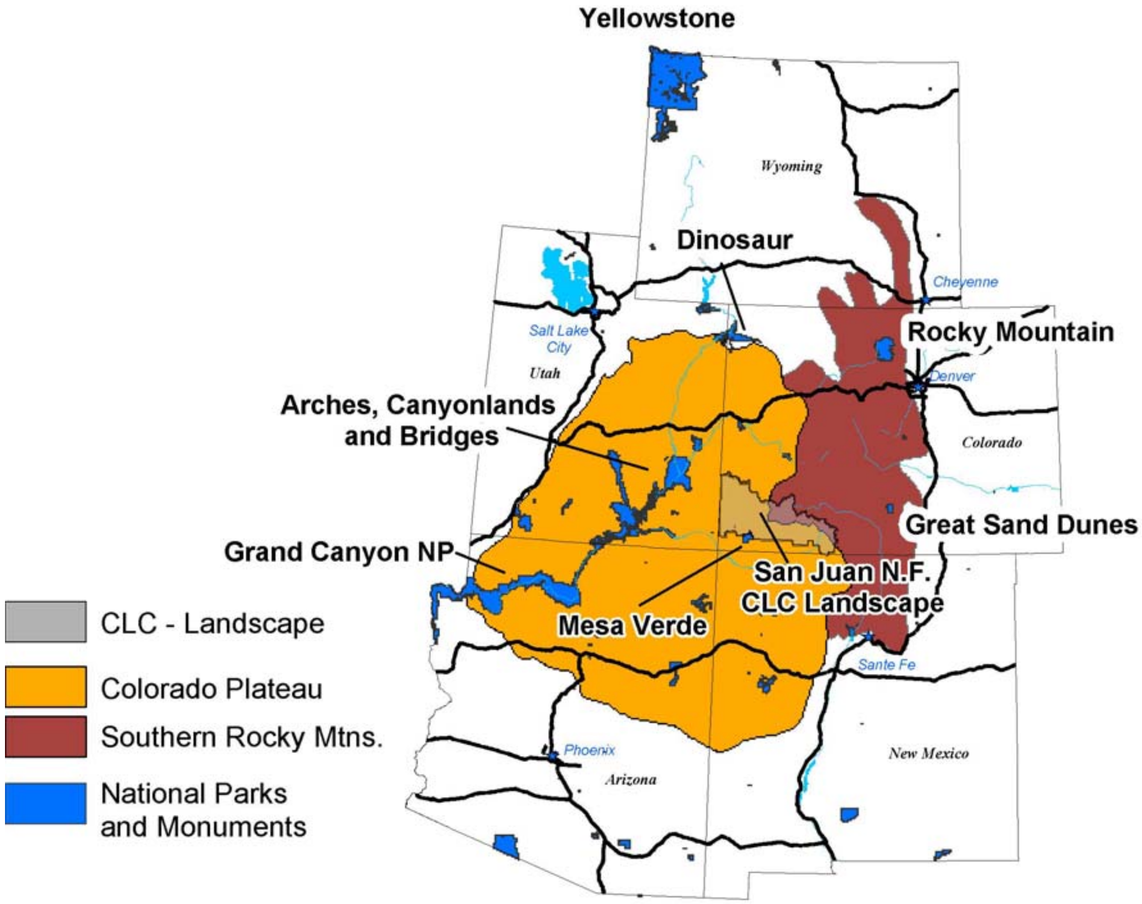


Figure 7-39 The CLC Landscape straddles both the Colorado Plateau and Southern Rocky Mountains physiographic regions.
And the landscape falls at the cross-roads for travelers from large regional metropolitan centers including Salt Lake City, Phoenix, Albuquerque, Santa Fe and Denver.

Importantly, studies show that over recent years there have indeed been significant increases in public participation in recreation activities nationally. This trend includes a wide spectrum of recreation opportunities and settings. The most significant increases in recreation activities include snow skiing, canoeing/kayaking, cycling, camping, sailing, swimming, fishing, horseback riding and hunting. With the exception of hunting, these activities have shown significant increases since 1995 (Cordell and Super, 2000). World class opportunities for these activities are found in the San Juan CLC landscape.

This trend of increasing public interest may be combined with changes in use patterns on National Parks to show that both BLM and National Forests in the landscape are of key importance to developed recreational activities in the landscape. As a result, levels of recreational activity, demand and influences on terrestrial and aquatic ecosystems are also increasing in character, degree and extent.

For example, visitation to Mesa Verde, Arches and Canyonlands National Parks steadily increased to maximums in the early 1990's. Then during the decade of the 1990's visitation leveled out and slumped (Fig. 7-40). While the slump in visitation may be at least partly attributed to a down turn in the national economy, or as a response to recent drought and fire, the overall trend may also attributed to a diminishing quality of user experience in these National Parks. This diminishing quality of experience may be an indication that National Parks are at or near their capacity to provide positive recreational experiences.

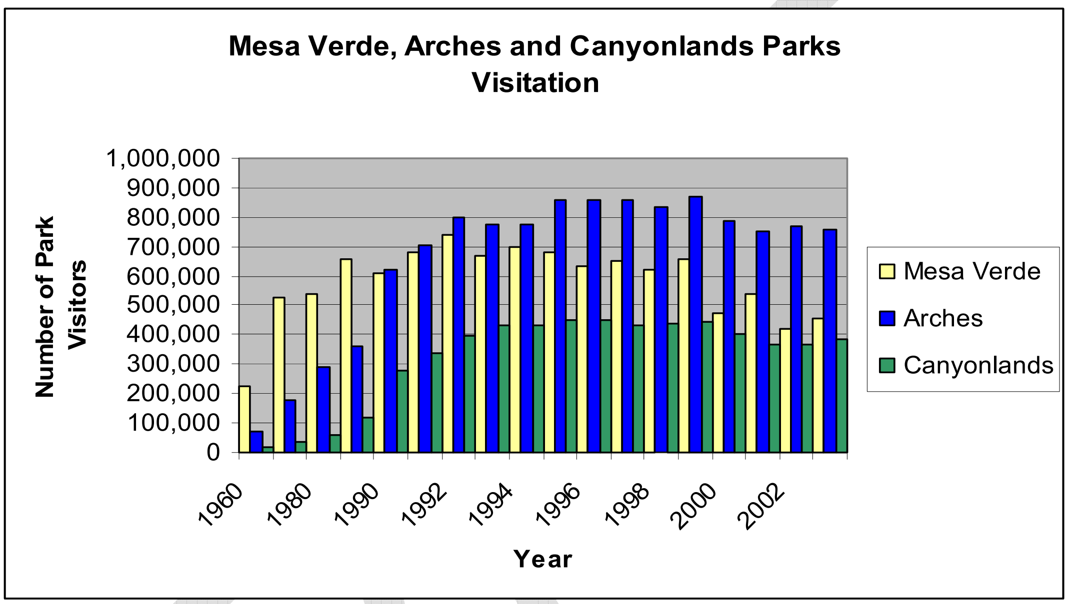


Figure 7-40 Visitation trends for the Mesa Verde, Arches and Canyonlands National Parks. These trends show and overall trend of diminishing visitation in these three parks (NPS, 2005).

So, as overall demand for recreational resources continues to increase, and quality of experience diminishes in National Parks, recreational users are seeking out new opportunities elsewhere. From 1986 to 1996 visitation levels for the National Forests, BLM lands, Federal lands managed by the Army Corps of Engineers and the Bureau of Reclamation (BOR) all increased while visits to National Parks dropped (Fig. 7-41).

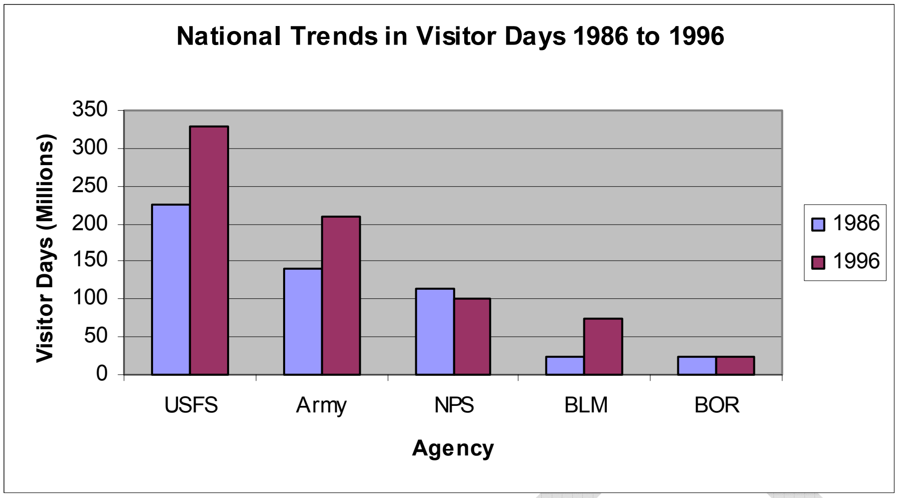


Figure 7-41 National trends in visitation suggest a shift away from National Parks to National Forests, Army Corps of Engineers lands and BLM lands.
(Adapted after Cordell and Super, 2000).

Large areas of these public lands, especially lands managed by the U.S. Forest Service (USFS) and Bureau of Land Management (BLM), are available in the CLC landscape (Table 7-28 and Fig. 7-42). Upland areas along the northern margin of the landscape are principally in Forest Service jurisdiction (San Juan and GMUG) while the desert canyon-lands and foothills include significant tracts of BLM land. Valley bottoms and table lands are principally in private jurisdiction.

Table 7-28 Land Ownership/Jurisdiction by agency in the CLC Landscape

Owner	Acres	Pct	Sum Pct.
USFS	1,860,333	49.4%	49.4%
Private	1,160,390	30.8%	80.2%
BLM	664,684	17.7%	97.9%
State	72,831	1.9%	99.8%
Tribal	6,728	0.2%	100.0%
NPS	448	0.0%	100.0%
Other	132	0.0%	100.0%
	3,765,545	100.0%	

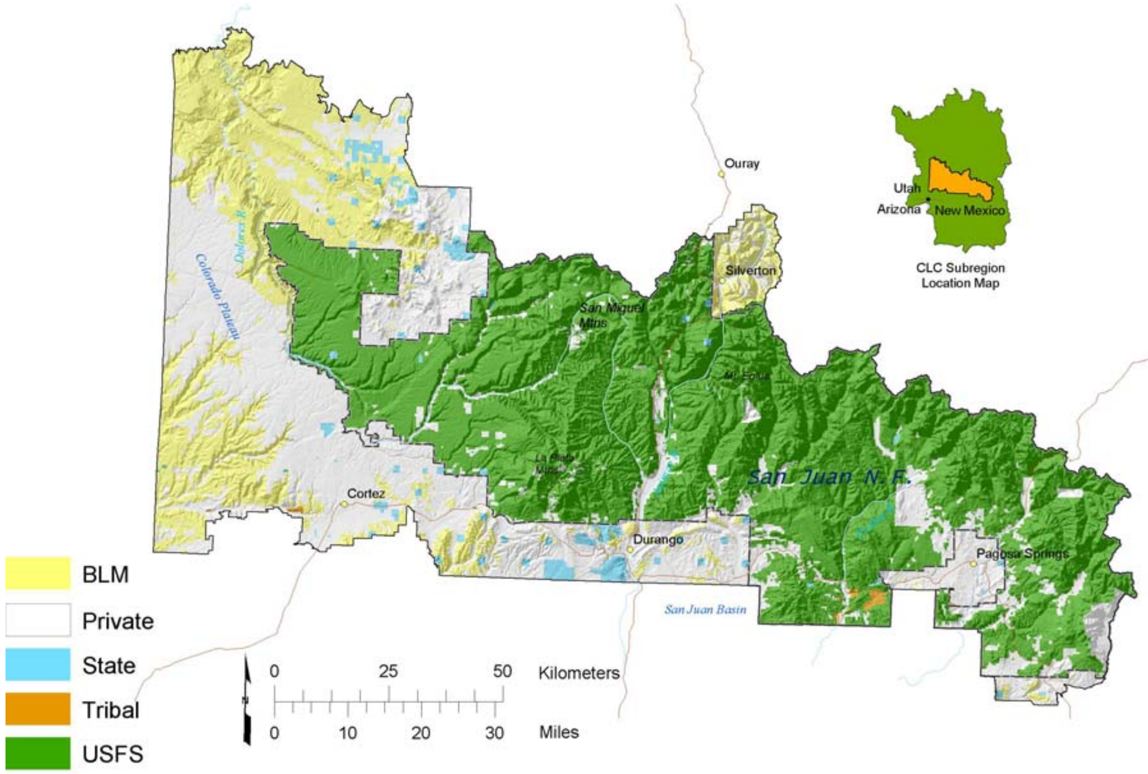


Figure 7-42 Large tracts of public lands in and around the CLC landscape.
Large tracts of public lands in and around the CLC landscape provide important recreational opportunities attracting visitors from both local and distant communities.

Local visitation is important too. U.S. Forest Service surveys for the San Juan Forests show that about half the number of Forest visitors report home zip code locations that are beyond 50 miles of the Forest (USDA, 2001). About 51% percent of visitors to the San Juan Forest (Fig. 7-43) and 44% of visitors to the GMUG Forest (Fig. 7-22) are from communities within 50 miles. The data show that the principal source communities for visits to the San Juan Forest include Durango, Farmington, Cortez and Pagosa Springs.

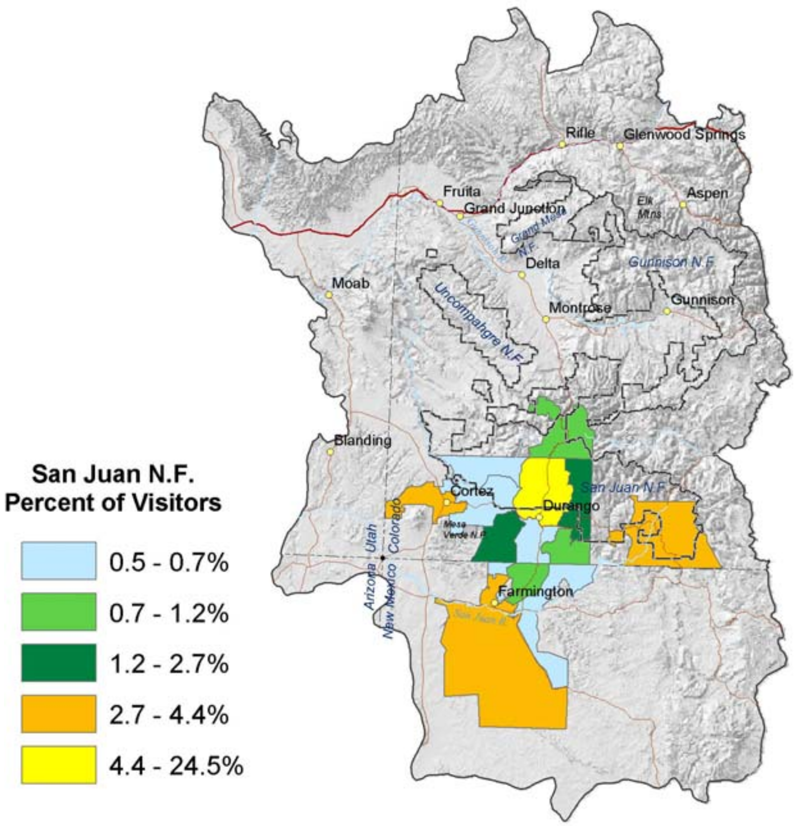


Figure 7-43 Home communities, by zip code, of local visitors to the San Juan National Forest displayed *within the CLC subregion.*

These western slope communities are expected to grow significantly over the coming years. By 2025 the population of Colorado is expected to grow by another 48% percent. Currently reported growth in the region for five years (1997 to 2001) averages just over 2% per year (Fig. 7-44). Much of this growth will take place in rural counties with access to public lands. (SCORP, 2003).

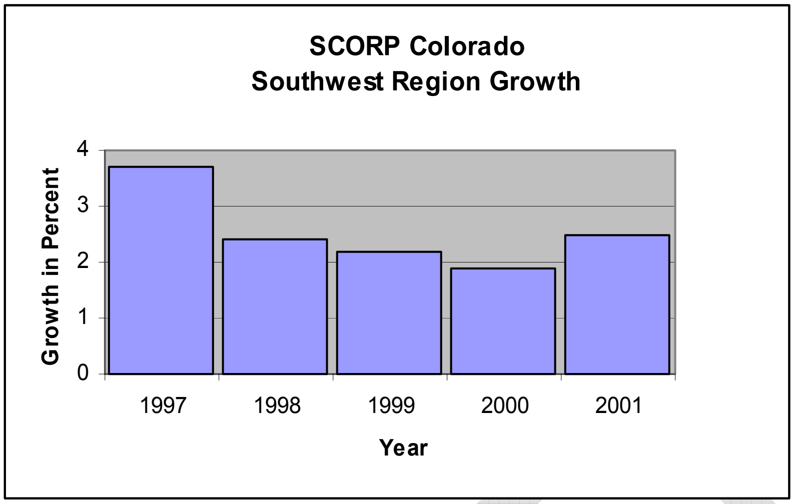


Figure 7-44 Graph showing percentage population growth in the SCORP southwest region. Graph showing percentage population growth in the SCORP southwest region, including the CLC landscape subject lands. Adapted after SCORP, 2003

To conclude, we can expect the demand for developed recreation on Forests in the ecoregion to continue to grow. Growth is likely to continue along upward trends as interest in National Forests and BLM lands increases. The increases will come as local communities continue to grow and as people look beyond the National Parks for their recreational experiences.

CLC Landscape Analysis

In this analysis we have identified 84 developed recreation sites on public lands in the CLC landscape (Fig. 7-45). These sites include campgrounds, picnic areas, trail-heads, fishing areas, observation and interpretive and boating sites (Table 7-29).

Various activities and installations around these sites are important disturbance factors. These include campground features such as fire rings, parking areas, picnic tables, toilets and disturbed congregation and tent pitching areas. Some sites include day use areas and facilities for recreational vehicle parking and facilities. In surrounding areas, trampling, cutting and foraging, littering and unsanitary wastes influence systems. Influences include vegetation loss, increased sedimentation and contamination.

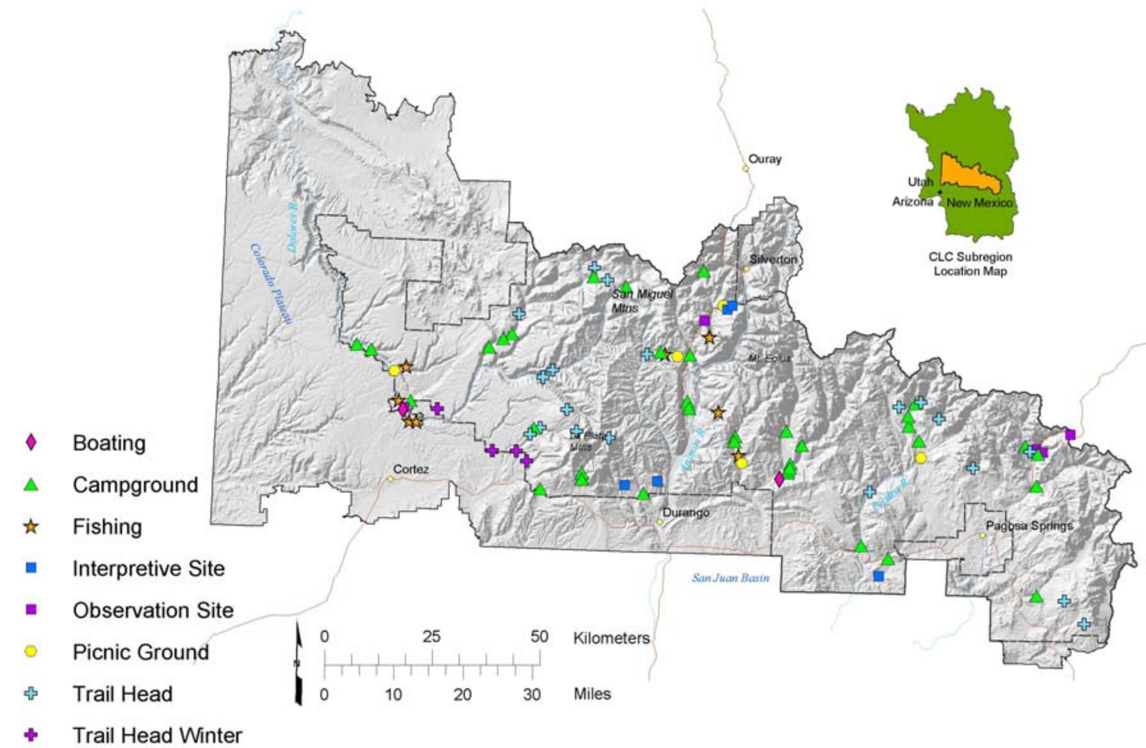


Figure 7-45 Developed recreation sites in the CLC landscape.
No sites were identified on BLM lands outside of the San Juan Forest.

Table 7-29 Developed recreation sites by type in the CLC landscape.

Site Type	Number of Sites	Pct	Sum Pct
Campground	36	42.9%	42.9%
Trail Head	19	22.6%	65.5%
Fishing	8	9.5%	75.0%
Picnic Ground	5	6.0%	81.0%
Observation Site	5	6.0%	86.9%
Interpretive Site	5	6.0%	92.9%
Trail Head Winter	4	4.8%	97.6%
Boating	2	2.4%	100.0%
	84	100.0%	

In 2000 there were approximately 1.9 million visits to the San Juan National Forest. A visit is defined as: *the entry of one person upon a national forest to participate in recreation activities for an unspecified period of time. A National Forest visit can be composed of multiple site visits.* Recent surveys indicate that about 38% percent of these visitors have primary interest in developed recreation site/activities (Table 7-30). Looking a bit closer, we find that a significant proportion of the remaining visitors, while having primary uses other than developed recreation sites in mind, still take advantage of one or more developed recreation site or activity. For example, a visitor who takes advantage of opportunities for dispersed camping (i.e. not aiming to stay at a campground) may take advantage of opportunities of hiking, or boating or fishing.

It is useful to examine and consider the average number of visitors per site over the year 2000 (Table 7-30). Daily averages approximating site visits allows one to more easily visualize the congestion, disturbance, litter and sanitation problems that may emerge at a site. For instance, daily averages at trail-heads are indicative of the need for parking, trash disposal and sanitary facilities at these sites. Conversely, absent these mitigation measures activities at these sites can strongly influence local terrestrial and aquatic systems.

Table 7-30 Developed recreation sites by type in the CLC landscape.
Percentages indicate number of visits where the given site type was of primary interest. Total visits and averages are calculated from the basic data.

Site Type	Number of Sites	Pct	Total Visits Per Year (Pct. x 1,900,000)	Average Yearly Visitation (Total/no.sites)	Average Daily Visitation (y/365)
Campground	36	5.00%	95,000	2,639	7.2
Trail Head	19	18.00%	342,000	18,000	49.3
Fishing	8	10.00%	190,000	23,750	65.1
Picnic Ground	5	2.00%	38,000	7,600	20.8
Observation Site	5		0	0	0.0
Interpretive Site	5		0	0	0.0
Trail Head Winter	4	3.00%	57,000	14,250	39.0
Boating	2		0	0	0.0
	84	38.00%	722,000	8,595	23.5

Developed Recreation Sites and Vegetation Communities

Developed recreation sites are located at elevations from 6,000 to 12,000 feet. Nearly three-quarters of all sites are found in elevations from 6,000 to 9,000 feet leaving the remainder, naturally, from 9000 to 12,000 feet. Sites in the lower elevations tend to be in dryer foothills to table/canyon-land settings. The higher sites are in upland and wetter settings. The characteristic vegetation communities reflect these basic settings associated with elevation.

Table 7-31 Number of sites per elevation class.
1000 foot elevation interval.

Number of Sites	Elevation Class	Percent of Sites	Sum of Percent
9	6000 - 7000	10.7%	10.7%
28	7000 - 8000	33.3%	44.0%
24	8000 - 9000	28.6%	72.6%
14	9000 - 10000	16.7%	89.3%
8	10000 - 11000	9.5%	98.8%
1	11000 - 12000	1.2%	100.0%
84		100.0%	

In the landscape, developed recreation sites are found in thirteen different vegetation communities (Fig. 7-36, Table 7-32). Just under 60% percent of sites fall in settings comprised of either Cool Moist Mixed Conifer, Mountain Grasslands, Spruce Fir or Ponderosa Pine. The remaining sites fall mostly in foothills and desert vegetation communities.

Table 7-32 Recreation sites in vegetation classes.

Number of Sites	Vegetation Code	Vegetation Description	Percent of Sites	Sum Of Percent
14	TMC-CM	Cool Moist Mixed Conifer	16.7%	16.7%
12	MT_GRA	Mountain Grasslands	14.3%	31.0%
11	TSF	Spruce-Fir	13.1%	44.0%
11	TPP-PP	Ponderosa Pine	13.1%	57.1%
9	TMC-WD	Warm Dry Mixed Conifer	10.7%	67.9%
8	TAA	Aspen	9.5%	77.4%
7	RIP	Riparian	8.3%	85.7%
4	MT_SHR	Mountain Shrublands	4.8%	90.5%
3	TPJ	Pinyon-Juniper	3.6%	94.0%
2	DS_GRA	Desert Grasslands	2.4%	96.4%
1	SSA	Sagebrush	1.2%	97.6%
1	NRS	Rock Soil	1.2%	98.8%
1	ALP	Alpine	1.2%	100.0%
84			100.0%	

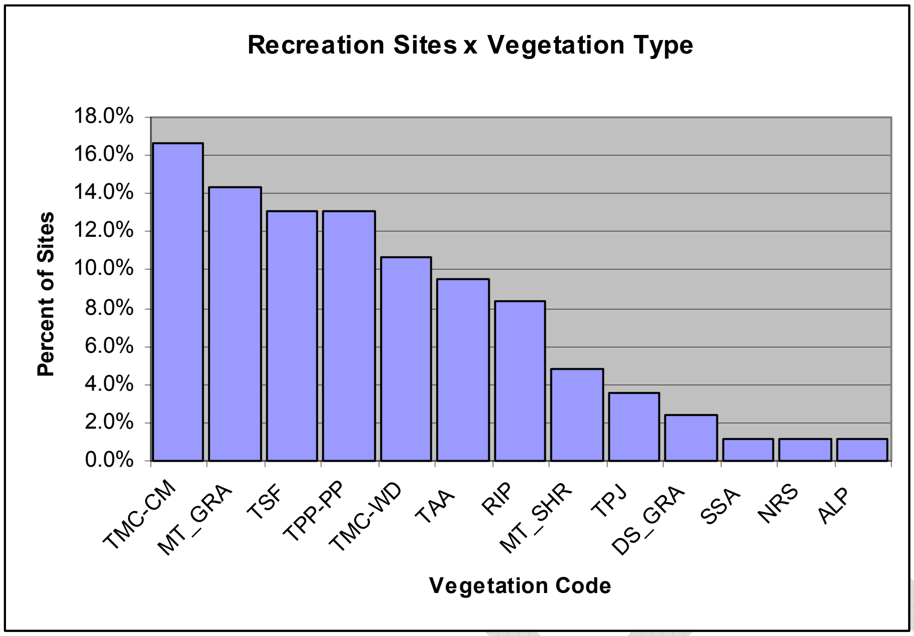


Figure 7-46 Percentage of all sites by Vegetation Class.

Estimation of Disturbance

The calculation of total area of disturbance related to developed recreation is dependent upon average site by site type. Developed campground average area has been estimated by visual examination of a subset of sites in high resolution image data. Data from that process of visual estimation has been combined with polygon area data for developed campgrounds on the Rio Grande National Forest. Furthermore, estimates, obtained by visual interpretation and averaging the Rio Grande polygon areas, are supported by reported recreation site area size in the literature (e.g. Chelan, 2001 and Spangle, 2004).

In this analysis the average site disturbance area is thus estimated at 101.6 acres. This estimated area includes highly disturbed areas such as camp sites, eating and congregation areas, paths, parking and roads. Importantly, it also includes a zone of activity and foraging where vegetation communities are subject to some level of disturbance (Table 7-33).

Table 7-33 Estimated disturbance by site type.
These estimates should be used with caution and are designed only to provide a qualitative measure of disturbance in this analysis. Identification of the need for better estimates is included in the Data Needs Section of this document.

Site Type	Estimated Area of Disturbance (Acres)
Campground	101
Trail Head	20
Fishing	20
Picnic Ground	50
Observation Site	10
Interpretive Site	10
Trail Head Winter	20
Boating	50

While estimates of developed campground average area are based in empirical data and some literature, the remaining site types in Table 5 are more crudely derived. These estimates should be used to provide insight into the potential magnitude of disturbance related to these sites.

Multiplying 101.6 acres times 36 sites yields an overall area influenced directly by campground activities at developed recreation sites of about 3,636 acres (Table 7-34). This represents just under half of all area disturbed by developed recreation activities on the CLC Landscape. Estimates show that Campgrounds combined with Trail Heads account for almost two-thirds of all disturbance area. Overall disturbance is estimated here at 4,706 acres.

Table 7-34 Estimated disturbance by site type.

Site Type	Number of Sites	Percent	Sum Percent	Total Disturbed Acres
Campground	36	42.9%	42.9%	3,636
Trail Head	19	22.6%	65.5%	380
Fishing	8	9.5%	75.0%	160
Picnic Ground	5	6.0%	81.0%	250
Observation Site	5	6.0%	86.9%	50
Interpretive Site	5	6.0%	92.9%	50
Trail Head Winter	4	4.8%	97.6%	80
Boating	2	2.4%	100.0%	100
	84	100.0%		4,706

Almost half, 41 of 84 sites are found in four vegetation classes. These four classes are Warm Dry Mixed Conifer, Riparian, Cool Moist Mixed Conifer and Ponderosa Pine (Table 7-35). Using approximated disturbance area figures, sites in these four classes disturb from 0.7830% down to 0.230% percent of the total area for each respective class. The remaining 43 sites are distributed among nine vegetation classes.

Table 7-35 Disturbance ranked as a percentage of total vegetation class in the landscape. At 0.783% percent, Warm Dry Mixed Conifer has the greatest relative degree of disturbance.

Vegetation Code	Vegetation Description	Number of Sites	Disturbance Acres	Vegetation Class Landscape Acres	Percent Of Landscape
TMC-WD	Warm Dry Mixed Conifer	9	747	95,351	0.783%
RIP	Riparian	7	383	74,621	0.513%
TMC-CM	Cool Moist Mixed Conifer	13	715	199,753	0.358%
TPP-PP	Ponderosa Pine	12	949	412,854	0.230%
MT_GRA	Mountain Grasslands	12	655	301,988	0.217%
TAA	Aspen	8	393	347,277	0.113%
TSF	Spruce-Fir	11	341	513,250	0.066%
MT_SHR	Mountain Shrublands	4	242	450,394	0.054%
DS_GRA	Desert Grasslands	2	151	303,690	0.050%
TPJ	Pinyon-Juniper	3	80	446,865	0.018%
NRS	Rock Soil	1	20	111,894	0.018%
ALP	Alpine	1	20	186,871	0.011%
SSA	Sagebrush	1	10	208,115	0.005%
		84	4706		

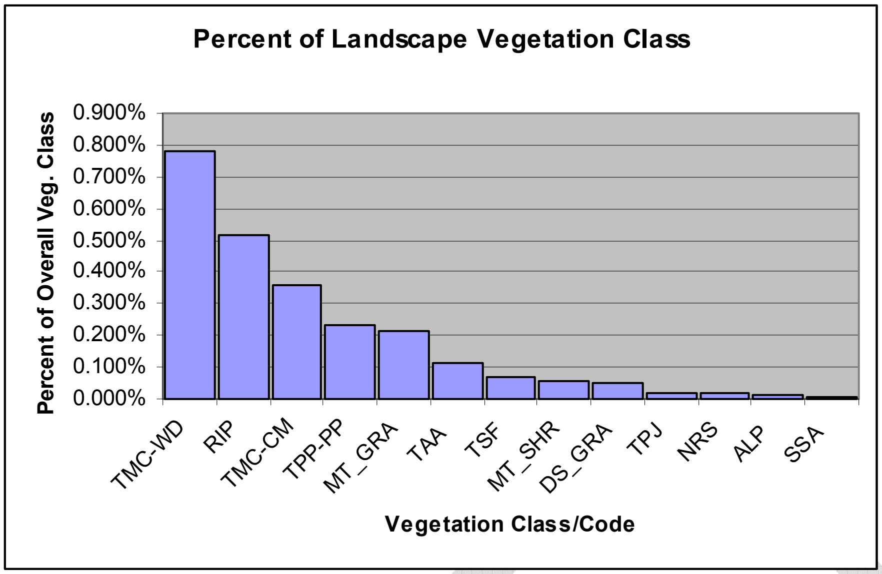


Figure 7-47 Vegetation Classes organized by percentage of landscape/veg. class area. The top three classes are shown in Figure 7-48 below. TMC-WD - Warm Dry Mixed Conifer, RIP - Riparian, TMC-CM - Cool Moist Mixed Conifer and TPP-PP - Ponderosa Pine.

The vegetation classes: Warm Dry Mixed Conifer, Riparian, Cool Moist Mixed Conifer and Ponderosa Pine are found at middle/foot hills elevations and settings (Fig. 7-47 and Fig. 7-48). The association of sites with these communities is evident and is even notable in the distribution of many sites that do not fall directly inside one of these four classes.

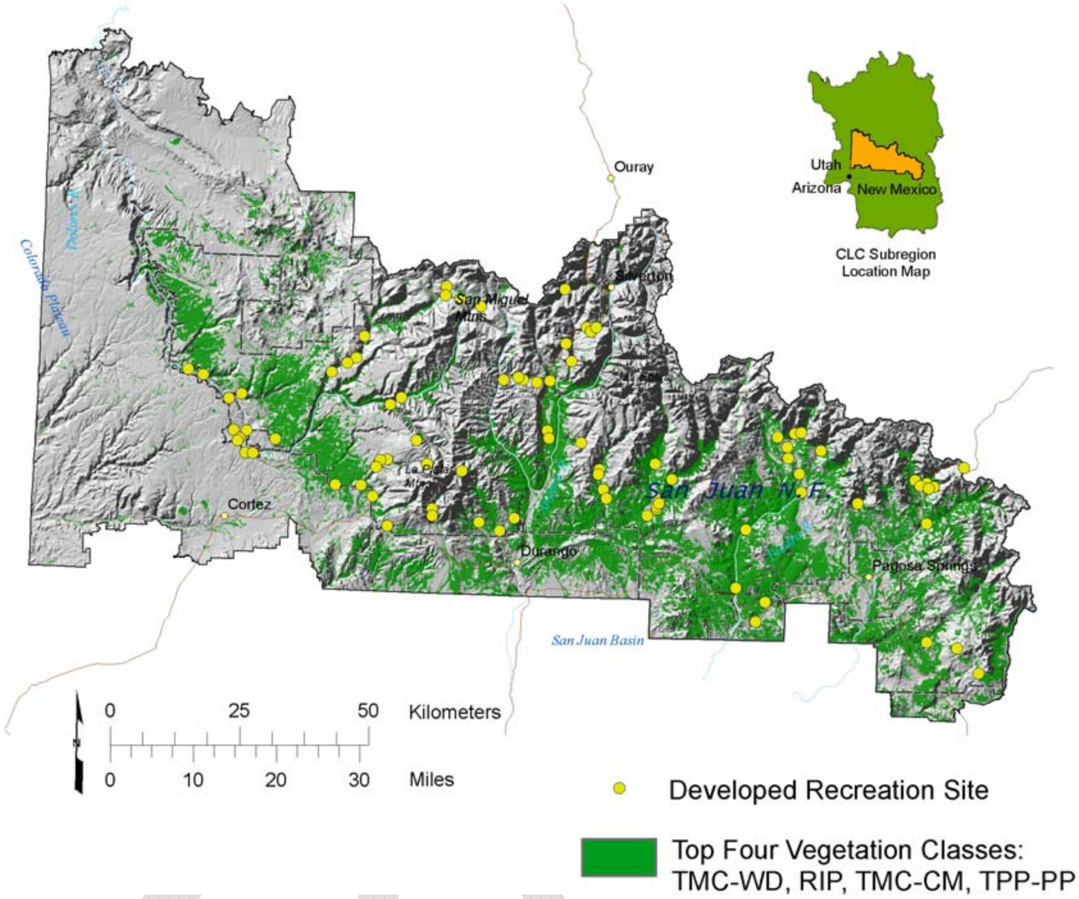


Figure 7-48 Distribution of the four top vegetation classes that contain just under half of all sites.
These four classes have relatively higher disturbance ratios than other vegetation classes in the landscape.

Data Needs

Better estimates of developed recreation site disturbed areas are important. Current data are point location only and provide little insight into geometry and extent of use areas.

Developed recreation databases should be standardized between agencies. Inventories of privately owned and managed developed recreation sites would add to our understanding of recreation.

Future analyses would be more complete with more robust measures of disturbance resulting from developed recreation such as boating ramps, picnic areas etc.

Future analyses would be enhanced by a comprehensive study of seasonal use patterns and duration.

Analyses should include recreational sites on private lands.

Bibliography

Canyonlands, 2005. Canyonlands National Park Visitor Study. Online: Address to be recovered.

Chelan County PUD. 2001. Recreation Resources Inventory Summary Report. Preliminary Discussion Draft. Rocky Reach Hydroelectric Project. FERC Project No. 2145. Online: http://www.chelanpud.org/rr_relicense/study/reports/2731_1.pdf

Cordell, Ken H. and Gregory R. Super. 2000. Trends in Americans' Outdoor Recreation. In: Trends in Outdoor Recreation and Leisure and Tourism. Eds. W.C. Gartner and D. W. Lime. CAB International.

Gramann, James H. 2003. Visitation Forecasting and Predicting Use of NPS Parks and Visitor Centers: Focus Group Report.

Meldrum, Bret H. Margaret A. Littlejohn and Steven J. Hollenhorst. 2004. Arches National Park Visitor Study, Spring 2003. Visitor Services Project, Report 150. July, 2004. Online: http://www.psu.uidaho.edu/files/vsp/reports/150ARCH_rept.pdf

NPS, 2005. Public Use Statistics Office, NPS Visitation Database Reports. U.S. Department of the Interior. Online: http://www2.nature.nps.gov/stats/

SCORP. 2003. Statewide Comprehensive Outdoor Recreation Plan, Colorado's Outdoor Recreation Future. Strategies for Colorado's Outdoors Heritage. State of Colorado State Parks. Online: http://parks.state.co.us/scorp/index.asp

Spangle, Steven L. 2004. Memo to E.J. Zieroth, Forest Supervisor, Apache-Sitgreaves National Forest. Online: http://arizonaes.fws.gov/Documents/BiologicalOpinions/2004/040107_BigLakeCampground.pdf

USDA, 1992. Record of Decision and EIS, Land and Resource Management Plan. San Juan National Forest. USDA Forest Service, Region 2. On file at Region 2 Offices.

USDA. 2001 National Visitor Use Monitoring Results, August 2001. San Juan National Forest. USDA Forest Service, Region 2. Online: http://www.fs.fed.us/recreation/programs/nvum/reports/year1/R2_San_juan_final.htm#_Toc524932697

Dispersed Recreation- CLC Landscape Scale- San Juan National Forest

Page intentionally blank.

Minerals, Oil and Gas Exploration and Development (Module 4G)

Influence of Mineral Exploration and Development

Mineral Development

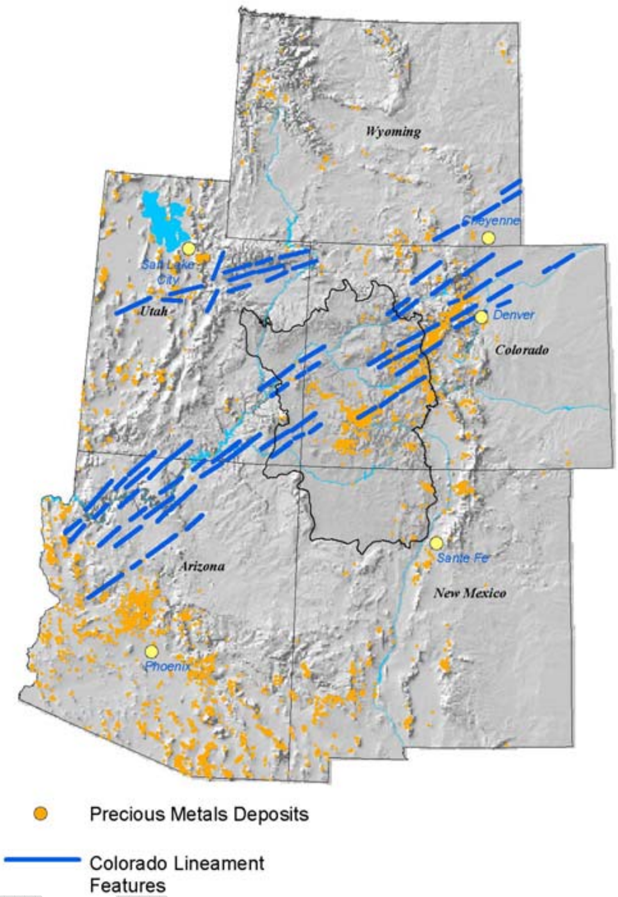
Historically, mineral exploration and development have played key roles in defining the character and landscape pattern in the Western U.S and importantly, within the ARW/CLC assessment area. Over the 19th century, prospectors combed the deserts and mountains in search of gold, silver and other precious metals (Murray, 1980; Preston, 2004). At the same time, workers developed coal, providing important energy resource to mining and other industries, agriculture and domestic needs. Deposits of limestone and aggregates were developed to build railroads, roads and provide a source for concrete along with clay for brick and ceramics. Beginning in the 1950s and 1960s, energy resources in the area, including coal, oil, gas, coal-methane and uranium deposits became the dominant mineral commodities produced in the region. Today, coal, oil, gas and common variety minerals (e.g. sand and gravel) development continues to be important in the sub-region and the surrounding western states.

The sub-region is particularly well endowed with mineral resources. World class deposits of precious and base metals occur along a northeast to southwest trend from Aspen through Silverton and Telluride southward to the La Plata Mountains. This trend follows a larger regional geologic trend called the *Colorado Lineament* (Warner, 1980). The Colorado Lineament is correlated regionally with important mineral deposits in Arizona, Colorado and Utah (Figure 7-49). These deposits include massive sulfides, vein and metallic replacement deposits (U.S.G.S, 2004). These deposits are largely associated with Tertiary volcanic centers. Notably, the volcanic center and mineralized areas of Silverton lay at the intersection of the Colorado Lineament and the northwest to southeast trend formed by the axis of the Uncompahgre Mountains (Ellingson, 1996).

Historical metallic mining areas and towns strongly contribute to the character of the subregion. Abandoned mines, mills and tailings are common landscape features lending themselves a sense of history and place. At the same time, abandoned hardrock mines pose a significant threat to water quality throughout the western U.S. (Schnitzer and Roberts, 2004).

Important uranium deposits of the Urvan mineral belt, along the Dolores River hosts important sandstone based deposits of uranium. These deposits were the object of significant development activity in the 1950s and 1960s leading to an economic boom at that time. The boom influenced the growth of important Colorado Plateau communities including Cortez, Durango and Grand Junction and Moab.

Figure 7-49 Regional Colorado Lineament features
Regional Colorado Lineament features adapted after Warner, 1980 superimposed on locations of precious metals deposits. The trend also corresponds to important structural controls for non-metallic deposits.



Oil and Gas deposits occur in sedimentary basins throughout the sub-region. Development of these deposits began early in the 20th century with discoveries in the Paradox Basin. World class gas deposits are currently in development in the San Juan Basin. Significant deposits occur also in the Piceance Creek and Paradox basins. Outside these basins wells have been drilled in plays along the Bookcliffs and Gunnison River valley. More than 30,000 wells have been drilled since the first wells appeared in the region. Disturbance from drilling, facilities development and road building is an important

Coal deposits occur in late Cretaceous and Tertiary rocks throughout structural basins in the subregion. These deposits appear near the surface along basin margins and have been exploited historically throughout the subregion. Today, large scale mining operations are located in deposits of the Bookcliffs, Upper

Gunnison River valley, Nucla-Naturita, Durango and areas west of Farmington New Mexico.

Today, growth in local communities drives up demand for common variety mineral materials used for road building and building construction. Sand and gravel are developed throughout the region along with quarries for building stone.

In the following summary we draw upon various data sources to characterize patterns of mineral development activity in the subregion and ARW landscape. The U.S. Geological Survey MILS/MAS (Causey, 1998) database provides important insights into the current and historical distribution of mineral sites of all types in the region. These data are augmented by BLM mining claim records to show mining claim distributions as a measure of current interest in locatable mineral development.

Oil and gas well location data sets from the state of Colorado, the state of Utah and BLM indicate those areas most significant for development and strongly influenced by development activity. Finally, coal mining data sets are used to illustrate locations of currently active mining. In each case, the character of mineral activity is illustrated and further developed by summary by GAP vegetation class and 4th level HUB.

Distribution of Mining Sites – Recent and Historic

Within the subregion there were 8,968 mineral development sites recorded in the MAS/MILS database as of 1997. These may be categorized into four status classes. These four are:

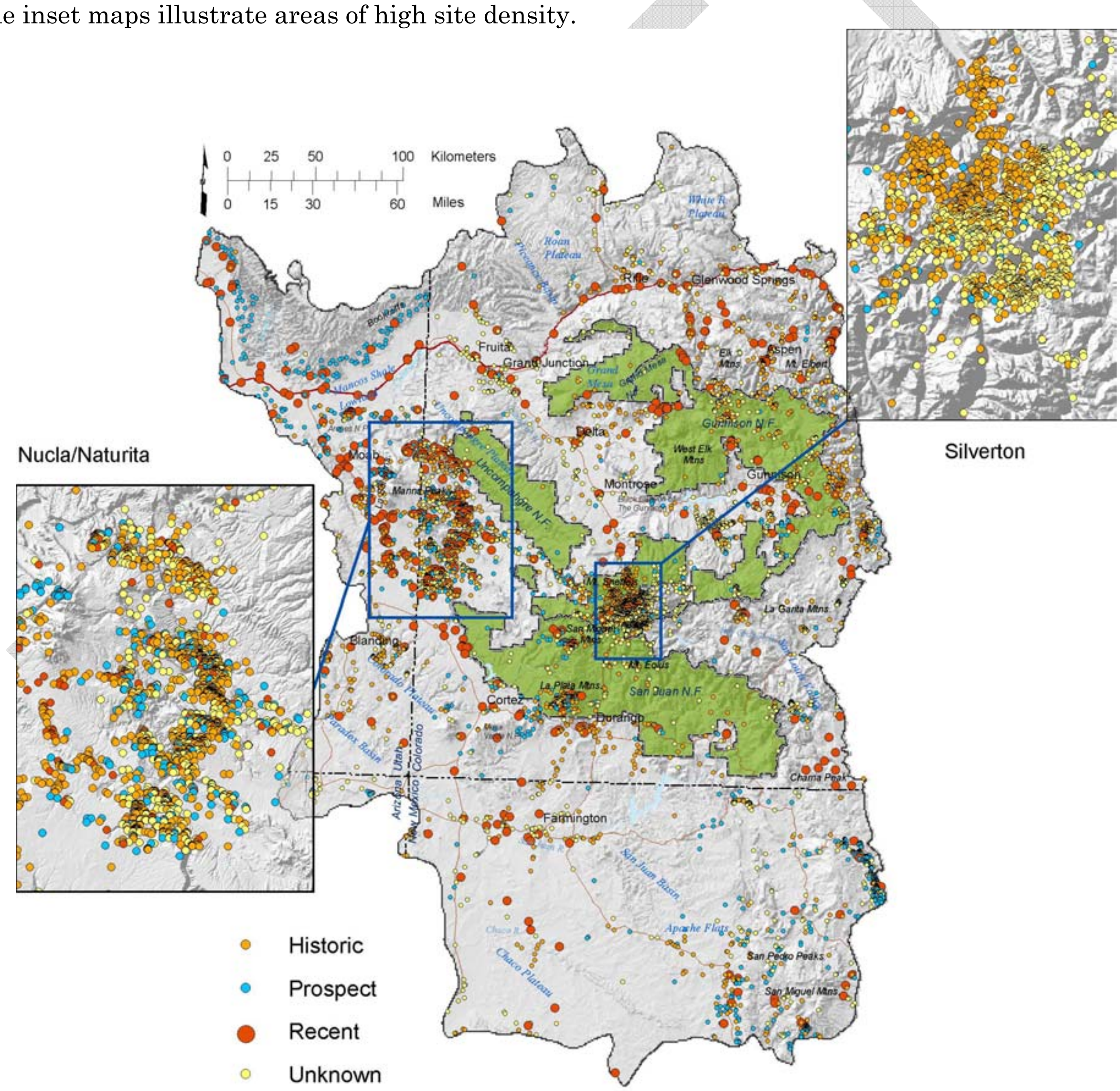
1) *Historic* – indicative of mineral development in the past;
2) *Prospect* – a site with prospecting but no development;
3) *Recent* – indicating active development currently or recently;
4) *Unknown* – indicating the possibility of prospecting and/or development. Likely to be historical.

Of the 8,968 sites, nearly 50% may be considered to be *Historic* (Table 7-36). Nearly fifteen percent are classified as *Prospects*. Thirty-three percent are of *Unknown* status leaving just under six percent are classified as *recent*. Historical sites tend to cluster in upland areas known for precious metals mining and in the Nucla-Naturita uranium mining areas (Figure 7-50). Recent sites are widely distributed and include a departure from the precious metals districts to areas of uranium, coal, mineral materials and base metals.

Table 7-36 Subregion mineral sites status as of 1997.
Adapted after U.S. Geological Survey MILS/MAS data (Causey, 1998).

Status	Number	Percent
Historic	4211	47.0%
Unknown	2961	33.0%
Prospect	1290	14.4%
Recent	506	5.6%
	8,968	100.0%

Figure 7-50 MILS/MAS mineral sites *status* as of 1997 in the subregion
MILS/MAS mineral sites status as of 1997 in the subregion (Causey, 1998).
The inset maps illustrate areas of high site density.



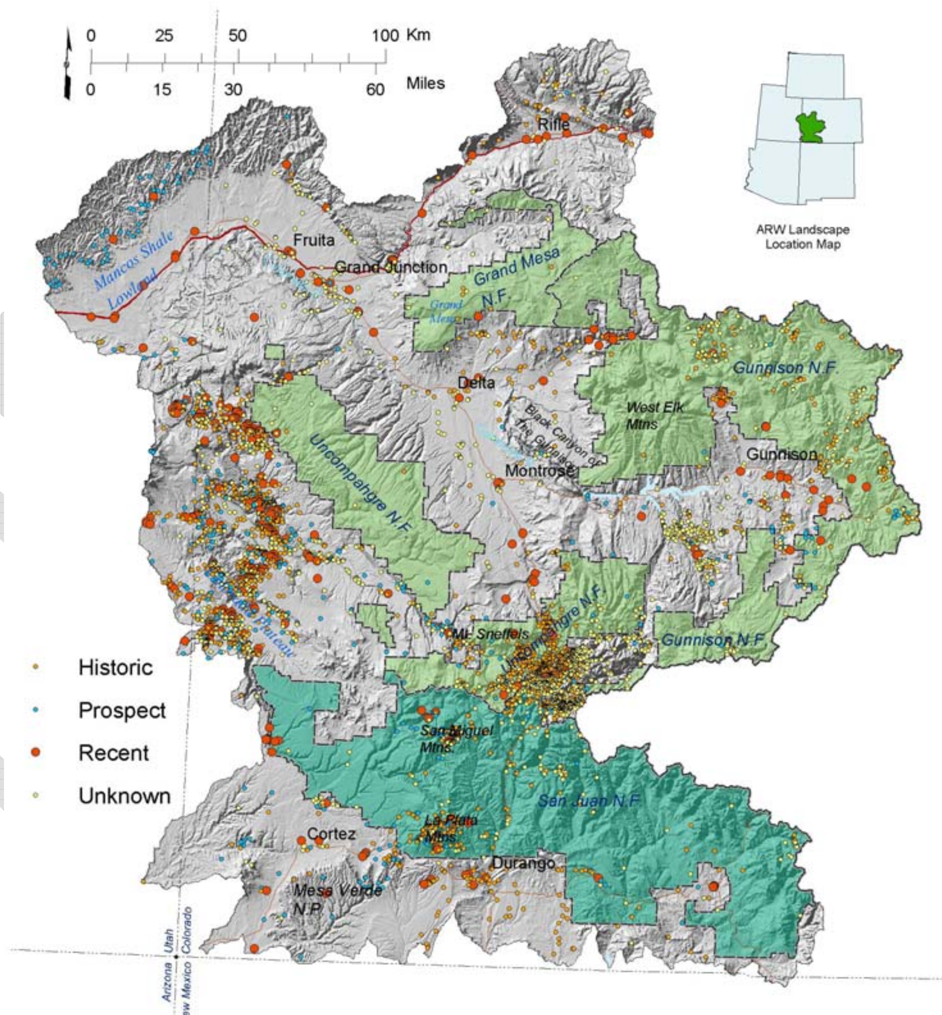
Within the ARW landscape there are 6,129 mineral development sites recorded in the MAS/MILS database. The distribution of theses by status is similar to the distribution of sites over all of the subregion (Table 7-37, Figure 3).

Table 7-37 ARW Landscape mineral sites status as of 1997.
Adapted after U.S. Geological Survey MILS/MAS data (Causey, 1998).

Status	Number	Percent
Historic	2923	47.7%
Unknown	2290	37.4%
Prospect	647	10.6%
Recent	269	4.4%
	6,129	100.0%

Figure 7-51 Mineral sites by *status* in the ARW landscape as of 1997 (Causey, 1998).

There are 6,129 sites in the ARW landscape.



In the subregion, most mineral sites are located outside of the San Juan and GMUG National Forest areas. About 10% of sites are located in the GMUG and about 9% are in the San Juan Forest. The 26 sites located in the Forests and classified as "*recent*" constitute less than 1% of the 8,968 in the subregion (Table 7-38).

Table 7-38 Distribution of 8,968 mineral sites inside and outside the San Juan and GMUG by status as of 1997.
Adapted after U.S. Geological Survey MILS/MAS data (Causey, 1998).

Forest	Status	Number	Pct
Other	Historic	3258	36.3%
	Prospect	1124	12.5%
	Recent	480	5.4%
	Unknown	2405	26.8%
	SubTotal:	7267	81.0%
GMUG	Historic	603	6.7%
	Prospect	38	0.4%
	Recent	14	0.2%
	Unknown	271	3.0%
	SubTotal:	926	10.3%
San Juan	Historic	350	3.9%
	Prospect	128	1.4%
	Recent	12	0.1%
	Unknown	285	3.2%
	SubTotal:	775	8.6%
	Total:	8,968	100.0%

Nearly 90% percent of the 506 sites are classified as recent and are comprised of common variety minerals, uranium, coal or base metal (Table 7-39). Common variety minerals include sand, gravel and building stone and development sites largely follow major roads. The higher proportions represented by both uranium and coal are indicative of the importance of energy development in the region today. Most uranium sites are located in the Nucla-Naturita area west of the Uncompahgre Mountains. Of the 56 coal mining sites, many constitute major operations with significant disturbance of surface and subsurface systems. The most significant of these are discussed further below.

Table 7-39 The principal commodities for sites classified as "*Recent*" across the subregion.
Common Variety minerals include sand and gravel and building stone.

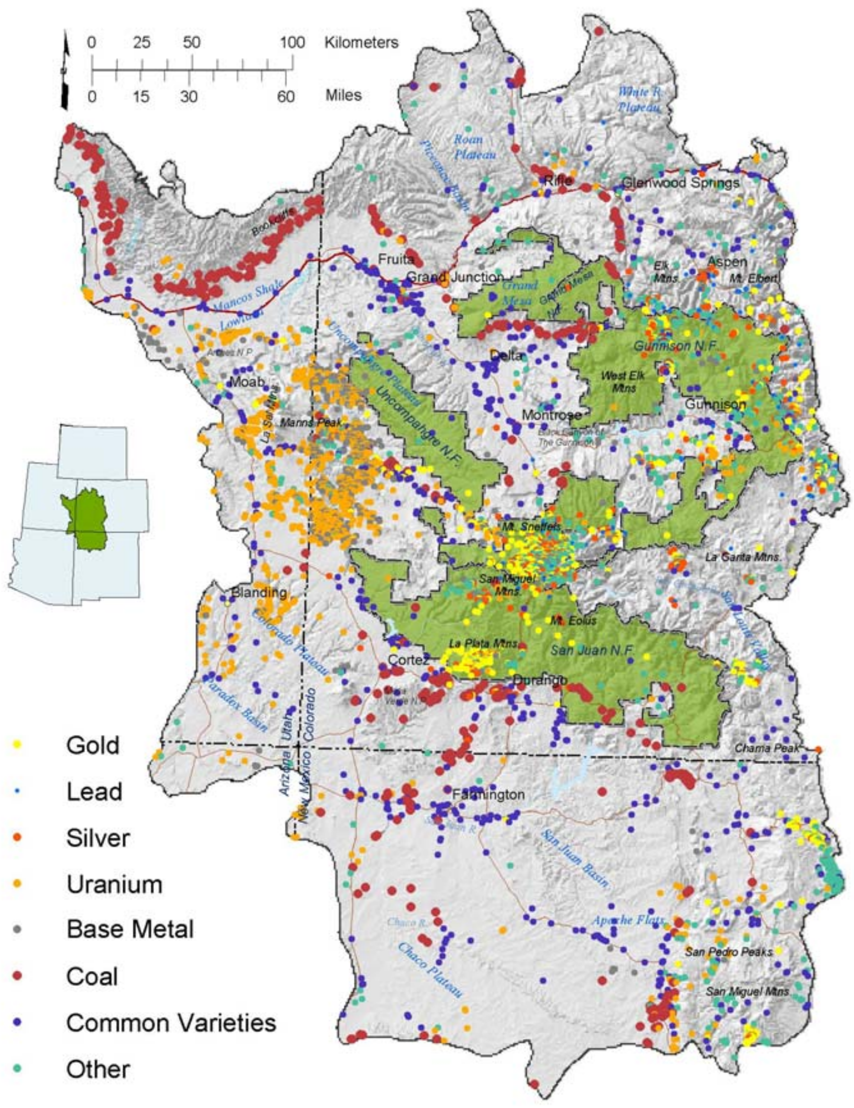
Category	Number	Pct	SumPct
Common Variety	185	36.6%	36.6%
Uranium	152	30.0%	66.6%
Coal	56	11.1%	77.7%
BaseMetal	54	10.7%	88.3%
Other	39	7.7%	96.0%
Unknown	7	1.4%	97.4%
Silver	6	1.2%	98.6%
Lead	4	0.8%	99.4%
Gold	3	0.6%	100.0%
	506	100.0%	

Of all the MAS/MILS sites in the subregion, regardless of status, nearly 90 percent fall within one of seven commodity categories. These seven include the four comprising the *"recent"* class plus an additional three (Table 7-40). Those three include: gold, silver and lead and have largely been the object of development in historical mining districts (Figure 7-52).

Table 7-40 Overall subregion principal mineral commodities.

Category	Number	Pct	SumPct
Uranium	1992	27.1%	27.1%
BaseMetal	1066	14.5%	41.5%
Gold	922	12.5%	54.1%
CommonVariety	768	10.4%	64.5%
Silver	626	8.5%	73.0%
Coal	535	7.3%	80.3%
Lead	427	5.8%	86.1%
Other	1025	13.9%	100.0%
	7361	100.0%	
Unknown	1607		
Total	8,968		

Figure 7-52 Seven commodity classes comprise nearly 90% of all mineral sites in the subregion (Causey, 1998).
The trend formed by coal sites reveals the margins of principal sedimentary basins.



Subregion Scale GAP Vegetation Analysis

Of the 8,968 mine sites in the subregion, just over 90% of mineral sites fall in five of twenty vegetation classes (Table 7-41). Of these, nearly 50% fall within two vegetation classes. These two vegetation classes include Pinyon-juniper and spruce-fir (Figure 7-53). The sites in Pinyon-juniper vegetation class are largely located outside of National Forests. Many of these are found on BLM and patented claims in the Nucla-Naturita area. Conversely, sites in the Spruce-fir type are strongly correlated to upland areas, much of which is in the Spruce-fir vegetation class.

Mining in both areas is largely historic. Disturbance there is generally less important that mine drainage from abandoned mines.

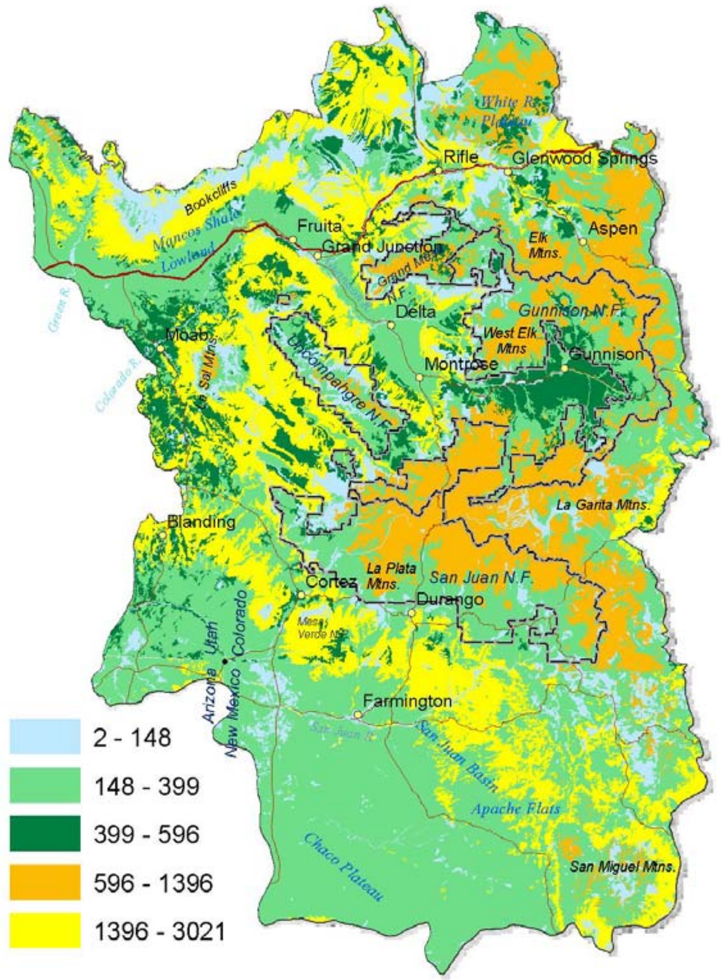
Table 7-41 Distribution of 8,968 historic and recent mineral sites inside and outside the San Juan and GMUG by vegetation class.

Adapted after U.S. Geological Survey MILS/MAS data. Note: Historic includes prospects and Recent includes Unknown

	Historic		Recent				
Veg Class	**Inside NF**	**Outside NF**	**Inside NF**	**Outside NF**	**Total**	**Pct**	**SumPct**
pinyon -juniper	6	2045	12	958	3,021	33.7%	33.7%
spruce - fir	525	309	272	290	1,396	15.6%	49.3%
alpine	347	285	156	430	1,218	13.6%	62.8%
sagebrush	16	261	13	306	596	6.6%	69.5%
aspen	115	161	54	69	399	4.4%	73.9%
desert shrub	0	196	0	179	375	4.2%	78.1%
ponderosa pine	32	206	19	76	333	3.7%	81.8%
crops	4	144	6	165	319	3.6%	85.4%
mixed conifer	19	142	19	125	305	3.4%	88.8%
desert grassland	0	152	0	60	212	2.4%	91.1%
lodgepole pine	40	105	23	33	201	2.2%	93.4%
deciduous oak	9	105	5	29	148	1.7%	95.0%
mountain grassland	0	93	2	31	126	1.4%	96.4%
urban	0	61	0	41	102	1.1%	97.6%
mountain shrubland	1	66	0	27	94	1.0%	98.6%
barren	2	26	1	30	59	0.7%	99.3%
woody riparian/wetland	0	13	0	14	27	0.3%	99.6%
water	3	4	0	12	19	0.2%	99.8%
greasewood	0	5	0	7	12	0.1%	99.9%
Not Classified	1	1	1	1	4	0.0%	100.0%
herbaceous riparian/wetland	0	1	0	1	2	0.0%	100.0%
Total:	1,120	4,381	583	2,884	8,968		
Pct:	12.5%	48.9%	6.5%	32.2%		100.0%	
Sum by Period	5,501		3,467				
Pct by Period:	61.3%		38.7%				

Figure 7-53 Mineral site counts by vegetation class.

Over half of the sites are Pinyon-Juniper and Spruce-fir vegclasses. Forest lands comprise significant proportions the Spruce-fir vegetation type. Mine sites in upland areas including Silverton and the La Plata Mountains are strongly correlated to Spruce-fir, aspen and alpine vegetation classes.



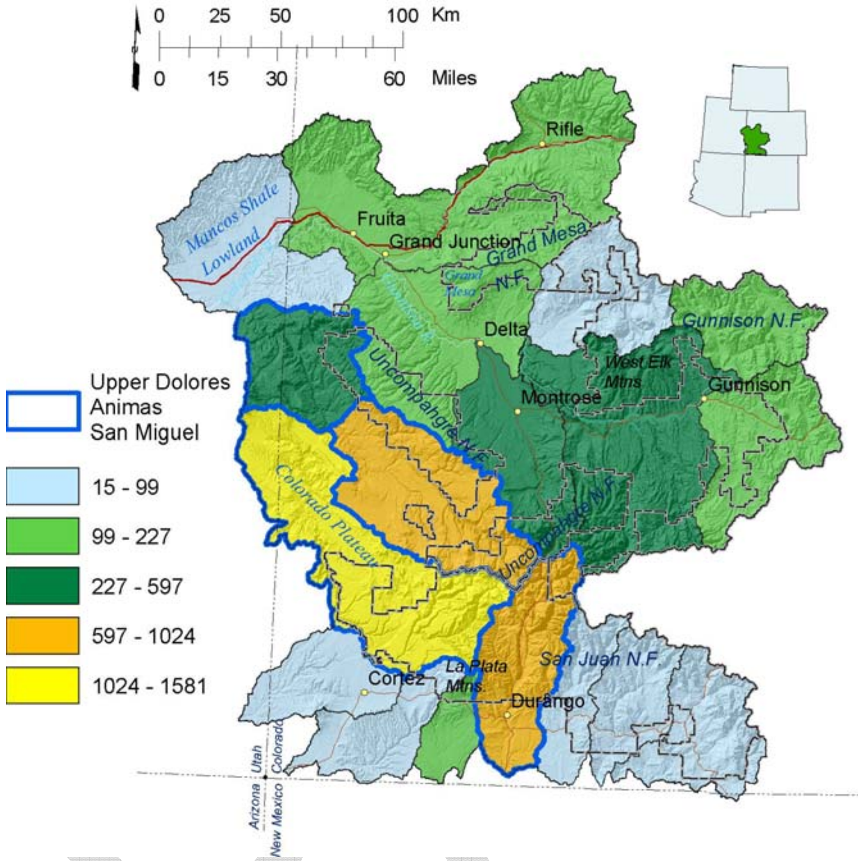
Landscape Scale 4th Level HUB Watershed Analysis

Of the 6,129 mine sites in the ARW Landscape, just over 90% of mineral sites fall in nine of seventeen fourth level watersheds (Table 7-42). Of these, over 50% fall within three watersheds. These three watersheds are the Upper Dolores, the Animas and San Miguel watersheds (Figure 7-54). Significantly, these watersheds host important proportions of both historic and recent mining operations. The Animas watershed is recognized for the impact historic precious metals sites have had and are having upon local ecosystems and especially local and downstream water quality and hydroecology (Schnitzer and Roberts, 2004; Robinson, 2002). More recent uranium mining impacts the Upper Dolores and San Miguel watersheds. Moreover, Forest Lands comprise majority proportions of the headwaters of these watersheds.

Table 7-42 Distribution of 6,129 historic and recent mineral sites inside and outside the San Juan and GMUG by HUB4 as of 1997.
Adapted after U.S. Geological Survey MILS/MAS data. Note: Historic includes prospects and Recent includes Unknown.

	Historic		Recent				
4th Level Watershed Name	Inside NF	Outside NF	Inside NF	Outside NF	Total	Pct	SumPct
Upper Dolores	187	901	99	394	1,581	25.8%	25.8%
Animas. Colorado	128	278	155	463	1,024	16.7%	42.5%
San Miguel	169	438	70	167	844	13.8%	56.3%
Lower Dolores	5	350	6	236	597	9.7%	66.0%
Upper Gunnison	9	93	51	328	481	7.8%	73.9%
Uncompahgre	314	55	27	54	450	7.3%	81.2%
Colorado Headwaters-Plateau	3	50	3	171	227	3.7%	84.9%
Tomichi	70	38	38	41	187	3.1%	88.0%
Middle San Juan	90	37	12	4	143	2.3%	90.3%
East-Taylor	64	0	69	0	133	2.2%	92.5%
Lower Gunnison	3	40	6	68	117	1.9%	94.4%
Mancos	48	40	4	7	99	1.6%	96.0%
Upper San Juan	18	28	22	6	74	1.2%	97.2%
Westwater Canyon	0	55	0	15	70	1.1%	98.3%
North Fork Gunnison	4	24	15	15	58	0.9%	99.3%
Mcelmo	0	21	0	8	29	0.5%	99.8%
Piedra	7	3	5	0	15	0.2%	100.0%
Total:	1,119	2,451	582	1,977	6,129		
Pct:	18.3%	40.0%	9.5%	32.3%		100.0%	
Sum by Period	3,570		2,559				
Pct by Period:	58.2%		41.8%				

Figure 7-54 Mineral site counts by fourth level watershed.
Over half of the sites are located in the Upper Dolores, Animas and San Miguel watersheds. Forest lands comprise significant proportions of these watersheds. At the same time, large numbers of sites are located, recently and historically, on Forest lands.



Mining Claims

Mineral development of precious metals, uranium, base metals and some classes of common variety minerals are administered as *"Locatable Minerals"* and are subject to mining claim as lode or placer. Excluding coal and sand and gravel, most of the 8,968 sites in the subregion may be associated with the exploration or development of locatable minerals. Where these minerals occur on federal land, they are thus subject to *"claim or location"* as lode or placer mining claims.

The distribution of mining claims on federal lands provides important insight into mineral potential. More significantly, areas with ongoing active claims may be considered areas of high interest and likely development under reasonably foreseeable economic conditions. It is important to management planning to call these areas to the attention to the public and land managers (Figs. 7-55 and 7-56).

Figure 7-55 Areas of open lode and placer mining claims in the subregion. These are areas of ongoing activity and high potential for future locatable mineral development activity as mining economics change. Naturally, these areas correspond to mine site maps above but are indicative of present and future interest while much the mine site maps are often indicative of historic interest only.

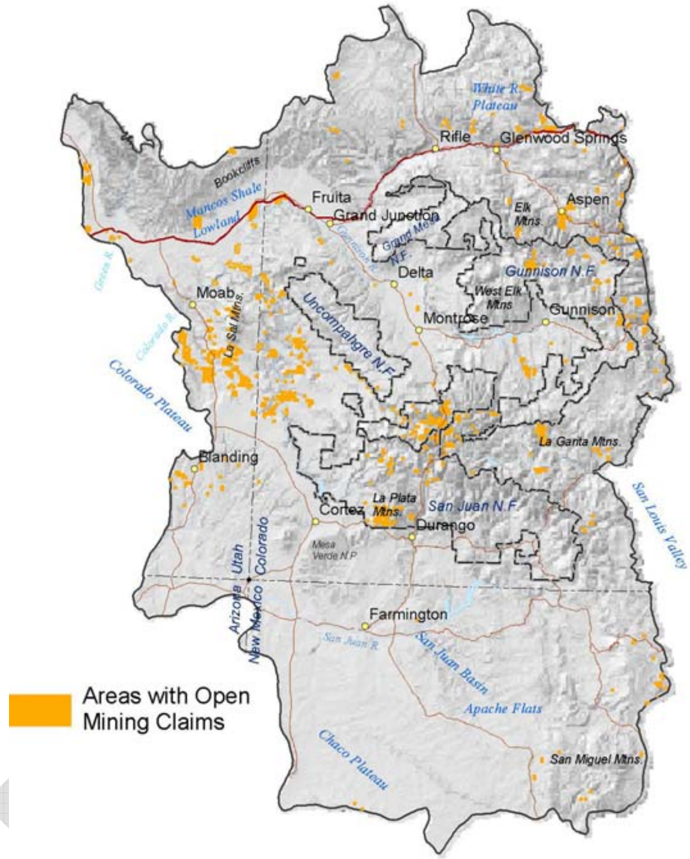
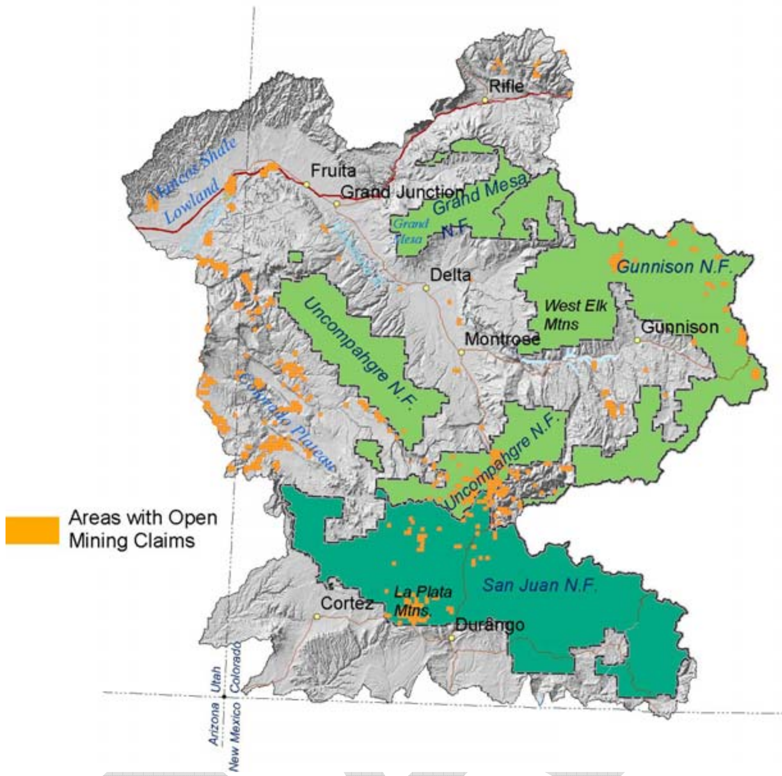


Figure 7-56 Areas of open lode and placer mining claims in the ARW Landscape.
In general, these claims are located on either BLM or Forest lands, with some minor exceptions.



Oil and Gas

Oil and gas development in the subregion has been ongoing since early in the last century. From the 1950s onward, development has surged and retreated periodically with new discoveries combined with swings in prices, and regulatory framework. In the 1980s, development of fields in Utah and Colorado surged. More recently, the discovery of coal-bed methane gas in the San Juan Basin has lead to a surge in development there and development in these fields is expected to continue in the San Juan Basin over the next 20 years (Engler, 2001).

According to available well data, there are 35,346 wells in the subregion and 8,870 wells in the ARW landscape (Table 7-43). The available data has been obtained the State of Colorado (COGCC. 2004) Utah (UTOG, 2004) and the Bureau of Land Management, New Mexico State Office (Ongard, 2001). This analysis does not include the relatively small proportion of wells located in Arizona due to data availability.

The Colorado, Utah and New Mexico data have been merged and clipped to the subregion and the ARW Landscape. Variable data standards and coding between states do not allow complete merging of all data and attributes. However, we have approximated well status to obtain a merged data set that may approximate the location of currently active wells contrasted with a backdrop of historic well sites.

Well drilling, development, production and abandonment all influence water quality and ecological integrity. Assuming about 3 acres of disturbance per well pad in developed fields (Engler, 2001) we can approximate overall disturbance of about 106,038 acres in the subregion and 26,610 acres in the ARW landscape.

Pipelines and road access adds significantly to this estimate of disturbed area.

Additionally, ground and surface water pollution can be caused by fracturing along with disposal of drilling fluids and produced water. Water loss by production of coal bed methane can be significantly higher than conventional drilling (USGS 2000). Altogether, exploration, development and production can lead to the introduction of noxious weeds, invasive species, changes in animal foraging, breeding and migration behaviors.

The following summaries and maps illustrate the distribution of wells in the subregion and landscape and current levels of activity.

Table 7-43 There are 35,346 wells in the subregion.

Just over 1 percent of these are located in the GMUG or San Juan . In the ARW Landscape there are 8,870 wells and the proportion of wells in the Forests to the total is slightly higher at almost 5% percent.

Subregion:

Forest	Number of Wells	Pct
Other	34,906	98.8%
GMUG	110	0.3%
San Juan	330	0.9%
	35,346	100.0%

ARW Landscape:

Forest	Number of Wells	Pct
Other	8,430	95.0%
GMUG	110	1.2%
San Juan	330	3.7%
	8,870	100.0%

In the subregion and the ARW landscape more than 90% are located on BLM, Private or Tribal lands (Table 7-44 and 7-45). As a consequence many of the influences resulting from drilling and production occur downstream from Forest Lands. Decisions affecting Forest lands directly may not directly influence or mitigate these influences.

Table 7-44 Distribution of 35,346 wells in the subregion by ownership.

Over 90% percent are located on BLM, Private or Tribal lands. Notably, less than 3% percent are located on Forest Lands.

Ownership	Num Wells	Pct	Sum Pct
BLM	14,891	42.13%	42.13%
Private	10,123	28.64%	70.77%
Tribal	7,274	20.58%	91.35%
State	1,746	4.94%	96.29%
USFS	1,037	2.93%	99.22%
BOR	188	0.53%	99.75%
DOD	80	0.23%	99.98%
NPS	7	0.02%	100.00%
	35,346	100.00%	

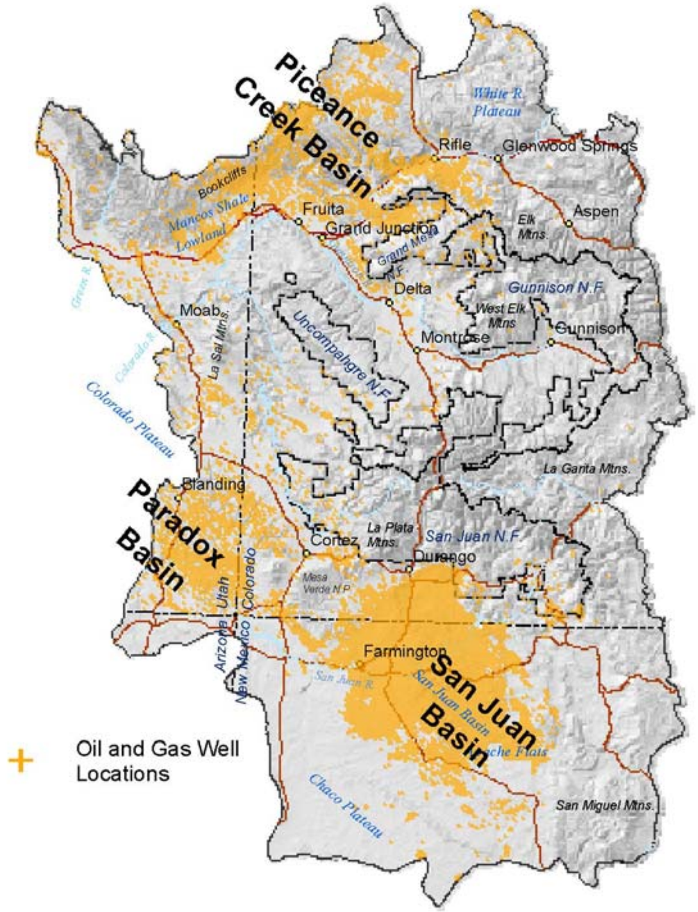
Table 7-45 Distribution of 8,870 wells in the ARW landscape by ownership.

(The overall distribution of wells in the landscape is similar to the distribution in the subregion.)

Ownership	Num Wells	Pct	Sum Pct
Private	4,859	54.78%	54.78%
BLM	2,493	28.11%	82.89%
Tribal	825	9.30%	92.19%
State	319	3.60%	95.78%
USFS	295	3.33%	99.11%
DOD	79	0.89%	100.00%
BOR	0	0.00%	100.00%
NPS	0	0	100.00%
	8,870	100.00%	

In the subregion, wells are largely concentrated within three major geologic basins. These three include the Piceance Creek, Paradox and San Juan Basins. Smaller fields are found within sedimentary rocks along local stratigraphic and structural trends beyond basin margins (Figure 7-57).

Figure 7-57 The 35,346 wells in the subregion are largely concentrated in Piceance Creek, San Juan and Paradox basins.



Today in the subregion, there are 7704 active or producing wells (Table 7-46) representing about 22% percent of the 35,346 wells in the subregion (Figure 7-58). In the ARW landscape, there are 3,427 active or producing wells (Table 7-47) representing almost 40% percent of the 8,870 wells in the landscape. Notably, areas with greatest well density and intensity of disturbance are on private and tribal lands. Well density and cumulative disturbance from exploration, drilling and production are most intense in the San Juan Basin.

Table 7-46 Distribution of 35,346 wells in the subregion by status.
Nearly 90% percent have been abandoned or are producing. Another 7.3%, are in "Shut In" or "Permit Location" status may become producers over time.

Status	Number of Wells	Pct	SumPct
Abandoned	23894	67.6%	67.6%
Producing	7704	21.8%	89.4%
Shut In	1428	4.0%	93.4%
Permit Location	1168	3.3%	96.7%
Unknown	765	2.2%	98.9%
Temp Abandoned	163	0.5%	99.4%
Injecting	78	0.2%	99.6%
Waiting Completion	65	0.2%	99.8%
No Designation	61	0.2%	99.9%
Domestic Well	20	0.1%	100.0%
	35,346	100.0%	

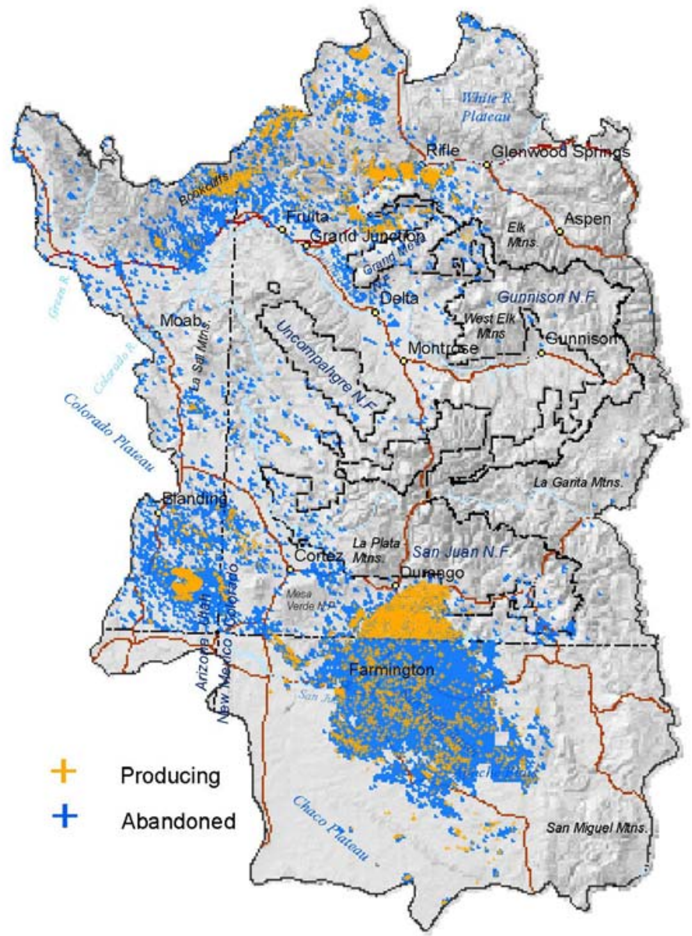
Nearly 80% percent have been abandoned or are producing. Another 15.1% are in "Shut In" or "Permit Location" status and may become producers over time.

Table 7-47 Distribution of 8,870 wells in the ARW Landscape by status.

Status	Number of Wells	Pct	SumPct
Abandoned	3898	43.9%	43.9%
Producing	3427	38.6%	82.6%
Permit Location	800	9.0%	91.6%
Shut In	544	6.1%	97.7%
Temp Abandoned	76	0.9%	98.6%
Injecting	51	0.6%	99.2%
Waiting Completion	29	0.3%	99.5%
No Designation	33	0.4%	99.9%
Domestic Well	12	0.1%	100.0%
	8870	100.0%	

Figure 7-58 Nearly 67% percent of the wells in the subregion may be classified as abandoned and about 21% are currently classified as active or producing.

Significant levels of production in the Paradox Basin are on Tribal lands. Very high levels in the San Juan Basin are associated permitting regimes on private and tribal lands that are relatively more permissive than on Federal and State lands.



Subregion Scale GAP Vegetation Analysis

Of the 35,346 wells in the subregion, just over 92% of these fall in five of twenty vegetation classes (Table 7-48). These are dominantly dry-land classes and more than half of these wells fall in two types: pinyon-juniper and dessert grassland. Figure 7-59 shows the distribution of vegetation classes containing the most significant levels of activity.

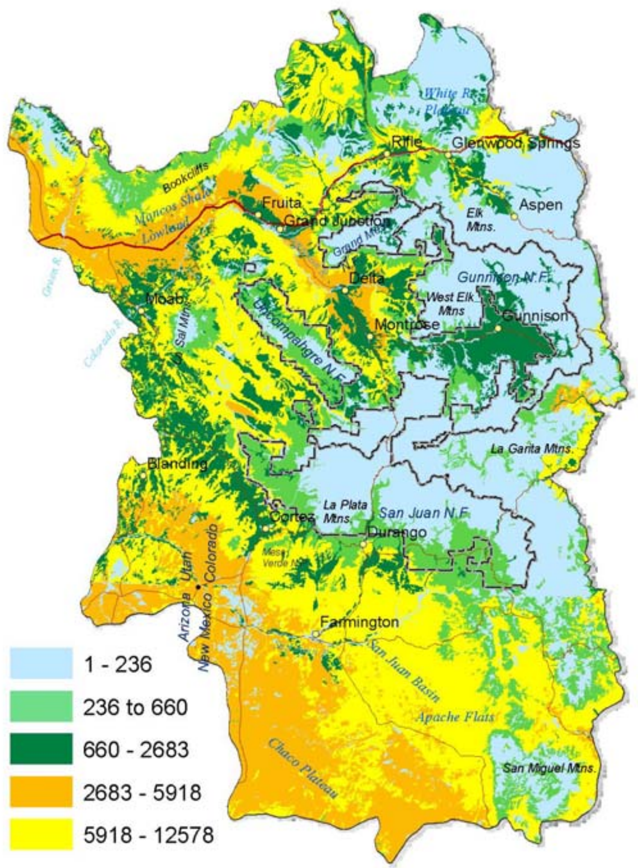
Table 7-48 Distribution of wells by vegetation class.

Gap Veg Class	Number of Wells	Pct	Sum Pct
pinyon -juniper	12,578	35.6%	35.6%
desert grassland	9,919	28.1%	63.6%
desert shrub	5,918	16.7%	80.4%
crops	2,683	7.6%	88.0%
sagebrush	1,528	4.3%	92.3%
ponderosa pine	660	1.9%	94.2%
deciduous oak	552	1.6%	95.7%
mountain grassland	462	1.3%	97.0%
aspen	236	0.7%	97.7%
woody riparian/wetland	226	0.6%	98.3%
mixed conifer	170	0.5%	98.8%
barren	104	0.3%	99.1%
urban	102	0.3%	99.4%
spruce - fir	90	0.3%	99.7%
greasewood	32	0.1%	99.8%
herbaceous riparian/wetland	32	0.1%	99.8%
alpine	28	0.1%	99.9%
mountain shrubland	19	0.1%	100.0%
water	4	0.0%	100.0%
Unclassified	2	0.0%	100.0%
lodgepole pine	1	0.0%	100.0%
	35,346	100.0%	

Within the dominantly dry-land classes, where the numbers of wells are highest, potential levels of disturbance are about 1% percent or less of the total vegetation type (Table 7-49). Overall levels of disturbance increase from here when accounting for roads, pipelines and other infrastructure.

Over half of the sites are Pinyon-Juniper and Spruce-fir vegclasses. Forest lands comprise significant proportions the Spruce-fir vegetation type. Mine sites in upland areas including Silverton and the La Plata Mountains are strongly correlated to Spruce-fir, aspen and alpine vegetation classes.

Figure 7-59 Mineral site counts by vegetation class.



Assuming 3 acres of disturbance per well, the following table shows the percentage of each vegetation class in the subregion potentially disturbed by wells.

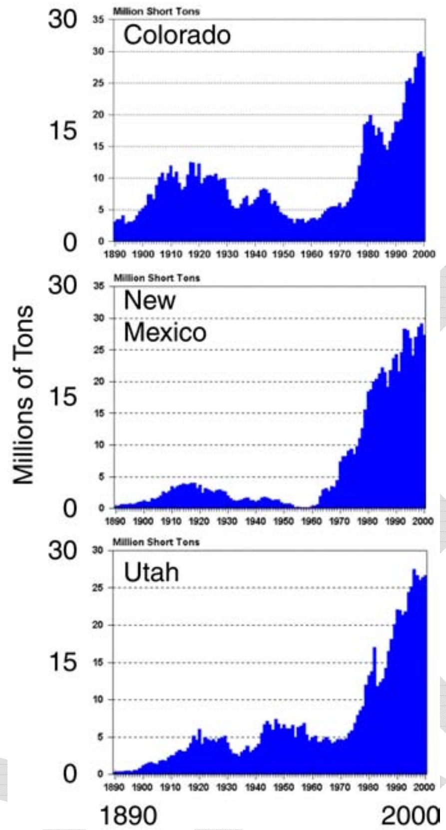
Table 7-49 The percentage of each vegetation class in the subregion potentially disturbed by wells.

Gap Veg Class	Veg ClassAcres	Number of Wells	Disturbed Acres	Pct of Veg Class
desert grassland	2,694,973	9,919	29,757	1.104%
crops	1,398,015	2,683	8,049	0.576%
woody riparian/wetland	118,598	226	678	0.572%
herbaceous riparian/wetland	17,141	32	96	0.560%
pinyon -juniper	7,265,382	12,578	37,734	0.519%
urban	64,333	102	306	0.476%
desert shrub	5,503,298	5,918	17,754	0.323%
mountain grassland	589,301	462	1,386	0.235%
sagebrush	2,965,185	1,528	4,584	0.155%
greasewood	63,032	32	96	0.152%
barren	289,403	104	312	0.108%
ponderosa pine	2,064,164	660	1,980	0.096%
deciduous oak	1,760,024	552	1,656	0.094%
mixed conifer	1,209,307	170	510	0.042%
aspen	2,272,967	236	708	0.031%
mountain shrubland	222,040	19	57	0.026%
water	63,941	4	12	0.019%
spruce - fir	3,485,003	90	270	0.008%
alpine	1,606,270	28	84	0.005%
lodgepole pine	462,096	1	3	0.001%
Unclassified	0	2	0	0.000%
	34,114,473	35,346	106,038	

Coal

Coal has been produced in Colorado, New Mexico and Utah since the middle of the 19th century. Coal production in Colorado expanded to become the largest in the West with the expansion of railroads in the region. Production in all three states grew from the turn of the century to a peak prior to the Great Depression. Production tapered off through the war years and 1950s with a resurgence beginning in the 1970s (Figure 7-60). Typical mine operations in the early period were small and served local markets. By contrast, modern mine operations are large and serve regional markets.

Figure 7-60 Coal production in Colorado, New Mexico and Utah.
(Adapted after EIA (2004)).



Upper Cretaceous rocks throughout the subregion contain coal bearing strata. Deposits crop-out along the margins of regional structural basins. Traces of these basin edges are evident by the spatial pattern of the 535 mine sites in the subregion (Figure 7-61). The majority of these sites consist of abandoned prospects and small scale underground mines. Most mines developed before 1970 were underground operations (EIA, 2004).

Within the subregion there are 535 coal mine sites and in the ARW landscape there are 287 coal mine sites (Table 7-50). About 10% of these locations, from the MAS/MILS (Causey, 1998) database, are considered to be in *Active* status. The remainder are classified as *Historic, Prospects* or *Unknown*. Today, only a handful of commercially viable mines remain. Most coal production is aimed at electrical generation. Four mines in the subregion are associated with electrical generation plants adjacent to the source mines.

Table 7-50 Number of (MAS/MILS) Coal Mine sites in the subregion and ARW Landscape.
Adapted after Causey (1998). About 10% percent of these are considered to be Recent.

Subregion

Forest	Num Sites	Pct
Outside	502	93.8%
GMUG	15	2.8%
San Juan	18	3.4%
	535	100.0%

ARW Landscape

Forest	Num Sites	Pct
Outside	254	88.5%
GMUG	15	5.2%
San Juan	18	6.3%
	287	100.0%

Today, in the subregion , four of twelve active operations are surface mining operations and eight are underground. Each of these operations are major producers of coal. From 1984 to 1987 these were among 28 operations whose cumulative production was over 221 million tons. Production over the same period from the twelve currently active operations was over 202 million tons (Table 7-51).

Figure 7-61 Subregion MAS/MILS coal sites.
Subregion MAS/MILS coal sites, twelve currently producing mines and associated power plants. MAS/MILS after Causey (1997). Currently producing mines and power plants after Kirschbaum (2000).

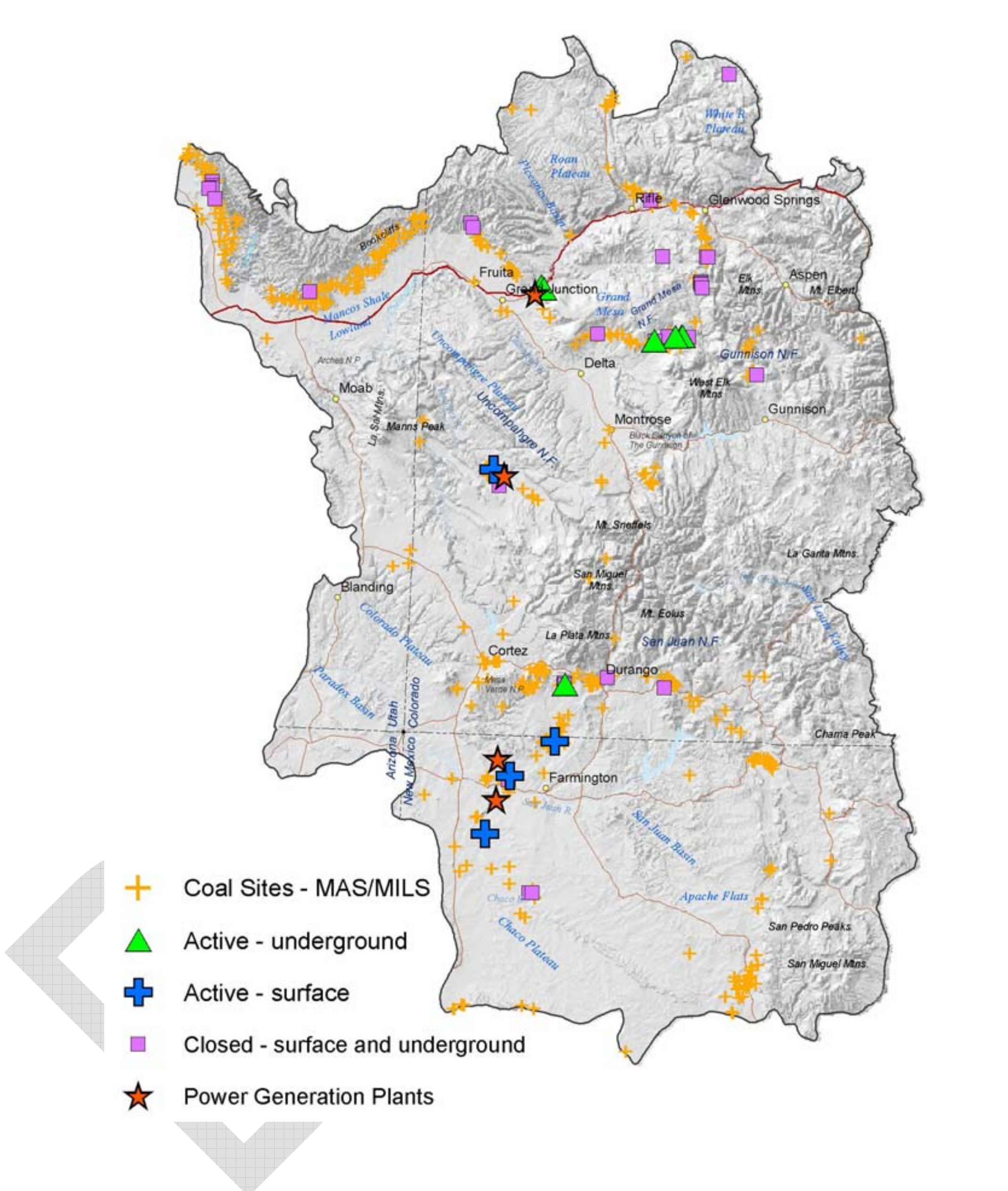


Table 7-51 Twenty-eight producing coal mines in the subregion from 1984 to 1997. Twenty-eight producing coal mines in the subregion from 1984 to 1997 sorted by cumulative tonnage. As of 1997 there were 12 Active mines.

MSHAID	Mine Name	Status	Type	1984 to 1997
2900097	NAVAJO MINE	Active	Surface	97,556,749
2901168	SAN JUAN MINE AND PLANT	Active	Surface	49,310,026
503672	WEST ELK MINE	Active	Underground	18,351,748
2901825	LA PLATA	Active	Surface	14,490,077
4200093	SUNNYSIDE MINE NO. 1	Permanently Abandoned	Underround	6,521,539
504184	BOWIE MINE #1	Active	Underground	6,239,472
503787	BEAR #3 MINE	Active	Underground	4,426,193
500281	ROADSIDE SOUTH PORTAL	Active	Underground	3,866,420
500301	DUTCH CREEK	Permanently Abandoned	Surface	3,449,170
504452	SANBORN CREEK MINE	Active	Underground	3,362,130
500469	DUTCH CREEK NO. 2	Permanently Abandoned	Surface	2,392,500
502898	CYPRUS ORCHARD VALLEY	Permanently Abandoned	Underground	2,084,162
500266	KING COAL MINE	Active	Underground	1,735,363
500294	SANBORN CREEK SURFACE FACILITI	Active	Underground	1,676,251
4200092	SUNNYSIDE MINE NO. 3	Permanently Abandoned	Underground	1,489,251
500299	NEW HORIZON MINE	Active	Surface	1,133,930
2901868	GATEWAY	Permanently Abandoned	Surface	819,208
4202093	SUNNYSIDE FACILITY	New - Under Construction	Surface	762,116
503013	MCCLANE CANYON MINE	Temporarily Closed	Underground	530,414
503012	ROADSIDE NORTH PORTAL	Active	Underground	365,680
2901833	DE-NA-ZIN	Permanently Abandoned	Surface	320,125
500300	L.S. WOOD	Permanently Abandoned	Underground	172,929
500259	O.C. COAL MINE	Permanently Abandoned	Underground	39,871
502421	EASTSIDE MINE	Permanently Abandoned	Underground	26,325
503683	CARBON JUNCTION MINE	Temporarily Closed	Surface	22,259
502658	THOMPSON CREEK NO. 1	Permanently Abandoned	Underground	20,724
4200094	SUNNYSIDE NO. 2 MINE	Permanently Abandoned	Underground	19,729
501962	RED CANYON #1	Permanently Abandoned	Underground	5,705
				221,190,066

Similarly, today, in the ARW landscape, 1 of 9 active operations is a surface mining operation and the remaining eight are underground. From 1984 to 1987 these were among 23 operations in the landscape whose cumulative production was over 43 million tons. Production over the same period from the twelve currently active operations was over 41 million tons (Table 7-52).

Figure 7-62 ARW Landscape MAS/MILS coal sites.
ARW Landscape MAS/MILS coal sites, nine currently producing mines and associated power plants. MAS/MILS after Causey (1997). Currently producing mines and power plants after Kirschbaum (2000).

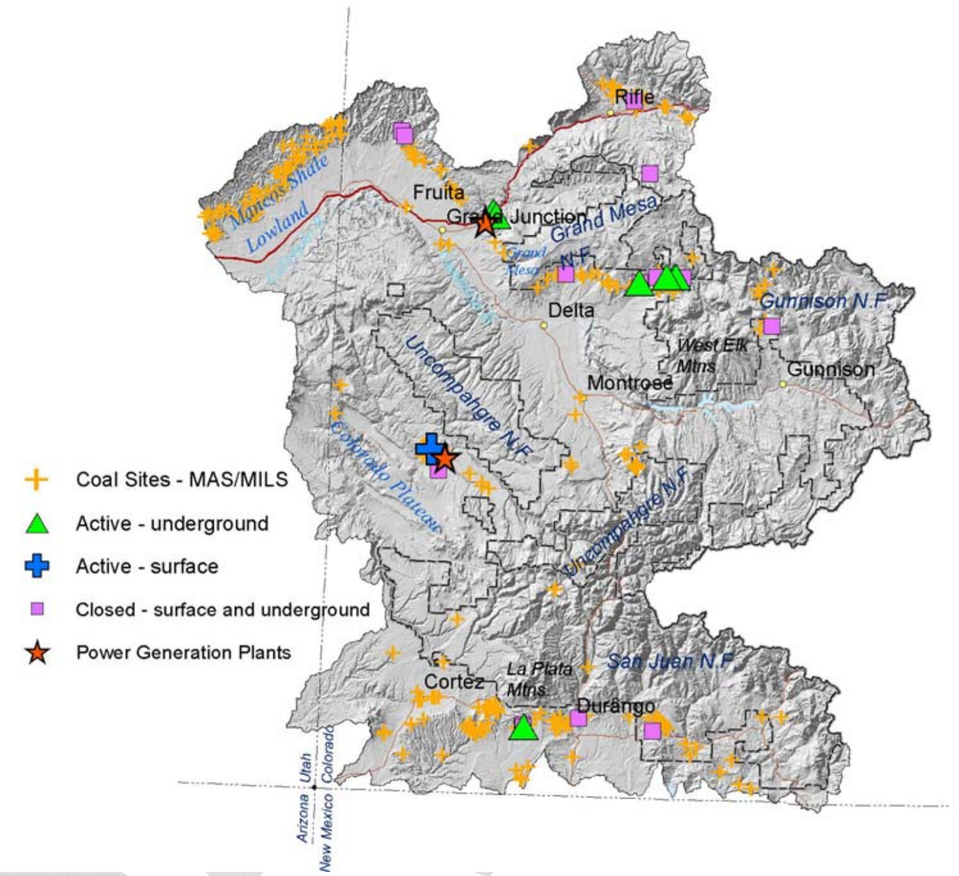


Table 7-52 Twenty-three producing coal mines in the ARW Landscape from 1984 to 1997.

Twenty-three producing coal mines in the ARW Landscape from 1984 to 1997 sorted by cumulative tonnage. As of 1997 there were nine active mines.

MSHAID	Mine Name	Status	Type	1984 to 1997
503672	WEST ELK MINE	Active	Underground	18,351,748
504184	BOWIE MINE #1	Active	Underground	6,239,472
503787	BEAR #3 MINE	Active	Underground	4,426,193
500281	ROADSIDE SOUTH PORTAL	Active	Underground	3,866,420
504452	SANBORN CREEK MINE	Active	Underground	3,362,130
502898	CYPRUS ORCHARD VALLEY	Permanently Abandoned	Underground	2,084,162
500266	KING COAL MINE	Active	Underground	1,735,363
500294	SANBORN CREEK SURFACE FACILITI	Active	Underground	1,676,251
500299	NEW HORIZON MINE	Active	Surface	1,133,930
503013	MCCLANE CANYON MINE	Temporarily Closed	Underground	530,414
503012	ROADSIDE NORTH PORTAL	Active	Underground	365,680
500259	O.C. COAL MINE	Permanently Abandoned	Underground	39,871
502421	EASTSIDE MINE	Permanently Abandoned	Underground	26,325
503683	CARBON JUNCTION MINE	Temporarily Closed	Surface	22,259
501962	RED CANYON #1	Permanently Abandoned	Underground	5,705
502303	BLUE FLAME COAL MINE	Permanently Abandoned	Surface	0
503119	MUNGER CANYON MINE	Permanently Abandoned	Surface	0
503133	LA PLATA #1	Permanently Abandoned	Surface	0
503644	COAL CREEK PREP PLANT	Permanently Abandoned	Surface	0
504457	HAMILTON MINE	Permanently Abandoned	Surface	0
500239	BOWIE MINE	Permanently Abandoned	Underground	0
500293	HAWKS NEST EAST	Permanently Abandoned	Underground	0
503134	COAL GULCH	Permanently Abandoned	Underground	0
				43,865,923

Bibliography

Amos, John F. 2003. Environmental Aspects of Modern Onshore Oil and Gas Development. Testimony to the Committee on Resources of the U.S. House of Representatives, Subcommittee on Energy and Mineral Resources. September 17, 2003. Online: http://www.wilderness.org/Library/Documents/upload/Testimony-J-Amos-of-Skytruth-on-Natural-Gas-Development-in-Wyoming.pdf

Causey, J. Douglas. 1998. MAS/MILS mineral location database information. MAS/MILS Arc/Info point coverage for the Western U.S. (excluding Hawaii). Open-File Report 98-512. U.S. Department of Interior, U.S. Geological Survey. Spokane, WA. Online: http://geo-nsdi.er.usgs.gov/cgi-bin/publication?open-file/98-512/

COGCC. 2004. Online Oil and Gas Well Locations Dataset. The Utah Division of Oil Gas and Mining. Online: http://ogm.utah.gov/oilgas/DOWNLOAD/downpage.htm.

EIA. 2004. State Coal Profiles. U.S. Department of Energy, Energy Information Administration. Online: http://www.eia.doe.gov/cneaf/coal/statepro/imagemap/usaimagemap.htm

Ellingson, Jack A. 1996. Volcanic Rocks, Chapter 6. In: The Western San Juan Mountains, Their Geology, Ecology, and Human History. Ed. Rob Blair. University Press of Colorado, Fort Lewis College Foundation. Niwot, CO.

Engler, Thomas W. Brian S. Brister, Her-Yuan Chen and Lawrence W. Teufel. 2001. Oil and Gas Resource Development for San Juan Basin, New Mexico. A 20 Year, Reasonable Foreseeable Development (RFD) Scenario Supporting the Resource Management Plan for the Farmington Field Office, Bureau of Land Management. U.S. Department of Interior, Bureau of Land Management, Albuquerque Field Office. Unpublished Report on file.

Kirschbaum, Mark A. 2000. Executive Summary – Geologic Assessment of Coal in the Colorado Plateau: Arizona, Colorado, New Mexico, and Utah. U.S. Geological Survey Professional Paper 1625-B. U.S. Department of Interior, U.S. Geological Survey, Denver, CO. Online: http://greenwood.cr.usgs.gov/energy/coal/PP1625B/

Murray, R. A. 1980. Multiple Use in the Bighorns – The Story of the Bighorn National Forest. Vol. 1. Narrative History and Notes. Sheridan, WY.

Ongard, 2001. Ongard petroleum well database for northwest New Mexico. New Mexico Institute of Mining and Technology. In: (Engler, 2001) Oil and Gas Resource Development for San Juan Basin, New Mexico, A 20-year, Reasonable Foreseeable Development (RFD) Scenario Supporting the Resource Management Plan for the Farmington Field Office, Bureau of Land Management

Preston, Michael. 2004. Productive Harmony Analysis, Interpretive Framework for Social and Economic Assessment of Southwest Colorado Communities and San Juan Public Lands, DRAFT. Fort Lewis College, Office of Community Services, Unpublished. Prepared report for and on file at the San Juan

Public Lands Center, U.S. Forest Service and Bureau of Land Management. Durango, Colorado.
Robinson, Rob. 2002. Upper Animas River Watershed, San Juan Field Office, Project Descriptions for Elk Tunnel, Forest Queen, Joe & John, Lackawanna, and Lark Mines. U.S. Department of Interior, BLM, Colorado State Office. Online: http://www.co.blm.gov/mines/upperaniproj/upaniprojdesc.htm
Schnitzer, Russ and Rob Roberts. 2004. Settled, Mined & Left Behind. The legacy of abandoned hardrock mines for the rivers and fish of the American West, and solutions for cleaning them up. Trout Unlimited. Online: http://www.tu.org/pdf/conservation/mining_report/mining_report04_full.pdf
U.S.G.S. 2000. Water Produced with Coal-Bed Methane. Fact Sheet FS-156-00. U.S. Department of Interior, U.S. Geological Survey. Online: http://pubs.usgs.gov/fs/fs-0156-00/
U. S. G. S. 2004. Resource Potential and Geology of the Grand Mesa, Uncompahgre, and Gunnison (GMUG) National Forests and Vicinity, Colorado. U.S. Geological Bulletin 2213. Viki Bankey Editor. U.S. Department of Interior.
UTOG. 2004. Online Oil and Gas Well Locations Dataset. The Utah Division of Oil Gas and Mining. http://ogm.utah.gov/oilgas/DOWNLOAD/downpage.htm
Warner, L. A. 1980. The Colorado Lineament In: Colorado Geology, Harry C. Kent, Karen W. Porter eds. Denver, Colorado: Rocky Mtn. Assoc. of Geologists

Chapter 8. Landscape Patterns (Module 5A & 5B)

Page intentionally blank.

Chapter 9. MODULE VI: AREAS OF SPECIAL BIODIVERSITY SIGNIFICANCE

INTRODUCTION

This module identifies and describes areas of special biodiversity significance throughout the San Juan Landscape, including plant species of special concern, plant communities of special concern, CNHP potential conservation areas (PCAs), TNC conservation areas, roadless areas, fens, old growth forests, and Research Natural Areas (RNAs).

PLANT SPECIES OF SPECIAL CONCERN

There are 35 Plant Species of Special Concern in the SJL (Table 9-1). They are designated as such because they appear on the R2 Forest Service Sensitive Species list, the CO BLM State Director's Sensitive Species list, or the CNHP Species of Special Concern list (G1 or G2 conservation rank).

Sensitive plant species are those species that typically have small and widely dispersed populations, and for which population viability is a concern as evidenced by significant current or predicted downward trends in population numbers or density, or significant current or predicted downward trends in habitat capability that would reduce a species' existing distribution (FSM 2670.5, BLM Bulletin CO-2000-014).

CNHP Species of Special Concern with a G1 rank are critically imperiled throughout their range because of extreme rarity (often known from five or fewer extant occurrences or very few remaining individuals) or because some factor of a species life history makes it vulnerable to extinction. A G2 rank indicates the species is imperiled throughout its range because of rarity (often known from 6-20 occurrences) or because of factors demonstrably making a species vulnerable to extinction.

Plant Species of Special Concern have special ecological significance because of their biodiversity importance and their rarity within the SJL. Since these species are vulnerable to management actions and extinction, their protection and persistence within the landscape and subregion is very important in terms of maintaining biodiversity.

In addition to their intrinsic value, these rare species have biodiversity importance because they may provide scientific, medical, and educational benefits, and their disappearance may adversely affect the plant communities and ecosystems they are a part of.

Table 9-1 Plant Species of Special Concern.

PLANT SPECIES OF SPECIAL CONCERN	STATUS
Amsonia jonesii *(K)*	BLM Sensitive
Astragalus deterior (K)	G2 CNHP Species of Special Concern
Astragalus iodopetalus (K)	G2 CNHP Species of Special Concern
Astragalus missouriensis var. *humistratus* (K)	FS Sensitive, G5T1 CNHP
Astragalus proximus (K)	FS Sensitive
Astragulus cronquistii (K)	BLM Sensitive, G2 CNHP
Calochortus flexuosus (K)	FS Sensitive
Carex diandra (L)	FS Sensitive
Carex viridula (K)	BLM Sensitive
Cryptantha gypsophila (L)	G1 CNHP Species of Special Concern
Cryptantha rollinsii (K)	BLM Sensitive
Cypripedium parviflorum (K)	FS Sensitive
Draba graminea (K)	G2 CNHP Species of Special Concern
Draba smithii (K)	FS Sensitive, G2 CNHP
Epipactis gigantea (K)	FS Sensitive
Erigeron kachinensis (K)	BLM Sensitive, G2 CNHP
Eriogonum clavellatum (L)	BLM Sensitive, G2 CNHP
Eriophorum altaicum var. *neogaeum* (K)	FS Sensitive, BLM Sensitive
Eriophorum chamissonis (K)	FS Sensitive
Eriophorum gracile (L)	FS Sensitive
Gilia sedifolia (L)	FS Sensitive, G1 CNHP
Ipomopsis polyantha (L)	FS Sensitive, BLM Sensitive, G1 CNHP
Lesquerella pruinosa (K) FS	FS Sensitive, BLM Sensitive, G2 CNHP
Machaeranthera coloradoensis (K)	FS Sensitive, G2 CNHP

PLANT SPECIES OF SPECIAL CONCERN	STATUS
Mimulus eastwoodiae *(K)*	BLM Sensitive
Parnassia kotzebuei (K)	FS Sensitive
Pediocactus knowltonii *(P)*	Federal Endangered, G1 CNHP
Physaria pulvinata (L)	G1 CNHP Species of Special Concern
Puccinellia parishii (L)	G2 CNHP Species of Special Concern
Salix arizonica (L)	FS Sensitive, G2G3 CNHP
Salix candida (K)	FS Sensitive, BLM Sensitive
Townsendia glabella (K)	G2 CNHP Species of Special Concern
Townsendia rothrockii (K)	G2 CNHP Species of Special Concern
Triteleia grandiflora (K)	FS Sensitive
Utricularia minor (K)	FS Sensitive

K – Known to occur in the SJL
L – Likely to occur in the SJL
P – Possible to occur in the SJL

PLANT COMMUNITIES OF SPECIAL CONCERN

There are 21 plant communities of special concern located within the SJL. They are designated as such because they are on the CNHP Communities of Special Concern list with a G1 or G2 conservation rank. CNHP Communities of Special Concern with a G1 rank are critically imperiled throughout their range because of extreme rarity (often known from five or fewer extant occurrences or very few remaining individuals) or because of factors demonstrably making it vulnerable to extinction. A G2 rank indicates the community is imperiled throughout its range because of rarity (often known from 6-20 occurrences) or because of factors demonstrably making it vulnerable to extinction.

Plant Communities of Special Concern have special ecological significance because of their biodiversity importance and their rarity within the SJL. Since these communities are vulnerable to management actions, their protection and persistence within the landscape and subregion is very important in terms of maintaining biodiversity.

In addition to their intrinsic value, these communities have biodiversity importance because they may provide scientific, medical, and educational benefits, and their disappearance may adversely affect the ecosystems they are a part of.

Abies concolor - Picea pungens - Populus angustifolia / Acer glabrum (G2)
Acer negundo - Populus angustifolia / Cornus sericea (G2)
Acer negundo / Betula occidentalis (G1G2)
Aquilegia micrantha - Mimulus eastwoodiae (G2G3)
Artemisia cana ssp. *viscidula* (G2G3)
Celtis laevigata var. *reticulata* / Mesic graminoid (G2G3)
Crataegus rivularis (G2)
Danthonia intermedia (G2G3)
Forestiera pubescens (G1G2)
Juniperus osteosperma / Stipa comata (G2)
Picea engelmannii / Betula glandulosa / Carex aquatilis - Sphagnum angustifolium (G2)
Picea engelmannii / Betula glandulosa / Carex aquatilis (G2)
Populus angustifolia / Crataegus rivularis (G2)
Populus angustifolia / Juniperus scopulorum (G2G3)
Populus angustifolia / Salix ligulifolia - Shepherdia argentea (G2)
Populus deltoides ssp. *wislizeni / Rhus trilobata*
Populus fremontii / Salix goodingii (G2)
Pseudotsuga menziesii / Paxistima myrsinites (G2G3)
Rhus trilobata (G2)
Salix brachycarpa / Carex aquatilis (G2G3)
Salix exigua - Salix ligulifolia (G2G3)
Salix ligulifolia (G2G3)

FENS

Fens are relict wetlands from the last glaciation that have unique characteristics found nowhere else on the landscape. They are connected to nutrient-rich groundwater that maintains perennial soil saturation. As groundwater percolates through the mineral soils, it collects nutrients and cations, which are absorbed by the fen. Fens have an accumulation of 40 cm or more of organic soil material and have soils that classify as histosols.

Fens are uncommon in the lower latitudes of the continental U.S. Although they occupy only a small percentage of the landscape in the Southern Rocky Mountains, fens are an important element of biological diversity, and often support globally rare plants, bryophytes, lichens, invertebrates, and unique species assemblages (Cooper 1994, Cooper 1996). Because the formation of fens is so slow, these ecosystems are essentially irreplaceable.

Fens in the SJL include the Chattanooga Iron Fen, the Cement Creek Iron Fen, the Burro Bridge Iron Fen, the South Mineral Creek Fens, the Harris Lake Fen, and the Grindstone Fen.

OLD GROWTH FORESTS

Old growth forests are unique ecosystems that are an important component of biological diversity (Kaufmann et al. 1992, Mehl 1992). Important biological values of old growth forests include habitat for a variety of animal and plant species, pools of genetic resources, and long-term biological records of climate (Kaufmann et al. 1992). Many species are known to require old growth forests for at least some part of their life cycle (Sobczak 1987, Maser et al. 1979, Walls et al. 1992, Kelly et al. 1993, Lattin 1993, Naslund 1993). Many organisms either find their optimum habitat in old growth forests or require old growth structures to survive, such as large snags and fallen trees (Forsman et al. 1984, Franklin et al. 1981, Harris 1984, Harris et al. 1984, Maser and Trappe 1984, Meslow et al. 1981). Because the formation of old growth forests is so slow, these ecosystems are essentially irreplaceable in our lifetime.

Old growth attributes for the Rocky Mountain Region are described by Mehl (1992) (Table 9-2). These, plus other attributes developed for the SJNF were used to define and inventory old growth forests for the spruce- fir, warm-dry mixed conifer, cool-moist mixed conifer, aspen, and ponderosa pine forest types. Attributes of old growth forests in the SJL include tree size, tree age, large trees/acre, standing dead trees (snags), trees with dead or broken tops, rotten trees, multiple canopy layers, and horizontal diversity.

Old growth ponderosa pine and warm-dry mixed conifer forests in the SJL have particular biodiversity significance because of their rarity, as these forests have been extensively harvested in the past. Large, old individual ponderosa pine trees that are not identified as being part of an old growth community also have particular biodiversity values associated with their genetics, their seed producing capabilities, and their future as snags.

Table 9-2 Old Growth Attributes for the Rocky Mountain Region

FOREST TYPE	acres	age	size	large tree/acre	rotten/dead tops	snags	layers
ponderosa pine	10,961	≥ 200	≥ 16	≥ 10	≥ 1/acre	≥ 2/acre	-
warm-dry mixed conifer	9,354	≥ 200	≥ 16	≥ 10	≥ 1/acre	≥ 2/acre	≥ 2
cool-moist mixed conifer	23,740	≥ 200	≥ 16	≥ 10	≥ 1/acre	≥ 2/acre	≥ 2
spruce- fir	117,482	≥ 200	≥ 16	≥ 10	≥ 1/acre	≥ 2/acre	≥ 2
aspen	25,195	≥ 100	≥ 14	≥ 10	≥ 1/acre	-	-

XXX MAP OF OLD GROWTH FORESTS XXX

CNHP POTENTIAL CONSERVATION AREAS (PCAs)

The CNHP has identified Potential Conservation Areas (Table 9-3) that contain high biodiversity values including occurrences of globally rare and endemic species and communities, Forest Service and BLM sensitive species, the presence of important habitat, and ecological systems that are critical to the diversity of the landscape. PCAs are polygons that represent the minimum area needed to ensure the long-term persistence of the species, communities, and ecological systems on which the PCA is based. Each PCA is assigned a biological diversity rank (B-rank) ranging from B1 (Outstanding Significance) to B5 (General Significance). In this assessment we consider only the PCAs with a B1 or B2 rank, as these are the areas with the highest biodiversity values.

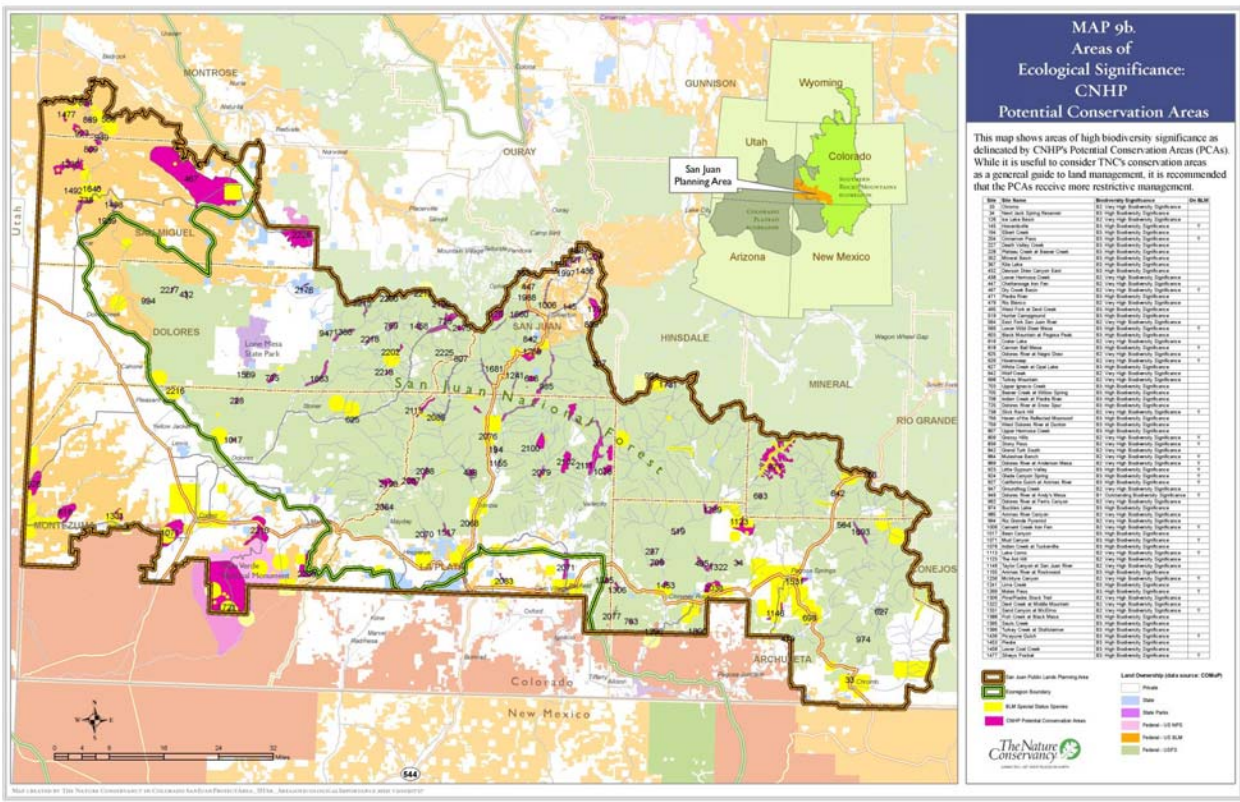
Protection and proper management of these sites will ensure the long-term persistence of the species, communities, and ecological systems on which they are the based, and will help to ensure the biodiversity they represent within the larger subregions they occur in. Since this assessment is only looking at the CNHP Potential Conservation Areas with the highest biological diversity ranks, these sites are the ones with the most ecological significance.

Table 9-3 CNHP Potential Conservation Areas- San Juan Public Lands

CNHP POTENTIAL CONSERVATION AREAS	RANK
Animas River Canyon	B2
Columbine Pass-Chicago Basin	B2
Endlich Mesa	B2
Junction Creek	B2
Lightner Creek	B2
Lime Mesa	B2
Lower Hermosa Creek	B2
Madden Creek	B2
East Fork San Juan River	B2
Nichols Draw (The Ant Hill)	B2
Stollsteimer Creek North	B2
Turkey Mountain	B2
Burro Bridge Iron Fen	B2
Cement Creek Iron Fen	B2
Chattanooga Iron Fen	B2
Crater Lake	B2
Grand Turk South	B2
Ice Lake Basin	B2
Lake Como	B2
South Fork Mineral Creek	B2
Stony Pass	B2
Archuleta Creek	B2
Beaver Creek at Willow Spring	B2
Chromo	B2
Coyote Wash	B2
Dawson Draw Canyon East	B2
Devil Creek at Middle Mountain	B2
Dolores River at Anderson Mesa	B2
Dolores River at Andy's Mesa	B1
Dolores River at Disappointment	B2
Dolores River at Ferris Canyon	B2
Dolores River at Negro Draw	B2
Glade Canyon Spring	B2
Grassy Hills	B2
Groundhog Creek	B2
Hovenweep	B2
McIntyre Canyon	B2
Mill Creek at Pagosa Springs	B1
Muleshoe Bench	B2
Pine/Piedra Stock Trail	B2
Rio Blanco	B2
Rio Grande Pyramid	B2
Sand Canyon at McElmo	B2

Table 9-3 CNHP Potential Conservation Areas- San Juan Public Lands	
CNHP POTENTIAL CONSERVATION AREAS	**RANK**
Slick Rock hill	B2
Slick Rock	B2
Taylor Canyon at San Juan River	B2
Wolf Creek	B2
Mesa Verde Entrance	B2
Navajo Basin	B2
Storm Peak	B2

Figure 9-1 CNHP Potential Conservation Areas- San Juan Public Lands (2005)[6]



TNC CONSERVATION AREAS

The San Juan Landscape is associated with The Nature Conservancy's Southern Rocky Mountains and Colorado Plateau Ecoregions. Assessments written for these ecoregions identify Conservation Areas that contain high biodiversity values including occurrences of globally rare, endemic and endangered species and

6 **The Nature Conservancy, San Juan Planning for Biodiversity Model Project, Phase 1 Report to BLM; October, 2005** http://conserveonline.org/workspaces/CO%20-%20San%20Juan%20Project/SanJuanPhase1Report/

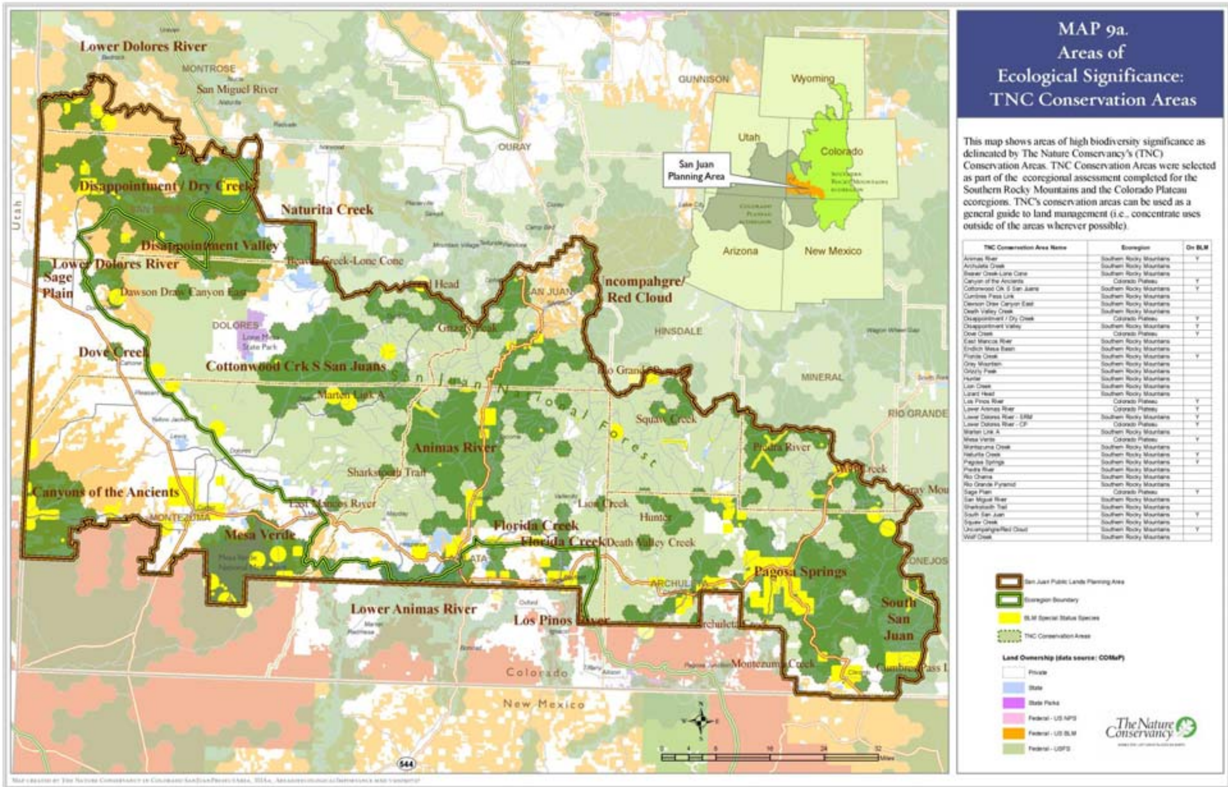
communities; species that face threats to their habitat; and examples of ecological systems that are critical to the diversity of the ecoregions (Table 9-4). Conservation areas, with proper management, would ensure the long-term persistence of the ecoregion's species, communities, and ecological systems, and will help to ensure the biodiversity they represent within the larger ecoregions and subregions they occur in. Since this assessment is only looking at the TNC Conservation Areas with the highest biological diversity ranks, these sites are the ones with the most ecological significance.

TNC uses biodiversity values to assess the overall biological diversity significance of a conservation area. High values reflect areas with high uniqueness (many rare species, communities, or ecological systems) and/or high landscape integrity (good viability of rare species, communities, or ecological systems). Low values reflect areas with less uniqueness and/or low landscape integrity. In this assessment we consider only the areas with a Medium or High rank, as these are the areas with the highest biodiversity values.

Table 9-4 TNC Conservation Areas- San Juan Public Lands

TNC CONSERVATION AREAS	BIODIVERSITY VALUE
Animas River	Medium
Archuleta Creek	Medium
Cottonwood Creek South San Juans	Medium
Death Valley Creek	Medium
Endlich Mesa Basin	Medium
Gray Mountain	Medium
Grizzly Peak	Medium
Hunter	Medium
Lizard Head	Medium
Montezuma Creek	Medium
Naturita Creek	Medium
Pagosa Springs	Medium
Disappointment Valley	Medium
Piedra River	High
Rio Grande Pyramid	High
San Miguel River	Medium
South San Juan	High
Squaw Creek	High
Uncompaghre/Red Cloud	High
Wolf Creek	Medium
Lower Dolores River (CO plateau)	High
Dove Creek	Medium
Canyon of the Ancients	Medium
Mesa Verde	Medium
Lower Animas River	Medium
Los Pinos River	Medium

Figure 9-2 TNC Conservation Areas- San Juan Public Lands6



RESEARCH NATURAL AREAS

The San Juan National Forest currently has two RNAs, Narraguinnep and Williams Creek. Twenty one other potential RNAs have been identified on the SJNF for possible inclusion in the RNA system. RNAs serve three important functions:

Reference Areas – RNAs provide excellent examples of ecosystems in a relatively natural condition, and in so doing serve as benchmarks for monitoring and evaluating the sustainability and impacts of land management practices on lands with similar ecosystems. To determine the impact of management on an area it is useful to have, as a control, a similar area maintained in natural condition for comparison.

Biological Diversity – RNAs provide protection for biological diversity and some degree of assurance that a wide array of plant and animal species (including threatened, endangered, or sensitive species), plant communities, and ecosystems will be afforded a high degree of protection in the future.

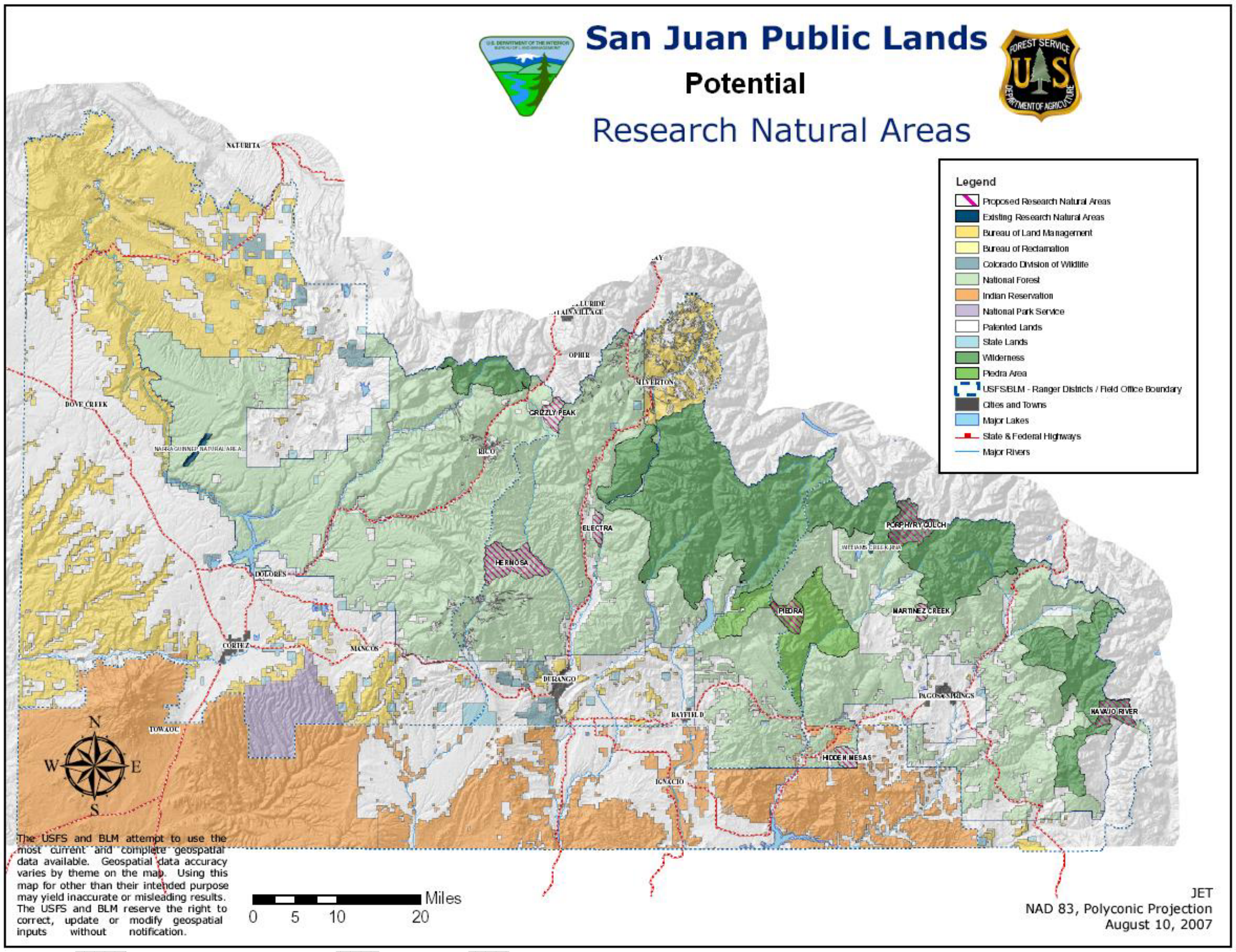
Research – RNAs provide sites for research into how ecosystems function, particularly sites where ecological and evolutionary processes are functioning in a relatively natural state. They serve as sites for monitoring long term ecosystem change including changes associated with global climate change and disturbances like fire, floods, and insect epidemics.

Table 9-5 Potential RNA's- San Juan National Forest

Potential RNA	Acres	Elevation (feet)	Quality*	Condition*	Viability*	Defensibility*
Bear Park	6934	7560 to 10,471	Excellent	Very good	Excellent	Excellent
Deadwood Mtn	2920	8440 to 12,400	Very good	Fair to Very good	Fair	Fair to Good
Electra	2265	7400 to 8841	Excellent	Excellent	Good	Good
Grizzly Peak	5672	10,140 to 13,752	Excellent	Excellent	Very good	Excellent
Grouse Rincon	8345	8800 to 12,915	Good	Very good	Very good	Excellent
Hermosa			Excellent	Good to Excellent	Excellent	Excellent
Hidden Mesas	6821	6340 to 8320	Excellent	Good to Excellent	Very good	Very good
Ignacio Creek	6193	6600 to 8936	Excellent	Good	Excellent	Excellent
Martinez Creek	1062	9600 to 11,460	Very good	Excellent	Excellent	Excellent
Middle Peak	2729	9680 to 13,290	Good to Excellent	Fair to Good	Excellent	Good
Navajo River	7085	9200 to 12,727	Excellent	Excellent	Excellent	Excellent
Needle Mtns			Excellent	Excellent	Excellent	Excellent
Porphyry Gulch	12,104	8560 to 12,593	Excellent	Excellent	Excellent	Excellent
Quien Sabe	8597	8760 to 12,139	Good	Good	Very good	Very good
Rio Blanco	9454	8360 to 12,887	Excellent	Excellent	Excellent	Excellent
Rock Lakes	1876	9632 to 12,401	Excellent	Excellent	Excellent	Excellent
Sand Creek	14,246	8000 to 12,410	Excellent	Very good	Excellent	Excellent
West Needles	14,040	7680 to 13,045	Good to Excellent	Good to Excellent	Very good	Good

* Quality: How well a site represents the key species or ecosystems.
Condition: How much the site has been altered from natural (pre-settlement) conditions.
Viability: Likelihood of long-term persistence of the key species or ecosystems.
Defensibility: Can the site be protected from human activities.

Figure 9-3 Potential RNA's



ROADLESS AREAS

Roadless Areas are ecosystems that are likely to be sustainable and self-regulating over the long-term. Because they are largely removed from ecologically damaging impacts caused by human access, resource extraction, and development, large roadless areas 1) support viable populations of native species, including wide-ranging species, and species that are sensitive to humans and require remote habitat; 2) perpetuate natural processes such as fire, flooding, and nutrient cycling; 3) provide habitat relatively free from exotic species; 4) protect aquatic habitats from pollution and degradation; 5) and provide baseline information by which to judge the ecological impacts of land management on similar ecosystems (Noss 1991 and Baker 1992).

For this assessment roadless areas include the original RARE II sites plus Wilderness Areas. RARE II define a roadless area as an area in a national forest or national grassland that (1) is larger than 5,000 acres or, if smaller, is contiguous to a designated Wilderness or primitive area, and (2) contains no roads and (3) has been inventoried by the Forest Service for possible inclusion in the Wilderness Preservation System.

Large roadless areas provide opportunities for maintaining intact ecosystem functions, including natural disturbance regimes, and for restoring some of the ecoregion's missing predator species such as grizzly bear, wolverine, and wolf. Many roadless areas provide linkages between protected areas, and connectivity between protected habitat that is essential to facilitate species movements and gene flow (Dobson et al. 1999). Roadless Areas also contain habitat for rare and imperiled species including federally listed threatened and endangered species and Forest Service and BLM sensitive species.

The many very large roadless areas (including Wilderness Areas) within the SJL may represent the most ecologically significance biodiversity attribute because not many other very large roadless areas exist in the subregion.

The San Juan National Forest manages four wilderness areas, all of which meet the RARE II definition of Roadless Areas. Approximately one fourth of the Forest totaling about 484,380 acres is designated as wilderness, most of which is above 9000 feet in elevation and encompasses much of the Forest's alpine and subalpine life zones.

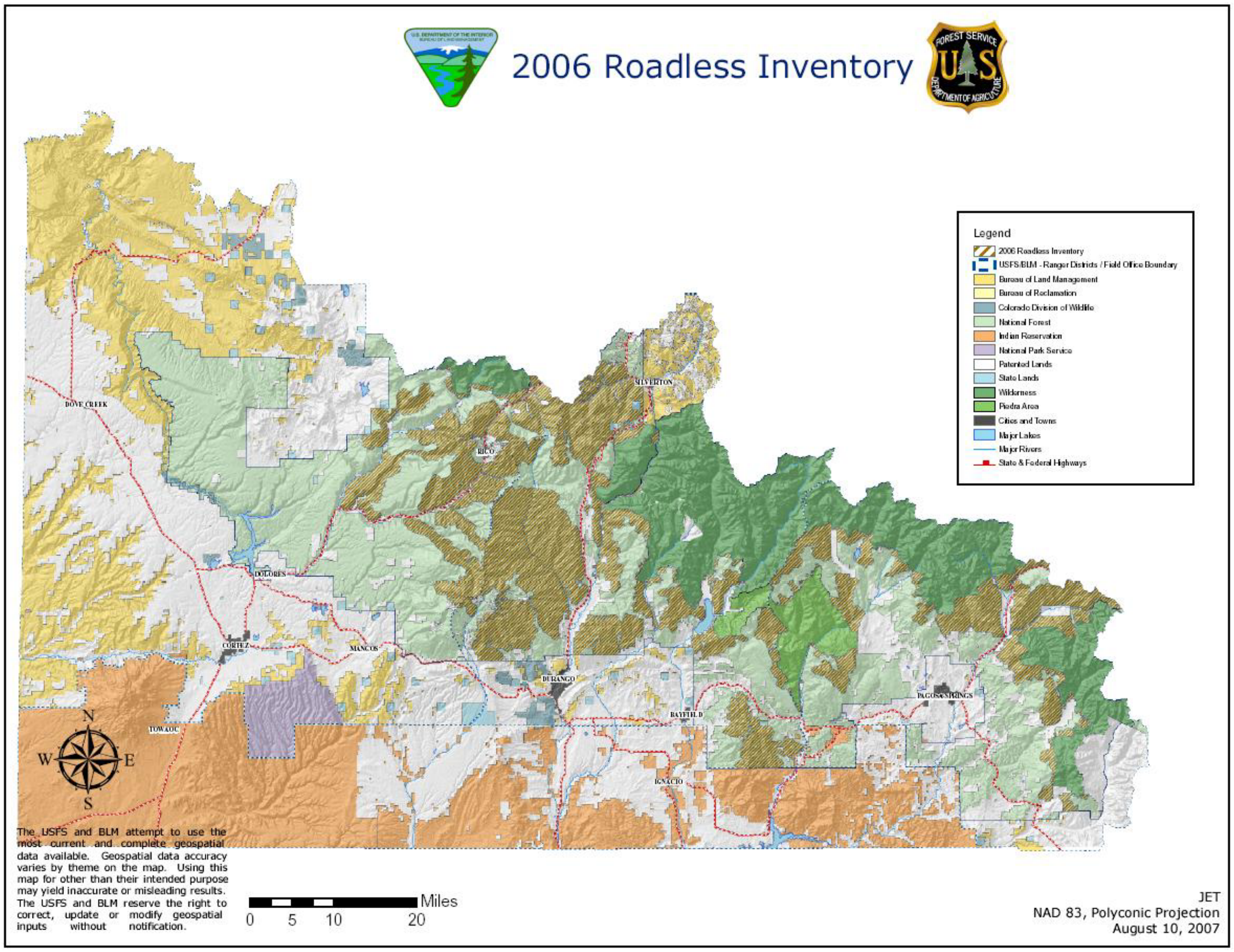
Table 9-6 Inventoried Roadless Areas- San Juan National Forest

ROADLESS AREAS IN THE SJL	ACRES
BLACKHAWK MOUNTAIN Roadless	17,172
DAVIS MOUNTAIN Roadless	1375
EAST ANIMAS Roadless	15,406
FLORIDA RIVER Roadless	35,536
GRAHAM PARK Roadless	10,614
HD MOUNTAIN Roadless	20,071
HERMOSA Roadless	14,2561
LIZARD HEAD Roadless	5241
MARTINEZ CREEK Roadless	4580
MONK ROCK Roadless	3029
PIEDRA Roadless	54,002

Table 9-6 Inventoried Roadless Areas- San Juan National Forest

ROADLESS AREAS IN THE SJL	ACRES
POISON PARK Roadless	8413
RUNLETT PARK Roadless	4970
RYMAN Roadless	7434
SAN MIGUEL Roadless	59,754
SHEEP MOUNTAIN Roadless	3777
SOUTH SAN JUAN Roadless	51,562
STORM PEAK Roadless	50,287
TREASURE MOUNTAIN Roadless	20,873
TURKEY CREEK Roadless	22,313
WEST NEEDLE Roadless	8440
WEMINUCHE Wilderness	331,605
SOUTH SAN JUAN Wilderness	71,714
LIZARD HEAD Wilderness	20,675
PIEDRA AREA Wilderness	60,388

Figure 9-4 Wilderness and Inventoried Roadless Areas



SPECIAL INTEREST AREAS

Special interest areas (SIAs) are designated to protect or enhance areas of the forest that contain unusual or significant characteristics, including botanical, ecological, geological, historical, paleontological, scenic, or zoological resources. They may be allocated to protect and manage threatened, endangered, or sensitive species and other elements of biological diversity, or for their emotional significance, historic importance, scenic values, or public popularity. SIAs that occur in the SJL include the Piedra Botanical Area which is habitat for *Lesquerella pruinosa,* a Forest Service and BLM sensitive plant species.

WILD AND SCENIC RIVERS

Wild and scenic rivers are managed to protect their free-flowing characteristics and their outstandingly remarkable values, including the ecological and biological values of their associated riparian areas (FSH 1909.12, Chapter 8). Since standards for wild and scenic rivers exclude timber production, dams & diversions, hydroelectric power facilities, flood control, mining, road construction, recreation development, structures, agriculture, and utilities the biodiversity values of these areas will remain high. Potential Wild and scenic rivers in the SJL include segments of the Piedra, Upper Pine, West Dolores, Dolores, Animas, San Juan, and Hermosa rivers.

THREATS TO AREAS OF SPECIAL BIODIVERSITY SIGNIFICANCE

Oil and Gas Activities: Oil and gas development in the SJL is expected to increase in the near future. In addition to direct disturbance that could cause mortality, habitat loss from wells, access roads, invasive plant species, and pipelines threaten native plants and communities.

Livestock Grazing: Much of the SJL is utilized for livestock grazing. Livestock foraging and trampling can adversely affect rare plant species and cause mortality. Livestock can also be a vector for the invasion of non-native plant species that can compete with rare plants.

Timber Harvest and Fuels Reduction Projects: Disturbance due to mechanical equipment used for timber harvest and fuels reduction operations, including road construction, can adversely affect and kill rare plants and old trees, and adversely affect habitats and ecosystems. Roads can also be a vector for the invasion of non-native plant species that can compete with rare plants.

Recreation: Recreation is increasing and becoming a threat to natural ecosystems in the SJL. ATVs can trample and kill rare plants. ATVs have also been identified as a vector for the invasion of non-native plant species, which can compete with rare plants. Non-motorized recreation, including hiking, mountain biking, and rock climbing, are also threats due to trampling (Cole and Knight 1990; Knight and Cole 1991).

Roads and Utility Corridors: Road and Utility Corridor construction is associated with a wide variety of impacts to rare plants/communities and old growth forests, including direct mortality, invasion by non-native plant species, fragmentation of habitats, hydrologic alteration, and erosion (Noss et al. 1997).

Invasive Species: Invasion of aggressive exotic species and the subsequent replacement of native species is one of the biggest threats to natural diversity (James 1993; D'Antonio and Vitousek 1992).

Natural Processes: Drought and natural fire may adversely affect rare plants and communities and cause mortality. Global warming could increase the risk of devastating wildfires.

Land development: Land development associated with residential and commercial growth on private lands adjacent to federal lands is occurring at a rapid rate. This is a major threat to rare plant species, since much of the land being developed is prime habitat for rare plants, and the construction can kill individuals and disturb habitat. Adverse affects on private lands could adversely affect rare plants on federal lands by disrupting gene flow and pollination.

Prescribed Fire: Prescribed fire and its associated suppression activities may adversely affect rare plants/communities and cause mortality.

Collecting is always a threat to rare plant species.

Hydrological Modifications: Hydrological alteration in the form of reservoirs, irrigation ditches or canals, and stream bank stabilization projects can adversely affect water flow, flood frequency, hydrophytic plants, and rare plants and communities (Chien 1985).

Fragmentation: Fragmentation is the disruption of continuity in predominantly natural landscapes (Lord and Norton 1990). It is human-caused and involves a reduction of total area of a natural landscape such as a roadless area (Buskirk et al. 2000). Fragmentation of natural habitats by human-caused structures and disturbance is blamed for wide-ranging impacts to biodiversity (Wilcove 1987).

INFORMATION GAPS

There is a lack of information on management activities and natural disturbances (fire), and their affect on rare plant species, communities of special concern, and on ecological systems.

More survey is needed for species and communities of special concern. An ongoing survey of rare plant species and communities of special concern through a challenge cost share agreement with the CNHP will continue through 2008.

More work is needed on the existing vegetation layer, including field verification of vegetation types.

More old growth work is needed, especially field verification of potential old growth sites.

More information about current and future threats is needed.

There is inadequate information to assess non-vascular plant species, lichens, mosses, fungi, ferns, and fern allies.

Need to research the pollination ecology and autecology of plant species of special concern.

An inventory of fens within the SJL is needed.

An analysis of potential Wild and Scenic Rivers is needed.

REFERENCES AND LITERATURE CITED

BLM Information Bulletin No. CO-2000-014. BLM State Director's Sensitive Species List

Colorado Natural Heritage Program, 2004. Element Occurrance Records for San Juan National Forest: GIS files and Database Reports.

Cooper, D.J., and Andrus, R. 1994. Patterns of vegetation and water chemistry in peatlands of the west-central Wind River Range, Wyoming. Can. J. Bot. 72: 1586 – 1597.

Cooper, D.J. 1996. Water and soil chemistry, floristics, and phytosociology of the extreme rich High Creek fen, in South Park, Colorado, U.S.A. Can. J. Bot. 74: 1801 – 1811.

Forest Service Handbook 1909.12. Chapter 8.

Kaufmann, Merill R., Moir W.H., and W.W. Covington 1992. Old Growth Forest: What do we know about their ecology and management in the SW and Rocky Mountain Region? Found in USDA Forest Service General Technical Report RM-213.

Mehl, Mel S. 1992. Old Growth Descriptions for the Major Cover Types in the Rocky Mountaon Region. Found in USDA Forest Service General Technical Report RM-213.

Neely, B., P. Comer, C. Moritz, M. Lammert, R. Rondeau, C. Pague, G. Bell, H. Copeland, J. Humke, S. Spackman, T. Schulz, D. Theobald, and L. Valutis. 2001. *Southern Rocky Mountains: An Ecoregional Assessment and Conservation Blueprint*. Prepared by The Nature Conservancy with support from the U.S. Forest Service, Rocky Mountain Region, Colorado Division of Wildlife, and Bureau of Land Management.

Ryan, Michael G., Linda A. Joyce, Tom Andrews, and Kate Jones. 1994. Research Natural Areas in Colorado, Nebraska, South Dakota, and Parts of Wyoming. Gen. Tech. Rep. RM-251. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 57p.

R2 (Rocky Mountain Region) Regional Forester's Sensitive Species List, 12/1/2003, FSM - R2 Supplement 2672.11 – Exhibit 01

Redders, Jeffery S., Vegetation of the San Juan National Forest, Working White Paper, 8/15/01, updated 5/24/04.

Redders, Jeffery S., 2003, A Classification of Riparian Area and Wetland Vegetation of
the San Juan National Forest.

US Fish and Wildlife Service Unit Species List of federally Threatened, Endangered, and Candidate species for the San Juan National Forest and the BLM San Juan Field Office, February 26, 2004.

USDA Forest Service. 1993. Research Natural Area Guide for the Rocky Mountain Region, USDA Forest Service, review draft. Rocky Mountain Region, Lakewood, CO. 38p.

Weber, William A. and Ronald C. Wittmann. 2001. Colorado Flora Western Slope, Third Edition. University Press of Colorado.