

Historic Variability for the Upland Vegetation of the Shoshone National Forest, Wyoming

Carolyn B. Meyer, Dennis H. Knight, and Gregory K. Dillon
Department of Botany, University of Wyoming
Laramie, WY 82071-3165

Phone: 303-231-9115
E-mail: meyerc@uwyo.edu

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EXECUTIVE SUMMARY

The challenges of sustainable land management have led to an increased emphasis on incorporating the results of science in the decision making process. One approach for accomplishing this objective is through the analysis of the historic range of variability (HRV) for key ecosystem variables that are affected by management activities. The rationale for HRV analyses is that the chances of sustainable forest management are greater if the variation in managed ecosystems is not greater than the range of conditions that are expected at various scales in ecosystems relatively uninfluenced by humans. This report provides an HRV analysis for the Shoshone National Forest (SNF) of northwestern Wyoming—an elongated National Forest of 2.5 million acres (~1 million hectares) extending about 180 miles (288 km) from north to south.

By definition, HRV analyses require the identification of specific variables and an estimation of how those variables fluctuated, at more than one scale, prior to the advent of resource extraction and management by European-Americans. A complete list of the variables that we considered is found in Table 7. The variables include live tree density, dead tree (snag) density, canopy cover, abundance of coarse woody debris, species diversity, fire return intervals, the abundance of various diseases, the proportion of the landscape in different land cover types, and the degree of patchiness in the landscape. We examined variables at two scales, namely, the stand and the landscape, and we separated high-elevation landscapes from low-elevation landscapes. Much of the report pertains to forests dominated by lodgepole pine, Engelmann spruce, and subalpine fir at high elevations, and by Douglas-fir, limber pine, and aspen at low elevations. Our analysis emphasizes forests rather than grasslands and shrublands because more historical information is available for the forests.

Two significant challenges for HRV analyses are 1) selecting a “reference period” for comparison to present day conditions, and 2) making decisions with very little data about the range of variability of a variable during the reference period. We defined the HRV reference period for the SNF as approximately 1600 to 1860. The influence of European-Americans in the SNF was minimal until about 1860. We also compared SNF ecosystems to comparable ecosystems in natural areas relatively uninfluenced by management activities, such as Yellowstone National Park (YNP). When data were lacking, our approach was to first make qualitative though plausible judgments about the conditions that must have existed in the reference period based on recent studies of plant adaptations and current knowledge about ecosystem structure and function. This approach is an example of deductive science. There will always be some uncertainty in such conclusions, but our conclusions for the Medicine Bow and Bighorn National Forests have been evaluated and generally supported by a panel of six anonymous peer reviewers. [The same kind of review is anticipated for this report, but has not yet occurred.] For each of our conclusions about whether a stand or landscape variable is within the HRV or trending away from that norm in managed landscapes, and the Forest as a whole, we indicate whether our confidence level is low, moderate or high. Our analysis cannot be site specific, but the information provided should be useful when developing site-specific management plans.

Our report describes how the forests of the SNF have evolved with regular disturbances, and how the kinds and frequency of disturbances have changed in some areas due to modern management practices. Still, by the year 2000 only ~3% of the forested area had been harvested, mostly in a relatively small area on the Wind River District (~200 km² or ~77 square miles). Fire suppression and livestock grazing, and possibly exotic plants, have had more widespread effects, but much of the SNF is wilderness, has a very rugged topography, and is not accessible by road. Livestock grazing has been intensive and widespread in the past, but in most areas the numbers of sheep and cattle have declined greatly in the last 20 years. Overall, compared to other National Forests in Wyoming, much of the SNF has been changed relatively little by European-Americans.

Section 7 presents a summary of our conclusions for each variable. The following variables appear to have exceeded their HRV in some areas affected by management activities:

At high-elevation forests (35% of SNF), the following variables, estimated at the stand scale, appear to be within the HRV where timber harvesting has not occurred (confidence level is indicated in parentheses):

Fire return interval, intensity, and size (low, moderate, and low, respectively)

Insect outbreaks (moderate)

Abundance of diseases, except for dwarf mistletoe in some partial-cut areas (low) and where the introduced white pine blister rust is found (high)

Blowdowns (high)

Tree, sapling and seedling density (high)

Canopy cover and canopy gap density (low)

Snag density, coarse woody debris abundance, and mineral soil disruption or displacement (high)

Age- and size-class structure of most stands (moderate)

Understory plant density and cover (moderate)

Plant species diversity (moderate)

Forest floor depth (high)

Similarly, the following variables estimated at the landscape scale at high elevations appear to be within the HRV for the entire National Forest:

Fire return interval, size, and intensity (low, low, and moderate, respectively)

Insect outbreaks (moderate)

Abundance of diseases (low) except where white pine blister rust is found (high)

Blowdowns (high)

Tree, sapling and seedling density (moderate)

Canopy gap density and cover and mineral soil disruption or displacement (moderate)

Snag density and coarse woody debris abundance (high)

Age- and size-class structure of most stands (moderate)

Understory density and cover (moderate)

Plant species diversity (moderate)

Forest floor depth (moderate)

Number and proportion of land cover types (moderate)

Forest/non-forest ratio (high)

Proportion of forests in different successional stands and old forests (moderate)

Proportion in low canopy cover (high)

Proportion with high snag and coarse woody debris densities (moderate)

The amount of edge, interior forest, and patch size and shape (moderate)

Rate of patch formation (high)

The situation in low-elevation forests (19% of SNF) is different because the land is generally more accessible and the historic low frequency and mixed-severity fire regimes are more subject to modification through management. **Still, at low elevations, the following stand variables appear to be within the HRV:**

Insect outbreaks (low)

Abundance and composition of diseases (moderate), except where the introduced white pine blister rust occurs (high)

Blowdowns (high)

Snag density and abundance of coarse woody debris, except where timber harvesting has occurred (moderate)

Similarly, the following low-elevation landscape variables appear to be within the HRV for the SNF as a whole:

Insect outbreaks (low)

Abundance and composition of diseases (moderate), except where the introduced white pine blister rust occurs (high)

Blowdowns (high)

Proportion of landscape with old trees (moderate)

Proportion of land with high-density snags and coarse woody debris (low)

Proportion of land in different land cover types in forests (low)

For aspen stands (1% of SNF), the following variables appear to be within the HRV at high and low elevations and at both the stand and landscape scales:

Insect outbreaks (moderate)

Disease abundance, except near campgrounds (low)

Blowdowns (high)

Conifer density in understory (low)

Understory plant composition (low), except in areas with exotic plants (high)

Extent of aspen in the landscape (low)

As noted, very little information is available for drawing conclusions about the HRV of grassland, meadow and shrubland variables on the SNF (33% of SNF). However, fire return interval, size, and intensity appear to be within the HRV (high confidence), probably at both the stand and landscape scales.

In contrast, the following variables probably are beyond their HRV within forest stands at high elevations where timber harvesting has occurred (~4% of the high-elevation forests on the SNF by the year 2000, primarily in the Wind River District):

1. Canopy cover probably is lower and the size and density of canopy gaps probably is higher in harvested stands than the HRV for these variables in

- unmanaged forests of comparable age and site conditions, due to selective and shelterwood cuts and fewer standing-dead trees (moderate confidence)
2. Snag density and the amount of coarse woody debris are lower than the HRV for unmanaged stands of comparable age and site conditions (high confidence). Whole-tree yarding can accentuate this variation by concentrating coarse woody debris and slash at landings in a way that has not occurred previously.
 3. Due to some methods of mechanical slash treatment, the intensity of soil scarification is beyond the HRV in some areas (high confidence). Burning can remove the forest floor, but the depth of soil disturbance typically is less than following mechanical harvesting. Similarly, skid trails probably have compacted soils beyond their HRV, at least temporarily.
 4. Age- and size-structure of managed stands within ravines or on some north or leeward slopes is skewed toward smaller and younger trees than would have occurred during the HRV period, because of harvesting where fires would have been less likely to burn (moderate confidence).
 5. Dwarf mistletoe abundance is probably above its HRV in stands subjected to partial timber harvests, but could be less than the HRV in clearcut stands (low confidence).
 6. White pine blister rust outbreaks cause some variables in affected stands of whitebark and limber pine to exceed the HRV, as this disease is not native (high confidence).

Across forested landscapes at high elevations where timber harvesting has occurred, primarily on the Wind River District (7% of SNF):

7. The estimated rotation times for timber harvest of forests judged suitable for harvesting are generally shorter, on average, than the estimated site-specific mean intervals between natural disturbances in high-elevation forests (high confidence). Old-growth forest is reduced, particularly in areas that have had long fire-free intervals, such as in valley bottoms and on some north or leeward

slopes (moderate confidence). Thus, ecosystem characteristics that depend on periods of forest growth longer than ~140 years may be lost in some areas.

8. The rate of patch formation and the size of disturbances across high-elevation landscapes are outside the HRV (high confidence). Timber harvest produces patches more frequently than natural disturbances, and mid-sized patches can become unusually common. Natural fires produce many small patches and a few very large patches in such areas.
9. Due to roads and clearcutting, the amount of edge is now higher and the amount of interior forest is lower than prior to the 1900s in small areas on the Wind River District (moderate confidence). Correlated with more edge is a decline in the average patch size, the development of a more simple patch shape, and a higher, more uniform level of landscape patchiness than would have occurred during the reference period.

Within stands at low elevations affected by timber harvest (<1% of the forested land), fire suppression, and livestock grazing:

10. Fire suppression and livestock grazing have led to the development of low-elevation forest stands with higher densities of young trees (high confidence), greater canopy cover (high confidence), and less understory plant cover (low confidence) than usually occurred on many sites before 1860. Understory composition has also changed in some areas (moderate confidence, partly due to more competition from trees and exotic plants), and forest floor depth has increased (high confidence).
11. By removing large trees and downed wood, harvesting has lowered the abundance of snags and coarse woody debris in managed stands of low-elevation forests (moderate confidence, but only in less than 1% of the forested area). Harvesting, where it has occurred, also has led to an increase in sapling density and a reduction in the average tree diameter, thus creating younger, more uniform-sized stands than existed before harvest (high confidence).
12. Mean fire return interval has become longer at low elevations than the HRV (high confidence), and consequently fire intensity is more likely to be higher due

to the amount and continuity of fuels, which enables more stand-replacing fires (low confidence).

13. White pine blister rust outbreaks cause some variables in stands with limber pine to exceed the HRV, as this disease is not native (high confidence).

Across forested landscapes at low elevations:

14. Similar to the stand-level, the size and intensity of fires have declined due to fire suppression at low elevations, and the amount and continuity of fuels is above the HRV in some areas, leading to the potential for more stand-replacing fires than occurred prior to 1860 (low confidence). Fire suppression has reduced the fire frequency (moderate confidence) and level of interspersion of tree stands with grasslands (low confidence), and average tree/sapling density has increased above the HRV (moderate confidence). Concomitantly, there is a more uniform canopy cover (moderate confidence), less understory vegetation (moderate confidence), and greater forest floor depth (moderate confidence) over larger areas than the HRV
15. Due to fire suppression, trees are smaller and younger on average than the HRV (moderate confidence)
16. White pine blister rust was absent during the HRV reference period, but is present today in stands of limber pine (high confidence).

In aspen forests (1% of SNF):

17. Livestock and/or native ungulate grazing and browsing in some areas have reduced aspen densities, but probably not to the point where this variable is beyond the HRV. However, exotic plants have affected the understory vegetation to a point where associated variables are exceeding their HRV (high confidence).

18. Fire suppression has reduced fire size and lengthened fire return intervals, thereby enabling an increase in conifer density in some areas to a level that is higher than the HRV (moderate confidence).
19. Due to human wounding of trees in recreational areas, some stands have an incidence of disease that is beyond the HRV (moderate confidence).

In non-forest vegetation (33% of SNF):

20. Some meadows, grasslands, and shrublands probably have been grazed more during the last century than during the HRV period, with the potential for an increase in forb abundance and soil erosion (moderate confidence). Deviations from the HRV also are caused by the invasion of exotic plants that displace native species and cause changes in other ecosystem properties (high confidence).

Frequently there has been a lack of data for the ecosystem variables we have addressed in this report (Table 7). Such deficiencies should be considered when identifying priorities for future research or monitoring. Research will also be required if decisions about the severity of a potential problem cannot be made without more information or if a consensus cannot be reached on ways of restoring the HRV where this is considered to be an important objective.

1. INTRODUCTION

Managers of public lands are now charged with achieving the societal goals of conserving biological diversity and maintaining ecosystem services while contributing to national needs for various other natural resources—an endeavor commonly referred to as ecosystem management (Covington and DeBano 1994, Grumbine 1994, Kaufmann and others 1994, Christensen and others 1996, Vogt and others 1997, Kaufmann and others 1998, Landres and others 1999). The proponents of ecosystem management recognize that our current knowledge of ecosystems is incomplete (Christensen and others 1996) but that resource managers should use the best scientific information available. An analysis of the historic range of variability (HRV) for key ecosystem variables is one means of synthesizing the available ecological information for an area. In this report we present an HRV analysis for the Shoshone National Forest (SNF) in north central Wyoming.

One example of how attitudes about resource management have changed in the last fifty years is greater emphasis on resilience rather than stability (in terms of resistance to change; Holling and Meffe 1996). Ecosystems have developed with constant change and fluctuation (Botkin 1990, Landres and others 1999, and others), with disturbances such as those caused by fire and insect epidemics having influenced ecosystems for millennia in the Rocky Mountain region (Knight 1987, Fall 1997). Managers are now using the variability caused by natural disturbances and other phenomena as a guideline. In outlining the conceptual framework of ecosystem management for the Forest Service, Kaufmann and others (1994) concluded, “. . . the most sound basis for ecosystem management is to assure that the variation characterizing [managed] ecosystems includes the range of conditions that are expected at various scales in ecosystems uninfluenced by humans.” The focus is on variability, not stability.

Various terms have been used in previous reports on variability, including "range of natural variability" (Swanson and others 1993), "natural variation" (Holling and Meffe 1996), "reference conditions" (Kaufmann and others 1994), "reference variability" (Manley and others 1995), and "historic range of variability" (Morgan and others 1994). We have adopted the term "historic range of variability" (HRV) because it avoids the potential problems associated with the word "natural." Regardless of the language, we interpret HRV as the spatial and temporal variation in various ecosystem characteristics when the influences of European-Americans were minimal. For our analysis, we focused on the period from 1600 to 1860, when European-American influence was minimal (Cochrane and others 1988).

We adopted four principles as guidelines. First, variables should be evaluated over at least two spatial scales. The variability that occurs in small tracts of land, commonly referred to as "stands," is quite different from that which occurs over larger areas, referred to as "landscapes." Stands are tracts of forest or other vegetation that are relatively homogeneous in terms of environment, plant species composition, and disturbance history. They generally occupy an area ranging from a few hectares to several hundred hectares. In contrast, "landscapes" are larger, usually ranging from several km² to several thousand km² or more, and they have considerable heterogeneity. The HRV of some variables, such as dead standing trees (snags), is typically higher at the stand scale than at the landscape scale (Fig. 1). Of course, some variables are not amenable to discussion in a practical way at both scales. For example, the proportion of a landscape in different cover types is not meaningful at the stand scale.

Just as there is more than one stand in a landscape, there are several landscape types in a National Forest. For this report, we separated high-elevation landscapes from low-elevation landscapes and defined them by forest type, not specific elevation. We evaluated four general vegetation types separately: 1) lodgepole pine/spruce-fir forest (*Pinus contorta*/*Picea engelmannii*-*Abies lasiocarpa*) and whitebark pine woodland (*Pinus albicaulis*), referred to as high-elevation forest; 2) Douglas-fir (*Pseudotsuga menziesii*) and limber pine (*Pinus flexilis*) woodlands, referred to as low-elevation forest; 3) quaking aspen (*Populus tremuloides*)

woodlands; and 4) non-forest vegetation. Our analysis emphasizes forests rather than grasslands and shrublands because much more historical information is available for the forests.

The second principle is that HRV analyses should consider variability in space as well as time. This principle is difficult to apply because historical data are rarely available for more than one or two watersheds or landscapes. However, it is possible to characterize spatial variability in vegetation patterns across watersheds on the SNF, as it exists today, and to compare that variability to the range of expected conditions during the HRV period based on our current understanding of ecosystem dynamics. By necessity, HRV analyses must often be deductive in nature—drawing conclusions about specific variables based on general knowledge about ecosystem

The third principle affecting HRV analyses is that the effects of climate change should be distinguished from the effects of human activities. There is no question that the climate has changed since Rocky Mountain glaciers began to recede about 10,000 years ago. It also is clear that the climate of the 1600s, 1700s, and much of the 1800s was different than at the present time (Millar and Woolfenden 1999). Working on the west side of the nearby Bighorn National Forest, Gray (2003) and Gray and others (2004) concluded from tree-ring data that, while the mean annual precipitation was about the same during the 1600s and 1700s as it is today, there are now fewer extended wet and dry periods than in previous centuries and the extremes of the wet and dry periods in the 1900s are less. Moreover, preliminary research suggests that the climate of areas near the SNF, such as the Bighorn Mountains, began to warm in the late 1800s (C. Fastie, Ph.D., paleoecologist, personal communication). While the HRV estimates of some ecosystem variables for the reference period 1600-1860 may be less applicable to present conditions because the climate has changed, we believe they still provide a useful context for management decisions. Distinguishing the effects of climate change from the effects of European-Americans is a difficult but important challenge for HRV reports.

The fourth principle is that the “variability” of an HRV can be characterized in several ways. To illustrate, variability can be thought of as the absolute range of an ecosystem variable or parameter, in which case extreme and rare conditions are

often included (Fig. 2). Alternatively, the HRV can be based on an estimate of the range of means (or frequency) for a variable over several consecutive periods of time, an approach that ignores extremes (Fig. 3). HRV analyses based on average conditions tend to produce a narrower range than when considering extremes (Fig. 1). A third approach involves the calculation of the standard deviation. Unfortunately, data availability and sample sizes are rarely, if ever, adequate for statistical analyses. In this report, we use the range of means approach, conceptually if not quantitatively.

The objective of this report is to characterize the HRV and compare existing conditions (or conditions after European-Americans settled in the area) to the HRV to determine if any ecological variables have been pushed beyond their historic variability by management activities. To characterize the HRV, a reference period first must be specified. Manley and others (1995) indicated that it would be ideal if the period could cover an evolutionary time scale (i.e., as long as possible within the past 10,000 years or so). It would be good to use a long, evolutionary time scale, but climate change and glaciation in the SNF during the last 10,000 years or more would create extremely broad ranges of variability that would be difficult to apply to present day management decisions. Also, it is difficult enough to estimate variability during the last few centuries, as we have tried to do.

Notably, an evolutionary perspective is adopted in our report, but in a different sense than implied by Manley and others (1995). Some of our interpretations are based on the prevailing consensus about the environmental conditions that must have led to the development of certain adaptations exhibited by the dominant plants of the SNF today. Such adaptations surely influence where species survive today. For example, the thick bark of ponderosa pine (*Pinus ponderosa*) suggests that the environment where it occurs, or has occurred in the past, is characterized by periodic surface fires. Similarly, the thin bark and serotinous cones of lodgepole pine suggest that the tree is characterized by periodic stand-replacing crown fires.

Estimates of HRV are inevitably limited by our ability to interpret the nature of past ecosystems and determine what is "natural" or "normal" (Swanson and others 1993, Holling and Meffe 1996). Because ecosystems change dramatically over a

period of a thousand years or more, or even a few centuries, we only capture a "snapshot in time" if our reference period is not long enough (Swanson and others 1993). For such reasons, estimates of HRV are not a panacea. Rather, they serve as one of several tools to aid managers in achieving the societal goal of conserving biological diversity while providing the benefits of various natural resources.

By considering the HRV for tangible, meaningful variables during defined time periods, land managers have an improved context in which to evaluate their decisions (Manley and others 1995). If a management activity pushes an ecosystem variable to a higher or lower level than ever before, based on the kind of evidence included in HRV reports, then there is reason to evaluate the environmental impacts of that activity—or even consider adopting a different approach. When quantitative data are not available, the goal of specifying variability must be done using qualitative information. For some observers, this approach is not satisfactory. However, deductive analyses can provide insights that are not available in any other way and which improve the chance that management decisions will not impede an ecosystem's inherent resilience following the inevitable disturbances that will occur, whether caused by fires, wind, insects and pathogens, or humans.

Landres and others (1999) concluded that, properly done, HRV analyses provide context and guidance for managing ecological systems. They also emphasized, however, that it is appropriate and unavoidable for HRV reports to include professional judgments, even though based on limited data. Given that fact, they recommended that the authors of such reports should specify their assumptions and value judgments. Briefly, our assumptions and biases are 1) there is much to be learned about how to manage National Forests within the context of ecosystem management, 2) there is value in the periodic evaluation of traditional management approaches to determine if they are appropriate for meeting an objective, 3) some parts of a National Forest are suitable for the careful production of commodities, while others are not—a policy that has been implemented for many years through the designation of, for example, lands suitable and not suitable for timber harvesting—and 4) decisions about the management of specific areas cannot be made based solely on a single report such as this one.

HRV analyses also should not be viewed as a step toward creating "pristine" conditions (Swanson and others 1993, Allen and Hoekstra 1994). Ecosystems may, in fact, be sustainable far outside the HRV for a variable if society is able and willing to engineer functions once carried out naturally (Allen and Hoekstra 1994). The question remains, however, whether or not we know enough to create substitutes for natural processes now done by the thousands of organisms, representing innumerable species, that occupy any ecosystem—and can afford to pay for them. Moreover, National Forests already have some features that are unprecedented and clearly outside the HRV, such as roads, dams, and exotic species (Swanson and others 1993). Restoration of the HRV may not be practical or desirable, or even possible, in some situations.

In this report for the SNF, we have attempted to provide 1) an estimate of the HRV for key elements of terrestrial ecosystems, including quantitative ranges for specific variables whenever possible; and 2) a comparison of our HRV analysis with current conditions on the SNF or conditions since European-Americans arrived. Our focus is on upland vegetation dynamics and plant-related ecosystem variables at the stand and landscape scales. After describing our methods, we describe the physical, ecological, and cultural characteristics of the SNF before proceeding with an HRV analysis for key variables in high-elevation landscapes, low-elevation landscapes, aspen forest, and non-forest vegetation. Our conclusions about human-caused deviations from the HRV are summarized in the Executive Summary, Table 7, and Section 7.

As noted, we emphasize forests over meadows, grasslands and shrublands because far more data are available for forested areas. Grazing certainly affects a large proportion of the SNF, but interpreting the historic development and variability of grassland and shrubland variables for our reference period (1600-1860) is much more difficult than for forests where the tree ring record provides insights into the past. Also, more ecosystem research has been done on forests than on non-forested areas, thereby providing a better basis for predicting the HRV through a deductive analysis. The HRV of riparian vegetation is of great importance and has more data than non-forests, but this landscape type is beyond the scope of our report.

2. METHODS

Manley and others (1995) suggested that estimating the HRV for biological systems should include the following steps: 1) determine key ecosystem elements (i.e., components, structures, and processes), 2) identify measurable environmental indicators for these elements (e.g., tree density and mean disturbance return interval), and 3) estimate values for the indicators over the selected HRV reference period. We tried to be more inclusive than exclusive in selecting key elements, as the importance of some ecosystem features or processes may be latent until the ecosystem is subjected to some kind of stress (Holling and Meffe 1996).

The challenge of HRV analyses is to make decisions about variables during the HRV period based on very little data. HRV studies are a relatively new endeavor and methods are changing rapidly as we gain new information on the best approaches to use. Our approach was to first make qualitative, though plausible, judgments about the conditions that must have existed in the past based on recent quantitative studies of plant adaptations and current knowledge about ecosystem structure and function. Knowledge about the organisms that exist in an ecosystem today provides clues about environmental conditions that probably existed in the past, and the general principles of ecosystem structure and function provide a basis for drawing tentative conclusions, deductively, about a specific area. Our conclusions can be considered as hypotheses requiring further testing in the future. Each conclusion is followed by an indication of our confidence level (low, moderate and high).

Second, we made judgments about the past from different kinds of historical records, such as contained in tree rings and the journals and photographs of early explorers. Specifically, we used 1) historic records and studies covering approximately the last century that provided quantitative or semi-quantitative information; 2) historical accounts, which provided mostly qualitative information back to the 1800s; and 3) reconstructive studies from tree-rings, pollen analysis and other

methods that provide the only quantitative information for periods prior to the 1800s (Kaufmann and others 1998).

Finally, valuable information was obtained from studies on the long-term dynamics of ecosystems in modern reference areas that have not been heavily influenced by humans, such as wilderness areas and some National Parks in the region. The research done in Yellowstone National Park (YNP) has been especially useful. The climate is similar to the SNF, though YNP is generally cooler and more moist than the eastern SNF and drier than the western SNF. The geological history is different in some areas, but the predominant vegetation is similar. Using data from any area as a proxy for another must be done with caution, but to ignore such pertinent information needlessly limits the applicability of research. Notably, no landscape is entirely free from human influences. Native Americans have caused some changes for thousands of years, and even our largest National Parks have been affected by the technological impacts that now characterize the earth.

Our analysis is based primarily on the compilation and synthesis of information on upland vegetation from published and unpublished sources relating to the history and ecology of the SNF specifically and the Rocky Mountain region in general. Funds were inadequate for initiating new field work or for incorporating an analysis of wetland vegetation. Information was derived from all available sources, including journals, books, USDA Forest Service publications, SNF reports, and SNF databases. Several sources were especially useful in quantifying historic and present conditions specific to the SNF, namely the 2000 Resource Information System (RIS) and the 2000 fire database of the SNF, the Forest Inventory and Analysis (FIA) program of the USFS (www.fia.fs.fed.us), and the grizzly bear habitat database. While there are some limitations to using these databases (Appendix, Table A-1), they provide the most comprehensive source of spatial information. We also examined old photographs, including those assembled by Meagher and Houston (1998) for YNP, and we conducted interviews of SNF personnel and others to gain a better understanding of the management and ecology of the SNF.

For this report, we have drawn conclusions about whether existing conditions have changed from conditions during the HRV reference period, from about 1600 to

1860, due to the management activities of European-Americans. We define existing conditions (EC) as conditions occurring after 1860, though primarily during the last 50 to 100 years. In general, we defined HRV as the range of means of a variable for several consecutive time periods during the period 1600 to 1860 (Fig. 3). We concluded that a change from the HRV probably had occurred if we felt there was evidence to suggest that the range of means for the variable over a series of similar, consecutive time periods during the HRV period probably would be different than the existing conditions (Fig. 3).

Unfortunately, only one mean is usually reported in the literature for the HRV period, if that, rather than the several required to obtain a range of means. In those cases, we estimated whether or not the mean during existing conditions probably differed by more than ~75% from the mean during the HRV period (Fig. 4). If a defensible rationale could be made for this level of change, we concluded that existing conditions are outside the HRV. This conclusion, based on only one mean, would not be possible if conditions vary substantially from half-century to half-century, or if they vary hardly at all. Additional research is required to evaluate this approach.

The above examples assume data for the HRV period are available for the SNF. In reality, pertinent HRV data are usually available only in YNP and other areas nearby where a considerable amount of research has been done. To partially offset this deficiency, we substituted "proxy" data for actual data on the SNF during the HRV period, as follows:

- 1) Data were assembled for the HRV period from studies in other places in the Rocky Mountains.
- 2) For many analyses, we used existing conditions (20th century) in YNP as the conditions on the SNF during the HRV period. The Park's vegetation has been less strongly influenced by European-American activities and probably reflects historic patterns that occurred in the SNF. In some cases involving fire regimes, we used Yellowstone data for the period in which fire suppression was attempted in the Park.

- 3) We used the pattern (20th century) of "natural" disturbances (e.g., blowdowns, stand-replacing fires) as the pattern during the HRV period for comparison to patterns created by artificial disturbances such as timber harvesting. Admittedly, some of the natural disturbances have been modified by humans (e.g., high elevation fires probably are smaller on average than during the HRV). Therefore, we only use this method when the modification in the pattern probably is relatively small compared to the modification caused by artificial disturbances.
- 4) We substituted space for time in some landscape analyses. In other words, the range of conditions at a point in time across all watersheds was assumed to reflect the range of conditions during the HRV period. This assumes the timing of disturbance patterns among watersheds is not correlated and that each watershed today is a snapshot of conditions that may have occurred in just one of the watersheds in the past. This is not a strong assumption because events such as large fires often cross watershed boundaries. Also, different soils, topography, and climate cause differences in the vegetation of watersheds. Consequently, we used this approach rarely and only when our conclusions could be supported by other information.

For the qualitative analyses, we used the same approach outlined above, but we substituted our knowledge of ecosystem processes and the effects of human-developments to estimate the direction and magnitude of any changes from the HRV. We describe the justification for each decision.

For each variable, we attempted to address 1) its HRV at the stand and landscape scale, 2) its spatial variability across the landscape as well as its temporal variability, when such data were available, 3) the influence of climate change when possible, and 4) our confidence level (low, medium and high) in concluding whether the variable under existing conditions is within or beyond the HRV. Our conclusions and confidence levels are summarized in Table 7, which is part of our summary for this report (Section 7).

Because it is easy to misinterpret how HRVs are estimated at the stand and landscape scales, we provide the following two illustrations for clarification of our approach. At the stand scale, the HRV focuses on changes through time, which ecological theory suggests would differ for every stand even if two stands are within the same vegetation type. A continuum of HRVs should be expected, just as there is a continuum of species composition. No two stands are exactly alike because of differences, however slight, in topography, soils, climate, disturbance history, seed availability, sprouting capacity of surviving plants, rate of successional development, and other factors. We cannot consider all the various stand types in our analyses, but general statements about changes in HRV through time are possible. For example, we can say that most pre-commercial thinning prescriptions would lower canopy cover and increase the number of canopy gaps beyond the individual HRV for canopy cover and canopy gap number in each stand that is thinned, at least for several decades, because no ecosystem processes mimic such an activity. Thus, the HRV of canopy cover for each stand averaged through those several decades would be reduced. When such a directional pattern is true of most but not all stands, we acknowledge the exceptions.

At the landscape scale, we looked for the same patterns over time. However, we averaged the variable across all stands to assess the HRV over a large spatial scale, as well as through time. As Figure 1 illustrates, the HRV narrows as the scale changes from the stand level to the landscape level. As an example, consider snag density. A stand may have many snags after a crown fire, but then none at all (or very few) if the snags fall before new snags are formed. Thus, the snag densities for the individual stand could range from zero to extremely abundant. However, it is unlikely that all snags across a landscape would fall before new ones are formed, or that all the live trees on that landscape would be converted to snags by a fire. Thus, snag density at the landscape scale would range from moderately low to moderately high. There would always be some snags in the landscape. A logical conclusion is that the HRV for a variable is dependent on the amount of land area being considered (Fig. 1). In other words, the probability of finding a stand or landscape with zero snags declines as the land area in the stand or landscape increases. This

is true simply because the probability of tree mortality somewhere in the area increases as the number of trees increases.

Regrettably, data for specific variables at the landscape scale are less common than for individual stands. Therefore, we sometimes used variables such as the percent of the landscape in stands that probably have high snag density instead of mean snag density for a landscape. This approach supplements our landscape-level, qualitative estimate of snag density.

Our conclusions generally pertain to the SNF as a whole. Notably, there is significant variation within the SNF from north to south, especially with regard to geology, and it would have been ideal to address the HRV for each of the five districts separately (Clark's Fork, Wapiti, Greybull, Wind River, and Washakie; Fig. 5) or by geological substrate. However, the range in elevation is about the same throughout the Forest (though the highest elevations are in the southern Wind River and Washakie Districts; Fig. 6) and significant differences from north to south in vegetation and ecosystem dynamics were not identified (Table 1).

3. THE SHOSHONE NATIONAL FOREST

3.1 PHYSICAL SETTING

The Shoshone National Forest (SNF) covers 982,982 ha (nearly 2.5 million acres) and is located in northwest Wyoming on three mountain ranges--the east side of the north/south trending Absaroka and Wind River Ranges (where the western boundary usually follows the Continental Divide) and, in the north, a small part of the Beartooth Mountain Range. The Forest extends more than 180 miles from north to south, and the elevation ranges from 1,402 to 4,221 (4,599 to 13,845 ft) (Figs. 5 and 6). The majority of the SNF is in the upper Missouri River Basin, which is subdivided by the Wind, Big Horn, and Clarks Fork Basins, from which the rivers flow into the Yellowstone River. The southern tip of the Forest is in the Sweetwater Basin, which drains into the Platte River (USDA Forest Service 1986). Located just east of YNP and the Bridger-Teton National Forest, the SNF is part of the Greater Yellowstone

Ecosystem. As such, wildlife and recreation management are very important parts of the Forest's mission. Fifty-five percent of the SNF has been designated by Congress as wilderness (Fig. 7).

The geology varies by mountain range. The Beartooth Range in the north is a northwest-trending, uplifted block of Precambrian rocks (granite, gneiss, and schist) that extends north into Montana (Polk and others 1981, Roberts and others 2001). This range is a gently rolling plateau incised by deep canyons. On the east flank of the range are highly faulted Paleozoic and Mesozoic sedimentary rocks (mostly limestone) that are nearly vertical palisades on the edge of the plateau. Three stages of Pleistocene glaciation have been provisionally recognized in the Beartooth Range. U-shaped canyons, lateral and terminal moraines, and outwash gravel deposits are prominent, and alpine lakes in cirques (Fig. 8) are numerous—though not as abundant as in the Wind River Range to the south.

The Clarks Fork of the Yellowstone River (a Wild and Scenic River) divides the Beartooth Range from the Absaroka Range to the south. The Absaroka Range is a broadly dissected plateau of Eocene volcanic rocks (Polk and others 1981, Roberts and others 2001) entrenched by a series of canyons varying between 610 to 1,525 m (2,001 to 5,002 ft) in depths. The North Fork of the Shoshone River divides the Absaroka Range into northern and southern sections. The northern section has high peaks and steep ridges, while the southern section is better characterized as a plateau above treeline dissected by canyons. The highest peak in the Absaroka Range is Francs Peak (4,018 m or 13,153 ft), located in the southern section. Often designated the Absaroka volcanic field, the Absaroka Range was created by episodic volcanism, periods of erosion, and local folding and faulting. Much of the volcanic rock (andesite, basalt, tuff, and breccia) became reworked by streams and mudflows, resulting in layers of sandstones and conglomerates in the southern half of the range. The entire volcanic field is bounded to the north by the Precambrian rocks in the Beartooth Range, to the east by the Bighorn Basin, to the south by an eroded margin of volcanic rocks, and to the west by rhyolite and basalt deposited during younger eruptions in YNP. The erosion of volcanic rocks to the south exposed the Washakie Range (Love 1939, Keefer 1970), which is considered part of the Absaroka Range

today. At least two stages of glaciation occurred in the Absaroka area and broad U-shaped valleys are characteristic. However, cirques seldom contain lakes because the highly erodible rocks formed valleys rather than lake basins. Overall, glaciers were apparently not as widespread in the Absaroka Mountains as in the Wind River and Beartooth Ranges, though small glacial deposits are found near Brooks Lake, the lower DuNoir Creek, and the middle reaches of Horse Creek (Keefer 1970).

Southward, the Wind River Range follows a large, northwest thrust fault on an anticline about 24 km wide by 250 km long (15 by 155 miles). Its highest peak (Gannet Peak) is 4,221 m, the highest in Wyoming. The range rises about 2,000 to 2500 m (6,560 to 8,200 ft) above the Wind River and Green River basins (Polk and others 1981, Roberts and others 2001). Like the exhumed Washakie Range, its core is predominantly Precambrian granite, granodiorites, and gneisses, with some schists and other metamorphic rocks exposed at the southern end. Sedimentary rocks (sandstone, shale, limestone, dolomite), ranging in age from Cambrian to Late Cretaceous, are exposed on the northeast flank (Fig. 8) where they are overlapped by younger rocks deposited during the Eocene to Pleistocene periods.

Much of the Wind River Range was glaciated several times during the Pleistocene Epoch (Richmond 1965). The oldest and largest glaciation (the Buffalo stade) covered about 13,000 km² (~5,000 square miles) in the Wind River Mountains and adjacent areas, and was over 1,000 m (3,280 ft) deep in some places. The next oldest glaciation (Bull Lake) covered about 3800 km² (~1,467 square miles), was more than 900 m (2,952 ft) thick in canyons, and was 300 to 500 m (984 to 1,640 ft) thick near the summit. Peaks above 3,350 m (10,988 ft) projected through the ice. Ice sheets extended 8 to 24 km (5 to 15 miles) down canyons on the east slope of the range. The youngest glaciation (Pinedale) started at 3200 m (10,496 ft) elevation, covered about 2,000 km² of land (~772 square miles), and was 250 to 300 m (820 to 984 ft) thick. Consequently, deep U-shaped valleys (about one km (0.62 mile) wide with 600 m (1,968 ft) side walls) extend from both sides of the Continental Divide, which coincides with the crest of the range. Many valleys contain cirque basins, some with small lakes. Large terminal and lateral moraines occur to the north, west, and east of the Wind River Mountains, commonly forming deep lakes.

Not surprisingly, the topography and associated soils of the SNF are diverse (USDA Forest Service 1986). About 25% of the land area is comprised of plateaus, glacial cirques, knife ridges, or steep walls near or above timberline; 20% is very steep, non-alpine rockland; and another 20% is comprised of moderately dissected and forested mountain slopes, with slopes often exceeding 40%. The remaining 35% is of low relief, has relatively deep soils and is quite heterogeneous with granitic uplands scoured by glacial ice, glacially scoured sedimentary landforms, valley bottoms, alluvial fans, landslide debris, and extensive deposits of glacial till (Fig. 8). The soils of the till, river terraces, alluvial fans and granitic uplands are coarse textured. Others are more fine-textured. Streams are abundant (Fig. 9) and most are free-flowing, and in some areas natural lakes are abundant, especially in the Beartooth and high-elevations of the Wind River Mountains. About 0.1% of the SNF is considered "badlands," with little soil development and high erosion rates. An extensive area of badlands—the Wind River Badlands—occurs outside the SNF to the east of Dubois (Keefer 1970, Ilians and others 1993, Kinter 2000)

As in other parts of the Rocky Mountains, the climate of the SNF varies greatly with elevation and topographic position (Fig. 10). Moisture, brought in by northwest winds flowing up and over the continental divide, falls as rain or snow depending on the elevation and season. Mean annual precipitation ranges from about 25 cm in some of the drier valleys to 178 cm along the crest of the Absaroka Range (Martner 1986). At the higher elevations, two-thirds or more of the precipitation falls as snow. Temperature is also variable across the National Forest, but all stations show average maximum and minimum temperatures occur in July and January, respectively. Monthly July averages range between 10 to 15°C, and monthly January averages range between –5 to –10°C. Some weather stations receive maximum precipitation in May or June, while others receive the maximum in January as snow. Heavy snow accumulations on steep slopes cause avalanches (USDA Forest Service 1986, Polk and others 1981).

Historically, the Rocky Mountain region has experienced great climatic and geological changes over time scales of millions of years (Table 2). Driven by these changes, the vegetation of the region also changed. By the beginning of the

Quaternary period, approximately 2 million years ago, many of the plant species present today were in place (Tidwell 1975). From approximately 2 million to 10,000 years ago, during the Pleistocene epoch, the glacial advances and retreats affected both the topography and vegetation (Roberts and others 2001). Since the end of the Pleistocene ~10,000 years ago, shifts in climate have continued, often triggering changes in the spatial and elevational distribution of many plant communities (Table 2).

The dramatic changes in climate have important implications for HRV analyses. Foremost, by considering variation observed in the past 400 years we are considering only a short period in the history of SNF ecosystems. Soils, for example, surely develop over a longer time period. Climatic conditions almost certainly will be different in the future, even without the potential for human-caused global warming.

3.2 ECOLOGICAL SETTING

Various classification systems have been applied to the upland vegetation of the SNF. At a broad scale, the vegetation is part of the Southern Rocky Mountain Steppe-Open Woodland-Coniferous Forest Province (Bailey and others 1994). At an intermediate scale, there are five broad community or ecosystem types: alpine/near alpine, coniferous forest, montane meadow-parkland, sagebrush-grassland, and riparian (USDA Forest Service 1986, Roberts and others 2001). The alpine/near alpine community includes mostly herbaceous plants, small shrubs, mats of mosses and lichens, and some coniferous trees such as whitebark pine and Engelmann spruce (*Picea engelmannii*) at treeline. The coniferous forest community is located below the alpine/near alpine community and above the sagebrush-grassland communities. At the higher elevations of this zone (2500 to 3000 m), Engelmann spruce and subalpine fir (*Abies lasiocarpa*) (Fig. 11) are most common, though lodgepole pine and whitebark pine are common also. At lower elevations (1980 to 2800 m), the two dominant tree species are lodgepole pine and Douglas-fir. Quaking aspen occurs in the SNF, but on only about one percent of the land area (2% of the forests). Limber pine occurs mostly on dry, rocky sites at low elevations, but it can also occur on calcareous substrates at upper treeline, substituting for the most

common high-elevation tree, whitebark pine (which tends to favor igneous substrates; Steele and others 1983).

More specific classifications are based on both overstory and understory species. Steele and others (1983) described seven forest series on the SNF based on potential overstory tree dominants, and 52 habitat types based on overstory dominants and understory indicator species (after Daubenmire and Daubenmire 1968; Table 3). The seven forest series are Douglas-fir, lodgepole pine, Engelmann spruce, subalpine fir, whitebark pine, limber pine, and aspen. Proponents of the more specific habitat type approach contend that, while the spatial distribution of each tree species is controlled by factors such as elevation and topographic position, the understory species are better indicators of soils and microclimatic conditions. Habitat types are named according to the potential natural vegetation that is predicted to develop after long periods with no major disturbance, rather than the existing vegetation.

As in the Rocky Mountains in general, some areas of lodgepole pine forest are considered to be self-perpetuating in the region (Reed 1976, Despain 1983, Steele and others 1983)—although Zimmerman and Roberts (2001) do not map any lodgepole pine forests on the SNF as a “potential natural vegetation (PNV) type” (Fig. 12). Zimmerman and Roberts created their predictive map of PNVs on the SNF using a model based on climatic, topoedaphic, and geologic information, but they lacked such data for the east side of the Wind River Range where stable lodgepole pine forests occur on dry, granitic soils (K. Houston, SNF soil scientist, personal communication). Where the species is seral, lodgepole pine is replaced by spruce and fir. The low-elevation ponderosa pine (*Pinus ponderosa*) forest that is so common in much of the Rocky Mountains is practically non-existent on the SNF (occurring on only 83 ha).

Many non-forested vegetation types have also been identified (Tweit and Houston 1980, Thilenius and Smith 1985, Kinter 2000, Roberts and others 2001). On upland sites, fifteen shrubland, six grassland, and six forbland plant communities occur on the SNF, and Walford and others (2001) identified 41 shrub or herbaceous community types in riparian/wetland areas. A discussion of these community types is

beyond the scope of this report, but common dominant species include mountain big sagebrush, Wyoming big sagebrush (*Artemisia tridentata ssp. wyomingensis*), silver sagebrush (*Artemisia cana*), shrubby cinquefoil (*Dasiphora fruticosa*), Idaho fescue, bluebunch wheatgrass (*Pseudoroegneria spicata*), junegrass (*Koeleria macrantha*), alpine hairgrass (*Deschampsia nubigena*), lupine (*Lupinus sp.*), balsamroot (*Balsamorhiza sp.*) and other shrubs, grasses and forbs. Walford and others (2001) also identified 10 riparian woodland types, dominated by aspen, balsam poplar (*Populus balsamifera*), Engelmann spruce, or subalpine fir. Blue spruce (*Picea pungens*) was found occasionally on floodplains at low elevations. The montane meadow-parkland community comprises non-wetland openings within the coniferous forests. These areas are dominated by grasses and forbs, but may have up to 15% canopy cover by coniferous or deciduous trees. Scott (1996) described the alpine flora of this part of the Rocky Mountains, and a classic paper on alpine plant ecology in the Beartooth Mountains was written by Johnson and Billings (1962).

If sparse forests (<50% cover in trees) are ignored, about 52% of the SNF is forested (USDA Forest Service 1994a). Including these sparse forests (i.e., woodlands with scattered trees), the proportion of land that is forested increases to 64% (based on the 2000 RIS database). For this report, we use the higher percentage. Only about 6% of the landbase on the SNF is considered to be available for timber production (forested areas outside wilderness areas and capable of producing industrial wood), and of this amount, about 3.5% have been designated suitable for timber harvest (34,722 ha, of which 14% is Douglas-fir, 33% Engelmann spruce and subalpine fir, and 53% lodgepole pine; USDA Forest Service 1986). By district, the area designated as suitable in 1994 was: Clark's Fork, 10,085 ha (24,900 acres); Greybull, 3,515 ha (8,680 acres); Washakie, 3,366 ha (8,310 acres); Wapiti, 895 ha (2,210 acres); and Wind River, 16,909 ha (41,750 acres) (USDA Forest Service 1994a)

For our HRV assessment, we discuss each variable in the context of four landscape types : (1) high-elevation forests and woodlands (dominated primarily by Engelmann spruce, subalpine fir, lodgepole pine and whitebark pine in the upper montane and subalpine zones; 45% of the entire SNF); (2) montane forests and

woodlands at lower elevations (primarily Douglas-fir and limber pine, 19% of the SNF); (3) aspen forest and woodlands (1% of the SNF); and (4) grasslands, forblands, and shrublands (33% of the SNF) (Fig. 13). There certainly are important ecological distinctions between different habitat types and community types within each of the landscape types that we consider, but the differences blur over the long time periods of HRV analyses. Further, because of successional relationships between lodgepole pine, subalpine fir and Engelmann spruce (see section 4.2), it is difficult to discuss HRV in isolation for each overstory cover type.

3.3 CULTURAL SETTING

For centuries humans have lived and traveled throughout the area that is today the SNF (Polk and others 1981, Cochrane and others 1988). Consequently, managers have inherited the legacies of past land uses and are not starting with a "blank slate" (Romme and others 2000). This is important for several reasons. First, our ability to interpret pre-European-American conditions will be limited by events that have removed various kinds of evidence during the last century. Also, it is difficult to distinguish events that were "natural" from those caused by Native Americans. In fact, it may be necessary and even desirable to include some human influences as a factor affecting the HRV. However, the unprecedented human influences of European-Americans present unique challenges for managers. Because human land uses on the SNF have changed over time, we provide a summary of four historic periods: Pre-European-American Arrival, European-American Arrival, Early Federal Administrative, and Late Forest Service Administrative.

3.3.1 Pre-European-American Arrival (1500-1860)

Native Americans had been in the region for at least 10,000 years prior to the arrival of European-Americans, but members of the Shoshone tribe arrived around 1500 (Cochrane and others 1988). They hunted bighorn sheep, elk, and deer and gathered wild berries, pine nuts, cactus and other foods. In the late 1600s, the Shoshone obtained the horse and then expanded eastward across the basins and plains as they became efficient at hunting buffalo. At that time, the Shoshone split

into two groups: the highly mobile buffalo eaters, who rode horses, and those who maintained their traditional mountain culture in the Absarokas without horses and were known as sheepeaters. The sheepeaters moved camp often in the summer to pursue game but established a more permanent village near a source of fish in the winter (Polk and others 1981, Cochrane and others 1988).

At this time, the Crow (or Absaroka) Indians were centered on the Powder River Basin and Big Horn Mountains to the east, but often they extended west into the Big Horn Basin and Northern Absarokas, using the area for winter hunting camps (Polk and others 1981, USDA Forest Service 1986). By 1800, the Crow obtained guns from fur traders to the north, took control of the Big Horn country, and challenged the Shoshone in the Wind River Valley (Cochrane and others 1988). Also, members of the Blackfoot tribe attacked the Shoshone from the north.

The impacts of these early inhabitants on the landscape are largely unknown. Except for a few intentional or unintentional fires and the harvest of small trees for needs such as lodge poles, their impacts probably were minor and most concentrated in the foothills, riparian areas, and along travel routes. Notably, Baker (2002) concluded that, for Rocky Mountain landscapes in general, Native Americans had little effect on the frequency and extent of fire regimes in lodgepole pine and spruce-fir forests.

European trappers arrived during the early 1800s (starting with John Colter in 1806) as they searched for beaver pelts and other furs. They developed excellent relations with the Shoshone and Crow in the area that would become the SNF toward the end of the century (Cochrane and others 1988), despite introducing diseases that sometimes caused epidemics. Trappers and traders were active from 1820 to 1830, but became less common in the 1840s (USDA Forest Service 1986). The trapping reduced beaver populations substantially, undoubtedly causing significant changes in riparian landscapes. Other furbearers were affected as well, such as fox, marten and mink (Cochrane and others 1988).

3.3.2 European-American Arrival (1861-1897)

European-Americans began to influence the area more significantly from about 1861 to 1897. Pioneers traveled along the Oregon Trail through South Pass at the southern end of the Wind River Mountains in the 1860s, bringing with them herds of cattle. A gold strike there in 1868 brought in over 2000 people to the South Pass area. A few small towns (South Pass City, Atlantic City, Miner's Delight) and a Fort had been established near South Pass by 1870 (USDA Forest Service 1986). Conflicts with the Native Americans soon developed, leading eventually to the establishment of Indian Reservations, including the Wind River Indian Reservation in 1868 on the eastern edge of the SNF in Wyoming. The initial boom in mining at South Pass lasted only four years, dying out by 1872—the same year that YNP was established to the north.

In the 1880s prospectors tried again to find gold, this time using hydraulic methods, but with little success (Cochrane and others 1988). The town of Kirwin (now a ghost town) sprung up at this time in the Wood River Valley of the Absaroka Mountains after gold and copper were found there in the 1890s, as did several other settlements northwest of Lander (USDA Forest Service 1986).

Sawmills were opened in many places, including Mill Creek, the Middle Fork of the Popo Agie River, Fort Washakie, Carter Mountain, the headwaters of the Wind River, and in an area near the Indian Bureau Agency. Timber was obtained from the nearby forests.

Cattle and sheep grazing in the SNF began in substantial numbers in the early 1880s. The use of mountain forage and the development of trails into the high country by livestock were unorganized at this time and caused considerable soil erosion, loss of soil water retention capability, and a reduction in plant diversity (Cochrane and others 1988). Cattle numbers peaked by 1883 and dropped significantly in 1887 when 30 to 50% of the cattle in the area died because of two consecutive harsh winters. By 1890 the cattle market was depressed, keeping cattle numbers relatively low (91,546 head in Fremont County, an area larger than today's county boundaries which included the entire SNF at that time). In contrast, sheep

were increasing and numbered 144,580 in the same area. Many sheep grazed on the Forest year round (at low elevations in the winter).

Sheepmen used fire as a tool to provide more forage. The southern section of the Wind River Range (in today's Washakie District of the SNF) was described as being stripped of its old trees by fires (Cochrane and others 1988). Early settlers could do little to combat the fires. Historical photos around 1910 indicate that large areas were burned by the settlers (K. Brauneis, SNF fuels manager, personal communication).

Due to its proximity to YNP, the tourism industry in the SNF started early. Subsistence big game hunters used the forest initially (1880s), but soon many of these hunters were guiding trophy hunters from the east (late 1880s, early 1890s). In the mid-1890s, working ranches took in a few hunters and summer guests (dudes); and in 1897 "Buffalo Bill" Cody founded the town of Cody. He was also instrumental in the creation of the Shoshone Dam and Reservoir (later named Buffalo Bill Dam), which provided irrigation and electricity to Cody and Powell by 1908.

Many wildlife populations were decimated during the late 1800s due to market and sport hunting (Cochrane and others 1988). Bison and elk numbers crashed by 1875, moose were nearly extirpated by the early 1890s, and trumpeter swans were nearly lost from along DuNoir and Torrey Creeks. Hunting restrictions were passed in 1886 by the Wyoming Territorial Legislature, but enforcement was ineffective initially. Conversion of winter range habitat into rangelands for livestock further diminished the abundance of big game.

3.3.3 Early Federal Administration (1898-1945)

The federal government became more involved in the area in the late 1800s, first by establishing the Yellowstone Timberland Reserve on the eastern border of YNP in 1891. Along with wildfires and illegal hunting, overgrazing was one of the management problems that federal managers hoped to correct. The SNF is considered the first National Forest in the nation because it was part of this first timber reserve. Within the same year, the Teton Reserve was created to the south

(and included parts of what are now the southern Shoshone and Bridger-Teton National Forests).

Little management occurred in the reserve until 1898, the year when the General Land Office took over management from the military. Within a few years, the new managers were taking an aggressive stance against overgrazing. The reserve's forage base was seriously threatened in 1900, when almost half a million sheep were in Fremont and Big Horn counties (Park County did not exist at that time). This number increased by 50% the following year (Cochrane and others 1988). In 1900 the Reserve required its first grazing permits, which were initially free, and grazing restrictions were vigorously enforced by 1902. In 1903 the Reserve was expanded and merged with the Teton Forest Reserve to cover much of northwestern Wyoming. By 1905 innovations such as salt blocks were used to distribute livestock more evenly. In 1906 there were about 150,000 head of livestock permitted on the reserve (135,738 sheep; SNF, undated brochure)

With the creation of the U.S. Forest Service in 1905, land uses were regulated still more carefully (Pinchot 1907). To create what was thought at that time to be more manageable units, the Yellowstone Timberland and Teton Reserves were divided into several National Forests, in 1908, and by 1945 the various National Forests had been merged to form the SNF as it exists today.

The type and amount of grazing varied across the two Forests. The Wapiti District of the SNF (main part of the Absarokas) had little grazing, but the Greybull District (southeastern Absarokas) had heavy use by cattle. On the Washakie National Forest (Wind River Range), the South Pass District had primarily sheep but the Sheridan Creek area had mostly cattle. Overall, less livestock grazing occurred on the Washakie National Forest than on the area then known as the Shoshone National Forest to the north.

By the time of World War I, sheep numbers had been reduced by half and cattle by about 5% on the area then known as the Shoshone National Forest (Cochrane and others 1988), but the pressures of wartime again caused overstocking. Peak numbers of grazing permits (>150,000) on the Forest were reached in 1918, which led to overgrazing in some areas. However, after the war

market conditions caused a 50% decline in cattle and 40% decline in sheep (between 1924 and 1925). Due to lack of funds to hire riders that would disperse the livestock, spotty use of allotments was common during this period, causing trampling and overgrazing of localized areas. Large herds of feral horses on the Washakie at this time also put excessive pressure on the forage base. With the droughts of the 1930s, conditions continued to deteriorate.

New Deal monies during the depression enabled the construction of stock driveways needed to properly access allotments and drift fences needed to curtail livestock trespass (Cochrane and others 1988). Although grazing fees were reduced by half during this period, the Forest Service did not increase the number of grazing permits. Grazing pressure remained high (mostly sheep) until the late 1930s, when some dude ranches started to gain priority for grazing their horses. With rising interest in recreation and big game hunting, the Forest Service cut back on the length of the grazing season and reduced the number of permits (Fig. 14).

In sharp contrast to World War I, overstocking did not occur when World War II started. Instead, the grazing season was shortened (to 3.5 months for cattle and 2.5 months for sheep on some National Forests). In 1940, only about 74,000 sheep and 10,000 cattle were on the forest (Fig. 14). As part of the program to protect livestock and wildlife, predator control programs were started that eventually greatly reduced wolf, mountain lion, lynx, and wolverine populations during this time. Rodents were also poisoned. Incidental poisoning apparently wiped out the bald eagle nesting populations on National Forest land (Cochrane and others 1988).

Mining and logging. In addition to cattle and sheep ranching, irrigated farming, tourism, hunting and some logging supported economic growth around the SNF in the early 1900s. Coal mining near Hudson and oil and gas development in the Big Horn Basin augmented this growth in the 1920s (USDA Forest Service 1986). Timber sales were initially low until the railroad came through the town of Shoshoni. Tie hacking began on the SNF in 1906, with major operations in place on the Wind River drainage by 1912 (Cochrane and others 1988, Neal 2000). Lodgepole pine trees of 25-38 cm diameter at breast height (DBH) were preferred, leaving behind trees that were too large or small for ties. In 1927, 61% of the harvested trees were

lodgepole pine, 29% were Engelmann spruce, 8% were limber or whitebark pine, 1% Douglas-fir, and less than 1% were subalpine fir (Neal 2000). Efforts were made to improve the forests by selecting diseased or forked trees, but sound trees were also selected to ensure 60 to 65% of the trees were cut (Cochrane and others 1988). Much slash was left in the forest, of which some was piled and burned (Cochrane and others 1988).

Railroad ties were floated down the Wind River to Riverton just after peak flow. To facilitate tie drives, large boulders and downed trees in the Wind River and East Fork of the DuNoir were removed, breakwaters were erected, splash dams built, and booms anchored in place. Riparian vegetation may also have been cleared, as occurred on the Medicine Bow National Forest (Young and others 1989, 1994). Tie hacking on the SNF continued through 1947 (Fig. 15). Millions of ties were cut over 33 years (1914-1947) on about 4500 ha (averaging 4 to 5 million board feet per year). Peak production occurred in 1927, with 700,000 ties driven to Riverton that year (Cochrane and others 1988).

Forest managers began in earnest to suppress fires in the early 1900s (Cochrane and others 1988). At that time, fire was personified as the enemy to forest managers and fire suppression became a primary objective, though suppression was not very effective initially. By 1913 lookout towers and a communication network had improved response time, but inaccessibility was still a problem. Fires were most effectively fought in the tie-hacking areas in the Wind River drainage, where a road network and a substantial number of laborers were available. Forest roads and trails were expanded in the 1920s and 1930s, which further assisted fire-fighting efforts. Fire danger was especially high in the dry years of the 1930s, when over half of the fires were human-caused. Radio communication and additional vehicles further improved fire-fighting efforts in the 1940s.

Work toward controlling insect outbreaks started later than fire control. Although infestations were present, primarily of different beetle species, the Forest administrators had a relatively tolerant and casual attitude toward forest "pests" up until the depression years of the 1930s (Cochrane and others 1988), when the Civilian Conservation Corps and Forest personnel worked on insect outbreak

problems by cutting, selling or burning infested forests and even by squashing beetles with their axes.

Recreation. In 1898, trails were built into the Shoshone River and Clark's Fork River Basins to meet hunting and recreational demands. The upper Wind River country and the Cody area gained a reputation for exceptional big game hunting (Cochrane and others 1988). Furthermore, to provide fishing opportunities, non-native trout were planted in lakes and rivers, many of them initially barren of fish. Tourists increased in the area after the North Fork Road was completed in 1903, which gave access from Cody to the east entrance of YNP. Recreational pressure was less in the southern portions of the SNF prior to World War I, but that changed after the war. With the growing demand from easterners to experience western life, dude ranches began to flourish on the fringes of the SNF everywhere, especially in the 1920s (e.g., Holm Lodge, Valley Ranch, Simon Snyder's Ranch, Diamond G Ranch, and CM Ranch). The boom years of the dude ranch industry ended with the Great Depression, but visitation increased again by the late 1930s.

Throughout the early 1900s, Forest Service officials tended to emphasize grazing and timber harvest over recreation. However, the wilderness movement gained momentum in the 1920s, and by 1932 four primitive areas (precursors to Wilderness Areas) had been established: the Stratified, Glacier Peak, South Absaroka, and North Absaroka Primitive Areas. A later one, the Popo Agie Primitive Area, was established in 1937. Livestock grazing could still occur in these areas, but other development was limited. These primitive areas became part of the National Wilderness System in 1964 (Fig. 7) and today over half (55%) of the SNF is designated wilderness.

Wildlife. Illegal hunting was a major problem in the first half of the 1900s. To illustrate, one could see about 5,000 pronghorn and several thousand elk and deer during a day's ride in the 1890s along Sage, Meeteetse, and Carter Creeks. By 1905 big game was considered scarce (Cochrane and others 1988). Hunting laws were initially enforced by state game wardens, but by 1907 Forest Service employees were allowed to act as deputy game wardens in their districts. Game preserves that overlapped Forest lands were also established to help populations recover. By the

mid-1930s, big game numbers were up again, and it was recognized that winter range protection and managed hunting were the keys to maintaining population sizes.

3.3.4 Recent Forest Service Administration (1946 to present)

Grazing. New range management practices were adopted after World War II (Cochrane and others 1988), when the number of livestock approved for grazing was reduced—cattle and sheep by 75% and 33%, respectively, on the Wapiti District (the main part of the Absaroka Range). In general, the number of cattle increased slightly or remained the same while sheep declined on the SNF as a whole (Fig. 14). In the 1970s and 1980s, rural subdivisions in the Wind River Valley reduced the land area available for livestock in general, and sheep numbers continued to drop because of poor markets. In 1986, only 17% of the Forest was considered to have grazing potential (Cochrane and others 1988), with 74,700 cattle animal unit months (AUMS) and 12,000 sheep AUMS permitted (USDA Forest Service 1986). Compared to 1935, these levels are similar for cattle but greatly reduced for sheep (Fig. 14). During recent years, emphasis has shifted away from grazing and more toward tourism.

Compared to the estimated 400,000 animals that grazed on the SNF at the beginning of the 20th century, today's levels of about 27,000 cattle, sheep and horses in the Shoshone high country are low (USDA Forest Service, undated brochure). Sheep numbers are now at an all time low of about 2,500 head and are on the forest for 1.5 months on average. Currently, cattle are grazed four months each year using deferred rotations or, in a few small allotments, open season-long management systems (Cochrane and others 1988, USDA Forest Service 1995).

Logging. After 1945, timber harvesting was still centered mainly in the Upper Wind River Valley, modernized with roving sawmills, trucks, and dozers (Neal 2000). In addition to ties, logs were cut and sent to mills to produce 2 by 4 studs. By the 1950s clearcutting of lodgepole pine became the dominant harvest method (Figs. 15 and 16). Economically, new levels of prosperity created larger markets for timber than ever before, and clearcutting created new stands of even-aged, rapidly growing,

economically valuable trees that could be more easily subjected to thinning and other management activities. Clearcutting also was attractive because of problems experienced with selective logging and desires to harvest stands that had been affected adversely by tie hacking (Langston 1995).

An average of ~8 million board feet was harvested on the SNF each year between 1952-62 (Cochrane and others 1988). Most of the cutting was in the Wind River Range near Dubois because of the relatively gentle topography and good road system, compared to elsewhere on the Forest. Not until 1962 was a modest size mill set up in Cody. The harvest then increased greatly. By the late 1960s, the SNF produced over 20 million board feet per year (Cochrane and others 1988; Fig. 17) and the size of clearcuts peaked (Fig. 18). Throughout the history of timber extraction from the SNF, nearly all of the logging has been in the Wind River District.

Although less than 400 ha (99 acres) were clearcut per year between 1953 and 1972, the public strongly criticized clearcutting and its effects on soil erosion, stream siltation, and aesthetics (Berntsen and others 1971, Cochrane and others 1988). The allowable cut on the SNF was reduced to 3.5 million board feet per year in 1975 (USDA Forest Service 1994a). During the 1970s, other harvest methods such as shelterwood and selective cutting were used more frequently than clearcutting (Berntsen and others 1983), and clearcuts were reduced in size, as mandated by the National Forest Management Act of 1976. In 1980, the maximum allowable cut rose to 10.4 million board feet. Nevertheless, the actual yearly cut on the entire SNF averaged over the period from 1974-83 remained the same as during the 1953-72 period, at ~8 million board feet.

In 1986 SNF personnel classified only 6% of the SNF landbase as available for timber production and only 3.5% as actually suitable for that purpose (USDA Forest Service 1986). This determination was based on various factors including new information about the potential for tree growth on the SNF, a large area already designated as wilderness (55% of the SNF), steep topography, and continuing debates about the relative value of timber, recreation, wildlife (including the endangered grizzly bear) and water yield in the region. Most of the suitable land was in the Wind River and Clark's Fork Districts, and the tree species most often

harvested over the years have been lodgepole pine (53%), Engelmann spruce (33%), and Douglas-fir (14%) (USDA Forest Service 1986). The cut level in 2001 was 4.4 million board feet (Neal 2000). In total, only about 3% of the forested land (2% of the entire SNF land area) has been affected by timber harvest (USFS RIS).

Although levels of timber harvest have been relatively low on the SNF and have declined in the last decade (Figs. 17, 18, and 19), the imprints of roads and timber harvesting during the last century are present, especially in about half of the Wind River District (Figs. 16 and 20).

Currently, the SNF forest management plan allows the cutting of approximately 2 million board feet per year for firewood and another 1 million board feet per year for other purposes such as hazard tree removal, wildlife habitat improvement, and the enhancement of scenic views (USDA Forest Service 1994b). Over 30 million board feet were salvaged from forests burned in 1988 (K. Houston, SNF soil scientist, personal communication). Thinning (commercial and precommercial) is a common practice in the small areas where timber harvest occurs. Between 1991 and 2000, 1,304 ha (3,221 acres) were thinned, averaging 130 ha (321 acres) per year (USDA Forest Service 2000). Between 1950 and 1990, the average land area thinned was lower, around 20 ha (D. Eckardt, SNF Silviculturist, personal communication).

Mining. Intensive mineral development (uranium, copper, iron) on the SNF began in the 1950s and 1960s, but only near South Pass and Kerwin. Digging at the U.S. Steel taconite mine started in 1961, but it was closed in 1983 following a decline in demand (Jo Anne Sterling, U.S. Steel, personal communication). Prospecting for copper was successful near Kirwin in the 1960s, but plans for an open pit mine were scrapped when opposition became strong and the value of copper dropped (Cochrane and others 1988).

Recreation. Visitation to the SNF was low during World War II, but after that travel through the Forest into YNP began to increase. New hotels and tourist facilities in the vicinity of the Forest, and camping areas in the Forest, were built to accommodate the demand. Off-road 4-wheeling also became popular (Cochrane and others 1988), although the rugged country and few back roads helped keep such

activities from being widespread. Erosion from off-road vehicles and overuse by horse-packers and their stock led to some areas becoming denuded, trampled, and excessively grazed. Consequently, outfitters were required to pack in their own horse feed. Off-road users were given information on how to protect the environment. In the 1960s snowmobiles became common. Backcountry travelers such as hunters, anglers, campers, and rock-climbers were a minority of the visitors, although they increased with the boom in backpacking in the late 1960s. The expanding timber harvest in the late 1950s and 1960s increased road construction and access to visitors and off-road vehicles. Recreation continued to increase in the 1970s and 1980s (when visitor days topped 1 million). In 1964, the primitive areas became Wilderness Areas in accordance with passage of the Wilderness Act, with over half of the SNF becoming designated wilderness. The 1984 Wyoming Wilderness Act added 57,464 more hectares (141,936 acres) of wilderness. Today, recreation is considered a primary resource of the SNF due to its proximity to YNP, its remarkable big game resources, and its spectacular scenery (Cochrane and others 1988).

Wildlife. After World War II, the increased number of logging roads and interest in outdoor recreation had significant effects on the native fisheries and wildlife of the SNF (Cochrane and others 1988). On the negative side, poaching increased as road access increased and off-road vehicles illegally entered wilderness areas. A massive stocking program of exotic fish such as lake trout (*Salvelinus namaycush*), golden trout (*Oncorhynchus aguabonita*), brook trout (*Salvelinus fontinalis*), and rainbow trout (*Oncorhynchus gairdneri*) was begun, which displaced native species; and river otters were over-harvested until otter trapping was banned in 1952. A positive effect of the increased recreation and logging was that the new roads helped disperse hunting on big game more evenly throughout the forest, reducing pressure on populations on the fringe of the Forest.

Big game thrived in the 1950s and 1960s after several elk and bighorn sheep winter ranges were purchased by the state of Wyoming. Mountain goats also recolonized the area, and winter feeding of elk was instituted (Cochrane and others 1988). By 1986, areas set aside as winter range for big game were estimated to be

15,514 ha (38,320 acres) (USDA Forest Service 1986). The Whiskey Mountain Winter Range has a large population of wintering bighorn sheep (900-1100). From 1965 on, elk numbers increased dramatically, although deer began to decline. On the SNF in 2002 there were approximately 37,000 deer (nearly all mule deer, but some white-tailed deer), 23,000 elk, 5,000 bighorn sheep, 1000 moose, 200 mountain goats, and 200 grizzly bears (L. Otto, SNF Land Management Analyst, personal communication).

Big game management is still faced with significant challenges (M. Hirschberger, SNF wildlife biologist, personal communication). For the bighorn sheep, migration paths traditionally used by a portion of the herd to reach their winter range have been closed by encroachment of conifers due to fire suppression. Also, there appear to be problems with unusually high abortion rates, possibly due to mineral deficiencies in the animals' diet. The high elk numbers on the forest have damaged winter range habitat and aspen in some areas, and managers have designed timber harvests that could be beneficial to wildlife. For example, the 1986 Forest Plan states some "climax" forests should be removed in the DuNoir area to create more forage for big game (USDA Forest Service 1986).

Bears and wolves have increased in number after their initial decimation in the late 1800s (Cochrane and others 1988). In 1973, the Interagency Grizzly Bear Study Team was established and management measures were established to protect the grizzly bear. Specifically, grizzly bear hunting was banned in 1975, a state law was passed to compensate stockmen for losses from the bear, important bear habitat was protected from development (except gas and mineral exploitation), and outfitter camps and backcountry campsites were developed with hanging poles for hanging food (Cochrane and others 1988). In 1995 wolves were reintroduced into the Greater Yellowstone Ecosystem and are now found on the SNF (Harris and others 1997). Black bear numbers have also been stable or slightly up. River otters have been slowly increasing since the ban on trapping; and the peregrine falcon, which disappeared from the Clark's Fork area when DDT began to be used after World War II, has increased since the ban on DDT in 1978 (USDA Forest Service 1994a).

Current standards for maintaining wildlife habitat on the SNF specify that at least 5% of the forest should be in grassland/forbland habitat and 10% in old-growth (>200 years old; USDA Forest Service 1994a). Actual amounts are estimated to be close to or higher than these targets, with 4% in grassland/forbland habitat (USDA Forest Service 1994a) and 22% in old-growth forest (FIA database). The loss of wildlife habitat, rather than over hunting, is the main concern for wildlife managers at the present time (USDA Forest Service 1994).

4. PLANT SPECIES DISTRIBUTIONS AND SUCCESSIONAL PATTERNS ALONG ECOLOGICAL GRADIENTS

4.1 DOMINANT PLANT DISTRIBUTION PATTERNS

Throughout the SNF, the forest vegetation changes dramatically from the lowest elevations on the eastern edge toward the highest elevations near or along the continental divide (Steele and others 1983, Roberts and others 2001). At the lowest elevations (below 1900 m), foothill grasslands and shrublands occur in small areas within the Forest (Figs. 11 and 12). Of course, they cover much of the Bighorn Basin to the east. Above the grasslands and shrublands, Douglas-fir and some limber pine woodlands and savannahs are common from about 1,900 to 2,800 m. Higher in the mountains, lodgepole pine dominates, particularly on areas of gentle terrain. Engelmann spruce and subalpine fir become abundant from about 2500 m (8200 ft) up to treeline at about 3000 m (9,840 ft). Whitebark pine is intermingled with spruce and fir at alpine treeline or on high, exposed ridges, especially slopes with southern to western aspects.

On severe sites, trees are stunted by wind, cold, and drought (Steele and others 1983). Whitebark and limber pines are common in such habitats. The distribution of whitebark pine is strongly affected by the Clark's nutcracker, a bird that commonly distributes and caches the large seeds of this species into wind-swept openings where snow cover is relatively low. The nutcracker also distributes the seeds of limber pine, which, while most common at low elevations, has a broad

elevational range (1850 to 3050 m)—extending from lower treeline where it may share habitat with Rocky Mountain juniper (*Juniperus scopulorum*) to calcareous sites at upper treeline (~3,000 m; Tomback 1983).

In addition to elevation, soils strongly affect the forest communities. As is typical in the Rocky Mountains, Douglas-fir thrives on soils derived from limestone or basic extrusive volcanics and is less common on soils from granitic rocks (Despain 1973, Reed 1976, Steele and others 1983). Douglas-fir normally is found at lower elevations, but if basic soils are present, Douglas-fir can occur at higher elevations and on mesic, north slopes with Englemann spruce and subalpine fir. Similarly, it is absent in some low-elevation areas in the Wind River and Absaroka Ranges that lack basic soils. In contrast, the more widespread lodgepole pine is common on acidic, coarse soils derived from granitic rocks and some sandstones. Lodgepole pine is frequently a seral species and is prevalent in disturbed areas, but, as mentioned earlier, it often shows no sign of succession to other species, especially on dry sites. Engelmann spruce and subalpine fir occur on any soil type at higher elevations. Whitebark pine is most competitive on acidic, igneous-derived soils near or at treeline, while limber pine predominates on calcareous soils (Steele and others 1983).

As noted previously, numerous coniferous forest habitat types have been identified on the SNF (Table 3). A review of each habitat type is beyond the scope of this report, as insufficient information for separate HRV analyses is available for each one. However, it is clear that the dominant understory plant species vary greatly with environmental conditions. Among the species used to distinguish the habitat types are snowberry (*Symphoricarpos oreophilus*), common juniper (*Juniperus communis*), Oregon grape (*Berberis repens*), buffaloberry (*Shepherdia canadensis*), dwarf huckleberry (*Vaccinium scoparium*), heartleaf arnica (*Arnica cordifolia*), and bluegrass (commonly, *Poa nervosa*).

As with coniferous forests, various habitat types are dominated by aspen (Table 3). Nonetheless, aspen is not abundant in the SNF (<1% of the land area; RIS database), occurring primarily as small groves at middle and low elevations. It is most common on relatively moist sites characterized by fine-textured soils (Hoff

1957, Reed 1971). Aspen reproduction typically is asexual, with new shoots produced from root sprouts (suckering; Barnes 1966, Bartos and others 1991). This, combined with the persistence of aspen in the understory of some mature forests, explains why aspen tends to develop where it occurred previously. Sexual reproduction is quite rare, though seedlings do occur when severe disturbances such as fire are followed by the extended moist conditions in the spring required for seedling establishment (McDonough 1985). For example, aspen seedlings were abundant in some areas in 1989 after the 1988 fires in nearby YNP (Romme and others 1995). Because of such requirements, sexual reproduction is thought to be episodic (DeByle and Winokur 1985, Romme and others 1997). There is considerable genetic diversity between clones, with some clones better adapted for higher elevations and some responding differently to weather conditions than others (Jelinski and Cheliak 1992). For example, it is not uncommon to see two adjacent aspen stands (clones) in the fall, one with yellow leaves and the other with green leaves.

Other tree species found on the SNF include white spruce (*Picea glauca*) and, infrequently, blue spruce (*P. pungens*) (Steele and others 1983). White spruce has been found in the Swamp Lake area in the Clark's Fork District and in nearby YNP (S. Jackson, University of Wyoming ecologist, personal communication), where it apparently hybridizes with Engelmann spruce, particularly in the northeastern part of YNP (J. Whipple, YNP botanist, personal communication). Blue spruce is found at lower elevations, often as a component of riparian habitat. Narrowleaf cottonwood (*Populus angustifolia*) and balsam poplar (*P. balsamifera*) are also found in the riparian zone at lower elevations, such as along the North and South Forks of the Shoshone River (Walford and others 2001).

Grasslands and shrublands at all elevations compose about 30% of the SNF (Fig. 13 and Table 1). They occur mostly on areas too arid to support trees, although a few are found on soils at mid to high elevations that are excessively wet during the growing season for tree growth (Tweit and Houston 1980). In the forest zone between the upper and lower timberline, areas dominated by shrubs, forbs and grasses typically include one or more of the following characteristics: convex or well-

drained landforms, thin or poorly developed soils that usually are quite dry, and high winds. Fires or landslides open up the forests in some areas, allowing early successional herbaceous and shrubby stages to flourish for a time. Above treeline in the alpine zone, the climate is too severe for trees.

Of the vegetated, non-forest area, about 79% has been classified as grassland by the USFS (RIS database). The dominant species in the majority of the grasslands above lower treeline is Idaho fescue (*Festuca idahoensis*) (Table 3). Below lower treeline, bluebunch wheatgrass dominates. Sometimes grazing pressure on these bunchgrasses allows shrubs to become more common (Tweit and Houston 1980).

Shrublands are mostly dominated by big sagebrush (*Artemisia tridentata vaseyana* and *Artemisia tridentata wyomingensis*) and are found below lower timberline as well as in the forest zone. Shrublands are found in site conditions intermediate between grassland and forest habitat types, having deeper, more developed soils and more available moisture. Bitterbrush (*Purshia tridentata*) is often mixed with mountain big sagebrush in the southern Wind River Mountains from 1800 to 2400 m, especially on soils derived from limestones and shales on mid to lower slope positions of south and west-facing exposures (Tweit and Houston 1980; Table 3). Shrubby cinquefoil is found in wet meadows at higher elevations. Willow-dominated shrublands (*Salix* spp.) are common in riparian zones (Roberts and others 2001).

Other vegetation types are found only at the higher elevations of the SNF. At alpine treeline (~3000 m), where the subalpine and alpine zones merge, subalpine fir, Engelmann spruce, and whitebark pine form dwarfed, shrubby patches referred to as "krummholz." Just above this, the alpine tundra is dominated by various grasses, sedges, small shrubs and forbs that are able to withstand the severe environment characterized by high winds, low humidity, cold soil temperatures, high ultraviolet radiation, short growing season, low soil moisture, and great daily temperature fluctuations (Bliss 1956, Knight 1994a).

Plant species distributions have been changing for thousands of years and likely will continue to change in the future. For example, investigators have found evidence in the pollen record that the relative abundance and distribution of

lodgepole pine, Engelmann spruce and subalpine fir at both the stand and landscape scales has shifted due to climate changes (Hansen 1940, Fall 1997, Whitlock 1993). Ponderosa pine arrived in the Bighorn Mountains about 2500 years ago (Jodi Norris, University of Wyoming ecologist, personal communication). As noted previously, HRV reports must, to every extent possible, separate climatic effects from management effects.

Some recent changes in vegetation distribution might have occurred due to climate change. For example, lodgepole pine, subalpine fir and Engelmann spruce have been observed establishing in formerly open meadows in the region (Patten 1969, Dunwiddie 1977, Gruell 1980, Schimpf and others 1980, Pearson and others 1987, Jakubos and Romme 1993, Hessl and Baker 1997, Meagher and Houston 1998). Some of the more recent observations of forest encroachment have been attributed, at least in part, to warmer conditions that began in the late 1800s (Fig. 21). Of note, temperature in the nearby Bighorn National Forest was higher in the year 2000 than it has ever been during the HRV period from 1600 to 1860 (Fig. 21). This may be true for the SNF as well.

4.2 SUCCESSIONAL PATTERNS

4.2.1 Succession in high-elevation forests

Lodgepole pine, because of its relative intolerance for the forest understory environment, traditionally is considered a pioneer, seral species (Baker 1949, Lotan and Perry 1983, Lotan and others 1985, Parker and Parker 1994), but in some drier areas on gentle slopes with large fluctuations in temperature, it appears to form self-perpetuating forests (Reed 1976, Despain 1983, Steele and others 1983). New trees become established in such stands after the creation of canopy gaps (Despain 1983). As a seral species, lodgepole pine is typically found extending from the lower-elevation cold, moist Douglas-fir sites up to all but the wettest spruce-fir sites, where the spruce and fir are often the pioneers following a disturbance (Steele and others 1983, Doyle 1997) and tree regeneration can be slow (Jack 1900). At higher elevations, Douglas-fir also can be seral to spruce-fir forests (often the spruce are

white spruce/Engelmann spruce hybrids), usually on sites too warm or calcareous for lodgepole pine (Steele and others 1983, Bradley and others 1992). Whitebark pine can also be seral to high-elevation spruce-fir forests on some sites. Often whitebark pine is the only tree species that survives near treeline. At lower elevations, especially forest edges and in moist sites, aspen can be seral to spruce-fir.

Subalpine fir and spruce forests are the most common and extensive on the SNF as a whole (Fig. 13; but see Table 1 for variation among districts). The abundance of subalpine fir is attributable to favorable climatic conditions and the fact that seedlings of this species are shade tolerant and often can become established on relatively thick forest floors (Steele and others 1983). Engelmann spruce is less successful at seedling establishment, but once established it lives longer (500 years or more). The invasion of subalpine fir can be retarded by moose browsing, maintaining dominance by lodgepole pine, Engelmann spruce, or Douglas-fir (Steele and others 1983). In some high-elevation areas where Engelmann spruce is capable of reproducing itself in the understory (i.e., on more calcareous soils), limber pine can be a seral species.

Lodgepole pine is well-adapted to disturbances because it often bears cones that remain closed for many years, thereby storing thousands of seeds on a tree. Known as serotinous cones, they open primarily when exposed to higher than normal temperatures, such as during a fire or when the cones are near the soil surface (which is warmer than the canopy). Notably, not all lodgepole pine produce serotinous cones and the proportion of "closed" and "open" cone trees is highly variable (Lotan and Perry 1983, Muir and Lotan 1985, Tinker and others 1994, Turner and others 2003). Steele and others (1983) concluded that serotinous lodgepole pine are not common on the SNF, but some observations suggest that serotinous cones are more common in the southern part of the Forest than in the north (about 60-70% and 10%, respectively, although both open and closed cone-trees are common (K. Houston, SNF soil scientist, personal communication). The resistance of lodgepole pine seedlings to cold and dryness (Cochran and Berntsen 1973, Steele and others 1983), combined with a history punctuated by disturbances over the last 150 years, help explain the prevalence of lodgepole pine on the SNF

today (17% of entire SNF land area, Fig. 13; see Table 1 for variation among the districts).

Because there is great variability in establishment times and forest conditions, deciding when a subalpine forest reaches "old growth" conditions can be difficult. The time required for a spruce-fir forest to develop after a fire may range from just over 150 years to over 1,000 years (Billings 1969, Mueggler 1976, Romme and Knight 1981). Villalba and others (1994) define "old growth" subalpine forests in Colorado as those with at least some trees greater than 300 years old. Important components of old-growth forest, such as relatively large trees, numerous snags, and relatively high amounts of coarse woody debris (CWD), may require several centuries to develop (Rebertus and others 1992). Notably, Mehl (1992) did not require that Rocky Mountain old-growth forests be 300 years or more, describing them instead as stands with relatively old and large trees for the species and site, high variation in tree sizes and spacing, presence of standing and down dead trees, decadence in the form of broken or deformed tops or bole and root decay, multiple canopy layers, gaps in the tree canopy, and understory patchiness. He provides minimum old-growth criteria for different forest cover types, which SNF managers are currently using when defining old-growth forests for the RIS database.

Based on estimates by the Shoshone National Forest personnel, only about 2% of the forested land on the SNF was estimated to have old-growth forests in 1994 (USDA Forest Service 1994a). However, most of the inventory was based on photo-interpretation (only 20% of the stands were inventoried on the ground and many of those were classified as suitable for timber harvesting). Thus, some skepticism exists as to the accuracy of that number. More recent data from 374 systematically spaced plots on the SNF, where each "plot" is composed of four 0.018-ha subplots (1999 FIA database; fia.fs.fed.us, USDA Forest Service 2001), indicates about 2% is >300 years old and 22% >200 years old (stands in the >200 age group meet Mehl's age criteria for old-growth). For the high-elevation forests only, 1% is >300 years old and 24% is >200 years old.

Using another database from the Interagency Grizzly Bear Study Team (Mattson and Despain 1985), 35% of the high-elevation forests in the northern SNF

(excludes the Wind River Range) were classified as dominated (>33% of the land unit) by late-successional stages in 1988 (LP3, SF, and WB, as described by Despain 1990).

The three different estimates indicate error may be high in the classification of old-growth. A good estimate of the actual amount of old-growth forest on the Shoshone will not become available until more of the forest is surveyed. Nevertheless, because much of the forest has not been strongly impacted by harvesting or firewood cutting (<6%), the higher estimates are more likely.

The HRV for succession and successional processes during the HRV period for both individual stands and the landscape surely would have been quite broad. Different sites on any specific burn can experience different successional trajectories, resulting in doghair stands of lodgepole pine in some areas, stands of average or below average tree density elsewhere, and a full variety of combinations of invading aspen, Douglas-fir, Engelmann spruce, subalpine fir, whitebark pine, limber pine, and lodgepole pine (Stahelin 1943; Turner and others 1997, 2003; Reed and others 1999). Moreover, secondary succession initiated today could have quite different trajectories than 200-300 years ago because of the warmer and sometimes wetter climatic conditions of the last century (Fig. 21). Franklin and Laven (1989) stress that forest development is not unidirectional, as moderate disturbances such as a minor wind storm or surface fires can set back succession to an earlier stage or create alternative pathways of succession, even at high elevations. Such disturbances probably are more common in warmer environments.

4.2.2 Succession in low-elevation forests

Douglas-fir is the dominant forest type at low elevations on the SNF (Steele and others 1983), but it covers only about 17% of the Forest as a whole (Fig. 13; see Table 1 for variation among districts). Lodgepole pine and aspen can be seral to Douglas-fir on moist sites (Steele and others 1983, Bradley and others 1992). On cool, dry sites, limber pine can be seral. However, in many areas, Douglas-fir is the only tree that becomes established, particularly where the soils are strongly calcareous. In 1988 the majority of Douglas-fir forests on the SNF were affected by

stand-replacing fire, and some of them have been replaced with lodgepole pine or aspen (Bradley and others 1992). Still, due to the prevalence of calcareous soils at lower elevations, most of the burned areas now have Douglas-fir regeneration (K. Houston, SNF soil scientist, personal communication).

In the northern part of the SNF, most of the Douglas-fir is late seral, with an average age of ~160 years. Using criteria of Mehl (1992), the old-growth definition for Douglas-fir is >200 years and >46 cm DBH. Based on the FIA database, 16% of the Douglas-fir forests can be classified as old-growth on the Forest as whole. At low elevations, 18% of the forests are old-growth (>200 years old). Limber pine dominates on the driest sites at low elevations, where it may be co-dominant with Douglas-fir (Bradley and others 1992) or it may perpetuate itself in areas too dry and rocky to support other tree species. Notably, some forests co-dominated by limber pine, Douglas-fir, and Rocky Mountain juniper are replacing shrubland dominated by sagebrush (K. Houston, SNF soil scientist, personal communication).

The warmer and sometimes drier conditions in recent years can lead to changes in the successional trajectory at lower elevations as well as at high elevations, particularly due to changes that have occurred in stand structure (see section 6.1.2 on stand structure at low elevations). Large conflagrations in the future due to increases in tree density and fuel conditions could set back some areas to earlier successional stages than would have occurred during the HRV period. Such landscape level changes in the proportions of late-successional forest are discussed in more detail in section 6.2 on landscape structure.

4.2.3 Succession in aspen woodlands

Aspen-dominated woodlands can be seral or relatively stable depending on site conditions and historical factors (Reed 1971, Mueggler 1985, Bartos and Campbell 1998). Where seral to conifers, a few aspen usually persist in the understory of coniferous forests and rapidly produce an abundance of new sprouts over large areas when the next stand-replacing disturbance occurs. Many new stands of aspen “ramets,” that part of the clone that most people think of as the aspen plants (trees), are produced in this way. A single clone of aspen can be large

with thousands of sprouts, covering a hectare or more, and is thought to be very long-lived, perhaps thousands of years—though individual ramets produced in the clone often do not live more than 100 or 125 years (Veblen and Lorenz 1991, Mehl 1992). Notably, while young aspen shoots within a clone share the same root system with the mature canopy dominants, they typically are unable to survive in the forest understory environment. A warmer climate than during the HRV period could lead to more fires and greater proportions of the landscape in seral stages containing aspen. Aspen dynamics are discussed further in subsequent sections.

4.2.4 Succession in non-forest vegetation

As with forests on the SNF, the composition and structure of non-forest cover types is affected by fire. During the HRV period and in areas where big sagebrush was dominant, vegetation changes were quite dramatic after fire because the sagebrush is not capable of sprouting. When a fire occurs, grasses and forbs become the dominants for periods of a decade or more. Eventually, big sagebrush will reinvade as new seedlings become established, returning to pre-burn densities after about 15 to 20 years for Wyoming big sagebrush/Idaho fescue communities and about 25 to 30 years for Wyoming big sagebrush/bluebunch wheatgrass communities (K. Houston, SNF soil scientist, personal communication). Effective fire suppression could lead to a larger amount of land area in the shrub successional stage than during the HRV reference period because big sagebrush is a common, dominant shrub.

If the dominant shrubs are capable of sprouting, then the effects of fire would be not only less dramatic, but some shrubs, such as snowbush ceanothus (*Ceanothus velutinus*), that require the heat of fire to break seed dormancy (Zavitkovski and Newton 1968), would become more common. Similarly, some grasses, such as bluebunch wheatgrass, may decline following hot fires (or heavy spring grazing) (Knight 1994a), contributing to increases in shrub dominance. The outcome after a fire varies depending on species present before the fire. Notably, long drought periods can shift the species composition and the outcome.

As noted, the invasion of trees into subalpine meadows is a common observation (Patten 1969, Dunwiddie 1977, Gruell 1980, Schimpf and others 1980, Pearson and others 1987, Jakubos and Romme 1993, Hessler and Baker 1997, Meagher and Houston 1998). In many instances, fire suppression must be a contributing factor, but grazing by livestock and climate change should be considered as well. The effects of warmer climate may have been more than counteracted by fire suppression activities in the last century, with the net result being an increase in the frequency of succession from grasslands to shrubland, especially shrublands dominated by big sagebrush. However, there is no evidence to suggest that the shift is large enough to be beyond the HRV. Prescribed burns by forest personnel may be keeping successional patterns in non-forest lands within the HRV.

Two other factors must be considered when addressing succession in non-forest vegetation below timberline—the livestock grazing that was especially intense early in the 1900s and the invasion of exotic plants that has occurred throughout the last century. The effects of grazing is discussed later in this report, but in general, recent range management practices are believed to have restored about one-third of SNF rangelands to relatively good condition (~5% are in “poor” condition; USDA Forest Service 1996). This has occurred through natural succession following shifts in management policies and has led to favorable shifts in species composition and the accumulation of biomass.

Some believe invasion by exotic plants is the greatest threat the ecosystems in SNF are facing (K. Houston, SNF soil scientist, personal communication). Undoubtedly, recovery patterns have been less desirable where exotic plants are common. There has been essentially no research on the effects of invasive plants in Wyoming National Forests. In Montana, studies show spotted knapweed (*Centaurea maculosa*) is increasing, reducing the value of winter range for elk (Rice and others 1997). Stohlgren and his associates (Stohlgren and others 1999a, 1999b) found that habitats in the Rocky Mountains of Colorado with high native species diversity, such as aspen groves and some meadows, often are the habitats where exotic plant invasion is most likely. In general, the presence of exotic plants is likely to push many stand variables beyond their historic range of variability.

Invasive plants apparently have not yet become a serious problem in the alpine tundra of the SNF, although yellow toadflax (*Linaria vulgaris*) is present and has the potential to invade such areas in the future (K. Houston, SNF soil scientist, personal communication). Succession occurs very slowly in the alpine tundra and recovery from invasions or human caused disturbances, especially if the soil is degraded, could take a century or more. While little is known about changes through time in the alpine zone, the causes of plant distribution patterns is better known. Johnson and Billings (1962) conducted a classic study on the effects of processes driven by freezing and thawing on alpine plant distribution on the Beartooth Plateau, and Thilenius and Smith (1985) describe plant-environment relationships on the alpine ranges of the Absaroka Mountains.

5. HISTORIC RANGE OF VARIABILITY FOR DISTURBANCES COMPARED TO EXISTING CONDITIONS

A variety of natural disturbances occur in the Rocky Mountains. Fire is the predominant one (Clements 1910, Brown 1973, Romme and Knight 1981, Fahey and Knight 1986, Despain and Romme 1991), but windstorms and insect or disease epidemics can be important as well. The successional processes described in the previous section have been initiated over and over by these disturbances, creating a diverse landscape mosaic. Although we discuss each type of disturbance separately, the various disturbances are interactive. For example, a windstorm that topples a large number of trees may increase the probability of an insect epidemic or fire in some areas (Schmid and Frye 1977, Lindemann and Baker 2001), an insect outbreak may increase fuel loadings, and new edges created between burned and unburned stands create environments favorable for the windthrow of both dead and live trees as well as the spread of certain insects and pathogens (Knight 1994a).

Today, the frequency, size and intensity of historic disturbances, and the forests that develop after different kinds of disturbances, have been altered by management activities in some areas. In this section we examine these changes in more detail and attempt to characterize the HRV for the major disturbances.

5.1 **FIRE**

The evidence of historical fires is recorded by the annual rings of trees that survive fire events (fire scars) or by the age of trees that become established within a few years after stand-replacing fires (e.g., Romme and Knight 1981, Kipfmueller 1997, Baker and Kipfmueller 2000). Evidence can also be obtained from charcoal and pollen deposited in bog and lake sediments (Mehring and others 1977, Millspaugh and Whitlock 1995, Whitlock and Millspaugh 1996, Fall 1997, Kipfmueller and Baker 1998a, 2000). Using these techniques, it is clear that the frequency, size, and intensity of fire disturbances have been altered in some areas by management activities—mostly at low elevations. As we discuss, these changes have been in the opposite direction of expected changes caused by global warming.

To quantify changes in fire occurrence in the Rocky Mountain region, high-elevation forests must be separated from low-elevation forests because fire frequency varies by elevation and forest type. Low elevation forests had the highest number of fire starts per year between 1970 and 2000 on the SNF, where, on average, 38 and 82 fires started per year per 100,000 ha in Douglas-fir and limber pine forests, respectively (SNF Fire and RIS databases). In contrast, only two fires started per year per 100,000 ha in the spruce-fir/lodgepole pine forests and 27 per year per 100,000 ha in the whitebark pine forests. Various studies indicate that low-elevation forests of Douglas-fir, limber pine, and ponderosa pine burn more frequently with low-intensity fires than forests dominated by lodgepole pine or spruce and fir, which typically have high-intensity crown fires (compare Tables 4 and 5). However, the number of fire starts in high-elevation forests of whitebark pine is similar to low-elevation forests on the SNF, with 27 per year per 100,000 ha between 1970 and 2000 (SNF Fire and RIS databases). The elevational and forest-type differences probably are not an effect of fire management activities, even though modern fire suppression efforts are usually more successful at low-elevations and in whitebark pine forests (Morgan and Bunting 1990).

Unfortunately, the available fire data are not separable into categories defined by elevation or forest type. Consequently, some of our analyses for high-elevation

forests are based on the entire SNF because the large fires that started at high elevations sometimes burned downward into low-elevation forest types. Because the high-elevation forests make up the majority of the forested land area (69%), we concluded that this deficiency was not serious, though caution should be used when interpreting the data.

5.1.1 Fire in high-elevation forests

As lodgepole pine and in some areas whitebark pine are often seral to spruce and fir, the historic variability of these forest types is closely linked. Thus, their fire regimes are discussed together. In this section we describe and compare the HRV for fire-related variables to existing conditions.

Fire intensity. Fire intensity in high-elevation forests varies according to weather, fuel and site conditions. Low-intensity surface fires or mixed-severity fires (having both low-intensity surface fires and stand-replacing fires) can occur (Brown 1973, Arno 1976, Franklin and Laven 1989, Kipfmüller 1997), but often such fires become stand-replacing crown fires if spruce and fir are present because of the vertical continuity of fuels they provide. Patchy fires in fuel-limited whitebark pine forests are of low intensity or mixed severity. Whitebark pine forests usually survive low and sometimes moderate-severity surface fires with their moderate bark thickness (thinner than ponderosa pine but thicker than lodgepole pine; Keane and others 2001). However, in general, high elevation forests are mostly affected by stand-replacing fires in the Greater Yellowstone Area (Turner and others 2003, and others).

Fire return interval. Fire frequency in high-elevation forests can be estimated using mean fire intervals (MFI, measured in years) at two spatial scales—within stands and across landscapes. Perhaps the most useful scale for managers, as a guide to determining harvest rotation times, is the MFI of stand-replacing fires (i.e., the mean fire-free period) within a particular stand. Research suggests that the MFI for a particular stand of subalpine forest that contains spruce-fir and/or lodgepole pine is commonly 150 to 700 years (Table 4). Shorter intervals (30 to 304 years) result when the MFI is estimated from fire scars, as they consider the occurrence of

all fire events, not just stand-replacing fires. Whitebark pine forests typically have a shorter and more variable MFI than spruce-fir and lodgepole pine stands of about 10-300 years (Table 4). Some research suggests that the longest fire-free intervals are often associated with mesic sites found in ravines, at high elevations, on leeward or north slopes, and in unusually dry sites where fires may not occur at all (Billings 1969, Hawkes 1980, Romme and Knight 1981, Tinker and others 1994). However, another study (Baker and Kipfmüller 2000) could not find a relationship between old forests and topographic position in the 3,241 ha (8,005 acre) watershed they studied on the Medicine Bow National Forest.

The second scale for characterizing fire frequency is to calculate MFI for a landscape that encompasses many stands (e.g., Veblen and others 1994, Wadleigh and Jenkins 1996, Kipfmüller 1997, Kipfmüller and Baker 1998a). This landscape-level MFI is calculated simply as the arithmetic mean of intervals between all known fire events within a given landscape, even though different portions of the area burn each time (Romme 1980b). Little information of this nature is available on the SNF, but the landscape MFI averaged 29 years (range of 10 to 67 years) in northcentral Idaho (Barrett 1993) and 12.5 years (range of 1 to 55 years) in southcentral Wyoming during our HRV reference period in comparable forests and watersheds (Kipfmüller 1997). The landscape MFI is area dependent, and thus larger areas have lower MFI estimates (Arno and Peterson 1983). While MFI estimates from different areas are difficult to compare unless standardized on a per unit area basis, MFI is useful as an index of change in fire ignition over time within one landscape.

Fire size and extent. Insight into the size of fire events on a landscape and the occurrence of such large fires can be gained by considering both long-term and recent fire histories in YNP. The Park is an area of similar latitude, topography and vegetation as the SNF, and it has been relatively undisturbed by the influences of European-Americans. Indeed, the fire management policy in the Park since 1972 has allowed lightning-caused fires to burn in many instances, thereby providing a reasonable proxy or reference area that is appropriate for the SNF. As important, a considerable amount of fire history research has been done there.

At the landscape scale, Despain and Romme (1991) reconstructed the fire history for a 129,600 ha study area in southcentral YNP (approximately one-eighth the size of the SNF). Although fire has apparently occurred in every decade, they found that large scale, stand-replacing fires have occurred during four main periods since 1690 AD: 1690-1710, 1730-1750, 1860-1870, and 1988 (Fig. 22). Similarly, but working in northeastern YNP, Barrett (1994) found that major fires occurred over four periods within 81,000 ha (1740-1750, 1770s, 1820-1830, and 1980s; Fig. 22). The two studies did not identify the same decades for major fires, but all large fires occurred during relatively dry or hot periods (Fig. 21). Again, their results do not represent the interval between fires in a specific stand, but rather between major fire events somewhere in the study area (fire frequency or MFI for the landscape).

During each major fire period, a large percentage of the entire area burned (7-26% in any one decade in southcentral YNP, 11-58% in northeastern YNP), creating extensive, relatively even-aged patches of forest. Despain and Romme (1991) concluded that such even-aged forests progress into later successional stages and become increasingly susceptible to burning as fuel loading and fuel continuity increase, and that very likely the area would burn again after another 200-400 years, whenever weather conditions permitted, as lightning ignitions are frequent (Figs. 23 and 24). Since that work was done, additional observations in YNP have suggested that stand-replacing fires at high elevations are more likely to be affected by weather conditions than successional stage or fuel accumulation (Turner and others 2003).

Looking at the area burned per year during the period of no fire suppression from 1972-1987 in YNP, three important trends in fire size emerge: 1) the area burned in a single year is highly variable, 2) the mean area burned per year is toward the low end of the range, and 3) the median, possibly a better indicator of a typical year because of the skewed distribution, is only 12 ha per million hectares (Fig. 25). The overall fire regime in YNP is characterized by numerous fires that are unable to spread over more than a few hectares and very few fires that burn large areas, but which have by far the greatest effect on the landscape mosaic (Fig. 26). The 1988 fires are one of the four major fire events of the past 300 years and burned more land area than the other large fires combined over many decades (Figs. 22 and 27).

Possibly this was due to the warmer conditions in the latter half of the 20th century, apparently warmer than at any other time during the HRV period (Fig. 21).

Using YNP as an HRV proxy. When determining if the YNP fire regime can be used as a proxy of the HRV regime for the high-elevation forest of the SNF, differences between the climate and vegetation types of the two areas must be evaluated. Located on the east side of the continental divide, the eastern edge of the SNF tends to be somewhat drier and warmer than YNP, which could create flammable conditions more frequently; but near the divide, the SNF can be wetter (Fig. 10). Similar to YNP, the majority of the forests in the SNF occur at high-elevations. The National Forest has less forested area (64% vs. 80%), which could cause a lower level of flammability, and generally has less highly flammable forested land than in the Park (14 - 35% of land area in LP3 and SF successional stages, compared to 71% in YNP; based on Despain 1990 and the FIA and grizzly bear habitat databases). Predictably, the mean and maximum number of fire starts per year has been lower on the Forest than in Park during the last 30 years (Fig. 23).

Both the Park and the Forest had relatively large forest fires in 1988, although the spatial extent of the fires was much smaller on the Forest. The Clover Mist Fire, the major fire on the SNF in 1988, started in YNP and burned 78,550 ha (194,019 acres) in the northern SNF (Fig. 26). However, per 100,000 ha, only 8,106 ha burned on the SNF in 1988 compared to 44,200 ha in YNP (Fig. 27). Also, the area burned in the early 1900s tended to be less on the National Forest than in the Park.

Evaluation of the stand ages on the forests of the SNF shows large areas were burned in the 1800s, particularly in the late 1800s and early 1900s, when large areas of even-aged lodgepole pine forest were created (Figs. 28 and 29). However, many areas were not burned at that time, as spruce-fir forests ≥ 160 years old also are common (Fig. 29). European-American settlers, especially in the Wind River Range, may have been responsible for some of the fires around the turn of the 19th century (see Cultural Section). The variation in stand ages over the last couple hundred years in YNP (Fig. 28—bottom graph) demonstrates that stand ages currently seen in the SNF are just a snapshot in time and large fires will periodically change the age composition over time.

Overall, the SNF can have large fires, but they probably burn less area than in YNP due to a somewhat lower ignition potential and a more rugged topography that creates more fire breaks.

Changes from HRV. From 1945 into the 1970s, fires were most often caused by humans rather than lightning and were typically small (especially in the south, Fig. 24), even without suppression. Although fires would often be hidden in the rugged landscape and not easily accessible, the SNF did not seem highly flammable. Fire suppression became more feasible with the development of a corps of smoke jumpers in the mid-1900s (Cochrane and others 1988), but suppression efforts probably never became as extensive on the SNF as on some other National Forests because of fewer roads and more rugged topography. SNF policies allowed some fires to burn without immediate suppression.

Due to insufficient data we could not evaluate quantitatively the changes in fire regimes that might have occurred at the stand level in lodgepole pine, whitebark pine, or spruce-fir forests on the SNF. Our qualitative assessment suggests that the MFI of stand-replacing fires at high elevations in the last 100 years is within the range of means during the HRV period, despite fire suppression efforts, because such fires tend to have very long intervals (150 to 700 years, Table 4; low confidence)—intervals that are longer than the past century of management (Morgan and others 1996). An exception could be high-elevation whitebark pine stands, where a change in MFI appears to have occurred based on one study on the SNF (Table 4; Morgan and Bunting 1990), although it was not necessarily beyond the HRV (Morgan and others 1996). Fire intensity at high elevations in general probably is within the HRV (moderate confidence), as the forests of lodgepole pine, Engelmann spruce and subalpine fir still are characterized by stand-replacing fires. Again, the exception may be whitebark pine stands which may be undergoing more rapid succession to spruce-fir forests because of successful fire suppression. As a result, mixed-severity fire regimes (in whitebark pine only) are being replaced by stand-replacing fire regimes (in stands with spruce, fir and whitebark pine), although possibly not beyond the HRV (low confidence).

At the landscape scale, fire intensity probably falls within the HRV (moderate confidence) because most fires in the high-elevation forests in the past and present have been stand-replacing fires (Barrett 1995, Brown and others 1995). Research suggests that large, stand-replacing fires occur more often at high elevations because of weather conditions rather than fuel accumulation (Bessie and Johnson 1995, Weir and others 1995, Turner and others 2003). Romme and Despain (1989) suggest that, while extensive fires could have occurred in YNP anytime between 1930 and 1988 based on fuel loading, they did not burn until 1988 largely because of weather conditions. Turner and others (2003) concluded that fire suppression had little effect in creating conditions favorable for the extensive fires of 1988 in YNP.

With regard to MFI at the landscape scale, one might expect it to have increased beyond the HRV in high-elevation forests due to fire suppression efforts in the 1900s. Data from northcentral Idaho support such an increase, where, on a 3,232 ha tract of forest with lodgepole pine, Engelmann spruce, subalpine fir and Douglas-fir, the MFI averaged 29 years during the HRV period and increased to 42 years after 1860 (data from Barrett 1993). However, Kipfmüller (1997) found the opposite in southeastern Wyoming (on a 3,241 ha area)—a MFI of 12.5 years during the HRV reference period compared to an MFI of only 4 years in the 20th century. Both studies focused on relatively small landscapes. Unfortunately, except for one whitebark pine study (Morgan and Bunting 1990), no fire history studies have been completed on the SNF, making it difficult to decide whether the Idaho or southeastern Wyoming results are most applicable to the SNF. By the late 1970s many fires on the SNF were more easily extinguished because of more effective fire suppression.

Using Barrett's (1995) summary of fire frequency and intensity data for the Interior Columbia River Basin, Morgan and others (1996) concluded that forests of spruce-fir, lodgepole pine, or whitebark pine have not had their fire regimes (frequency and intensity) change since the HRV period. The SNF was included in the analysis of Morgan and others. Because many parts of the SNF have been inaccessible in the past century, and considering the results of Morgan and others, we conclude that the landscape-level (averaged across the entire forest) fire frequency (MFI) and intensity at high elevations on the SNF have not changed

beyond the HRV because of European-American activities (low and moderate confidence, respectively). Even if fire size has been reduced by fire suppression, the size change probably has not been great enough to affect the MFI in a measurable way.

With regard to fire size, Brown and others (1995) found that the total area burned on the Selway-Bitterroot National Forest in Montana was 1.7 times greater for high elevation forests (lodgepole pine, Engelmann spruce-subalpine fir, whitebark pine) during the HRV period than in recent years (1979-90), even though many fires (44%) were not suppressed or contained. The range of mean and standard deviations of total area burned per decade from the 1930s to 1970s on the SNF currently is somewhat lower than the historical conditions in YNP (1690-1860; Fig. 30). From the 1950s to 1990s, the mean falls within historical conditions in YNP (due to the 1988 Clover Mist Fire on the SNF), but the standard deviation is slightly higher. Additionally, the average area burned/hectare/year on the SNF was one-eighth the area burned during years in YNP when fire suppression was not practiced (1972-1987) (Fig. 25), and individual fires were small relative to YNP (Figs. 25, 27, and 31). The apparent lower ignition potential for sizeable fires on the SNF may be partly responsible for the reduced amount of area burned.

Firebreaks and mop-up efforts certainly can extinguish the potential for future flare-ups, thereby reducing fire sizes, and fire suppression efforts outside a management unit can affect fire frequency within a unit. For example, if fire suppression had been practiced in YNP in early summer 1988, SNF fire personnel believe the Clover Mist Fire in the SNF could have been extinguished at a small size (D. Sisk, SNF fire specialist, personal communication; Rothermel and others 1994). Such observations suggest that, without fire suppression, much larger areas would have burned on the SNF, and that the above differences between YNP and SNF are due more to fire suppression than lower fire ignition potential. Still, considering the few roads in the SNF, the large area in wilderness where fires often have been allowed to burn, and the abundant fire breaks created by rugged topography, we conclude that fire size and distribution of sizes (Fig. 32) is currently within the HRV at the landscape scale on the SNF (low confidence).

Other management activities on the SNF can alter ecosystems following a large fire. After the Clover Mist Fire, approximately 9,292 ha (23,000 acres) of burned land was seeded for sediment control (K. Houston, SNF soil scientist, personal communication). Most of the planted species were native grasses, but some areas outside wilderness areas were seeded with a non-native, alsike clover (*Trifolium hybridum*). This non-native changed the species composition in those areas, beyond the HRV, until it was naturally eliminated by canopy closure of the maturing trees.

On a broader geographic scale, fire suppression and clearcuts (which can act as fuel breaks) appear to be less important than climate change and fuel buildup in determining the extent of large fires. More specifically, weather conditions on the SNF may not have been conducive to large fires during the early and mid-1900s, when conditions probably were relatively wet, but by the late 1900s temperatures had become the warmest since 1500 AD (Fig. 21). Possibly for this reason, the area burned each year in the National Forests in all eleven western states more than doubled from the mid-1900s (1950-60) to recent years (the late 1980s and 1990s), despite more sophisticated fire-fighting techniques (Pyne 1997). Baker (2002) provides a detailed review of various factors affecting the frequency and extent of fires in the Rocky Mountain region, concluding that climate, fuels and elevation are linked causes and that ongoing land uses can be confounded with climatic effects. His analysis encompassed both high- and low-elevation landscapes.

Studies on fire history in the Greater Yellowstone Area suggest that up to 15% of the entire SNF landscape could burn in one year. Dealing with such large-scale disturbances presents a challenge to forest managers. On one hand, disturbances of the magnitude of the 1988 Yellowstone fires appear to be characteristic of our HRV reference period and have had very important ecological effects (e.g., Christensen and others 1989). Preventing such events in the future, if that is possible, could have unknown consequences for biological diversity. On the other hand, the large land area involved with such disturbances is an extreme in the HRV that the public hesitates to support (Swanson and others 1993, Manley and others 1995). Managers must consider such issues when deciding whether to manage high-elevation forests within the HRV.

Though our results suggest that fire suppression on a landscape at high elevations with limited harvests has probably had limited effects, this conclusion is still quite controversial. Many fires, whether started by lightning or humans, certainly have been put out, but without the wind and drought that accompany large fires, suppressed fires might not have burned over a large area anyway. Fire suppression in the SNF may have been most effective at reducing fire sizes during wetter periods, but such fires probably would have been small anyway.

It is probable that big fires will once again burn across the SNF at high elevations, regardless of modern fire suppression technology and silvicultural practices designed to reduce fuel loadings. In unmanaged forests, stand-replacing fires may occur in the same stand, on average, every 200-400 years, and they may burn over a large proportion of the landscape approximately once each century. Where landscape patterns and stand structures are modified by humans, as in a few areas on the SNF, such intervals may be more difficult to predict. Harvesting often is done with the objective of reducing fuel loads and flammability as well as providing a source of wood. However, as noted, wild fires that burn large areas probably are determined as much by drought and wind as by fuel loadings, if not more so (Clark 1990, Johnson and Larsen 1991, Bessie and Johnson 1995, Weir and others 1995, Baker 2002, Turner and others 2003). Where there are enough live trees to be economical for timber harvesting, there probably is enough fuel to carry a stand-replacing fire under conditions of drought and high winds.

Another controversial issue associated with fire in high-elevation coniferous forests is the frequency of surface fires burning through the understory, thinning some of the trees in the process and thereby altering stand structure in a way that could be emulated with selective harvesting. The abundance of flammable low branches on the shade tolerant spruce and fir suggests that such fires would be rare where these species are common. However, the lower branches of lodgepole pine are shed by the tree, creating an open understory. Surface fires in lodgepole pine have been observed, and because of the thin bark, some trees are killed (Turner and others 2003). Others survive but develop fire scars, sometimes multiple scars. Data presented in Table 4 and in Lotan and others (1985) suggest that surface fires may

have burned through lodgepole pine forests in the Rocky Mountains at intervals as short as 40 to 80 years—considerably shorter than the 200+ intervals usually reported for YNP. Non-lethal fires in lodgepole pine have been documented to occur on gently sloped, dry areas and are common in whitebark pine stands (Brown and others 1995), but the amount of forest burned in this way on the SNF is not known. Stand-replacing fires accounted for 72% of the total area burned in western Montana and northern Idaho (Brown and others 1995), but some of this land area was at low elevations.

In summary, stand-replacing crown fires dominate the fire regime in high-elevation forests, but surface fires did and continue to occur periodically—especially on the edges of crown fires where flame lengths (a measure of fire intensity) are shorter and in whitebark pine stands. MFI, fire size, and intensity at high elevations are within their HRV in most of the SNF (low, low, and moderate confidence, respectively). Exceptions would be around buildings and along highways where fires are fought vigorously whenever they occur.

5.1.2 Fire in low-elevation forests

Fire intensity and return intervals. Commonly the fire regime of low-elevation forests in the Rocky Mountains during the HRV period is characterized as frequent, low-intensity surface fires that kill most of the small trees but few of the older, thick-barked trees—thus maintaining low density woodlands. Historic accounts of low-elevation forests that include ponderosa pine and/or Douglas-fir in other parts of the Rocky Mountains describe "open and park-like" stands with understories rich in grasses, such as in the American Southwest (Cooper 1960), Colorado (Veblen and Lorenz 1991), and northern YNP (Houston 1973).

In northern YNP, Houston (1973) found a mean *fire return interval* of 20-25 years in Douglas-fir stands during the HRV period. Arno and Gruell (1986) found mean intervals of 35-40 years in southwestern Montana. Other intervals reported for low-elevation stands of Douglas-fir in the region with a surface fire regime are generally of similar magnitude, except in areas where Douglas-fir occurs with grand

fir (where the mean fire intervals are 107 to 216 years; Table 5). Grand fir does not occur on the SNF. Notably, Baker and Ehle (2001) concluded that the ranges that have been reported for low-elevation forests are too narrow and are biased toward shorter intervals. At the landscape scale, little is known about the MFI of low-elevation forests.

In addition to surface fires, there is evidence of crown fires that kill tracts of Douglas-fir and ponderosa pine over large areas. Littell (2002) found a mixed-severity fire regime in low-elevation forests in and around northern YNP, and photographs in the late 1800s provide evidence for mixed-fire regimes in stands of Douglas-fir on the Bridger-Teton National Forest (Gruell 1980). This fire regime is most common in the moist, cool Douglas-fir habitat types, whereas frequent surface fires are most common in dry Douglas-fir habitats (Bradley and others 1992). The limber pine habitat type also tends to have a mixed-fire regime (Morgan and others 1996), but often with longer intervals than for Douglas-fir woodlands (about 74-100 years, Bradley and others 1992), perhaps because fuel production is slower on the exposed sites where limber pine occurs. Thus, low intensity and mixed-severity fire regimes probably characterized the low-elevation Douglas-fir forests and woodlands on the SNF.

The vegetation database developed for the grizzly bear habitat analysis (Mattson and Despain 1985) shows that about 62% of the 1988 Clover Mist Fire in Douglas-fir stands on the SNF was in the crown and 38% was on the surface. The high percentage of stand-replacing fires may be a shift from historic conditions (see Brown and others 1995), which would be consistent with conclusions drawn for the region (Morgan and others 1996). In northern YNP, Douglas-fir stands also experienced mixed-severity burning in 1988, but it is unknown what proportions were surface and crown fires (Barrett 1994). In low-elevation forests in the Interior Columbia River Basin, Morgan and others (1996) concluded that non-lethal, frequent fire regimes are being replaced with infrequent mixed-severity fire regimes and that frequent mixed-severity regimes are being replaced with stand-replacing, infrequent fire regimes.

A factor complicating the interpretation of what may be natural at low-elevations is the role that humans have had in influencing the fire regime. Prior to the arrival of European-Americans, fires set by Native Americans probably were most common at low elevations (e.g., Barrett and Arno 1982, Gruell 1983a) and were inseparable from "natural" fires (Martin and Sapsis 1991). Fisher and others (1987) found evidence that fire in ponderosa pine at nearby Devil's Tower National Monument became more frequent from 1770 to 1900 (mean interval = 14 years) than it had been from 1600-1770 (mean interval = 27 years), possibly, they concluded, because of the expansion of Sioux Indians into the area after 1770.

Heavy livestock grazing on the SNF may also influence fire return intervals, especially at low elevations. Fine fuels in the form of grasses are reduced by grazing, thereby reducing the likelihood of a fire starting in the area and extending into the forests. Fires in the Bighorn Basin to the east of the Forest could easily have spread into the forests of the nearby mountains, but such fires also are more readily suppressed. Fewer fires in the Basin, because of either grazing or suppression, leads to fewer fires on the SNF.

Fire size. Fire size and intensity probably is directly proportional to the length of the fire-free interval, with more large, severe fires occurring where or when fire intervals (or climate oscillations) are longer and there has been more time for fuels to accumulate. At Devil's Tower National Monument in Wyoming, fires caused by individual lightning strikes were historically common, but typically they would damage only one or two trees before burning out due to weather conditions or fuel discontinuity (Fisher and others 1987). As at higher elevations, larger fires were less frequent, occurring only at times when weather and fuel conditions facilitated fire spread. In parts of the Black Hills with ponderosa pine, the fires apparently could be larger, less frequent, and more intense—like those in lodgepole pine forests (Shinneman and Baker 1997). Historic fires in the Black Hills may have been as large as 20,000 ha (Dodge 1876), or even 60,000 ha (Shinneman and Baker 1997, based on Graves 1899)—although, with one exception, fires since 1860 have been less than 9,000 ha. The exception was the Jasper Fire in 2002 that burned 33,850 ha (83,580 acres). Notably, these fires were in ponderosa pine forests, which do not

occur in the Shoshone National Forest. Nevertheless, based on FIA data (Fig. 29), smaller, stand-replacing fires in the low-elevation forest on the SNF might have occurred in the 1840s and 1920s. During 1988, the Clover Mist Fire burned 4,752 ha in Douglas-fir forests and an unknown area (<1919) of limber pine forests (USDA Forest Service 1994a). Much of the Clover Mist Fire burned at high elevations (26,000 ha). Fire sizes at high elevations on the SNF generally have been much larger than at low-elevations (Fig. 29).

Changes from HRV. Surface and mixed-severity fires are much easier to suppress than the crown fires that develop so rapidly at high elevations, partly because low-elevation forests are more accessible and surface fires are extinguished more easily. Thus, there is the potential for greater changes due to fire suppression. Houston (1973) found the fire return interval for Douglas-fir in the Northern Range of YNP lengthened from a mean of about 20 to 25 years during the HRV period to >70 years (range 70-150 years, and no fires between 1900 and 1970), probably due to fire suppression (Table 5). In his study fire frequency also was reduced, as the number of fires ranged from 1 to 5 per 50-year period historically (1570-1870) and dropped to 0 to 2 in modern times. In another study at low elevations in the northern Greater Yellowstone Ecosystem, Littell (2002) found evidence for 1 to 5 fires during 50-year intervals from 1570-1870 but only 0 to 1 fire per 50-year interval after that.

Other studies in low-elevation forests also show that the mean fire intervals became longer with management by European-Americans (Table 5), such as in the Ashenfelder Basin, just north of Laramie Peak (Medicine Bow National Forest), where the last fires recorded by Brown and others (2000) occurred in 1911 and 1909 at two sites within the basin. With few exceptions (Goldblum and Veblen 1992, Table 5), fire suppression probably has lengthened the stand-scale MFI (Table 5) beyond the HRV in most low-elevation forests, and we believe the same is true for the SNF (high confidence). Although less information is available at the landscape scale, we suspect the MFI is lengthened beyond the HRV at that scale, too (moderate confidence).

In Colorado, Brown and others (1999) found evidence to suggest that widespread fires on low elevation forests occurred on average every 59 years, but

that such fires did not occur after European-American settlement in the late 1800s. Similarly, since 1970, the surface and mixed-severity fire regimes at the lower elevations of the SNF have been dominated by numerous small fires with relatively few widespread fires (see data for Douglas-fir and limber pine in Fig. 29). From 1970-2000, 96% of all fires in Douglas-fir on the SNF were <4 ha, partially because they usually were suppressed. In contrast, fires of at least 20 ha appeared to be common historically in YNP (Littell 2002). This kind of evidence suggests that fire suppression has been effective and that fire size (low confidence) as well as frequency has exceeded the HRV for stands and landscapes.

In general, because of successful fire suppression in low-elevation forests, some savannas have changed into dense forests during the past century. Such forests now have high densities of small-diameter trees (Covington and Moore 1994, Knight 1994a, Fule and others 1997). Historic photo comparisons (1885 to 1970) in northern YNP, near the SNF, show that Douglas-fir forests have increased in density (Houston 1973, Houston 1982). Additional factors such as livestock grazing may also have contributed to this increase in density by reducing the competition for tree seedlings from grasses and forbs, as was found in Utah (Madany and West 1983), though livestock trampling could have offset such increases. Unique climatic conditions during the early 1900s (especially the wet, warm conditions from 1910-1920) also may have played a role in the increase of tree density and fuel loadings (Savage and others 1996). The result is that, regionally, intense stand-replacing fires in low-elevation forests generally have become more common in recent years than they were during the 18th and 19th centuries (Agee 1997). Notably, despite the relatively small extent of most fires at low elevations in Wyoming during the last 50 years, two very large crown fires occurred in 2002 in the low-elevation ponderosa pine forests of the Laramie Peak area—the Hentzel Fire that burned 6,016 ha (14,855 acres) and the Reese Fire that burned 7,633 ha (18,848 acres). These large fires suggest that fire intensity in stands and across the landscape may have increased because of the increased fuel loading, possibly beyond the HRV (low confidence).

Some surface fires are now prescribed on the SNF for low-elevation forests and non-forested vegetation types (K. Houston, SNF soil scientist, personal communication), and such burns can help restore the fire regime to the HRV. However, the timing of such fires usually does not match the timing of fires during the HRV reference period, as prescribed fires typically are spring burns rather than summer burns (K. Brauneis, SNF fuels manager, personal communication).

5.1.3 Fire in aspen woodlands

Aspen populations have changed considerably since the beginning of European-American occupancy in the Rocky Mountains. Extensive fires during the 1800s appear to have created large stands of aspen, both in Colorado (Veblen and Lorenz 1986) and on the west slope of the Sierra Madre in southern Wyoming (burned in 1841). It is unclear if these large stands developed largely from sprouts (ramets) or seedlings (genets). It is also unclear what trees dominated the vegetation prior to the fires (von Ahlefeldt and Speas 1996, from Severson 1963). Aspen occurs on only about 1% of the SNF, but, where it occurs, fire probably has played a similarly important role in the establishment of new stands (Brown and DeByle 1989, Romme and others 1997).

Fire frequency. Mean fire return intervals for aspen groves during the HRV period probably are essentially the same as for the other forest types with which they occur. This is because, while aspen is not considered to be highly flammable, the stands often burn when the adjacent coniferous forests burn (DeByle and others 1987, 1989). Veblen and others (1994) found fire return intervals in aspen forests of 160 and 240 years (mean = 202) in Colorado, which was similar to some nearby conifer stands in the same area. Lightning-caused ignitions in aspen stands are probably rarer than in adjacent coniferous forests. High intensity fires may kill the root systems of some aspen, thus favoring development of conifers (Parker and Parker 1983). However, the relatively mesophytic herbaceous understory in many aspen stands probably results more often in cooler fires than in adjacent stands of conifers. Such fires favor the development of large numbers of aspen root sprouts

even though the aboveground part of the older trees (shoots) is killed (Veblen and Lorenz 1991).

Changes from HRV. The abundance of aspen surely has fluctuated with the frequency, size and intensity of fires, although grazers and browsers also may have had an effect. Across the Rocky Mountains, aspen stands are apparently declining in some areas but not in others (Campbell and Bartos 2001, Shepperd and others 2001).

Though covering only about 1% of the SNF land area, most aspen stands here have trees (ramets) that are 60 years old or less (Fig. 29). However, the grizzly bear habitat database for the northern and higher elevation parts of the SNF indicates 69% of the aspen stands are in a late successional stage (ASP3) and only 9% are of recent regeneration (ASP0). For the SNF as a whole, the data are inadequate for making judgments about the representation of different age-classes now or in the past.

The data that are available suggest that seedlings and young sprouts are uncommon, perhaps more so than during the HRV period. Fire disturbances during the last century may have enabled the establishment of new groves through seedlings or the regeneration of old groves through root sprouting, but it is difficult to determine if they were previously more or less common because aspen shoots (the ramets) usually live less than about 100 years (in contrast to the root system which lives much longer). Fire suppression during the last 75 years has likely reduced fire frequency and extent below the HRV in some aspen stands (Shepperd and others 2001). Houston (1973) noted apparent declines in aspen to only 2 to 4% of his study area in northern YNP, which he postulated was due primarily to fire suppression and secondarily to elk browsing of young root sprouts (Loope and Gruell 1973, Kay 1990). A cursory survey on the SNF shows conifers are slowly replacing the seral aspen stands in a number of areas, probably more than would have occurred without fire suppression. Overall, similar to low-elevation coniferous forests, the fire frequency and extent in aspen today, at both the stand and landscape scales, is probably lower than during the HRV period (moderate confidence).

5.1.4 Fire in non-forest vegetation

As noted previously, the composition and structure of non-forest cover types can be affected by fire, especially where non-sprouting big sagebrush occurs and where the fire regime has been altered by fire suppression and grazing. Historic photographs (late 1800s) show that there was much less sagebrush on bunchgrass steppes in northern YNP than in 1970 (Houston 1973) and in some areas on the Bridger-Teton National Forest (Gruell 1980), probably due to fire suppression (although some areas also showed an increase in sagebrush). In shrubland communities in northern Idaho and western Montana, fire intervals averaged 54 years during the HRV period and the stands were estimated to burn 1.3 times more area than have burned in recent years (1979-90) under a prescribed fire regime, where 38% of the fires have been contained (Brown and others 1995). The Forest Service has conducted prescribed burns to reduce encroachment of woody plants such as big sagebrush (Cook and others 1994), which may help maintain the fire regime within the HRV. Although fire suppression could cause increases in shrub cover and tree encroachment, we suspect the change is not large enough to have exceeded the highly variable HRV at the stand or landscape level (low confidence). Fires are still frequent in non-forested habitats and can burn areas up to 700 ha (e.g., the Clover Mist Fire of 1988; Fig. 33).

Exotic plants can become a problem after fire and other disturbances, especially at lower elevations and in riparian zones. Cheatgrass (*Bromus tectorum*) and muskthistle (*Carduus nutans*) increased after prescribed fires in Sinks Canyon (K. Houston, SNF soil scientist, personal communication), and studies done elsewhere indicate that increases in cheatgrass can cause a dramatic reduction in the MFI and the abundance of some native plants (Knight 1994a, Morgan and others 1996). The MFI probably has been reduced in comparable areas on the SNF as well (high confidence). More than in forest vegetation, dramatic changes in non-forest vegetation could occur in the future if exotic plants become more widespread.

5.2 INSECTS

Mortality caused by insects has been an important form of disturbance in the high-elevation forests of the Rocky Mountains. The spruce beetle (*Dendroctonus rufipennis*), mountain pine beetle (*Dendroctonus ponderosae*), and Douglas-fir beetle (*Dendroctonus psuedotsugae*) are capable of reaching epidemic population sizes in spruce-fir, lodgepole pine, and Douglas-fir forests on the SNF. As with fire, historic epidemics can be detected in the annual growth-ring patterns of trees, providing insight into pre-settlement dynamics (e.g., Veblen and others 1994). No studies of this nature have been done on the SNF, and therefore our current understanding of the HRV for insect populations since the 1500s must be derived from general observations during the last 100 years and from studies conducted elsewhere.

5.2.1 Insects in high-elevation forests

Mountain pine beetle. Based on historical reports that date back 100 years, the mountain pine beetle and other associated bark beetles periodically have been a problem on the SNF. A 1905 report stated that “very little insect infestation is in evidence upon this Forest” (cited in Cochrane and others 1988). However, such a finding may reflect a high tolerance for insect damage at that time, as that report and other SNF memoranda make it clear that infestations by different beetle species were present in large enough abundance that managers sought advice from entomologists. In the 1930s, a forest entomologist referred to the insect problem on the SNF as possibly the most complex of any National Forest in the system (Cochrane and others 1988). He further commented that insects (primarily bark beetles) had destroyed more timber than fires on that portion of the present-day SNF known at one time as the Washakie National Forest. Extensive infestations by beetles have continued to be a problem to the present time (Cochrane and others 1988, Harris and others 1997).

Areas with beetle outbreaks were treated by harvesting infected trees between 1930 and 1960, and in the 1960s some large areas (>400 ha) were clearcut to reduce the rate of infestation (Berntsen and others 1971). Although recent investigations have found that cold, high-elevation environments like much of the

SNF may inhibit mountain pine beetle epidemics (Amman and others 1977, Amman 1989), they clearly still occur on the SNF, even in the high-elevation whitebark pine forests (Cochrane and others 1988). Often, the high-elevation pines appear to have been attacked by beetles flying from lodgepole pines at lower elevations (Ciesla and Furniss 1975, McGregor and Cole 1985, McGregor and others 1987).

The mountain pine beetle survives in the phloem of the inner bark throughout most of its life cycle (McCambridge and others 1979). While healthy trees can often defend themselves by producing resin to expel the beetle, trees that are stressed or damaged may lack this ability. Stand thinning is sometimes prescribed to minimize the stress of competition for water and nutrients (McGregor and others 1987, Christensen and others 1987). Once a tree becomes infested, it typically dies within a year, primarily because the blue-stain fungus introduced by the mountain pine beetle impedes the flow of water and nutrients through the sapwood (Christiansen and others 1987).

Mountain pine beetle epidemics typically start in large diameter trees that are diseased or stressed. Thus, most outbreaks occur in stands that are at least 80 years old and have trees at least 20 cm DBH (Koch 1996, Schmid and Mata 1996). Typically, the larger trees are killed in groups of 100 or more (Schmid and Mata 1996). The result is to reduce the average live tree diameter in a stand while creating small to large openings (Schmid and Mata 1996). By killing large trees, mountain pine beetle epidemics increase the number of standing dead trees that can become snags, which eventually fall and become coarse woody debris, thereby improving habitat quality for a number of species (Balda 1975, Amman and others 1977). Also, the canopy gaps stimulate the growth of understory plants and small trees (Stone and Wolfe 1996), which creates a ladder of fuel from the forest floor to the canopy as they grow taller. Hence, flammability increases because of the beetles. The large size of the Dinwoody Fire in 1935 (606 ha) in the Wind River Range was blamed on a large outbreak of mountain pine beetles that had occurred earlier in a whitebark pine area (Cochrane and others 1988). Notably, the total productivity of the forest can return to pre-epidemic levels within 10-15 years (Romme and others 1986).

Hazard ratings have been developed for the mountain pine beetle based on tree diameter, age, and location by latitude and elevation (Amman and others 1977). Older stands of large lodgepole pine at lower elevations are of highest susceptibility. More specifically, stands with diameters of <18 cm DBH, 18-20 cm DBH, and >18 cm DBH are of low, moderate, and high susceptibility, respectively. Stands <60 years, 60-80 years, and >80 years are considered to be of low, moderate, and high susceptibility, respectively. Elevation of stands >2590 m (8,495 ft), 2290-2590 m (7,511-8,495 ft), and <2290 m (7,511 ft) are also of low, moderate, and high susceptibility, respectively. Overall, the susceptibility of many pine stands in the SNF is probably low to moderate based on elevation (mostly >2290 m), but even moderately infested areas still can receive 20-50% loss of their overstory trees (Harris and others 1997). At the latitude of the SNF, Amman and others (1977) estimated that lodgepole pine forests up to 2500 m (8,200 ft) elevation are susceptible to 25 to 50% mortality, while those above 2500 m (8,200 ft) would likely incur less than 25% mortality. Some research suggests that infestation risks may be even lower where *Vaccinium scoparium* is dominant in the understory, as is common throughout the SNF (Steele and others 1983).

Frequency and severity of mountain pine beetle epidemics. The historic frequency and severity of MPB epidemics during the HRV period vary considerably in the Rocky Mountains, depending on environmental conditions such as elevation and climate. On the scale of an entire National Forest (landscape-scale), an epidemic may occur every 20 to 40 years. Return intervals for an individual stand are longer, 50-100 years in some areas (Johnson 1995, Schmid and Mata 1996). The duration of individual epidemics may vary from one to ten years, with the average being six (Schmid and Mata 1996). Epidemic severity has been highest in the northern Rockies, where millions of trees have been killed in a single year (Klein and others 1979), although outbreaks also have been large in Colorado (Schmid and Mata 1996). Notably, such mountain pine beetle outbreaks can change the forest species composition. In seral lodgepole pine stands, mountain pine beetle epidemics can eliminate the overstory pines and accelerate succession toward a forest dominated by subalpine fir and Engelman spruce (Veblen 2000).

The fact that mountain pine beetle epidemics are more frequent at lower elevations suggests a relationship with warmer temperatures. Thus, it is reasonable to suggest that epidemics have become more common with climate warming during the last century and that this trend will continue if global warming continues.

Spruce beetle. As with mountain pine beetle, the spruce beetle occurs in the spruce-fir forests of the SNF at endemic levels and some of the large beetle infestations in the last century reported by Cochrane and others (1988) probably were of this species. Spruce beetle epidemics have been known to cause heavy spruce mortality throughout much of Colorado and central New Mexico (e.g., Baker and Veblen 1990, Veblen and others 1991a, Veblen and others 1994). The two largest recorded outbreaks in Colorado occurred in the mid- to late-1800s (Sudworth 1900, Baker and Veblen 1990) and in the 1940s (Hinds and others 1965).

Frequency and severity of spruce beetle epidemics. As no specific information is available for the SNF, our estimates for the HRV of spruce beetle epidemics are based primarily on studies and observations from Colorado and New Mexico. The 1940s epidemic, which killed 99% of overstory spruce trees across more than 282,800 ha on the White River National Forest, is believed to be an extreme example (Hinds and others 1965, Schmid and Mata 1996). Historically, it appears that spruce beetle epidemics kill anywhere from 5 to 99% of large spruce trees (>10 cm DBH) over areas ranging from small patches to thousands of hectares (Schmid and Mata 1996). Based on tree-ring records in a part of the White River National Forest, Veblen and others (1994) estimated that the average return interval for spruce beetle epidemics on the landscape was 116 years (range = 111-122 years), with a period of 259 years required for the entire stand to be affected (turnover time). Because the spruce beetle typically does not attack trees less than 70 years old, epidemics are not likely in stands that have experienced disturbance in less than that time (Veblen and others 1994). Notably, because the spruce beetle does not attack subalpine fir, a rapid shift in tree species abundance occurs that may persist until the next crown fire or wind storm (Schmid and Hinds 1974).

Forest susceptibility to spruce beetle epidemics in general has not been defined as precisely as for mountain pine beetle, but Schmid and Frye (1977)

identified four characteristics of highly susceptible stands: 1) average DBH >41 cm; 2) basal area >34 m²/ha; 3) >65% spruce in the canopy; and 4) locations in well drained creek bottoms. Some spruce stands on the SNF, such as in the Ramshorn Area, have been rated as moderately susceptible to spruce beetle overall because they have large spruce trees and high basal areas, although a lower proportion in spruce (Harris and others 1997). Other areas that have not yet been inventoried may have a greater incidence of high susceptibility ratings.

Areas of large blowdowns tend to be more susceptible to spruce beetle outbreaks because endemic populations are normally found in windthrown or otherwise fallen spruce trees. After large blowdowns, endemic populations can increase to epidemic levels and many trees are killed. The 1940s epidemic in Colorado, for example, was apparently triggered by an extensive blowdown in 1939 (Hinds and others 1965). Large blowdowns are not common on the SNF, but they have occurred on the other side of the Continental Divide in the adjacent Bridger-Teton National Forest (Knight 1994a). Notably, the large blowdowns in the 1950s and 1990s on the Bighorn National Forest in northcentral Wyoming did not cause large spruce beetle outbreaks (1953 Bighorn National Forest historical report, Allen and Harris 1999). Hence, it is not clear whether blowdowns always increase susceptibility.

Western balsam bark beetle and "fir decline". In conjunction with other disease agents, the western balsam bark beetle (*Dryocoetes confusus*) may be causing widespread mortality of subalpine fir throughout the Rocky Mountains (Allen and Harris 1999). It is a native insect like the mountain pine beetle and spruce beetle, but it mainly attacks subalpine fir. Root disease pathogens such as *Armillaria* spp. and *Annosus* (*Heterobasidion annosum*) may contribute to what has been referred to as a region wide decline in subalpine fir (Morrison 1981, Wargo and Shaw 1985). To date, the exact cause and disease agents responsible for the decline in subalpine fir are unknown, but the decline is obvious in numerous areas, such as the Ramshorn Area on the SNF (Harris and others 1997). This disease has the potential to spread rapidly throughout the SNF as it has in other Wyoming forests (e.g., Bighorn National Forest). Affected areas typically are circular patches of several

dead and dying trees. Young seedlings and saplings growing in the area are also dead, and surrounding trees have poor growth (Shaw and Kile 1991). Because very little is known about the cause of the fir decline, no information is available about its HRV.

Other insects. Other insects are known to occur in the high-elevation forests of Wyoming and Colorado, such as the pandora moth (*Coloradia pandora*) (Schmid and Mata 1996), but they have not been mentioned as potential concerns on the SNF (Harris and others 1997). As with the other insects, they typically cause mortality or reduced vigor in individual trees or small groups of trees, thereby contributing to the structural diversity of the forests in the same way they have for centuries.

Changes from HRV. Thinning of lodgepole pine stands can reduce losses during mountain pine beetle and spruce beetle epidemics (Waring and Pitman 1985, MacGregor and others 1987, Amman and others 1988, Gibson 1989, Cole 1989, Schmid and Mata 1996). Moreover, clearcutting removes all suitable host trees and thus reduces the threat of outbreaks. Removal of large diameter lodgepole pine trees, the first trees that are attacked by mountain pine beetle, can also reduce losses during epidemics. On the SNF, clearcutting, thinning, and selective removal of large trees have occurred in only 3% of the forests, mainly in the northern Wind River Range (Figs. 15 and 16). It is unlikely that this small amount of harvesting has had a significant effect on the populations of the insects that cause tree mortality. Given that this harvesting has been located primarily in the Wind River Range, and that epidemics still occur there (e.g., in the 1980s), it is unlikely that the frequency or severity of mountain pine beetle, spruce beetle or western balsam bark beetle outbreaks (and the current episode of fir decline) are outside the HRV at the stand or landscape level in high-elevation forests (moderate confidence).

Overall, Schmid and Mata (1996) suggested that all insect populations and their effects are currently within their respective HRVs in the southern Rocky Mountains, with some important caveats. Specifically, measurements of epidemics are imprecise, making the comparison of past, present and future epidemics difficult. It is clear, however, that the insects affecting Rocky Mountain forests are native, and probably they have reached epidemic population sizes from time to time throughout

the post-glacial history of these forests. Human influences such as silvicultural treatments can slow the spread of outbreaks, but fire suppression can have the opposite effect. Intensification of either management activity, combined with climate change, could cause epidemic frequency and severity to move either above or below historic levels (Schmid and Mata 1996).

5.2.2 Insects in low-elevation forests

Douglas-fir beetle. The primary insects affecting low-elevation forests on the SNF are the Douglas-fir beetle (*Dendroctonus pseudotsugae*) and the western spruce budworm (*Choristoneura occidentalis*). Typically, the Douglas-fir beetle reproduces in scattered trees that are highly stressed, such as defoliated or fire-scorched trees (Lessard and Schmid 1990, McMillin and Allen 2000). If enough stressed trees are available, the beetles can increase and infest nearby healthy trees (Furniss and others 1981). Thus, epidemics frequently occur after extensive fires, which can create large areas of weakened Douglas-fir (McMillin and Allen 2000). This occurred in 1988, when extensive wildfires scorched many Douglas-fir in the SNF that apparently increased beetle infestations. Subsequent beetle generations have moved from injured trees to undamaged neighboring stands (Pasek 1990), killing about 23,000 trees over the 7-year period following the fires (McMillin and Allen 2000). Additional mortality is continuing, making the Douglas-fir beetle an important disturbance agent at low elevations. Similar events are taking place in burned Douglas-fir stands in YNP (Rasmussen and others 1996).

The Douglas-fir beetle produces one generation per year, with most new attacks by the flying adults in the late spring to early summer (McMillin and Allen 2000). Broods develop under the bark and overwinter as either adults or larvae. Douglas-fir beetle attacks are most successful on the large-diameter, mature or overmature trees in high-density stands that contain a high proportion of Douglas-fir in the overstory (Schmitz and Gibson 1996). McMillin and Allen (2000) found that three factors, namely, total basal area, Douglas-fir basal area, and the relative abundance of Douglas fir, were most important in determining the level of tree mortality on the SNF.

The Douglas-fir beetle can cause widespread mortality of Douglas-fir in the southern Rocky Mountains, with outbreaks occurring every 15 to 35 years in the same area that last from 5 to 10 years (Hadley and Veblen 1993, Schmid and Mata 1996). This beetle has affected very large areas on the SNF. From 1992 to 1999, 7,569 ha on the SNF (Clarks Fork Ranger District) were infested with Douglas-fir beetle and 37,548 trees were killed (58,627 m³ of timber) (McMillin and Allen 2000). An additional ~90,200 ha were infested in this district and other districts of the SNF, and 573,450 trees killed, from 2000 to 2005 (904,885 m³ of timber; K. Allen, Entomologist, Forest Health Management, U.S. Forest Service, personal communication). Tree age in the area averaged 165 years (Negron and others 1999), so it will probably require many years to return to pre-infestation tree densities and size, depending on the severity of tree mortality for a given stand (McMillin and Allen 2000). The infestations have changed the forest stand structure, reducing the percentage of Douglas-fir in the overstory by >15%, total basal area by 40 to 70%, and tree diameter by 8 to 40%. The understory vegetation increased three-fold, though the proportion in grass, shrubs or forbs did not change. Conifer regeneration increased nearly four-fold, with 90% of the regeneration being Douglas-fir. Although progression toward a steady-state Douglas-fir stand is slowed by such infestations, they are not changing long-term successional patterns (McMillin and Allen 2000). Similar to mountain pine beetle, Douglas-fir beetle outbreaks increase the flammability of the forest (Veblen 2000). The practice of harvesting in old, mature stands and thinning the younger stands can reduce the susceptibility of Douglas-fir to the beetle (McMillin and Allen 2000).

Western spruce budworm. The western spruce budworm (*Choristoneura occidentalis*) is the most destructive defoliator in western forests in general when population sizes are high. Its main host is Douglas-fir, with spruce and subalpine fir attacked much less frequently (Van Sickle 1985, Schmid and Mata 1996). The larvae feed on buds and new needles in the spring, after which the adult moths disperse into other trees (mostly in July and August). Heavy outbreaks can completely defoliate overstory trees in 4 to 5 years. Defoliation reduces growth rates, reproduction, kills tree tops, and eventually kills whole trees. In contrast to mountain

pine beetle, the smaller, suppressed trees are killed first, resulting in a higher average stand DBH but less total basal area (in live trees).

The western spruce budworm can cause extensive defoliation and tree mortality over several years (Veblen 2000). The frequency of budworm epidemics in the Southern Rocky Mountains during the HRV period ranged from 20 to 33 years in the same stand, with individual epidemics lasting a variable amount of time but averaging 11 years (Lynch and Swetnam 1992, Swetnam and Lynch 1993). Epidemics of western spruce budworm have occurred on the SNF, primarily on Douglas-fir, but currently the population levels are low (W. C. Schaupp, Jr., USFS Rocky Mountain Research Station Entomologist, personal communication).

Mountain pine beetle. An extensive outbreak of the mountain pine beetle occurred in the Dunoir area of the Wind River Range on the SNF in 1980, mostly in limber pine at low elevations (Raimo and Haneman 1981). Too little research on the SNF or nearby has been done to estimate epidemic frequency and severity, particularly in limber pine, but mountain pine beetle epidemics in the Southern Rocky Mountains have been more severe and more frequent in low-elevation forests of ponderosa pine than in lodgepole pine, perhaps because of warmer winters at lower elevations. On the landscape scale, Schmid and Mata (1996) estimate that an epidemic in ponderosa pine occurred in Colorado every 11 to 20 years during the HRV period. Beal (1943) found that the epidemics often coincided with periods of dry climatic conditions. Return intervals for an individual stand, however, are probably longer, with epidemics occurring once every 50 to 100 years (Johnson 1995, Schmid and Mata 1996). The reported duration of an individual epidemic varies from 2 to 14 years (Schmid and Mata 1996).

Severity of mountain pine beetle epidemics can also vary considerably. In extreme cases, outbreaks result in the mortality of over a million trees in a single year (McCambridge and others 1979). In the Black Hills, historical reports indicate that 50,000 ha of mostly ponderosa pine were killed by a MPB epidemic in 1895 (Shinneman and Baker 1997).

Changes from HRV at low elevations. Douglas-fir has likely evolved with the influence of the insects, as it has with the influence of fire. Fire exclusion during the

past 80 years in areas from New Mexico to Montana appears to be creating a more homogeneous landscape of suppressed low-elevation Douglas-fir forests having increased susceptibility to western spruce budworm outbreaks (Veblen 2000). For example, in the Colorado Front Range, budworm outbreaks were relatively infrequent from the 1880s through the 1920s, when fires were more common, but outbreak severity and extent is thought to have increased since that time (Veblen 2000). On the SNF, however, this does not appear to be the case. Douglas-fir are presently relatively large on the SNF (mean DBH = 36 cm) and they average 160 years in age (McMillin and Allen 2000). While not very susceptible to western spruce budworm epidemics, they are quite susceptible to the Douglas-fir beetle (especially after the 1988 fires injured many live trees). Because large fires and insects are a natural part of the ecosystem, it is probable that the current severity and frequency of insect outbreaks at low-elevations is within the HRV (low confidence). Far less clearcutting, sanitation salvaging, or thinning occurs in Douglas-fir forests at low elevations than in higher forests (SNF RIS database), and thus such practices probably have had a small effect on epidemic frequency.

Like Douglas-fir, limber pine also has likely evolved with the influence of bark beetles, as it has with the influence of fire. On the Bighorn National Forest, outbreaks in ponderosa pine appeared to be within the HRV, and the same probably is true for limber pine on the SNF. Very little limber pine has been harvested on the SNF (D. Eckardt, SNF Silviculturist, personal communication). Therefore, harvesting probably has not caused insect outbreak frequency or severity in that forest type to exceed the HRV (low confidence). The white pine blister rust (see Disease section) appears to be a much larger problem for the limber pine forests.

5.2.3 Insects in aspen woodlands

Several insects defoliate aspen trees, but two of special concern are the forest tent caterpillar (*Malacosoma disstria*) and the large aspen tortrix (*Choristoneura conflicta*) (Allen and Harris 1999). Both can affect large areas over longer periods than most other insects. After the overwintering eggs hatch, the larvae of the tent caterpillar feed on the growing leaves. When population sizes are high, this

caterpillar quickly defoliates entire trees and stands. The trees may produce new leaves (though infestations for several years in a row is lethal). Such epidemics have not been reported for the SNF, but where they occur new aspen groves develop through sprouting from the surviving root system. Without an epidemic, a coniferous forest can develop when young conifers in the understory grow into the canopy. Epidemics occur every 10 to 20 years in Northern New Mexico and Southern Colorado, with a single epidemic lasting for up to 10 years or longer (Hinds 1976).

Similarly, the aspen tortrix has larvae that feed at the time of leaf expansion. They also mine buds and can reach epidemic population sizes for two to three years (Allen and Harris 1999). However, like the tent caterpillar, epidemics of the aspen tortrix are unknown or infrequent on the SNF.

In general, there is no basis to conclude that the characteristics of insect outbreaks in aspen stands are beyond their HRV (moderate confidence).

5.2.4 Insects in non-forest vegetation

Insects causing epidemics on non-forest vegetation have not been reported in any SNF documents, nor have we found information on such events in other parts of the Rocky Mountains that would suggest they are outside the HRV (low confidence).

5.3 DISEASE

Pathogens such as dwarf mistletoe (*Arceuthobium americanum*), comandra blister rust (*Cronartium comandrae*), root diseases (*Armillaria* spp., *Heterobasidion annosum*), and broom rusts (*Melampsorella carophyllaceum* and *Chrysomyxa arctostaphyli*) are all native elements of the SNF (Harris and others 1997). They often have the effect of reducing tree growth in a stand, or even causing localized mortality. Consequently, along with insects, they are commonly viewed as detrimental to "forest health." Despite negative effects on short-term productivity, plant pathogens have been recognized for their positive contributions to biological diversity and various ecosystem processes (Gill and Hawksworth 1964, Hawksworth 1975, Tinnen and others 1982, Dinoor and Eshed 1984, Tinnen 1984, Zimmerman and Laven 1984, Johnson 1995).

5.3.1 Disease in high-elevation forests

Comandra blister rust. A prominent, and often the most damaging, disease in high-elevation forests on the SNF is comandra blister rust, a native fungus that attacks lodgepole pine (Harris and others 1997). This rust frequently occurs with dwarf mistletoe, but it is unknown if either of the two diseases increases susceptibility to the other. In the Ramshorn Area of the Wind River Range, 57% of the lodgepole pines sampled were infected with the rust (Harris and others 1997). Of all the lodgepole pines in the sampled area (306 trees in 53 stands), 21% were dead or dying, apparently due to the rust.

Comandra blister rust requires proximity of its two host plants, lodgepole pine and the herbaceous plant called comandra (*Comandra umbellata*). The spores are spread by wind from comandra (also known as bastard toadflax) to infect pine needles and shoots. The disease generally girdles the stem and kills the top of the tree, which leads to death within a few decades (Geils and Jacobi 1993, Harris and others 1997). Trees with topkill usually are 80 to 90 years old (K. Allen, entomologist, Forest Health Management, U.S. Forest Service, personal communication). The younger trees (up to 80 years) are more susceptible to death than older trees because a small diameter provides a greater chance of the infection developing into stem-girdling cankers (Johnson 1986, Geils and Jacobi 1990). The canopy opens when trees are topkilled, enabling smaller lodgepole pines and other species to grow into the canopy (Peterson 1962, Krebill 1967). Sometimes a lower branch forms a new leader, creating a fork or crook in the trees.

From 1910 to 1945 the incidence levels of comandra blister rust increased dramatically in the Rocky Mountains, apparently due to unusually warm, moist summers (Krebill 1965). In the mid-1900s, Andrew and Harrison (1959) reported that about 25% of the lodgepole pine stands in Wyoming were infected. During the HRV period, the disease probably occurred at relatively low levels when cool, dry conditions prevailed and at higher levels during relatively warm, moist periods, such as the early 1500s and at the present time (Fig. 21).

Dwarf mistletoe. Another widespread disease on the SNF is dwarf mistletoe, a parasitic vascular plant that grows primarily on lodgepole pine, ponderosa pine and limber pine in Wyoming. Several species are involved, but *Arceuthobium americanum* appears to be the most common (Hawksworth and Wiens 1972). The effects of dwarf mistletoe on a host tree include slowed growth, spike tops, witches brooms, dead branches and eventual tree death. It can develop on trees of all sizes, but incidence of the disease is greatest in large trees and it appears to spread most rapidly in uneven-aged landscapes (Kipfmueller and Baker 1998b). While its effects can have negative consequences for silviculture and the enjoyment and safety of forest visitors, mistletoe can also be beneficial for nesting birds and other animals (Hawksworth and Johnson 1989). Some research suggests that bird abundance and diversity is higher in ponderosa pine stands infected with mistletoe (Bennetts and others 1996).

Extensive infestations of dwarf mistletoe have been common on the SNF (Cochrane and others 1988). During a roadside survey on the SNF, Johnson and others (1979) found 64% of the stands surveyed (mostly lodgepole stands) were infected, creating a loss of 698,000 cubic feet in wood volume annually due to slower growth. In the Ramshorn area in the Wind River Range, 37% of the lodgepole pines were infected (Harris and others 1997). Infection rates were highest in lodgepole pine trees found within the lodgepole pine-whitebark pine forest type than in lodgepole pines within the spruce-fir type (types were based on RIS classification of dominant tree species; Table 3). Mistletoe severity ratings for infected trees were moderate to low in the lodgepole and spruce-fir forests, but were high in the whitebark pine forest. Occasionally, whitebark pine was a host for the disease and may be increasing transmission. Whitebark pine has been documented as a host in other areas (Hawksworth and Wiens 1996), although it is also possible the mistletoe infecting whitebark pine is a different species than the one most common on lodgepole pine (Harris and others 1997). Selmants (2000), working in the Medicine Bow National Forest, found that as many as 25% of the lodgepole pine trees in stands developing after clearcutting could become infected within 40 years (see also Selmants and Knight 2003)

Historically, the geographic spread of heavy infection centers was most likely limited by stand-replacing fires. Where such fires kill all infected trees, they essentially eliminate dwarf mistletoe from the stand until re-infection occurs, which can take years to decades depending on the size of the fire (Hawksworth and Wiens 1972). However, the relationship between fire and dwarf mistletoe is complex. Where infected trees survive a fire, the mistletoe can infect new seedlings, creating new infection centers during the early stages of succession.

Mason (1915), in one of the earliest surveys of the Rocky Mountains, described the distribution of severe mistletoe damage as “scattered and localized.” Although precise levels of dwarf mistletoe infection in the 1800s are not known, it probably has been present on the SNF and other areas in Wyoming at all levels of severity for centuries or longer (Kipfmüller and Baker 1998b).

White pine blister rust. This rust, *Cronartium ribicola*, was introduced to western America from Europe around 1910 (Kendall 1998, Harris and others 1999) and is damaging to whitebark pine, limber pine, and other five-needle pines (Harris and others 1997). The disease requires currant or gooseberry (*Ribes* spp.) as an alternate host to complete its life cycle (Krebill and Hoff 1995), and most infections on the SNF are in areas with *Ribes* nearby (Harris and others 1997). Wind-dispersed spores from the shrubs infect pine needles and eventually create cankers that girdle and kill branches and eventually the whole tree. Because pine cones form in the top third of the tree and the rust tends to kill trees from the top down, the tree’s ability to reproduce is lost long before it dies (Kendall 1998). The disease may also predispose white pines to other pathogens or damaging insects (Krebill and Hoff 1995).

The rust had spread into YNP by 1945 and probably into the SNF about the same time. Despite the extremely low resistance of whitebark pine to the disease (Kendall 1998), the incidence of the rust in Wyoming was still low in the 1960s (only 6% of whitebark pine and limber pine were infected). In 1999, white pine blister rust occurred at low (<10%) to moderate (12-15%) incidence levels on the SNF and in eastern YNP (Harris and others 1999). Forty-two percent of sites surveyed in these areas still had no infections in 1999. Only one site in high-elevation forests had an

infection rate of 50% or higher on the SNF—the mixed conifer and whitebark pine stands on Republic Mountain in the north part of the Clarks Fork District.

Incidence levels of white pine blister rust have been lower thus far on the SNF than farther north, where epidemics have been rampant since the 1970s. Nearly half of the whitebark pine trees in the Glacier National Park/Bob Marshall Wilderness complex in northwestern Montana are dead and over 80% of the remaining trees are infected (Kendall and others 1996). More than a third of the cone-bearing crowns of these trees are dead. In YNP, Grand Teton National Park, and National Forests west of the Continental Divide in Wyoming, the disease is spreading (Kendall and others 1996); and it's also now present on limber pine in the Laramie Mountains of southeastern Wyoming and southward into Colorado. The drier conditions on the SNF appear to be keeping the disease from spreading rapidly (Harris and others 1999). However, it is found at low to moderate incidence levels on both whitebark pine and limber pine, and it could become more widespread in the future (Kendall 1998). Where it occurs, some ecosystem variables (such as density, basal area, and cover by this species) can be expected to exceed their HRV (high confidence)

Other diseases. Broom rusts (such as *Chrysomyxa arctostaphyli* and *Melampsorella caryophyllacearum*) infect subalpine fir and spruce and are another pathogen of importance on the SNF (Harris and others 1997). However, this disease and others seem to have minor effects on high-elevation forests. An exception is *Armillaria* spp., which, in combination with the western balsam bark beetle, may cause patch mortality in the fir, as discussed under the section on insects.

While the combined effects of minor diseases and their interaction with large and small disturbance agents (including porcupines, which feed on tree bark) undoubtedly contribute to a variety of ecological processes, very little is known about such interactions. Overall, diseases are important elements of montane and subalpine forests in the SNF. Most stands are likely to have numerous pathogens as concurrent members of the biological community (Lundquist 1993), which helps to maintain biodiversity (Lundquist 1995a).

Changes from HRV. Because diseases in western coniferous forests are very difficult to control, the influence of management on existing conditions is primarily to

enable the diseases to become more abundant, such as through fire suppression or some silvicultural practices. For example, fire suppression increases the amount of older forest on the landscape. Although dwarf mistletoe can spread in young as well as older trees, incidence levels are higher when stands have relatively large trees.

Dwarf mistletoe also increases when landscapes contain uneven-aged stands and small clearcuts or group-selection cuts (Johnson 1995). Such conditions occur on portions of the SNF and, where fire suppression has been effective, more older stands with large trees may be found (only about 5% of current forests considering lower forest flammability than YNP; see YNP vs. SNF in Fig. 22). Tie-hacking and selection harvests, which produce the more susceptible uneven-aged stands, make up only about 1% of the SNF high-elevation forests, but small clearcut patches (<8 ha, Harris and others 1997) were commonly created in the 1980s and 1990s in some areas (Figs. 15 and 18)—though still less than 1% of the forested land on the SNF.

By contrast, other areas on the SNF probably have reduced incidences of dwarf mistletoe due to management. For example, the clearcuts (>8 ha) common in the 1960s (Fig. 17) on the Wind River District may have slowed the spread of mistletoe and the resultant younger stands (only 1% of the forested lands) may now be less susceptible. Of course, large, even-aged stands are common on the landscape even without timber harvest.

Because partial cutting is only on a small portion of the forest and fire suppression probably has not changed the size or frequency of fires beyond the HRV, dwarf mistletoe incidence may be within its HRV at the landscape scale (low confidence). Incidences in local uneven-aged stands, that would have been even-aged without harvest (group or individual selection cuts, especially the tie-hacking area), are probably beyond their HRV at the stand scale (low confidence). Similarly, mistletoe in stands that are older due to fire suppression, or younger due to clearcutting, may be above and below the HRV, respectively (low confidence). However, most forests on the SNF have not been affected by these management practices and should be within the HRV (low confidence).

Comandra blister rust commonly infects stressed trees of any age, and such stress can increase with high tree densities. It is a common perception that fire

suppression has increased tree densities and stress in high-elevation forests, but there is little evidence to support that. Stand-replacing fires dominate the high-elevation forests, and even light fires can kill the thin-barked trees that inhabit those forests (i.e., lodgepole pine, spruce, and subalpine fir). Because fire suppression probably has not increased tree density or stress at high elevations, the severity and distribution of comandra blister rust is probably still within the HRV (low confidence), especially on the landscape scale. The current relatively warm climatic conditions (Fig. 21) probably support the moderate to high levels of infection seen on the SNF.

White pine blister rust infections definitely create conditions outside the HRV, as this fungus never existed in the SNF prior to its introduction in the mid-1900s (high confidence). Given the right climatic conditions (relatively warm periods with higher precipitation), this disease has the potential of decimating whitebark and limber pine over large areas. It appears that only natural or artificial selection for disease-resistant trees will prevent this disease from destroying forests dominated by the five-needle pines (Kendall 1998).

5.3.2 Disease in low-elevation forests

White pine blister rust. Incidence levels of white pine blister rust in low-elevation forests are higher than for high-elevation forests. Nonetheless, most of the low-elevation forests on the SNF are only at low to moderate levels (0 to 15%). It is damaging to a few stands with limber pine, such as near Dead Indian Pass in the northern part of the Forest, where the infection rate is 52%). There, limber pine stands also suffer from dwarf mistletoe, mountain pine beetle, a bark beetle in the genus *Ips*, and possibly a needlecast disease—suggesting various factors may be causing the high mortality found at the Dead Indian Pass site (Harris and others 1999). Mortality and infection rates in limber pine are much higher in Montana and southern Alberta, where over a third of the limber pine trees are dead and 90% of the live trees are infected (Kendall 1998).

Other diseases. There are several other diseases that concern forest managers working at low-elevations on the SNF, including western gall rust, Armillaria root disease (*Armillaria mellea*), red rot (*Dichomitus squalens*), Schweinitzii

root rot (*Phaeolus schweinitzii*) and limb rust (*Peridermium filamentosum*) (Lundquist 1993, Lundquist 1995a). While such diseases can cause localized mortality and tree damage, none of them individually appears to have a great impact on the SNF as a whole. As Lundquist (1995a) noted, it is the combined effects of diseases and other disturbance agents that influence stand structure and composition.

Changes from HRV. With the exception of white pine blister rust, the diseases found in the low-elevation forests of the SNF are native and probably have not been greatly affected by human influences, simply because those influences have been relatively minor over most of this relatively roadless National Forest. Therefore, the native diseases in low-elevation forests currently appear to be within their HRV (moderate confidence). However, the presence in some areas of white pine blister rust probably affects the dynamics of ecosystems with whitebark pine and limber pine in ways that are deviations from the HRV for some variables (high confidence). Further analysis of such effects is warranted.

5.3.3. Disease in aspen and non-forest vegetation

Aspen is susceptible to root diseases, canker-causing fungi, and a heartrotting fungus (*Phellinus tremulae*; Allen and Harris 1999). Cankers form in wounds caused by animals, humans, or other agents, and they can weaken and kill the trees (ramets) (Hinds 1976, Walters and others 1982). Heartrotting fungi colonize the sapwood and produce a fruiting conk. Conversion of younger aspen stands to older stands caused by fire suppression may be increasing the occurrence of this disease beyond the HRV at the stand, landscape, and regional scales (Hinds 1985), because older trees are more susceptible. Recent large fires could help to offset such increases in some areas. Also, human activities in developed recreational areas may increase tree wounding (Johnson 1985), thereby increasing infection rates beyond the HRV. At present, however, we have concluded that diseases in aspen stands are still within their HRV, except possibly near recreation areas (low confidence).

With regard to non-forested vegetation, little is known about diseases or their HRV.

5.4 **WIND**

5.4.1 Wind in high-elevation forests

Wind speeds are highly variable in the Rocky Mountains, with storms that commonly blow down small groups of trees (Veblen 2000) and occasionally topple whole stands of trees over thousands of hectares (Alexander 1964, Veblen and others 1991b). In 1987 a tornado blew down 6000 ha of forest in the Teton Wilderness (Knight 1994a), and in 1997 winds of 200-250 km/hr toppled over 10,000 ha (>24,700 acres) of forest on the western slope of the Routt National Forest in northern Colorado (Baker and others 2001). In YNP, winds blew down patches of 1 to 40 ha (2.5 to 88 acres) in 1984, covering 267 ha (659 acres) total (Despain 1990); and 304 ha (750 acres) were toppled in the Park in the 1970s. On the Bighorn National Forest in northcentral Wyoming, about nine blowdowns occurred between 1958 and 1998, averaging 211 ha (521 acres) for each event. Nearly a quarter of those were partial blowdowns in which many trees were left standing. On the SNF, large blowdowns have been rare or not recorded in the last century. Blowdowns may be more frequent and extensive to the west of the Forest, on the west side of the Continental Divide.

Blowdowns interact with other kinds of disturbances. To illustrate, a dense accumulation of fine fuels is created when windfall occurs, greatly enhancing flammability. Also, as discussed above, the potential for an outbreak of spruce beetles increases. Notably, many trees in a stand are not toppled during blowdowns, producing an effect more similar to a selective harvest than a clearcut (Lindemann and Baker 2001). Older post-fire lodgepole stands appear to be more susceptible to blowdowns than old-growth spruce-fir stands (Veblen 2000), and when blowdowns occur in lodgepole pine stands, succession towards spruce-fir is accelerated.

Recent studies of the Routt-Divide blowdown in northcentral Colorado indicate that various features of the physical environment and the vegetation determine where blowdowns are likely to occur. Physical factors seemed to be most influential, especially wind exposure, elevation, and aspect (Lindemann and Baker, 2002). Unexpectedly, soil permeability, soil water-holding capacity, and distance to natural

edges were not significantly correlated with the blowdown pattern, and the effect of cover type varied with elevation. At lower elevations, aspen forests were less susceptible to blowdown and spruce-fir forests were more susceptible. At higher elevations, closer to the Continental Divide, younger stands were less susceptible than older stands (Baker and others 2001). These results suggest that both topography and vegetation structure can influence the extent and pattern of damage, but that the effects vary with elevation and wind intensity.

In another study of the Routt-Divide blowdown, Lindemann and Baker (2001) found that blowdown patches were larger and had more compact shapes than patches in the surrounding forest (which were derived from past fires, insect epidemics, topographic influences, and possibly previous blowdowns). In contrast, patch shapes of blowdowns on the Bighorn National Forest were long and linear (Meyer and others 2005). Lindemann and Baker also discussed the potential for a spruce beetle outbreak in the vicinity of the blowdown, concluding that it is difficult to predict the spread of possible outbreaks because they could continue for a decade or more and the beetles can fly long distances (300 m or more in Alaska; Werner and Holsten 1997).

Changes from HRV. Blowdown has been a common process associated with timber harvest, particularly after shelterwood (2- and 3-step canopy removal) and group selection cuts or on the edges of new clearcuts (Alexander 1964, Vaillancourt 1995). However, clearcut, shelterwood or group selection harvests have affected only about 2% of the forested landscape at high elevations on the SNF; clearcuts alone have affected only about 1% (2000 SNF RIS database). Consequently, the effects of harvest on blowdown frequency or extent surely must have been small. We can safely conclude that blowdowns, like fire, were an important factor in shaping landscape patterns and stand structure of high-elevation forests during our HRV reference period, and that current blowdown frequencies are within the HRV (high confidence).

5.4.2 Wind in low-elevation forests (including aspen woodlands)

No major blowdowns have been observed in low-elevation forests or aspen groves on the SNF. As at higher elevations, the presence of predisposing factors such as disease, insects, fire, and a shallow water table will affect the size of wind-caused disturbances (Alexander 1986, Lundquist 1995a, Shinneman and Baker 1997). Limber pine often occurs on very windy sites, but its "limber" branches probably are an adaptation to reduce breakage.

Changes from HRV. Because few blowdowns have been documented in the low-elevation forests of the SNF, and in stands of aspen, the differences between existing conditions and the HRV are unknown. However, timber harvest is not common in aspen woodlands and low-elevation forests (<1% of low elevation forests), and current conditions affecting blowdown frequency and extent must be similar to conditions during the HRV period (high confidence).

1. HISTORIC RANGE OF VARIABILITY FOR STAND AND LANDSCAPE STRUCTURE COMPARED TO EXISTING CONDITIONS

Of the numerous human influences on upland vegetation, three have had the most significant impact— fire suppression, timber harvest, and livestock grazing. In this section we focus on the effects of these human activities on variables associated with both stand and landscape structure. The effects of exotic plants are considered also, though very little information is available for this type of disturbance. Our discussion includes our conclusions with regard to whether the variables probably are within or have exceeded the historic range of variability.

6.1 STAND STRUCTURE

Both stand and landscape structure change through time as well as space because of natural and human-caused influences. In this section we focus mainly on stands that have been harvested for timber, assessing probable changes within individual stands rather than comparing all harvested and unharvested stands on the

landscape. Landscape structure is discussed in section 6.2.

As of December 2000, harvested areas on the SNF comprised only about 3% of the forested land (2% of the SNF as a whole). This does not include firewood collection, which is about 25% of the timber cut at the present time, but the land area involved is not known. Because the road system is not extensive on the SNF, the land area affected by firewood gathering is small. Considering the various silvicultural methods, 1.1% of the forests have been harvested selectively, 0.9% have been clearcut, 0.7% was with the shelterwood or seed tree systems, 0.5% was salvage harvesting, and 0.03% was commercial thinning (Fig. 19). Ninety-three percent of the harvests on the SNF have been in the northern part of the Wind River District (Fig. 16).

Of forests classified as suitable for harvest (3.5% of land on SNF, USDA Forest Service 1994a), 32% of suitable lands have been harvested since 1900 (about 37% of timber removed was done prior to the time of classifying stands as suitable or unsuitable and was not on lands currently classified as suitable). Thus, if the current definitions of suitable lands remain in effect, more hectares could be cut in the future (at least 68% of the suitable forests, or 23,400 ha; the number is higher if tie-hacked areas or shelterwood cuts are opened for harvesting again).

The 3% of the forests on the SNF known to have been harvested at least once, of which nearly 25% was for railroad ties, are the focus of our stand structure analysis for high elevation forests. Unharvested high-elevation forests classified as "unsuited for timber harvest" probably have experienced few changes at the stand scale that can be attributed to European-American activities. Although fire suppression can change the proportions of the landscape in different successional stages, and cause higher tree densities at low elevations, research suggests that fire suppression thus far has not greatly affected the internal structure of high-elevation forest stands (i.e., the mean tree density at high elevations has not exceeded the HRV relative to stand age) (Turner and others 2003). The importance of stand-level changes across the landscape is discussed in section 6.2. In evaluating how management activities might have caused some variables to exceed their HRV, we focus on the effects of timber harvesting at high elevations and the effects of fire

suppression at low elevations (where very little harvesting has occurred). We also consider the effects of livestock grazing and exotic plants, but much less information is available on these topics. Only about 14% of the SNF is considered as suitable for livestock grazing.

6.1.1 Stand structure in high-elevation forests

Some effects of timber harvesting are similar to those of fire, insects, disease and wind storms, but others are different. For example, canopy gaps of varying sizes are created after nearly all disturbances. In contrast, silvicultural treatments are designed partially, if not entirely, to manipulate stand structure in a way that favors the growth of trees. Notably, natural processes involved with forest development and succession do not lead to maximizing the rate of tree growth, or even total forest productivity (tree growth plus understory plant production). Instead, plant adaptations facilitate persistence and the production of seed rather than growth rates.

Consequently, our evaluation of many of the variables discussed below (e.g., canopy gaps) is based on the recognition of such fundamental differences between natural processes and silvicultural treatments. Our focus in this section is on the following stand structure variables that are potentially affected by human-caused disturbances: tree density, regeneration time, seedling and sapling density, percent canopy cover, the density of canopy gaps, the density and cover of understory plants, plant species diversity, the distribution of trees in a series of age classes and size classes, forest floor depth, the amount of exposed mineral soil, the density of snags, and the amount of coarse woody debris. Small-scale patches in the canopy of one to several hundred square meters, such as those created by the death of one or a few trees, are considered an aspect of stand structure. Larger gaps of one to several ha are considered when discussing landscape structure (section 6.2).

As noted, there are essentially no data on stand structure for the HRV period (1600-1860). HRV analyses are still possible, but they must be based on a qualitative review of what is known about how current stand structure changes during succession and from one place to another, along with our impressions of what the forests must have been like prior to the arrival of European-Americans—based on the

journals of early explorers and information about the biology of the dominant species. This analysis cannot be site specific, but the information we provide could be useful when developing site-specific management plans.

Based on available information, and the nature of disturbances in the mid- to high-elevation forests of the SNF, we draw the following conclusions about HRV deviations for variables pertaining to stand structure in forests dominated by lodgepole pine, Engelmann spruce, and subalpine fir:

Tree density. The naturally occurring number of trees per hectare is highly variable through space and time, ranging from a few hundred mature trees in open stands to over 15,000/ha in 100-year-old doghair stands of lodgepole pine. Various studies have shown that high lodgepole pine seedling densities are partially a function of fire intensity and the number of serotinous trees in a stand (Franklin and Laven 1989; Anderson and Romme 1991; Turner and others 1997, 2003). Ten years following the 1988 fires in YNP, tree seedling density ranged from essentially no seedlings to extremely high densities (sometimes exceeding 100/m²) in different stands across the landscape. With succession, densely stocked stands may be thinned by natural mortality, but many doghair stands persist for well over 100 years—probably until the next stand-replacing disturbance.

Tree density also is quite variable on the SNF. In the Wind River Range, lodgepole stands tend to be dense with small trees on the relatively infertile granitic soils there, whereas forests are sparser in the Absarokas (K. Houston, SNF soils scientist, personal communication). When thinning is done for enhancing tree growth, the goal is to have approximately 750 trees per hectare (D. Eckardt, SNF Silviculturist, personal communication). It seems clear that timber harvesting and associated thinning, in that small amount of area where it occurs, results in stand densities that are within the HRV, and that fire suppression is unlikely to affect significantly the HRV of tree density at high elevations (high confidence).

Regeneration time and seedling/sapling density. Concerning forest regeneration, some characteristics of clearcuts are similar to conditions created by large fires, such as an open environment for seedling establishment. Lodgepole pine has proven to be well adapted for establishment and regeneration in the patches

created by both fire and harvesting. In some areas, Engelmann spruce and subalpine fir are also pioneer species (Doyle and others 1998). On most sites with lodgepole pine present before the disturbance, subsequent regeneration by this species has been very good (especially at <2,900 m (9,512 ft) elevation). In fact, lodgepole pine often regenerates so prolifically after a clearcut that precommercial thinning has been recommended after about 20 years (Grasso and others 1981, Geils and Jacobi 1991).

Notably, seedling establishment can be slower above 2900 m (9,512 ft) (Stahelin 1943), partially because high-elevation clearcuts often are subjected to high winds that blow snow into downwind forests. Consequently, new seedlings and saplings often die, apparently because of the lack of snow protection from desiccation and high-intensity solar radiation during the winter. From the late 1940s to the present, some clearcuts on the SNF have been in high-elevation spruce-fir forests (see RIS database and Fig. 34), where the development of new forests is slow (regardless of the type of disturbance).

Snags and CWD provide micro-environments favorable for snow accumulation, along with shade and less wind, which favors new tree establishment. However, too much slash also can delay regeneration and fuel fires that are hot enough to burn the seed source for natural regeneration. Nonetheless, very hot fires did occur in YNP in 1988 that burned a significant amount of seed (Turner and others 1997), and these hot fires are not likely attributable to fire suppression. Therefore, conditions that create hot fires capable of burning the seed bank are within the HRV (high confidence).

In general for lodgepole forests, there is no evidence to suggest that seedling and sapling density is beyond the HRV in managed stands (high confidence); but on soils at high elevations where conditions are more stressful, certain types of slash management (leaving low amounts of coarse woody debris) could lower mean seedling densities from the mean for this variable during the HRV period. However, as already noted, regeneration is often slow at high elevations (D. Eckardt, SNF Silviculturist, personal communication), whether after fire or harvest. Because of the high natural variation in regeneration and seedling density from site to site (Turner

and others 1997), the presently variable conditions probably are within the HRV (high confidence).

Percent canopy cover and rate of gap formation. When stands are thinned naturally, the canopy is opened up less than is usually done for silvicultural purposes. Surface fires kill some of the trees in small groups, as do insects and diseases, but the dead trees (snags) remain in the forest. Consequently, thinning and selective harvesting probably push mean percent canopy cover below the HRV in forests of comparable age and site conditions, where timber harvesting occurs (moderate confidence). Such effects may last no more than about 20 years, as the canopies of the remaining trees normally expand to fill the openings or new trees grow into the gaps from below. Group selection harvests would create larger gaps (0.1 to 0.4 ha), which would take longer to fill.

Shelterwood cutting in two- or three-steps also produces canopy cover patterns that are not within the HRV (moderate confidence). This system removes about one-third of the trees at intervals of about 10 to 30 years, producing large gaps that create temporary uneven-aged forests. Once the overstory is completely removed (after 10 to 30 years, possibly longer), the stand becomes even-aged. Some of the gaps from shelterwood cuts may mimic partial blowdowns or insect outbreaks for short periods, but the process of removing part and then all of the top canopy every 30 years or so creates gaps faster than during the HRV period because insect epidemics or fire typically have a within-stand return time that is longer by 20 years or more than shelterwood harvesting return time.

In summary, though timber harvesting can cause canopy cover to drop below its HRV, so little selective or shelterwood harvesting or thinning has occurred on the SNF that this variable probably still is within the HRV on all stands except where timber harvesting is intensive (Fig. 16) (moderate confidence). As noted, fire suppression probably does not affect significantly the canopy cover of stands at high elevations.

Density of canopy gaps. Trees are gradually dying in healthy, natural forests, creating canopy gaps or openings in the forest canopy. Also, if a sufficiently large group of neighboring trees die, perhaps 10-20, a root gap is created as well as a

canopy gap (Parsons and others 1994), thereby reducing competition for water and nutrients. Both canopy and root gaps enhance the growth of the remaining trees as well as understory plants (including saplings). Indirectly, canopy and root gaps affect age-class structure, size-class structure, species diversity, and the abundance of some kinds of animals and other organisms.

Silvicultural systems that include group selection and thinning can create densities of canopy and root gaps that are similar to those found in unmanaged forests, and individual selection harvests create small gaps that are similar to natural gaps caused by the death of a single tree or several neighboring trees—though standing dead trees remain after natural disturbances, as noted previously. Shelterwood systems create larger, temporary gaps. Overall, the average number of gaps in a managed forest is likely to be higher than the HRV (moderate confidence), because of the motivation of silviculturists to enable most remaining trees to grow more rapidly.

Density and cover of understory plants. If managed forests have lower canopy cover or a higher density of canopy gaps than unmanaged forests, then the density and cover of understory plants usually will be higher. However, timber harvesting can leave an abundance of slash that could suppress some understory plants, as could surface soil compaction or other changes caused by the machinery that might be used. Species composition can be affected by slash treatments that disrupt the soil profile, which reduces the sprouting capacity of some understory plants and can be favorable for the invasion of new species (Selmants 2000, Selmants and Knight 2003), especially those with wind-borne propagules. Some exotic plants invade both managed and burned forests, but they apparently do not persist in 30- to 50-yr-old post-harvest stands on the Medicine Bow National Forest (Selmants and Knight 2003). The same is probably true for the SNF, though exotic plants often are common in road ditches there. In the Medicine Bow National Forest, exotic plants have not yet invaded the adjacent forests in large numbers (Dillon 1998), which also seems to be the case on the SNF. Exotic plant species did not increase in extent or abundance after the 1988 fires in YNP, perhaps because of rapid sprouting from

roots and other belowground parts by native understory plants that were not killed by the fires (Turner and others 2003).

Given that the density and cover of understory plants is highly variable in natural stands as well as in managed stands, there is no indication, to date, that timber harvesting, in the small area where it occurs, would push variables associated with the understory vegetation beyond the HRV (moderate confidence). Some plant species may be more sensitive to timber harvesting than to fire, wind storms, or other natural disturbances, and, of course, road building associated with harvesting decimates the native vegetation in the road corridor and provides habitat for introduced, weedy species. Grazing probably has not yet affected the understory plants significantly because, unlike some meadows and shrublands, high-elevation spruce-fir or lodgepole pine forests on the SNF are not heavily grazed (J. Hicks, SNF range staff, personal communication). Sheep often entered high-elevation timbered areas in the late 1800s (Cochrane and others 1988), but it is difficult to know about their long-term effects on the forest understory vegetation.

Plant species diversity. All of the tree species present in the 1700s and 1800s are still common on the SNF today. High-grading, which occurred earlier in the 20th century during the tie-hacking era, may leave an abundance of trees with less desirable wood characteristics, such as subalpine fir, or other species with malformed or small trees. More recently the practice of clearcutting may have increased the abundance of lodgepole pine in some areas, though fire could have had the same effect. Nevertheless, within-stand tree species diversity is variable and probably within the HRV (moderate confidence), particularly since no new tree species have been introduced.

Some plant species may be rare and easily extirpated from an area, but evidence of this happening over a large area is not now available. Some plants are interior forest species that have been extirpated locally because of depth-of-edge effects associated with patch cutting (fragmentation), such as various species of orchids and herbaceous ericads (Dillon 1998, Baker and Dillon 2000). The cover of dwarf huckleberry declines in proportion to the intensity of slash treatment, but it tends to persist in post-harvest stands and it can gradually increase in abundance in

some areas, such as in the Medicine Bow National Forest (Selmants 2000, Selmants and Knight 2003). Understory vegetation of post-harvest stands dominated by lodgepole pine at lower elevations appears to recover more rapidly than in stands of spruce and subalpine fir at higher elevations (Selmants and Knight 2003). Where timber harvesting has occurred in the past, the species that are present today are the species that can tolerate timber harvesting practices. Additional study is required to determine which species cannot.

Notably, the species diversity in some areas may now be higher because of the invasion of exotic plants. However, while the presence of such species increases the number of species, one potential effect of exotics is to reduce the abundance of native species. Insufficient research has been done to determine the net effect of introduced species on species diversity and other variables in forests, but many scientists believe they are considerable.

In general, there are few introduced plants in forests at high-elevations in Wyoming, and a very small proportion of the SNF has been subjected to harvesting. Significant changes could have occurred locally, especially in areas of severe disturbance and where exotic plants or other organisms have become abundant. Where that is the case, a variety of ecosystem variables could have exceeded their HRV.

Age- and size-class structure of stands. Because of periodic fires and other disturbances, a full range of age- and size-class structures must have existed within individual stands during the HRV period. Stand structure would have ranged from even-aged stands of lodgepole pine, with a variety of ages, to uneven-aged stands that had not burned for centuries and were dominated primarily by Engelmann spruce and subalpine fir, with possibly some lodgepole pine and aspen. The fires would have killed many of the big trees from time to time, and they probably burned over large areas (possibly over 25% of the SNF; 44% of YNP was subjected to varying degrees of fire in 1988; Balling and others 1992). Consequently, tree size would have ranged from uniformly small trees after the stand was recently burned to very large trees coexisting with small trees when the stand had not been disturbed for long periods.

Since the late 1800s, both human-caused fire and timber harvest have converted areas with large, old trees to stands of younger, smaller trees, with the important exception that timber harvesting could extend into areas that had not burned for centuries (such as on leeward slopes or in ravines, both of which are thought to burn less frequently). In such protected areas, the age- and size-class structure would exceed the HRV (moderate confidence) if harvesting does not allow sufficient time for old, large trees to replace those that were cut. Across the rest of the landscape (i.e., where disturbances were more common historically), the variety of age and size-class structures with and without clearcutting, at the stand level, is probably within the HRV because both clearcutting and intense fires would create the kind of size-class structure characteristic of even-aged stands (moderate confidence). Similarly, insect epidemics, partial blowdowns and partial cutting could create a forest structure characteristic of uneven-aged stands.

Forest floor depth. High-elevation forests in the Rocky Mountains are characterized by an abrupt boundary between the mineral soil and the forest floor, which is comprised mostly of fine and coarse litter in varying stages of decomposition. The abrupt boundary is caused by periodic fires that burn the forest floor and expose much of the mineral soil. A new forest floor gradually accumulates because of the slow rate of decomposition (Fahey 1983), but the litter does not become well incorporated into the mineral soil because of the absence of earthworms. Just as there is considerable variation in the ages of the forests due to natural disturbances, there is considerable variation in forest floor depth in a stand.

In addition to time since disturbance, the variation in forest floor depth is caused partially by the fact that, on slopes, litter often drifts downslope due to gravity and flowing water, which creates deeper forest floors on toe slopes than nearer the ridge tops. Moreover, fallen logs often accumulate detritus on their upper side, and squirrels create middens of cone scales under some trees. Usually the forest floor of unmanaged stands is 2-6 cm thick, but the range of forest floor depth during the HRV period is much broader, ranging over time from essentially no forest floor to a forest floor that can be several decimeters thick (Maxwell and Ward 1976).

Silviculturalists often recommend that much of the forest floor be scarified or removed, thereby creating a better seed bed for new tree establishment (Alexander 1966, 1987); or they tend to suppress fires, which minimizes disturbance to the forest floor, thereby allowing for a continuation of forest floor thickening until the next fire that cannot be controlled. Under a variety of such treatments, harvested forests most likely still have forest floor depths that are within the HRV (high confidence).

Mineral soil exposed, disrupted, or compacted. In general, the forest floor is important for minimizing soil erosion because it increases the rate of infiltration during periods of snowmelt and rainfall. Less surface runoff leads to less soil erosion. However, exposed mineral soil is thought to improve the chances of successful establishment for lodgepole pine and spruce seedlings (Alexander 1987). The machinery used for yarding and slash treatments, such as piling and burning, roller chopping and tractor walking, scarify the soil in ways that favor the establishment of seedlings, similar to a natural fire (which can expose the mineral soil just as a timber harvesting operation can). Such scarification occurred on the SNF after 1960 to 1975 clearcuts in the Wind River District near Dubois (K. Houston, SNF soil scientist, personal communication).

A notable difference between mineral soil exposed by fire compared to scarification practices is the depth to which the mineral soil under the forest floor is disrupted (Bockheim and others 1975, Halpern and Franklin 1990, Dion 1998). Scarification by a bulldozer blade or the blades of a roller chopper disrupt soil structure to a greater depth than does a fire, which often burns the litter without breaking up the mineral soil (though some erosion may occur). In the process of mechanical scarification, the surface roots of surviving trees and understory plants may be broken, which leads to a higher mortality of plants than would be expected for plants that survive a fire (Selmants and Knight 2003). An example of a plant in which scarification may slow recovery is dwarf huckleberry. In contrast, though fires typically burn plant shoots severely, the soil temperature typically is not hot enough to kill the roots (except under patches where fuels have accumulated). Thus, most plants other than conifers survive fires because they are capable of sprouting from underground tissues that are not affected.

Other soil changes, such as soil displacement (including ruts on temporary roads) and compaction, can occur after logging to greater extents than after fire. In a study of two clearcuts on the Bighorn National Forest, 25 to 52% of the soil was displaced (duff and top soil removed), mostly from machine piling of slash (Nesser 1990); and on one of the clearcuts, 16% of the soil was compacted (a loss of >10% porosity compared to an undisturbed site), mostly under skid trails. Because caterpillar tractors with wide tracks are commonly used on the SNF, compaction may not be a widespread problem there (D. Eckardt, SNF Silviculturist, personal communication). Logging can cause soil displacement, but natural erosion processes, particularly in areas with large debris flow systems, probably far exceed erosion caused by timber or road activity. Overall, soil displacement probably is still within the HRV (low confidence).

In sum, mechanized yarding, slash treatment, and scarification can create a disturbance that rarely occurs following a fire, with the consequence that, in the small area where such practices have occurred on the SNF, the HRV for surface soil disruption is exceeded (high confidence). For most of the forest stands at high elevation, however, soil erosion, displacement and compaction appear to be within the HRV (high confidence).

Snag density. The importance of dead trees as a component of wildlife habitat is widely acknowledged (Hutto and others 1992, Hutto 1995, Bull and others 1997, Parks and others 1997, Hagan and Grove 1999). Reflecting this fact, SNF management prescriptions call for leaving 6 to 10 snags per 10 acres (~4 ha) that are at least 20 to 25 cm DBH (20 cm for aspen and lodgepole pine; 25 cm for Douglas-fir, ponderosa pine, spruce and subalpine fir). Nonetheless, in areas that have been subjected to clearcutting (only 1% of the forested land on the SNF), there are far fewer snags than prior to the arrival of European-Americans (Harris 1999).

Every natural disturbance, with the possible exception of wind, creates dead standing trees. For example, a fire typically leaves a forest of snags. In contrast, past clearcuts left few, if any. Another fire can burn through a stand not long after a previous one (called a reburn), removing some of the standing or fallen snags (Lotan and others 1985) and reducing snags to low densities. However, our impression

from the literature is that such reburns are uncommon because many young forests tend to be relatively inflammable due to the removal of so much of the fuels by the previous fire. Also, removal of large, living trees during timber harvest leads to a decline in the number of snags in the future. Early photographs in unlogged areas often show an abundance of snags, giving the “salt and pepper” effect in black and white photos. Native Americans did not remove many snags during the HRV period because they probably used standing dead trees infrequently, especially those of large size.

In sum, with the advent of thinning, partial cutting, stand improvement, firewood gathering, salvage harvesting and wood harvesting in general, the rate at which old snags are replaced with new snags has been reduced at the stand level in stands with these activities. Because most snags fall after 25 years (Lotan and others 1985), the absolute range of conditions within an unmanaged stand for actual snag density would be zero to abundant—which clearly is within the range of historical conditions (high confidence). However, using our definition of HRV (range of means), the mean snag density in post-harvest stands would be lower than the range of means for snag densities during the HRV reference period (high confidence).

Coarse woody debris. As noted, natural disturbances do not remove bolewood from a forest. Even an intensive fire leaves most of the wood in the form of snags, and most of it becomes coarse woody debris within two decades (Lotan and others 1985). A reburn can occur, but there is still considerable coarse woody debris remaining on the forest floor (Tinker and Knight 2000). After a century or more, the downed wood becomes incorporated into the surface soil (Graham and others 1994), giving the impression that there is very little coarse woody debris, but the residual of log decay is still present under the forest floor. Raking reveals the decomposed logs. New coarse woody debris is added as the larger trees die and fall, one by one during stand development or in large numbers during a windstorm or after the next fire.

Thus, harvesting is fundamentally different from any kind of natural disturbance and produces a forest that is beyond the HRV for dead standing trees (snags), coarse woody debris, and probably the ecosystem processes associated with structural features dependent on big pieces of wood (high confidence).

Moreover, variation from natural disturbances has resulted from some machine yarding, where whole-trees are brought to the landing before delimiting. This results in a concentration of slash and residual coarse woody debris on landings. Following natural disturbances, this material would be distributed more evenly throughout the stand. Salvage operations after blowdowns or fires also reduce the amount of coarse woody debris.

The consequences of deviations from the HRV for coarse woody debris are still poorly understood, but both standing and downed tree boles provide important habitat for some species of fungi and a variety of insects, all of which can be important sources of food for vertebrates (Harmon and others 1986, Hagan and Grove 1999); and Turner and others (2003) found that the labyrinth of fallen trees after the 1988 fires provided sufficient protection from elk browsing to enable some aspen seedlings to grow into saplings. Coarse woody debris is also known to be important for diversifying the structure of streams, and the organic compounds derived from decomposing wood undoubtedly influence the underlying soils.

One detailed study has been done on the temporal and spatial dynamics of coarse woody debris in Wyoming forests. Tinker and Knight (2001) worked in the Medicine Bow National Forest, but the results are relevant to the SNF. They found a wide range of coarse woody debris cover and mass on harvested stands (11.4-26.8 m²/ha or 15.3-35.9 Mg/ha). Using a simulation model, Tinker and Knight concluded that such amounts provide lower coarse woody debris after several clearcuts than after the same number of fires on comparable sites, supporting our conclusion that mean coarse woody debris amounts in harvested stands are lower than the HRV. They also found that the time required for coarse woody debris to cover 100% of the ground surface area after a fire was much less than after clearcutting (Fig. 35). Only when slash amounts left on the ground were at least double the average amounts left using current management practices was the time required similar for a clearcut and fire (in other words, 200% of current coarse woody debris amounts in slash could keep practices within the HRV on some sites). Coarse woody debris can be found elsewhere on the landscape, such as on lands that have not been harvested, but the

coarse woody debris dynamics of individual stands—the focus of this section—is changed beyond the HRV by harvesting (high confidence).

Could a changing climate have altered snag and coarse woody debris dynamics from the HRV reference period? Information available for the climate of the reference period suggests that it was somewhat cooler, sometimes drier and sometimes more moist. When moist conditions favored tree growth and density, there may have been more snags and coarse woody debris during the reference period than now. However, such conditions also would have favored decomposition, causing a more rapid loss rate for both snags and coarse woody debris. For now, we believe that the safest conclusion is that the change in climate would not have had a significant net effect on snag and coarse woody debris abundance. Further research is required to verify this conclusion.

In summary for high-elevation forest stands, the HRV for many variables appears to be quite broad and many stand-scale variables on the SNF remain within what we believe is their HRV. Variables that are possibly beyond their HRV in high-elevation stands, and only in those subjected to timber harvesting—less than 4% — include canopy cover and gaps, mineral soil disrupted, snag density, and the amount of coarse woody debris.

6.1.2 Stand structure in low-elevation forests

Only 1% of the timber harvested on the SNF has been in low-elevation forests, and 99% of that has been Douglas-fir. As in other National Forests in the region, the low elevation forests as a whole have changed more due to fire suppression, livestock grazing, and the introduction of exotic plants than to timber harvesting. In this section, we address the changes in stand structure that have occurred in low-elevation forests since the late 1800s, and whether or not various stand variables are beyond their HRV. Here we focus on all low-elevation forests, not just those that have been harvested, so our approach in this section is different than for high-elevation forests.

Tree and Sapling Density. Tree density is the most frequently addressed variable in comparisons of early and modern conditions in low-elevation Rocky Mountain forests. As discussed in section 5.1.2, studies in northwestern Wyoming (Houston 1973, Gruell 1980), the Southwest (Covington and Moore 1994) and the Front Range of Colorado (Veblen and Lorenz 1991) suggest that the stands would have been more open prior to fire suppression by European-Americans. Reconstructions from fire scars and other sources indicate that such stands often were maintained for hundreds of years by frequent, low intensity surface fires (White 1985, Covington and Moore 1994, and many others). The thick bark of Douglas-fir and ponderosa pine would appear to be adaptations for frequent surface fires. In more mesic Douglas-fir stands, however, mixed-severity fire regimes may have been more typical, with a combination of surface and stand-replacing fires (Littell 2002). In such stands, canopy cover can range from complete closure to very open (Bradley and others 1992).

Notably, while low-elevation forests often were described as open with low tree density, "dense" stands (with 50 to 80% canopy cover) were described in the 1800s by several explorers in the Black Hills area (Dodge 1876, Newton and Jenny 1880, Graves 1899). Such stands probably occurred on more mesic sites, but they could have been characterized by infrequent, stand-replacing fires (Shinneman and Baker 1997). Working in the ponderosa pine forests of the Black Hills, McAdams (1995) estimated that pre-European tree densities ranged widely, from 15 to 1600 trees/ha. Similarly, Ehle and Baker (2003) reported historic ponderosa pine densities of 70 to 3,000 trees/ha for Rocky Mountain National Park in northern Colorado. Whether or not this research applies to low-elevation Douglas-fir forests in the SNF is yet to be determined, but we believe the studies provide a frame of reference from which to work.

Whether dominated by Douglas-fir, ponderosa pine or limber pine, some areas previously dominated by open stands are now doghair stands with several thousand trees per hectare (Covington and Moore 1994, Knight 1994a, Knight 1994b, Savage and others 1996), most likely because of fire suppression and the elimination of Native American fires. Historic photos indicate that many stands of Douglas-fir or

ponderosa pine were very open (Houston 1973; Progulsk 1974; Gruell 1983b; Veblen and Lorenz 1986, 1991; Johnson 1987) and that low-elevation tree density in many Rocky Mountain areas has increased greatly since the beginning of successful fire suppression. Selective cutting can create more open stands again, but this has occurred on only about 41 ha on the SNF. Most of the harvests, though few in number and area, have been shelterwood cuts (245 ha).

Livestock grazing also may have affected low elevation forest structure, but probably only to a small extent in modern times (J. Hicks, SNF range staff, personal communication). A comparison of historic and modern photographs of a pine savanna in northcentral Wyoming that was grazed in the past (but not currently) shows an increase in tree density and canopy cover (Wyoming State Historical Society 1976). The heavy livestock grazing in the late 1800s and early 1900s could have facilitated an increase in tree density in low elevation forests by removing sufficient herbaceous vegetation to reduce competition for young tree seedlings. However, insufficient information is available at this time to distinguish an increase in tree density due to fire suppression from an increase due to grazing. Large numbers of cattle and sheep (or elk and deer) could cause damage to seedlings and saplings, which would reduce tree density.

Based on the studies that have been done thus far, our conclusion is that both surface and crown fires could have occurred at low elevations on the SNF, and those areas that had frequent surface fires are now developing tree densities beyond the HRV because of fire suppression (high confidence). The stands on the SNF that would have escaped frequent fires during the reference period probably are still within the HRV for tree density (high confidence).

Plant species diversity and understory plant composition and cover. No data are available to suggest that the plant species diversity of low-elevation forests is either within or beyond the HRV, whether for trees or understory plants. Dramatic shifts in tree species composition (and tree species diversity) surely have not occurred on the SNF (high confidence). As trees and saplings become more dense, canopy cover increases and there is a concomitant reduction in the amount of understory cover (Moir 1966, McPherson 1992). Also, shade-tolerant species

become more abundant as canopy cover increases. In areas where tree density has increased due to management activities, or where harvesting has not been used to thin the trees, it seems highly probable that the species composition of the understory vegetation could have changed and the average amount of understory vegetation would have declined. On average, such changes probably are beyond the range of means during the HRV period (low confidence). As discussed for high-elevation forests in section 6.1.1, the presence of exotic plants could cause further changes in the understory vegetation. Such plants are more likely to become a problem at low elevations than higher in the mountains (Selmants and Knight 2003).

Unfortunately, changes in herbaceous species composition due to domestic grazing in low-elevation forests are difficult to detect by comparing historic to present-day photographs, which makes the determination of their HRV essentially impossible. Indeed, livestock grazing was so widespread for so many years that the long-term effects are difficult to identify, even for trained observers. Because of livestock grazing, some understory plant species may be quite rare today, overall cover may be less, and some soils degraded by livestock may not have been restored (based on studies in the Central Rockies by Thilenius 1975, Turner and Paulsen 1976, and Fleischner 1994).

Size- and age-class structure of trees. An effect of harvesting and fire suppression has been to reduce the natural variability in stand structure and age distribution caused by historic disturbances. The increased tree and sapling density resulting from fire suppression prevents most trees from reaching large sizes and reduces the stand average. Similar to tree density, the average size- and age-structure of unharvested low-elevation stands probably have been reduced below the HRV due to fire suppression (high confidence).

The partial cutting of the few stands of Douglas-fir that have been harvested on the SNF tends to skew age-structure toward young trees and does not mimic fire effects. Partial cutting may or may not mimic insect outbreaks, depending on the insect. For example, the western spruce budworm tends to have the opposite effect as partial cutting—skewing age-structure toward larger, older trees because it attacks the small trees in a stand. In contrast, the Douglas-fir beetle has effects similar to

partial cutting, killing some or all of the larger trees. Because there has been very little harvesting at low elevations, and because the most common insect damage is caused by the Douglas-fir beetle, we conclude that age-structure at low elevations is within the HRV (low confidence).

Forest floor depth. The short return interval of surface fires in low-elevation forests would probably keep the depth of the forest floor from becoming as deep as in high-elevation forests, where the fire return interval was much longer. Fire suppression probably is causing the development of deeper layers of litter than normally would be expected, pushing this variable beyond the HRV (high confidence), thereby increasing flammability (along with increased tree density and more continuous fuels). Because frequent surface fires typically keep the forest floor litter amounts low, harvesting in such forests may be further increasing the forest floor depth beyond the HRV if much of the slash is left in place.

Snags and coarse woody debris. Lundquist (1995a) studied the diversity of snags and coarse woody debris in low elevation forests in the Black Hills. He identified ten classes of snags and five classes of logs (mostly ponderosa pine), finding that different types of disturbance create different types of snags and logs. For example, the stems of beetle-killed trees tend to break, leaving a relatively short snag, while trees killed by root rot are commonly uprooted. Based on such studies, timber harvest, in the small amount of area where it occurs at low elevations on the SNF, probably tends to 1) reduce the number and diversity of snag types and 2) distribute log types among the five classes more evenly on the forest floor (Lundquist 1995a, 1995b). This evidence, combined with old photographs in Wyoming that show a larger number of dead trees in low-elevation forests than occurs today, suggests that snags and downed wood are now lower than the HRV (moderate confidence) in the managed Douglas-fir woodlands, due to harvesting (including casual fire-wood cutting). Notably, some coarse woody debris is burned by fires in low-elevation forests because of frequent surface fire, possibly more than observed by Tinker and Knight (2000) in high-elevation forests where fires are less frequent. Thus, with regard to coarse woody debris, low-elevation forests where timber

harvesting has occurred may not be as far from the HRV as are high-elevation forests.

Considering that most of the low-elevation forests on the SNF have not been harvested but have been subjected to fire suppression, the bigger effect of management on snags and coarse woody debris may be less frequent fire. Fire tends to create snags, but insect and disease epidemics do the same. In the absence of fire (or harvesting), do pathogens become more abundant? That is a contention of some who worry about forest health problems in western landscapes. However, we doubt that mortality rates are higher or lower than the HRV (moderate confidence).

With regard to coarse woody debris, periodic surface fires could consume downed and decaying wood on the forest floor. With less frequent fires, this wood could persist for a longer time. Moreover, if wood inputs remain constant, then coarse woody debris may increase. We suspect, however, that fire suppression has not occurred frequently enough to raise the amount of coarse woody debris to above the HRV because of slowed fire consumption (moderate confidence). This opinion is based partly on the considerable longevity of the trees that dominate the low-elevation landscapes (Douglas-fir and limber pine) and the fact that pathogenic problems are still rather localized where these species occur. A notable exception may be near Dead Indian Pass, where the exotic white pine blister rust has caused what appears to be an unusually high level of limber pine mortality.

In summary, a combination of human influences, and possibly climate change, have caused many structural characteristics of low-elevation forest stands to exceed their HRV in other parts of the Rocky Mountains. The same probably has occurred in the SNF. Great attention has been given to this situation, often in the context of "forest health" discussions. Low-elevation forests on the SNF cover a relatively small proportion of the terrain (19%) and provide a rather small amount of the wood that is harvested (less than 1% of the low-elevation forests have been subjected to harvesting). However, the effects of fire suppression probably have been more widespread at low elevations. Also, the effects of the exotic white pine blister rust are

most evident there. Attention should be given to the potential adverse consequences of the unusual conditions now found in limber pine forests, where the disease occurs.

6.1.3 Stand structure in aspen woodlands

Following a disturbance, aspen often becomes more abundant because of root sprouting—though seedlings can be abundant from time to time in some areas, such as in some parts of YNP after the 1988 fires (Romme and others 1995). With fire suppression and natural succession, conifers are now becoming co-dominants with aspen in some areas, to the point that managers have begun cutting the conifers to maintain this highly valued species in the landscape (SNF RIS database; J. Hicks, SNF range staff, and J. Emmerick, Wyoming Game and Fish biologist, personal communication; Bartos 2001, Campbell and Bartos 2001). In other areas, stands of aspen have not been invaded by conifers. Both kinds of stands probably occurred on the SNF during the HRV reference period, but the average number of conifers in aspen stands now is probably higher than in the past in some low-elevation areas because of fire suppression, where aspen is found. Nevertheless, the change is probably not large enough to cause changes that exceed the HRV, because historically many stands had encroaching conifers (low confidence).

The addition of conifers can change the understory because plant growth under aspen is typically more abundant than in coniferous forests, partially because of the relatively open canopies of aspen stands that allow light to reach the forest floor. Notably, such forests support a high diversity of birds (Merrill and others 1993) relative to other areas. Loose and Anderson (1995) found that woodpecker nests are more common in aspen forests than in other forest types, particularly in large aspen that are >18 cm DBH and have heart rot or dead tops (average DBH 26.7 cm; range 18-45 cm), and Struempf (1999) found higher nest survival in aspen than in conifer forests. Good forage and ground cover for a variety of animals is provided, and aspen contributes significantly to the aesthetic quality of an area wherever it occurs. Nevertheless, we expect that the changes in the understory vegetation from the increase in conifer cover probably is not enough to push understory variables beyond their HRV (low confidence).

Variables pertaining to aspen stand structure probably have not exceeded the HRV because there has been very little cutting of aspen on the SNF (about 115 ha, SNF RIS database). When the bolewood is left on site after felling, some aspects of fire disturbance are mimicked. When bolewood is removed, however, coarse woody debris and snags could be reduced to below the HRV for young and mature stands. Large changes in snags and coarse woody debris probably have not occurred on the SNF because of small amount of aspen that has been harvested.

Rather than fire suppression, the most widespread influence of humans on the understory of aspen stands may be through grazing and the introduction of non-native plants, both of which could cause changes in the understory plant species composition. Numerous exotic plants, including musk thistle (*Carduus nutans*), Canada thistle (*Cirsium canadensis*), houndstongue (*Cynoglossum officinale*), oxeye daisy (*Chrysanthemum leucanthemum*), Kentucky bluegrass (*Poa pratensis*), and various species of knapweed (*Centaurea* spp.), are becoming increasingly more abundant in some areas, causing shifts in native species composition (unpublished report, Fremont Country Weed and Pest Control District; and K. Houston, SNF soil scientist, personal communication). The addition of non-native species in the understory changes the condition to beyond the HRV (high confidence). The recent field guide by Houston and others (2001) for identifying invasive plants facilitates the solution to this sometimes staggering problem, by helping one find and remove these plants.

It is less clear if grazing by elk and/or livestock is pushing aspen stand variables beyond the HRV on the SNF. Elk grazing (and sometimes livestock) is extensive in some aspen stands on the SNF, such as on both sides of Carter Mountain. A few stands cut for regeneration have been able to develop into new stands only because they were protected from grazing with fences (J. Hicks, SNF range staff, personal communication). The loss of areas dominated or co-dominated by aspen is discussed in Section 6.2.2 on landscape change. For changes in stand structure, grazing may be slowing tree regeneration and reducing the typically lush understory structure in some areas. However, where the understory species composition is still mostly native, we suspect that this variable is within the HRV

because understory composition is highly variable, depending in part on the highly variable overstory composition (low confidence).

In summary, because of very little timber harvesting, and the fact that some stands probably would have changed to coniferous forest through succession during the HRV reference period, it can be argued that the variables pertaining to many modern day aspen stands are within their HRV on the SNF. However, we are hesitant to draw this conclusion because of the effects of long-term grazing in these forage-rich habitats and the widespread presence of introduced plants. Stand variables pertaining to the trees probably are within their HRV, but variables pertaining to the understory vegetation in areas with exotic plants probably are not (high confidence).

6.1.4 Stand structure in non-forest vegetation

Exotic plants also are becoming more widespread in non-forested areas (i.e., grasslands, meadows and shrubland; K. Houston, SNF soil scientist, personal communication). The invasive species include spotted knapweed (*Centaurea maculosa*), Alsike's vetch, Dalmation toadflax (*Linaria genistifolia* ssp. *dalmatica*), and a variety of non-native grasses, including cheatgrass (unpublished report, Fremont Country Weed and Pest Control District; and K. Houston, SNF soil scientist, personal communication). Some exotic plants have been treated with herbicides, which affects some native species as well. Smooth brome (*Bromus inermis*) is the most extensive introduced plant in non-forested areas, having been introduced in grass mixes as part of road rehabilitation programs (J. Hicks, SNF range staff, personal communication). Kentucky bluegrass and common timothy (*Phleum pratense*) are found in a few places, but are less common. In grasslands and shrublands with exotic species, both species composition and diversity are outside the HRV (high confidence), as would the dynamics of some still poorly understood ecosystem processes.

Although 51% of the SNF is in commercial allotments for grazing, only about 14% was classified as capable and suitable rangelands for livestock in 1996 (USDA

Forest Service 1996). Most of the riparian and upland areas in the 36 allotments that are used are considered in fair (61-63%) or better condition (32-34%), with isolated cases of deteriorated range (5-6%) caused by excessive grazing by livestock, wildlife, or both. Canada thistle (*Cirsium arvense*) and shrubby cinquefoil are management problems in some grazing allotments, but only Canada thistle is introduced. Shrubby cinquefoil increases with overgrazing in relatively wet areas and its abundance may be beyond the HRV in some places.

Both the composition and structure of non-forest cover types are also affected by grazing. However, herbivory has been a natural process in non-forest vegetation. Before European-Americans arrived, bison and other herbivores grazed the montane grasslands in Wyoming (Dorn 1986), and many of the plants dominating the different kinds of non-forest vegetation evolved adaptations for such herbivory (Mack and Thompson 1982).

Since then, however, the characteristics of herbivory have been altered by the introduction of domestic livestock. In the late 1800s and early 1900s, grazing by sheep and cattle was intense and poorly controlled—probably exceeding the levels of ungulate herbivory during the HRV period (moderate confidence). However, because population estimates of herbivores for the 1600 to 1860 HRV period are not available, confident comparisons of herbivory levels at that time and today are impossible. Working in the Bighorn National Forest, Jack (1900) documented and photographed the damage from livestock grazing in the late 1800s. Comparisons of these early photos to photos taken in 1975-76 suggest that the initial heavy grazing caused forbs to replace grasses, particularly at high-elevations. Stream downcutting and less coverage by willows and sedges in riparian areas were apparent. Based on the 1900 photographs, it appears topsoil erosion was accelerated by the grazing. Similarly, some areas apparently have not fully recovered from past heavy grazing on the SNF (USDA Forest Service 1986).

Regionally, the effects of domestic livestock grazing on non-forest areas include the following: 1) the creation of patches from grazing, trampling, and nutrient input (urine and feces); 2) increases in soil erosion where more than 50% of the herbaceous cover is removed (Currie 1975); 3) declines in some native species of

bunchgrass that are sensitive to grazing (e.g., Johnson 1956, Madany and West 1983); 4) increases in the abundance of grazing-resistant species; 5) the introduction of exotic species such as crested wheatgrass (*Agropyron cristatum*) and smooth brome; and, 6) perhaps most significantly in terms of stand structure, a reduction in grasses and an increase in tree seedlings (as observed in Utah by Madany and West 1983). Additionally, because of the value of forage for livestock, grassland fires were suppressed whenever possible, which sometimes led to an increase in shrub and tree cover.

Typically, an increase of needle-and-thread grass (*Stipa comata*), junegrass (*Koeleria cristata*), and Sandberg's bluegrass (*Poa sandbergii*) has occurred with livestock grazing pressure on the SNF, while bluebunch wheatgrass and western wheatgrass (*Pascopyrum smithii*) have decreased, particularly in the watershed of the South Fork of the Shoshone River (K. Houston, SNF soil scientist, personal communication). These species shifts and the low vegetation cover in areas that were heavily grazed may be beyond the HRV (moderate confidence). However, because such intense levels of grazing are no longer allowed, some of the vegetation probably is recovering.

The number of livestock permittees has declined during the last several decades on the SNF, partially because of the concern over the increased predation pressure from grizzly bears and wolves (J. Hicks, SNF range staff, personal communication). Therefore, most of the rangelands are moving toward (or are in) satisfactory condition (USDA Forest Service 1996). In other areas in Wyoming, livestock grazing has caused loss of ground cover, loss of litter, increased soil compaction, and changes to stand structure and species composition, including increases in non-native species. Wild herbivores may have also caused such damage, but, unlike domestic stock, wildlife are less likely to concentrate for long periods within specific areas (van Vuren 1982) and they cause less damage to riparian areas because they have a more dispersed grazing behavior than livestock usually do. Also, livestock are more concentrated because they are fenced. Since the late 1960s and early 1970s, rotation grazing (modified deferred) has been

implemented on the SNF to reduce impacts (J. Hicks, SNF range staff, personal communication).

Another management activity that may be changing the species composition beyond the HRV is controlled burning in non-forest areas. Such burning is commonly done in the spring (K. Houston, SNF soil scientist, personal communication) rather than during the more typical late summer or early fall. When the timing of such disturbances changes, some species (e.g., grasses) are favored over others, possibly changing the species composition in the burned area if the practice continues.

Overall, exotic species today, and the intense grazing levels on some parts of the SNF in the early 1900s, have pushed the structure and species composition of some of the grasslands, forblands, and shrublands beyond their HRV (high confidence). Prescribed fire or chaining (and to a limited extent, chopping and range pitting) are sometimes used to reduce the abundance of big sagebrush (K. Houston, SNF soil scientist, personal communication), which might help to restore HRV conditions when that seems desirable, but there is now the potential of exotic plants growing where such control practices are attempted.

Our discussion is relatively brief for non-forest vegetation types because much less is known about them. In contrast to forests, it is difficult if not impossible to learn about their disturbance history (no annual rings, and old photographs reveal less information), and many ecosystem variables are more difficult to measure than in forests. This is due in part to the fact that, unlike forests, half or more of the biomass is belowground. Moreover, successional changes are more difficult to interpret than in forests. It seems clear that the ongoing trend of more careful management of livestock and wildlife will restore at least some variables to within their HRV, if indeed that has not happened already. The presence of exotic plants creates significant challenges. Monitoring various soil variables and variables associated with plant species composition, including the abundance of exotic plants, should help ensure that important values are not lost due to inappropriate human activities.

6.2 LANDSCAPE STRUCTURE

The spatial and temporal variability of some stand variables, with and without European-American influence, can also be considered at the landscape scale. In addition, large scale disturbances cause natural landscape changes that are obvious from aerial photographs and satellite images. Changes in landscape patterns are known to influence the movement of animals, water, and nutrients, as well as the spread of disturbances.

In this section, we first address stand structure variables at the landscape scale in relation to the HRV (specifically, tree density, canopy cover, number and rate of formation of gaps, understory plant cover, age structure, forest floor depth, mineral soil disruption, snag density and coarse woody debris). Then we discuss variables specific to the landscape scale, namely, the number and proportion of different vegetation types, the ratio of forest to non-forest, the proportion in different successional stages, the amount of land in edge and interior forest habitat, patchiness and patch shape, the proportion in high-density classes for snags and coarse woody debris, and disturbance return interval.

As with our stand-level analysis, this section is divided into high and low-elevation landscapes. Aspen and non-forest vegetation are discussed in the context of these more general landscape categories.

6.2.1 High-elevation landscape structure

The high-elevation, forested landscapes encountered by early explorers in the early 1800s in northwestern Wyoming must have had large areas of even-aged cohorts of trees (stands) that originated following large crown fires. However, the landscapes probably were patchy due to uneven burning (Foster and others 1998). Working in YNP, Turner and others (2003) found that most areas subjected to crown fires were within 50-200 m (164-656 ft) of unburned or lightly burned areas, which would be sources of propagules. Large fires would burn for several weeks or months, with greater intensity some days than others. Winds would shift from time to time, causing the flames to move in different directions. Spotting would occur, and typically the flames would move more slowly downhill. More mesic areas were less

likely to burn, such as in ravines or on north slopes. Uneven-aged stands of older trees, including old-growth forests, would have occurred where, by chance, there had not been a stand-replacing fire for several centuries (Romme and Knight 1981).

Large, infrequent fires cause forest fragmentation just as timber harvesting can. Landscape fragmentation is a complex phenomenon (Buskirk and others 2000) that has received considerable attention. For this analysis we define it as an increase in the number of abrupt transitions on a landscape between forest and non-forest or young-forest (whether caused by humans, fires or wind) that reduce the area of interior forest and increase the area affected by edges. Notably, the forests over much of the SNF have always been patchy because of the many abrupt transitions caused by environmental changes associated with the rugged topography.

To evaluate changes in stand variables across the landscape due to management, the scale of management activities must first be considered. We focus on timber harvesting and fire suppression as the major management activities that could move stand structure variables averaged across the landscape beyond the HRV. As noted previously, the HRV for a variable generally becomes narrower as the size of the land area increases (Fig. 1). To illustrate, 10 years following the 1988 fires in YNP, tree seedling density is broad at the stand level, ranging from essentially no seedlings to very high densities in different stands (Turner and others 1997). However, at the landscape scale, information on this spatial variability is lost because the stand-level values for a variable are averaged across all stands on the landscape. We define HRV at the landscape scale as the variability of the entire landscape through time, not space, which conceptually is the range of variable means averaged over 30 to 50 year periods through time.

Change from the HRV of landscape variables. The magnitude of the percentage of the landscape affected by management activities must be evaluated to assess the change from the HRV of landscape variables. On high elevation landscapes, the percentage of SNF forests that have been subjected to some form of harvesting (as of 2000) is ~4%. Specifically, in 2000 the land area in clearcuts comprised 1.2% of the total forested area at high elevation; shelterwood cuts, 1.0%; selection cuts (including most tie-hacking), 1.6%; and salvage cuts, 0.7% (SNF RIS

database) (Fig. 16). The forest area that had been thinned (commercial and pre-commercial) was about 0.6%. Under existing conditions, tree densities still fall within the moderately broad HRV because new tree establishment on the few areas that were harvested is usually quite good. Tree, sapling, and seedling density in harvested landscapes probably falls within the fairly broad HRV (moderate confidence).

Canopy gap density and cover at the scale of the entire SNF are probably still within the HRV because such a small proportion of the forested landscape on the SNF has been thinned or selectively cut (~2%; moderate confidence). Understory plant density and cover, forest floor depth, soil displacement, and plant diversity averaged across the landscape probably are not outside the HRV, either (moderate confidence). Stand age structure also is likely within the HRV, as a variety of ages are well-represented (see section that follows on proportion in successional stages). At the District scale, conclusions are slightly different for canopy cover. The Wind River District comprises about 20% of the entire forest, yet most of the timber harvest (about 93%) has occurred there. Within this District alone, canopy gap density and cover are probably below the HRV due to the relatively high amounts of timber harvest (moderate confidence; over 20% of Wind River landscape affected). However, 20 years or so after harvest, such variables return to their HRV.

Mean snag density averaged over the landscape may be lowered by management as burned areas have been salvaged, snags removed for firewood, and bolewood removed on 4% of the forests, preventing future development of many snags in harvested areas. Quantification of areas with heavy firewood gathering is required to estimate the percentage of stands on the National Forest that have snag density below the HRV, but it is probably relatively low given the low density of roads in the forest. Within the Wind River District, mean snag density is probably below the HRV (moderate confidence). On the scale of the SNF, the changes from management probably are not enough to cause snag density to drop below the HRV. The same is probably true of coarse woody debris abundance. Wood has been removed from the landscape and not allowed to decay on site, as occurred for millennia prior to the arrival of European-Americans, but the proportion of land area

where this has occurred is too small to reduce abundance below the HRV for the entire SNF (high confidence). Snag density on the Wind River District, where harvesting has affected over 20% of the landscape, probably would have exceeded the HRV. The same would be true for other variables associated with logging, such as mineral soil disruption (moderate confidence).

Number and proportion of land cover types at high elevations. The proportion of the landscape in different vegetation cover types can give additional insights into landscape changes through time. Just as Romme (1982) found for YNP, all major land-cover types present in modern times on the SNF were also present during the HRV period. The number of land cover types (landscape richness) probably has not changed beyond the HRV (moderate confidence).

Some evidence suggests that the lodgepole pine forest type in Wyoming (such as on the Bighorn National Forest) has become less abundant in the last century while the spruce-fir type has increased because of the maturing of the forest after heavy forest fires produced abundant lodgepole pine in the late 1800s (Fig. 36). Fire suppression also could be increasing the extent of forests dominated by Engelmann spruce and subalpine fir. We would expect the opposite to occur with harvesting, where an increase in lodgepole pine is expected. About 2.2% of the forested SNF landscape has been clearcut or cut using the shelterwood system, leading to more lodgepole pine. The warmer conditions in the latter part of this century may also be increasing lodgepole pine, relative to spruce-fir. This could have been counteracted by fire suppression, which could have facilitated the development of more of the spruce-fir type over possibly 5% of the high-elevation forested landscape, but fire suppression efforts often have not been effective on the SNF. Overall, variability in cover types due to climatic change is probably greater than variability due to minor changes caused by management. The proportions of different cover types are naturally dynamic and probably are within the HRV (moderate confidence).

Forest/non-forest ratio at high elevations. Because of forest encroachment into meadows and shrublands, and natural reforestation following fire and timber harvesting, the proportion of the SNF in forested vegetation may have been increasing since the late 1800s, as it has on the Bighorn National Forest (by 9% on

the Bighorn National Forest, Fig. 36). The percentage of the entire SNF that was forested in the early 1980s was reported to be only 40% (Berntsen and others 1983), and currently it is just above 50% (52% if sparse forests are not included; 64% if they are included). Because most of the SNF is at high elevation, the percentages may be similar for high-elevation landscapes. The current variability of forest/non-forest percentages across the landscape from one Ecological Subsection of the SNF (as defined by National Hierarchy of Ecological Units, Cleland and others 1997, see Fig. 8) to another is high, from 22 to 90%—a range of 68%. Much of this spatial variability is due to distribution patterns discussed previously in section 4, including topographic and edaphic patterns, but fire suppression may be narrowing this range, as is occurring in other areas nearby (Gallant and others 2003). Due to the variability of local climatic conditions over the past 500 years, however, the present ratio of forest to non-forest land on the SNF (64% forest, 36% non-forest) is most likely still within the HRV on high-elevation landscapes (high confidence).

Proportion of forests in different successional stages and old-growth at high elevations. The vegetation mosaic of Rocky Mountain landscapes is known to vary greatly through time, primarily because of large-scale fires and other natural disturbances. In YNP, Romme and Knight (1981) found that the amount of land area in forests of early, middle, and late successional stages varied temporally and that it was unlikely that the subalpine forests of YNP as a whole are in a "shifting mosaic steady-state." In the mid- and late-1700s, young forests dominated more land area; in the mid-1900s, old forests were more common in the same area (Romme and Knight 1981, Romme 1982; Fig. 37). Old-growth forests (>200-300 years old) are important ecologically because they provide considerable large wood on the ground, a relatively large number of snags, and habitat that seems to be required for some species of plants and animals.

A similar dynamic change in successional stages must have occurred during the HRV period at high-elevations in the SNF, but the amount of old-growth forest today probably is within the HRV except in those few areas where timber harvesting has been intensive (e.g., Wind River District, moderate confidence). To evaluate this quantitatively, the proportion of the forested landscape at high elevations in different

age classes in the SNF in the northern part of the Park was divided into successional stages and compared to the HRV in YNP (Table 6, and Fig. 37). Using FIA data, most of the successional stages fell within or close to the HRV, when the HRV was defined as the absolute range, but many were outside the HRV when compared to a historic range of means (Fig. 37). The SNF appears to naturally have low numbers of stands with trees averaging >300 years old because the present low amounts (2%) cannot be attributed to timber harvesting (only 3% of forests have been harvested) or firewood cutting (unquantified but probably not more than a few percent). When the oldest stage was defined more liberally as >200 years old (old-growth), all stages were within the HRV at the scale of the entire Forest (Fig. 38). Old-growth proportions in the Wind River District must be determined to judge if such proportions are still within the HRV there, where most of the timber harvesting has occurred. When the little-harvested northern part of the Forest was evaluated using a different database, the grizzly bear database, all the stages fell within the HRV of YNP (Fig. 37) except LP1 (18, 2, 32, and 32% for LP0, LP1, LP2, and LP3 + SF on the SNF, respectively). The low LP1 value (40-100 year old trees) may be a result of fire suppression or inaccuracies in the database (see Appendix). Overall, for the SNF as a whole, we conclude that the proportions in different successional stages in high elevation (subalpine) environments are within the HRV (moderate confidence).

Proportion of landscape with low canopy cover at high elevations. Thinning or partial cutting of stands may raise the proportion of low-canopy-cover, mid-successional forest on the landscape (<40% canopy cover) beyond the HRV. Such forests are present on the SNF landscape (6% of landscape; Fig. 39), but mostly in areas with Douglas-fir, which are at low elevations and which tend to grow with more open canopies. Only about 2.1% of the landscape has been affected by thinning or selective cutting. Thus, it is unlikely that the proportion of the forests in low canopy cover is beyond the HRV (high confidence). Low canopy cover is not common in the Wind River District, indicating that even where timber harvest has been most common, the HRV has not been exceeded.

Proportion of landscape with high density classes for snags and coarse woody debris at high elevations. Snag and coarse woody debris density is often highest in recently burned forests (Tinker 1999) and in old-growth forests (Mehl 1992). If the abundance of such forests falls below the HRV, then numbers of snags and the amount of coarse woody debris on the landscape (at the whole Forest scale) is probably below the HRV also. Fire suppression has reduced the area of recently burned forests by possibly 5% and timber harvesting has reduced the area of old-growth forests by perhaps 1-2% (a significant proportion of the land area that has been harvested). Such changes are probably not large enough to shift snag and coarse woody debris densities below the HRV for the entire SNF, although they probably were below the HRV for the Wind River District (moderate confidence).

Edge, interior forest habitat, patch shape, and patchiness at high elevations. The spatial configuration of the landscape patches has been changed due to timber harvesting, but only over about 1% of the SNF. Whereas fires produce many small patches and a few large patches, clearcuts produce a few small patches and many mid-size (4 – 40 ha) patches (Figs. 32 and 33). Hence, as opposed to the large, contiguous, even-aged cohorts that would have developed after fires during the HRV period, clearcut harvesting has tended to produce smaller patches at any given time (mostly dominated by lodgepole pine; Fig. 32), which are dispersed throughout the high-elevation, forested landscape. In this sense, clearcuts “perforate” areas of interior forest (Forman 1995) and do not perpetuate large areas of even-aged forest, as would have occurred with fire (Figs. 15 and 16).

Also, although edges were certainly created between areas of different burn intensities, human-created edges (around clearcuts and along roads) usually are more abrupt. In the 1970s, the creation of edges by clearcutting was encouraged to increase “game” species that prefer edges, such as elk and deer, but the effects of such edges on plants and animals that require interior forest conditions or security cover can be detrimental (Murcia 1995, Beauvais 1997). Fragmentation apparently has not affected bird species diversity significantly in at least some parts of Wyoming (Merrill and others 1993), but the loss of interior-dependent wildlife species is of regional concern (Beauvais 2000). For these reasons, many investigators have

argued that timber harvesting and other human activities that cause the fragmentation of high-elevation forests in Wyoming (Tinker and others 1998, Baker 1994; Reed and others 1996a, 1996b) and elsewhere in the Rocky Mountains (R.L. Knight and others 2000) are of concern. Roads and clearcuts can reduce the core area of patches (interior area of patch with edge depth of 50 m), increase edge and edge convolution, reduce patch size, and increase patch diversity on the landscape (Tinker and others 1998).

Using a map of recorded fires on the SNF in Arc/Info, we compared a number of landscape patterns that include clearcuts (Fig. 16) on the SNF to the more natural patterns produced by fires since the 1930s (Fig. 26), to determine if such effects were occurring on the SNF. We used the program FRAGSTATS (McGarigal and Marks 1995) to quantify the patterns of the patches created by the disturbances. We found one area (the Wind River District) that had more edge per ha (e.g., 24 m/ha) due to clearcuts than ever occurred with fire in any Ecological Section (Fig. 8) of the SNF since the 1930s (maximum 4 m/ha), but most Sections fell within the natural fire patterns for edge density (Fig. 40). However, clearcuts created more patches than fires, even though they covered less area on the forest than fires. On average, clearcuts were smaller, were closer together, had less complex (convoluted) shapes, and less variation in patch size (Figs. 18 and 40). If roads had been included, the difference would probably be even greater. Our conclusion is that patch variables are beyond the HRV in areas where clearcuts and roads are common (moderate confidence), though such areas comprise a small proportion of the SNF.

Thus, the few watersheds with heavy cutting, mostly on the Wind River District, have more homogeneous patch sizes than before, and the level of patchiness probably has exceeded the HRV for those watersheds (moderate confidence), assuming high-elevation fire in the last half century is somewhat similar to the HRV, which seems reasonable. Patchiness averaged across the entire landscape of the SNF is within the HRV (moderate confidence)

Additionally, shelterwood cuts are not equivalent to natural disturbances because all the bolewood is removed in stages within 10 to 30 years. Possibly, some partial blowdown areas will have complete blowdown or insect attack that destroys

the rest of the trees in the future, but in general, the 2 to 3-step shelterwood system is a new kind of disturbance on the landscape because it occurs in stages. The staged patterns that are created are beyond the HRV, but only on the Wind River District (high confidence). Also, the numerous meadows and shrublands in the SNF give the landscape a rather high amount of natural edge and relatively low amount of interior forest. The additional fragmentation by clearcuts and shelterwood cuts on the Wind River District may further reduce the amount of already limited interior forests.

Compared to fire, timber harvesting has affected a much smaller area on the SNF, but harvesting every year in similar patch sizes creates an unnatural pattern on the landscape where it occurs. For example, Tinker and others (2003) compared landscape pattern dynamics during the last several centuries in the subalpine forests of YNP to comparable forests on the east side of Targhee National Forest, adjacent to the Park, where timber harvesting has been common (largely by clearcutting). They found that none of the landscape metrics calculated for the Targhee National Forest fell within the HRV for subalpine forest landscape structure on YNP.

For the Medicine Bow National Forest in southern Wyoming, Tinker and Baker (2000) found that if practices observed from 1985-1993 continue over time, the spatially variable landscape patterns created by fire would cease to exist. In their place would be a patchwork of more or less equal-sized patches with staggered age distributions. All age classes would continue to be represented on the landscape, but patches of the same age class would become small and isolated rather than large and contiguous. Tinker and Baker concluded that future landscape structure will have little resemblance to conditions in the 1700s and 1800s if the practices observed from 1985-1993 continue (which included the rather unusual patch-cutting associated with the Coon Creek Project on the Medicine Bow National Forest).

Other studies of landscape patterns created by clearcut harvesting and road building have reached the same conclusion (Reed and others 1996a, 1996b; Miller and others 1996; Tinker and others 1998). Such studies make assumptions that timber harvesting will continue with the same approach as in the past on all suitable lands. This assumption may be unrealistic because size and amount of area

harvested have changed over time (Figs. 17 and 18), but the studies show that fragmentation has increased steadily.

Disturbance return intervals and rate at which new patches are formed at high elevations. Another aspect of silvicultural practices that can be compared to natural disturbances at the scale of landscapes is the return interval for successive disturbances at a site. The 1986 SNF Management Plan (pages III-59 to III-62) specifies the rotation age for clearcutting in subalpine/montane landscapes at 50-140 years for lodgepole pine, 100 to 180 years in spruce-fir, 80-120 years for aspen, and 70+ years for other cover types. For two- and three-step shelterwood cuts, the rotation ages are 50-180 years for stands dominated by Engelmann spruce and subalpine fir, 90-140 years for lodgepole pine, and 70+ for other cover types. If a mean return interval for stand-replacing fires of 150-300 years (spruce-fir typically would be on the long-end of the range) is assumed for individual stands during the HRV (Arno 1980, Romme and Knight 1981, Romme and Despain 1989), then the silvicultural rotation ages in general tend to be somewhat shorter, depending on the fire-return interval at specific sites.

The timing of thinning and partial harvests also is different from natural disturbance frequency. For example, both the mountain pine beetle and spruce beetle typically affect stands that are at least 70 to 80 years old. In contrast, precommercial thinning, which sometimes has the same effect as beetles in reducing competition and increasing growth of surviving trees, is often prescribed for 20 or 35 year old stands in the SNF. Because shelterwood cutting involves two or three cuts per rotation, the landscape disturbance frequency is much higher than with natural processes, especially with the concomitant road-building and maintenance.

The result of the above changes in the site-specific return interval for disturbances is a change, where timber harvesting occurs, in the rate at which new patches are formed across the landscape. The shorter return times result in the increased patchiness on the landscape that is occurring in some heavily-cut areas (e.g., in the Wind River District). Thus, in the small amount of area where timber harvesting occurs, it is likely that the rate at which new landscape patches are formed

now exceeds the HRV (high confidence). Across the landscape of the entire forest, the disturbance rate is still within the HRV (high confidence).

In summary, our general conclusion for landscape structure at high-elevations is that European-Americans have left a significant imprint in the Wind River District but not on the other districts. Deviations from the HRV in such landscapes appear to be driven mostly by the introduction of timber harvest and roads as additional disturbance agents. A moderate level of human-caused fragmentation is apparent on the Wind River District. However, other districts have a high-elevation landscape structure that probably is within the HRV.

6.2.2 Low-elevation landscape structure (including aspen and non-forest vegetation)

Low elevation landscape patterns are very different than the high-elevation patterns on the SNF, and fire suppression has impacted the low-elevation landscape far more than timber harvesting (less than 1% of the low-elevation forests have been harvested; 2000 RIS database). As in the previous section, we first examine stand structure variables averaged at the landscape scale; then we discuss variables that apply to landscape mosaics.

Forest stand structure at the landscape scale. During the HRV period, the low-elevation landscapes of the SNF probably contained grasslands, shrublands, savannas, and relatively open woodlands as well as small tracts of forests. There probably were always some stands that escaped the surface fires, resulting in high tree and sapling density, low understory densities, and fewer and smaller canopy gaps than in neighboring stands that burned more frequently (Romme and others 2000). When forested stands are averaged with the savanna or woodlands, and projected through time during the HRV period, tree and sapling density on low-elevation landscapes probably ranged from moderately low to moderate. With the frequently open tree canopy, the density of grasses, forbs and shrubs would have ranged from moderately high to high. Forest floor depth did not accumulate across most of the landscape, due to the frequent fires, and the HRV probably remained

fairly narrow at low values. Average size and age of trees was probably moderate to moderately high. Unfortunately, it is impossible to quantify variables such as these at the present time.

The HRV of species diversity at the stand level was probably somewhat broad, but when averaged over the landscape, the HRV of this diversity was more likely narrow, not changing much through time because of the great diversity of environments on the SNF. In contrast, the HRV of snag density must have been considerable across the landscape, from zero in mid-successional stands to abundant after fires. When averaged, however, snag density probably was moderately low to moderately high during the HRV period. There always would have been some decaying wood on the ground.

Where fire suppression has been effective, tree and sapling densities and canopy cover averaged across the landscape probably would be above the HRV; canopy cover, gaps and understory vegetation density would be below the HRV; stand age younger than the HRV; and more stands would be even-aged as more stand-replacing fires occur due to the currently high tree densities (moderate confidence). Species diversity (including non-forests) probably has not changed, but forest floor depth probably has increased because of less frequent fires (moderate confidence). Snag density could be below the HRV where fires have been suppressed, as surface fires create some snags, but this trend is probably counter-balanced because more of the trees are susceptible to disease as they age. Although less wood is burned today, not enough time has passed to have increased coarse woody debris density in low elevation forests (low confidence). In the future, with continued fire suppression and more disease, snags and coarse woody debris could accumulate to higher levels than the HRV (low confidence). Where firewood cutting has been extensive at low elevations (currently not quantified), the abundance of snags and coarse woody debris could be below the HRV. Spatial distribution may now be different from the HRV at low elevations, with more continuous forests instead of the tree clumping that could have characterized some areas during the HRV period (low confidence).

Patch sizes and configuration at low elevations. As mentioned previously, the low-elevation landscapes during the HRV period probably were composed of distinct and separate clumps of relatively even-aged stands of Douglas-fir (but with some variation in ages within the clumps), in which new cohorts of tree establishment across the landscape were linked to climatic oscillations (Kerr 1988, Swetnam 1990, Savage 1991, Savage and others 1996). Such landscapes probably resulted from major fires creating openings of various sizes, as well as leaving lightly burned or unburned areas (Kaufmann and others 2000). Today, due to fire suppression, such areas may be more homogeneous, with continuous dense forests that are beyond the HRV for the landscape (low confidence).

In low-elevation Douglas-fir forests on the SNF, clearcutting practices have been almost non-existent (only 10.1 ha cut). Rather, shelterwood cuts have been more common. These cuts can increase mid-sized patches (4-40 ha) over and above the HRV under natural fire regimes (Fig. 32), just as they do in high elevation-areas. Timber harvesting affected a very small percentage of the forest, and thus it would not affect the HRV on the National Forest scale. But, where the harvests occurred, the distribution pattern of such perforated landscapes is probably beyond the HRV (moderate confidence).

Proportion of landscape dominated by old trees at low elevations. Because only a small percentage of the Douglas-fir has been harvested (1%), timber harvest has probably not yet reduced the percentage of the low-elevation landscape in old trees below the HRV (moderate confidence). So little harvesting has occurred at low elevations that it probably has not caused an old-growth decline (16% of Douglas-fir are currently old-growth >200 years old and 3.5% are >300 years old). Old-growth forests certainly could be lost in the future if suitable lands are harvested at the planned rotation rate. Timber harvest has reduced the number of old trees in other Rocky Mountain low-elevation landscapes. For example, on an unlogged landscape of ponderosa pine and Douglas-fir in Colorado, about one-third of the sampled tree patches contained old trees (>300 years), whereas a nearby logged landscape had only one-fifth of the patches with old trees (Kaufmann and others 2000). Notably, when old trees were defined as >200 years rather than >300 years in that study, the

old-growth (which comprised half the patches) did not differ between logged and unlogged landscapes. There is not yet widespread agreement on the definition of old-growth, and which definition should be used for HRV studies.

The harvested landscape used for the Colorado study also had more dense trees and fewer openings than the unharvested one. The oldest and largest trees had been cut, leaving a more dense forest with smaller trees. As noted, if harvest becomes more extensive in low-elevation forests on the SNF (14% of Douglas-fir stands are now considered suitable), the planned timber harvest rotation of 50+ to 180 years (USDA Forest Service 1986) would cause the proportion of old trees in that area to fall below the HRV.

Proportion of landscape with high-densities of snags and coarse woody debris at low elevations. Areas with extensive cutting for firewood or lumber would have reduced snag and coarse woody debris across the landscape. Cutting for lumber (at least since the turn of the century) has not yet been extensive at low elevations on the SNF, but the extent of firewood cutting is unknown. Therefore, without such information, we cannot conclude that harvesting, for whatever purpose, has reduced the percentage of the landscape with high density snags and coarse woody debris below the HRV (low confidence).

Proportion of landscape with different land cover types at low elevations. Historically, low-elevation forests were interspersed on the landscape with grasslands, shrublands, and savannas, and, as noted previously, surface fires probably occurred more frequently than they do today (Romme and others 2000). Where trees have invaded grasslands, or have become more dense in savannas, the amount of land area in grasslands and savannas probably has been reduced below the HRV (low confidence). Our low confidence in making this statement for the SNF is due to the fact that tree distribution at low elevations may be controlled more by soils and topography than the periodicity of fires.

Another species in low-elevation forests, limber pine (2% of the entire SNF), is not usually harvested for wood (only 24 ha with this species have been harvested on the SNF). Because fire in limber pine communities is relatively infrequent (Bradley and others 1992), human activities such as fire suppression have probably not

altered limber pine forest. Thus far, there is no evidence that limber pine distribution and density have changed significantly from the HRV because of human activity, although that could change with the introduction of the non-native and spreading white pine blister rust. More extensive surveys of the incidence of this disease are needed to evaluate changes occurring in the limber pine forests.

While aspen stands do not occupy a large area on the SNF (<1%), they contribute significantly to the overall diversity of the landscape. It is difficult to say whether the proportion of the landscape in aspen has declined below the HRV, as aspen has declined on some National Forests (Shepperd and others 2001). On the Bighorn National Forest, aspen was not extensive in the 1930s nor in the 1990s (<1%). Yet, on the Targhee National Forest in Idaho, where it is more abundant, aspen stand size and number have declined substantially since 1920, apparently due to fire exclusion (Gallant and others 2003, Parmenter and others 2003). On the nearby Bridger-Teton National Forest, a photographic analysis of changes between 1878-93 and 1968-72 also shows a decline in aspen (Gruell 1980). It is possible that aspen has declined beyond the HRV on the SNF, but aspen may not have been as extensive on the drier SNF than the National Forests to the west, on the wetter side of the Continental Divide. In the Greater Yellowstone Ecosystem, aspen is relatively rare (1.4% of region) and aspen loss rates average only 10% (Brown and others in press) which still may be within the HRV.

Declines in aspen cover have been attributed to not just fire suppression, but also excessive browsing by large populations of ungulates—especially elk (DeByle 1979, Kay 1990, 1993, Kay and Wagner 1996, Baker and others 1997, Smith and others 2001) but secondarily domestic livestock. This research has been done primarily in the Greater Yellowstone Area and in Rocky Mountain National Park where ungulate populations have been high in recent years. Heavy browsing and grazing are causing declines in some aspen groves on the SNF, particularly on or near low-elevation winter ranges (e.g., in Sunlight Basin and North Fork of the Shoshone Basin; K. Houston, SNF soil scientist, personal communication). After wolf reintroduction in Yellowstone in 1995, elk began to avoid predation, apparently by moving more frequently, finding more dense hiding cover. Consequently, browsing

was less in the more open aspen stands, thereby increasing aspen growth (Ripple and others 2001, Fortin and others 2005). The recent migration of the wolf into the SNF may cause the same trophic cascade on the SNF. However, other causes of aspen decline may be as important or more so in some areas, such as fire suppression and various diseases that accompany the aging of trees established a century or more ago (Veblen and Lorenz 1986).

To offset declines in aspen, clearfelling and removal of encroaching conifers has been used on about 115 ha since 1980 to regenerate aspen. Photographs taken in 2000 show that such treatments have stimulated aspen regeneration in some areas (M. Hirschberger, SNF wildlife biologist, personal communication). This practice and the 1988 fires may be helping to maintain the extent of aspen on the landscape within the HRV, though Smith and others (2001) found that small scale clearcuts of aspen near the National Elk Refuge were not successful in restoring aspen because of heavy browsing by elk. Chong and others (2001) also cautioned about the possible invasion of exotic plants into aspen habitats after such treatments. With so little aspen on the SNF today (<1%), and because we have not yet found evidence for large areas having been dominated by aspen during the HRV reference period, we have concluded that aspen abundance is still within the HRV (low confidence).

The grassland cover type may also be declining relative to sagebrush on the SNF. On the Bighorn National Forest, grasslands declined by 15% and sagebrush increased by 4% between 1931 and 1996 (Fig. 36), which may have been due to fire suppression. However, given the variability in proportion of other cover types over time (e.g., forest vs. non-forest discussed above), sagebrush cover may not be above its HRV on either the Bighorn or Shoshone National Forests. Prescribed burning since the 1980s may also be reducing the rate of sagebrush invasion into grassland. Research is needed to determine if the amount of prescribed burning that is possible each year is adequate to prevent undesirable trends in vegetation change through natural succession when fires are suppressed.

Overall, management strategies that recognize the long-term, historic importance of fire, insects, and other natural disturbances—and aim to maintain large trees and snags—will most likely allow low-elevation forests to stay within their HRV (Romme and others 2000).

7. SUMMARY OF PROBABLE HRV DEVIATIONS

Based on the literature and data that we reviewed, and our understanding of the ecology of the SNF, it seems clear that the majority of the SNF is within the HRV. However, human-caused deviations from the HRV are occurring in some areas where timber harvesting, fire suppression, and livestock grazing have been important management activities and where exotic plants and diseases have become established. Does this matter? An answer is beyond the scope of this report. However, variables with deviations from their HRV should be considered for monitoring, and there may be good reason to initiate research designed to determine if the deviations could lead to undesirable consequences.

As our report describes, disturbances in the ecosystems of the SNF are not new. Indeed the ecosystems have evolved with regular disturbances. The effects of European-Americans have been significant in some areas, but only ~3% of the forested area has been harvested, mostly in a relatively small landscape on the Wind River District (~700 km² or ~270 square miles, 7% of SNF, 13% of non-wilderness on the SNF). Fire suppression and livestock grazing, and possibly exotic plants, have had more widespread effects, but much of the SNF is classified as wilderness, has a rugged topography, or is inaccessible by road. Some fires have been allowed to burn or were not extinguished until a large area was burned. Livestock grazing has been intensive and widespread in the past, but in most areas the numbers of sheep and cattle have declined greatly in the last 20 years (Fig. 14). Overall, compared to other National Forests in Wyoming, the SNF has been changed relatively little by European-Americans.

Based on our understanding of SNF ecosystems, we have attempted to draw conclusions about whether or not certain ecosystem variables are presently beyond their HRV during the period 1600-1860, or trending in that direction (Table 7). In the wilderness at high elevations, which comprise ~55% of the land area and where disturbances caused by fire, insect, disease and wind probably are still within their HRV, most variables pertaining to stand and landscape structure are still within their HRV (Table 7). Only in small areas near roads and where timber harvesting has been feasible is this not the case. Livestock grazing was intensive at one time, but the numbers of domestic animals is now greatly reduced and rangelands are undergoing succession that probably will bring ecosystem variables back within their HRV in the future, if indeed they had been exceeded previously. Exotic plants, once introduced, tend to persist and they reduce the abundance of some native species, but the extent of this potential problem is not widely known. Information on the HRV of grassland, forbland and shrubland ecosystem variables is scarce.

At high-elevation forests (35% of SNF), the following variables, estimated at the stand scale, appear to be within the HRV where timber harvesting has not occurred (confidence level is indicated in parentheses):

Fire return interval, intensity, and size (low, moderate, and low, respectively)

Insect outbreaks (moderate)

Abundance of diseases, except for dwarf mistletoe in some partial-cut areas (low) and where the introduced white pine blister rust is found (high)

Blowdowns (high)

Tree, sapling and seedling density (high)

Canopy cover and canopy gap density (low)

Snag density, coarse woody debris abundance, and mineral soil disruption or displacement (high)

Age- and size-class structure of most stands (moderate)

Understory plant density and cover (moderate)

Plant species diversity (moderate)

Forest floor depth (high)

Similarly, the following variables estimated at the landscape scale at high elevations appear to be within the HRV for the entire National Forest:

Fire return interval, size, and intensity (low, low, and moderate, respectively)

Insect outbreaks (moderate)

Abundance of diseases (low) except where white pine blister rust is found (high)

Blowdowns (high)

Tree, sapling and seedling density (moderate)

Canopy gap density and cover and mineral soil disruption or displacement (moderate)

Snag density and coarse woody debris abundance (high)

Age- and size-class structure of most stands (moderate)

Understory density and cover (moderate)

Plant species diversity (moderate)

Forest floor depth (moderate)

Number and proportion of land cover types (moderate)

Forest/non-forest ratio (high)

Proportion of forests in different successional stands and old forests (moderate)

Proportion in low canopy cover (high)

Proportion with high snag and coarse woody debris densities (moderate)

The amount of edge, interior forest, and patch size and shape (moderate)

Rate of patch formation (high)

The situation in low-elevation forests (19% of SNF) is different because the land is generally more accessible and the historic low frequency and mixed-severity fire regimes are more subject to modification through management. **Still, at low elevations, the following stand variables appear to be within the HRV:**

Insect outbreaks (low)

Abundance and composition of diseases (moderate), except where the introduced white pine blister rust occurs (high)

Blowdowns (high)

Snag density and abundance of coarse woody debris, except where timber harvesting has occurred (moderate)

Similarly, the following low-elevation landscape variables appear to be within the HRV for the SNF as a whole:

Insect outbreaks (low)

Abundance and composition of diseases (moderate), except where the introduced white pine blister rust occurs (high)

Blowdowns (high)

Proportion of landscape with old trees (moderate)

Proportion of land with high-density snags and coarse woody debris (low)

Proportion of land in different land cover types in forests (low)

For aspen stands (1% of SNF), the following variables appear to be within the HRV at high and low elevations and at both the stand and landscape scales:

Insect outbreaks (moderate)

Disease abundance, except near campgrounds (low)

Blowdowns (high)

Conifer density in understory (low)

Understory plant composition (low), except in areas with exotic plants (high)

Extent of aspen in the landscape (low)

As noted, very little information is available for drawing conclusions about the HRV of grassland, meadow and shrubland variables on the SNF (33% of SNF). However,

fire return interval, size, and intensity appear to be within the HRV (high confidence), probably at both the stand and landscape scales.

In contrast, the following variables probably are beyond their HRV within forest stands at high elevations where timber harvesting has occurred (~4% of the high-elevation forests on the SNF by the year 2000, primarily in the Wind River District):

1. Canopy cover probably is lower and the size and density of canopy gaps probably is higher in harvested stands than the HRV for these variables in unmanaged forests of comparable age and site conditions, due to selective and shelterwood cuts and fewer standing-dead trees (moderate confidence)
2. Snag density and the amount of coarse woody debris are lower than the HRV for unmanaged stands of comparable age and site conditions (high confidence). Whole-tree yarding can accentuate this variation by concentrating coarse woody debris and slash at landings in a way that has not occurred previously.
3. Due to some methods of mechanical slash treatment, the intensity of soil scarification is beyond the HRV in some areas (high confidence). Burning can remove the forest floor, but the depth of soil disturbance typically is less than following mechanical harvesting. Similarly, skid trails probably have compacted soils beyond their HRV, at least temporarily.
4. Age- and size-structure of managed stands within ravines or on some north or leeward slopes is skewed toward smaller and younger trees than would have occurred during the HRV period, because of harvesting where fires would have been less likely to burn (moderate confidence).
5. Dwarf mistletoe abundance is probably above its HRV in stands subjected to partial timber harvests, but could be less than the HRV in clearcut stands (low confidence).
6. White pine blister rust outbreaks cause some variables in affected stands of whitebark and limber pine to exceed the HRV, as this disease is not native (high confidence).

Across forested landscapes at high elevations where timber harvesting has occurred, primarily on the Wind River District (7% of SNF):

7. The estimated rotation times for timber harvest of forests judged suitable for harvesting are generally shorter, on average, than the estimated site-specific mean intervals between natural disturbances in high-elevation forests (high confidence). Old-growth forest is reduced, particularly in areas that have had long fire-free intervals, such as in valley bottoms and on some north or leeward slopes (moderate confidence). Thus, ecosystem characteristics that depend on periods of forest growth longer than ~140 years may be lost in some areas.
8. The rate of patch formation and the size of disturbances across high-elevation landscapes are outside the HRV (high confidence). Timber harvest produces patches more frequently than natural disturbances, and mid-sized patches can become unusually common. Natural fires produce many small patches and a few very large patches in such areas.
9. Due to roads and clearcutting, the amount of edge is now higher and the amount of interior forest is lower than prior to the 1900s in small areas on the Wind River District (moderate confidence). Correlated with more edge is a decline in the average patch size, the development of a more simple patch shape, and a higher, more uniform level of landscape patchiness than would have occurred during the reference period.

Within stands at low elevations affected by timber harvest (<1% of the forested land), fire suppression, and livestock grazing:

10. Fire suppression and livestock grazing have led to the development of low-elevation forest stands with higher densities of young trees (high confidence), greater canopy cover (high confidence), and less understory plant cover (low confidence) than usually occurred on many sites before 1860. Understory composition has also changed in some areas (moderate confidence, partly due to more competition from trees and exotic plants), and forest floor depth has increased (high confidence).

11. By removing large trees and downed wood, harvesting has lowered the abundance of snags and coarse woody debris in managed stands of low-elevation forests (moderate confidence, but only in less than 1% of the forested area). Harvesting, where it has occurred, also has led to an increase in sapling density and a reduction in the average tree diameter, thus creating younger, more uniform-sized stands than existed before harvest (high confidence).
12. Mean fire return interval has become longer at low elevations than the HRV (high confidence), and consequently fire intensity is more likely to be higher due to the amount and continuity of fuels, which enables more stand-replacing fires (low confidence).
13. White pine blister rust outbreaks cause some variables in stands with limber pine to exceed the HRV, as this disease is not native (high confidence).

Across forested landscapes at low elevations:

14. Similar to the stand-level, the size and intensity of fires have declined due to fire suppression at low elevations, and the amount and continuity of fuels is above the HRV in some areas, leading to the potential for more stand-replacing fires than occurred prior to 1860 (low confidence). Fire suppression has reduced the fire frequency (moderate confidence) and level of interspersion of tree stands with grasslands (low confidence), and average tree/sapling density has increased above the HRV (moderate confidence). Concomitantly, there is a more uniform canopy cover (moderate confidence), less understory vegetation (moderate confidence), and greater forest floor depth (moderate confidence) over larger areas than the HRV
15. Due to fire suppression, trees are smaller and younger on average than the HRV (moderate confidence)
16. White pine blister rust was absent during the HRV reference period, but is present today in stands of limber pine (high confidence).

In aspen forests (1% of SNF):

17. Livestock and/or native ungulate grazing and browsing in some areas have reduced aspen densities, but probably not to the point where this variable is beyond the HRV. However, exotic plants have affected the understory vegetation to a point where associated variables are exceeding their HRV (high confidence).
18. Fire suppression has reduced fire size and lengthened fire return intervals, thereby enabling an increase in conifer density in some areas to a level that is higher than the HRV (moderate confidence).
19. Due to human wounding of trees in recreational areas, some stands have an incidence of disease that is beyond the HRV (moderate confidence).

In non-forest vegetation (33% of SNF):

20. Some meadows, grasslands, and shrublands probably have been grazed more during the last century than during the HRV period, with the potential for an increase in forb abundance and soil erosion (moderate confidence). Deviations from the HRV also are caused by the invasion of exotic plants that displace native species and cause changes in other ecosystem properties (high confidence).

Frequently there has been a lack of pertinent quantitative data for the ecosystem variables we have addressed in this report (Table 7). Such deficiencies should be considered when identifying priorities for future research or monitoring. Research will also be required if decisions about the severity of a problem cannot be made without more information or if ways of restoring the HRV for important variables cannot be identified.

Much has been learned about ecosystem structure and function, but the kinds of influences to which National Forests are now subjected are different and are occurring more rapidly than ever before—and in a cumulative way. Human experience with modern forest management has been too short to speak with

confidence about sustainability, but HRV analyses can help achieve that goal. Our report, which focuses primarily on upland vegetation, should be supplemented with HRV reports on riparian ecosystems, aquatic ecosystems, and the population dynamics of plant and animal species thought to be potentially threatened by management activities.

Attitudes during the last century have set in motion changes that have pushed some ecosystem variables outside their HRV. The unusually high density of some low-elevation forests on the SNF is an example of how negative attitudes toward fire have influenced the landscape. Timber harvesting is sometimes promoted as a way of solving problems thought to be associated with fire suppression and the resultant development of what appear to be unhealthy forests. However, sick and dying trees surely have always been a part of the SNF—sometimes over large areas. Such trees create habitat for organisms that cannot survive on or around healthy trees. The biological diversity of the forest, which assures the long-term survival of forest ecosystems without human intervention, is higher because of periodic waves of mortality that oscillate through the landscape.

For some observers of management practices, forests with an abundance of diseases and insects should be treated by harvesting slow growing or dying trees. That may be desirable in some situations, but such judgments should be preceded by analyses to determine whether or not fire suppression has been effective on specific sites and whether or not the "treatment" creates more issues than it resolves. A stand of healthy trees surely is the goal of a tree farm on private land, but widespread stands of fast growing trees over much of a National Forest, where the pathogens are essentially all native, would not be within the HRV. Similarly, to harvest a forest to the point where it could not burn also would exceed the HRV for some ecosystem variables. Notably, and in contrast to some other western National Forests, many of the ecosystem variables that we have examined for this report on the SNF are still within their HRV.

In general, a continuing challenge is to determine if there are options for extracting wood, suppressing fires, and grazing livestock without long-term

undesirable effects on the various amenities of National Forests. Considering the HRV for key variables can be useful in achieving that goal.

8. LITERATURE CITED

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9. APPENDIX

Table A-1. Problems known to be associated with using data in the Shoshone National Forest RIS and Fire databases.

<p>Fire database (1970-2000)</p>	<p>Locations of the start point for each fire are mapped in the GIS. These data were used to locate fires by vegetation type. However, the fire may have started in one vegetation type but burned most extensively in another, and that information is not recorded, just the type where it started. Also, the program that National Forest personnel used to plot the points in the GIS occasionally shifted the location of the point from the start point to the center of the township section in which the fire started. Therefore, the location of some of the fires may be inaccurate, although the shift would not be large enough to make comparisons between high and low elevations inaccurate. Additionally, the vegetation type of the fire was recorded as the vegetation class following the fire, not before the fire. To avoid some of these problems, we attempted to use the cover type recorded with each fire in the database, but the coding system has changed over the years.</p>
<p>Large fire database (1930-1999)</p>	<p>The spatial extents of large fires were originally drawn on topographic maps. The polygons were crude on early maps and became more refined by the 1980s.</p>
<p>2000 RIS database and FIA database</p>	<p>Stand ages were estimated using the FIA database, not the RIS database because the stand ages for the RIS database were only for the 3% of the forest that was harvested. A problem with the more representative FIA database is that it only covers 374 plots on the forest, but the systematic distribution of the plots should give a relatively unbiased, though imprecise, estimate of stand ages. The IRI database might have improved on the RIS database but was unavailable for this report.</p>
<p>2000 RIS database</p>	<p>Mean patch size of clearcuts was based on the size of clearcuts that foresters recorded in the GIS database, not on the mapped polygon size. Many clearcuts were smaller than the polygon in the GIS, and the map of clearcuts shows more area cut than actual area because the entire polygon is shown. When clearcuts occurred in the same month and same polygon, they were considered one clearcut event. However, if clearcuts occurred in the same month and were in adjacent polygons, they remained separate, even though ideally they should be joined. Thus, the size of the clearcut patches may be underestimated somewhat (but total area is correct). Similar to the fire database, the vegetation type for a clearcut is the one after the clearcut occurred, not before.</p>

Appendix A-1 (continued)

<p>Romme (1979, 1982), Romme and Despain (1989) data compared to RIS data</p>	<p>The classifications used in Yellowstone National Park were only roughly comparable to the classifications that could be developed from the RIS. Ages associated with structural stages are very approximate, based on the best estimates available. Also, area of the landscape in each of Romme's classifications by seral stage (Fig. 38) was estimated using a dot grid on his maps in his 1979 dissertation and by estimating areas burned during the 1988 fires by examining a 1988 satellite image of the Park.</p>
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10. TABLES

Table 1. Percent of land area for common land cover types in the five districts of the Shoshone National Forest, arranged from north (Clarks Fork) to south (Wind River). See Fig. 5 for district location, Fig. 10 for a map of vegetation types, and Fig. 13 for comparable land cover composition data for the entire National Forest. Source: SNF RIS database.

Land Cover Type	Clark's Fork	Wapiti	Greybull	Wind River	Washakie
<u>Forests</u>					
Aspen	<1%	<1%	<1%	1%	1%
Limber pine	3	2	2	<1	<1
Douglas-fir	22	30	7	6	4
Lodgepole pine	21	10	4	19	38
Engelmann spruce- subalpine fire	14	19	24	26	10
Whitebark pine	7	10	8	9	10
<u>Non-forests</u>					
Grassland	24	22	47	25	18
Forblands	1	<1	<1	1	1
Shrublands	6	5	8	7	7

Table 2. Chronology of some major geologic and climatic events in Wyoming and Colorado, as summarized by Knight (1994a) and von Ahlefeldt and Speas (1996). mybp = million years before present; ybp = years before present

TIME	CLIMATE/GEOLOGY	VEGETATION
350 mybp	Tropical climate; submerged under saltwater	
300 mybp	swamps, lagoons, tidal flats over much of area	Clubmosses, horsetails, and ferns in swamps; pine, spruce and fir on upland
100 mybp	Ancestral Rocky Mountains leveled, still near equator	
245-66 mybp (Mesozoic era)	Overthrust belt formed (ca. 66 mybp); climate still tropical	Magnolia, palm, fig, breadfruit, sassafras, cinnamon, sweetgum, and willow in wetlands; conifers, ginkgo, and cycads on uplands
66-50 mybp	Mountains uplifted during Laramide orogeny; drifting apart of continents; climate still tropical, snow rare, much wetter than today	Flowering plants more common in upland forests; woody species include alder, beech, birch, black locust, chestnut, cottonwood, cyprus, dogwood, elm, fir, ginkgo, hickory, maple, oak, pine, redwood, spruce, sweetgum, sycamore, walnut, and willow; palm and breadfruit also present
50-36 mybp	Tremendous erosion, volcanism, and basin filling	
10 mybp	Regional uplifting creating present mountain landscape and rainshadows	
7-5 mybp	Increased aridity	Demise of some forests, giving rise to shrublands and grasslands; drought tolerant species persisted or immigrated; numbers of xerophytic plants increased in grasslands, fires became more frequent, leading to demise of trees and spread of grassland; woodlands restricted to ravines, valley bottoms, and ridges
660,000 ybp	Volcanism (creating Yellowstone plateau); with climate cooling, heavy snows and formation of glaciers	
2 mybp - 10,000 ybp (Pleistocene epoch)	Six glacial advances and retreats; flooding, creation of outwash plains, loess, and soil development	Coniferous forests, shrublands, and grasslands (many broad-leaved trees now extinct from the region)
127,000 ybp	Interglacial period	Douglas-fir and limber pine common in Yellowstone
15,000 ybp	Glacial advance; temperatures 10-13° C colder than today; permafrost	Trees in Yellowstone confined to a narrow band at lower elevations of mountains; tundra above the trees and tundra-like vegetation in basins below
11,500 ybp	Temperatures 5-6° C cooler than today; retreat of glacial ice	Upper treeline about 600 m (1,968 ft) lower than today in Yellowstone; gradually colonized by Engelmann spruce, and later by subalpine fir and whitebark pine in some areas (11,000-9,500 ybp)
9,500 ybp	Continued warming	Establishment of lodgepole pine at higher elevations and Douglas-fir in foothills in Yellowstone
9,000-7,000 ybp		Upper treeline at its lowest in Colorado Front Range
7,000-4,000 ybp	Altithermal period, comparatively warm and dry conditions	Expansion of sagebrush, greasewood, juniper and grasses; spruce and fir retreated to higher elevations; elevational range of spruce and fir reduced
4,000 ybp - present	Neoglacial period of gradual cooling	Expansion of forests to previous elevational ranges
ca. 1350 - 1500 AD and 1700 - 1900 AD	Characterized by cooler temperatures than at present time; see Figure 21	Probable glacial advances in the mountains
1850 AD - present	Generally warmer and wetter; 20 th century warmer than previous 1000 yrs	Increased tree recruitment near upper treeline (Hessl and Baker 1997), in subalpine meadows (Jakubos and Romme 1993) and in montane forests (Savage and others 1996)

Table 3. Vegetation of the Shoshone National Forest, as classified at various levels of detail. Source: SNF, Tweit and Houston (1980), Steele and others (1983), Roberts and others (2001), and Zimmerman and Roberts (2001).

This Report	SNF (dominant species)	Habitat and community types (Steele and others 1983, Tweit and Houston 1980, Roberts and others 2001)	Potential natural vegetation types (PNV) (Zimmerman and Roberts 2001)
High-elevation forests	Engelmann spruce-subalpine fir Lodgepole pine Whitebark pine	16 Subalpine fir habitat types 11 Engelmann spruce habitat types 6 Lodgepole pine habitat types 5 Whitebark pine habitat types	6 Engelmann spruce-subalpine fir PNV types 1 Whitebark pine PNV
Low-elevation forests and woodlands	Douglas-fir Limber pine Rocky Mountain juniper Ponderosa (very little)	10 Douglas-fir habitat types 3 Limber pine habitat types	4 Douglas-fir PNV types
Aspen and riparian woodlands	Aspen Cottonwood Blue Spruce	11 Aspen community types (1 aspen series) 9 Deciduous riparian woodland community types (includes those dominated by cottonwoods, subalpine fir, and blue spruce)	1 wetland riparian PNV

Table 3 (continued)

<p>Non-forest vegetation (grasslands, shrublands and forblands)</p>	<p>Grasslands dominated by Idaho fescue, tufted hairgrass, bluegrass, wheatgrass, kobresia, rushes, and sedges</p> <p>Shrublands dominated by sagebrush, serviceberry, willow, and including krummholz at treeline</p> <p>Forbland</p> <p>Cropland</p>	<p>Grasslands: 2 wheatgrass habitat types, 4 fescue habitat types</p> <p>Shrublands: 10 sagebrush habitat types, 1 bitterbrush habitat type, 2 shrubby cinquefoil habitat types, 1 greasewood habitat type, 1 sumac habitat type, and 33 riparian/wetland communities</p> <p>Forblands that include 5 alpine community types (of which two are also wetland types) and 3 non-wetland forb community types, and 9 wetland/riparian community types</p>	<p>Grasslands: 2 Idaho fescue PNVs</p> <p>Shrubland: 2 big sagebrush PNVs</p> <p>All forblands classified as one alpine PNV</p>
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Table 4. Examples of ranges for average fire intervals in high-elevation forests in the Rocky Mountain region, as estimated by various investigators. These ranges represent the typical amount of time between successive fire events in an individual stand. YNP = Yellowstone National Park.

Study	Forest type / location	Fire return interval (years)	Type of fire event	Basis for estimates
Billings (1969)	Engelmann spruce-subalpine fir in southeastern Wyoming	up to 600-700	stand-replacing	stand age structure
Arno (1980)	Subalpine forest in the Northern Rockies	150-350	stand-replacing	stand age structure
Romme (1980a, 1982)	Lodgepole pine, subalpine fir, and Engelmann spruce in YNP	300+	stand-replacing	stand age structure / fuel accumulation
Romme and Knight (1981)	Engelmann spruce-subalpine fir in southeastern Wyoming	300-500+	stand-replacing	stand age structure / fuel accumulation
Romme and Despain (1989)	Lodgepole pine, subalpine fir, and Engelmann spruce in YNP	200-400	stand-replacing	stand age structure / fuel accumulation
Kendall (1998)	Whitebark pine in the Rocky Mountains	50-300 historically; changed to 3000 in last 25 years	all	fire scars
Hawkes (1980)	Lodgepole pine, subalpine fir, Engelmann spruce, and western larch in Alberta	90-304 ¹	all	fire scars
Kipfmüller (1997)	Lodgepole pine, subalpine fir, and Engelmann spruce in southeastern Wyoming	39-149 ²	all	fire scars
Barrett (1993)	Lodgepole, subalpine fir, and Engelmann spruce in northcentral Idaho	213--stand replacing 39--underburn	all	stand age structure and fire scars
Arno (1976), Arno and Peterson (1983)	Whitebark pine in the Bitterroot National Forest, Montana and Idaho	30-94	all	fire scars
Morgan and Bunting (1990)	Whitebark pine in the Shoshone National Forest	10-77 before 1867; no fires after 1867	all	fire scars on at least two trees in 100-ha stand

¹This range represents overall averages calculated for forests at different elevations and topographic positions, including: 1) lower subalpine/lower elevation (90-101 years); 2) upper subalpine/upper elevation (153-304 years); 3) north aspects (187 years); and 4) south, east and west aspects (93-104 years).

²This range represents four different methods for calculating point-scale MFI: 1) the interval between stand origin date and first scar (82 years); 2) the interval between scars on the same tree (39 years); 3) the interval between the last scar and the present (149 years); and 4) the overall average of all intervals (129 years for 54 different stands in the Rock Creek Basin).

Table 5. Some fire-return-intervals reported for stands containing Douglas-fir in the Rocky Mountain region. The range in parentheses is the range in interval sizes for one stand or area and is not the range of means because it considers extremes (see text for explanation). When the mean shifted by more than 75% between pre- and post-settlement, we considered the mean fire return intervals to be outside the HRV, which was true for all studies in this table with pre- and post-settlement estimates except the last one. YNP = Yellowstone National Park.

Study	Forest type and location	Mean fire return interval estimates and/or ranges (years)		Type of fire event	Basis for estimate
		Pre-settlement	Post-settlement		
Loope and Gruell 1973	Douglas-fir in Jackson Hole, Wyoming	25-100 ¹			
Houston 1973	Douglas-fir in YNP	before 1890: 20-25	70+	surface	fire scar analysis
Barrett 1994	Douglas-fir in YNP	before 1870: 28 (15-50)	(48-119+)	all types	stand age and fire scars
Littell 2002	Douglas-fir in the northern part of the Greater Yellowstone Ecosystem	before 1870: 21-33	132+	mixed	fire scars
Arno and Gruell 1986	Douglas-fir in Montana	35-40	rare	mixed	stand age and fire scars
Brown and others 1995	Douglas-fir and grand fir in northern Idaho and western Montana	before 1935: 119		stand replacing and mixed	stand age and fire scars
Barrett 1993	Douglas-fir, grand fir, and western red cedar ² in northcentral Idaho	107-216		stand-replacing	stand age and fire scars
Barrett 1993	Douglas-fir and ponderosa pine in northcentral Idaho	before 1890: 10-44 (3-60)	91-105+	surface	fire scars
Brown and others 1999	Ponderosa pine and Douglas-fir in central Colorado	1496-1851: 59.2 (27-128+)	1880-1996: no fires	surface	fire scars
Goldblum and Veblen 1992	Ponderosa pine and Douglas-fir near Boulder, Colorado	before 1859: 31.8	1859-1920: 8.1 ≥1920: 28	surface	fire scars

¹A range of medians or means indicates more than one stand was sampled.

²*Psuedotsuga menziesii*, *Abies grandis*, and *Thuja plicata*

Table 6. Successional stage characterizations for lodgepole pine and Engelmann spruce-subalpine fir forests, used when comparing Yellowstone National Park (YNP) to the Shoshone National Forest.

Yellowstone National Park—from Romme and Despain 1989			Shoshone National Forest—from the FIA database
Successional Stage	Years after stand-replacing fire	Characteristics	Stand age
Lodgepole pine seedlings/saplings (LP0)	0 to 40	Vegetative sprouting of herbaceous plants; lodgepole pine seedling establishment; fuel consists of large, dead tree boles; does not ignite easily or support intense fires	about 0 to 40 years
Immature lodgepole pine (LP1)	40 to 150	Canopy closure; stands are relatively dense with little groundlayer vegetation; period of tree growth, competition, and thinning; fuels are discontinuous from forest floor to canopy, do not burn readily, and fires die in absence of wind	about 40 to 160 years
Mature lodgepole pine (LP2)	150 to 300	Tree growth, slower thinning; canopy more open; groundlayer more dense and diverse; mixed understory of lodgepole pine, subalpine fir and Engelmann spruce common; dead, woody fuels increase but still somewhat resistant to fire; most fires will die out in absence of wind	about 160 to 300 years
Lodgepole pine, Engelmann spruce, and subalpine fir (LP3)	more than 300	Canopy mortality increases; understory trees reach 2-3 m (6.5-10 ft) tall and grow into canopy gaps; fuels and organic materials build up, burn relatively easily when weather conditions are right; last stage before next stand-replacing fire event in southcentral YNP.	more than 300 years

Table 7. Summary of variables describing upland vegetation characteristics on the Shoshone National Forest relative to their estimated historic range of variability (HRV) from about 1600 to 1860, with an indication of our confidence in making each comparison at the stand (S) and landscape (L) scales. For our discussion of each variable, refer to the pages listed in the column on the right. WRD = Wind River District.

Variable	Within HRV	Outside HRV	Confidence	Pages with discussion of variable
High-elevation forests				
<i>Disturbances</i>				
Mean return interval for stand-replacing fires	S,L		Low	54-61
Fire intensity	S,L		Moderate	54-61
Fire size	L		Low	54-61
Insect epidemics	S,L		Moderate	72-78
Dwarf mistletoe abundance	S,L in general	S with partial harvests	Low	82-88
White pine blister rust		S,L	High	82-88
All other diseases	S,L		Low	82-88
Wind	S,L		High	90-91
<i>Stand structure at high elevations (96% of stands are within HRV for stand structure)</i>				
Density of trees in all age and size classes	S,L		S-High L-Moderate	94-95
Regeneration time and seedling/sapling density	S,L		S- High L-Moderate	95-97
Tree canopy cover and the density and size of canopy gaps	S,L in general	S,L on WRD temporary	Moderate	97-98
Understory plant density and cover	S,L		Moderate	98-99
Diversity of all plants	S,L		Moderate	99-100
Age and size-class structure of trees	S,L in general	S,L on WRD on north & leeward slopes, ravines	Moderate	100-101
Forest floor depth	S,L		S-High L-Moderate	101-102
Mineral soil structure disruption/compaction	S,L in general	S,L on WRD	S-High L-Moderate to High	102-103
Mineral soil displacement	S,L		S-Low L-Moderate	103

Snag density and coarse woody debris	S,L in general	S,L on WRD	S-High L-Moderate to High	103-106
<i>Landscape features at high elevations</i>				
Proportion of landscape in low canopy cover	L		High	123
Number and proportion of different land cover types	L		Moderate	121
Forest/non-forest land area ratio	L		High	121-122
Proportion of forest land in old-growth and mid-successional stages	L in general	L on WRD	Moderate to High	122-123
Proportion of land with high snag and coarse woody debris densities	L	L on WRD	Moderate	124
Proportion of land in edge, interior forest, and patchy forest	L in general	L on WRD	Moderate	124-127
Rate at which new, uniformly distributed, small- to mid-sized patches are formed	L in general	L on WRD	High	127-128
Low-elevation forests				
<i>Disturbances</i>				
Mean return interval for surface fires		S,L	S-High L-Moderate	54-55, 64-69
Fire size and intensity		S,L	Low	54-55, 64-69
Insect epidemics	S,L		Low	78-81
White pine blister rust		S,L	High	88-89
Other diseases	S,L		Moderate	88-89
Wind	S,L		High	92
<i>Stand structure at low elevations</i>				
Tree density and canopy cover		S,L	S-High L-Low	107-108
Plant species diversity, canopy cover and gaps, and understory plant composition and cover	S for diversity	S,L	S-Low L-Moderate	108-109
Size- and age-class structure		S,L	S-High L-Moderate	109-110
Forest floor depth		S,L	S-High L-Moderate	110

Snag density and coarse woody debris	S, L	S in harvested areas only	S-Moderate L-Low	110-111
<i>Landscape features at low elevations</i>				
Proportion of landscape with high-density trees and high fuel continuity		L	Moderate	128-129
Proportion of landscape with low understory plant cover and relatively deep forest floors		L	Moderate	129
Proportion of landscape dominated by old trees	L		Moderate	130-131
Proportion of landscape with high-densities of snags and coarse woody debris	L		Low	131
Proportion in forest and non-forest		L	Low	131-134
Proportion of landscape in different cover types	L		Low	131-134
Forest patch sizes and configuration		L	Low	130
Aspen woodlands				
Insect epidemic frequency	S,L		Moderate	81-82
Blowdown frequency	S,L		High	92
Fire size and mean return interval at stand and landscape scales		S,L	Moderate	54-55, 69-70
Disease abundance	S,L	S near recreation areas	Low	89
Conifer density in understory	S		Low	112-114
Understory plant composition and stand structure	S without exotic plant species	S with exotic species	High-exotic Low-native	112-114
Representation in landscape	L		Low	112-114, 131-134
Non-forest vegetation	Little information available for drawing conclusions about specific variables, but see summary and text for discussion			54-55, 71, 82, 89, 114-117, 131-134

11. FIGURES