CHAPTER 2. EXISTING VEGETATION

Landscape Scale

Current Vegetative Description

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 percent, is within Colorado, with only 26 percent in New Mexico, 14 percent in Utah, and one percent 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 1).

Table 1. Generalized vegetation GAP vegetation classes in the subregion. Percentages are relative to the sub-region area.

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%
	SubTotal:	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%
	SubTotal:	1,526,301	2,385	4.5%	
	Total:	34,114,695	53,304	100.0%	

Tables 2 and 3 summarize the distribution of GAP classes in the GMUG and San Juan National Forests, respectively. The upland settings of the Forests are 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 2. Generalized vegetation GAP vegetation classes in the GMUG. Percent values are relative to the Forest area.

Forest	Class	Name	Acres	SqMiles	Pct	SumPct
GMUG	Vegetation	spruce - fir	943,115	1,474	29.6%	29.6%
		aspen	648,763	1,014	20.4%	49.9%
		alpine	397,266	621	12.5%	62.4%
		sagebrush	274,863	429	8.6%	71.0%
		lodgepole pine	236,601	370	7.4%	78.4%
		deciduous oak	207,282	324	6.5%	84.9%
		ponderosa pine	198,087	310	6.2%	91.2%
		pinyon -juniper	116,641	182	3.7%	94.8%
		mixed conifer	92,296	144	2.9%	97.7%
		barren	13,865	22	0.4%	98.2%
		mountain shrubland	13,566	21	0.4%	98.6%
		woody riparian/wetland	8,911	14	0.3%	98.9%
		mountain grassland	6,956	11	0.2%	99.1%
		desert shrub	331	1	0.0%	99.1%
		SubTotal:	3,158,543	4,981	99.1%	
	Non-Veg	crops	23,608	37	0.7%	99.8%
		water	5,493	9	0.2%	100.0%
		SubTotal:	29,102	45	0.9%	
		Total:	3,187,645	5,026	100.0%	

Table 3. 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%
		SubTotal:	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%	
		SubTotal:	25,533	40	1.2%	
		Total:	2,093,203	5,021	100.0%	

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.

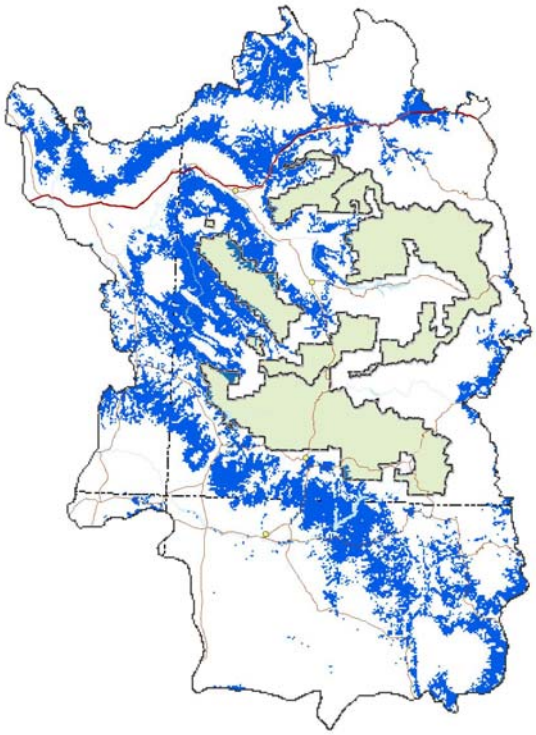
Vegetation/HRV Description

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.

Pinyon-Juniper

The pinyon-juniper vegetation type compromises 7,265,426 acres (21.3 percent) of the Sub-Region and 116,641 acres (3.7 percent) within the GMUG and 54,924 acres (2.6 percent) within the SJ National Forest (Figure 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. Pinyon pine (*Pinus edulis*), Utah juniper (*Juniperus osteosperma*) and Rocky Mountain (*Juniperus scopulorum*) dominate the pinyon-juniper type. *Juniperus* spp. dominates lower elevation/xeric sites and pinyon pine dominates at higher elevation/mesic sites.

Figure 1. The Pinyon-Juniper GAP class in the subregion covers 7,265,426 acres (21.3 **percent**) of the subregion.



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 pinyon 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 pinyon-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 pinyon-juniper forests are outside their HRV. For example, the large increase in pinyon 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 1950s 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 pinyon stands that are the result of the post 1950s drought recruitment.

Baker and Shinneman (2004) and Romme and others (2003a) have identified three major types of pinyon-juniper stand structures and fire regimes for the western U.S. All three of these major pinyon-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 pinyon-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 pinyon-juniper grass savanna is present but rare, the pinyon-juniper shrub woodland has low abundance, and the pinyon-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 pinyon-juniper type is referred to as the pinyon-juniper grass savanna. The pinyon-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 pinyon and juniper trees have become established in previously open grass/woodlands (Allen 1989). As bare ground was exposed from cattle grazing, pinyon 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 pinyon-juniper type is referred to as the pinyon-juniper shrub woodland. The pinyon-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 pinyon-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.

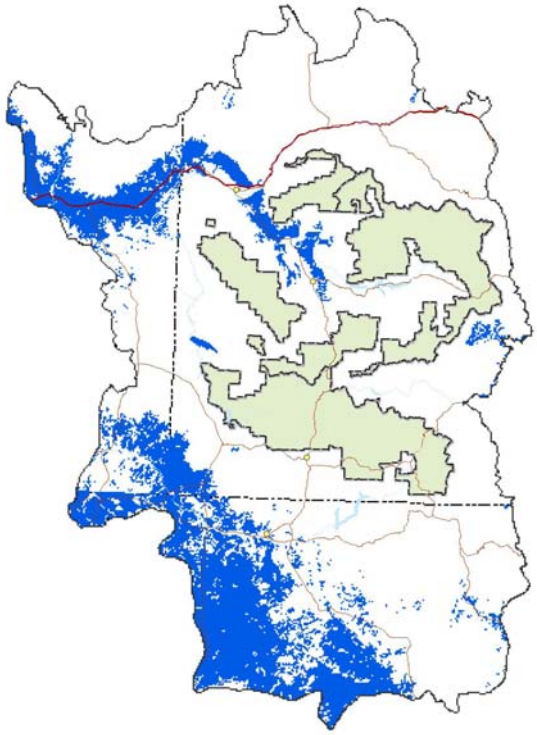
The third pinyon-juniper type is referred to as the pinyon-juniper forest. The pinyon-juniper forest's structure is characterized as having dense trees, sparse to moderate shrubs, and few herbs with an infrequent, high-severity crown fire (Romme et al. 2003a). Pinyon-juniper stands on Mesa Verde (Floyd 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 pinyon 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 percent) of the Sub-Region and only 331 acres within the GMUG and 0 acres within the SJ National Forest (Figure 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 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.

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



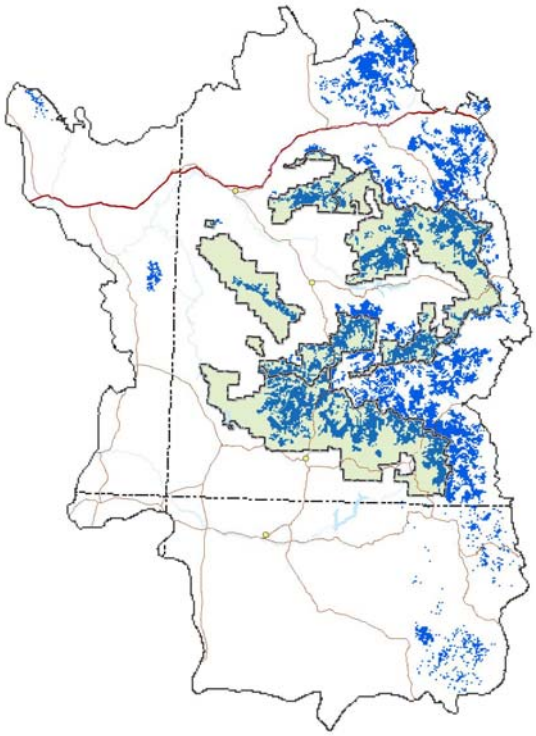
The winterfat shrub steppe is also included within this vegetation type and is found within the Upper Gunnison Basin at higher elevations, 7,500-9,000 ft (2286-2896 m), than the desert shrub community described above (Johnston 1997). The winterfat shrub steppe is compromised of dwarf shrubs, primarily 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 percent) of the Sub-Region and 943,115 acres (29.6 percent within the GMUG and 665,361 acres (31.8 percent) within the SJ National Forest (Figure 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 3. The Spruce-Fir GAP class in the subregion covers 3,485,025 acres (10.22 percent) 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).

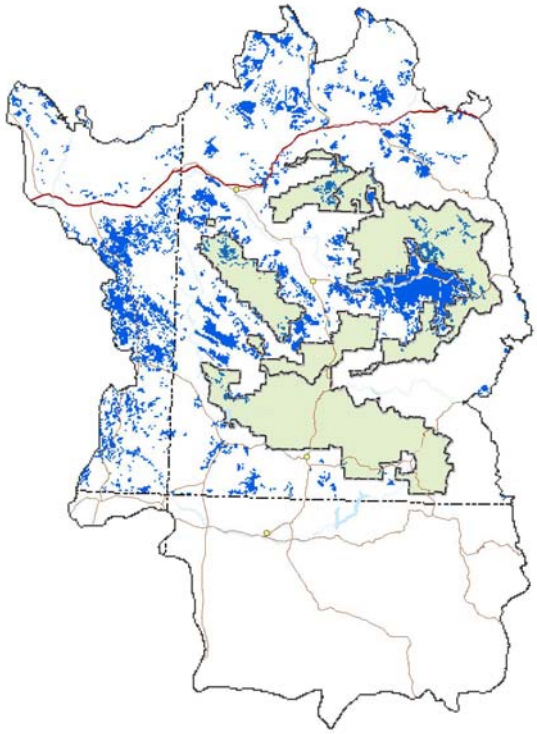
Determining HRV is more difficult in spruce/fir forests than other forest types because fire is generally lethal to trees leaving no fire scars and lags can exist between disturbance events and tree establishment. Therefore, post-fire age and size data combined are normally used to identify past disturbances (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 percent) of the Sub-Region and 274,863 acres (8.6 percent) within the GMUG and 21,520 acres (1.0 percent) within the SJ National Forest (Figure 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 areas 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. The Sagebrush GAP class in the subregion covers 2,965,204 acres (8.69 percent) 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 pinyon-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 percent 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.

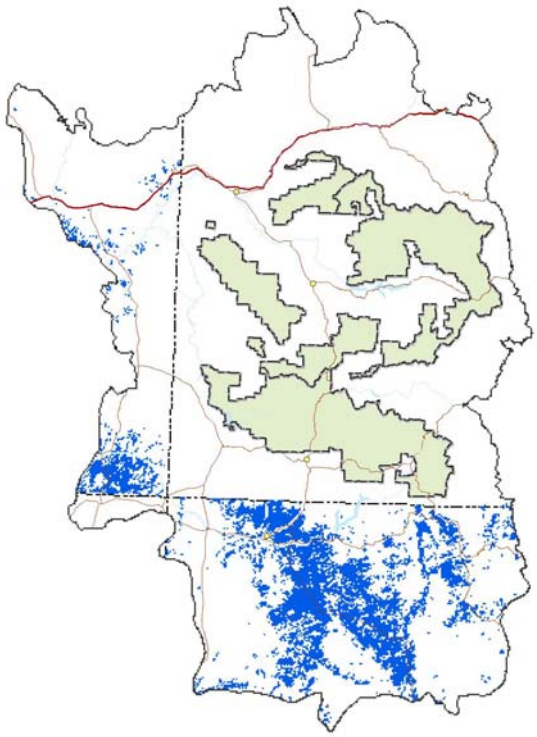
Areas with livestock grazing, increased non-natives and increaser shrub species, and increased fire frequency are outside their HRV. Successional dynamics indicate that these areas have crossed a critical threshold and have moved to an alternative community state or new equilibrium that can not be returned 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 percent) of the Sub-Region and is not present within the GMUG or the SJ National Forest (Figure 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 5. The Desert Grassland GAP class in the subregion covers 2,694,989 acres (7.90 percent) 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.

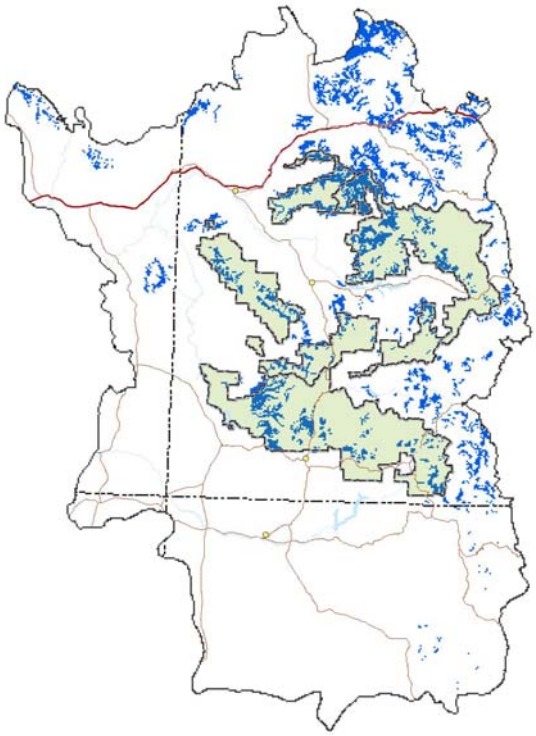
Aspen

The aspen vegetation type compromises 2,272,984 acres (6.66 percent) of the Sub-Region and 648,763 acres (20.4 percent) within the GMUG and 273,778 acres (13.1

percent) within the SJ National Forest (Figure 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). 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.

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



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 1800s and early 1900s 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 1700s to early 1800s in the Southwest and therefore high densities of old aspen stands are most likely not outside their HRV; however, more detailed studies

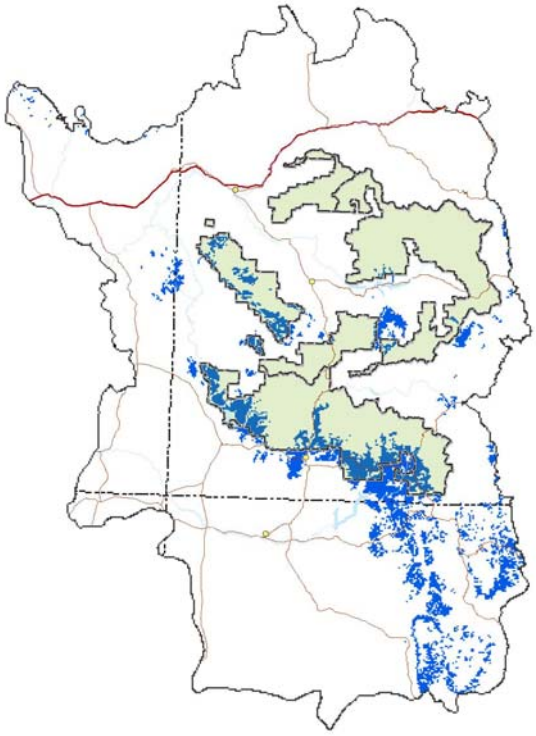
are needed to confirm or reject this hypothesis (Romme et al. 2003d; Kulakowski, in press).

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 percent) of the Sub-Region and 198,087 acres (6.2 percent) within the GMUG and 543,573 acres (26.0 percent) within the SJ National Forest (Figure 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 7. The Ponderosa Pine GAP class in the subregion covers 2,064,177 acres (6.05 percent) 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 1600s 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 1800s 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 1950s. 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.

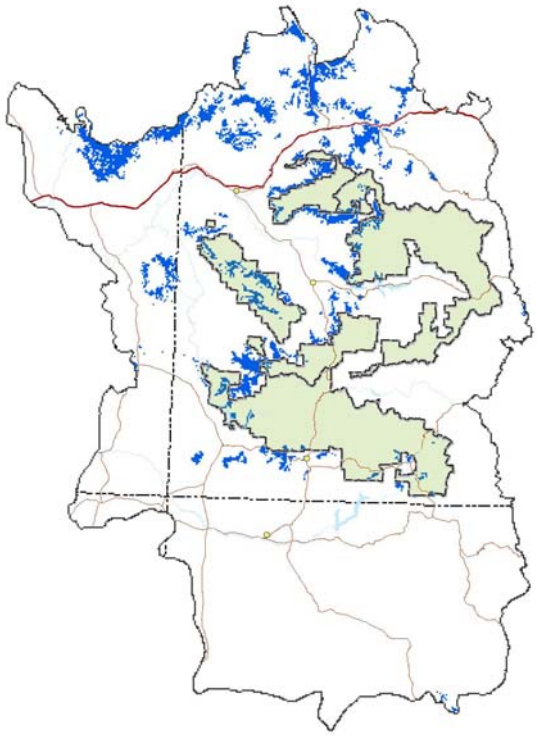
Ponderosa pine forests at the Sub-Regional-scale were not homogenous open park-stands prior to Euro-American settlement, but rather it appears that some dense stands of small

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 percent) of the Sub-Region and 207,282 acres (6.5 percent) within the GMUG and 65,747 acres (3.1 percent) within the SJ National Forest (Figure 8). This vegetation type is often considered part of the Mountain

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



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. 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 pinyon 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.

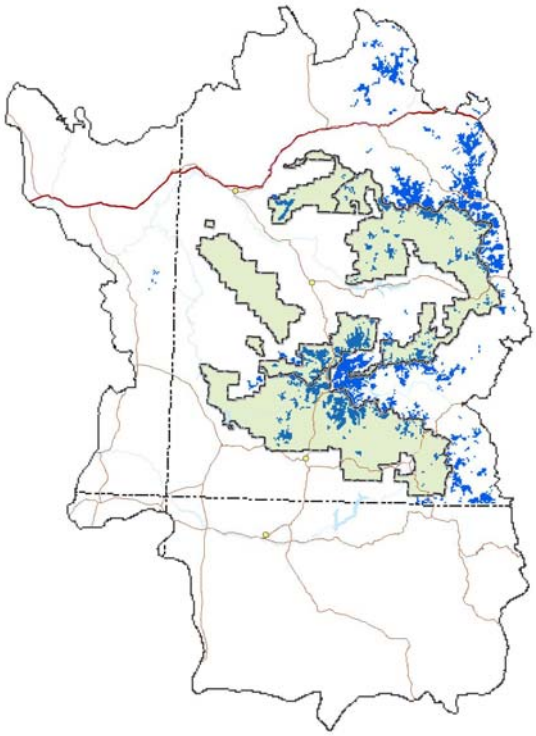
Alpine

The alpine vegetation type compromises 1,606,281 acres (4.7 percent) of the Sub-Region and 397,266 acres (12.5 percent) within the GMUG and 252,406 acres (12.1 percent) within the SJ National Forest. 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 9. The Alpine GAP class in the subregion covers 1,606,281 acres (4.71 percent) of the subregion.

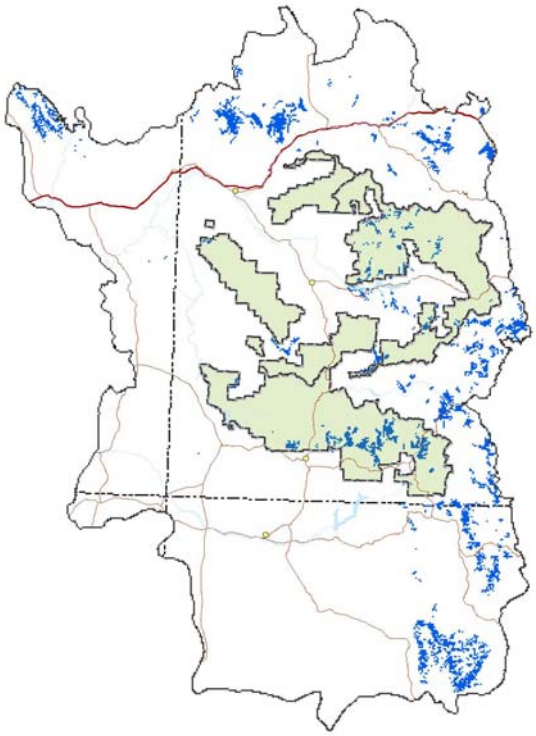


Mixed Conifer

The mixed conifer vegetation type compromises 1,209,315 acres (3.54 percent) of the Sub-Region and 92,296 acres (2.9 percent) within the GMUG and 145,760 acres (7.0 percent) within the SJ National Forest (Figure 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 one 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 10. The Mixed Conifer GAP class in the subregion covers 1,209,315 acres (3.54 percent) of the subregion.



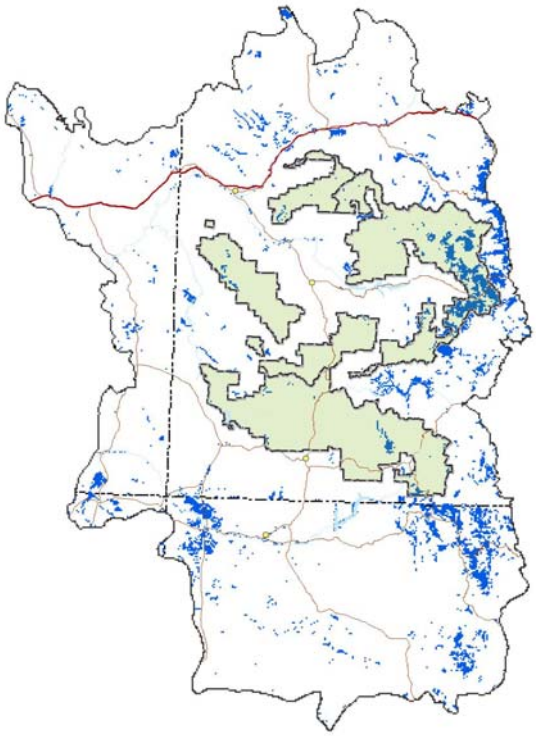
The warm-dry mixed conifer type is found between ~7500-9000 ft (2286-2743 m) and is dominated by ponderosa pine, Douglas-fir, white fir, and occasionally aspen. The abundance of individual tree species is dependent on local site characteristics (soil, aspect, slope, topographic position) and natural 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 year 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 percent of the subregion. These classes are listed and shown in Figures 1 through 10 above. The remaining seven classes, combined cover about five percent of the subregion. Combined the seventeen vegetation classes cover about 95 percent of the subregion. These are described in the following sections and are combined in Figure 11.

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



Mountain Grassland

The mountain grassland vegetation type compromises 589,306 acres (1.7 percent) of the Sub-Region and 6,956 acres within the GMUG and 32,274 acres within the SJ National Forest. 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 percent) of the Sub-Region and 236,601 acres (7.4 percent) within the GMUG and 0 acres within the SJ National Forest. 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 1800s 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 1800s-early 1900s, 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 percent) of the Sub-Region and 13,566 acres within the GMUG and 3,032 acres within the SJ National Forest. 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 pinyon 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 percent) of the Sub-Region and 8,911 acres within the GMUG and 709 acres within the SJ National Forest. 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

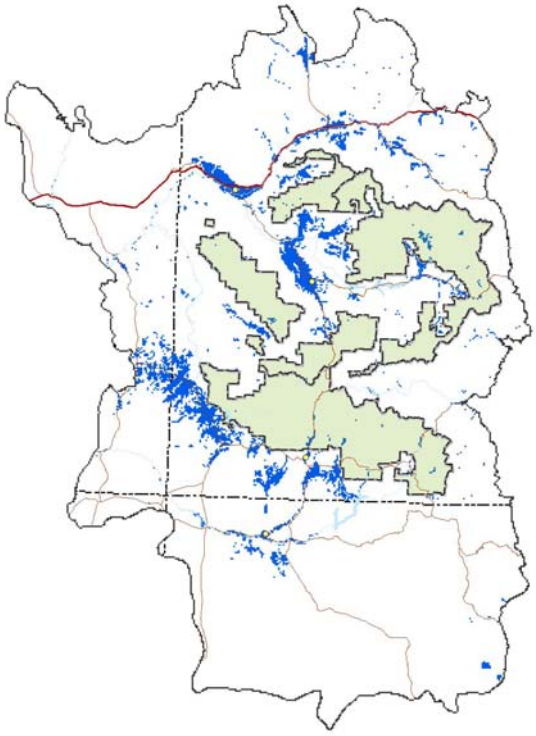
difficult (Redders, 2003a). Within the Sub-Region there are dozens of specific riparian woodland and wetland plant associations, which is beyond the scope of this discussion (see the Forest-Scale Riparian Woodland/Wetland section for descriptive riparian woodland/wetland vegetation types). In addition, Redders (2003a) provides an in-depth discussion of these community types for the San Juan and Johnston (1997) for the GMUG.

Greasewood

The greasewood vegetation type compromises 63,033 acres (0.18 percent) of the Sub-Region and is not present within the GMUG or SJ National Forest. 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 percent of the subregion. The remaining five percent of the subregion are comprised of crops, urban areas and water bodies Figure 12.

Figure 12. Three GAP non-vegetation classes in the subregion combined cover 1,526,301 acres (4.47 percent) 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., pinyon dieback in pinyon-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 1970s 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 one 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 pinyon bark beetle (*Ips confusus)* killing pinyon 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 percent 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 percent of all land located within the Southern Rockies is within two 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 1800s to early 1900s (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 1950s, 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 percent 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 from light grazing pressure in the 1800s to intense pressure in the early-mid

1900s 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 percent of Forest Service land in the Southern Rockes has active grazing allotments and 93 percent 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 1800s 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 1900s. 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.

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