

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

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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. Such analyses provide a means of synthesizing the scientific literature. The rationale for HRV analyses is that the chances of sustainable forest management are greater if the variation in managed ecosystems includes the range of conditions that are expected at various scales in ecosystems relatively uninfluenced by humans. This report provides an HRV analysis for the Bighorn National Forest (BNF) of northcentral Wyoming.

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 examined is found in Table 7. Examples 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. For this report 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 ponderosa pine, Douglas-fir, and aspen

at lower 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 BNF as approximately 1600 to 1890, as some data sets on fire history go back several centuries and the influence of European-Americans in the BNF was minimal until about 1890. We also compared BNF ecosystems to comparable ecosystems in natural areas relatively uninfluenced by management activities, such as Yellowstone National Park. When data were lacking, our approach was to first make plausible though qualitative judgements 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 were evaluated and supported by a panel of six anonymous peer reviewers. 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 report describes how the forests of the BNF have evolved with regular disturbances, and how the kinds and frequency of disturbances have changed in some areas due to modern management practices. Only ~18% of the forested

area has been harvested for timber (high- and low-elevation landscapes combined). Roughly 12% has been harvested on two occasions. The effects of fire suppression and livestock grazing are more widespread, especially at the high elevations.

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

Within stands at high elevations affected by timber harvest (~20% of the high-elevation forests on the BNF):

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. Where timber harvesting has occurred, snag density and the amount of coarse woody debris is 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 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 may be below its HRV in clearcut stands (low confidence).

Across forested landscapes at high elevations:

6. 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 has been 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. The proportion of other age-classes are within the HRV, except for the now more abundant mid-successional forests (low confidence)
7. The average snag density and coarse woody debris biomass across the landscape have been reduced below the HRV (moderate confidence). The loss of some old-growth forests and the suppression of fires that create abundant snags have reduced the percentage of the forested landscape that contains high density snags/coarse woody debris to a level that is below the HRV (high confidence).
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 are over-represented. Natural fires produce many small patches and a few very large patches at high elevations. Fire return intervals, size, and extent also have been reduced below the HRV due to fire suppression (low confidence).
9. The percentage of the landscape with low canopy cover probably is higher than the HRV due to partial cutting and thinning (low confidence).

10. 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 parts of the BNF (low confidence). Correlated with this and fire suppression is a decline in the maximum 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 (high confidence).
11. Some kinds of mechanical slash treatment have led to a higher level of mineral soil disturbance than would have occurred during the HRV reference period (high confidence)
12. Dwarf mistletoe may be somewhat above its HRV on the landscape scale due to the tie-hacking, selective cutting and thinning that has occurred on the forest (low confidence).

Within stands at low elevations affected by harvest, fire suppression and livestock grazing:

13. Fire suppression, livestock grazing, and possibly climate change have led to the development of low-elevation forest stands with higher densities of young trees, greater canopy cover, less understory plant cover, and deeper forest floors than usually occurred on many sites before 1890 (see Table 7 for confidence levels associated with each variable). Understory composition also has changed where introduced plants occur.
14. By removing large trees and downed wood from the system, wood harvesting has lowered the abundance of snags and coarse woody debris in managed stands of low-elevation forests (moderate confidence). Harvesting also has led to an increase in sapling density and a reduction in the average diameter of trees, thus creating younger, more uniform-sized stands than existed before harvest (moderate confidence). However, only about 1% of such forests have been subjected to harvesting thus far.
NOTE: Without harvesting and with fire suppression, snags and coarse woody debris probably are still within the HRV.
15. Mean fire return interval has become longer than the HRV due to fire suppression (moderate confidence). Consequently, there is a trend for tree

distribution to be more uniform and to have tree invasion into some meadows.

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

Across forested landscapes at low elevations:

17. Similar to the stand-scale, the sizes of most fires have declined due to fire suppression, and the mean fire interval, tree/sapling density, and amount and continuity of fuels is above the HRV in a larger proportion of the BNF than during the reference period (moderate confidence).
18. Understory plant density and cover have declined below the HRV (low confidence) and forest floor depth has increased above the HRV in a larger proportion of the landscape than during the reference period (moderate confidence).
19. White pine blister rust was absent during the HRV reference period, but is present today (high confidence).
20. The age- and size-class structure of trees (moderate)

Aspen forests:

21. In some areas, livestock or native ungulate grazing and browsing have reduced aspen densities and affected the understory vegetation to a point where these variables are exceeding their HRV (high confidence). These changes are confounded with the introduction of exotic plants.
22. 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).
23. Due to human wounding of trees in recreational areas, some stands have an incidence of disease that is beyond the HRV (low confidence).

Non-forest vegetation:

24. Some meadows, grasslands, and shrublands are probably grazed more, or have been in the past, than during the HRV period, with the potential for an increase in soil erosion and forb and shrub abundance (high confidence). Deviations from the HRV are also caused by the presence 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 variables can be identified as 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 ways of restoring the HRV for important variables cannot be identified.

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 et al. 1994, Christensen et al. 1996, Vogt et al. 1997, Landres et al. 1999). The proponents of ecosystem management recognize that our current knowledge of ecosystems is incomplete (Christensen et al. 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 Bighorn National Forest (BNF) in northcentral 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 et al. 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. Kaufmann et al. (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 et al. 1993), "natural variation" (Holling and Meffe 1996), "reference conditions" (Kaufmann et al. 1994), "reference variability" (Manley et al. 1995), and "historic range of variability" (Morgan et al.

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 1890. Historical evidence suggests that timber harvesting and livestock grazing did not begin in the BNF until after 1890 (Martorano et al. 1985).

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. We also evaluated the following general vegetation types separately: lodgepole pine/spruce-fir forest (referred to as high-elevation forest), Douglas-fir/ponderosa pine/limber pine forest (referred to as low-elevation forest), aspen, and non-forest. Because the BNF timber and fire databases were often not separated into these vegetation types, our interpretation for the HRV of some landscape-level variables (e.g., fire regimes and mean shape of disturbance patches) required some assumptions (identified

later in the report). 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 BNF, 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 ecosystems

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 the 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 BNF, Gray (2003) and Gray et al. (submitted) 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 the Bighorn Mountains began to warm in the late 1800s (Christopher Fastie, personal communication). While the HRV estimates of some ecosystem variables for the reference period 1600-1890 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.

To characterize the HRV, a reference period must be specified to which the last century or so can be compared. Manley et al. (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), but we selected 1600 to 1890 (Martarano et al. 1985). It would be good to consider a longer, evolutionary time scale, but climate change and glaciation in the BNF 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.

Notably, an evolutionary perspective is adopted in our report, but in a different sense than implied by Manley et al. (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 BNF 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 (*Pinus contorta*) 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 et al. 1993, Holling and Meffe 1996). Because ecosystems change dramatically over 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 et al. 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 et al. 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 et al. (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 judgments are 1) there is much yet 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 et al. 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 et al. 1993). Restoration of the HRV may not be practical or desirable, or even possible, in some situations.

In this report for the BNF, 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 BNF 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 BNF 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 BNF, but interpreting the historic development and variability of grassland and shrubland variables for our reference period (1600-1890) is much more difficult, if it is even possible, 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 of considerable interest, but is beyond the scope of our report.

2. METHODS

Manley et al. (1995) suggested that estimating 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 plausible, though qualitative, 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 et al. 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 BNF, though YNP is generally cooler and more moist. The geological history is different, 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 BNF 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, BNF reports, and BNF databases. Several sources were especially useful in quantifying historic and present conditions specific to the BNF, namely the Resource Information System (RIS) and the fire database of the BNF. Both databases are maintained at the Supervisor's Office in Sheridan, Wyoming (USDA Forest Service 1986, 1996). While there are some limitations to using these databases (Appendix, Table A-1), they provide the most comprehensive source of spatial information. We also used maps of disturbance patterns across

the landscape that had been derived from satellite imagery by the Wyoming Game and Fish Department (1996). Finally, we conducted interviews of BNF personnel and others to gain a better understanding of the management and ecology of the BNF.

For this report, we have drawn conclusions about whether existing conditions have changed from conditions during the HRV reference period, from about 1600 to 1890, due to the management activities of European-Americans. We define *existing conditions* (EC) as conditions occurring during any period after 1890, though primarily during the last half-century. In general, we defined HRV as the range of means of a variable for several consecutive time periods during the period 1600 to 1890 (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 argument could be made, 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. Future research is required to help evaluate the accuracy of the assumptions associated with our approach.

The above examples assume data for the HRV period are available for the BNF. In reality, pertinent HRV data are usually available only in Yellowstone National Park 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 BNF during the HRV period, as follows:

- 1) We used data collected for the HRV period from studies in other places in the Rocky Mountains.
- 2) For many analyses, we used existing conditions (20th century) in Yellowstone National Park as the conditions on the BNF during the HRV period. The Park's vegetation has been less strongly influenced by European-American activities but probably reflects historic patterns that occurred in the Bighorn Mountains. 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 existing 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 very artificial disturbances (e.g., timber harvest). Admittedly, some of the natural disturbances have been modified by humans (e.g., high elevation fires are smaller on average than during the HRV). Therefore, we only use this method when the modification in the pattern is small relative to the modification caused by the artificial disturbances (e.g., present-century fire patterns are compared to present-century clearcut patterns on the landscape).
- 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 timing of disturbance patterns in watersheds are uncorrelated and each watershed 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, 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 applied to the stand compared to the landscape scale, 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 a result, the HRV narrows as the scale changes from the stand level to the landscape level (Fig. 1). For example,

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. By contrast, it is highly unlikely that all of the snags across a landscape would all fall before new ones are formed, or that all live trees 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 from this discussion is that the HRV for a variable is dependent on the amount of land area being considered. In other words, the probability of finding a stand or landscape with zero snags declines as the size of the stand 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. This approach supplements our landscape-level, qualitative estimate of snag density.

3. THE BIGHORN NATIONAL FOREST

3.1 PHYSICAL SETTING

The Bighorn National Forest (BNF) is located in northcentral Wyoming (Fig. 5) and covers 447,500 ha. The Forest occurs over most of the Bighorn Mountains, a relatively isolated mountain range that rises from rolling plains at 900-1200 m up to 4,018 m on Cloud Peak (USDA Forest Service 1985). On the eastern and western forest boundaries, the flanks of the mountains drop steeply to the surrounding Bighorn and Powder River Basins. High peaks rise from a subsummit plateau (at 2400-2800 m) in the southcentral portion of the mountain range (Fig. 6). The glaciated peaks of granite slope off to gently rolling and timbered drainages, all of which contribute water to the Yellowstone River (Fig.

7). The Forest is characterized by several spectacular canyons through sedimentary strata on the west slope, large expanses of non-forested vegetation on fine-textured soils derived from sedimentary strata at mid-elevations, extensive forested land in the north and south, and the large Cloud Peak Wilderness Area, much of which is above treeline. Many lakes occur in glaciated areas.

The anticlinal mountains were formed during the Laramide Orogeny and are composed primarily of igneous and sedimentary rocks with a Precambrian granite foundation (Fig. 8). Much of the sedimentary rock has eroded away to expose the granite, but the northern and southern ends of the mountains are still overarched by sedimentary strata (sandstone, shale, dolomite, limestone). Because the central granitic block was uplifted to higher elevations, its peaks were glaciated during the Pleistocene. The Cloud Peak Wilderness is now centered on these peaks. No glaciers reached below 1990 m and the lower ends of the canyons retain their angular, preglacial form. Tilted sedimentary rock layers form steep flatirons and escarpments on the eastern and western flanks.

Soils derived from the granite are acidic, shallow, coarse-textured, and low in nutrients. Shale-derived soils are deep, fine-textured, high in nutrients, and acidic to neutral. Soils derived from interbedded limestone, sandstone, and dolomite are of intermediate texture, basic, and high in nutrients (Despain 1973, Nesser 1986).

As with other portions of the Rocky Mountains, the climate of the BNF varies greatly with elevation and topographic position (Fig. 9). The lowlands east and west of the Bighorns are dry, with a mean annual precipitation from 28 to 38 cm on the plains to the east and 13 to 18 cm in the drier Bighorn Basin to the west. In the mountains, precipitation averages about 38 cm at the lower limits of tree growth on the eastern foothills (1,520 m) and increases to ~63 cm at higher elevations (2740 m) (Nesser 1986). At the highest elevations, cumulative snowfall is 400-500 cm from October to May, with an average snowpack depth of 178 cm. Temperatures range from -40°C to 43°C , with an annual mean of 8°C at Hyattville in the foothills at 1367 m to 1°C at Burgess Junction on the plateau

at 2450 m. Frost can occur during any month of the year at higher elevations. Mean annual air temperature decreases about 1.5°C per 300-m increase in elevation (Nesser 1986). South slopes are considerably warmer and drier than north slopes or creek bottoms, causing great spatial variability in the distribution of different vegetation types. The western side of the BNF receives less precipitation than the eastern side because it lies within the rain shadow of the higher Absaroka Mountains to the west as well as the rain shadow of the Bighorn Mountains, which intercepts moisture caused by upslope air circulation from the east (Fig. 9). Cloud Peak also appears to create a rain shadow to the southeast (Despain 1973).

Historically, the Rocky Mountain region has experienced great climatic and geological changes over time scales of millions of years (Table 1). 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 species present today were in place (Tidwell 1975). From approximately 2 million to 10,000 years ago, during the Pleistocene epoch, several glacial advances and retreats (Table 1) affected both the topography and vegetation in the center of the Bighorn Mountains (Nesser 1986). Since the end of the Pleistocene around 10,000 years ago, shifts in climate have continued, often triggering changes in the spatial and elevational distribution of many plant species (Table 1).

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 BNF ecosystems. Soils, for example, surely develop over a longer time period. Moreover, future climatic conditions may be different than those of the past, 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 BNF (Table 2). On a broad scale, the vegetation is classified as part of the Southern Rocky Mountain Steppe-Open Woodland-Coniferous Forest Province

(Bailey et al. 1994). At an intermediate scale, the vegetation can be classified by the dominant overstory species, which is the approach adopted for the BNF for its RIS database. Using this intermediate system, the most dominant forest cover types are lodgepole pine and Engelmann spruce-subalpine fir (*Picea engelmanni-Abies lasiocarpa*) (Fig. 10). Ponderosa pine, aspen (*Populus tremuloides*), limber pine (*Pinus flexilis*), and Douglas-fir (*Pseudotsuga menziesii*) forests and woodlands are more common at low-elevations, but are much less abundant (Fig. 10). Grasslands (meadows) are scattered throughout the BNF, but are more common in the north where fine-textured soils are more widespread. Forb-dominated areas (mapped with willows in Fig. 10) are mostly in the alpine and subalpine meadows. Shrublands containing mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) are common. Juniper (*Juniperus osteosperma*) and shrublands dominated by curlleaf mountain mahogany (*Cercocarpus ledifolius*) can be found on the BNF, but they are more common in the foothills outside the Forest boundary.

More detailed vegetation classifications have been developed for the BNF. As in other parts of the Rocky Mountains (Moir 1969; Loope and Gruell 1973; Whipple and Dix 1979; Despain 1983, 1990; Lotan and Perry 1983; Peet 1988), large areas are dominated by forests of lodgepole pine (Fig. 10). Compared to the Medicine Bow National Forest in southeastern Wyoming, there is a lower tree basal area per hectare in the spruce-fir zone and more of the lodgepole pine stands are self-perpetuating rather than seral to Engelmann spruce and subalpine fir (Hoffman and Alexander 1976, Despain 1973). Hoffman and Alexander (1976) described six principle forest series on the BNF, based on overstory dominants; and 15 habitat types, based on overstory dominants and understory indicator species (after Daubenmire and Daubenmire 1968; Table 2). Proponents of the 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. Also, such classifications focus on the potential natural

vegetation given long periods without disturbance, rather than the actual vegetation at the present time.

Because of concerns about protecting riparian habitats, Girard et al. (1997) developed a detailed classification system for riparian vegetation on the BNF. Their system described 53 communities. Each community was placed into the following broader classification based on the dominant riparian species: sedge (*Carex* sp.), grass, Booth's willow (*Salix boothii*), Geyer's willow (*Salix geyeriana*), planeleaf willow (*Salix planifolia*), wolf's willow (*Salix wolfii*), other willow (*Salix* sp.), shrubby cinquefoil (*Potentilla fruticosa*), narrowleaf cottonwood (*Populus angustifolia*), aspen (*Populus tremuloides*), lodgepole pine, and Engelmann spruce. Discussion of the HRV for riparian ecosystems is planned for a future report.

About 62% of the BNF is forested (USDA Forest Service 1994) and rock type has a strong effect on the location of forests (Despain 1973). On sedimentary substrates, forests cover less than 50% of the area and are composed of Douglas-fir or Engelmann spruce and subalpine fir. On granitic substrates, forests cover over 80% of the land area and are dominated by lodgepole pine at mid-elevations and Engelmann spruce and subalpine fir at higher elevations (Despain 1973). The environmental preferences of some shrubs, forbs and grasses species are described by Despain (1973) and Hoffman and Alexander (1976).

Thirty-seven percent of BNF forests have been judged suitable for timber harvesting (102,645 ha, BNF RIS Database). About 25% of this suitable land was subjected to tie hacking from 1893 to 1939 (R. Laurent, BNF archaeologist, and B. Bornong, BNF Ecologist, personal communication). Another 15% of the suitable land has been subjected to harvesting by European-Americans since 1940 (RIS database). Thus the total area judged suitable for timber harvesting and known to be affected by harvesting is about 40% (though this estimate drops slightly when numbers are adjusted to account for the fact that some old harvests occurred on unsuitable land—about 5900 ha). About 23% of the tie hacked forests were harvested again after 1940. Harvests after World War II have

affected about 21% of the suitable land base. When the Forest Plan is revised, the percentage of lands deemed suitable will be re-evaluated.

When discussing each variable, we addressed four broad landscape types: 1) high-elevation forests (i.e., spruce-fir/lodgepole pine forests in the upper montane and subalpine, 49% of BNF); 2) montane forests and woodlands at lower elevations, 12% of BNF (i.e., Douglas-fir, ponderosa pine, limber pine); 3) aspen forest and woodlands, 1% of BNF; and 4) non-forest vegetation, namely, upland grasslands, forblands, shrublands, and juniper woodlands (24% of BNF). The remainder of the Forest is wetlands or barren habitats with little or no vegetation. Note that high elevation forests are not defined by an elevation boundary, but rather as all spruce-fir or lodgepole forests on the BNF. Similarly low elevation forests are all of the Douglas-fir, ponderosa pine, or limber pine forests/woodlands. These landscapes could be broken into smaller classification units, as there certainly are important ecological distinctions between different habitat types and community types within each of the general forest types. However, the differences blur over the long time periods of HRV analyses. Furthermore, because of successional relationships between lodgepole pine, subalpine fir and Engelmann spruce, and in some areas between Douglas-fir and ponderosa pine, it is difficult to discuss HRV separately for each overstory cover type.

3.3 CULTURAL SETTING

For centuries humans have lived and traveled throughout the area that is today the BNF (Murray 1980). Consequently, managers have inherited the legacies of past land uses and are not starting with a "blank slate" (Romme et al. 2000). This is important for several reasons. First, our ability to interpret pre-European-American conditions will be limited by events caused by Native Americans during the last several centuries or more. 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, many of the unprecedented

human influences of European-Americans have left legacies that present unique challenges for managers. Because human land uses on the BNF have changed over time, we provide a brief summary of three periods that relate to natural resource management.

3.3.1 Historic Period (1600-1890)

The first and earliest time period is from about 1600-1890, a period when European-Americans had not yet directly impacted the area to any extent (Martorano et al. 1985). During the early years of this period, the Native American tribe called the Snakes probably occupied the Bighorn Basin and Bighorn Mountains (Murray 1980). The Snakes and their ancestors, which later became the Shoshoni and Comanche tribes, had a mixed hunting/foraging economy. The mountains were most likely used during the summer to escape the heat at low elevations, to hunt or trap the abundant bison and elk, and to obtain medicinal and food plants. As evidenced by the Medicine Wheel constructed during this period, sometime after 1620, the mountains were also used for ceremonial purposes (R. Laurent, personal communication).

By 1812, the Crows (or Absaroka) had obtained horses and guns and were in control of the Bighorn Mountains. The Cheyenne and Arapaho later fought the Crows and succeeded in gaining some control of the land (Scott 1981). With the decline of bison in the east, the Dakota (Sioux) tribe also began to move into the region, forming an alliance with the Cheyenne and Arapaho and forcing the Crow and other tribes to retreat. Thus, the Bighorn Mountains were part of an Indian “war” ground between 1830 and 1860. Away from a tribes’ established hunting territory, war zones probably provided a refuge for the hunted game species (Scott 1981, Martin and Szuter 1999). For short periods in the fall, tribes from the west, such as the Shoshone, would use the area as a hunting ground, but hostilities prevented long-term use. The impacts of these early inhabitants on the landscape are largely unknown, but, except for a few intentional or unintentional fires, impacts were likely minor and probably were limited primarily to foothills, riparian areas and travel routes. Grasslands and big

game animals were abundant in the mountains, and thus, intentional Indian-set fires were probably not as common as in other parts of the Rocky Mountains (R. Laurent, personal communication)—where fire was helpful for creating openings that would attract game. Notably, Baker (2002) concluded that, for Rocky Mountain landscapes in general, Native Americans had little effect on fire regimes in lodgepole and spruce-fir forests.

In general, Native American influences probably had less impact than those of others that gained control of the region (R. Laurent, personal communication). Native Americans usually harvested small trees that would meet their needs, such as for lodge poles, and impacts from their harvesting were undoubtedly minor. Native Americans could have started fires that burned over large areas, and this may have been their greatest influence on the landscape, but such fires also could have been ignited by lightning.

Beginning in the early 1800s, trappers and adventurers began to explore the Rocky Mountain West. Human land uses probably remained fairly minor during this time, primarily because travelers in the region would follow courses around rather than through the mountains. Forts and camps were also on the plains rather than in the mountains (Murray 1980, Conner 1940), although some wood was cut from low elevation forests. Conner (1940) reported that, following the Battle of the Little Big Horn of 1876, the Sioux Indians retreated into the Bighorn Mountains and set a large forest fire. However, the age of the trees in the area does not support this claim. More likely, the fire occurred on the adjacent plains (R. Laurent, personal communication).

Because of the low density of beaver in the area, fur trading in the Bighorn Mountains was relatively minor (Martorano et al. 1985). Numerous prospectors entered the Bighorns, but none stayed long. In the 1870s, after most of the bison were gone (last one seen in the early 1880s; Murray 1980), the Bozeman and Oregon Trails brought cattle and sheep ranching into the plains surrounding the Bighorn Mountains and small settlements such as Sheridan, Story, Buffalo, and Tensleep sprang up. Minor grazing and a small sawmill operation did not begin inside the present BNF boundaries until the mid-1880s (Martorano et al. 1985).

3.3.2 European-American Settlement and National Forest Administration (1890-1945)

The second period of human occupation, 1890 to 1945, began with the arrival of large numbers of European-Americans. In the 1890s, livestock grazing increased and lumber mills and gold mining towns, such as Bald Mountain, were established within the area that would become the BNF (Martorano et al. 1985). Due to poor results, the large mines closed less than a decade later. Mining activity was marginal thereafter. From 1893 to the 1920s, the cutting of railroad ties (tie hacking) and timber for posts, poles, and fuelwood was practiced in low and high elevation forests (Murray 1980, Martorano et al. 1985). Tie hacking occurred on about 25,500 ha of the forest (0.2 ha patches were cut in some areas and over 40% of the trees were removed in other areas, R. Laurent, personal communication). Northern watersheds of the Tongue River, Wolf Creek, Sheep Creek, Big Goose Creek, and Little Bighorn River were affected (drainages 2,3,and 4 on Fig. 7; Granum 1990).

In the mid-1920s to late 1930s, tie hacking was extensive in two high-elevation watersheds in the south—Clear Creek and Sourdough Creek watersheds (drainages 7 and 8 on Fig. 7; USDA Forest Service 1997). Lodgepole pine trees of 28-45 cm diameter at breast height (DBH) (usually over 150 years old; Jack 1900) were often selectively cut for railroad ties, which left only trees too large or too small for ties and trees that had defects. About half of the wood in a tree was “wasted” (Martorano et al. 1985). Selective cutting removed the best quality trees, leaving trees of less value for lumber until the next disturbance (Howe 1997). The rate of timber harvesting gradually dwindled in the 1940s until after World War II (USDA Forest Service 1985).

To deliver logs to the sawmills from high elevations, water was diverted from streams into large wooden tie flumes built above streams (Granum 1990). Later, in the 1920s, dams were built on some streams to provide streamflow adequate to float logs downstream. Riparian vegetation was undoubtedly damaged during this time (Martorano et al. 1985, USDA Forest Service 1997).

The frequency of human-caused fires probably increased with the arrival of European-Americans, primarily due to accidents associated with saw mills and the dwellings of the workers. Ember-producing wood-burning stoves provided heat during the winter. In the summer of 1898, about 21,000 ha of the forests burned, 9,000 ha of which were purportedly caused by a shepherd's campfire near Piney and Little Goose Creeks (Town 1899). The causes of other fires that year are unknown (although Town reported rumors that Indians had set them).

From 1890 to the early 1900s, the native herbivores (elk, deer, bighorn sheep, pronghorn antelope, and bison) and furbearers (pine marten, beaver) were hunted for eastern markets and sport (Dary 1986). Some species were driven to near extinction; the elk disappeared. Populations of the native predators, such as wolves, mountain lion, lynx, and grizzly bear, were also decimated during this time by poisoning and shooting. Simultaneously, domestic livestock grazing was extensive and uncontrolled until 1899. Town (1899) estimated 3,000 cattle and 450,000 sheep on the Forest Reserve in 1898. There were many more by 1900, when Jack (1900) observed that the BNF was badly overgrazed south of the 13th standard parallel.

Concerns about grazing, timber harvesting, fire and water quality led to the establishment of the Bighorn Forest Reserve in 1897 by President William McKinley. More careful attention was given to finding the best ways to avoid depleting and destroying valuable natural resources (Pinchot 1907). Because fire was personified as the "enemy" to forest managers, fire suppression became a primary objective (Town 1899, Conner 1940). Grazing permits also were required and sheep grazing was reduced to 150,500 head in 1899. However, the number of sheep permitted to graze on the Reserve increased again to 374,734 in 1904, and the number of cattle permits increased from 3,000 to over 30,000 during the same period. Notably, the number of sheep often exceeded permitted levels, and uneven livestock distribution led to overgrazing near streams while some rangelands were left mostly unused. In 1907, approximately 16,000 ha of the Forest (22% of grasslands) were used by cattle and 24,000 ha (34% of grasslands) were used by sheep (Conner 1940, Martarano et al. 1985). The

Reserve became Bighorn National Forest in 1905, but over a decade passed before grazing became more tightly controlled and various kinds of range improvement were attempted (including fencing and the provision of water). Grazing continued to increase until 1919, when it was reduced to about 117,000 sheep and between 36,000 to 48,500 cattle. In 1931, sheep use was 126,765 head for a 2.5 month season and cattle use was 32,352 head for 3.5 months. By 2002 the number of cattle would be slightly lower (28,025) but sheep use would decline by nearly 90% to 13,830 animals.

Demand for tourism increased in the 1910s and 1920s and resort hotels and dude ranches flourished, such as Arrowhead Lodge, Dome Lake Club, Eaton Ranch, and Paradise Guest Ranch (BNF Historical Files). Road-building became a major activity on the Forest, primarily to assist tourism and fire-fighting (Martarano et al. 1985). Wildlife management improved and elk were reintroduced beginning in 1909 (Fig. 11).

3.3.3 Post-World War II (1945 to present)

The post-World War II period was characterized by new technology that rapidly increased the rate of human modifications of the landscape. The rate of timber harvesting was slow in the 1940s and 1950s (Fig. 12), but large clearcuts were common in the 1960s and 1970s (Fig. 13). New levels of prosperity created larger markets for timber than ever before and the creation of new stands of rapidly growing, economically valuable trees became a common goal (Langston 1995). The cutting of larger trees was encouraged. Gasoline-powered sawmills, some that were portable, had been in use since the 1920s, and they now became more common. Also, additional road construction occurred in the 1950s, making more of the forest accessible for fire-fighting and timber harvest. By 1960, aerial techniques improved fire-suppression dramatically (Martorano et al. 1985). Desires to harvest older stands with slow tree growth and remove stands that had been affected by tie hacking (in which the poorest quality trees were left to reproduce) made clearcutting an attractive practice. Throughout the national forest system at this time, the old forests

commonly were viewed as decadent, with the best prescription thought to be a clearcut to establish a new stand of rapidly growing trees that could be more easily subjected to thinning and other management activities (Langston 1995).

Ten of the twelve major watersheds on the BNF have been subjected to logging in the past 40 years (Fig. 13). Almost half of this logging was in the form of dispersed clearcuts. In the 1960s, clearcuts averaged 22 ha. In the 1980s and 1990s, harvested patches were generally smaller than before, averaging 8 ha (Fig. 14). Shelterwood, group, and individual selection silvicultural systems became popular in the 1970s (Fig. 15), though such methods sometimes increased the chances of windthrow (USDA Forest Service 1994).

The target rotation time for high-elevation clearcuts typically has ranged from 60 to 180 years on the BNF (USDA Forest Service 1985). Most of the timber was harvested in high-elevation forests dominated by lodgepole pine, Engelmann spruce and subalpine fir. At lower elevations, some Douglas-fir was logged (580 ha or ~3% of total wood harvested), but ponderosa pine and limber pine have been harvested on only ~14 ha (0.0007% of wood harvested) on the BNF. Harvest rotation times for low-elevation forests were set at 80 to 100 years in 1985 (USDA Forest Service 1985). Tree regeneration, especially on spruce-fir forests on sedimentary soils, often has not met the standards of the National Forest Management Act and thus, tree planting was and is required on about 30% of the land harvested in the BNF (USDA Forest Service 1994; Fig. 15). To reduce fuels or prepare sites for regrowth, thinning has occurred on 5.5% of the forested area and prescribed burning has occurred in ~2% of the forests (Fig. 15).

Domestic grazing has declined in the region during the last 60 years, due largely to a dramatic reduction in sheep grazing (Cochrane et al. 1988), from 126,765 head in 1931 to 13,830 in 2002. In the 1980s, the number of cattle grazing on the BNF was similar to the 1930s (33,000), but sheep grazing declined from about 127,000 to 58,000 head on the Forest (USDA Forest Service 1985) with the decline in the market for wool. Wild herbivores such as deer and elk increased during this period (Fig. 11). Bighorn sheep have been re-

introduced periodically (Fig. 11), but in 1999 their population was down to 20, probably due to disease transmission from domestic sheep (B. Bornong, personal communication). Moose were introduced in the 1950s, reaching a population of over 400 by 1994 (USDA Forest Service 1994). Deer peaked in the 1960s and then declined, possibly caused by lower winter habitat quality resulting from fire suppression, tree encroachment, and other habitat alterations (Leege and Hickey 1971; Hobbs and Spowart 1984; B. Jellison, Wyoming Game and Fish Biologist, personal communication). In 1984, winter carrying capacity on the Forest for deer and elk was estimated to be only 1,053 and 544, respectively (USDA Forest Service 1985). Many animals migrate off the BNF during the winter. Winter rather than summer habitat tends to limit the ungulate populations in the region (Davis 1977).

Although levels of timber harvest have declined in the last decade (Fig. 15), many human influences still affect the BNF. Imprints of timber harvesting over the last 50 years are conspicuous in some areas (Fig. 13), and clearcut or group selection harvesting is still common (though with greater sensitivity to wildlife and aesthetic values). The road system is now extensive in most watersheds away from the Cloud Peak Wilderness and steep canyons (Fig. 16). Notably, an important effect has been the influence of people from throughout the nation who are engaged in a wide range of recreational activities on the Forest. Due to accessibility and the popularity of outdoor recreation, such influences can have widespread impacts.

4. DOMINANT PLANT DISTRIBUTION AND SUCCESSIONAL PATTERNS ALONG ECOLOGICAL GRADIENTS

4.1 DOMINANT PLANT DISTRIBUTION PATTERNS.

The zonation of vegetation with elevation on the mountain ranges of the BNF is similar to other parts of the Rocky Mountains. At the lower elevations (1,320-1,840 m), ponderosa pine forests or savannas occur in the foothills of the Bighorn Mountains (Fig. 10 and Despain 1973), extending down to the alluvial fans where they are bordered by grasslands at the base of the mountains on the east and juniper woodlands on the west. Ponderosa pine forests are nearly absent on the drier, western flanks of the range, where a well-developed juniper-sagebrush plant association replaces the ponderosa pine or is situated between the pine and the grasslands below. On the western flanks, the lowest closed forest type is Douglas-fir, which grows along the north-facing sides of canyons. Curlleaf mountain-mahogany is common on rocky, calcareous soils in the foothills of the mountains, though mostly outside the Forest boundary. At slightly higher elevations on the eastern flanks of the Bighorns, from 2,000 to 2,300 m elevation, Douglas-fir forests become more common, occurring between the lodgepole pine forests above and ponderosa pine woodlands below. Higher on the mountain, from 2,150 to 2,900 m, lodgepole pine is common and creates an almost continuous forest on soils derived from granite (Fig. 17). Above the lodgepole pine, from 2500 to 2900 m, forests dominated by Engelmann spruce and subalpine fir are more common where tree growth is possible. The lodgepole pine and spruce-fir forests that occur at mid- to high-elevations are commonly known as upper montane or subalpine forests. Interspersed with the forests are extensive subalpine meadows, and above the subalpine forests is a large alpine zone, especially in the Cloud Peak Wilderness.

Soils strongly affect forest communities on the BNF. Woodlands dominated by juniper and curlleaf mountain-mahogany are common on very shallow, sedimentary soils or fractured bedrock in the dry foothills above the

Bighorn Basin (Fig. 17; Despain 1973, Knight et al. 1987). The mountain-mahogany communities are most common on calcareous soils derived from dolomite, red shales and sandstones. Douglas-fir thrives on soils derived from limestone or dolomite and is rare on soils derived from granite (Fig. 17 and Despain 1973). At the lower elevations of the Douglas-fir zone, limber pine ranges from a minor to a dominant overstory species on the dry, rocky and often sedimentary soils of windswept ridges and the sides of canyons where other species are not able to survive (Despain 1973, Nesser 1986). Limber pine tends to be found on exposed ridges and south-facing slopes, partially because the Clark's nutcracker prefers such habitats for its food caches—which are composed primarily of limber pine seed (Tomback 1983). Spruce-fir forests are best developed in the northern part of the Forest, especially on shales (Figs. 8 and 10, and Despain 1973). Lodgepole pine is more common on granitic soils and only infrequently is associated with Engelmann spruce and subalpine fir on soils derived from sandstone (Figs. 8 and 10, and Despain 1973).

An interesting feature of the Bighorn Mountains is the numerous sharp boundaries between grassland and forest, particularly in the northern part of the forest (Despain 1973). Such boundaries appear to be caused by differences in soil texture, with forests occurring on the more coarse soils and meadows on finer textured soils. The physical location of many mountain meadows and grasslands appears to be quite stable (Despain 1973). In some instances, however, changing climatic conditions or other factors have allowed tree seedling establishment on areas that were historically dominated by grasses and forbs (Jack 1900, Bornong 1996a). Similarly, lodgepole pine has invaded some subalpine meadows in Yellowstone National Park (Jakubos and Romme 1993), Douglas-fir is invading meadows in southwestern Montana (Arno and Gruell 1986), and subalpine fir and Engelmann spruce seedlings are becoming established near treeline in Rocky Mountain National Park (Hessl and Baker 1997). Of course, the converse may be true, with climatic conditions following forest disturbances leading to the development (at least temporarily) of a

mountain meadow. Thus, while edaphic conditions appear to be very important in causing forest/meadow boundaries, climate change could be important as well.

The extensive montane grasslands of the BNF tend to occur on fine-textured sedimentary soils, mostly at low-elevations with low precipitation (<2,300 m) or at high-elevations too cold or snow-covered for too long to support continuous forests (>2,750 m; Jack 1900, Despain 1973). Idaho fescue (*Festuca idahoensis*) is the most common grass from 1,840 m to about 3000 m (timberline); bluebunch wheatgrass (*Elymus spicatum*) dominates the grasslands at lower elevations. Both grasses also can be found with curleaf mountain-mahogany and big sagebrush. The relatively high precipitation on the east slope has resulted in more forest there than on the west side, even on sedimentary substrates (Despain 1973).

Aside from elevation and soils, the distribution of lodgepole pine, subalpine fir and Engelmann spruce is affected by moisture availability. Most precipitation falls in early spring in the Bighorn Mountains and is not evenly distributed (Fig. 9). Consequently, less moisture is available in the critical summer months, when seedling establishment of Engelmann spruce and subalpine fir are limited by their need for at least 2.5 cm of precipitation per month or 1.3 cm per week during the growing season (Alexander and Nobel 1971, Alexander and Sheppard 1984). Even at higher elevations, this level of precipitation often does not occur on the BNF. Only 24% of the weeks in July and August from 1979-1995 had more than 1.3 cm precipitation per week near Powder River Pass (Bornong 1996b).

Topographic position also affects moisture availability and thus species distributions. Spruce-fir forest is typical on the more mesic sites throughout the BNF, such as in ravine bottoms or on north slopes, while lodgepole pine is common on drier sites (Despain 1973). Small patches of spruce-fir forests occur in ravines at lower elevations, where cold air drainage occurs and the potential evapotranspiration is low. High-elevation landscapes often have abrupt changes in topography, with steep slopes causing many subalpine forests to occur in small patches (Knight 1994b). Occasionally white spruce (*Picea glauca*) occurs

in the subalpine forests, though it is quite rare compared to in the Black Hills to the east.

Aspen is not abundant on the BNF (Despain 1973), occurring primarily as small, clonal groves at middle and low-elevations (but with some extensive groves north of Burgess Junction). Aspen is most common on the more moist sites characterized by deeper, fine-textured soils (Hoff 1957). Aspen reproduction typically is asexual, with new shoots produced from root sprouts (suckering; Barnes 1966, Bartos et al. 1991). This, combined with the way aspen is able to persist in the understory of some mature forests, explains why aspen tends to develop where it occurred previously. Sexual reproduction is rare, though seedlings do occur when severe disturbances such as fire are followed by the extended moist conditions required for seedling establishment (McDonough 1985, Romme et al. 1997). Because of such requirements, sexual reproduction is thought to be episodic (DeByle and Winokur 1985, Romme et al. 1997). There is considerable genetic diversity between clones, with some clones better adapted for high 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.

In riparian habitats on the western slopes, narrowleaf cottonwood (*Populus angustifolia*) and Rocky Mountain maple (*Acer glabrum*) are common, particularly on soils derived from limestone. Only a few isolated stands of narrowleaf cottonwood occur on the east slope, primarily along the Tongue River and in Crazy Woman Canyon.

As with the forests, the distribution of the various shrublands on the BNF is controlled by climate, elevation, and topographic position. Mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*), for example, often occurs where moisture is available but the soils are well drained and temperatures are not too cold (Burke et al. 1989). As mentioned, curleaf mountain-mahogany (*Cercocarpus ledifolius*) is found primarily on rocky, sedimentary substrata (Knight 1994a). Bitterbrush (*Purshia tridentata*) is often mixed with mountain big

sagebrush in the foothills, especially where snow accumulates, such as in ravines. At higher elevations, in association with sub-irrigated wet meadows, shrublands dominated by mountain silver sagebrush (*Artemisia cana* ssp. *viscidula*) and shrubby cinquefoil (*Potentilla fruticosa*) are common. Willow-dominated shrublands (*Salix* spp.) are common in riparian zones.

Other types of vegetation are found only at higher elevations. Where the subalpine and alpine zones merge, subalpine fir and Engelmann spruce trees form dwarfed, shrubby patches referred to as krummholz. Just above this, the alpine vegetation is dominated by grasses, 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). The extensive alpine zone is one of the primary attractions of the Cloud Peak Wilderness, which covers about 13% of the BNF.

Plant species distributions have been changing for thousands of years and likely will continue to change in the future (Table 1). 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 change (Hansen 1940, Fall 1997, Whitlock 1993). Ponderosa pine arrived in the Bighorn Mountains about 2500 years ago (Jodi Norris, personal communication). As noted previously, HRV reports must, to every extent possible, separate climatic effects from management effects.

Some changes in vegetation distribution might have occurred due to climate change. For example, as noted, lodgepole pine, subalpine fir and Engelmann spruce have been observed establishing in formerly open meadows in the region (Patten 1969, Schimpf et al. 1980, Pearson et al. 1987, Jakubos and Romme 1993, Hessler and Baker 1997). Such forest encroachment occurs on the BNF (Jack 1900, Bornong 1996a), although not commonly (Despain 1973); and juniper appears to be expanding its range and becoming more dense in the western foothills (mostly outside the Forest boundary). Some of the more recent

observations of forest encroachment in the region have been attributed, at least in part, to warmer and wetter conditions that began in the late 1800s (Fig. 18). Of note, temperature in the BNF was higher in the year 2000 than it has ever been during the HRV period from 1600 to 1890, or even earlier to 1200 AD (Fig. 19). Nevertheless, the invasion of mountain meadows by trees, due to climate change, probably has not been widespread because of the relatively strong influence of soil characteristics in maintaining the forest/meadow boundary.

4.2 SUCCESSIONAL PATTERNS

4.2.1 *Succession in high-elevation forests*

Because of its relative intolerance for the forest understory environment, lodgepole pine traditionally has been considered a pioneer or seral species to forests dominated by Engelmann spruce and subalpine fir (Baker 1949, Lotan and Perry 1983, Lotan et al. 1985, Parker and Parker 1994). As mentioned, however, some of the BNF is too dry in the summer to support spruce-fir forests. Lodgepole pine perpetuates itself on such sites (Despain 1973). Some very old lodgepole pine trees have been found in what appear to be stable, lodgepole pine forests (~500 years old in the North Fork of the Powder River Watershed; Bornong 1996a). Reproduction occurs in these stands after the creation of canopy gaps (Despain 1983). In some areas, Engelmann spruce and subalpine fir together may be the pioneer species following a disturbance, such as Doyle (1997) observed in Grand Teton National Park. In such situations, regeneration can be very slow (Jack 1900; B. Bornong, personal communication). Although subalpine fir is the most shade-tolerant and has the greatest potential to dominate the forest, it is usually a co-dominant with Engelmann spruce because spruce lives considerably longer (fir up to ~200 years; spruce over 500 years). On limestone/dolomite substrates at the higher end of the elevational range of Douglas-fir, Douglas-fir rather than lodgepole pine can be seral to spruce and fir (Despain 1973).

Lodgepole pine is well-adapted to disturbances because it often bears cones that remain closed for many years, thereby storing large numbers of seed 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, Muir 1993, Tinker et al. 1994). The resistance of lodgepole pine seedlings to cold and dryness (Cochran and Bernsten 1973), combined with a history punctuated by disturbances over the last 150 years, help explain the prevalence of lodgepole pine on the BNF today (33% of the BNF land area; Fig. 10).

Because there is great variability in rates of forest development, 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 et al. (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, may require several centuries to develop (Rebertus et al. 1992). Mehl (1992) also developed criteria for defining old-growth forests in the Rocky Mountain region, but he did not specify that Rocky Mountain old-growth forests be at least 300 years old. Instead, he concluded that the stands should be 150 to 200 years old, depending on species, and have large trees for the species and site, high variation in tree sizes and spacing, presence of standing and downed 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 BNF managers are currently using when defining old-growth forests for the RIS database.

The BNF's forest inventory is not yet complete (only 50% of spruce-fir and 67% of lodgepole pine stands have been surveyed), so it is not now certain what

proportion of the forests are classified appropriately as "old-growth". The inventory incorporates Mehl's age and size criteria, which are that spruce-fir stands should be ≥ 200 years with trees ≥ 41 cm DBH, and lodgepole should be ≥ 150 years with trees ≥ 25 cm DBH. When the entire forest was classified with photointerpretation using tree crown size, only 5% of the forested land was classified as old-growth. This estimate probably will change when the old-growth forest inventory is completed.

The HRV for succession and successional processes during the reference 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, spruce, fir and lodgepole pine across the landscape (Stahelin 1943, Turner et al. 1997, Reed et al. 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. 18). 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 and ponderosa pine are the dominants of late-successional forests at low elevations on the BNF (Hoffman and Alexander 1976). Sometimes Douglas-fir replaces ponderosa pine as the stand matures, but most ponderosa pine stands are dominated by ponderosa pine saplings in the understory (Despain 1973). Unlike areas in the Colorado Front Range, lodgepole pine is generally not seral to Douglas-fir, because of the separation of the two species by soil type in the Bighorns (Despain 1973). Occasionally, limber pine has been

viewed as a seral species (Hoffman and Alexander 1976), though it also tends to occur on sites where the other species are less tolerant.

Based on the criteria of Mehl (1992), old-growth definitions for low elevation forests are as follows: Douglas-fir ≥ 200 years and ≥ 46 cm DBH, and ponderosa pine ≥ 200 years and ≥ 41 cm DBH. The photointerpretation for the BNF shows only 1% of the forests in Douglas-fir old-growth and another 1% in ponderosa pine old-growth (USDA Forest Service 1984). Again, these estimates may change substantially when the old-growth inventory for the BNF is completed.

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). Large conflagrations in the future due to both warming and increases in tree density and fuel loading (see Section 6.2.2 for low-elevation landscapes) could set back some stands to earlier successional stages than would have ever occurred during the HRV period. Such landscape level changes in the proportions of late-successional forest are discussed in more detail in the Landscape Structure section of this report (Section 6.2).

4.2.3 Succession in aspen woodlands

Aspen can be seral or relatively stable depending on site conditions and historical factors (Mueggler 1985), but typically it is thought of as a pioneer species (Bartos and Campbell 1998). Usually a few aspen persist in the understory of coniferous forests and produce an abundance of new sprouts when the next stand-replacing disturbance occurs. 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 understory. The clones (genets) of aspen are thought to be very long-lived, perhaps thousands of years, though individual trees (ramets) produced in the clone often do not live more than 100 or 125 years (Veblen and Lorenz 1991, Mehl 1992). Some ramets may live much longer. Aspen dynamics is discussed further in subsequent sections.

4.2.4 Succession in non-forest vegetation

As with forests on the BNF, the composition and structure of non-forest vegetation varies considerably during the time period of a century or more. In areas where big sagebrush was dominant, vegetation changes were probably quite dramatic after fire during the HRV reference period because the sagebrush is not capable of sprouting. When a fire occurs, grasses and forbs become the dominants for periods of several decades. Eventually, big sagebrush reestablishes with new seedlings, which is a common successional pattern. Some grasses, such as bluebunch wheatgrass, also may have declined following hot fires (or heavy spring grazing) (Knight 1994a). In contrast, some shrubs, such as snowbush ceanothus (*Ceanothus velutinus*), require the heat of fire to break seed dormancy (Zavitkovski and Newton 1968). Sufficiently long drought periods during the past century could cause shifts in species composition.

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. Whether or not the increase is beyond the HRV is uncertain because there is no evidence to support such a conclusion. In recent decades, the Forest Service has conducted prescribed burns to reduce the extensive invasion of woody plants such as big sagebrush (Cook et al. 1994), thereby restoring what many consider to be a natural process to the landscape (i.e., one that was common during the HRV reference period).

Two other factors must be considered when addressing succession in non-forest vegetation—the effects of livestock grazing, which was especially intense in the early 1900s, and the invasion of exotic plants that has occurred throughout the last century. The effects of grazing are discussed later in this report, but in general, range management practices are better now than at the turn of the century and rangeland condition is probably better than at that time. This has occurred through natural successional processes, including the accumulation of biomass and favorable shifts in species composition.

Undoubtedly, where exotic plants have become common, the recovery patterns have been different. Unfortunately, there has been essentially no research on this topic in Wyoming national forests. The work of Stohlgren and his associates (Stohlgren et al. 1999a, 1999b) is pertinent. They found that habitats in the Rocky Mountains of Colorado with high native species diversity 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.

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

A variety of natural disturbances occur in the Rocky Mountains, but fire is the predominant one (Clements 1910, Brown 1973, Romme and Knight 1981, Fahey and Knight 1986, Despain and Romme 1991). Occasional windstorms and insect or disease epidemics can be important as well, but fire has shaped the vegetation mosaic for thousands of years by causing clearly defined disturbances on a variety of scales. The successional processes described in the previous section have been initiated over and over, creating a diverse landscape mosaic. Although we discuss each disturbance type separately in this report, the various disturbances have been 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 1976, Lindemann and Baker 2001), and new edges 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, often have been altered by management activities. In this section we examine these changes in more

detail and attempt to characterize HRV for the major disturbances. Then we compare the HRV of disturbances to existing conditions.

5.1 FIRE

The evidence of historical fires is recorded by the annual rings of trees that survive fire events (fire scars) or of trees that become established within a few years after the fire (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 et al. 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.

When separated by forest type, ponderosa pine forests currently have the greatest number of fires. On average, 15 fires started per year per 100,000 ha in ponderosa pine forest since 1970. In contrast, only five fires started per year per 100,000 ha in the spruce-fir/lodgepole pine forests (BNF Fire and RIS databases). Various other studies also indicate that low elevation forests burn more frequently than forests dominated by lodgepole pine or spruce-fir (compare Tables 3 and 4). Of course, low intensity surface fires are typical of the low-elevation forests (ponderosa pine and Douglas-fir), while intense crown fires are more common at high elevations. The fire regimes of high and low elevations should be considered separately, but some of the fire data were not easily separated in this way. For example, many of the quantitative fire and landscape analyses for high-elevation forests are based on the entire BNF. Because the forested area is 80% at high elevations, this deficiency is not serious when considering the HRV at high elevations.

5.1.1 *Fire in high-elevation forests*

As lodgepole pine is often seral to spruce and fir, the historic variability of these two forest types is closely linked. Thus, the fire regimes for the two forest

types are discussed together. In this section we 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. In certain cases, low intensity surface fires may occur (Brown 1973, Arno 1976, Franklin and Laven 1989, Kipfmüller 1997), but often such fires become stand-replacing crown fires if spruce, fir and lodgepole pine are present. Thus, high elevation forests are affected primarily by stand-replacing fires.

Fire return interval. Fire frequency in high-elevation forests can be estimated using mean fire intervals (MFI) at two spatial scales: within a stand and across a landscape. Perhaps the most useful of such data to managers, as a guide to determining harvest rotation times, are estimates that focus on the fire return interval of stand-replacing fires (fire-free period) within a particular stand. A common approach to estimating such site-specific fire return intervals involves age-class sampling and the evaluation of stand structure and fuel accumulation (e.g., Billings 1969, Romme and Knight 1981, Romme 1980a, 1982, Romme and Despain 1989; Kipfmüller and Baker 1998a, 2000). As a stand ages, the amount and continuity of fuels increases and stand-replacing fires become more likely. Based on this approach, the fire-free interval for a particular stand of subalpine forest varies commonly from 150 to 700 years (Table 3). Of course, shorter intervals have been observed as well.

Average fire return intervals have also been estimated by counting the rings on fire-scarred trees (Hawkes 1980, Bornong 1996a, Kipfmüller 1997, Kipfmüller and Baker 1998a). Such estimates are often low because they consider the occurrence of all fire events, not just stand-replacing fires. Such average fire scar intervals for high-elevation forests in the central and northern Rockies range from 39 to 304 years, depending on site characteristics, species composition, and the method of calculation (Table 3). The longest fire-free intervals tend to be associated with mesic sites found in ravines, high elevations, leeward or north slopes, and on unusually dry sites where fires may not occur at all (Billings 1969, Hawkes 1980, Romme and Knight 1981, Tinker et al. 1994)—

though 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 in the Medicine Bow National Forest.

The second way of characterizing fire frequency is to calculate the MFI for a large landscape that encompasses many stands, rather than just one stand (e.g., Veblen et al. 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). The MFI for a high-elevation watershed on the BNF during the HRV period was estimated to average 13 and 32 years for lodgepole pine and spruce-fir forests, respectively (Bornong 1996a; Fig. 20). Because this statistic is area dependent, larger study areas typically have more fire events, leading to lower MFI estimates (Arno and Peterson 1983, Kipfmüller 1997). 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 changes in fire ignition over time within one location.

Fire size and extent. Insights into the size of fire events and the occurrence of large fires can be gained by considering both long-term and recent fire histories in Yellowstone National Park (YNP), which is similar in latitude, topography and vegetation to the BNF and has remained relatively undisturbed by human influences. Thus, despite geologic differences, it provides a reasonable reference area—indeed, one of the only reference areas that is appropriate for the BNF and where a considerable amount of research has been done.

At the landscape scale, Despain and Romme (1991) reconstructed the fire history for a 129,600 ha study area in southcentral YNP (approximately one-fifth the size of the BNF). 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. 21). Similarly, Barrett (1994) found that major fires occurred over four periods within

81,000 ha in northeastern YNP (1740-1750, 1770s, 1820-1830, and 1980s; Fig. 21), but not always during the same decades identified by Despain and Romme. The periods identified by both studies often corresponded to very dry or very hot periods (Fig. 18). The time between such periods of major fires does not represent the interval between fires in a specific stand, but rather between major fire events somewhere in the study area (fire frequency on the landscape).

During each major fire period, a large proportion of the entire area burned (7-26% in the decade with the maximum area burned in southcentral YNP, 11-58% in northeastern YNP), creating extensive, relatively even-aged forests. Despain and Romme (1991) concluded 1) that such even-aged cohorts progress into later successional stages and become increasingly susceptible to burning as fuel loading and continuity increase, and 2) that the same area burns again after another 200-400 years whenever weather conditions permit. Lightning ignitions are frequent (Fig. 22) but usually do not burn more than a few hectares (Despain and Romme 1991). Most of the area burned during a century is attributable to only a few large fires (Figs. 23 and 24).

Since 1972, the fire management policy in YNP has been to allow lightning-caused fires to burn in most instances, providing a proxy for possible dynamics in the HRV period. Only during a few years from 1972-1988 were weather conditions dry enough to allow more than 100 ha to burn (Fig. 24).

Considering 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 (Fig. 23). The 1988 fires are one of the five major fire events of the past 300 years, and they burned more land area than the other large fires combined over decades (Figs. 21 and 24). Possibly this was due to the warmer conditions in the latter half of the century than had occurred during the HRV period. The driving factors that create this fire regime appear to be weather conditions and the type and distribution of fuel (Despain and Romme 1991).

Using YNP as an HRV proxy. When determining if the YNP fire regime can be used as a proxy for the high-elevation forest of the BNF, differences between the climate and vegetation types of the two areas must be evaluated. The BNF tends to be somewhat drier and warmer than YNP (Fig. 9), which could create flammable conditions more frequently; but the BNF also has less forested area and less land in highly flammable forest (Despain 1990), which reduces fire potential. The mean number of fire starts per year is similar between the two areas (Fig. 22). Similar to YNP, the majority of the forests in the BNF are high-elevation forests. Both YNP and BNF had relatively large fires in 1988, but only about 1,600 ha burned on the BNF per 100,000 ha of land that year, compared to 44,200 ha per 100,000 ha of land in YNP (Fig. 24). Despite major fire suppression efforts, the largest fire on the BNF in 1988 (5,300 ha), started by lightning, stopped only due to insufficient fuel as it headed up into the rocky Cloud Peak Wilderness (Fig. 25), where the fire probably would have extinguished naturally. The topography in the southern portion of the BNF may limit fire size more than in much of YNP.

Due to more rocky barriers and less continuous forests, one might think that the BNF never had the very large fires that burned in YNP during the HRV period. Yet, YNP had the same number of large fires as the BNF from 1900 to 1970 when fire suppression was practiced in both areas (three that were > 500 ha/100,000 ha), suggesting that large-fire dynamics may be comparable (Fig. 24).

Clearly, however, more data are needed on the behavior of large fires without fire suppression on the BNF before we can conclusively state that the HRV for the YNP reference area is adequate for comparing HRV fire size and frequency in the BNF. Notably, evaluation of the stand ages of the BNF suggests extensive and large fires did create large areas of more or less even-aged lodgepole pine forest during the HRV period on the drier portions of the BNF (Figs. 26 and 27).

To further examine the similarity between YNP and the BNF, we examined stand age in lodgepole pine forests over time and found that the majority of the

forest on the BNF originated in the 1890s (Fig. 27). Although some of the burned areas were ignited by early settlers, many large areas had burned before that time (Figs. 27 and 28). Estimated lodgepole pine stand ages during the HRV period on the BNF (Fig. 27), compared to lodgepole pine/spruce-fir forests in YNP (Fig. 26), suggest that large fires occur periodically in both areas and that differences between YNP and the BNF in topography and forested area are not sufficient to negate the use of YNP as a proxy for the BNF.

Changes from HRV. Due to insufficient data, we could not evaluate quantitatively the changes in fire regimes that might have occurred at the stand level. However, our qualitative assessment suggests that the mean fire return interval of stand-replacing fires in the last 100 years is still within the range of means during the HRV period, despite fire suppression efforts (Table 3; low confidence (Morgan et al. 1996). Fire intensity probably has not changed either, as the forests are still dominated by stand-replacing fires (high confidence).

At the landscape scale, a study of lodgepole pine above 2670 m elevation in the North Fork of the Powder River (Bornong 1996a) found the mean landscape-level fire interval (MFI) more than doubled after European-Americans arrived (after 1877) (Fig. 20). For his analysis, Bornong combined fire scars of trees that survived stand-replacing fires (at the edge of eight stands). His mean of current fire intervals since European-Americans arrived was greater than 75% of the HRV mean, indicating that the current intervals probably are longer than the HRV (low confidence). Recent fire records for the BNF suggest that fires still play an important role on the BNF, but fire suppression efforts have lengthened the MFI during the last 50 years. Notably, Kipfmüller (1997) found that the MFI was shorter after the arrival of European-Americans in the 1860s, when he investigated a slightly smaller watershed in the MBNF.

Comparing the BNF to YNP, the mean and standard deviation of total area burned per decade on the BNF currently is lower than what appears to be the historical conditions in YNP (1690-1890; Fig. 29). Additionally, the average area burned/year/ha on the BNF was less than half the area burned during years when fire suppression was not practiced in YNP, even when 1988 was excluded

(1972-1987; Fig. 23). Construction of fire breaks in the BNF probably facilitated the reduction of fire sizes. Moreover, mop-up efforts can completely extinguish any potential for flare-ups several days later. In contrast, areas with unregulated fire during the HRV period, such as in YNP, can wane and then flare up again, enlarging the original area burned and creating very large fires. Recent clearcuts without much tree growth can also act as fire breaks, plus roads increase accessibility for fire suppression crews. Both factors must have contributed to the reduction in the number and size of fires on the BNF.

Because many fires are apparently put out quickly, any increase in ignition from human activities may not be important when balanced against the effects of aggressive fire suppression, climate change, and fuels, as discussed previously. Notably, the median size of all fires on the BNF was larger than the median size of fires in YNP during the era of suppressing fires, wherever they occurred (Fig. 23). Definitive conclusions are not possible, but the data suggest that fire suppression plus removal of fuels by timber harvest on the high-elevation forests of the BNF may have reduced the number of large fires but not the mid-sized fires (Fig. 30).

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 the largest fires. Weather (or fuel) conditions on the BNF may not have been conducive to large fires during the early and mid-1900s, when conditions on the Bighorns were relatively wet (Figs. 18 and 19). By the late 1900s, temperatures had become the warmest since 1200 AD. 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). Warmer conditions may have triggered the big fires in 1988. The fact that fire suppression on the BNF is successfully reducing the size of fires relative to YNP despite this warming strengthens the argument that the fire regime has changed.

It is possible that the reduced incidence of large fire on the BNF during much of the last century has occurred because many high elevation-forests on the BNF have been in a relatively inflammable stage of succession, rather than because of fire suppression or timber harvesting. Today, many stands are still only about 100-150 years old (Fig. 26), and they probably will become more flammable over the next 100 years, as various reports on forest health predict. However, research suggests that large stand-replacing fires are more likely to occur because of weather conditions than fuel accumulations (Bessie and Johnson 1995, Weir et al. 1995). Romme and Despain (1989) suggest that, while extensive fires could have occurred in YNP since about 1930 based on fuels, they did not burn until 1988 because of weather conditions. The relatively low amounts of fuel on the BNF at the present time must have some influence on fire occurrences, but probably not as much effect as more frequent periods of drought with strong winds. Baker (2002) also recognized the importance of climatic conditions, but concluded that climate, fuels and elevation are linked in determining the frequency and extent of fires in the Rocky Mountain region.

Dealing with the large-scale disturbances caused by fire in high-elevation forests presents a challenge to forest managers. On one hand, disturbances of the magnitude of the 1988 Yellowstone fires appear to have had very important ecological effects (e.g., Christensen et al. 1989), and 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 also is an extreme in the HRV that the public hesitates to support (Swanson et al. 1993, Manley et al. 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 partially-harvested landscape reduces fire sizes, this conclusion is still quite controversial. Many fires, whether started by lightning or humans, certainly have been put out. However, without the wind and drought that accompanies the large fires that cannot be extinguished, the suppressed fires might not have burned over a large area anyway. There is little basis for arguing that fire suppression in YNP from

1872 to 1972 had a big effect on the magnitude of the 1988 fires there, except at lower elevations in the Douglas-fir woodlands in the northern part of the park (where fire history is more similar to the ponderosa pine forests). We have no records of fires during the reference period burning as much land area during a single year as occurred in 1988 in YNP, but just as much land burned during two decades around 1700 (Romme and Despain 1989). Whether or not fire suppression in YNP and the BNF has been effective in high-elevation forests is still debatable. More research is needed to be confident.

It is probable that big fires will once again burn across the BNF, 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 (5-25%) of the landscape approximately once each century. Where landscape patterns and stand structures are modified by humans, as in the BNF, 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 et al. 1995, Baker 2002). 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 is the frequency of surface fires burning through the understory of high-elevation coniferous forests, 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 low branches on 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 a relatively open understory. Surface fires in lodgepole pine have been observed, and because of the thin bark, some trees are killed. Others survive but develop fire scars, sometimes multiple scars. Data presented in Table 3, and in Lotan et

al. (1985), suggest that surface fires may have burned through lodgepole pine forests in the BNF at intervals as short as 40 to 80 years—considerably shorter than the 200+ intervals usually reported for YNP. Our impression is that stand-replacing crown fires are most common in all high-elevation forests, but that some are burned periodically by surface fires—especially on the edges of crown fires where flame lengths (a measure of fire intensity) are shorter.

5.1.2 Fire in low-elevation forests

Fire intensity. The most common fire regime during the HRV period in ponderosa pine and Douglas-fir forests (which comprise only 10% of BNF (9% Douglas-fir and 1% ponderosa pine) but which are common on adjacent lands) is characterized by 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 in other parts of the Rocky Mountains describe "open and park-like" stands of ponderosa pine with understories rich in grasses, such as in the American Southwest (Cooper 1960) and in Colorado (Veblen and Lorenz 1991). Photographs from the 1920s to 1950s show open, park-like stands of ponderosa pine forests on the east slope of the Bighorn Mountains (historic photos on file at the BNF). Notably, some lumber was taken from these ponderosa stands in the late 1800s (Jack 1900), which may have contributed to the low tree-density observed in photos taken after that time.

In addition to surface fires, there is evidence of fires that kill groups of trees, creating openings. Kaufmann et al. (2000) refers to a fire regime with both fire types as a mixed-fire severity pattern. The mixed-fire severity pattern probably occurred on the BNF during the HRV period. As discussed below, stand-replacing fires could have occurred also.

Fire return interval. Mean fire return interval estimates for stands at low elevations in portions of the Black Hills and Front Range of Colorado have generally been on the order of 15-45 years (Laven et al. 1980, Fisher et al. 1987, Goldblum and Veblen 1992, Brown 1994, Brown and Sieg 1996), compared to less than 10 years in the Southwest (Weaver 1951, Dieterich 1980, Fule et al.

1997; Table 4). In Wyoming, ponderosa pine fire scars indicate that, during the HRV reference period, a median fire interval of 26 to 33 years occurred in the stands of the Laramie Range (Brown et al. 2000), and that a mean fire interval (MFI) of 14 to 27 years occurred at Devil's Tower National Monument (Fisher et al. 1987; Table 4). Notably, the range of fire intervals during the HRV period can be quite broad in Wyoming and Colorado (e.g., range of 3-161 years, Laven et al. 1980 in Table 4; 8-82 years for Ashenfelder Basin, Fig. 31), though Baker and Ehle (2001, in press) concluded that such ranges are biased toward shorter intervals.

The fire return intervals vary widely among studies partly because of differences in methodology (Veblen 2000) and partly because of differences in climate, fuel conditions and topographic position. Most likely, warmer and drier sites (or years) experience more frequent fires than areas where (or when) conditions are cooler. Veblen et al. (2000) conducted a detailed study of tree rings in the ponderosa pine forests of the Colorado Front Range, finding that alternating wet and dry periods with a 5-year periodicity is conducive to fire spread. Wet periods, believed to be caused by El Niño conditions, allowed the production of fine fuels that increased flammability during subsequent dry periods associated with La Niña. The effect of the El Niño/La Niña oscillation, however, may not be as strong in areas north of the Colorado Front Range (Veblen 2000). Another factor causing variability could be livestock grazing, which reduces the abundance of fine fuels and the likelihood of a fire extending into adjacent forests.

Because most of the ponderosa pine in the Bighorn Mountains is on land outside the BNF boundary, the predominant forest type at low elevations within the national forest is dominated by Douglas-fir (9% of the BNF). Houston (1973) found a MFI of 20-25 years in Douglas-fir stands during the HRV period in YNP. Where Douglas-fir and ponderosa pine stands are located near each other in Colorado, fire intervals for stands of Douglas-fir appear to be slightly longer (Goldblum and Veblen 1992), possibly due to the occurrence of Douglas-fir on somewhat more mesic sites.

A factor complicating our ability to interpret what may be natural in lower elevation forests and woodlands 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 lower elevations (e.g., Barrett and Arno 1982, Gruell 1983a) and were inseparable from "natural" fires (Martin and Sapsis 1991). Fisher et al. (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 (Table 4).

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 nearby Devil's Tower National Monument (Fisher et al. 1987), fires caused by individual lightning strikes (i.e., spot fires) were historically common, but typically they would damage only one or two trees before burning out due to weather conditions or fuel discontinuity. Larger fires were less frequent, occurring only at times when weather and fuel conditions facilitated fire spread. In the Southwest, where summer conditions are typically warmer and drier than Wyoming, large surface fires in the 1800s averaged only 1,300 ha (Swetnam 1990). In parts of the Black Hills, the fires apparently could be larger and more intense (Shinneman and Baker 1997), like those in lodgepole pine forests. 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 1890 have been less than 9,000 ha. The exception was the intense, stand-replacing Jasper Fire in 2002 that burned 33,850 ha (83,580 acres). Notably, the few ponderosa pine woodlands and savannas in the BNF and nearby are patchy. This could have limited the spread of fire to smaller areas than in the Black Hills, except where the trees were separated by flammable grasslands.

Changes from HRV. Surface fires are much easier to suppress than the crown fires that develop so rapidly at higher elevations, and thus the change in fire regime from the HRV tends to be more noticeable at low elevations. The mean fire return interval for low-elevation ponderosa pine stands at Devil's Tower appears to have lengthened from a range of 14 to 27 years before 1900 to 42 years after 1900, possibly due to fire suppression (Fisher et al. 1987, Table 4). Fire intervals also are thought to have lengthened beyond the HRV in the nearby Ashenfelder Basin just north of Laramie Peak in the Medicine Bow National Forest during the last century, with the last fires recorded by Brown et al. (2000) occurring in 1911 and 1909 at two sites within the Basin (Fig. 31). The resultant 86- and 88-year intervals since the last fire are four and six years longer than any recorded at these sites in the 1800s. In Colorado, Brown et al. (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. Brown et al.'s results showed that the number of widespread fires per unit area (fire frequency) in the past century did not fall within the range of mean fire intervals during the HRV period. This kind of evidence suggests that fire suppression has been effective and that fire size as well as mean fire return interval has exceeded the HRV in low elevation forests (moderate confidence).

Similar to high-elevation forests, the fire regime on the BNF at the lower elevations since 1970 has been dominated by numerous small fires with relatively few widespread fires (Figs. 32 and 33). From 1970-1996, 93% of all fires in the forests at low elevations on the BNF were < 4 ha, while only two fires were larger than 125 ha and none were larger than 400 ha (BNF fire database). Although such fire sizes are much smaller than the averages of >1300 ha reported for the Black Hills (Shinneman and Baker 1997), the historical fire sizes in the less extensive, patchy ponderosa pine and Douglas-fir stands of the Bighorn Mountains is not known.

With some exceptions (e.g., Goldblum and Veblen 1992, Table 4), fire suppression apparently has lengthened the mean fire return interval in many

ponderosa pine forests in the Rocky Mountains. Extremely long intervals still occurred in the HRV period, however, because the ranges of post-settlement intervals often fall within the ranges of pre-settlement intervals (Table 4). Probably the same pattern has occurred in Douglas-fir stands, although data comparing pre- and post-European-American settlement are lacking (Table 4). Some research suggests that such ranges are too narrow and are biased toward shorter intervals (Baker and Ehle 2001), but frequent, low-intensity fires probably characterized the scattered low-elevation forests and woodlands of the BNF. Grazing removes herbaceous fuels and may further increase mean fire intervals after the introduction of livestock. Because the 20th century probably was warmer than the HRV reference period, fire intervals should be shorter, not longer than the HRV based on climate alone.

At the landscape scale, no data are available on fire frequency in the BNF during the HRV period. However, surface fires were probably pervasive throughout the low-elevation forests and attempts at fire suppression have become increasingly more effective. Thus, mean fire return interval at the landscape scale probably is also outside the HRV (high confidence).

The extent of area burned per year also probably was reduced below the HRV on low-elevation forests of the BNF (low confidence). The total area burned on low-elevation forests during the entire period from 1970 to 1996 on the BNF was no more than 2,000 ha (of 53,676 ha of low-elevation forest; BNF Fire database). Additionally, based on stand ages, fires in the low-elevation forest on the BNF appeared to be common in the 1880s, prior to major European-American settlement. Although the percent of low-elevation forests sampled for stand age was low (26-34%), the data suggest that many of the low-elevation forests (ponderosa pine, Douglas-fir, and limber pine) have trees that originated in the late 1800s and that fewer originated in the 1900s (Fig. 27, although over 400 ha of ponderosa pine originated in the 1970s).

In general, because of successful fire suppression in lowland forests, many areas of ponderosa pine in the Rocky Mountains that were once maintained in 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 et al. 1997). Historic photos throughout the 1900s show that ponderosa pine woodlands in the BNF have increased in density as well. Additional factors such as livestock grazing may 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). However, trampling may 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 ponderosa pine density and fuel loadings (Savage et al. 1996). As a result, intense stand-replacing fires in low elevation forests generally have become more common in recent years, regionally, than they were during the 18th and 19th centuries (Agee 1997). Notably, two large crown fires occurred in ponderosa pine forests in the Laramie Peak area of the Medicine Bow National Forest in 2002—the Hentzel Fire that burned 6,016 ha (14,855 acres) and the Reese Fire that burned 7,633 ha (18,848 acres).

Some surface fires are now prescribed on the BNF for low-elevation forests and non-forested vegetation types (B. Bornong, personal communication), and such burns can help restore the fire regime to the HRV. Without such surface fires, stand-replacing fires will occur. Stand-replacing fires probably are outside the HRV for the low elevation forests on the BNF (high confidence).

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 southcentral Wyoming (burned in 1841). It is unclear if these large stands developed largely from sprouts (ramets) or seedlings (genets) and what trees dominated the vegetation prior to the fires (von Ahlefeldt and Speas 1996,

Severson 1963). On the BNF, fire has probably played a similarly important role in the establishment of new aspen stands (Brown and DeByle 1989, Romme et al. 1997).

Fire frequency. Mean fire return intervals for aspen groves during the HRV period probably essentially are the same as for the other forest types with which they occur because, while aspen is not considered to be highly flammable, the stands often burn when the adjacent coniferous forest burns (DeByle et al. 1987, 1989). Veblen et al. (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. If fires are of high intensity, some aspen root systems may be killed, thus favoring development of conifer stands (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 or ramets) is killed (Veblen and Lorenz 1991). The abundance of aspen surely has fluctuated with the frequency, size and intensity of fires driven by climatic conditions, although grazers may also have had an effect (see Sections 4.2.3, 6.1.3, and 6.2.2).

Changes from HRV. Most aspen stands that were surveyed on the BNF appear to have originated during two periods, namely, 1880 to 1920 and 1980 to the 1990s (Fig. 27). The extensive fire disturbances during the late 1800s (1870-1890) and early 1900s may have caused aspen seedlings or sprouts to establish at that time, but it is difficult to determine if they were previously common because aspen shoots (the ramets) often live only about 100 years (although much older stems have been found and the aspen root system lives much longer and is capable of sprouting). Fire suppression since then probably has reduced fire frequency and extent because aspen regeneration has declined in some areas (Shepperd et al. 2001). Bornong (1996a) documented a recent decline in the extent of aspen in the watershed of the North Fork of the Powder River,

where clones are now usually less than one hectare in size. Across the Rocky Mountains, there is considerable variability in aspen regeneration, with stands in Utah apparently declining (Bartos and Campbell 2001) more so than in some other areas (Shepperd et al. 2001).

The cause of the recent aspen regeneration in some parts of the BNF after 1980 is likely due to mechanical treatment (clearfelling) as well as the increase in fires in the late 1980s (Fig. 24). However, the stand age data for aspen are limited (28% surveyed) and the age distribution data (Fig. 27) may be biased. Similar to adjacent coniferous forests, the mean fire interval is longer and the mean fire extent is lower in aspen forests today, at both the stand and landscape scales, than during the HRV reference period (moderate confidence).

5.1.4 Fire in non-forest vegetation

As noted previously, the composition and structure of non-forest cover types is affected by fire, especially where the non-sprouting big sagebrush occurs and where the natural fire regime has been altered by fire suppression and grazing. The Forest Service has conducted prescribed burns to reduce encroachment of woody plants (Cook et al. 1994), which may help maintain the fire regime of some areas within the HRV. Although fire suppression may be causing increases in shrub cover and tree encroachment, such changes probably are not beyond the HRV at the stand or landscape level (low confidence). Fires are still very frequent in non-forested habitats and have burned areas up to 400 ha annually from 1970 to 1996 (BNF fire database; Fig. 33).

Cheatgrass (*Bromus tectorum*) is rare on the BNF (S. Gall, BNF Range Conservationist, personal communication). Rapid increases in cheatgrass could cause an increase in the mean fire return interval of some sites to beyond the HRV and a decline in the abundance of some native plants (Knight 1994a, Morgan et al. 1996) (high confidence). More so than in forest vegetation, dramatic changes in non-forest vegetation could occur in the future if exotic plants become 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*) and mountain pine beetle (*Dendroctonus ponderosae*) are capable of reaching epidemic population sizes in spruce-fir, lodgepole pine, and ponderosa pine forests. 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 et al. 1994). No studies of this nature have been done on the BNF, 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 (e.g., *Ips* spp.) do not appear to be a significant factor in the lodgepole pine forests of the BNF, at least after European-Americans arrived. Referring to conditions during the European-American settlement period, a 1906 report on the Bighorn Reserve states that "insects have never made serious or extensive ravages on lodgepole pine in this reserve. Bark borers kill or assist in killing scattered trees, but their work is unimportant" (Coolidge 1906). Recent investigations have found that cold, high-elevation environments like much of the BNF may inhibit mountain pine beetle epidemics, apparently due to higher mortality of dormant larvae under the bark (Amman et al. 1977, Amman 1989). Indeed, few lodgepole pine trees have been affected at high elevations on the BNF (USDA Forest Service 1985). Nevertheless, beetle epidemics may occur from time to time in some warmer, low- to middle-elevation lodgepole pine forests, particularly in older stands (Amman et al. 1977, Amman 1989).

The mountain pine beetle survives in the phloem of the inner bark throughout most of its life cycle (McCambridge et al. 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. 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 et al. 1987).

Mountain pine beetle epidemics typically start in large diameter trees that are diseased or stressed. Therefore, most outbreaks occur in stands that are at least 80 years old, with trees at least 20 cm diameter at breast height (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 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 et al. 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. Notably, the total productivity of the forest can return to pre-epidemic levels within 10-15 years (Romme et al. 1986).

Hazard ratings have been developed for the mountain pine beetle based on tree diameter, age, and location by latitude and elevation (Amman et al. 1977). Stands with diameters of <18 cm DBH, 18-20 cm DBH, and >20 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, 2290-2590, and <2290 are also of low, moderate, and high susceptibility, respectively. Thus, older stands of large lodgepole pine on the eastern side of the BNF, where elevations are lower, are of highest susceptibility.

The historic frequency and severity of mountain pine beetle 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; 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 appears to be highest in the northern Rockies, where millions of trees have been killed in a single year (Klein et al. 1979), although outbreaks also have been large in Colorado (Schmid and Mata 1996). At the latitude of the BNF, Amman et al. (1977) estimated that lodgepole pine forests up to 2,500 m elevation are susceptible to only 25 to 50% mortality, while those above 2,500 m 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 (Amman 1977), as is common throughout the BNF.

The fact that mountain pine beetle epidemics are more frequent at lower elevations suggests a relationship with warmer temperatures. Thus, it is plausible to suggest that epidemics have been more common with climate warming during the last century (Fig. 19) and that they will become more common if this warming trend continues.

Spruce beetle. As with mountain pine beetle, the spruce beetle has not been an important cause of disturbance on the BNF during the last 100 years (Allen and Harris 1999), though it has caused heavy spruce mortality in Colorado and central New Mexico (e.g., Baker and Veblen 1990, Veblen et al. 1991a and b, Veblen et al. 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 et al. 1965).

As no specific information is available for the BNF, 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 et al. 1965, Schmid and Mata 1996). Historically, it appears that spruce beetle epidemics kill anywhere from 5 to 99% of large spruce trees (greater than 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 et al. (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 amount of time (Veblen et al. 1994).

Susceptibility of spruce stands on the BNF to future spruce beetle epidemics is uncertain. Endemic populations are normally found in windthrown or otherwise fallen spruce trees (i.e., logging residue). The 1940s epidemic in Colorado was triggered by an extensive blowdown in 1939 (Hinds et al. 1965). However, the large blowdowns on the BNF in the mid-1900s and early 1990s have not led to the expected spruce beetle outbreaks (1953 BNF historical report; Allen and Harris 1999). It is possible that an outbreak could still result from the 1993 blowdown, but that is unlikely after a lapse of 10 years (K. Allen, personal communication). Forest susceptibility to spruce beetle epidemics has not been defined as precisely as for mountain pine beetles, but Schmid and Frye (1976) outlined the following 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. The effects of a spruce beetle epidemic on the BNF would likely be similar to a mountain pine beetle epidemic, with the creation of snags, coarse woody debris, and fuel continuity. Flammability would increase as well. 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).

Western balsam bark beetle and "fir decline". In conjunction with other disease agents, the western balsam bark beetle (*Dryocoetes confusus*) appears to 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 attacks subalpine fir primarily. Root disease

pathogens such as *Armillaria* (*Armillaria* spp.) and *Annosus* (*Heterobasidion annosum*) may contribute to what has been referred to as a regionwide decline in subalpine fir (Morrison 1981, Wargo and Shaw 1985). To date, the exact cause and disease agents responsible for the mortality of subalpine fir are unknown (western balsam bark beetle and *Armillaria* are suspects), but the decline is obvious in numerous and extensive areas on the BNF (Allen and Harris 1999). Affected areas typically are circular patches of several dead and dying trees, with some that have fallen over. Young seedlings and saplings growing in the area are also dead and surrounding trees have poor growth (Shaw and Kile 1991). Because so little is known about the cause of the fir decline, interpreting its HRV has not been done.

Other insects. Numerous 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 BNF (Allen and Harris 1999). The western spruce budworm (*Choristoneura occidentalis*) may attack Engelmann spruce and Douglas-fir (Allen and Harris 1999). Such insects also cause mortality or reduced vigor in individual trees or small groups of trees, thereby contributing to the structural diversity of a forest in the same way that they have for centuries.

Changes from HRV. Given that outbreaks were rare before clearcutting and continue to be rare, there is no evidence that the frequency of mountain pine beetle or spruce beetle outbreaks is outside the HRV (moderate confidence). It is unknown if the suspected causes of the fir decline (western balsam bark beetle and associated root diseases) are outside the HRV, but its increasing and widespread prevalence is of concern to BNF managers (C. Thomas, personal communication). Schmid and Mata (1996) suggested that insect populations and their effects in the southern Rocky Mountains are currently within their respective HRVs, with some important caveats. Specifically, measurements of epidemics are imprecise, making the comparison of past, present and future epidemics difficult. However, it is clear 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. Both management activities, combined with climate change, could cause epidemic frequency and severity to move either below or above historic levels in the future (Schmid and Mata 1996).

5.2.2 *Insects in low-elevation forests*

As at higher elevations, the primary insects affecting low elevation forests on the BNF are the mountain pine beetle and the western spruce budworm (Allen and Harris 1999). The Douglas-fir beetle (*Dendroctonus psuedotsugae*) may be present in scattered areas, though its effects so far have been relatively minor (Allen and Harris 1999). The introduced gypsy moth (*Lymantria dispar*) has been observed in Wyoming, but no populations of this insect are known to have persisted in the state (K. Allen, personal communication).

Mountain pine beetle. A fluctuating population of mountain pine beetles has killed large numbers of ponderosa pine along the east face of the Bighorns, with serious outbreaks in the 1950s and the 1970s when an estimated 6,000 or more trees were killed. From 1961-65, trees were treated with chemicals to reduce mountain pine beetle infestations (Allen and Harris 1999). Areas of concern in the 1980s include the Sand Turn, Story, Red Grade, and Hospital tracts (USDA Forest Service 1985). The majority of the infested trees in these areas were not on the BNF but the epidemic was of considerable concern to local residents. Additionally, many limber pine trees were killed by the beetles in Tensleep Canyon from 1991 to 2000 (C. Thomas, personal communication).

As in lodgepole pine forests, epidemics in stands of ponderosa pine often start when the beetles kill small groups of 3 to 10 of the larger trees (Schmid and Mata 1996). If sufficient stresses are present in the “starter trees” (e.g., old age, water stress, or disease), the epidemic may spread across the landscape, killing groups of over 100 trees (Schmid and Mata 1996). The results of such outbreaks can cover large areas of the landscape, reduce the average tree diameter in a stand, and create small canopy gaps as well as large openings

(Schmid and Mata 1996). As beetle-killed trees become snags and downed logs, they contribute to structural diversity (and therefore habitat diversity), at the same time providing fuel for the next fire (Lundquist 1995a, Schmid and Mata 1996). Other species not susceptible to mountain pine beetle, such as Douglas-fir, may become more dominant (Schmid and Mata 1996). Based on the hazard rating developed for the Black Hills (Schmid and Mata 1992, Schmid et al. 1994), large, old trees (18-33 cm DBH) in dense stands with $>28 \text{ m}^2$ of basal area/ha are very susceptible to mountain pine beetle outbreaks.

Too little research in the BNF or nearby has been done to estimate epidemic frequency and severity, but mountain pine beetle outbreaks in the Southern Rocky Mountains have been more severe and more frequent in ponderosa pine than lodgepole pine—perhaps because of warmer winters at lower elevations where ponderosa pine occurs. On the landscape scale, Schmid and Mata (1996) estimate that an epidemic occurred in Colorado every 11 to 20 years in ponderosa pine 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). Duration of an individual epidemic varies from 2 to 5 years for short-lived epidemics to 7 to 14 years for long-term outbreaks (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 et al. 1979). In the Black Hills, historical reports indicate that 50,000 ha of mostly ponderosa pine were killed by a mountain pine beetle epidemic in 1895 (Shinneman and Baker 1997).

Western spruce budworm. The western spruce budworm is the most destructive defoliator in western forests. Its main hosts include Douglas-fir, with spruce and subalpine fir of lesser preference (Van Sickle 1985, Schmid and Mata 1996). In the spring, the larvae feed on buds and new needles, 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 frequency of western spruce budworm epidemics on the BNF at the stand or landscape scale in the past century is generally unknown. In Colorado and New Mexico, where most research on this insect has been done, the average outbreak intervals during the HRV period are similar to existing conditions there, ranging from 20 to 33 years (Swetnam and Lynch 1989, 1993) in individual stands. The epidemics last an average of 11 years (Lynch and Swetnam 1992, Swetnam and Lynch 1993). Records for the northwestern part of the BNF indicate that the budworm defoliated about 640 ha of Douglas-fir forest in 1985 and 400 ha in 1994 (2.5% of the Douglas-fir over 10 years). Outbreaks have also been observed over approximately 3,000 ha of ponderosa pine woodlands near Buffalo, Wyoming.

Douglas-fir beetle. As with other bark beetles, the Douglas-fir beetle produces one new generation of young per year, with most new attacks by the flying adults occurring in the late spring to early summer (McMillin and Allen 2000). Broods develop under the bark and overwinter as either adults or larvae. Attacks of this beetle 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 percentage of the stand that is Douglas-fir, were most important in determining the level of tree mortality.

Typically, the Douglas-fir beetle reproduces in scattered trees that are highly stressed, such as windfall, defoliated or fire-scorched trees (Lessard and Schmid 1990, McMillin and Allen 2000). If enough stressed trees are available, beetles can increase and infest nearby healthy trees (Furniss et al. 1981). Thus, Douglas-fir beetle epidemics frequently occur after extensive fires, which create large areas of weakened Douglas-fir trees (McMillin and Allen 2000). Such

epidemics have not yet been observed on the BNF, possibly because extensive fires in Douglas-fir forests have not occurred there (Fig. 33).

The Douglas-fir beetle can cause widespread mortality of Douglas-fir in the Southern Rocky Mountains. Outbreaks have been observed to occur every 15 to 35 years in the same stand and they last from 5 to 10 years (Hadley and Veblen 1993, Schmid and Mata 1996). Outbreak frequencies at the scale of larger landscapes are unknown, but 7,569 ha on the Shoshone National Forest (5% of the Douglas-fir on the Forest) were infested from 1992 to 1999 (37,548 trees were killed, including 58,627 m³ of timber; McMillin and Allen 2000).

Changes from HRV. For low-elevations forests, ponderosa pine has likely evolved with the influence of the mountain pine beetle, as it has with the influence of fire. In the past 50 years, however, fire suppression has created densities of young trees not often seen in previous centuries (as discussed previously and below). Such increased densities can contribute to water and nutrient stress, thereby reducing tree vigor and potentially increasing the susceptibility of some stands to mountain pine beetle epidemics. Given the influences of fire suppression, it is possible that the severity of ponderosa pine infestations in the latter half of the past century have increased. Intervals between outbreaks on the landscape have been about 10 to 20 years in recent decades on the BNF. Such levels are close to estimates during the HRV period in Colorado, so we conclude tentatively that outbreaks are probably still within the HRV on the BNF (low confidence). If the return interval for beetle outbreaks is shorter, the cause could be the warm and dry conditions of the 1980s. If true, then the most recent epidemics may be natural events rather than events caused by management activities. Timber harvest essentially has not occurred in the BNF ponderosa pine forests. Therefore, harvesting could not have affected the HRV of insect outbreak frequency or severity in that forest type.

Fire suppression could also have increased the frequency of western spruce budworm outbreaks, but, on the other hand, silvicultural practices such as clearcutting could have the opposite effect. Relative to high-elevation forests, less clearcutting occurs in Douglas-fir forests (Fig. 34). Therefore, harvesting

probably has had a small effect on the frequency of budworm epidemics in that forest type (low confidence). Western spruce budworm outbreak frequencies across the landscape are unknown, but at the stand level, the average outbreak intervals during the HRV period in Colorado and New Mexico are similar to existing conditions in those states, ranging from 20 to 33 years (Swetnam and Lynch 1989, 1993) and lasting for an average of 11 years (Lynch and Swetnam 1992, Swetnam and Lynch 1993). Thus, western spruce budworm incidence on the BNF is probably also within the HRV (low confidence).

Fire suppression and the low level of harvest in Douglas-fir forests (1%) may be increasing the susceptibility of Douglas-fir forests to the Douglas-fir beetle on the BNF. Douglas-fir beetle presently occurs only in scattered areas and large outbreaks have not been recorded. Even if outbreaks occur in the future, they probably will not create conditions that are beyond the HRV (moderate confidence), as they occur quite frequently in other parts of the Rocky Mountain region.

5.2.3 *Insects in aspen woodlands*

Various insect species defoliate aspen trees, but two are of special concern—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, though epidemics for these two insects have not been reported for the BNF. 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 a reflush of leaves, but infestation for several years in a row is lethal. When this happens, new aspen groves develop by sprouting or the conifer saplings in the understory grow into the canopy more rapidly. 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). The tortrix similarly has larvae that feed at the time of leaf expansion, but they also feed on buds. This insect can reach epidemic

population sizes that persist for two to three years in some areas (Allen and Harris 1999).

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 BNF documents, nor have we found information on such events in other parts of the Rocky Mountains that would enable an interpretation of HRV.

5.3 DISEASE

Pathogens such as dwarf mistletoe (*Arceuthobium americanum* most commonly), comandra blister rust (*Cronartium comandrae*), various root diseases (*Armillaria* spp., *Heterobasidion annosum*), broom rusts (*Melampsorella carophyllacearum* and *Chrysomyxa arctostaphyli*), and western gall rust (*Peridermium harknessii*), and heart rots are all native elements of the BNF and other Rocky Mountain forests (Johnson 1995, Allen and Harris 1999). 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 widely recognized also for their positive contributions to biological diversity and various ecosystem processes (Gill and Hawksworth 1964, Hawksworth 1975, Tinnen et al. 1982, Dinoor and Eshed 1984, Tinnen 1984, Johnson 1995).

5.3.1 Disease in high-elevation forests

Comandra blister rust. The most widespread tree disease in high-elevation forests on the BNF is comandra blister rust, which is a native fungus that attacks lodgepole pine (Lundquist 1993). On six permanent plots on the BNF, 18% of the lodgepole pine trees were infected in 1995, of which 30% had severe infections and topkill (Allen and Harris 1999). At the landscape scale on

the BNF, Lundquist (1993) found that 77% of the townships he surveyed from the roadside in 1991 had infected lodgepole pine stands, and 20% of the townships had severely infected stands (two or more infected stands per 1.6 km of road; Fig. 35). Infections were scattered throughout the BNF, but the most severe infections were on the east side at high elevations.

Comandra blister rust requires proximity of two host plants, lodgepole pine and comandra (*Comandra umbellata*). The spores are spread by wind from the comandra (also known as bastard toadflax) to infect pine needles and shoots. The fungus can eventually form a girdling canker that causes topkill and reduces tree growth thereafter (Geils and Jacobi 1991). Trees with topkill usually are at least 80-90 years old (K. Allen, personal communication). The younger trees (up to 80 years) are more susceptible to death than older trees because a smaller diameter provides a greater chance of the infection developing into stem-girdling cankers (Johnson 1986a, 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 of comandra blister rust increased dramatically, 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 may have occurred at relatively low levels during cool periods and at higher levels during warm, moist periods, such as the early 1500s and at the present time (Figs. 18 and 19).

Dwarf mistletoe. The second-most widespread disease at the present time on the BNF 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 most common on BNF tree species (Hawksworth 1958, 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 larger trees and it appears to spread most rapidly in uneven-aged landscapes (Kipfmüller and Baker 1998). While its effects can have negative consequences for silviculture and the 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 et al. 1996).

During an extensive roadside survey on the BNF in the 1980s, Johnson (1986b) found 36% of surveyed stands were infected with mistletoe. Later, in 1991, Lundquist (1993) found 71% of surveyed townships had at least one stand with mistletoe infections (6% had severe infections; Fig. 35). Although infections occurred throughout the BNF, most of the severe infections were in the south. The symptoms of mistletoe infection usually were most severe in localized areas. Notably, the severity level of dwarf mistletoe varied more from place to place than all the other major diseases on the Forest (Lundquist 1993). Kipfmüller and Baker (1998b) stressed that a "healthy lodgepole pine forest" probably has always had varying levels of dwarf mistletoe infection, and that some old forests are not infected.

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 several years to decades depending on the size of the fire because of the slow rate of growth and spread of the mistletoe plant (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 (Kipfmüller and Baker 1998b). In the earliest surveys of the Rocky Mountains, Mason (1915) 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 BNF at all levels of severity for centuries or longer (Kipfmüller and Baker 1998b).

Other diseases. Broom rust (*Chrysomyxa arctostaphyli* or *Melampsorella caryophyllacearum*), which infects fir and spruce, is another pathogen of importance (Lundquist 1993; Fig. 35). However, this disease and others seem to have minor effects on the high-elevation forests of the BNF. An exception is *Armillaria*, which in combination with the western balsam bark beetle may cause patch mortality in subalpine fir (Fig. 35), as discussed in Section 5.2.1 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 subalpine forests in the BNF. Most stands are likely to have numerous pathogens as concurrent members of the biological community (Lundquist 1993), which contributes to the biodiversity of an area (Lundquist 1995a).

Changes from HRV. Dwarf mistletoe can spread in young as well as older trees, but incidence levels are higher when stands have larger trees. Dwarf mistletoe also increases when landscapes contain stands with uneven-aged trees and small clearcuts or group-selection cuts (Johnson 1995). Such conditions occur on portions of the BNF. Some older stands with large trees are present across the landscape (about 10% of current forests; see Fig. 21) that probably would not have existed today without fire suppression (taking into consideration that much of the Forest is in young, less flammable stages). Tie-hacking and selection harvests, which produce the more susceptible uneven-aged stands, make up about 9% of the land on the BNF (15% of the forested area), and small clearcut patches (<8 ha, Harris et al. 1997) were commonly created in the 1980s and 1990s (Figs. 13 and 14). Working in the Medicine Bow National Forests, Selmants (2000) found that as many as 25% of the lodgepole pine trees in stands developing after clearcutting could become infected within 40 years.

By contrast, other areas on the BNF probably have reduced incidences of dwarf mistletoe. For example, the large clearcuts (> 8 ha) common in the 1960s (Figs. 13 and 14) may have slowed the spread of dwarf mistletoe. Some

younger stands in the high elevation forests would have been older without clearcuts (4% of the forest), and now they are less susceptible. Of course, large, natural, even-aged stands are common on the landscape even without timber harvest, keeping dwarf mistletoe at lower levels in large areas.

Because the extent of clearcutting has not fully counteracted the more extensive effects of partial cutting and fire suppression on the BNF, dwarf mistletoe incidence is probably above its HRV at the landscape scale (low confidence). Incidences in local uneven-aged stands where dwarf mistletoe occurs, and that would have been even-aged without harvest (group or individual selection cuts), 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 clearcuts may be above and below the HRV, respectively (low confidence).

Comandra blister rust attacks 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 (lodgepole pine, Engelmann spruce, subalpine fir). Because fire suppression probably has not increased tree density or stress at high elevations, the severity and distribution of comandra blister rust and other diseases, except for dwarf mistletoe, probably are still within the HRV (low confidence), especially on the landscape scale and given the current relatively warm climatic conditions that probably support the moderate levels of infection seen on the BNF.

5.3.2 Disease in low-elevation forests

Blister rusts. Various diseases also are present in the low-elevation forests and woodlands of the BNF, particularly two blister rusts. Comandra blister rust infects ponderosa pine as well as lodgepole pine, but little is known about its occurrence in ponderosa pine on the BNF (Allen and Harris 1999). The second, white pine blister rust (*Cronartium ribicola*), was introduced to North America and

is damaging to some stands with limber pine. It was apparently introduced to the Tongue River drainage of the BNF in the early 1960s (Lundquist 1993). Like comandra blister rust, the white pine blister rust requires an intermediate host to complete its life cycle, namely, currant or gooseberry (*Ribes* spp.) (Krebill and Hoff 1995, Allen and Harris 1999). Wind-dispersed spores from the shrubs infect pine needles and eventually create cankers that girdle and kill branches, if not the whole tree. Based on limited surveys, most of the forest has low incidence levels of white pine blister rust (landscape-level), although many limber pine trees have been killed by the disease in Ten Sleep Canyon (Lundquist 1993, Allen and Harris 1999). More surveys are needed to identify hotspots and the spread of the disease. White pine blister rust can have serious impacts on limber pine forests over time, such as in Montana and southern Alberta, where over a third of the limber pine trees at the landscape scale are dead and 90% of the remaining live trees are infected (Kendall 1998).

Other diseases. Other diseases in low-elevation forests include 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 damage to various tree species, none of them, individually, appears to have much effect on the BNF as a whole. Rather, it is the combined effects of diseases and other disturbance agents that influence stand structure and composition (e.g., Lundquist 1995a). Notably, dwarf mistletoe is not common on ponderosa pine in the BNF, though it is common in southern Wyoming and in Colorado. The rarity of dwarf mistletoe on the ponderosa pine of Wyoming may be due to geographic isolation or factors that either make the trees more resistant to mistletoe invasion or provide an environment not suitable for mistletoe (Hawksworth and Wiens 1972).

Changes from HRV. With the exception of white pine blister rust, the diseases found in the low-elevation forests of the BNF are native and probably have not been greatly affected by human influences. Silvicultural treatments probably have been too limited in the BNF to increase or reduce the spread of

disease across the landscape. Our observations suggest that the native diseases in low-elevation forests currently appear to be within their HRV at the stand and landscape level (moderate confidence). However, the presence of an exotic disease such as the white pine blister rust creates conditions that definitely are beyond the HRV (high confidence).

5.3.3 Disease in aspen and non-forest vegetation

Root diseases, canker-causing fungi, and a heartrotting fungus (*Phellinus tremulae*) infect aspen (Allen and Harris 1999). Cankers form in wounds commonly caused by animals, humans, or other agents, and they can weaken and kill the trees (defined as ramets in this case; Hinds 1976, Walters et al. 1982). Heartrotting fungi colonize the sapwood and produce a fruiting conk. Conversion of young aspen stands to older stands, a result of fire suppression, may be increasing the occurrence of this disease at the stand, landscape, and regional scales (Hinds 1985), although clearfelling has been used to offset the increase where that seems desirable (Johnson et al. 1995). Human activities in developed recreational areas may increase tree wounding (Johnson 1985), thereby increasing infection rates beyond the HRV in such localities (low confidence).

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

5.4 WIND

5.4.1 Wind in high-elevation forests

Wind speeds are highly variable in the Rocky Mountains, ranging from very strong at ridge crests to light in some valleys and in the lee of peaks (Martner 1986). Based on cursory observations, the BNF appears to have more storms capable of windthrow than most other national forests in the Northern Rockies (Howe 1997). The wind can blow down trees in small clumps (i.e., one to several trees) or topple whole stands of trees over thousands of hectares

(Alexander 1964, Veblen et al. 1991a). Large blowdowns typically produce a long, linear disturbance pattern oriented in the BNF in a southwest-northeast direction (Fig. 36). Most severe storms that cause blowdowns on the BNF come from the southwest (B. Bornong, personal communication). The wind follows a straight path but the actual blowdown is not continuous, as the wind often skips 2 to 3 km or more. Many large trees are snapped off at 6 to 9 m, but some are uprooted. Small trees may remain standing.

In 1991, about 670 ha of forest blew down in the northcentral portion of the BNF. Two years later about 1212 ha of trees were blown down farther north, north of Highway 14 (USDA Forest Service 1994). About nine long, linear patches on the landscape were produced from blowdowns between 1955 and 1998 (long, linear yellow or blue patches in Fig. 36), covering a total of about 1896 ha (0.4 % of landscape) and averaging 211 ha per patch. The actual area affected by blowdowns in total was probably greater than that estimate because it included only patches identifiable on satellite imagery. Of the 1896 ha identifiable on the imagery, 405 ha were partial blowdowns, in which many large trees were left standing (Fig. 36). The recurrence interval for such windstorms on the BNF is unknown.

Elsewhere, in 1987 a windstorm leveled 6000 ha of forest in the Teton Wilderness (Knight 1994a), and in 1997 winds of 200-250 km/hr toppled over 10,000 ha of forest on the western slope of the Routt National Forest in northern Colorado (Baker et al. 2001; Lindemann and Baker 2001, in press).

Blowdowns interact with other kinds of disturbances. To illustrate, a dense pile of fine fuels is created when windfall occurs, greatly enhancing flammability. Also, as discussed above, there is the potential for an outbreak of spruce beetles on downed spruce trees (although the potential appears to be low on the BNF, based on historical records of the past 100 years). Standing trees weakened by fire, insects, and disease may be more susceptible to blowdown. Notably, many trees in a stand are not toppled during blowdowns, making the effect more similar to a selective harvest than a clearcut (Lindemann and Baker 2001).

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, in press). Unexpectedly, soil permeability, soil water-holding capacity, and distance to natural edges were not significantly correlated with the blowdown pattern. 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 et al. 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.

Changes from HRV. Blowdown has been a common process associated with timber harvest, particularly for shelterwood (2- and 3-step canopy removal) and group selection cuts or on the leeward side of clearcut edges (Alexander 1964, USDA Forest Service 1994, Vaillancourt 1995). However, clearcut, shelterwood, or group selection harvests have affected only 11% of the forested landscape at high elevations; clearcuts alone have affected only about 4% (BNF RIS database). Consequently, the effects of harvest on blowdown frequency or extent surely must have been small. We can safely say that blowdowns, like fire, have always been and always will be an important factor in shaping landscape patterns and stand structure of high-elevation forests, and that current blowdown frequencies probably are within the HRV (high confidence).

5.4.2 *Wind in low-elevation forests and aspen woodlands*

No major blowdowns have been observed in the low-elevation forests or aspen groves on the BNF. Yet, just as in high-elevation forests, wind can have various effects on forests of ponderosa pine, Douglas-fir, and limber pine. As at higher elevations, the presence of predisposing factors such as disease, insects, and fire will affect the size of disturbance (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.

In the Black Hills, Graves (1899) found evidence for one tornado in the 19th century; another occurred in 1940 (Shinneman and Baker 1997). The more recent tornado leveled 1.5 million board feet of ponderosa pine. Aspen is also easily affected by blowdowns, as it has relatively weak stems and brittle branches (Veblen and Lorenz 1991). Despite its severe effects, wind is another natural phenomenon that will always be a potential disturbance agent in low-elevation forests and woodlands, as it is at higher elevations.

Changes from HRV. Because only a few blowdowns have been documented in the low-elevation forests on the BNF, and in stands of aspen, the differences between existing conditions and the HRV are unknown. However, timber harvest is not common in the low-elevation forests (<2% of low elevation forests) and aspen woodlands, and the current conditions affecting blowdown frequency and extent in both cover types must be similar to conditions during the HRV period (high confidence).

6. HISTORIC RANGE OF VARIABILITY FOR STAND AND LANDSCAPE STRUCTURE COMPARED TO PRESENT 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 impacts on variables associated with stand structure and landscape structure. As in the previous section on disturbances, our discussion includes our conclusions on whether or not the variables probably are exceeding or remaining within the apparent historic range of variability. Fire suppression and livestock grazing could have affected large portions of the BNF. Timber harvesting, in contrast, has affected only about 18% of the forested land area.

6.1 STAND STRUCTURE

Stand and landscape structure both change through time as well as in space because of natural and human-caused influences. In this section we focus mainly on stands that have been harvested for timber or subjected to fire suppression, assessing probable changes through time within individual stands rather than comparing all stands on the landscape. Landscape structure, which includes unharvested and harvested stands, is discussed in Section 6.2.

As of August, 2001, harvested areas on the BNF comprised only about 9% of all forested land (5% of the BNF as a whole; but does not include tie hacking, commercial thinning or salvage). If tie hacking, commercial thinning and salvage harvests are included, the proportion of the forests affected by harvesting increases to approximately 18%. About 4.7% was harvested using the shelterwood system, 3.3% by clearcutting, 0.4% with the seed tree approach, and 0.3% by selection harvesting. Most of the clearcuts have been concentrated in three watersheds (watersheds 3, 7, and 8 in Fig. 7; see Fig. 13 also). Today, it is estimated that 5% of the forested land is old-growth (based on photointerpretation as described in Section 4.2.1, but this estimate is preliminary because the old-growth inventory has not been completed).

As described previously, about 21% of the forests classified as suitable for harvesting (37% of the forested land) have been harvested since 1940 and 9% have been clearcut. If tie hacking is included in harvest estimates, 40% of suitable lands have been harvested since the late 1800s. Thus, many more hectares could be subjected to silvicultural treatments in the future (at least 79% of the suitable forests or about 82,000 ha, including about 20,000 ha that already have been harvested for railroad ties).

The 40% of suitable lands (5% of the BNF as a whole, and 18% of the forests) that are known to have been harvested at least once, of which nearly half was for railroad ties, is the focus of our stand structure analysis for high-elevation forests. High-elevation forests classified as "unsuited for timber harvest" probably have experienced few changes at the stand scale that were caused by European-American activities. Fire suppression can change the proportions of

the landscape in different successional stages (discussed in Sections 4.2 and 6.2), as well as tree density at lower elevations, but we do not believe that fire suppression thus far has greatly affected the internal structure of high-elevation forest stands (e.g., tree density at high elevations is not increased beyond the HRV relative to the age of the stand). In evaluating how management activities might have caused some variables to have exceeded their HRV, we focus on the effects of timber harvesting at higher elevations and the effects of fire suppression at lower elevations (where very little harvesting has occurred). Stand variables are considered in this section; landscape changes are discussed in Section 6.2.

6.1.1 Stand structure in high-elevation forests

The occurrence of fire at high elevations appears to have declined over the past century, but the amount of land affected by timber harvesting has increased. 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 both types of disturbances. In contrast, silvicultural treatments are designed partially, if not entirely, to manipulate stand structure in a way that favors the growth of certain species. 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 tend to facilitate persistence and the production of seed rather than growth rates. Consequently, our evaluation of many of the variables discussed below 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 timber harvesting: 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, age- and size-class structure (the number of trees in a series of age classes and size classes), forest floor depth, the extent of disturbed 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 part of the stand structure. Larger gaps are considered when discussing landscape structure (Section 6.2).

As noted, there are essentially no data on stand structure for the HRV period (1600-1890). 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 is necessary when developing site-specific management plans.

Based on the information we reviewed in Sections 4 and 5, and the nature of silvicultural practices on the BNF, we draw the following conclusions about 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/hectare in 100-year-old doghair stands of lodgepole pine. Various studies have shown that high lodgepole pine seedling densities are partially a function of the number of serotinous trees in a stand and fire intensity (Franklin and Laven 1989, Anderson and Romme 1991, Turner et al 1997). Ten years following the 1988 fires in YNP, tree seedling density ranged from essentially no seedlings to very high densities (sometimes exceeding 100/m²) in different stands across the landscape. We have observed the same on the BNF. 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, after which the area may develop low-density stands. When thinning is done for enhancing tree growth, the goal in the Bighorn National Forest is to have approximately 750 trees per ha (C. Thomas, personal communication). Thus, timber harvesting and associated thinning result

in stand densities that are within the HRV (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 the creation of 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 can also invade after these disturbances (Doyle et al. 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 elevation). In fact, lodgepole pine often regenerates so prolifically after a clearcut that precommercial thinning has been recommended after about 20 years (Grasso et al. 1981, Geils and Jacobi 1991).

Notably, seedling establishment is much slower above 2900 m on the BNF (Jack 1900) and elsewhere (Stahelin 1943). High-elevation clearcuts often are subjected to high winds that blow snow into downwind forests. Consequently, new seedlings and saplings in the harvested areas often die, apparently because of the lack of snow protection from winter desiccation and high-intensity solar radiation (Knight 1994a). Livestock grazing also can retard seedling establishment; C. Thomas, personal communication). Snags and coarse woody debris provide micro-environments favorable for snow accumulation along with shade and less intense wind, all of which favor new tree establishment.

From the late 1940s to the present, some BNF clearcuts have been in high-elevation spruce-fir forests (see RIS database and Fig. 34). Slash treatments that leave relatively low amounts of coarse woody debris could lower mean seedling densities from the mean for this variable during the HRV period. However, regeneration is often slow on rocky or sedimentary soils relative to granitic soils on the BNF, whether after fire or harvest (B. Bornong, personal communication), and because of the high natural variation in regeneration and seedling density from site to site, the presently variable conditions for seedling establishment and density are probably still within the HRV (high confidence).

Management activities or prescribed fire have been used to facilitate tree regeneration and to reduce flammability through the removal of excessive slash. The Intermission Fire on the BNF in 1988 was sufficiently hot, due to the abundant slash, that much of the seed source burned (Howe 1997). In contrast, several one-hectare patches that had reduced fuel loadings through slash treatment did not burn or burned less intensively and had good regeneration. Hot fires in Yellowstone National Park burned a significant amount of seed in the 1988 fires (Turner et al. 1997). Apparently, conditions that create fires that burn much of the stored seed are still within the HRV.

Percent canopy cover and rate of gap formation. By thinning forests to maximize tree growth, silviculturists create a canopy that may be more open than would have occurred otherwise. When stands are thinned naturally, the canopy is opened up less than is usually done for silvicultural purposes. This assumes that surface fires and outbreaks of insects and diseases do not thin forests on the BNF in a comparable way as selective harvesting. Surface fires kill some of the trees in small groups, as do insects and diseases, but the dead trees (snags) remain in the forest. Unlike thinning, selective harvesting includes tree removal that probably changes mean percent canopy cover below the HRV period in forests of comparable age and site conditions (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 that were created, or new trees grow into the gaps from below. Group selection harvests would create somewhat larger canopy gaps (0.1 to 0.4 ha) that would take longer to fill.

Shelterwood harvesting removes about one-third of the trees at intervals of about 10 to 30 years, producing large gaps that temporarily create uneven-aged forests. Once the overstory is completely removed (after 10 to 30 years, possibly longer), the stand becomes essentially 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 canopy trees creates gaps faster than during the HRV period because insect epidemics or fire typically have a within-stand turnover time that is considerably longer (at least 50 to 100

years). Thus, shelterwood cutting also produces canopy cover patterns that are not within the HRV.

Density of canopy gaps. As mentioned above, trees are gradually dying in the most healthy of 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 et al. 1994). Light penetration to the forest floor increases if a canopy gap is created, and competition for water and nutrients is greatly reduced if a root gap (large or small) is created. Both 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 there is the difference of standing dead trees, as discussed 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 managers to enable most of the remaining trees to grow faster.

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), especially those with wind-borne propagules. For example, the machinery used for yarding, as well as for piling, burning, roller chopping, and

tractor walking, can scarify the soil in ways that favor the establishment of lodgepole pine and Englemann spruce seedlings. However, such scarification may also slow the recovery of some understory plants, such as dwarf huckleberry (*Vaccinium scoparium*), as suggested by von Ahlefeldt and Speas (1996) and the results of Selmants (2000) in the Medicine Bow National Forest. Thus, species composition and abundance may be altered in ways that would not have occurred during the HRV reference period.

Grazing probably has not yet affected the understory plants significantly because high-elevation spruce-fir or lodgepole pine forests on the BNF are not heavily grazed (S. Gall, personal communication). However, grazing has been reported to affect tree seedling success in regenerating stands (C. Thomas, personal communication). Such potential impacts of grazing should be quantified. Sheep often entered high-elevation timbered areas in the late 1800s (Jack 1900), but it is difficult to know about the long-term effects on the forest understory.

Exotic plants can 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 2000). The same is probably true for the BNF. Exotic plants often are common in road ditches, but, in the Medicine Bow National Forest, they apparently have not yet invaded the adjacent forests in large numbers (Dillon 1998).

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 where the road bed was located and often provides habitat for introduced, weedy species. However, 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 would push variables associated with the understory vegetation beyond the HRV (high confidence).

Plant species diversity. All of the tree species present in the 1700s and 1800s apparently are still very common on the BNF. High grading, which

occurred earlier in the 20th century, may leave a relatively high number 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, though fire could have caused the same shift. Nevertheless, within-stand tree species diversity is variable and probably within the HRV (moderate confidence), particularly since no new tree species have been introduced.

Research is insufficient to indicate the degree to which management activities during the last century have changed the genetic diversity of stands. If fire suppression has been effective, or becomes more effective in the future, the abundance of serotinous trees in managed forests may eventually decline below the HRV. To counteract such an effect, some management practices on the BNF are specifically designed to select for cone serotiny (Howe 1997).

The total species diversity of organisms that exist in any ecosystem is determined in part by disturbances, but timber harvesting, with the concomitant removal of coarse woody debris and the effects of machinery and slash disposal practices, is a new kind of disturbance to which some species may not be adapted. Also, some plants may be 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 gradually increases in abundance, based on research done in the Medicine Bow National Forest (Selmants 2000). Understory vegetation of post-harvest stands dominated by lodgepole pine at lower elevations appears to recover more rapidly than in stands of spruce and fir at higher elevations (Selmants 2000). Some plant species may be rare and easily extirpated from an area (e.g., *Festuca hallii* disappeared after 1898 in the head of Clear Creek). Some stands (about 5,900 ha or 5.8% of suitable land) have been subjected to at least two entries for the harvesting of many large trees, with the first one (tie-hacking) occurring around the turn of the century. In such stands, the species

that are present today may be the species that can tolerate timber harvesting practices. Additional study is required to determine which species cannot.

Significantly, the species diversity in some areas may have increased because of the invasion of exotic plants. However, while the presence of such species increases the number of total species, one potential effect of exotics is to reduce the abundance of native species. Insufficient research has been done in Wyoming to determine the net effect of introduced species on species diversity and other variables in national forests.

In general, there are few introduced plants in high-elevation forests in Wyoming, and a relatively small proportion (16%) of the forests has been subjected to harvesting thus far. Thus, we conclude that native plant diversity probably is not lower or higher than the HRV in high-elevation stands (low confidence). 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, several ecosystem variables are beyond their HRV.

Size- and age-class structure of stands. A full range of size- and age-class structures must have existed within individual stands throughout time during the HRV period (Figs. 27 and 28; the proportion of the landscape in forests of different ages is discussed in Section 6.2). They would have ranged from young to older even-aged stands of lodgepole pine, many having developed after fires in the 1600s and 1700s, to uneven-aged stands that had not burned for centuries and were dominated primarily by Engelmann spruce and subalpine fir (and 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 BNF; 44% of YNP was subjected to varying degrees of fire in 1988; Balling et al. 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). Where timber harvesting does not provide sufficient time to allow the growth of large, old trees to replace those that were harvested, the age- and size-class structure is probably outside the HRV (moderate confidence). 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—as both clearcutting and intense fires would create the kind of size-class structure characteristic of even-aged stands, and currently, the forest stands have not been cut more frequently than the rate at which fires have burned stands (moderate confidence). Similarly, insect epidemics, partial blowdowns and partial cutting could create a structure characteristic of uneven-aged stands. However, because the planned rotation times are shorter than natural fire return intervals (see Section 6.2.1 on disturbance return intervals and rate at which new patches are formed), the average harvested stand in the future will not be able to achieve old-growth characteristics, and thus stand age and size structure may eventually exceed the HRV.

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 slow decomposition (Fahey 1983), but the litter does not become well incorporated into the mineral soil because of the absence of earthworms. Consequently, 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 moves 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.

Forest managers 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 in lodgepole pine have forest floors averaging 2-3 cm thick and range from no forest floor to one with a depth of several decimeters (Maxwell and Ward 1976), which falls within the expected HRV (high confidence).

Mineral soil exposed, disrupted, or compacted. In general, the litter on 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 is associated with 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 roller chopping, tractor walking and piling and burning, 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).

A notable difference between the two kinds of disturbances is the depth to which the mineral soil under the forest floor is disrupted (Bockheim et al. 1975, Halpern and Franklin 1990, Dion 1998). Scarification by a bulldozer blade or the blades of a roller chopper often disrupts 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 higher mortality of plants than would be expected for plants that survive a fire (Selmants 2000). An example of a plant in which scarification may slow recovery is dwarf huckleberry. In contrast, though fires typically burn plant shoots, the soil temperature typically is not high 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.

Soil displacement and compaction also occur after logging to greater extents than after fire. For example, on two clearcuts on the BNF, 25 to 52% of the soil was displaced (duff and top soil removed), mostly from machine piling of slash (Nesser 1990). On one of the clearcuts, 16% of the soil was compacted (a loss of $\geq 10\%$ porosity compared to an undisturbed site), mostly under skid trails. Natural disturbance processes do not create skid trails or such levels of compaction and soil displacement. Mass movement events such as landslides can displace soil, but only on steep slopes. Logging can create such disturbances on gentle slopes. It seems clear that mechanized yarding, slash treatment, and scarification can create a disturbance that rarely occurs following a fire, with the consequence that the HRV for surface soil disruption is exceeded (high confidence).

Snag density. The importance of dead trees as a component of wildlife habitat is widely accepted (Hutto et al. 1992, Hutto 1995, Bull et al. 1997, Parks et al. 1997, Hagan and Grove 1999). Reflecting this fact, BNF management prescriptions now call for leaving 5 to 7 snags per ha that are ≥ 25.4 cm DBH (USDA Forest Service 1985). Still, in areas that have been subjected to clearcutting (3% of the forested land on the BNF), there are far fewer snags right after the cut and in post-harvest mature forests 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 standing dead snags; a clearcut leaves few. Another fire can burn through a stand (called a reburn) not long after a previous fire, removing some of the standing or fallen snags (Lotan et al. 1985) and reducing snags to low densities. However, reburns are uncommon because young forests tend to be relatively inflammable due to the removal of so much of the fuels by the last 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 (not quantified), 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. Because most snags fall after 25 years (Lotan et al. 1985), the range of conditions within a stand for actual snag density would be zero to abundant—which clearly is within the range of historical conditions. However, 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. 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 et al. 1985). A reburn can occur, but there is still considerable woody debris remaining in the stand. The leaves, branches, and smaller wood are consumed by fires, but Tinker and Knight (2000) found that only about 8% of the wood >7.5 cm was burned after an intense fire in YNP. After a century or more, the downed wood becomes incorporated into the surface soil (Graham et al. 1994), giving the impression that there is little coarse woody debris. However, the residual of log decay is still present under the forest floor. Raking reveals the decomposed logs. New 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.

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 (MBNF), but the results are relevant to the BNF. They found a wide range of coarse woody debris cover and mass on cutover stands (11.4-26.8 m² and 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, suggesting 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. 37). Only when slash amounts left on the ground were doubled from 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 land classified as unsuitable for harvesting, but the coarse woody debris dynamics of individual stands—the focus of this section—is changed beyond the HRV by harvesting (high confidence).

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 et al. 1986, Hagan and Grove 1999); and the organic compounds derived from decomposing wood undoubtedly influence future soil productivity. Coarse woody debris is also known to be important for diversifying the structure of streams.

Could a changing climate have altered snag and coarse woody debris dynamics from the HRV reference period? As discussed previously, information available for the climate of the reference period suggests that it was somewhat cooler and was sometimes drier and sometimes moister. When moist conditions

avored tree growth and density, then there may have been more snags and coarse woody debris during the reference period than there are now. However, such conditions also would have favored decomposition, causing a more rapid loss rate for both snags and coarse woody debris. We conclude tentatively 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.

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.

In summary for high-elevation forest stands, the HRV for many variables appears to be quite broad and many stand-scale variables remain within what we believe is their HRV in managed forests. Stand variables that are possibly beyond their HRV are canopy cover and gaps, mineral soil exposed, snag density, and the amount of coarse woody debris.

6.1.2 *Stand structure in low-elevation forests*

Only 3% of the timber harvested on the BNF thus far has been in low-elevation forests, with most of that in forests dominated by Douglas-fir. There has been essentially no timber harvesting in ponderosa pine forests, and only a few hectares with limber pine have had timber sales. As in other national forests, the low elevation forests as a whole have changed more due to fire suppression, and possibly livestock grazing, than to timber harvesting. In this

section, we address the changes in stand structure that have occurred due to these activities in low elevation forests since the late 1800s, and whether or not various stand variables are beyond their HRV.

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. Studies in the Southwest and the Front Range of Colorado have focused primarily on stands of ponderosa pine, with the results suggesting that the stands would have been more open prior to fire suppression by European-Americans (e.g., Progulské 1974 and Veblen and Lorenz 1991). 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). Notably, some research suggests that crown fires also occurred in stands of ponderosa pine (Shinneman and Baker 1997), most likely on more mesic sites where tree density could have been higher. Indeed, "dense" stands were described in the 1800s by several explorers in the nearby Black Hills area (Dodge 1876, Newton and Jenny 1880, Graves 1899), probably on more mesic sites, and such stands probably were characterized by infrequent, stand-replacing fires. The pre-1900 canopy cover was 50 to 80% in such areas (Graves 1899). Working in the Black Hills, McAdams (1995) estimated that pre-European tree densities ranged widely, from 15 to 1600 trees/ha. Similarly, Ehle and Baker (in press) reported ponderosa pine densities of 70 to 3,000 trees/ha for Rocky Mountain National Park in northern Colorado.

Based on this evidence, Shinneman and Baker (1997) concluded that some ponderosa pine stands periodically would become susceptible to crown fires similar to those in forests at higher elevations. However, the thick bark of ponderosa pine, and the way that terminal clusters of needles on saplings protect the buds from fires, would appear to be adaptations to frequent surface fires.

Whether dominated by Douglas-fir, ponderosa pine or limber pine, some areas throughout the Rocky Mountains formerly dominated by open stands now have doghair stands with several thousand trees per hectare (Covington and

Moore 1994, Knight 1994a, Knight 1994b, Savage et al. 1996), mostly likely because of fire suppression and the elimination of Native American fires. Historic photos from the Black Hills, Northern Rockies, and to the south indicate that many stands of ponderosa pine and Douglas-fir were very open (Progulske 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. Indeed, historic photos indicate that BNF low-elevation forests that once were open now have become more dense (Wyoming State Historical Society 1976). Selective cutting can create open stands again, but this has not occurred in low-elevation forests on the BNF. Most of the harvests, though few in number, have been clearcuts that occurred in the 1960s. Since the 1980s, harvests at low elevations have mostly been shelterwood cuts (BNF RIS database). Neither silvicultural system restores the low tree-density structure that many believe was most prevalent on the BNF during the HRV reference period.

Livestock grazing also may have affected low elevation forest structure, as this land use practice has been widespread in the BNF. Indeed, reference areas unaffected by livestock are difficult to find. A comparison of historic and modern photographs of a BNF ponderosa pine savanna that was grazed in the past (but not currently) shows an increase in tree density (Wyoming State Historical Society 1976). Livestock grazing could facilitate this increase in density by removing sufficient herbaceous vegetation to reduce competition for young tree seedling. 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. Notably, large numbers of cattle and sheep (or elk and deer) could cause damage to seedlings and saplings, which would reduce tree density.

In summary, we conclude that the few stands at low elevations on the BNF that would have escaped fires probably are still within the HRV for tree density (high confidence). However, tree density in the majority of low-elevation stands probably has increased substantially above the HRV (high confidence). By contrast, the regeneration of Douglas-fir following clearcutting has been slow

at the upper elevation limits of Douglas-fir on the BNF (B. Bornong, personal communication). This slow regeneration may not be outside the HRV for this species at the limits of its range.

Plant species diversity, canopy cover and gaps, and understory composition and cover. No data are available to suggest that plant species diversity 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 BNF (high confidence), though Fule et al. (1997), working in the Southwest, found that trees less adapted to fire than ponderosa pine have become more important in many stands. Notably, Douglas-fir can eventually replace ponderosa pine as succession proceeds in relatively moist areas where fires are suppressed or occur at long return intervals (Goldblum and Veblen 1992).

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 or where harvesting has not been used to thin the trees, it seems probable that the species composition of the understory vegetation would have changed and the average amount of understory vegetation would have declined. The range of means for such variables is probably beyond the HRV (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.

Unfortunately, changes in herbaceous species composition due to domestic grazing 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 on the BNF 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 that were 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).

It is interesting to speculate on the effects that management activities might have on genetic diversity. A change in fire regime could reduce the fitness (ability to produce offspring that bear seed) of trees at low-elevations because bark thickness is heritable and repeated surface fires over time probably select for trees with thick bark. Without the selection pressure of repeated surface fires, the insulative quality of the bark of ponderosa pine and Douglas-fir could be reduced below the HRV over the long term (Howe 1997). We suspect, however, that such adaptations for surface fires are still well represented in the gene pool.

Size and age structure of trees. An effect of harvesting and fire suppression has been to reduce the natural variability in size- and age-class structure caused by historic disturbances. The increased tree and sapling density resulting from fire suppression prevents most trees from reaching large sizes; and with many more young trees, the average age of the trees in the stands is less. In ponderosa pine, the mountain pine beetle also reduces the average tree diameter by selectively killing more of the larger, older trees. Additionally, the beetles create canopy gaps and large openings. Similar to tree density, the average size and age structure of unharvested low-elevation stands probably are beyond the HRV due to fire suppression (moderate confidence).

For harvested stands, clearcutting on the BNF skews age and size distributions toward young trees and creates more even-aged stands. In contrast, natural disturbances, such as the western spruce budworm and fire, tend to have the opposite effect—an increase in the mean DBH per tree because the smaller, suppressed trees are killed first, leaving the larger ones, and through the more rapid growth of surviving trees. Less stand basal area results because the augmented growth rates don't compensate for the trees that fall due to the disturbance. Even partial cutting tends to skew age toward young trees and does not mimic the effects of insects and fire. Thus, age and size structure in harvested Douglas-fir stands is probably beyond the HRV.

In the Front Range of Colorado, harvesting specifically designed to improve forest health, vigor and appearance also produces a greater uniformity in density and age structure (Myers 1974, Alexander 1986). Various cutting methods have been proposed that could produce more stand structures typical of the HRV reference period (Merrill Kaufmann, personal communication), but, as discussed for high-elevation forests, their effects differ from those of natural processes in notable ways (less coarse woody debris and snags; Alexander 1986, Lundquist 1995a, Romme et al. 2000). Such experimentation has not yet been done on the BNF (C. Thomas, personal communication).

Spatial distribution of trees. An important pre-1900 characteristic of ponderosa pine forests appears to be the aggregation or clumped distribution of tree cohorts (a group of trees of the same species and age) in some areas at low elevations. Historical observations and reconstructions of open-grown ponderosa pine in the Southwest indicate that trees grew in clumps or patches of less than 0.4 ha (1 acre) and 3 to 44 trees (Cooper 1960, White 1985, Moore et al. 1993). Covington and Moore (1994) found that age structure within and between clumps varied depending on scale. Such patterns could occur at low elevations in the BNF as well, though Ehle and Baker (in press) found that dense, even-aged stands were common in Rocky Mountain National Park in northern Colorado. Further research is necessary to determine if the ponderosa pine distribution patterns observed farther to the south are also characteristic of woodlands in the BNF.

Although, the spatial distribution of both ponderosa pine and Douglas-fir forests on the BNF during the HRV period is unknown, we suspect that the distribution of ponderosa pine now is greater and more uniform than the HRV (moderate confidence), based on historic photographs. Also, the warmer conditions in recent years, relative to our HRV reference period, indicates that fire frequency and forest openness should be higher than during the HRV period. Yet, fire suppression is causing the opposite to occur. This observation further strengthens the argument that many characteristics of low-elevation forests are outside the HRV.

Forest floor depth. The short return interval of surface fires in low-elevation forests would probably keep the depth of the forest floor quite thin. Thus, it is plausible to assume that the forest floor did not become 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 small, 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, 1995b) studied the diversity of snags and coarse woody debris in ponderosa pine forests in the nearby Black Hills. He identified ten classes of snags and five classes of logs, 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 and become coarse woody debris on the forest floor. No snags are created. Based on this research in the Black Hills, timber harvest in the small area where it occurs at low elevations on the BNF 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. This evidence, combined with old photographs in Wyoming that show a much 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 ponderosa pine and Douglas-fir woodlands, due to harvesting (including casual fire-wood cutting). Notably, some coarse woody debris is burned by fires in low-elevation forests, possibly more than observed by Tinker (1999) in high-elevation forests because of frequent surface fires. Thus, with regard to coarse woody debris, low-elevation forests where timber harvesting has occurred may not be as far from the HRV as high-elevation forests.

Considering that most of the low-elevation forests on the BNF have not been harvested, but have been subjected to fire suppression, the bigger effect of management on snags and coarse woody debris at lower elevations may be less frequent fire. Fire tends to create snags, but insect and disease epidemics can do the same. In the absence of fire (or harvesting), are pathogens likely to be more abundant? That is a contention of some concerned about forest health problems in western landscapes, but it is not clear if this is occurring on the BNF. Our opinion is that there is not yet an unusually high or low number of dead trees (snags) due to fire suppression (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, and if coarse woody debris 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 (low confidence). This opinion is based partly on the considerable longevity of the trees that dominate the low elevation landscapes (ponderosa pine, Douglas-fir, and limber pine) and the fact that pathogenic problems are still rather localized where these species occur. A notable exception may be Tensleep Canyon, 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 stands of conifers to exceed their HRV in other parts of the Rocky Mountains. The same probably has occurred in the BNF. Great attention has been given to this situation, often in the context of "forest health" discussions. Low-elevation forests on the BNF cover a relatively small proportion of the terrain and provide a rather small amount of the wood that is harvested. Only about 2% 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, this is

where the effects of the exotic white pine blister rust are most evident. Attention should be given to the potential adverse consequences of the unusual conditions now found in this forest type, where they occur.

6.1.3 *Stand structure in aspen woodlands*

Following a disturbance, aspen often becomes more abundant, invading primarily from root sprouts. Seedlings can be abundant at times, such as after the 1988 fires in some parts of YNP (Romme et al. 1997). With fire suppression and natural succession, conifers are now becoming co-dominants with aspen in some areas, to the point that managers have begun harvesting or burning aspen to maintain this highly valued species in the landscape (BNF RIS database; B. Bornong, personal communication; Bartos 2001, Campbell and Bartos 2001). In other areas, stands of aspen have not been invaded by conifers. Both invaded and non-invaded stands probably occurred on the BNF during the HRV reference period, but the average number of conifers in aspen stands probably is increasing.

The addition of conifers can change the understory because plant growth under aspen is typically quite abundant, compared to coniferous forests, partially because of the relatively open, seasonal canopies of aspen stands that allow light to reach the forest floor. Notably, such forests support a high diversity of birds relative to other areas (Merrill et al. 1993). 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. Of course, aspen contributes significantly to the aesthetic quality of an area wherever it occurs.

Variables pertaining to aspen stand structure probably have not exceeded their HRV due to timber harvesting because there has been very little harvesting of aspen on the BNF (about 25 ha in the 1950s and 1960s were clearcut; BNF RIS database). However, in the 1980s, numerous groves were clearfelled to

stimulate aspen root sprouting. Such efforts were successful in many areas. When the bolewood is left on site during clearfelling, this management could mimic some aspects of the effect of fire. When bolewood is removed, the abundance of coarse woody debris and snags would be reduced below the HRV in the affected stands.

The most widespread influence of humans on aspen stands in the BNF may be through livestock grazing and the introduction of non-native plants, both of which could be changing the understory vegetation in still unknown ways. Less than one-quarter of the stands cut for regeneration have been fenced and protected from grazing (B. Bornong, personal communication). Consequently, assuming wild herbivores did not graze as heavily in aspen stands as domestic livestock do today (van Vuren 1982, Kay and Wagner 1996), grazing may be slowing tree regeneration and reducing the typically lush understory vegetation. Notably, an exotic species, Kentucky bluegrass (*Poa pratensis*), has invaded most aspen stands on the forest, which changes the understory species composition.

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 modern day aspen stands are probably within their HRV on the BNF. 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, such as Kentucky bluegrass. Stand variables pertaining to the trees probably are within their HRV, but variables pertaining to the understory vegetation probably are not (moderate confidence).

6.1.4 Stand structure in non-forest vegetation

Exotic plants are becoming more widespread in non-forested areas (i.e., grasslands, meadows and shrublands) on the BNF (USDA Forest Service 1998; S. Gall, BNF Range Conservationist, personal communication). In some areas, grasslands have been invaded by Kentucky bluegrass, Canada thistle (*Cirsium*

arvense), houndstongue (*Cynoglossum officinale*), Orchard grass (*Dactylis glomerata*), common timothy (*Phleum pratense*), Japanese brome (*Bromus japonicus*) and crested wheatgrass (*Agropyron desertorum*). Notably, cheatgrass is not a problem at the present time. Kentucky bluegrass is the most extensive introduced plant, particularly in intermediate moisture zones where it can outcompete native species (Girard et al. 1997). This grass is found in every grazing allotment and in all aspen stands (S. Gall, personal communication). Timothy is found in <5% of the allotments, but where it occurs, it occurs in big patches. Some non-native species were seeded in road rehabilitation sites. In stands with exotic species, the species composition and diversity is definitely outside the HRV (high confidence).

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, sheep and cattle grazing was intense and poorly controlled--probably exceeding the levels of ungulate herbivory during the HRV period (moderate confidence). However, because population estimates of herbivores from the 1600 to 1890 HRV period are not available, confident comparisons of herbivory levels at that time and today are impossible. Jack (1900) documented and photographed the damage from livestock grazing in the BNF in the late 1800s. Comparisons of such early photos to 1975-76 suggests that the initial heavy grazing caused forbs to replace grasses, as well as stream downcutting and less coverage by willows and sedges in riparian areas. Based on the 1900 photographs, it appears that some topsoil has eroded since 1900. Because such heavy grazing intensities are no longer allowed, some of the vegetation is now recovering.

Although the numbers of livestock have declined during the last several decades, today the BNF is still one of the most heavily grazed national forests in the region. Possibly over a third of the rangelands are less than satisfactory, based on percentages of plant species that are increasers compared to decreaseers (S. Gall and D. Beard, BNF Range Conservationists, personal communication). The dominant grassland type affected by grazing on the BNF is the Idaho fescue community, where 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. Exotic species also have increased in riparian zones (Beetle 1956, Girard et al. 1997), where the elimination of the litter layer warms and dries the soil, encouraging the growth and reproduction of certain plants at the expense of others. In the 1950s, grazing reduced grass and grass-like species by 15%. Consequently, forbs increased (Beetle 1956), although some forbs such as larkspur (*Delphinium* spp.) declined (S. Gall, personal communication). 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. Livestock are more concentrated because they are fenced and only in the last 10-15 years has rotation grazing been implemented on the BNF. Grazing can also promote the spread of big sagebrush in some areas, which, once established, is competitive with grasses and may persist for many years.

Overall, exotic species today, and the intense grazing levels on the BNF during the last century, probably have pushed the structure and species composition of grasslands, forblands, and shrublands beyond their HRV (high confidence). Prescribed fire or chaining are sometimes used to reduce big sagebrush (B. Bornong, personal communication), which might help restore HRV conditions when that seems desirable, but there is now the potential of exotic plants growing where such control practices are attempted. Fire suppression may have allowed woody plants to occupy meadows and shrublands in some areas,

but we doubt this change has been as significant as introduced plants and livestock grazing.

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 tree rings, and old photographs reveal less information), and many ecosystem variables are more difficult to measure than in forests. To illustrate, much of the biomass and important ecosystem processes occur belowground. Moreover, successional changes are more difficult to interpret than in forests. With more careful management of livestock and wildlife, some variables could be restored to within their HRV—to the extent that is possible without the eradication of exotic plants such as Kentucky bluegrass. Monitoring various soil variables and variables associated with plant species composition, including the abundance of introduced plants, will help provide information required for judging sustainability

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, stand-replacing fires, wind storms, and some insect epidemics 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 forest 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 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, proportion in different classes of 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 intervals.

As with our stand-level analysis, this section emphasizes high- and low-elevation landscapes. Aspen and non-forest vegetation are included in the discussion of these two, more general landscape categories.

6.2.1 High-elevation landscapes

The high-elevation, forested landscapes encountered by early explorers in the 1800s in northcentral 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 et al. 1998). 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, such as in ravines or on north slopes, were less likely to burn. Uneven-aged stands of older trees, including old-growth forests, would have occurred in mesic areas where, by chance, there had not been a stand-replacing fire for several centuries (Romme and Knight 1981).

Large, infrequent fires cause fragmentation of the forest just as timber harvesting can. Landscape fragmentation is a complex phenomenon (Buskirk et al. 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. Where the topography is comparatively flat outside the Cloud Peak Wilderness, extensive forest stands can develop that are accessible for timber harvesting and susceptible to fragmentation (Tinker et al. 1998), particularly in the southern part of the BNF (Fig. 13). The northern half has always been more fragmented because of the natural parks that occur on fine-textured soils (forests commonly are on coarse-textured soils).

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).

We base our estimates of landscape HRVs on our more lengthy discussions in the previous stand structure section and on the average of values (mean) estimated across the landscape through time. To illustrate, consider that ten years following the 1988 fires in YNP, tree seedling density ranged from essentially no seedlings to very high densities (exceeding 100/m²) across different stands (Turner et al. 1997). Information on this spatial variability among different stands is lost when the stand-level values for a variable are averaged, but there is still the potential for change in the average values through time. If the stands did not change over time, the landscape-level average would remain at a constant value. However, stands do change, and therefore the temporal variability in average values can be used to describe the historic range of variability.

Many characteristics of stand structure surely had high variability through time during the HRV period. Average tree seedling density and adult tree density across landscapes were probably quite different in years following large fires than in years with few fires. To illustrate, serotinous lodgepole pine cones release abundant seeds following fires, which can lead to high seedling densities in many stands, increasing average seedling (and eventually sapling and tree) density across the landscape as a whole. In other years, densities could be lower in most stands, perhaps because of slower seed dispersal from non-serotinous cones, such as might occur after insect epidemics that cause high levels of tree mortality. Similarly, the density, species diversity, and cover of understory plants change as succession occurs in stands (e.g., intermediate stages have high tree canopy cover and sparse understory vegetation). Thus, the temporal variability of the average density and cover of understory vegetation

probably would be moderately high. The same is true of canopy gap density and cover. The range of the averages of these variables would be expected to be from moderately low to moderately high values—never very low to very high because that would require all stands on the landscape, at one time, to have very low or very high values. That would be unlikely.

Similarly, stand age structure averaged over the landscape probably had a moderate range of values during the HRV period (Figs. 38 and 39). Stand age must have ranged from very young to quite old over time, and from predominantly even-aged to predominantly uneven-aged—the result of infrequent stand-replacing fires and long periods of succession combined with other natural disturbances that would affect both stand and landscape composition. The average of the stands on a landscape would range from moderately young and even-aged to moderately older and uneven-aged. For the same reason, forest floor depth must have had a moderate range of values because it is shallow or absent immediately after a hot crown fire but deep in stands that have not burned for a century or more, such as in old-growth forests.

In contrast, mineral soil disruption and compaction surely were very limited in spatial extent throughout the HRV period, because machinery was not used to assist with timber harvesting. Thus, the variability in average forest floor depth should have been moderate. Snag density and the abundance of coarse woody debris must have varied greatly across the landscape, from near zero in mid-successional stands to abundant after fires or disease and common in stands of old-growth forest; but, similar to the other variables, with a narrower range (moderately high to moderately low) at the landscape scale than at the stand scale (Fig. 1).

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 BNF forests that have been subjected to some form of harvesting is about 20%. Specifically, the proportions of the total forested land area at high elevations in shelterwood cuts, clearcuts, selection

cuts, and seed tree cuts are 5.7%, 4.1%, 0.4%; and 0.5%, respectively (similar data in Section 6.1 are for high- and low-elevation landscapes combined). About 8.7% has been tie-hacked without re-harvest (11% if re-harvested stands are included). The forested area that has been thinned (commercial and pre-commercial) is 5.5% (BNF RIS database). Forests that might have burned but were not due to fire suppression, and thus are older, comprise an estimated 10% of the landscape. Under existing conditions, tree densities would still fall within the moderately broad HRV because, even though trees have been partially or completely removed on about 20% of the landscape, fires may have removed half of that anyway if fire suppression had not been practiced (moderate confidence). Also, because new tree establishment on harvested areas is usually quite good, seedling and sapling density in harvested landscapes probably falls within the fairly broad HRV (moderate confidence). Canopy gap density and cover may be slightly below the HRV, but after 20 years these variables return to more natural conditions (moderate confidence). Understory density and cover, forest floor depth, and plant diversity averaged across the landscape probably are not outside the HRV (moderate confidence).

Stand variables averaged at the landscape scale that could be beyond their HRV—again on the 20% of the high-elevation landscape affected by timber harvesting—may be the abundance of older forests, mineral soil disruption or compaction, snag density, and the abundance of coarse woody debris. Until recently, timber harvesting tended to reduce the land area in old-growth forests; and with the advent of mechanical harvesting, the soil would have been subjected to mineral soil disruption or compaction in ways that would not have occurred during the HRV period. Even though mechanized yarding and slash treatment have been used on a small portion of the BNF, the average amount of disruption/compaction would be higher than the HRV because the HRV would have been narrow. Mean snag density averaged over the landscape may be below the HRV because burned areas are often salvage logged and snags are removed for firewood (firewood collection affects areas other than the 20% harvested). Also, bolewood is removed, preventing future development of as

many snags in harvested areas. Similarly, coarse woody debris abundance is probably below the HRV because much 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.

Number and proportion of land-cover types. 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 BNF probably were present during the HRV period. The number of land cover types (landscape richness) probably has not changed beyond the HRV (moderate confidence). Also, the proportions of different cover types are naturally dynamic (Fig. 40) and probably are within the HRV (high confidence). Evidence suggests that the lodgepole pine type on the BNF has become less abundant in the last century while the spruce-fir type has increased (Fig. 40), because of the maturing of the forest after heavy forest fires in the late 1800s (Fig. 26) and some successful attempts at fire suppression. We would expect the opposite to occur with timber harvesting, which would increase the amount of lodgepole pine. About 10% of the forested BNF landscape has been clearcut or cut using the shelterwood system, yet this has not increased the prevalence of lodgepole pine relative to spruce and fir. Clearly, the effects of climate and harvesting on lodgepole pine abundance appears to be small relative to the effects of successional processes. Thus, the proportions of the various types of coniferous forests in the landscape probably are still within the HRV (moderate confidence).

While aspen stands do not occupy a large area on the BNF, they contribute significantly to the overall diversity of the landscape. Aspen occur in high and low-elevation landscapes, but mostly at low elevations on the BNF. Therefore, extent of aspen on the landscape is discussed under low-elevation landscapes.

The grassland cover type may have declined relative to sagebrush on the landscape during the decades preceding 1985. The 15% decline in grasslands and 4% increase in sagebrush (Fig. 40) documented for the whole BNF (no data

are available for just the high-elevation landscapes) may have been due to fire suppression, but, given the variability in proportion of other cover types over time, sagebrush cover probably is within its HRV (moderate confidence). Prescribed burning since the 1980s may be reducing the rate of sagebrush invasion into grassland.

Forest/non-forest ratio. Because of forest encroachment into meadows and shrublands, and natural reforestation following fire and timber harvesting, the proportion of the BNF in forested vegetation has increased substantially since the late 1800s, from 35% in 1898 to 62% at present—an increase of 27% (USDA Forest Service 1994). The accuracy of the early 1898 and 1900 estimates used for this comparison (Town 1899, Jack 1900) is probably low, but it seems clear that the area in forests has increased substantially (even between 1931 and 1993, when an increase of 9% occurred; USDA Forest Service 1994). The current variability of forest/non-forest percentages across watersheds is also high, from 40 to 90%. Much of this spatial variability is due to distribution patterns discussed in Section 4.1, including those caused by varying topography and soils, but this range may have been much broader because fire suppression might have narrowed this range. Nevertheless, due to the variability of local climatic conditions over the past 500 years, the present ratio of forest to non-forest land is most likely within the HRV on high-elevation landscapes (high confidence).

Proportion of forests in different successional stages and old-growth. 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 middle and late 1700s, young forests dominated more land area; but in the mid-1900s, old forests were more common in the same area (Romme and Knight 1981, Romme 1982, Fig. 39). Such shifts

in the landscape mosaic surely have ecological effects on species composition, biodiversity, nutrient cycling, and other ecological phenomena.

A similar shifting mosaic must have occurred during the HRV period at high-elevations in the BNF. To evaluate this quantitatively, the present vegetation on the BNF was divided into successional stages (using information on habitat structure in the BNF RIS database) and compared to the HRV in YNP (Table 5). The inventory of old-growth forest on the BNF is not yet complete, but, based on present estimates of habitat structural stages in the RIS database, current old-growth forest percentages on the BNF fell below HRV levels of old-growth forest (LP3) from 1735 to 1985 on 129,600 forested ha in YNP (Fig. 38). The lowest old-growth percentage ever observed in YNP during that period was 15%; the BNF classified only 5% of its forests as old-growth in 1994 (USDA Forest Service 1994, based on aerial photo interpretation). Averaged over the last 30 years, making the estimate more comparable to the YNP estimate, the mean percentage in old-growth on the BNF is probably closer to 7%--which is still below the range of means over 30-year periods during the HRV period (see Fig. 38). The BNF's current definition of old-growth is probably more liberal (includes more hectares) than the one used in YNP, which suggests further that old-growth is below the HRV on the BNF.

Notably, the effect of the 1988 fires is not included in this assessment. To include the 1988 fires, we divided the successional stages into early (0-40 years, middle (~40-200 years), and late stages (>200 years, late includes transitional stages to spruce-fir, not just old-growth), as Romme (1982) had done in YNP's Little Firehole watershed (Table 6). We estimated percentages of each successional stage just after the 1988 fires in that watershed using satellite imagery, adding the 1988 data to Romme's (1982) data (Fig. 39). We found that late-successional stages were less common and mid-successional stages more common in many watersheds in the BNF than the YNP HRV proxy—1738 to 1988 on the Little Firehole watershed (Fig. 39). Even with inclusion of the 1988 fires as part of the HRV period, older forests in the Little Firehole watershed were

never below 7%. Comparable values for old forests in some BNF watersheds were lower than that (Fig. 41; specifically, watersheds 3, 4, 5, 6, 7 and 10).

The Little Firehole watershed is not the best reference area for comparison to BNF watersheds because two of the BNF watersheds that had fewer late successional stages than YNP had no timber harvests (watersheds 5 and 6). These two watersheds had experienced large fires in the late 1800s during the HRV period, which naturally reduced older forest percentages. Thus, the high level of mid-successional stages and low old-growth in these specific areas is probably within the HRV.

The observation that late successional stages of all BNF watersheds are either lower or at the low end of the HRV, based on research done in YNP, suggests that the percentage in some heavily clearcut watersheds is probably beyond the HRV (Fig. 41). Because large-fire dynamics in the BNF appear to be similar to YNP, one would expect at least one watershed to have at least medium amounts (about 40%) of late successional stage forest. Apparently, due to the extensive fires in the late 1800s on the BNF, the BNF was already at a low point in amounts of old-growth. The additional harvesting (9% affected by shelterwood cuts, clearcuts, and seed tree cuts) appears to have pushed old-growth or late-successional stage percentages below the HRV (moderate confidence).

Forest planning that promotes the harvest of larger, slow-growing trees of old forests, and an ability to harvest such forests on north slopes and ravines where fires might not have burned as often, may help to explain why the amount of old forest today in some watersheds appears much lower than at any time during the YNP reference period. Fire history studies and more complete old-growth inventories are needed on the BNF to support or refute the conclusion that old-growth forest is below the HRV.

Landscape proportion in low canopy cover. 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 relatively abundant on the BNF landscape (14% of landscape with low-density forest; Fig. 41), partially because of management and partially from natural conditions.

About 6% of high-elevation forests were thinned or selectively cut, but only about 4% were thinned in the last 20 years. Only those thinned in the last 20 years may still have open canopies, as the branches of neighboring trees gradually fill in gaps created by harvesting. Thus, nearly one-third of the stands with low canopy cover appear to have resulted from management activities, a landscape trait that probably is beyond the HRV (low confidence)—even with the prevalence of dwarf mistletoe infections on the forest (which still leave snags that provide some canopy cover). Considerable uncertainty is involved with this conclusion because of the lack of information on the amount of forested land where trees could have been thinned due to surface fires, insects or pathogens during the HRV period.

Landscape proportions in high-density classes for snags and coarse woody debris. Snag and coarse woody debris density is often highest in recently burned forests (Tinker 1999) and in old-growth forests (Mehl 1992). When the percentage of the landscape in such forests is below the HRV, the numbers of snags and the amount of coarse woody debris on the landscape also is probably below the HRV. Fire suppression has reduced the area of recently burned forests by at least 10%, and timber harvesting has reduced the area of old-growth forests. Both observations suggest that areas with high snag densities are less common than during the HRV period (moderate confidence).

Edge, interior forest habitat, patch shape, and patchiness. The spatial configuration of the landscape patches has been changed due to timber harvesting. Whereas fires and blowdowns produce many small patches and a few large patches, clearcuts produce a few small patches and many mid-size (4-40 ha) patches (Figs. 30, 32, 34, and 36). Hence, in contrast 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. 34), 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 and blowdowns (Fig. 36).

Also, although edges were certainly created between areas of different burn intensities, human-created edges (around clearcuts and along roads) usually are more abrupt. 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 on the BNF (Merrill et al. 1993), but loss of the interior-dependent wildlife species is of regional concern (Beauvais 2000). For these reasons, many investigators have argued that timber harvesting and other human activities cause the fragmentation of high-elevation forests, both on the BNF (Tinker et al. 1998) and elsewhere in the Rocky Mountains (Baker 1994; Reed et al. 1996a, 1996b, R.L. Knight et al. 2000).

Using a LANDSAT-derived database in Arc/Info (Fig. 36), we compared a number of landscape patterns of clearcuts across watersheds on the BNF to the more natural patterns produced by fires and blowdowns in the last 43 years on the BNF, to quantify timber harvest effects. We used the program FRAGSTATS (McGarigal and Marks 1995) to quantify the patterns of the patches created by these disturbances. Rather than evaluating the patterns of the entire landscape (as was done in another study on the BNF by Tinker et al. 1998), we focused on the configuration of just the disturbance patches shown in Fig. 36. The mean of pattern variables created by clearcuts (e.g., edge density and percentage of land in clearcut) averaged across watersheds fell within the range of patterns from fires, but some individual watersheds had more edge per ha and total land in patches caused by clearcuts than ever occurred with fire or wind (Fig. 42). On average, clearcuts were smaller, had less complex (convoluted) shapes, less variation in patch size, and smaller maximum patch sizes than natural patches (Figs. 13 and 42). These patch variables probably are beyond their respective HRVs (moderate confidence).

Thus, the watersheds with heavy cutting 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 and blowdown patterns in the last half century are somewhat similar to the HRV. This assumption is violated for fire patterns, since current fire sizes are, on average, smaller than fire sizes during the HRV period due to fire suppression. Because fire sizes were actually bigger during the HRV period, the contrast in patch sizes is even more evident (large for fire, small for clearcuts).

When we compared patterns of shelterwood cuts (which begin as partial cuts) to patterns of partial blowdowns in the last 43 years, using the same database (Fig. 36), shelterwood cuts in some watersheds had more edge per patch area, more total area, and a larger area in the biggest patch than the HRV of partial blowdowns (Fig. 43). All partial blowdowns may not have been detected on the LANDSAT imagery, but shelterwood cuts probably are still more abundant on the landscape than the partial blowdowns. Additionally, shelterwood cuts are not equivalent to partial blowdowns because the rest of the trees are removed within 10 to 30 years (and bolewood is removed in the former). Possibly, some partial blowdown areas 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. The patterns that are created are beyond the HRV (moderate confidence). Other disturbances in the past 43 years (from insects or disease) apparently did not create large enough patches to be detected on the LANDSAT imagery.

Tinker et al. (1998) looked at the effects of roads and clearcuts on landscape patterns on the BNF (the extensive road network is shown in Fig. 16 and clearcuts are in Fig. 13). They found that both roads and clearcuts appeared to reduce core area (interior area of patch with edge depth of 50 m) of all land cover types, increased edge and edge convolution, reduced patch size, and increased patch diversity on the landscape. In two of the twelve watersheds, clearcuts and roads reduced the area of interior forest lower than values before roads and timber harvests. In their analysis, they removed the roads and replaced harvested areas with old-growth to simulate "natural" conditions. Addressing spruce-fir and lodgepole pine forests only, roads and clearcuts

reduced area, patch size, and core area (interior forest) for each type. Edge and patch density also were increased for spruce-fir forests. Up to three watersheds had variables outside the range found in watersheds without clearcuts and roads. If spatial variability (range) in the 1990s, after roads and harvests are removed, substitutes for temporal variability during the HRV period, then these variables fall outside the HRV (low confidence). Moreover, the numerous meadows and shrublands in the BNF gave the landscape a rather high amount of natural edge and relatively low amount of interior forest (Fig. 10) during the HRV reference period. Thus, additional fragmentation by timber harvest may further reduce the amount of already limited interior forests (Knight and Reiners 2000).

Compared to fire, timber harvesting affects a much smaller area on the BNF (Figs. 21), but harvesting every year in similar patch sizes creates a pattern on the landscape over time that is different than would have occurred during the reference period. For example, on the MBNF in southeastern 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 (high confidence). 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 unusually small patch-cuts 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 et al. 1996a, 1996b; Miller et al. 1996; Tinker et al. 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. 12 and 14), but the studies show that fragmentation has steadily increased.

Disturbance return intervals and rate at which new patches are formed.

Another aspect of silvicultural practices that can be compared to disturbances during the reference period is the return interval for successive disturbances at a site. Currently, the 1985 BNF Management Plan (pages II-232) specifies the rotation age for clearcutting in subalpine/montane landscapes at 60-180 years for lodgepole pine, 100 to 240 years in spruce-fir, 80-120 years for aspen, and 100+ years for other cover types. For two- and three-step shelterwood cuts, the rotation ages are 100-240 years for stands dominated by Engelmann spruce and subalpine fir, 90-180 years for lodgepole pine, and 80+ 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 fire-return interval at specific sites.

The timing of thinning and partial harvests also are different from natural disturbance frequency. For example, both mountain pine beetles and spruce beetles 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 BNF. On the BNF, thinning 90 to 100 year-old stands was the standard practice in the 1970s and early 1980s. Monitoring has shown that this practice has not been economically effective for increasing tree growth and that it creates excessive slash (B. Bornong, personal communication). Because of the current thinning of young stands and shelterwood cutting, which involves two or three cuts per rotation, the landscape disturbance frequency is higher than with natural processes, especially with the addition of the concomitant road-building and maintenance.

The result of the above changes in the site-specific return interval for disturbances is a change in the rate at which new patches are formed across the landscape. For example, in one BNF watershed, the mean clearcut return interval across the landscape (not in one site) was shorter with timber harvest

than the mean interval for all fires that occurred prior to and after 1877 (Fig. 20). The increased harvest return rate results in the increased patchiness on the landscape that is occurring in some heavily-cut watersheds (watersheds 2, 3, 7, 8, and 9 in Fig. 7; Fig. 36). Thus, it is likely that the rate at which new landscape patches are formed now exceeds the HRV (high confidence).

In summary, the general conclusion is that European-Americans have made a significant imprint on high-elevation landscapes through timber harvesting on about 20% of the high-elevation forests on the BNF. Since the late 1800s, a large amount of old-growth forest has been cut, probably lowering the amount of this forest type to below the HRV. A large number of roads now exist also, contributing to a level of fragmentation that probably exceeds the HRV in locations where forests occur over large, flatter areas (Fig. 13). The affected forests occur primarily north and south of the Cloud Peak Wilderness.

6.2.2 Landscape structure at low-elevations (including aspen and non-forest vegetation)

Because the majority of forested area on the BNF occurs at high-elevation (81%), the RIS data for the BNF enable conclusions that are largely pertinent to high elevations. It is clear, however, that the low elevation landscape patterns are different. Fire suppression probably has impacted the low-elevation landscape more than timber harvesting, as less than 2% of the low-elevation forests have been harvested (2001 RIS database)—about 1% shelterwood cuts and 1% mostly clearcuts. As in the previous section, we first examine stand structure variables at the landscape scale; and then we discuss variables that apply to landscape mosaics.

Forest stand structure at the landscape scale. Most of the low-elevation landscape probably had grasslands, shrublands and relatively open forests during the HRV reference period because of frequent surface fires and the patchy distribution of soils and microclimate in the topographically variable foothills where ponderosa pine, Douglas-fir and limber pine could survive.

However, there probably were always some stands that escaped the surface fires, resulting in higher tree and sapling density, lower understory plant densities, and fewer and smaller canopy gaps than in neighboring stands that burned more frequently (Romme et al. 2000). When the few forested stands were averaged with the abundant woodlands and shrublands, and projected through time during the HRV period, tree and sapling density on low-elevation landscapes probably ranged from low to moderately low. With the open canopy cover, density of grasses and forbs in the understory probably ranged from moderately high to high. Forest floor depth did not accumulate, 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. It is impossible to quantify variables such as these at the present time.

Where fire suppression has been effective, tree and sapling densities and canopy cover averaged across the landscape are now relatively high, while canopy gap area and the cover of understory vegetation are relatively low. Snag density and coarse woody debris could be relatively low in areas subjected to timber harvesting, but as most of the landscape has not been harvested (only ~2% of the landscape at low elevations), the majority of the low-elevation forest probably still has snag density and coarse woody debris amounts within the HRV. Species diversity across the landscape is relatively high and relatively constant through time because of the great variety of environments and successional stages in the BNF, and diversity probably has not been affected in an adverse way at the landscape scale by management practices. On average tree size and age probably have declined due to fire suppression increasing the number of young trees. Overall, where fire suppression has been effective, tree and sapling density, mean age and size of trees, forest floor depth, and some understory vegetation variables are probably outside the HRV (see Table 7 for confidence levels).

Proportion of landscape in different land cover types. The proportion in different land cover types in low-elevation landscapes has changed between 1931 and 1994 (Fig. 40). Specifically, the amount of forested area has increased

~4%, with Douglas-fir increasing by ~6% and ponderosa pine declining by ~2%. Aspen was not extensive in the 1930s on the BNF, nor is it today with $\leq 1\%$ land cover (Fig. 40), and thus, detecting any change is difficult. The abundance of aspen probably is not outside the HRV (low confidence). Notably, in comparing historic and modern photographs taken on the BNF, it is clear that up to half of the aspen stands in some photographs died between 1900 and 1975 (Wyoming State Historical Society 1976).

As discussed in Section 6.1.3, aspen cover has declined in some areas due to a variety of factors (DeByle 1979; Veblen and Lorenz 1986; Kay 1990, 1993; Kay and Wagner 1996; Baker et al. 1997; Hessler 2002, and several papers in Shepperd et al. 2001). Livestock and large ungulates are possibly causing declines in aspen in the small area where it occurs on the BNF, particularly on low-elevation winter ranges, but other causes of aspen decline may be as important. To help offset declines in aspen, clearfelling was used in the 1980s to regenerate some aspen stands (about 200 ha), and many of the new shoots (ramets) have now grown into the sapling-pole stage (RIS database, USDA Forest Service 1994).

With regard to the non-forest vegetation at low elevations, there would have been some fluctuation in the the abundance of grassland and shrublands during the HRV reference period due to the effects of fire and the varying sprouting capacity of of the different shrub species, as discussed in Section 5.1.4. Fire suppression may have caused the land area in shrublands to become relatively high. However, we have no basis for concluding that this variable is beyond its HRV (low confidence).

In sum, we believe the proportion of the different vegetation types is still within the HRV on low-elevation landscapes in the BNF (moderate confidence).

Forest patch sizes and configuration. The low-elevation landscapes of the BNF during the HRV period probably were composed of distinct and separate stands of Douglas-fir, ponderosa pine and limber pine (with variation in age-structure within each stand). Such landscapes probably resulted largely from topographic control, but fires also would have created openings of various sizes

(Kaufmann et al. 2000). Today, due to fire suppression, some areas are more homogeneous than during the reference period (moderate confidence).

Using the RIS and fire databases for the BNF, we found that clearcutting practices (mostly in the 1960s) increased mid-sized patches (4-40 ha, Fig. 34) over and above the HRV under natural fire regimes (Fig. 33) in low-elevation Douglas-fir forests, which are the most abundant low-elevation forests on the BNF.. Notably, the timber harvesting comprised a small percentage of the landscape, but, in the small areas where the harvests occurred, the distribution pattern of the openings created by harvesting on these landscapes is probably beyond the HRV for comparable openings created by fire and other disturbances (low confidence).

Proportion of landscape dominated by old trees. 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). However, the available preliminary data indicate that only 1% of the land area with Douglas-fir is classified as old-growth (habitat structure data in BNF RIS database). Because little harvesting has occurred at low elevations, we are hesitant to postulate a cause for such apparent low proportions. Old-growth forests certainly could be lost in the future if suitable lands are harvested at the planned rotation rate. Fire suppression could cause more of the low-elevation forests to have old trees. However, this practice has been in effect for too short a time for that to happen.

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 et al. 2000). Notably, if old trees are defined as >200 years rather than >300 years in the Colorado study, then old-growth was more common on the logged landscape. The number of patches with >200-year-old trees comprised half the patches and were similar between the logged and unlogged areas. 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.

Proportion of landscape with high-densities of snags and coarse woody debris. Areas with extensive cutting for firewood or lumber would have reduced snags and coarse woody debris across the landscape. Cutting for lumber (at least since 1940) has not yet been extensive at low elevations on the BNF, and the extent of firewood cutting is unknown. Therefore, 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 (high confidence). Coarse woody debris at low elevations may have been less than at high elevations during the reference period because of frequent surface fires that provided more frequent opportunities for its combustion. As at the stand level, we doubt that coarse woody debris and snags are outside the HRV. Of course, outbreaks of insects or various diseases can lead to rapid increases in coarse woody debris at both high and low elevations.

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 et al. 2000).

7. SUMMARY OF PROBABLE HRV DEVIATIONS

Based on the literature and data that we reviewed, and our understanding of the ecology of the BNF, it seems clear that human-caused deviations from the HRV are occurring in some areas, primarily because they are subjected to timber harvesting, fire suppression, and livestock grazing. Does this matter? An answer is beyond the scope of this report. However, deviations from the HRV, where they appear to be occurring, should be evaluated through monitoring and research to determine if the deviations could lead to undesirable consequences.

As our report describes, forest disturbances on the BNF are not new. Indeed the forests have evolved with regular disturbances, and the historical record indicates that significant human-caused disturbances have occurred since the late 1800s. Moreover, many forested areas have been subjected to numerous episodes of burning during the HRV period. Still, only ~18% of the forested area has been harvested on the BNF (high and low-elevation forests combined). Roughly 12% has been harvested on two occasions. In contrast, the effects of fire suppression and livestock grazing have been widespread.

Based on our understanding of BNF ecosystems, we have attempted to draw conclusions about whether or not certain ecosystem variables are now beyond the HRV they experienced during the period 1600-1890, or trending in that direction (Table 7).

At high elevations the following variables estimated at the stand scale appear to be within the HRV (confidence level is indicated in parentheses):

- Fire return interval and intensity (high)
- Insect outbreaks (moderate)
- Abundance of diseases, except for dwarf mistletoe in some areas (low)
- Blowdowns (high)
- Tree, sapling and seedling density (high)
- Plant species diversity (low)
- Understory density and cover (high)
- Age- and size-class structure of most stands (moderate)
- Forest floor depth (high)

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

- Fire intensity (high)
- Insect outbreaks (moderate)
- Abundance of diseases except for dwarf mistletoe (low)
- Blowdowns (high)

Tree, sapling and seedling density (moderate)
Understory density and cover (moderate)
Plant species diversity (low)
Forest floor depth (moderate)
Number and proportion of land cover types (high)
Forest/non-forest ratio (high)

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
Blowdowns (high)
Tree species diversity (high)
Snag density and abundance of coarse woody debris, except where timber harvesting has occurred (moderate and low, respectively)

Similarly, the following landscape variables appear to be within the HRV at low elevations:

Insect outbreaks (low)
Abundance and composition of diseases, except where the introduced white pine blister rust occurs (moderate)
Blowdowns (high)
Tree species diversity (high)
Proportion of landscape with old trees (moderate)
Proportion of land in high density snags and coarse woody debris (low)
Proportion of land in different land cover types (low)

For aspen stands, 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 (low), except near campgrounds
Blowdowns (high)

Very little information is available for drawing conclusions about the HRV of grassland, meadow and shrubland variables on the BNF. However, fire return interval, size, and intensity appear to be within the HRV, probably at both the stand and landscape scales.

In contrast, the following variables probably are beyond their HRV:

Within stands at high elevations affected by timber harvest (~20% of the high-elevation forests on the BNF):

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. Where timber harvesting has occurred, snag density and the amount of coarse woody debris is 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 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 may be below its HRV in clearcut stands (low confidence).

Across forested landscapes at high elevations:

6. 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 has been 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. The proportions of other age-classes are within the HRV, except for the now more abundant mid-successional forests (low confidence)
7. The average snag density and coarse woody debris biomass across the landscape have been reduced below the HRV (moderate confidence). The loss of some old-growth forests and the suppression of fires that create abundant snags have reduced the percentage of the forested landscape that contains high density snags/coarse woody debris to a level that is below the HRV (high confidence).
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 are over-represented. Natural fires produce many small patches and a few very large patches at high elevations. Fire return intervals, size, and extent also have been reduced below the HRV due to fire suppression (low confidence).

9. The percentage of the landscape with low canopy cover probably is higher than the HRV due to partial cutting and thinning (low confidence).
10. 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 parts of the BNF (low confidence). Correlated with this and fire suppression is a decline in the maximum 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 (high confidence).
11. Some kinds of mechanical slash treatment have led to a higher level of mineral soil disturbance than would have occurred during the HRV reference period (high confidence)
12. Dwarf mistletoe may be somewhat above its HRV on the landscape scale due to the tie-hacking, selective cutting and thinning that has occurred on the forest (low confidence).

Within stands at low elevations affected by harvest, fire suppression and livestock grazing:

13. Fire suppression, livestock grazing, and possibly climate change have led to the development of low-elevation forest stands with higher densities of young trees, greater canopy cover, less understory plant cover, and deeper forest floors than usually occurred on many sites before 1890 (see Table 7 for confidence levels associated with each variable). Understory composition also has changed where introduced plants occur.
14. By removing large trees and downed wood from the system, wood harvesting has lowered the abundance of snags and coarse woody debris in managed stands of low-elevation forests (moderate confidence). Harvesting also has led to an increase in sapling density and a reduction in the average diameter of trees, thus creating younger, more uniform-sized stands than existed before harvest (moderate confidence). However, only about 1% of such forests have been subjected to harvesting thus far.

Without harvesting and with fire suppression, snags and coarse woody debris probably are still within the HRV.

15. Mean fire return interval has become longer than the HRV due to fire suppression (moderate confidence). Consequently, there is a trend for tree distribution to be more uniform and to have tree invasion into some meadows.
16. White pine blister rust outbreaks cause some stand variables to exceed the HRV in affected stands, as this disease is not native (high confidence).

Across forested landscapes at low elevations:

17. Similar to the stand-scale, the sizes of most fires have declined due to fire suppression, and the mean fire interval, tree/sapling density, and amount and continuity of fuels is above the HRV in a larger proportion of the BNF than during the reference period (moderate confidence).
18. Understory plant density and cover have declined below the HRV (low confidence) and forest floor depth has increased above the HRV in a larger proportion of the landscape than during the reference period (moderate confidence).
19. White pine blister rust was absent during the HRV reference period, but is present today (high confidence).
20. The age- and size-class structure of trees (moderate)

Aspen forests:

21. In some areas, livestock or native ungulate grazing and browsing have reduced aspen densities and affected the understory vegetation to a point where these variables are exceeding their HRV (high confidence). These changes are confounded with the introduction of exotic plants.
22. 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).

23. Due to human wounding of trees in recreational areas, some stands have an incidence of disease that is beyond the HRV (low confidence).

Non-forest vegetation:

24. Some meadows, grasslands, and shrublands are probably grazed more, or have been in the past, than during the HRV period, with the potential for an increase in soil erosion and forb and shrub abundance (high confidence). Deviations from the HRV are also caused by the presence 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. Such variables can be identified as 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 ponderosa pine in some areas 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 BNF—sometimes over large areas. Such trees create habitat for organisms that cannot survive on or around healthy trees. The biological diversity of an ecosystem, which assures their long-term survival 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 really has been effective on specific sites and whether or not the "treatment" creates more problems 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.

In general, a continuing challenge is to determine if there are options for reducing the undesirable effects that can be associated with extracting wood, suppressing fires and grazing livestock while providing the other resources and amenities that are desired from national forests. Considering the HRV for key variables can be useful in achieving that goal.

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8. APPENDIX

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

Fire database (1970-1996)	Locations of the start point for each fire is 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 BNF 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.
Large fire database (1909-1996)	The spatial extents of large fires from 1909 to 1996 were originally drawn on topographic maps. The polygons were crude on early maps and became more refined by the 1980s. The smallest mapped fire was 9 ha, but that does not necessarily mean all fires > 9 ha were always mapped. It is unknown if the mappers used a size criteria when deciding which large fires to map. Also, a couple large fires in the late 1980s, recorded in the above fire database, have not been mapped out spatially; these two fires were \leq 540 ha.
RIS database	Stand ages were calculated as follows: Cores were taken at breast height, and then 15 years were added to the number of rings counted to the pith for softwoods, 4 years were added for hardwoods, and 30 years for spruce to account for growth to breast height. Error of \pm 10% in age was acceptable. Each forest stand had 1 sample point per 4 ha and a minimum of 5 points per stand. At each point, the first two live trees in the northward direction that were > 12.5 cm dbh were measured for age. The age was averaged over trees and plots. Only certain proportions of stands in each forest type were sampled (Fig. 27 for percentages).
RIS database	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 (31% more) 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.
RIS database	The seral stage map in RIS appears very approximate and sometimes appears to conflict with the tree size map. It seems to be acceptable for broad-scale landscape patterns.
RIS database	Cover type classifications were conducted by separate teams across the Forest, resulting in some inconsistencies. A new (IRI) database improves on these greatly. However, at the Forest-level of analysis, the differences between the RIS and IRI maps is small. The IRI database was not fully available at the time of the preparation of this report.
Romme (1979, 1982), Romme and Despain (1989) data compared to RIS data	As Tables 5 and 6 demonstrate, the classifications used in YNP 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 estimate of the BNF silviculturalists (USDA Forest Service 1994). Also, area of the landscape in each of Romme's classifications by seral stage (Fig. 26) was estimated using a dot grid on his maps in his 1979 dissertation and by crudely estimating areas burned during the 1988 fires by examining a 1988 satellite image of the park.

9. TABLES

Table 1. 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	Tropical
300 mybp	swamps, lagoons, tidal flats	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 of BNF uplifted during Laramide orogeny; drifting apart of continents; climate still tropical, snow rare, much wetter than today	Flowering plants more common in upland forests, 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	
7-5 mybp	Increased aridity, creation of rainshadows	Demise of some forests, giving rise to shrublands and grasslands; draught 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); climate cooling, heavy snows, further cooling 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 regionally extinct)
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 elevational band; lowlands tundra-like
11,500 ybp	Temperatures 5-6° C cooler than today; retreat of glacial ice	Upper treeline about 600 m 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; Ponderosa pine expanded into the Bighorn Mountains during the last 2000 years (Jodi Norris, personal communication).
Ca. 1350 – 1500 AD and 1700 – 1900 AD	Characterized by cooler temperatures (during “Little Ice Age”), but also long, severe droughts (Gray 2003; Gray et al., in press).	Potential glacial advances in the mountains
1850 AD – present	Generally warmer and wetter than Little Ice Age; Twentieth century warmer than previous 1000 yrs, but less severe droughts than in previous centuries (Gray 2003)	Increased tree recruitment near upper treeline (Hessl and Baker 1997), in subalpine meadows (Jakubos and Romme 1993) and in montane forests (Savage et al. 1996)

Table 2. Vegetation of the Bighorn National Forest, as classified at various levels of detail. RIS = Resource Information System;

This Report	BNF RIS (dominant species)	Hoffman and Alexander 1976
High elevation forests	Engelmann spruce-subalpine fir Lodgepole pine	3 subalpine fir habitat types 1 Engelmann spruce habitat type 2 lodgepole pine habitat types
Low elevation forests and woodlands	Ponderosa pine Douglas-fir Limber pine Rocky Mountain juniper	5 ponderosa pine habitat types 2 Douglas-fir habitat types
Aspen and riparian forests	Aspen Cottonwood Willow	1 aspen habitat type
Non-forest vegetation (grasslands, shrublands and forblands, including alpine vegetation)	Grasslands: fescue, oatgrass, tufted hairgrass, bluegrass, wheatgrass, needle and thread, kobresia, bluejoint reedgrass, rushes, sedges, general Shrublands: sagebrush, mountain mahogany, bitterbrush, skunkbrush, shrubby cinquefoil, general Forbland: forb willow, general	

Table 3. Examples of ranges for average fire intervals in high elevation forests, as estimated by various investigators. These ranges represent the typical amount of time between successive fire events in an individual stand. MBNF = Medicine Bow National Forest, YNP = Yellowstone National Park, BNF = Bighorn National Forest (not available for pre- and post-settlement periods).

Study	Forest type / location	Fire return interval estimate (years)	Type of fire event	Basis of estimates
Billings (1969)	Spruce-fir / MBNF	up to 600-700	stand-replacing	stand age structure
Arno (1980)	Subalpine forest / N. Rockies	150-350	stand-replacing	stand age structure
Romme (1980a, 1982)	Lodgepole, fir, spruce / YNP	300+	stand-replacing	stand age structure / fuel accumulation
Romme and Knight (1981)	Spruce-fir / MBNF	300-500+	stand-replacing	stand age structure / fuel accumulation
Romme and Despain (1989)	Lodgepole, fir, spruce / YNP	200-400	stand-replacing	stand age structure / fuel accumulation
Hawkes (1980)	Lodgepole, fir, spruce, larch / Alberta	90-304 ¹	all	empirical calculations based on fire scarred trees
Kipfmüller (1997)	Lodgepole, fir, spruce / MBNF	39-149 ²	all	empirical point-scale calculations based on 73 fire scarred trees
Bornong (1996a)	Lodgepole habitat type, BNF	59 ³	all	empirical point-scale calculations based on 15 apparently fire scarred trees
Bornong (1996a)	Spruce-fir, BNF	88 ³	all	empirical point-scale calculations based on 4 apparently fire scarred trees

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).

³Based on overall average of all intervals (8 different stands in North Fork of the Powder River Watershed). Note: Bornong was unsure whether or not some scars were caused by insects.

Table 4. Examples of fire return intervals reported for stands of ponderosa pine and Douglas-fir in the Rocky Mountains. 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 the first three studies in this table.

Study	Forest type / location	Mean fire return interval estimate (years)		Type of fire event	Basis of estimates
		Pre-settlement	Post-settlement		
Brown et al., 2000	Ponderosa pine / Laramie Range, WY (Ashenfelder Basin)	<i>median</i> = 26-33.5 (8-82)	>86 – 88	surface	fire scar analysis
Brown et al. 1999	Ponderosa pine-Douglas-fir Central Colorado	<i>widespread fires</i> 1496-1851: 59.2 (27-128+)	<i>Widespread fires</i> 1880-1996:> 116	surface	fire scar analysis
Fisher et al. 1987	Ponderosa pine / Devils Tower Nat'l Mon., WY	1600-1770: 27 1770-1900: 14 <i>overall</i> : 19	≥ 1900: 42	surface	fire scar analysis
Dieterich 1980	Ponderosa pine / northern Arizona	(2-35)	----	surface	fire scar analysis
Goldblum and Veblen 1992	ponderosa pine-Douglas-fir / Boulder, CO	<i>before 1859</i> : 31.8	1859-1920: 8.1 ≥ 1920: 28	surface	fire scar analysis
Brown 1994; Brown and Sieg 1996	Ponderosa pine Jewel Cave Nat'l Mon., SD	20-32 (5-79)	----	surface	fire scar analysis
Laven et al. 1980	Ponderosa pine Roosevelt Nat'l Forest, CO	<i>before 1840</i> : 66 (5-157) 1840-1905: 17.8 (3-161)	1906-1973: 27.3 (8-46)	surface	fire scar analysis
Shinneman and Baker 1997	Ponderosa pine Black Hills Nat'l Forest	≤ 250-300+	----	crown fires	historical observations; stand ages
McCune 1983	Douglas-fir Montana	60	----	not specified	
Arno and Gruell 1986	Douglas-fir Montana	35-40	----	not specified	
Houston 1973	Douglas-fir Yellowstone Nat'l Park, WY	20-25	----	surface	fire scar analysis

Table 5. Successional stage characterizations for lodgepole pine and spruce-fir forests, used when comparing Yellowstone National Park (YNP) to Bighorn National Forest.

Yellowstone National Park (YNP) –Romme and Despain 1989			Bighorn National Forest (BNF)—from RIS database and USDA Forest Service 1994	
SUCCESSIONAL STAGE	YEARS AFTER STAND-REPLACING FIRE	CHARACTERISTICS	SUCCESSIONAL STAGE	YEARS AFTER STAND-REPLACING FIRE (from USDA BNF 1994)
Lodgepole 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	Grass-forb-shrub-seedling (structural stage 1 and 2)	about 0 to 40 years
Immature lodgepole (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	Sapling-pole (structural stage 3)	about 40 to 100 years
Mature lodgepole (LP2)	150 to 300	Tree growth, competition, and thinning decrease; canopy more open; groundlayer denser and more diverse; mixed understory of lodgepole, spruce, and fir common; dead, woody fuels increase but still somewhat resistant to fire, most fires will drop and die out in absence of wind	Mature (structural stage 4)	about 100 to 300 years
Lodgepole, spruce-fir (LP3)	> 300 years	Canopy mortality increases; understory trees reach 2-3 m tall and penetrate gaps; fuels and organic materials build up, burn relatively easily when weather conditions are right, burn more or less continuously and intensely. Last stage before next stand-replacing fire event in southcentral YNP.	Old-growth ¹ (structural stage 5)	Trees are usually > 300 years, but can be > 200 years. Lodgepole may be as young as 150 years; this category still needs to be verified in the field.

¹Old-growth on the BNF is not based just on age and is structurally defined in Mehl (1992), but not all of the old-growth on the BNF has been inventoried on the ground, so any conclusions from these data are tentative.

Table 6. Comparison of community type definitions used in comparisons of Little Firehole River watershed with Bighorn National Forest watersheds.

YNP Successional Stage (Romme 1982) ¹	Age of stand (Romme 1982)	BNF Successional Stage (from RIS) ²
Early successional stages	0- 40 years	Grass-forb-shrub-seedling (< 3 cm dbh)
Middle successional stages	40 – 200 years	Sapling-pole, mature (3 – 23 cm dbh, > 40% canopy cover)
Late successional stages	> 200 years	Very large mature (> 23 cm dbh)
Riparian forest	--	Riparian forest
Steep slope, sparse forests	--	Low density sapling-pole, mature forest (< 40% canopy cover)
Meadow	--	Non-forest

¹Only lodgepole and spruce-fir vegetation types occur in the study area.

²Romme's classifications are based on stand ages (different than in Table 5) that were best matched by searching for the successional stage or size structure that matched the correct stand age. Very large mature trees were almost always > 200 years old in the RIS database. However, this approach ignores old trees > 200 years that are small, such as those in doghair stands (Table 5 better captures those trees). The emphasis of this approach is on late-successional stage structure that includes large trees. Such large trees provide important habitat for many species.

Table 7. Summary of variables describing upland vegetation characteristics on the Bighorn National Forest relative to their estimated historic range of variability (HRV) from about 1600 to 1890, with an indication of our confidence in making each comparison at the stand (S) and landscape (L) scales. Stand characteristics are evaluated for areas modified by human activities only. Landscape characteristics are evaluated across the landscape for parts or all of the BNF. For our discussion of each variable, refer to the pages listed in the column on the right. At high elevations, stand scale analyses focus on harvested areas, whereas at low elevations the stand scale analyses focus on unharvested stands. This is because very little timber has been harvested at low elevations and fire suppression has had a greater effect there.

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	S-High L-Low	36-45
Fire intensity	S,L		High	36-45
Fire size		L	Low	36-45
Insect epidemics	S,L		Moderate	53-58
Dwarf mistletoe abundance		S,L	Low	64-67
All other diseases	S,L		Low	63-64, 66-69
Wind	S,L		High	69-71
<i>Stand structure at high elevations</i>				
Density of trees in all age and size classes	S,L		S-High L-Moderate	74-76
Regeneration time and seedling/sapling density	S,L		S- High L-Moderate	76-77
Tree canopy cover and the density and size of canopy gaps		S,L	S-Moderate L-Low	77-78
Understory plant density and cover	S,L		S-High L-Moderate	78-79
Diversity of all plants	S,L		Low	79-81
Age and size-class structure of trees in most stands	S	L	Moderate	81-82
Age- and size-class structure of trees on some north slopes, leeward slopes and ravines		S	Moderate	81-82
Forest floor depth	S,L		S-High L-Moderate	82-83

Mineral soil affected		S,L	High	83-84
Snag density		S,L	High	84-85
Coarse woody debris		S,L	High	85-87
<i>Landscape features at high elevations</i>				
Proportion of landscape in low canopy cover		L	Low	107-108
Number and proportion of different land cover types	L		High	104-105
Forest/non-forest land area ratio	L		High	105
Proportion of forest land in old-growth and mid-successional stages		L	Moderate	107
Proportion of land with high snag and CWD densities		L	Moderate	108
Proportion of land in edge, interior forest, and patchy forest		L	Low	108-112
Rate at which new, uniformly distributed, small- to mid-sized patches are formed		L	High	112-113
Low elevation forests				
Disturbances				
Mean return interval for surface fires		S,L	Moderate	45-50
Fire size		L	Moderate	45-50
Insect epidemics	S,L		Low	58-62
White pine blister rust		S,L	High	67-69
Other diseases	S,L		Moderate	67-69
Wind	S,L		High	71-72
<i>Stand structure at low elevations</i>				
Tree density and canopy cover		S,L	S-High L-Moderate	87-90
Tree species diversity	S,L		High	90-91
Understory plant composition and cover		S,L	Low	90-91
Size- and age-class structure		S,L	Moderate	91-92
Spatial distribution of trees		S	Moderate	92
Forest floor depth		S,L	S-High L-Moderate	93
Snag density	S,L		S-Moderate L-Low	93

Coarse woody debris	S,L		Low	93-94
<i>Landscape features at low elevations</i>				
Proportion of landscape with high-density trees and high fuel continuity		L	Moderate	113-114
Proportion of landscape with low understory plant cover and relatively deep forest floors		L	Moderate	113-114
Proportion of landscape dominated by old trees	L		Moderate	116
Proportion of landscape with high-densities of snags and coarse woody debris	L		Low	117
Proportion of landscape in different cover types	L		Low	114-115
Forest patch sizes and configuration		L	Moderate	115-116
<i>Aspen woodlands</i>				
Insect epidemic frequency	S,L		Moderate	62-63
Blowdown frequency	S,L		High	71-72
Fire size and mean return interval at stand and landscape scales		S,L	Moderate	50-52
Disease abundance	S,L	S near recreation areas	Low	69
Conifer density in understory		S	Moderate	95-96
Understory plant composition with exotic plant species present		S	High	95-96
Representation in landscape	L		Moderate	95-96, 113-117
<i>Non-forest vegetation</i>	Little information available for drawing conclusions about specific variables, but see summary and text for discussion			52, 69, 96-99, 113-117

9. Figures

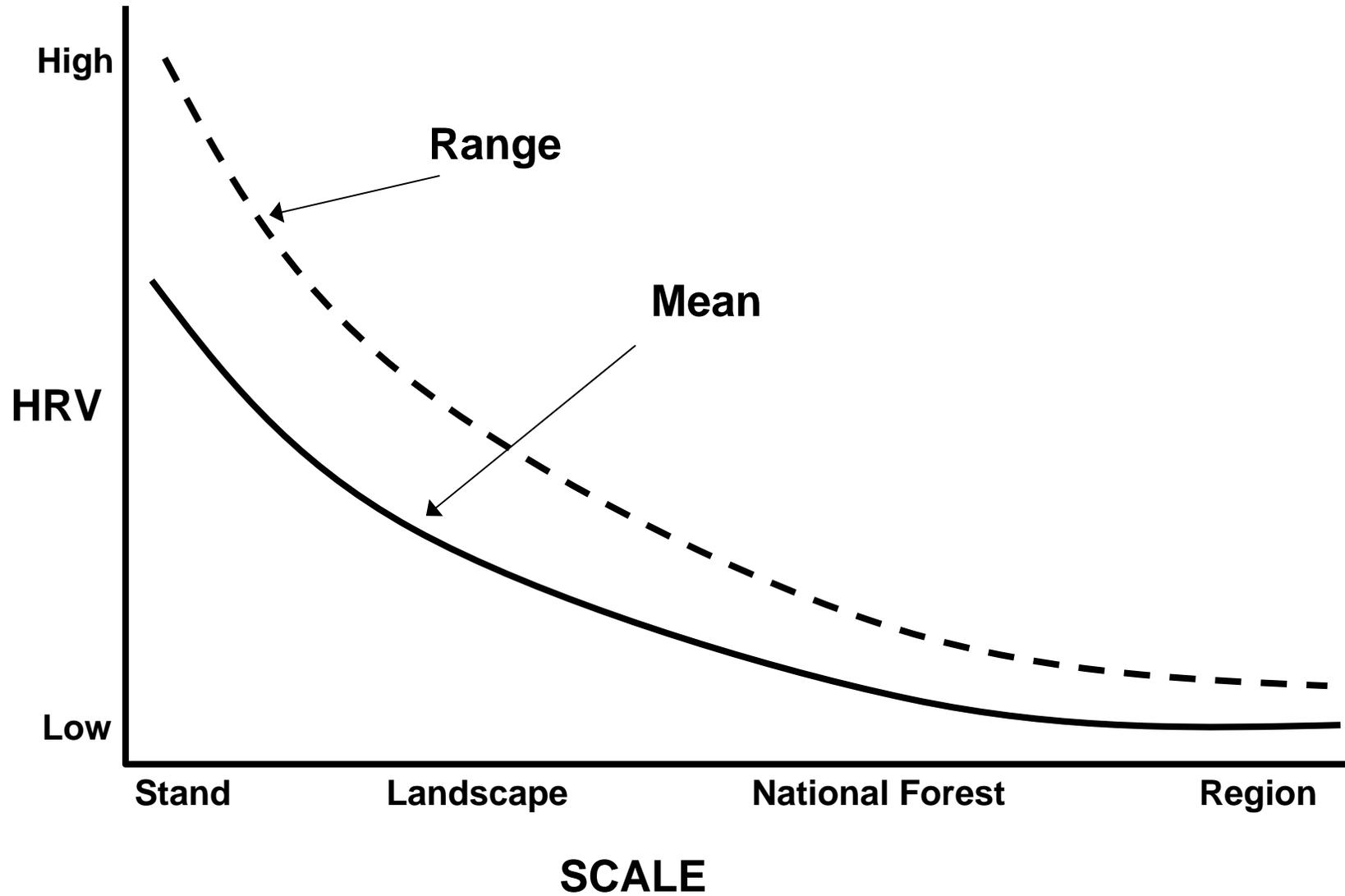


Figure 1. Illustration of the decline in the HRV of any variable as the spatial scale increases. Also, the mean (average) HRV of a variable is always less than the range between the maximum and minimum for the entire period.

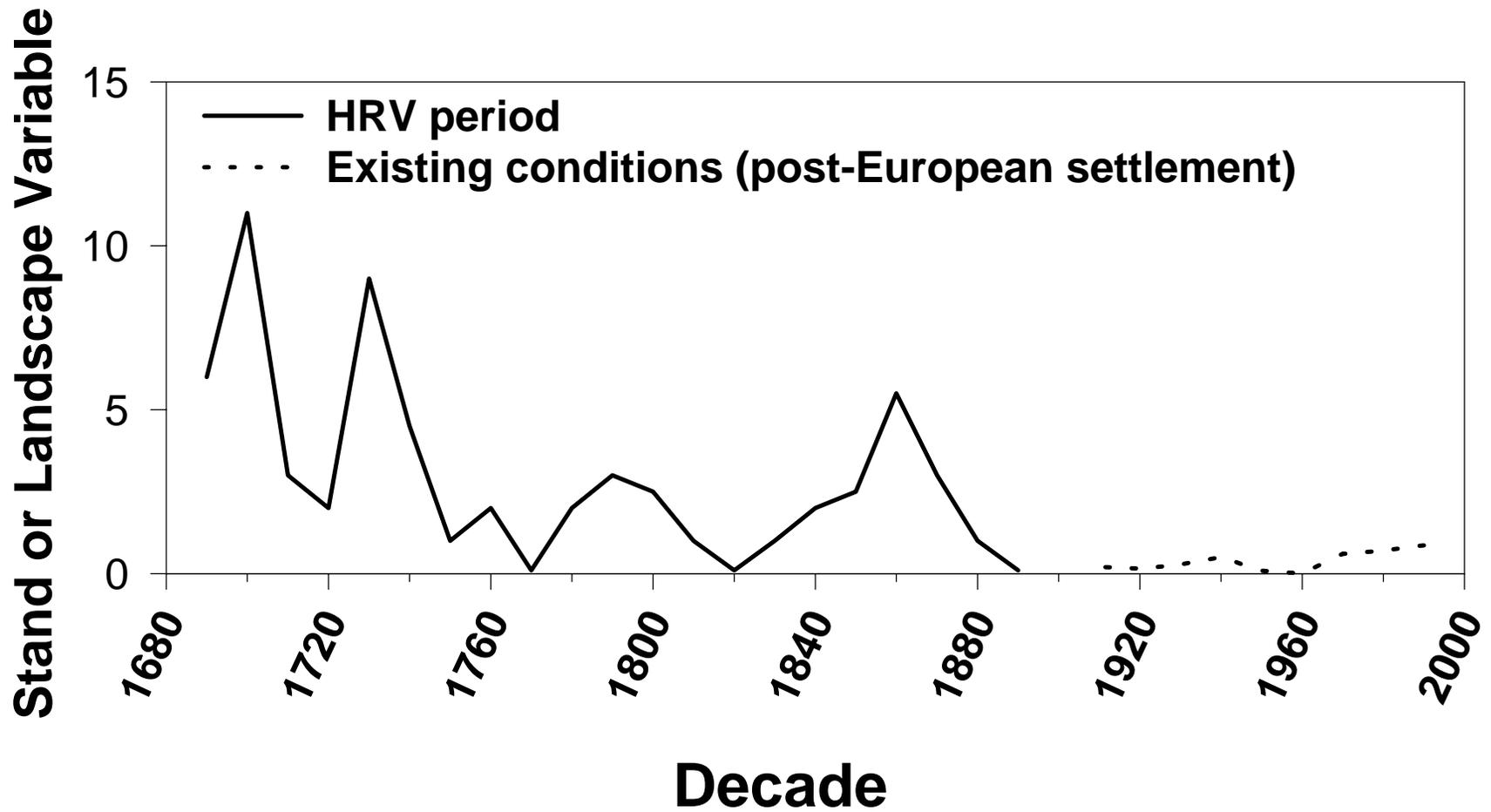


Figure 2. Illustration of the historic range for a variable, which incorporates extremes from 0 to 11 in the above example. If HRV is defined as the historic range, then existing conditions in the above example (ranging from 0 to 2) fall within the HRV.

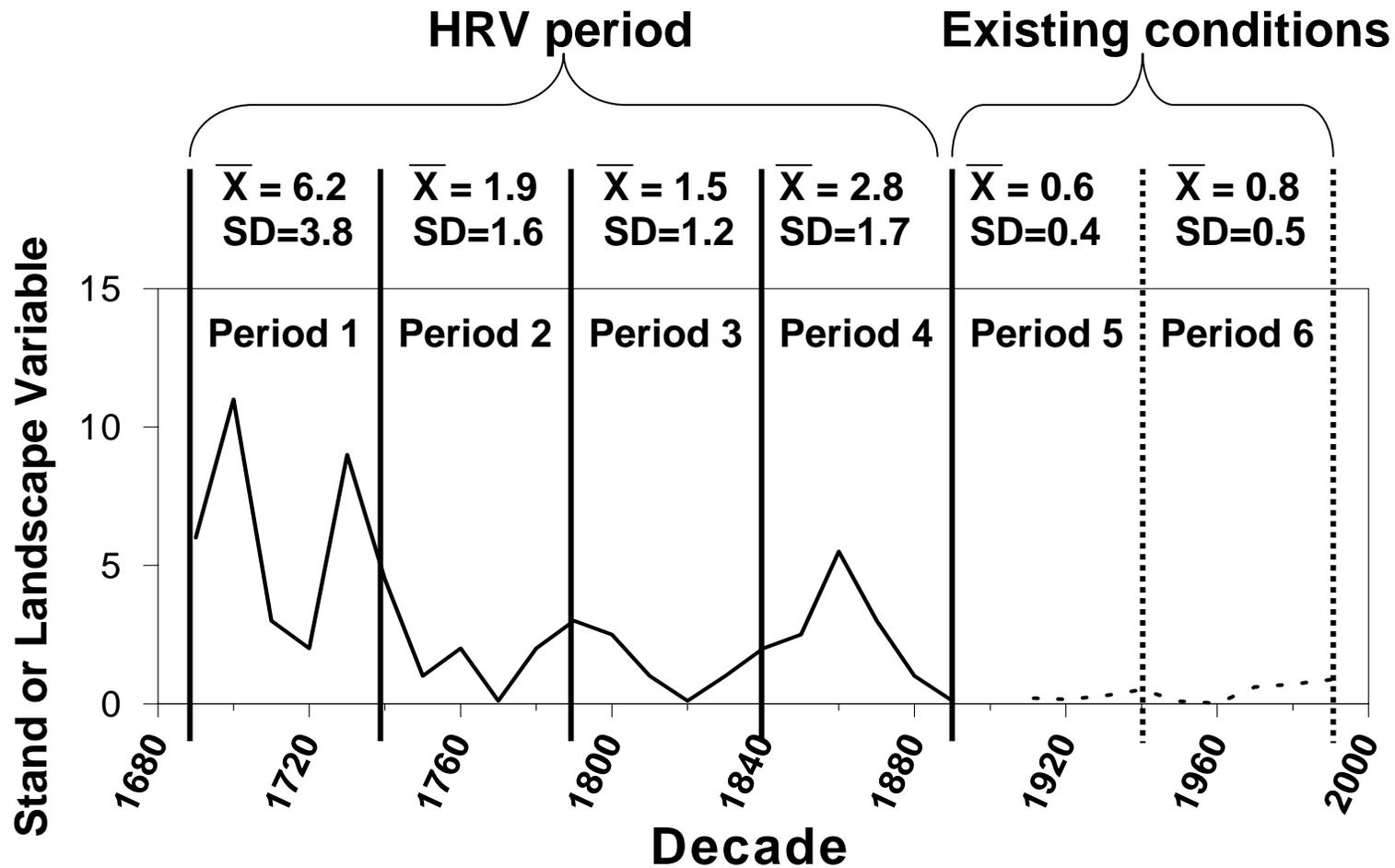


Figure 3. Illustration of the range of historic means (\bar{x}) for a variable, which ignores extremes. If HRV and Existing Conditions (EC) are defined as the range of means, the range of EC in this example (0.6-0.8) falls outside the HRV (1.5-6.2). Also illustrated is a comparison of the range of standard deviations (SD) of the variable, where the SD of the existing conditions (0.4-0.5) falls outside the SD for the historic conditions (1.2-3.8) during a series of consecutive time periods. We used the range of means (mean = average for each period) and standard deviations (SD = how variable the values are for the period) as the definition for HRV in this report, conceptually if not quantitatively. Note: Using a t-test on the means with $\alpha = 0.05$, the HRV of our example does not differ significantly from the EC ($P = 0.11$), probably due to the low sample sizes of $n = 4$ and $n = 2$.

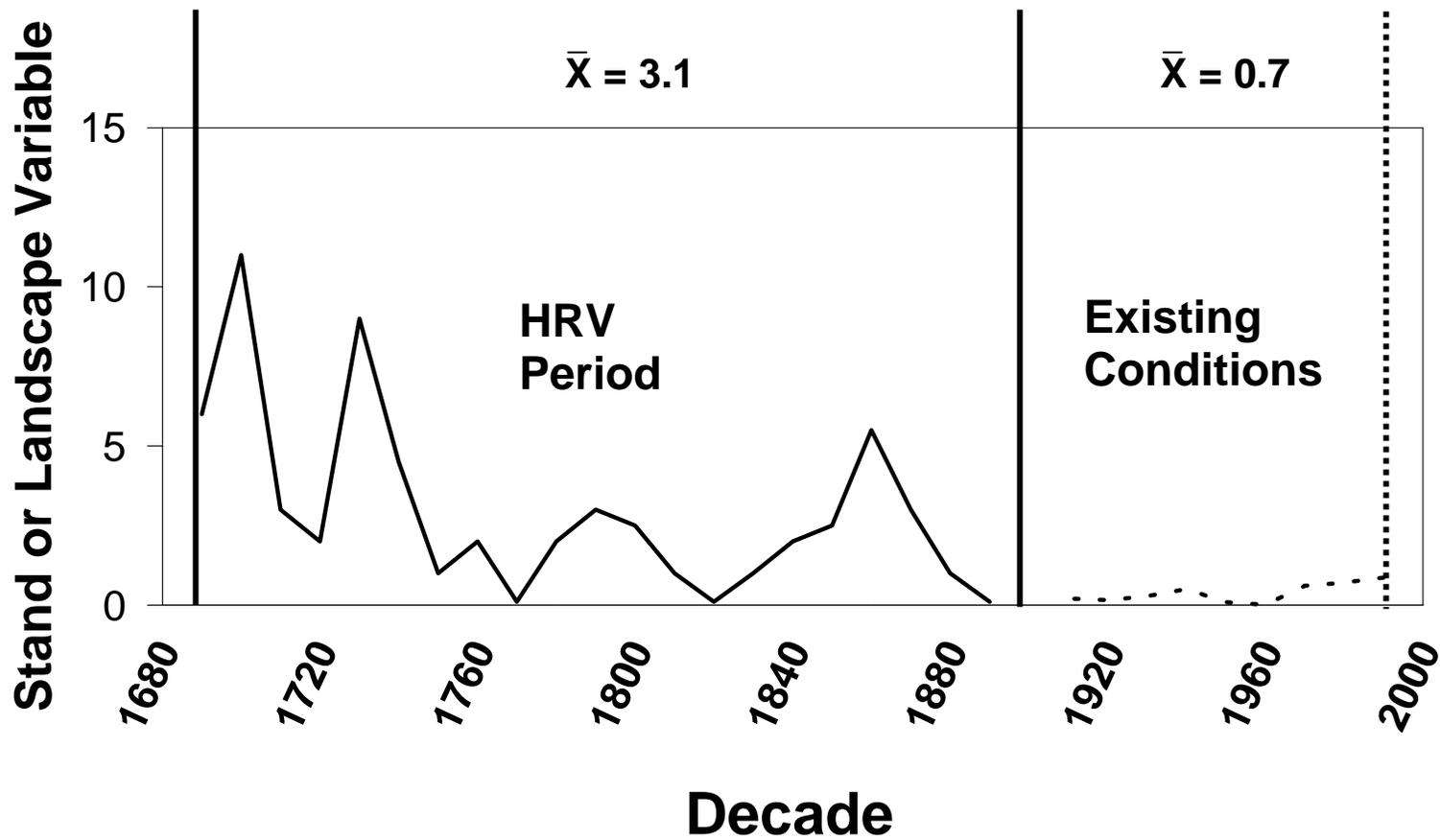


Figure 4. In many studies, only one mean is reported for the reference period, and thus, no range of means can be determined for the HRV (as in Fig. 3). In such cases, the mean for the existing conditions must be substantially greater or lower than the HRV mean to be considered a deviation from the HRV. In this example, an estimate of the HRV would have to be no more than $\pm 75\%$ of the historic mean of 3.1 to have the EC mean (0.7) still fall outside the HRV (0.8-5.4) using the same data as in Fig. 3.

Bighorn National Forest

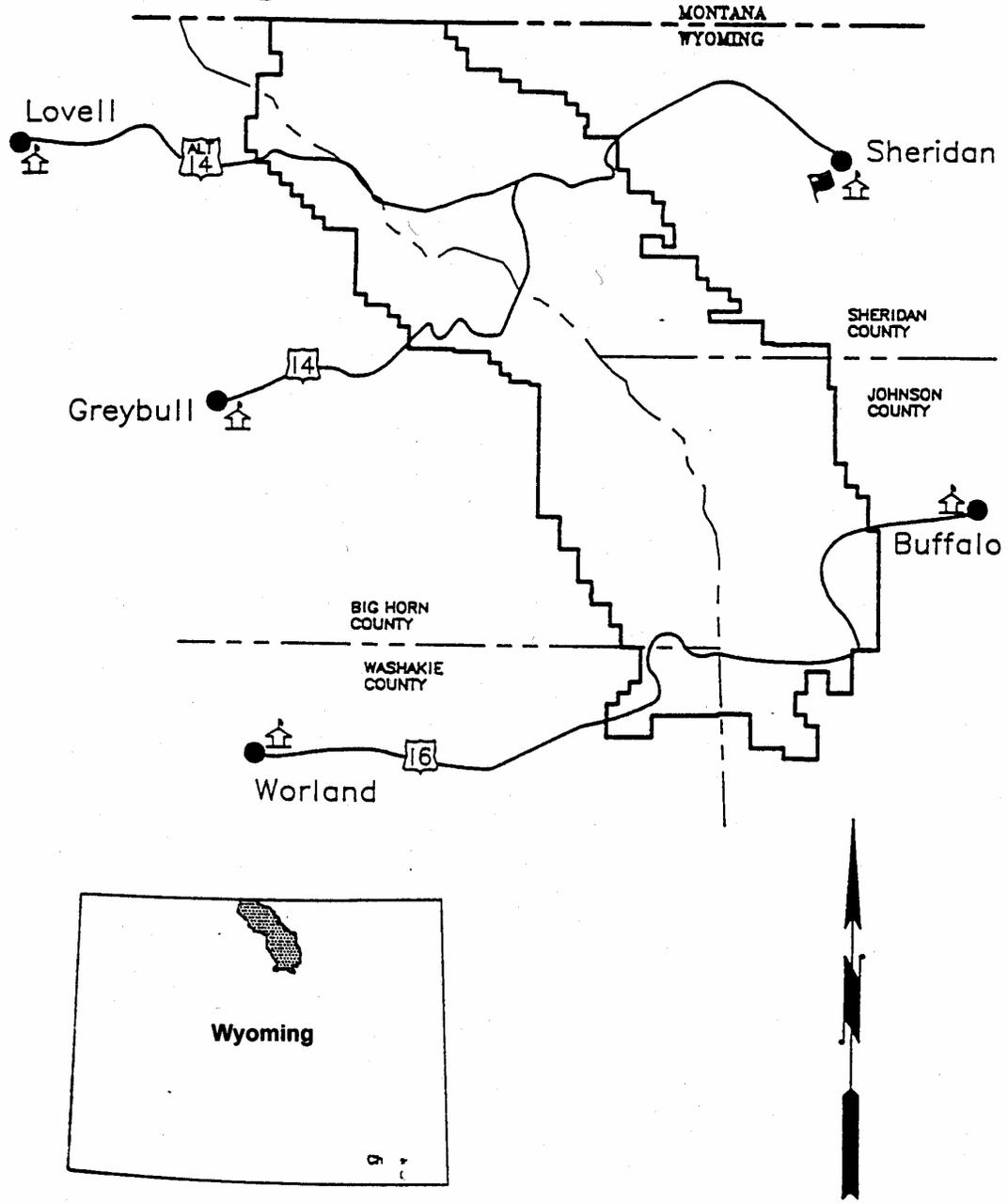


Figure 5. Location of the Bighorn National Forest in northcentral Wyoming (from USDA Forest Service 1994).

Bighorn National Forest Elevation Contours (m)

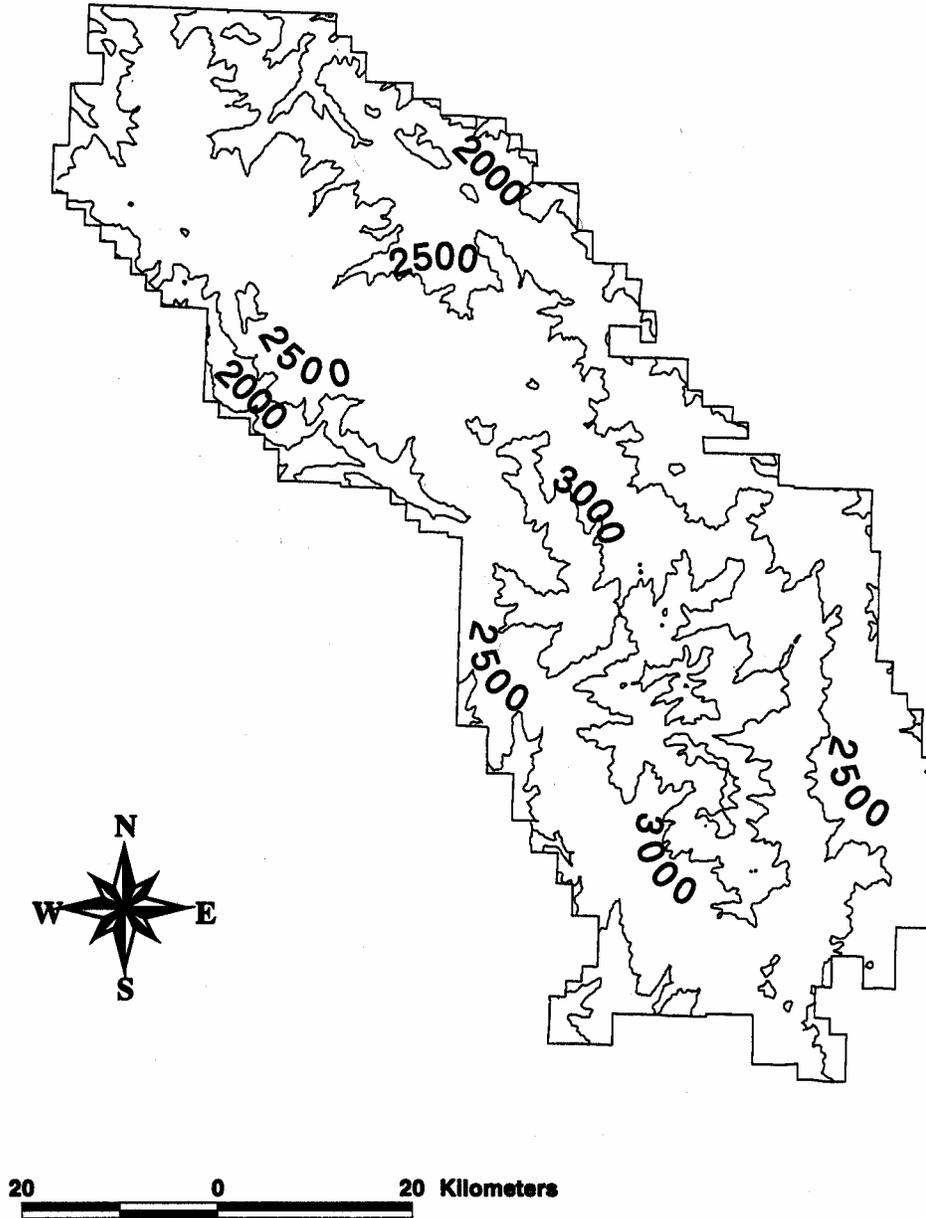


Figure 6. Elevation contours in meters on the Bighorn National Forest (from USGS DEM).

Bighorn National Forest Watersheds

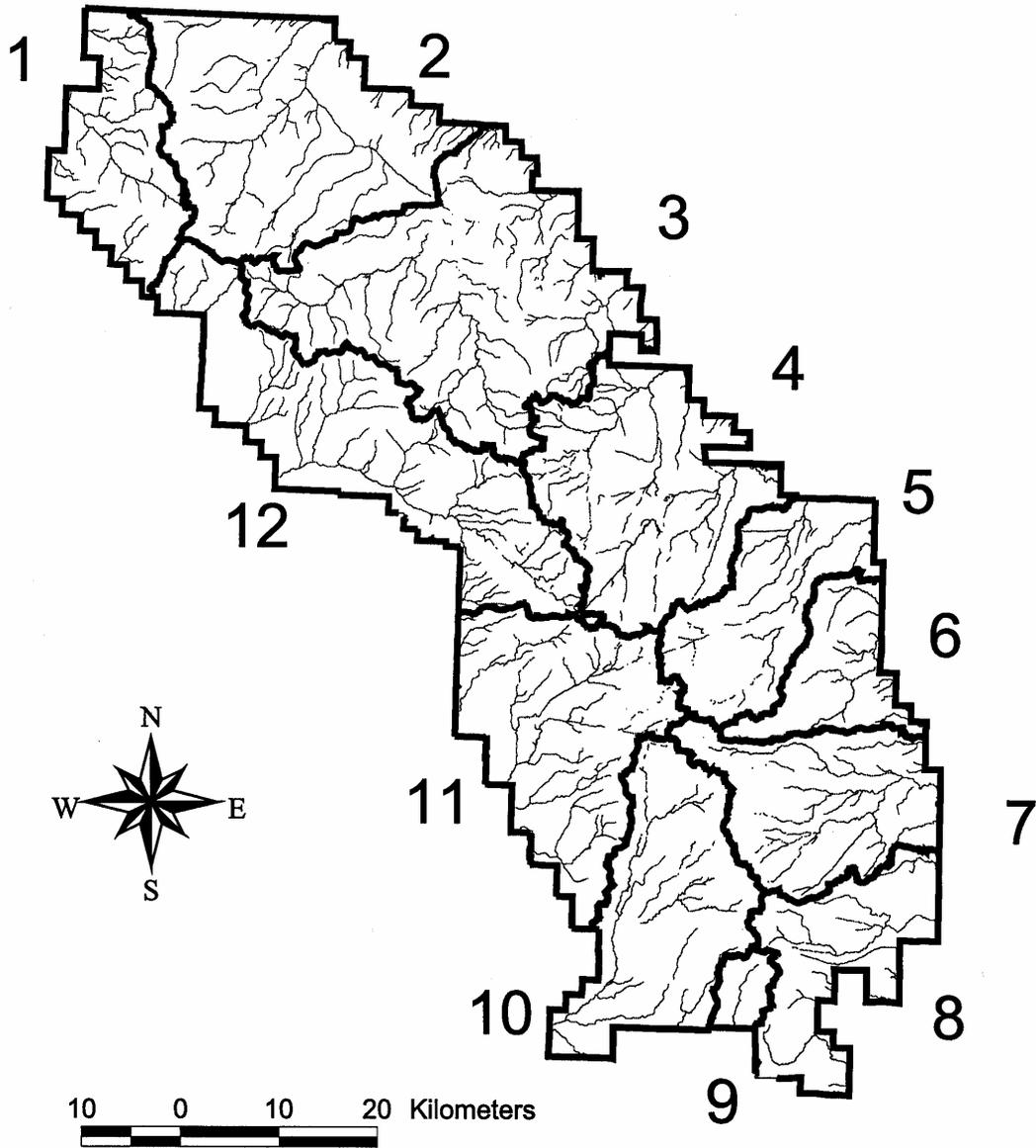


Figure 7. Stream network and watersheds on the Bighorn National Forest (from USGS DLG).

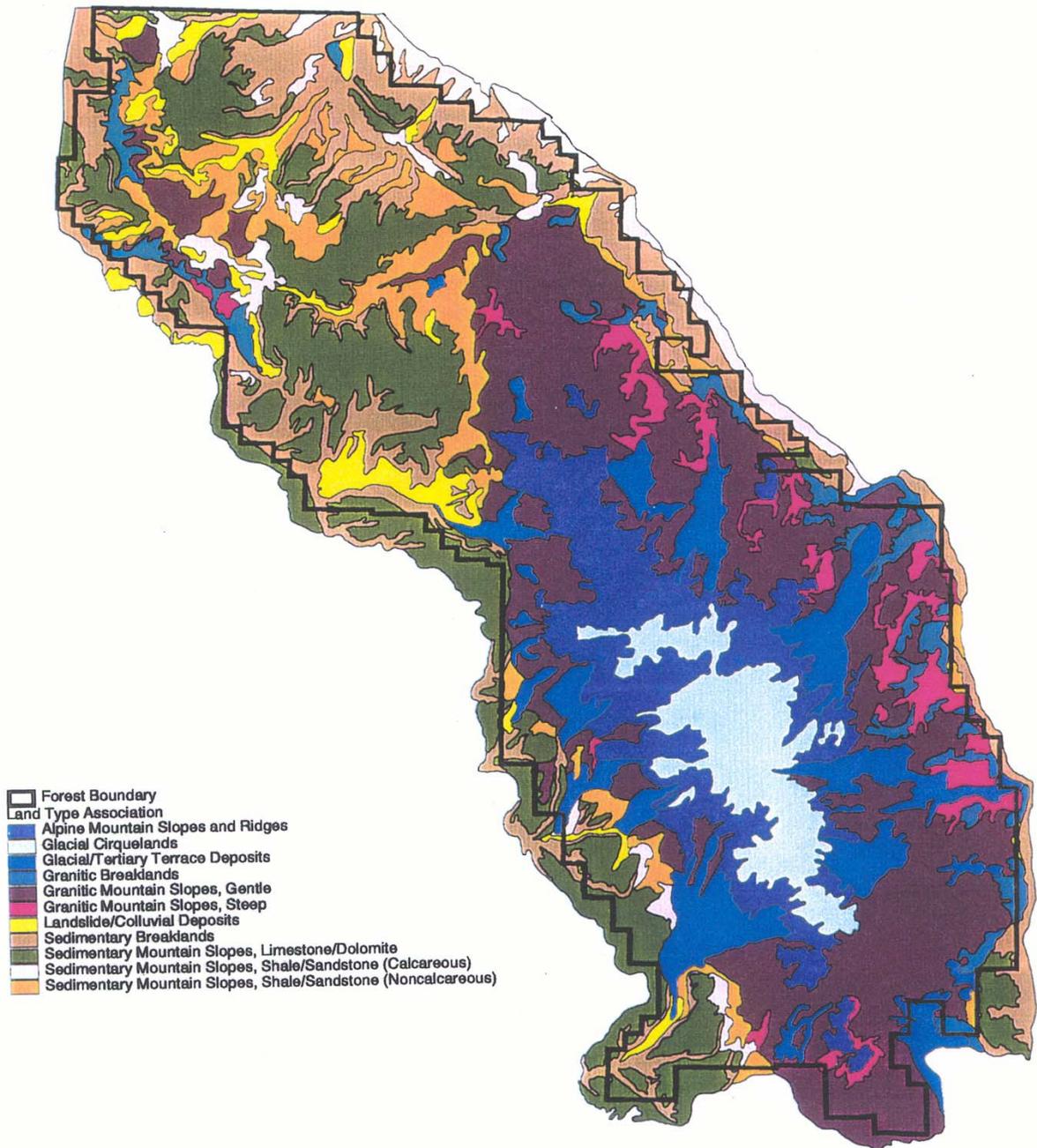


Figure 8. Land type associations (based on soils, topography, and geology) on the Bighorn National Forest (from Integrated Resources Inventory Center 1999).

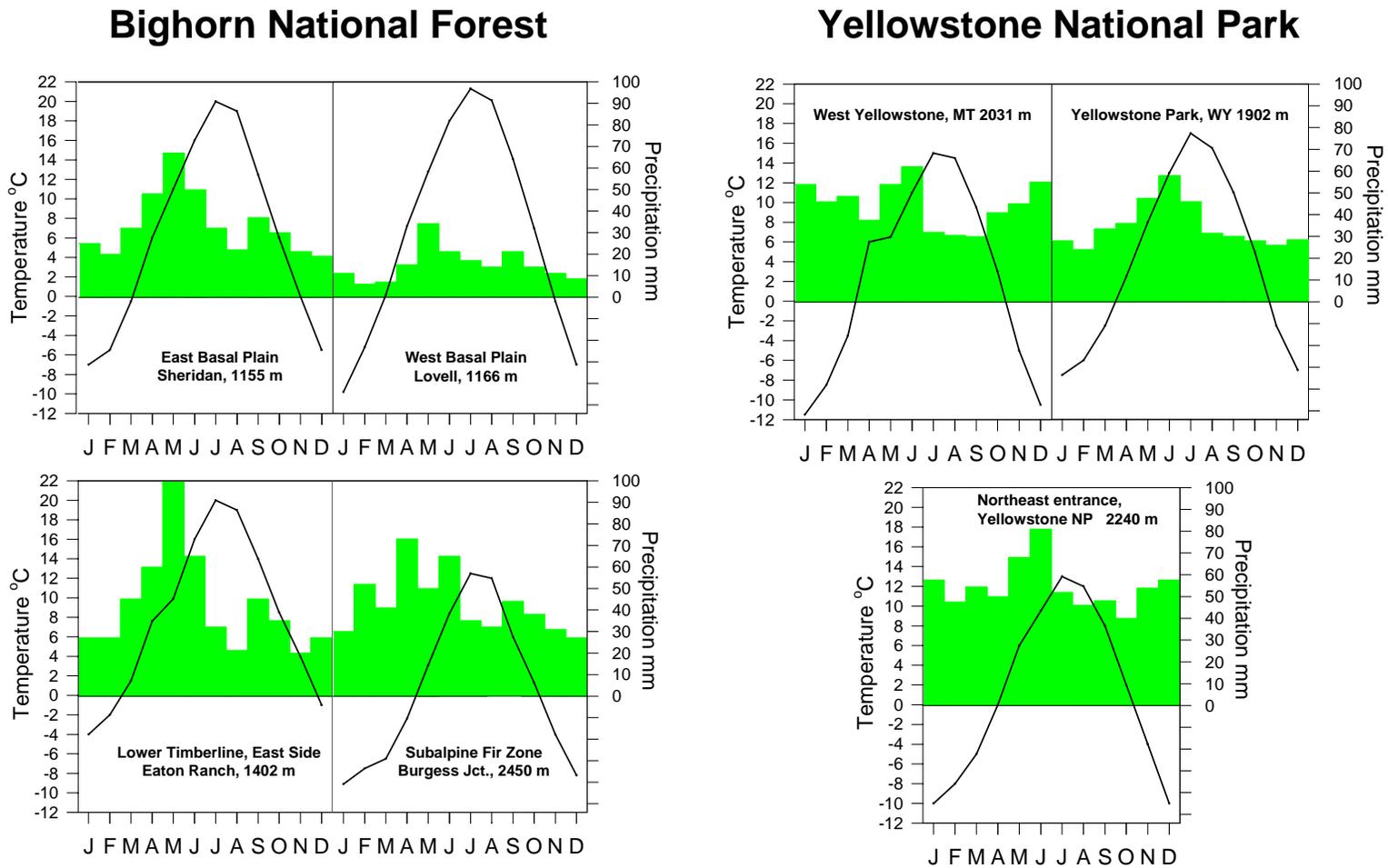


Figure 9. Climographs of mean temperature and precipitation for four weather stations in and near the Bighorn National Forest (from Hoffman and Alexander 1976) and Yellowstone National Park (from Steele et al. 1983).

Cover Types Bighorn National Forest

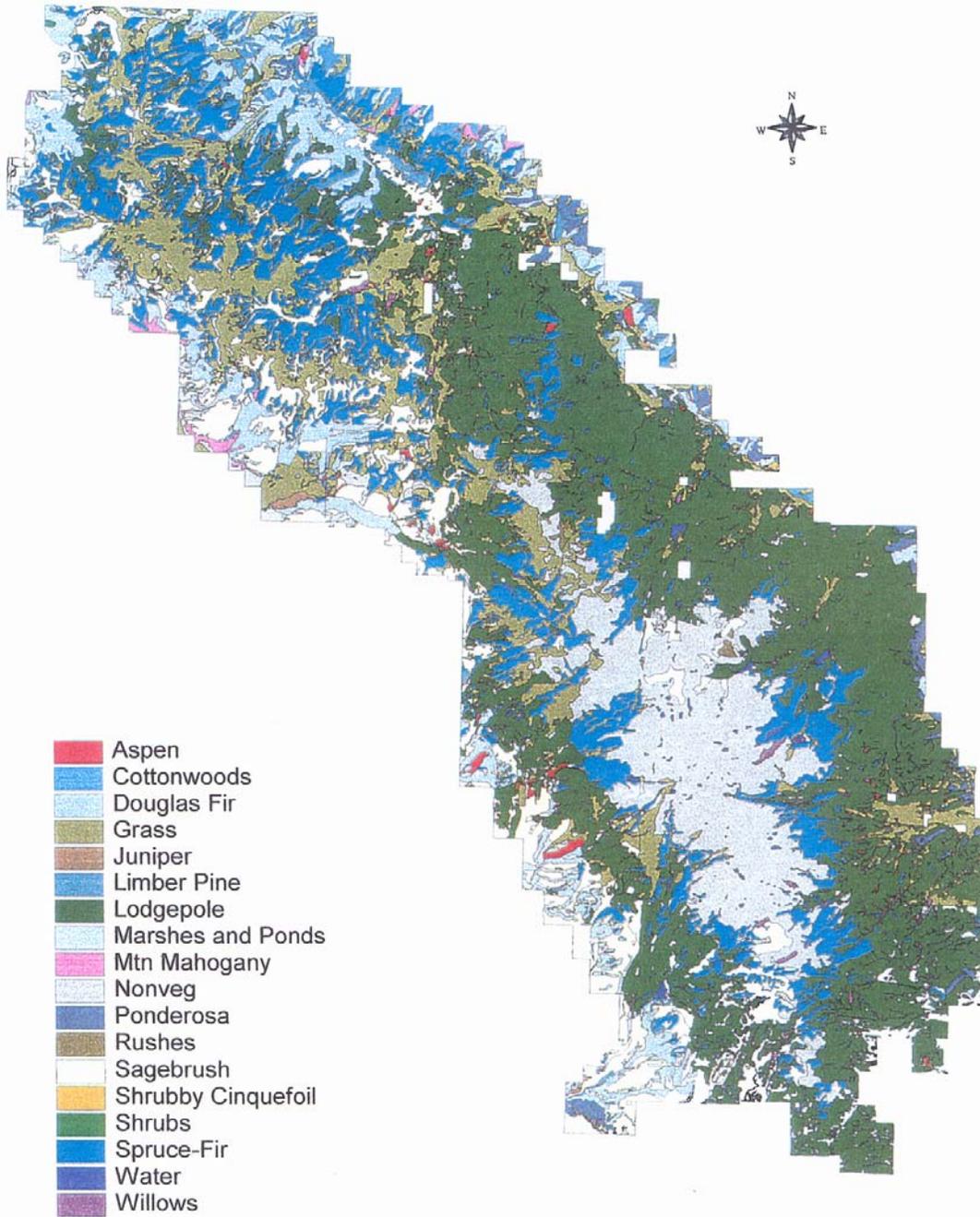


Figure 10. Vegetation cover types on the Bighorn National Forest (from RIS database).

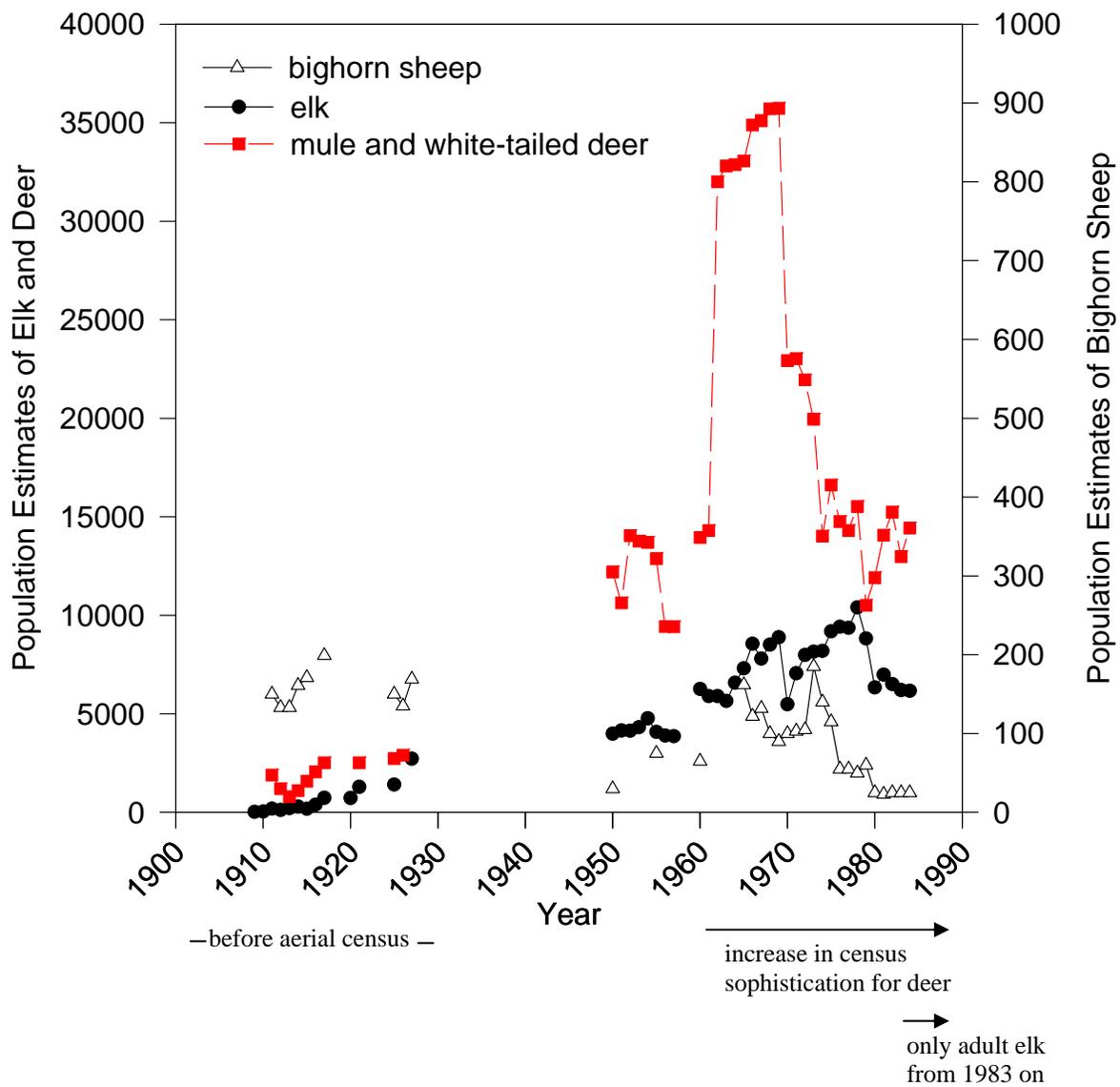


Figure 11. Population estimates for elk and deer (left axis), and bighorn sheep (right axis), on the Bighorn National Forest (data from USDA Forest Service 1984, 1994, and from 1909-1929 Annual Fish and Game Reports for the Forest).

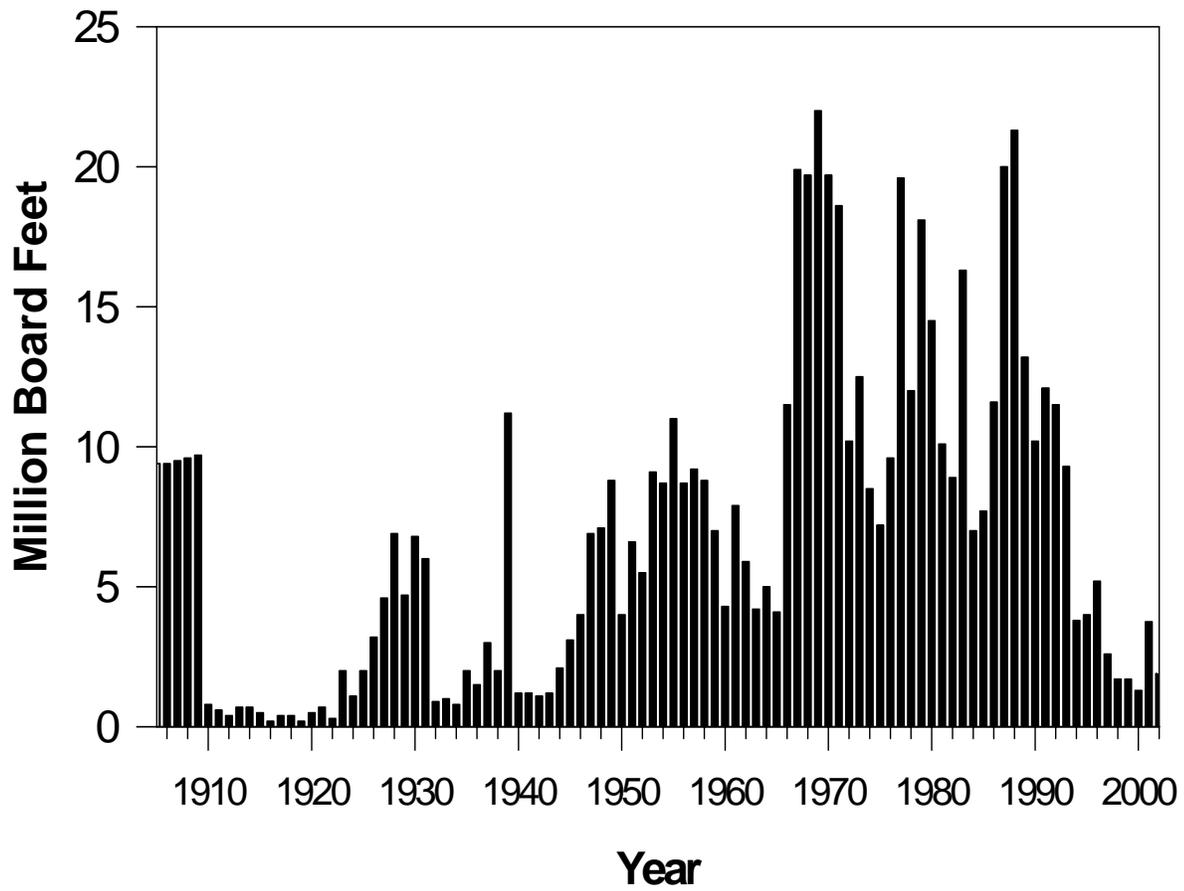


Figure 12. Million board feet harvested each year on the Bighorn National Forest from 1909 to 2002.

Bighorn National Forest

Clearcuts by Decade

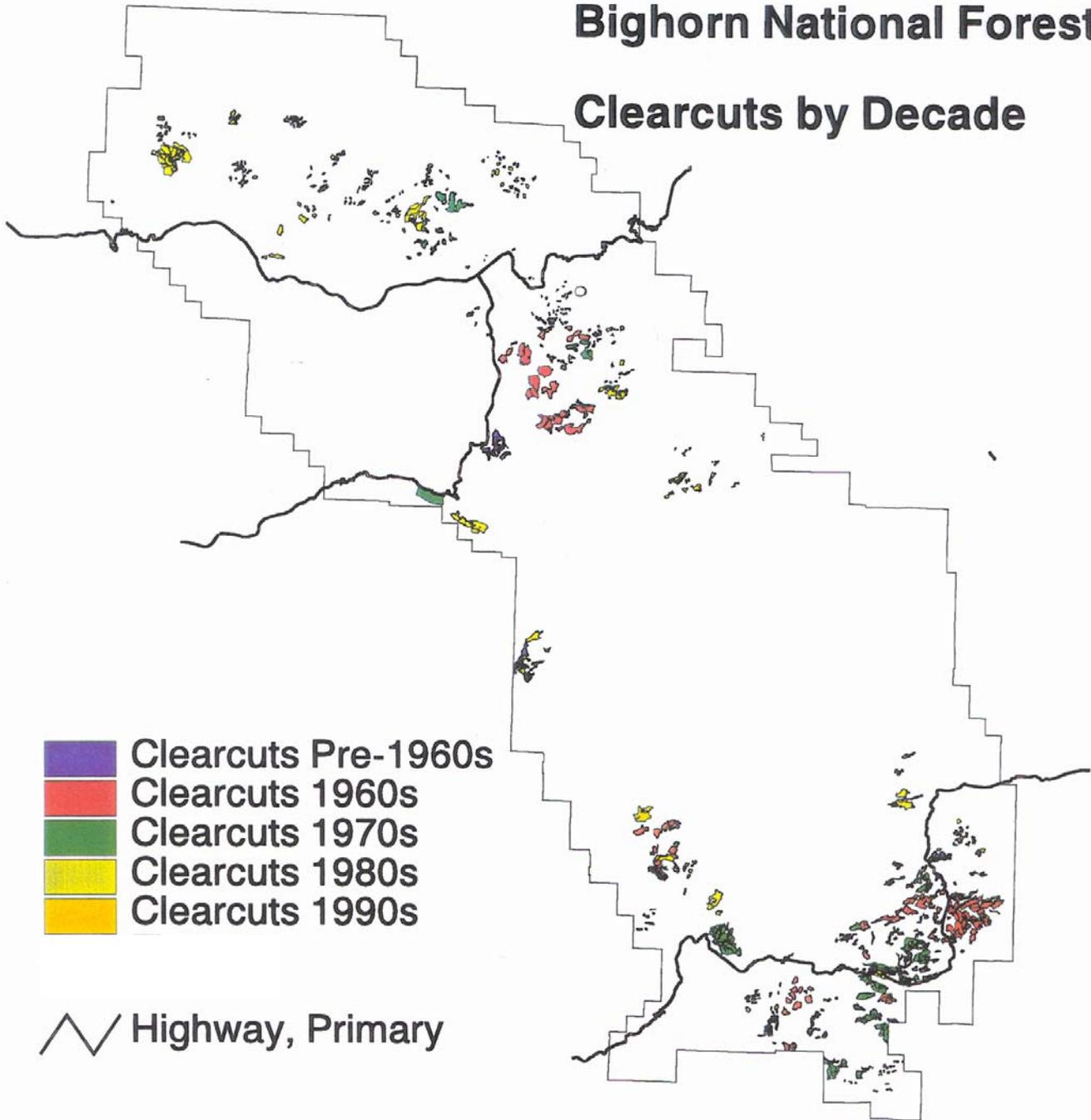


Figure 13. Location of clearcuts by decade on the Bighorn National Forest (from RIS database). Some areas had been affected by fire before they were cut. See Fig. 7 for watershed boundaries.

Bighorn National Forest

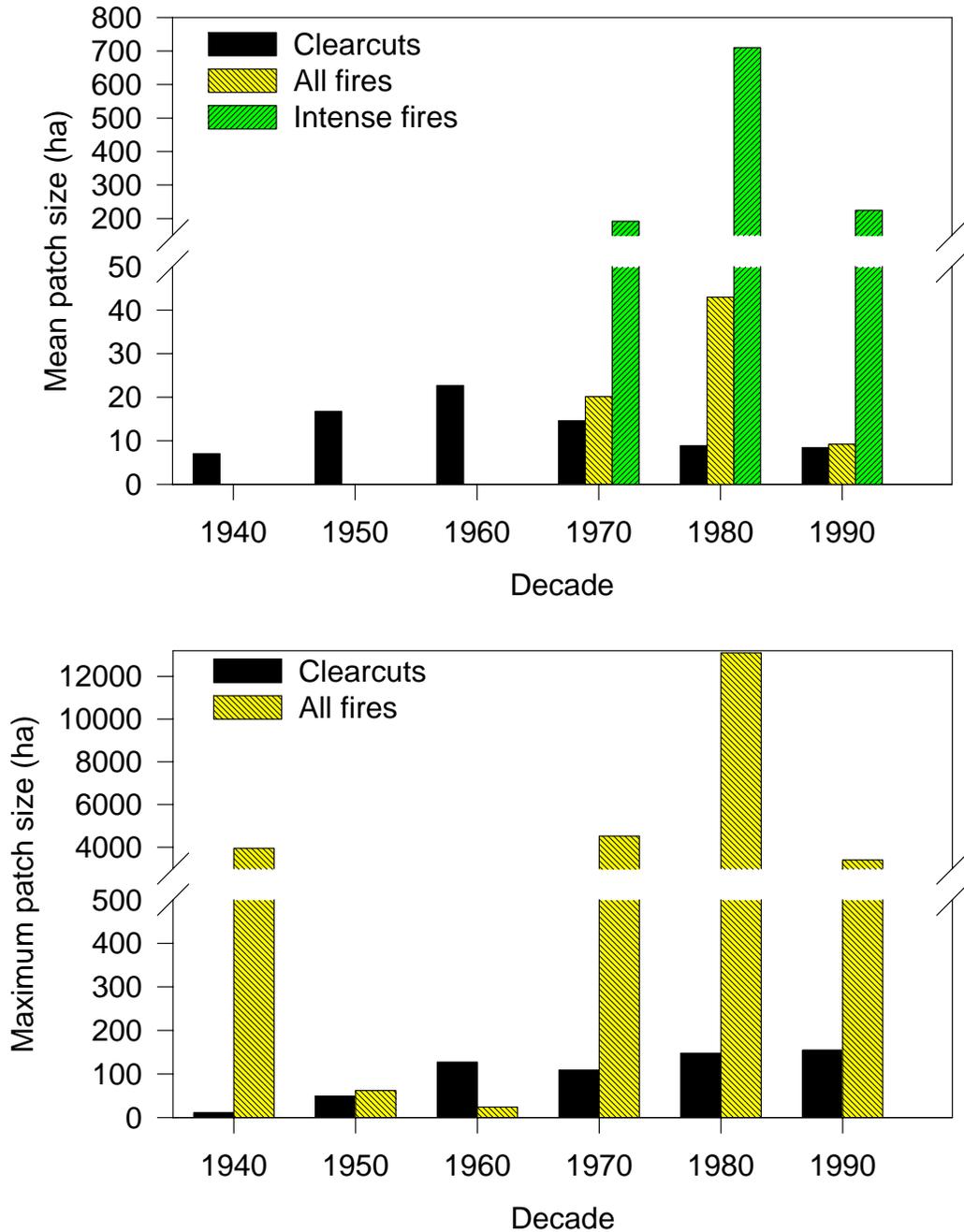
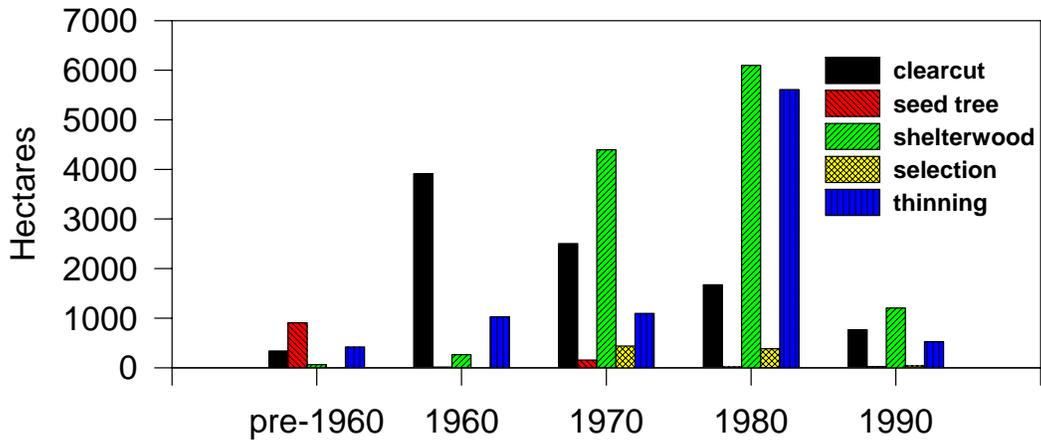


Figure 14. Mean and maximum patch size of clearcuts compared to fires on the Bighorn National Forest by decade (clearcuts from RIS database, fires from 1970-1996 fire database for top graph; data for bottom graph is from a map of fires for each decade). A clearcut patch was defined as an area clearcut over a month period that created one contiguous patch. An intense fire had flame lengths > 0.6 m.

Timber Harvest Activity on Bighorn NF



Site Activity on Bighorn NF

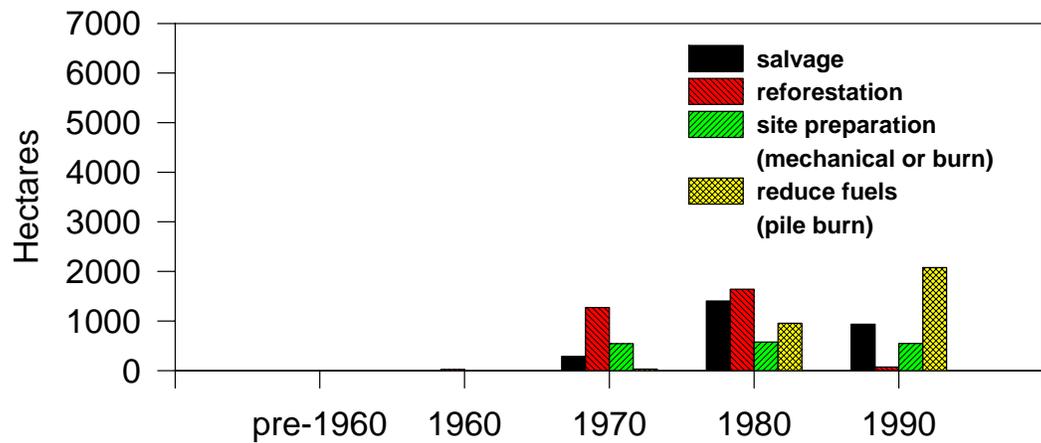


Figure 15. Hectares of the Bighorn National Forest affected by timber harvest and other related activities over decades (from RIS database). Data prior to 1960 are incomplete and do not include tie-hacking.

Bighorn National Forest Roads

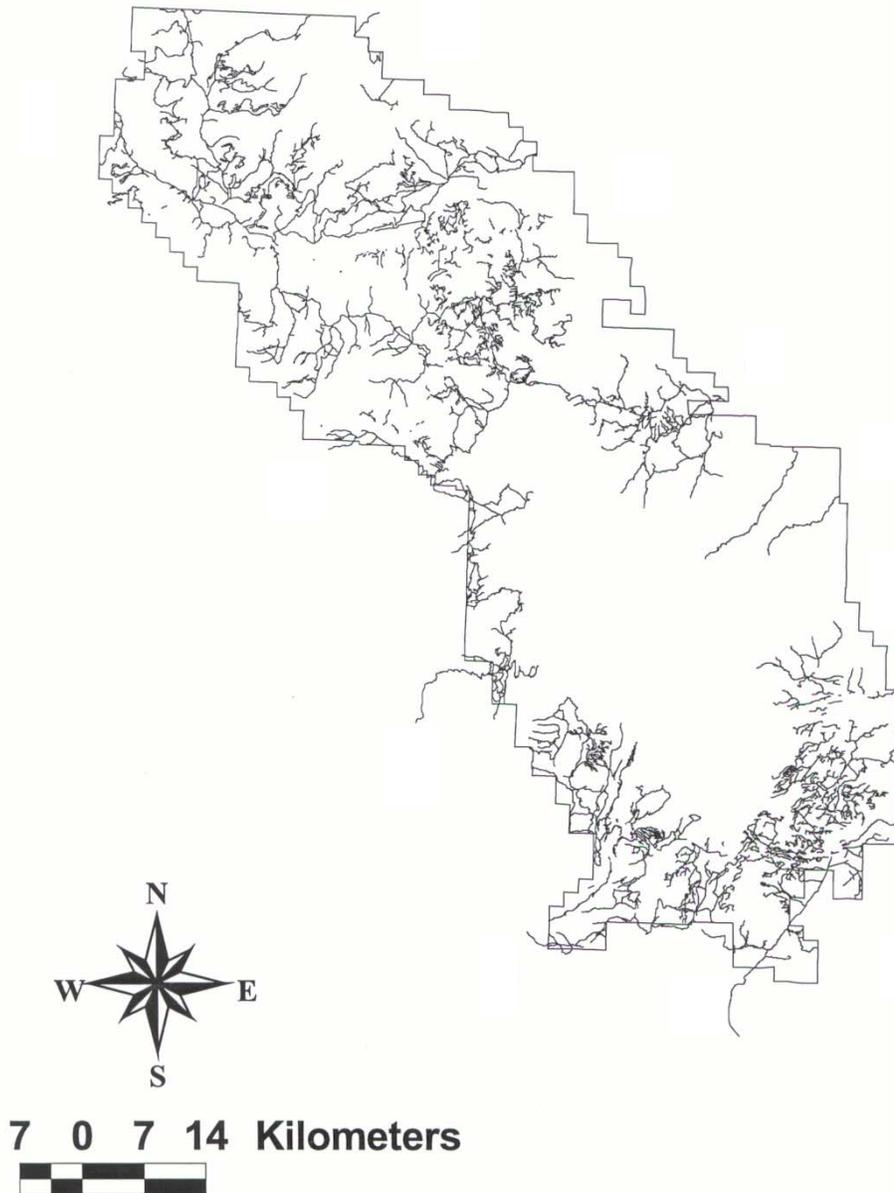


Figure 16. Location of roads on the Bighorn National Forest (from USFS road system inventory).

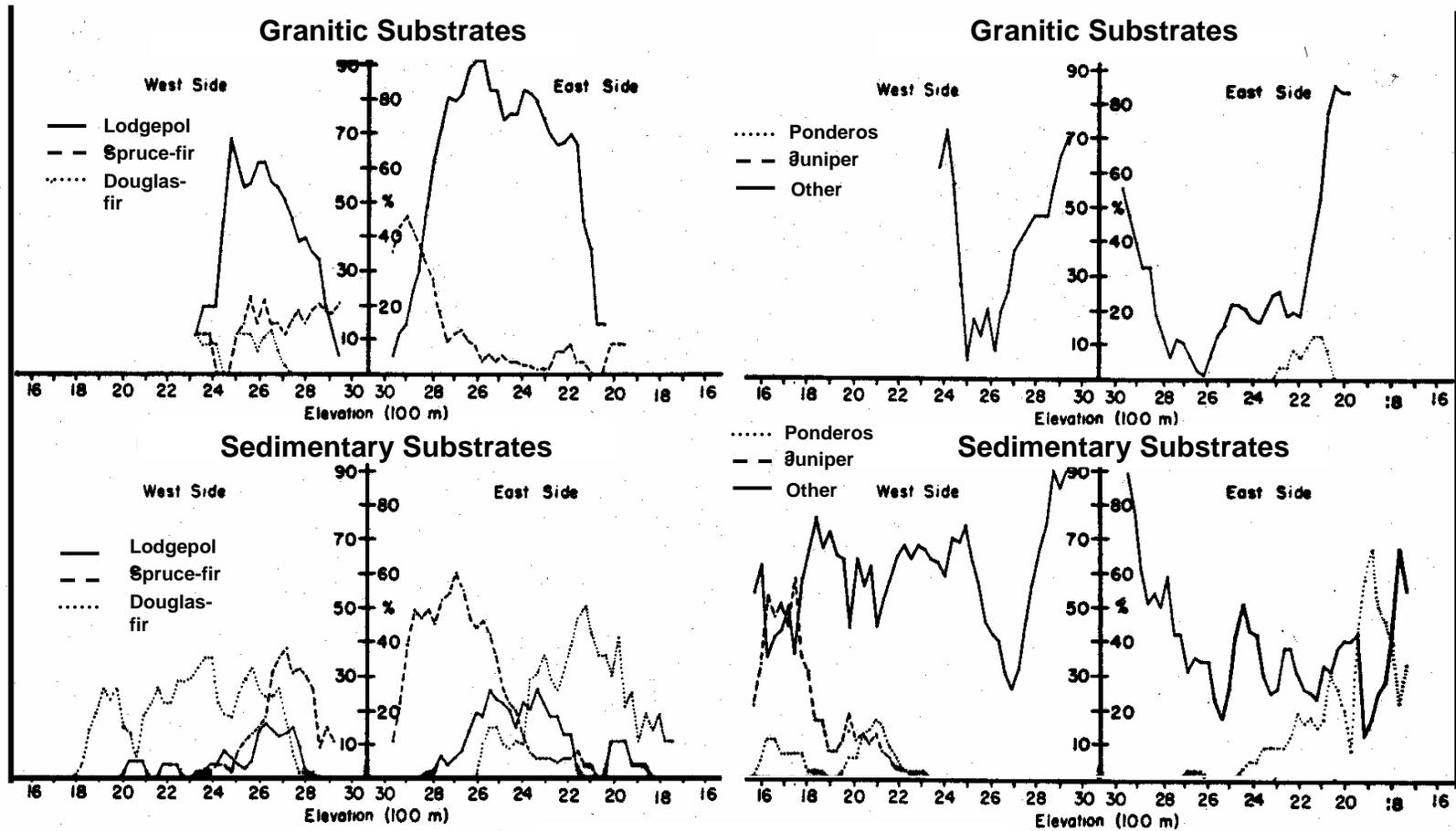


Figure 17. Relationship of percentage of each community type to elevation and substrate in the Bighorn Mountains, Wyoming. The curves were smoothed with a moving average ($n = 3$) (from Despain 1973).

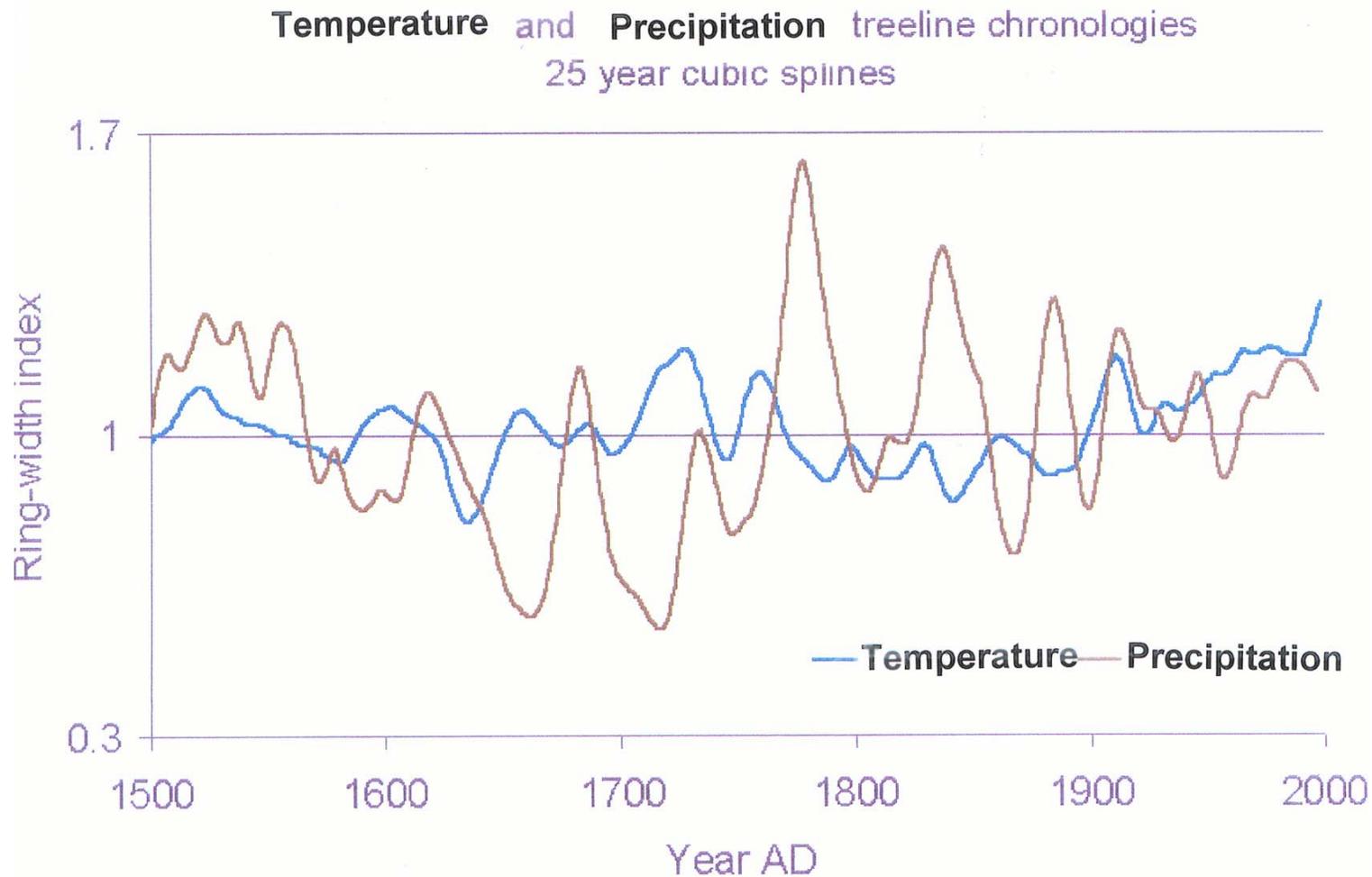


Figure 18. Tree-ring based reconstructions at upper treeline for temperature (Powder River Pass) and lower treeline for precipitation (west side) in the Bighorn Mountains. Values > 1 are wetter or warmer than average, values < 1 are drier or colder than average (unpublished data from C. Fastie, personal communication). Cubic spline is a mathematical equation that smooths the raw data in 25-year periods, which helps remove trends unrelated to climate.

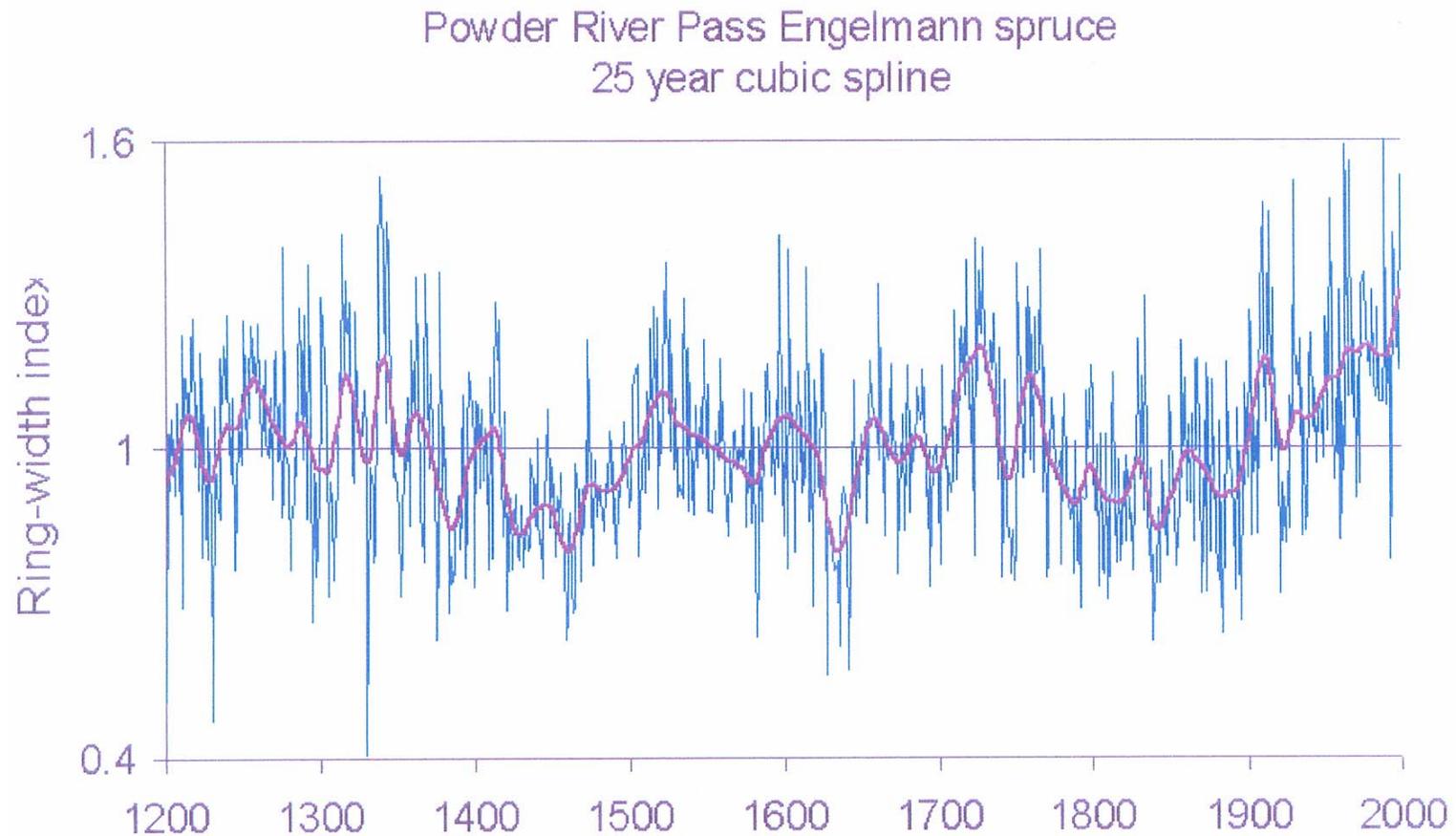


Figure 19. Tree-ring based reconstruction of temperature from 1200 AD to present before and after application of the 25-year cubic spline in an Engelmann spruce forest at Powder River Pass (treeline) in the Bighorn Mountains, Wyoming. Values > 1 are wetter or warmer than average, values < 1 are drier or colder than average (unpublished data from C.Fastie. personal communication).

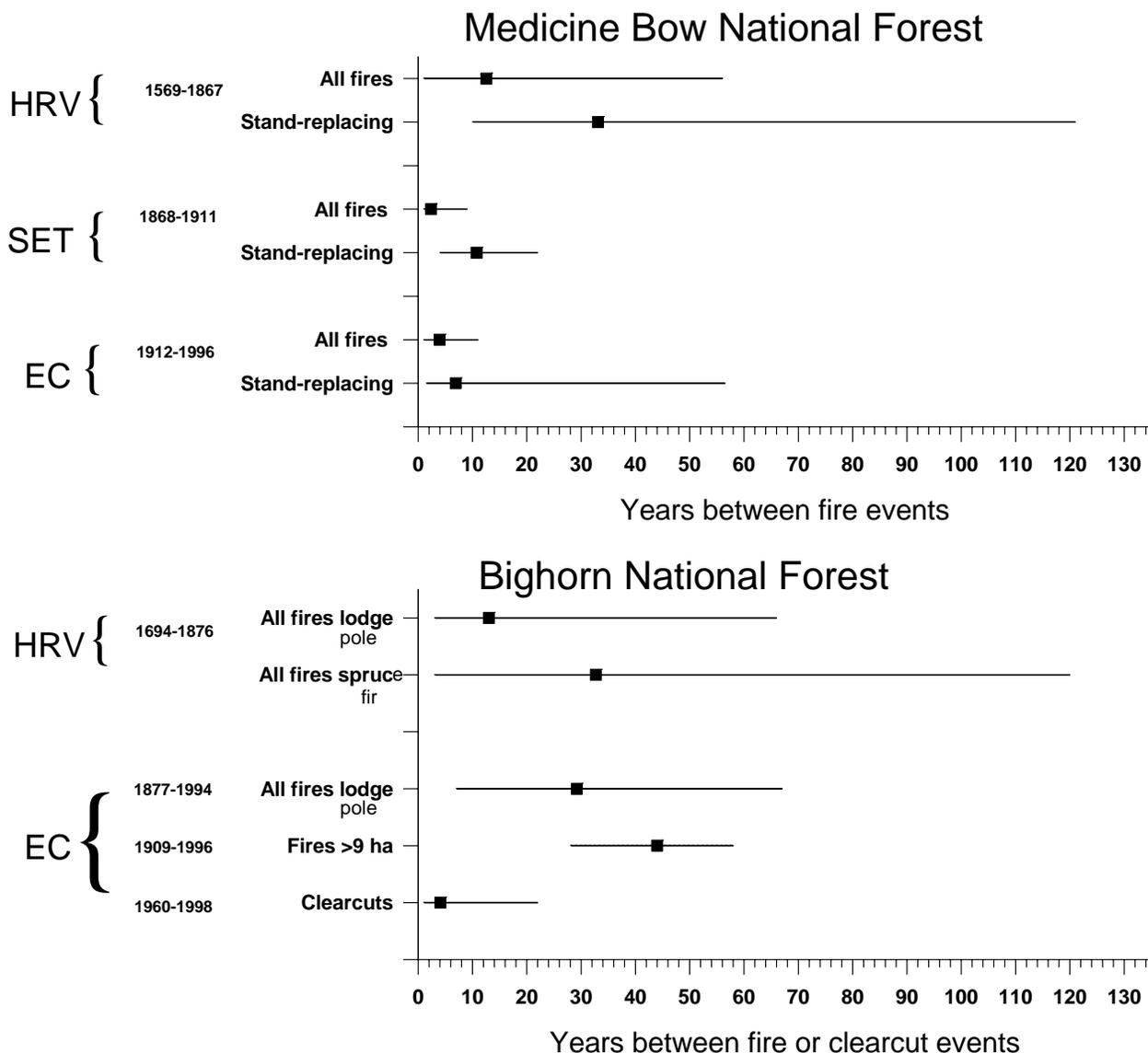


Figure 20. Mean (box) and range (line) of fire intervals across the landscape on a watershed on the Medicine Bow National Forest (Rock Creek, 3,241 ha) and on a watershed on the Bighorn National Forest (watershed 9 in Fig. 7, 4,162 ha) during the historic range of variability (HRV) reference period and during existing conditions (EC). The Medicine Bow National Forest (MBNF) also had a period after Europeans arrived but before fire suppression (SET), resulting in abundant fires during that period (short fire intervals) compared to the HRV period. Fire intervals lengthened again during post-settlement (EC) with fire suppression, but were still shorter than during the HRV period. In contrast, significant influences by European-Americans occurred later in the BNF, and large fires became less frequent in the lodgepole pine forests with fire suppression. Mean clearcut intervals on the BNF were shorter than fire intervals and thus are shown for comparison (data from Kipfmüller 1997 for the MBNF and Bornong 1996a for the BNF). Note: Fires > 9 ha include all cover types.

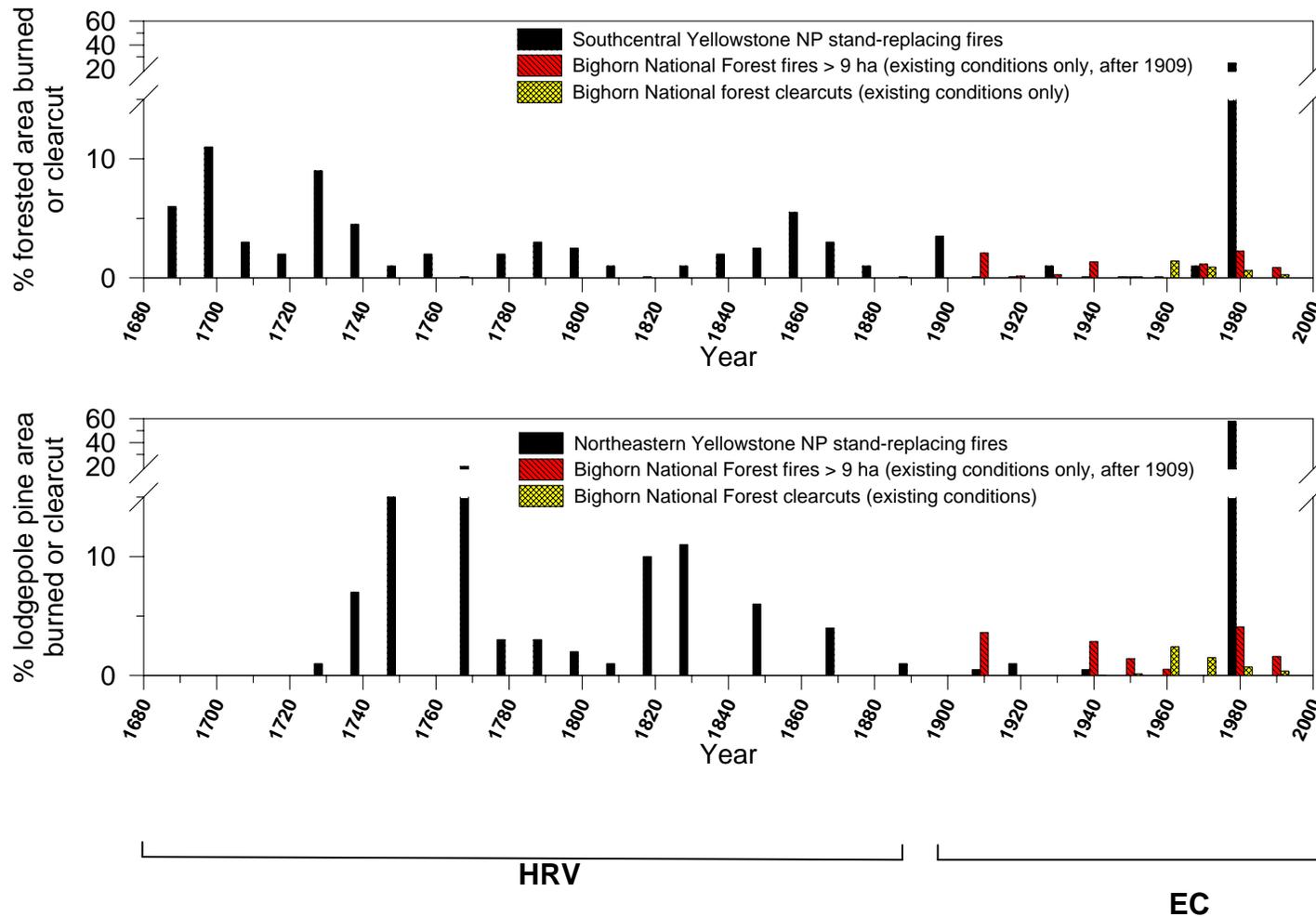


Figure 21. Comparison of the historic range of variability of area burned in Yellowstone National Park (proxy for BNF), to more recent conditions for area burned or clearcut in the Bighorn National Forest (from Romme and Despain 1989 for southcentral YNP, Barrett 1994 for northeastern YNP, and BNF fire and RIS databases). Top graph is for all forests; bottom graph is for lodgepole pine forests only. EC = existing conditions after European-Americans arrived. Data are summed by decade, not individual years. To illustrate, the 1988 fires in YNP are part of the 1980 dataset.

Fire starts per year

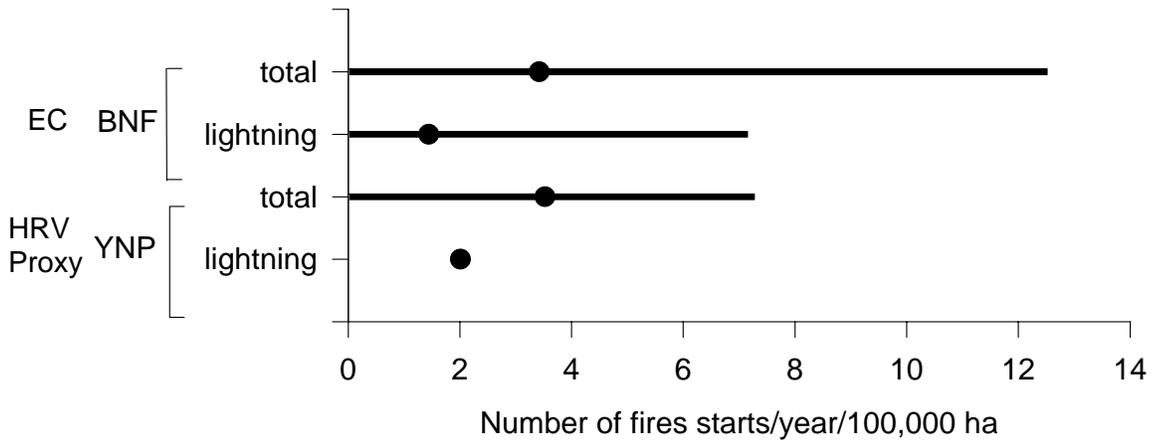


Figure 22. Mean (dots) and range (lines) of number of fire starts per year on the entire Bighorn National Forest compared to Yellowstone National Park, our HRV reference area (based on the BNF fire database for the periods, 1909-1936 and from 1970-96, and on YNP from 1972-1988, Despain and Romme 1991). From 1930 to 1970, 57% of the YNP fires were started from lightning (Taylor 1973 cited in Houston 1973), but the range of the number of lightning-caused fires is unknown. EC = existing conditions after European-Americans arrived.

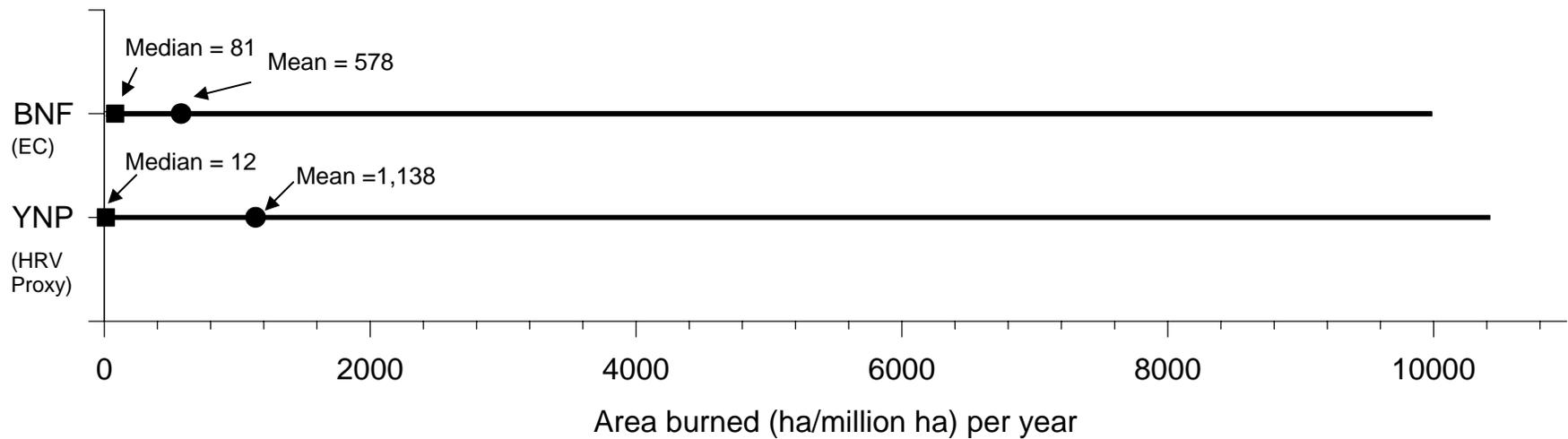


Figure 23. Area burned per year per million hectares within the entire Bighorn National Forest (BNF; EC = existing conditions, i.e., 1972-1987) and an area within Yellowstone National Park (HRV proxy) without fire suppression between 1972 and 1987 (YNP data from Despain and Romme 1991). Note: If 1988 fires in YNP had been included, the YNP range would extend to 441,890, and the mean (41,810) would shift to the right, further increasing the difference between the BNF and YNP means.

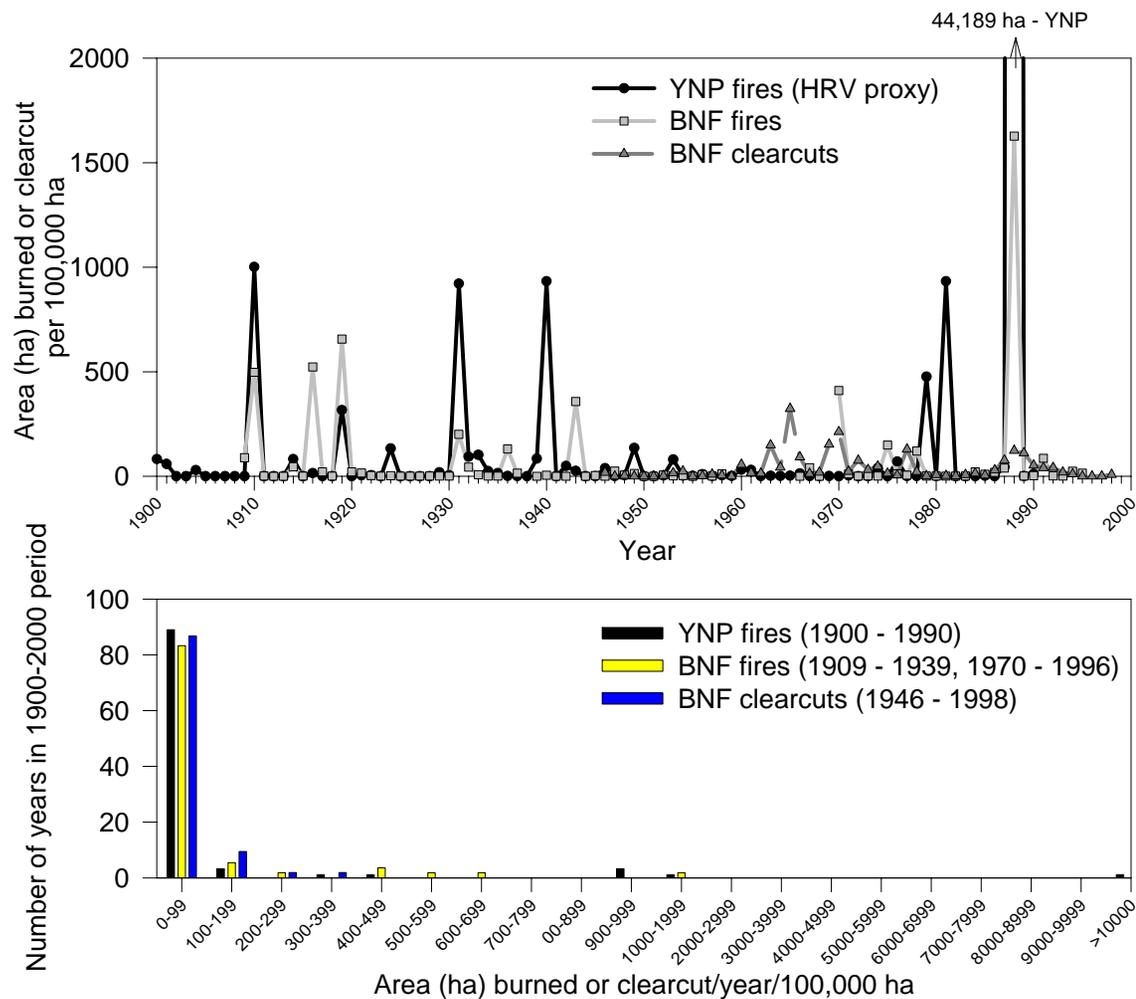


Figure 24. Area burned or clearcut each year during the last century (1900-1996) on the entire Bighorn National Forest (BNF), compared to area burned in all of Yellowstone National Park (YNP) during a similar period, which may serve as an HRV proxy. However, fire suppression was practiced in YNP for all but the period from 1972 to 1987. Also, the BNF has less forested area to burn (62%) than YNP (80%) and less flammable forest (about 41% on BNF in flammable LP2, LP3, and LP of Despain (1990) classifications, compared to 63% on YNP), but a slightly drier climate. Data from small fires (< 9 ha) on the BNF from 1940-1969 are missing. Those years are not used in the lower graph, but results for the remaining years were standardized to 100 years (as were clearcuts). The area burned includes all fires, not just stand-replacing fires (sources: Balling et al. 1992 for YNP; BNF fire and RIS databases).

Large Fires by Decade (> 9 ha)

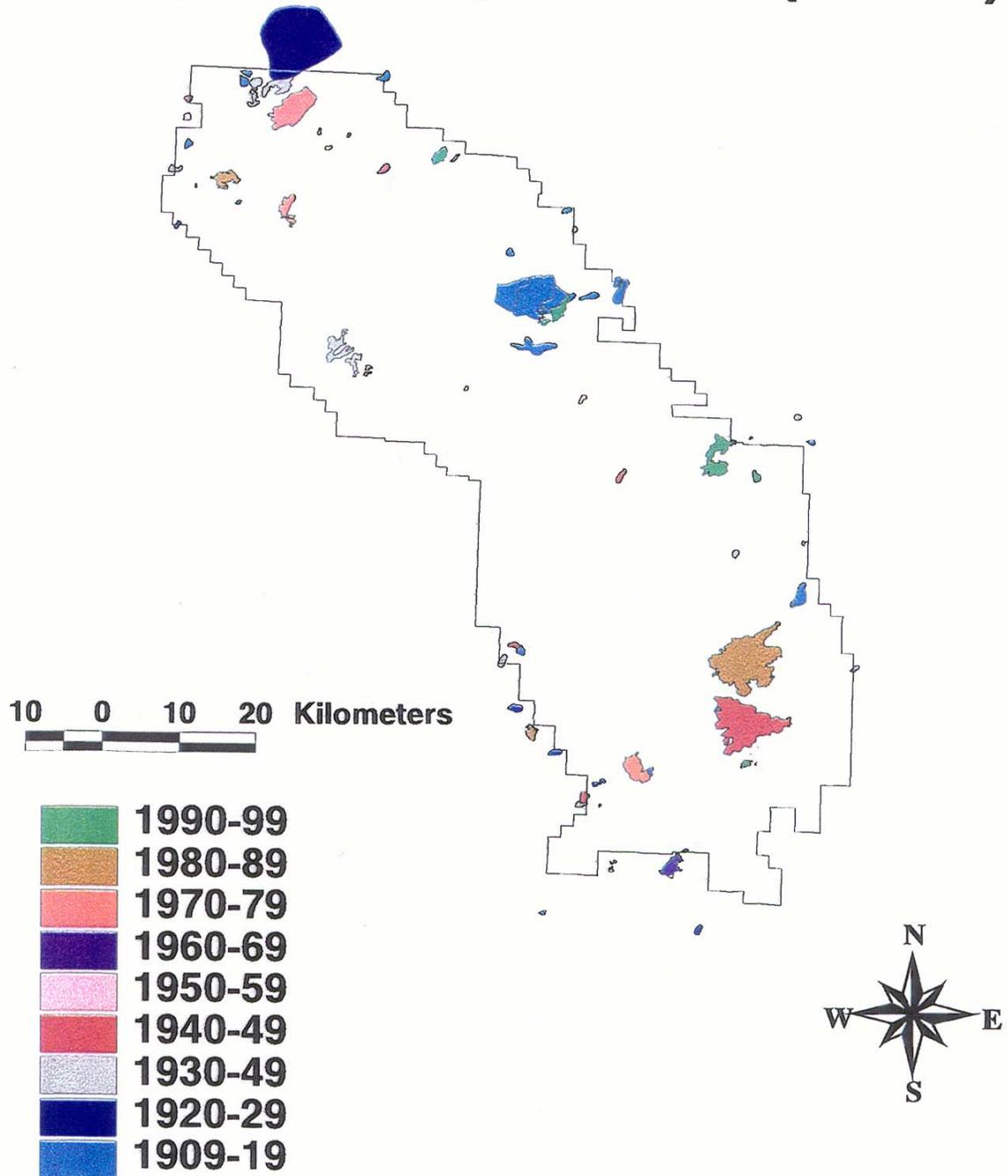
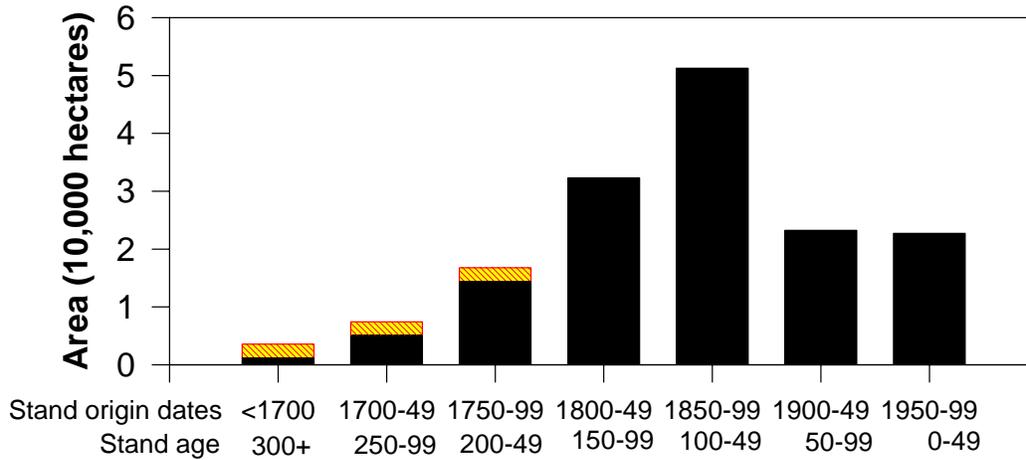


Figure 25. Location of large fires (> 9 ha) on the Bighorn National Forest for each decade from 1909 to 1996. The largest fires shown on this map burned about 5,000 ha.

Forest Ages in Bighorn NF (EC)



Forest Ages in Little Firehole Watershed, YNP (HRV proxy)

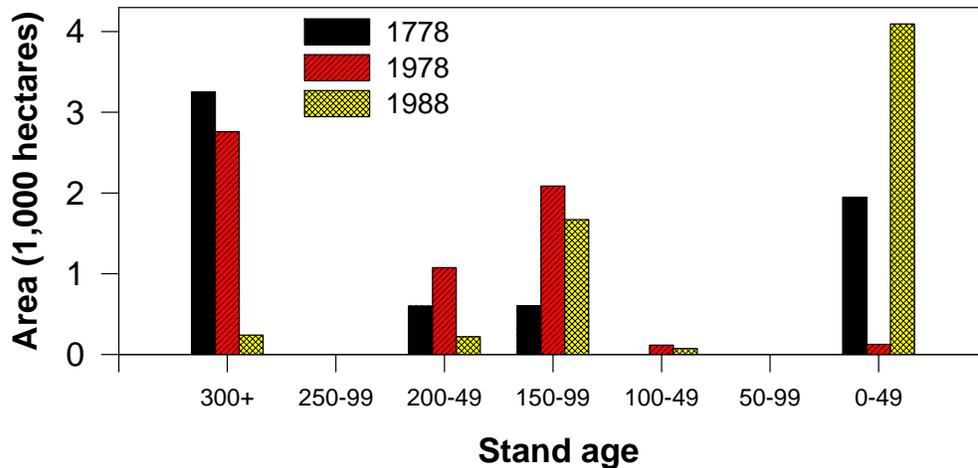
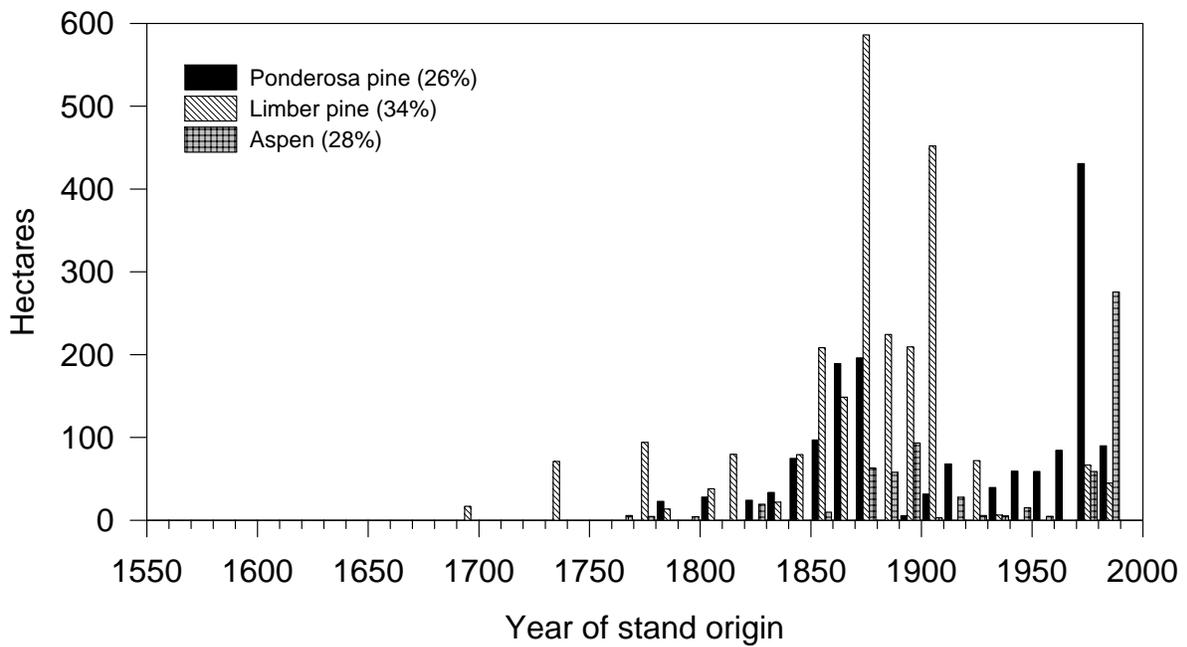
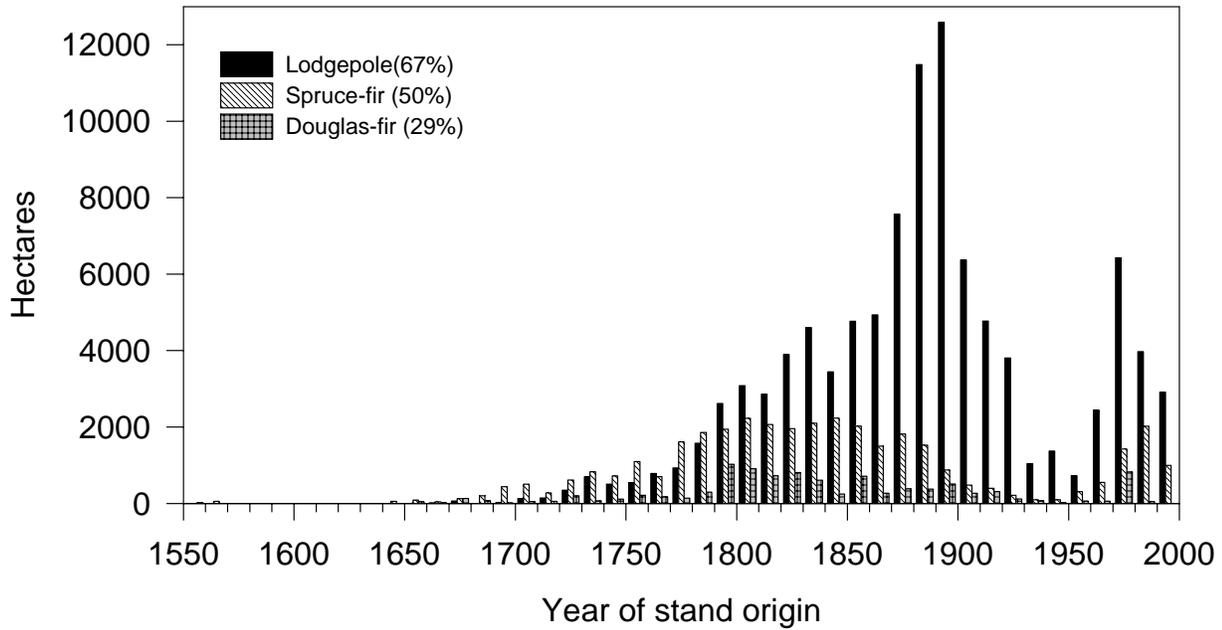


Figure 26. Stand age and origin dates for trees presently on the Bighorn National Forest (represents 60% of forests only, RIS database) compared to stand age for forests in the Little Firehole Watershed in Yellowstone National Park. In Yellowstone (HRV proxy), stand ages are shown during three time periods (1778, 1978, and 1988), which came from maps of the reconstructed forest in Romme (1979) and from our estimates using 1988 satellite imagery of the park. In the upper graph, the lighter part of the stacked bar is the area clearcut that would have been in older age classes without clearcutting (we divided the total clearcut area into thirds and placed them in the three oldest age classes). Although not shown, the clearcut area would also have to be subtracted from the youngest age class to represent conditions without clearcutting.



HRV
EC

Figure 27. Stand origin dates for forests on the Bighorn National Forest (from RIS database). Percentages represent relative area of each forest type surveyed for age. Note that for non-seral forests, origin dates do not approximate time since last stand-replacing disturbance, just the date the majority of the present trees became established. Note also that vertical axes differ between the two graphs.

Stand Age

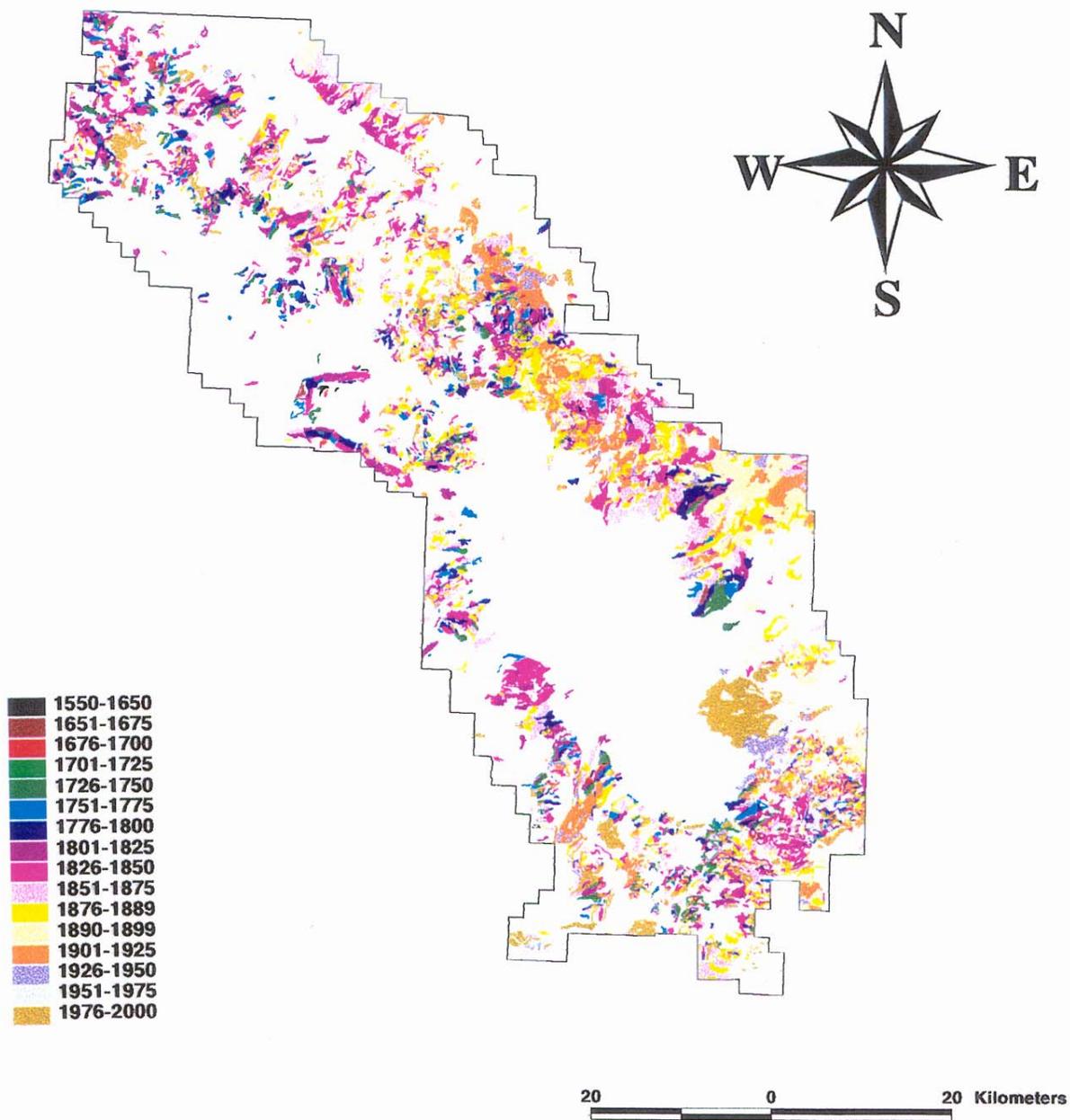


Figure 28. Origin dates for stands that have been aged on the Bighorn National Forest (from RIS database).

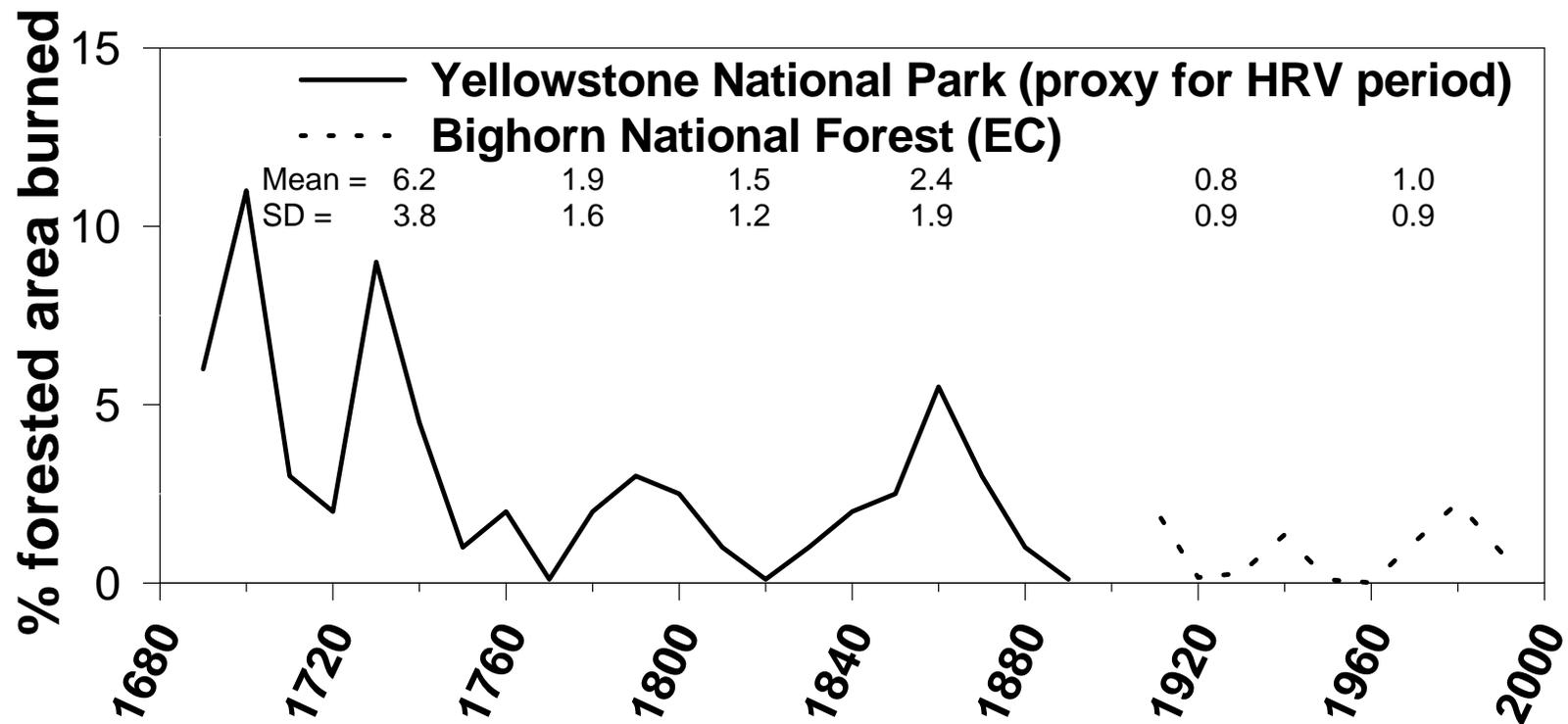


Fig. 29. The historic range of means for percent of forested area burned each decade (means shown for each 50-year period, starting in 1690) is 1.5 to 6.2 in Yellowstone National Park (used as HRV proxy for BNF). The range of means for the Existing Conditions (EC) in the Bighorn National Forest is 0.8 to 1.0 for the 50-year periods, which suggests that the forested area burned on the BNF is outside the HRV, probably due to fire suppression. The range of standard deviations for existing conditions (0.9-0.9) in the BNF also falls outside the historic range of standard deviations for this variable (1.2-3.8).

Lodgepole and Spruce-fir Forest Fires

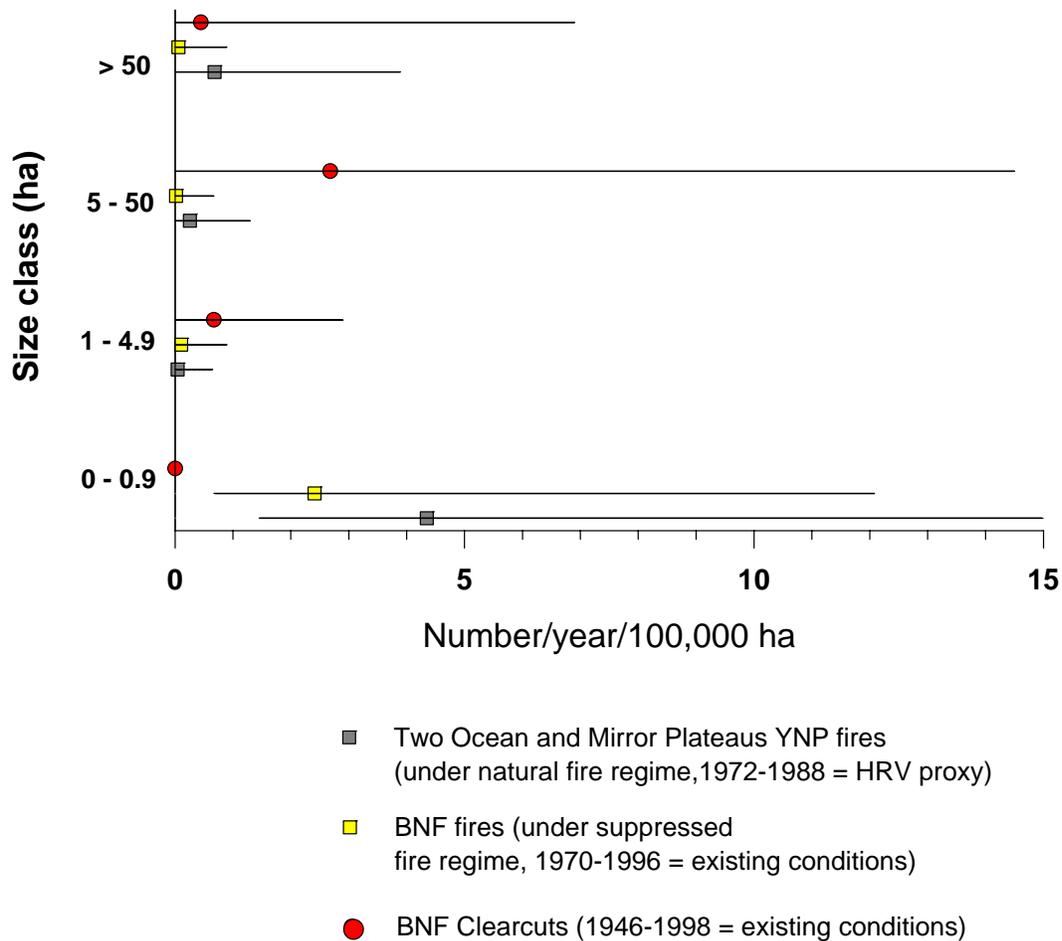


Figure 30. Mean (box or circle) and range (line) of number of fires and clearcuts of various size classes that occur per year across the landscape in spruce-fir and lodgepole pine forests for the entire Bighorn National Forest (BNF) compared to fires that occur on the Two Ocean and Mirror Plateaus in Yellowstone National Park (YNP) under a nearly natural fire regime. Percent of spruce-fir and lodgepole pine forest is similar between the two areas, although YNP has additional whitebark pine forest (from Despain 1990 for YNP and BNF Fire and RIS databases). Absolute ranges are shown, but based on the means, more large fires (5-50 and > 50 ha) tend to occur in YNP, which serves as an HRV proxy for the BNF, than for existing conditions with fire suppression on the BNF. However, the number of clearcuts of size 1-4.9 and 5-50 ha is much higher than the number of fires of the same size in either YNP or the BNF.

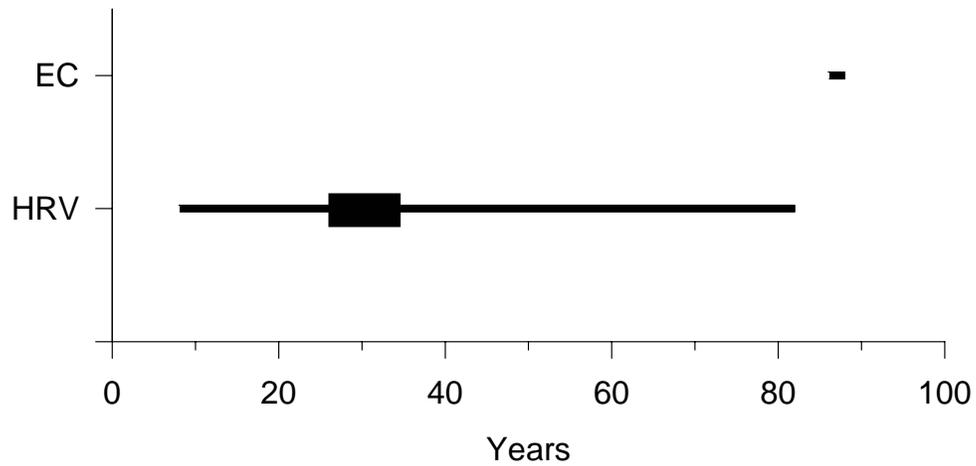


Figure 31. Range of fire return intervals for surface fires in stands of *Pinus ponderosa* in the Ashenfelder Basin, Medicine Bow National Forest, southeastern Wyoming. The HRV period is prior to 1900 and the block represents the range of median intervals during that period (from Brown et al. (2000)). The value during the EC is essentially time since last fire because no fires have occurred since the last fire in the early 1900s.

Bighorn National Forest Fires from 1970-1996 and Clearcuts from 1946-1998

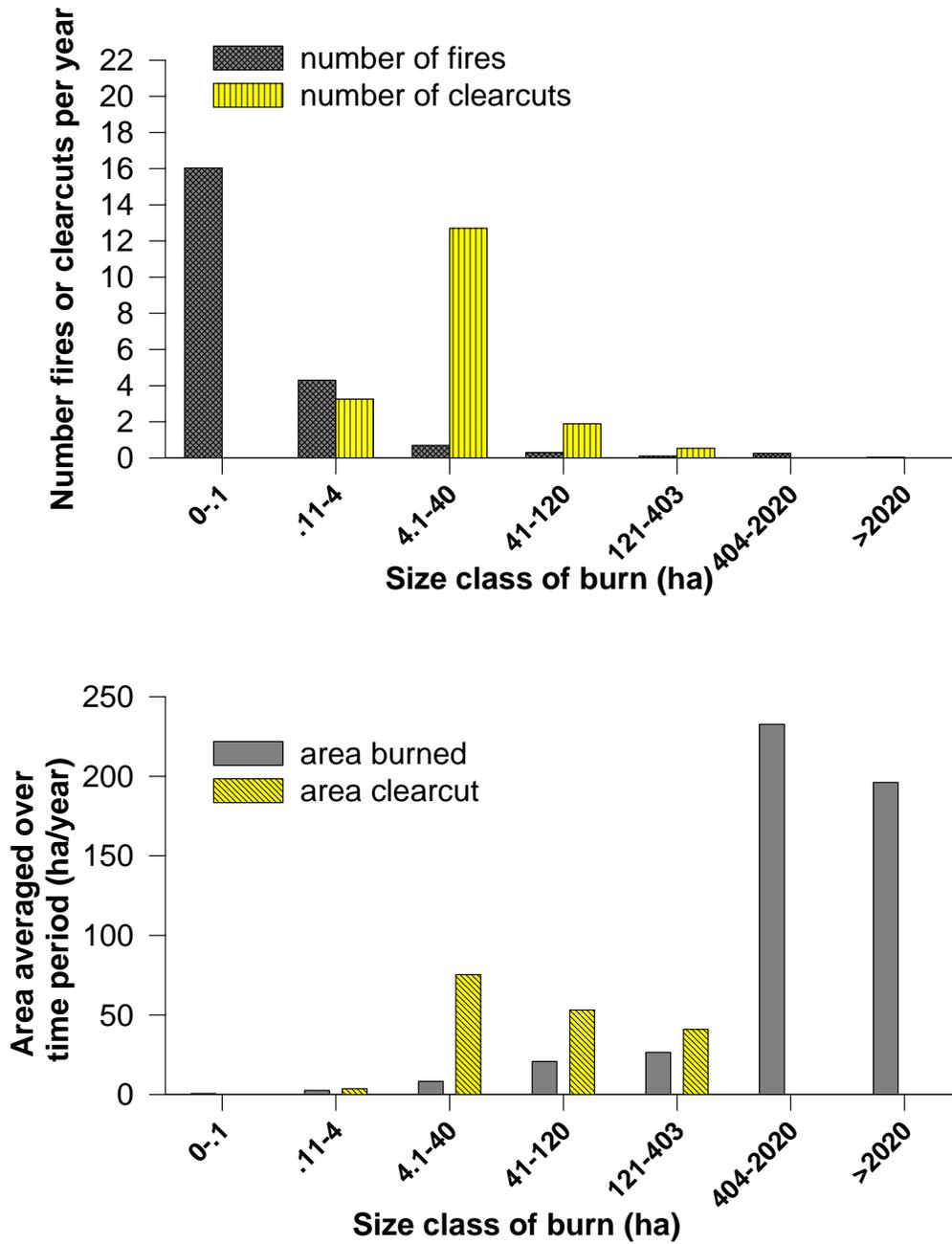


Figure 32. Number (top) and area (bottom) of individual fires and clearcuts by size class on the Bighorn National Forest (from BNF RIS and fire databases). Note: data are not standardized per ha.

Fires by Vegetation Type on the Bighorn National Forest from 1970-1996

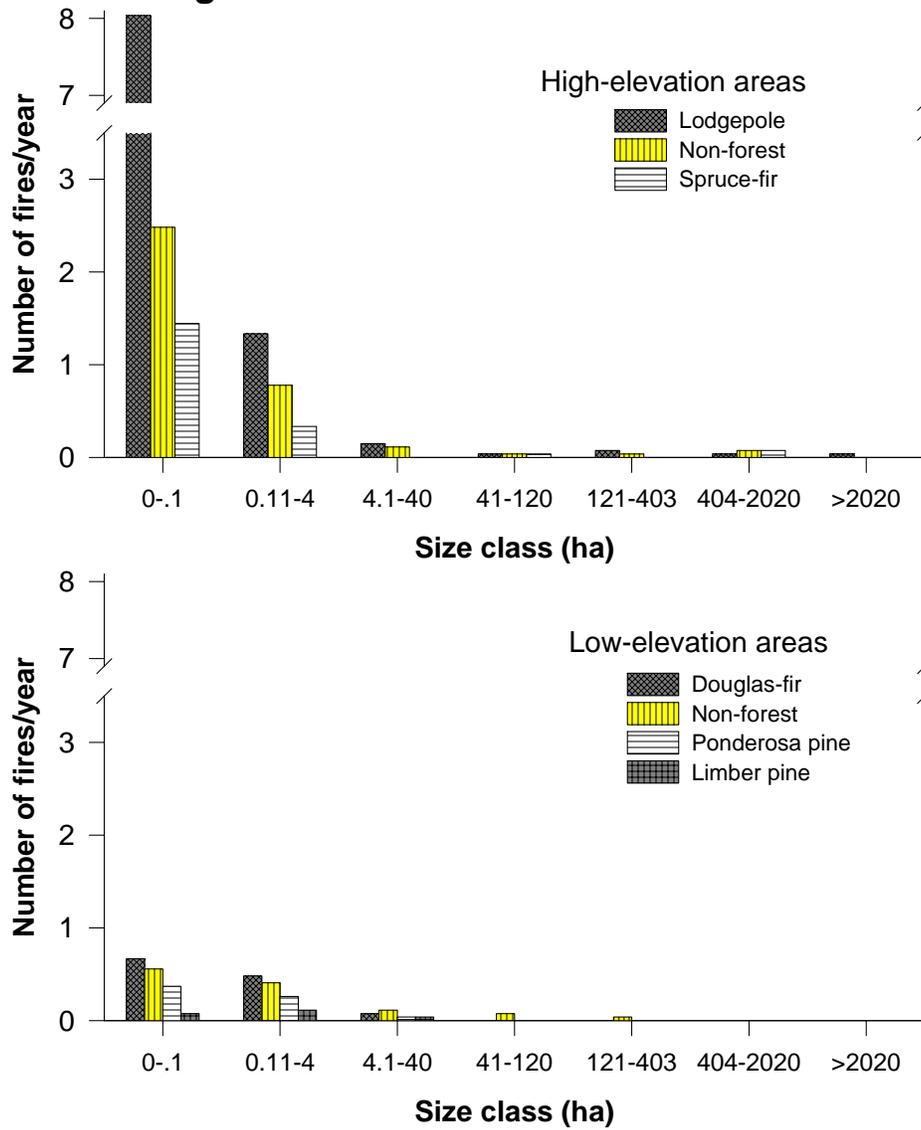


Figure 33. Number of individual fires by size class and vegetation type on the Bighorn National Forest (from BNF RIS and fire databases). Note: the number of fires per year is not standardized per hectare.

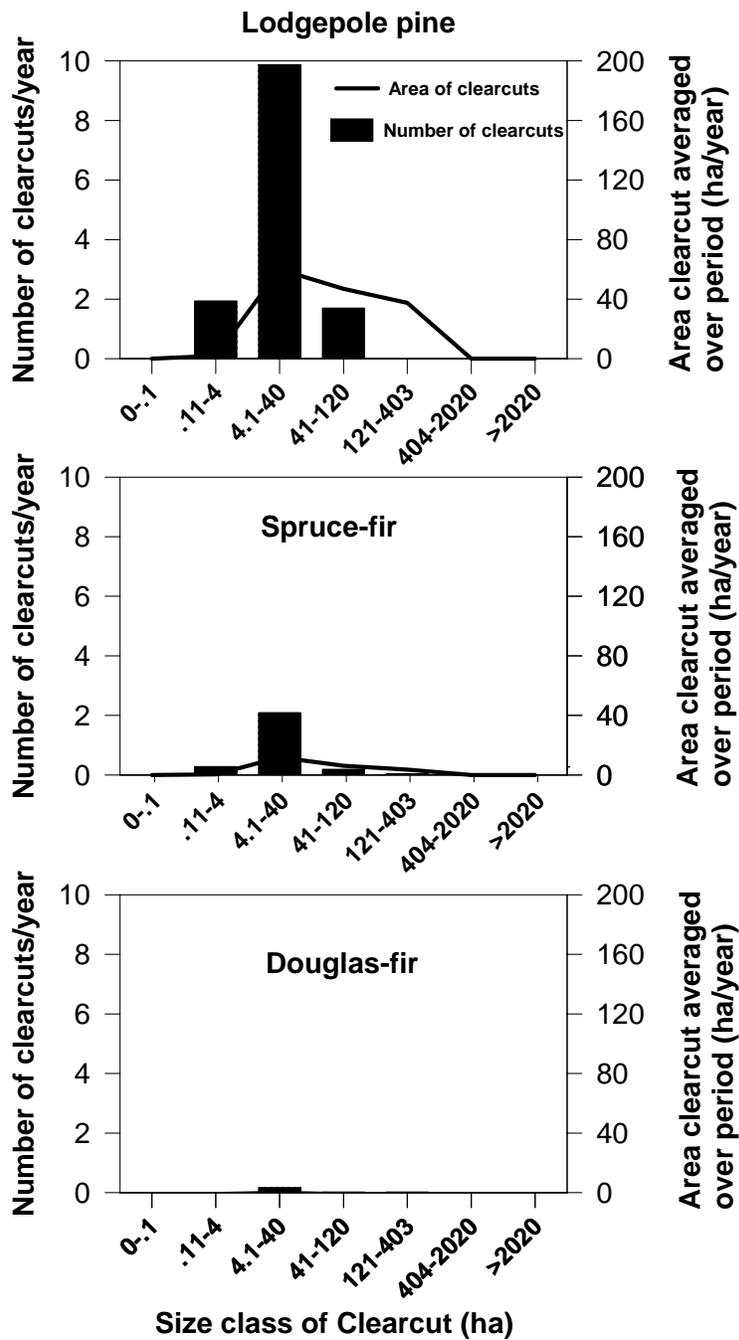
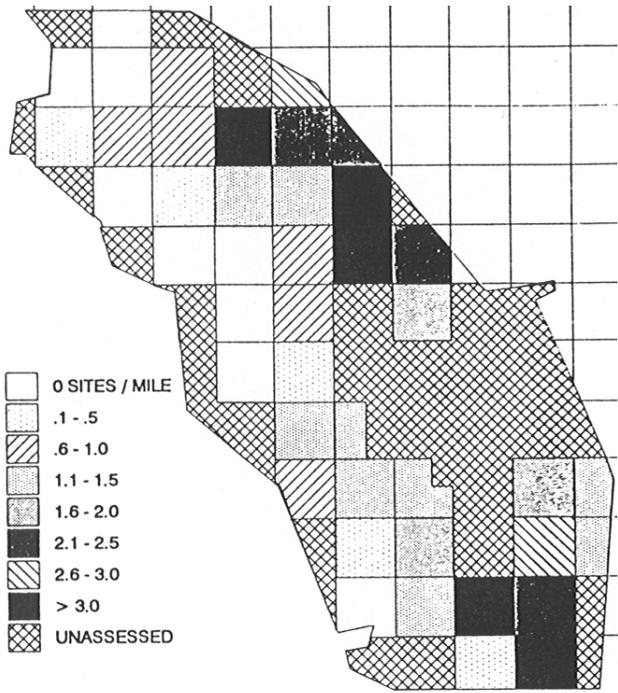
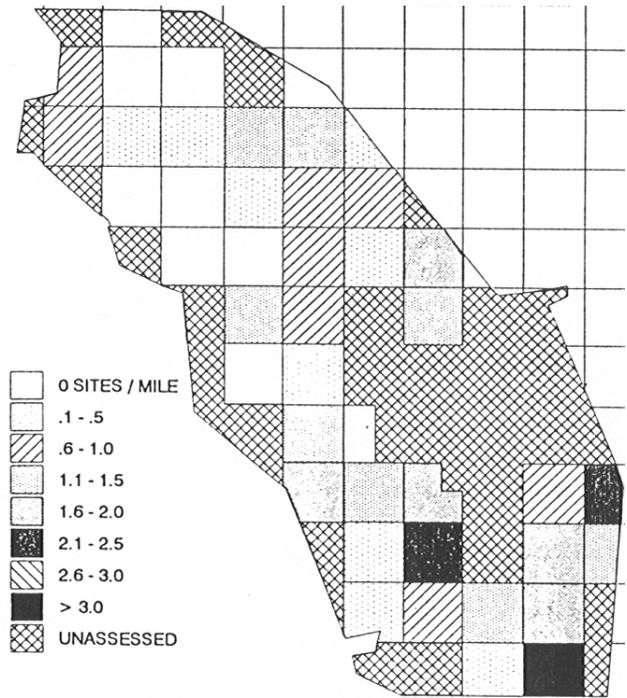


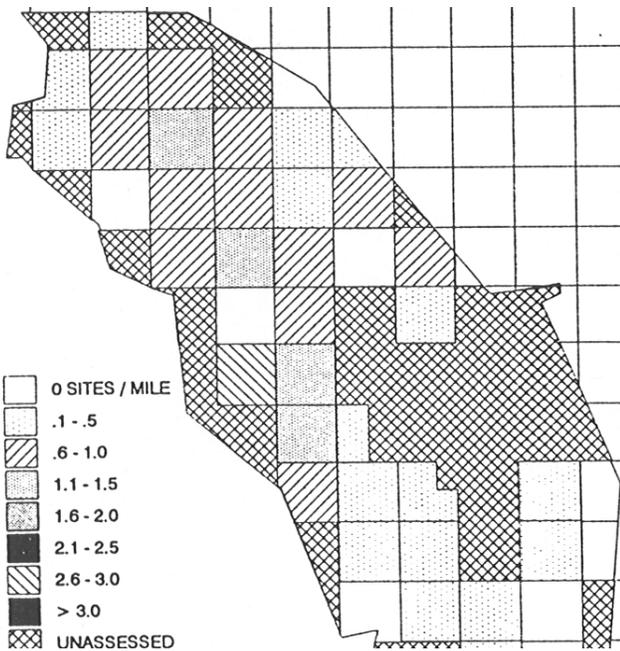
Figure 34. Number (bars) and area (lines) of individual clearcuts by size class and vegetation type on the Bighorn National Forest (from BNF RIS database, 1946 - 1998). A clearcut patch was defined as an area clearcut over a month period that created one contiguous patch. Note that the data are not standardized per hectare.



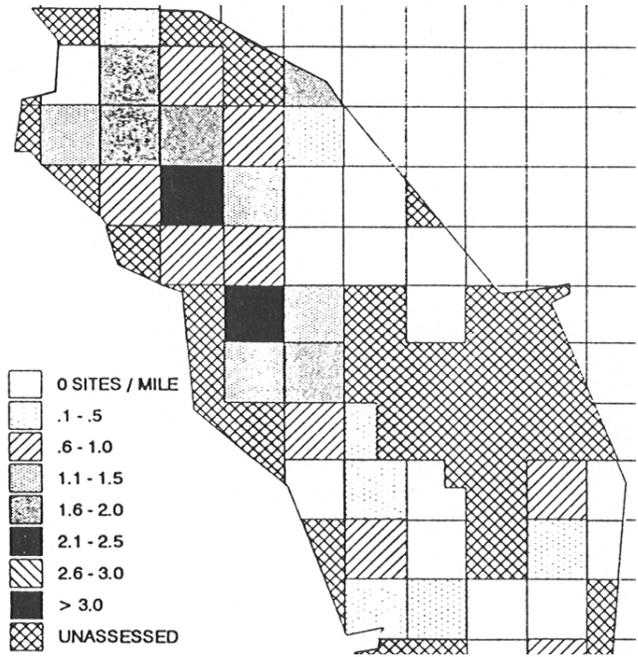
Commandra Blister Rust of Lodgepole Pine



Dwarf Mistletoe of Lodgepole Pine



Broom Rust of Engelmann spruce and Subalpine fir



Patch Mortality (fir decline) of Engelmann spruce and Subalpine fir

Figure 35. Distribution by township of various diseases surveyed on the Bighorn National Forest in 1991 (Lundquist 1993).

Landscape Disturbances (1955-1998)

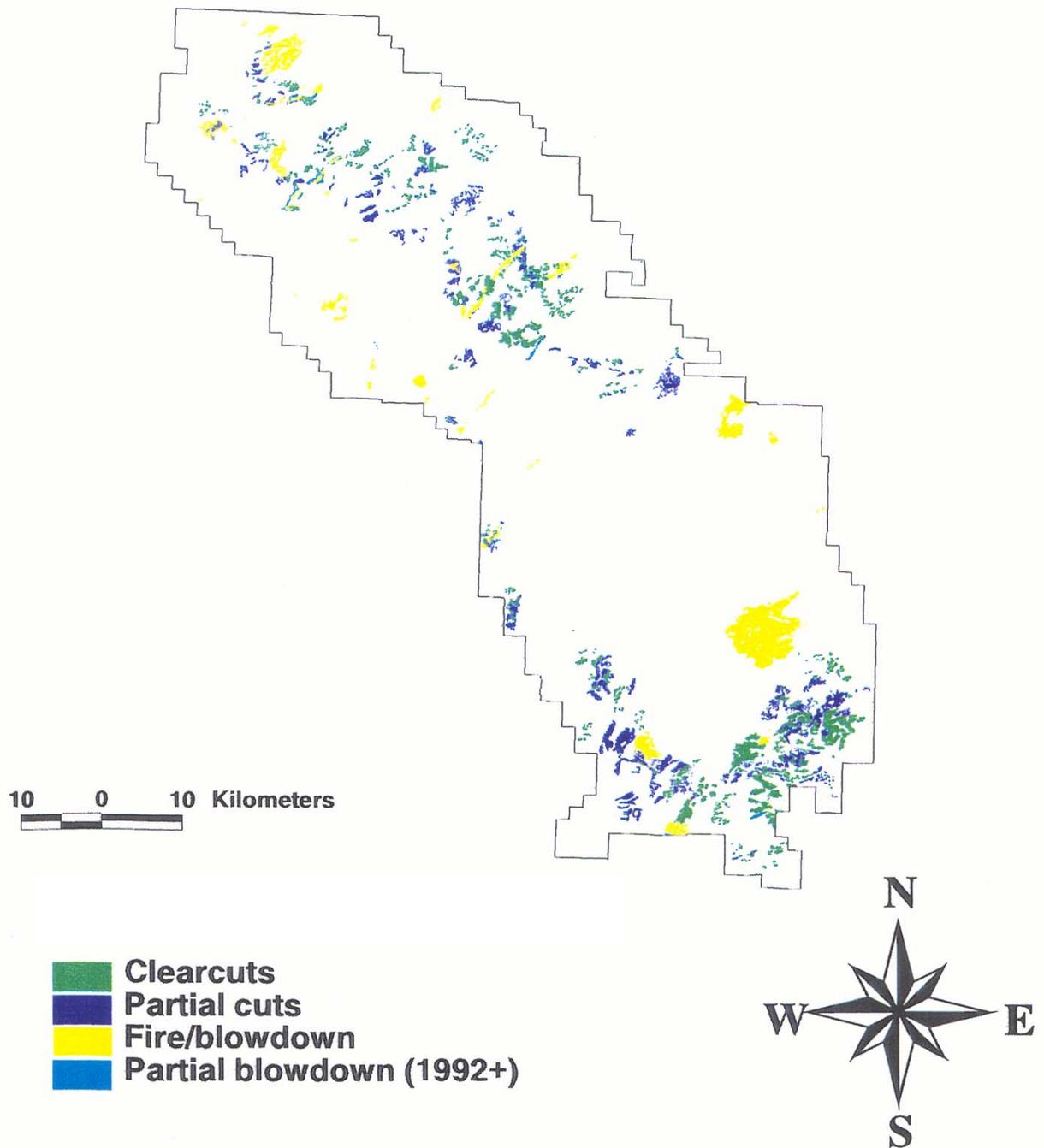


Figure 36. Location of landscape disturbances on the Bighorn National Forest from 1955 to 1998 (from Wyoming Game and Fish Department, LANDSAT-derived database). The linear patches in yellow are blowdowns.

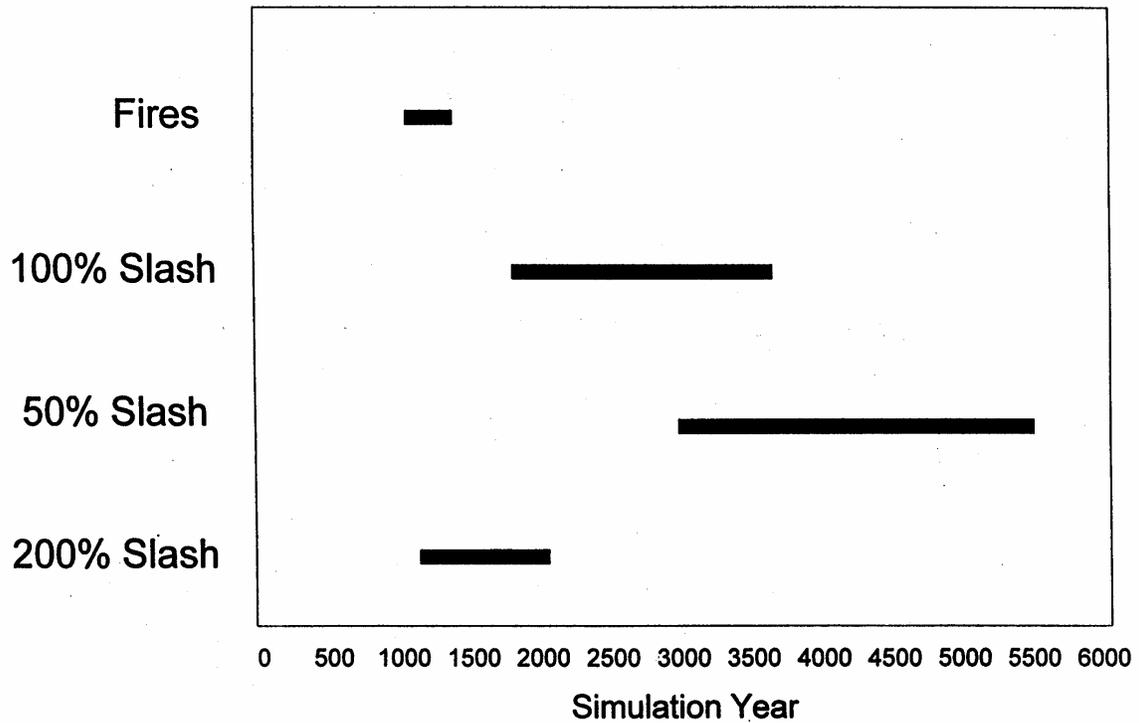


Figure 37. Time required for 100% of the forest floor to have been affected by coarse woody debris following fires and clearcuts, with different amounts of slash left behind after harvesting. Horizontal bars indicate the range of years required for coarse woody debris (CWD) to have affected (covered) 100% of the forest floor. A range of random simulations are shown for fires (HRV conditions) and for clearcuts (existing conditions), both with a 100-year mean return interval. Slash cover from clearcuts was 100% (average), half (50%) and double (200%) average amounts of slash usually left on site in recent clearcuts on the Medicine Bow National Forest. The time required for the forest floor to be completely covered under the natural fire regimes ranged from 1125-1350 years. In contrast, 100% clearcut slash simulations required 1800-3600 years to completely cover the forest floor, and 50% clearcut slash simulations required 3000-5600 years. Leaving double the amount of CWD in slash (200%), simulations required 1100-2150 years for 100% of the forest floor to have had the influence of CWD. This computer simulation study suggests that leaving double the amount of CWD in slash could possibly bring CWD to within the HRV. Source: Tinker and Knight (2001).

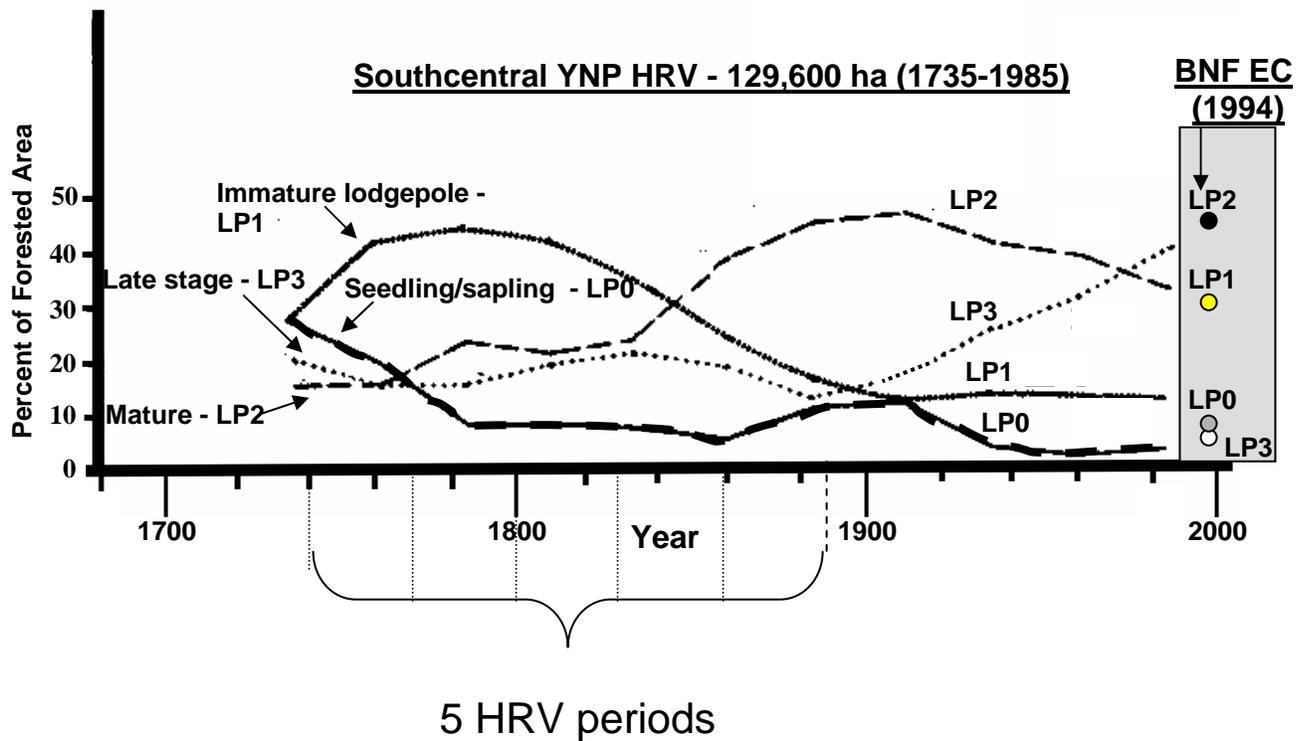


Figure 38. Historic range of variability of percentages in different successional stages of lodgepole pine/spruce-fir forests within several watersheds of Yellowstone National Park (HRV proxy, Romme and Despain 1989) compared to existing conditions on the Bighorn National Forest (BNF, far right). See Table 5 for definitions of successional stages. Existing conditions on the BNF fall within the range of means within five 30-year periods (from 1740 to 1890), except for the late successional stage (LP3, HRV = 18-22%, EC = 5%) and mature stage (LP2, HRV = 17-40%, EC = 49%). Note: The effect of the 1988 fires is not shown, which probably would have extended the upper limit of the HRV of LP0 and the lower limit of LP3 and LP2.

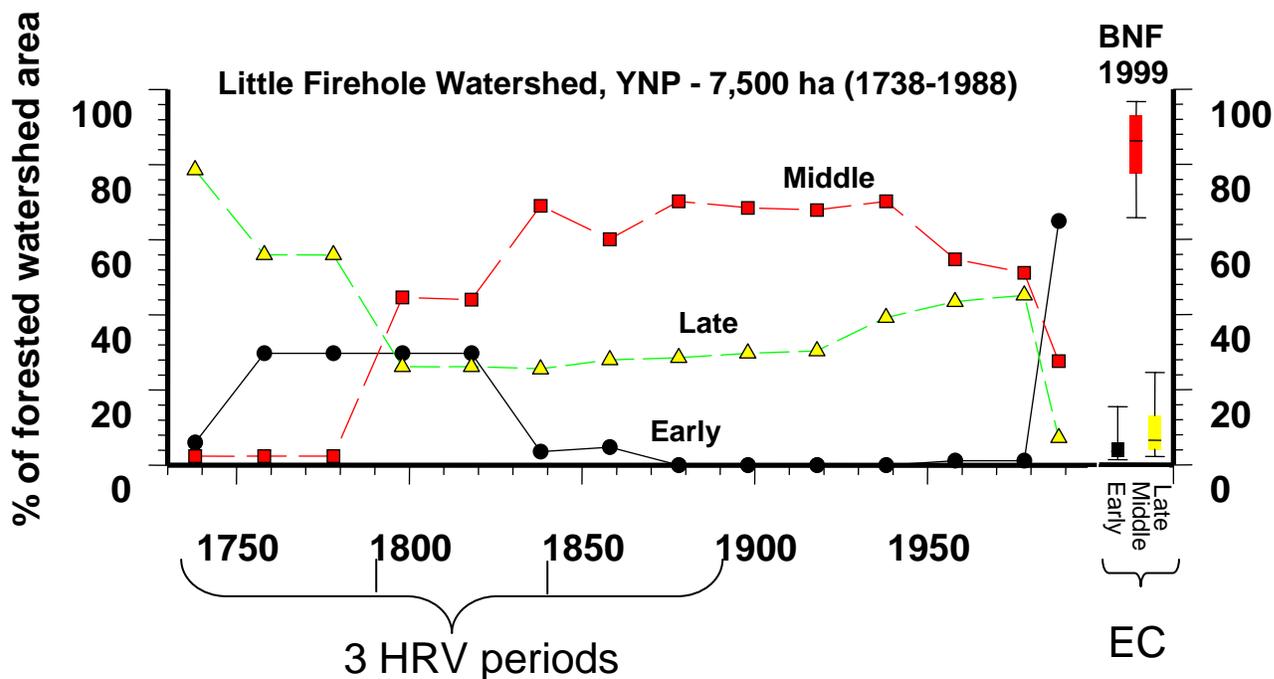


Figure 39. Historic range of variability of percentages in different successional stages of lodgepole/spruce-fir forests within one watershed of Yellowstone National Park (HRV proxy, from Romme 1982 and recent 1988 data) compared to existing conditions on the Bighorn National Forest (BNF, far right). See Table 6 for definitions of early, middle and late successional stages. The distribution of percentages of each stage for each of the 12 watersheds on the Bighorn National Forest on the bottom figure are shown using boxplots (median is the line in the box; box ends are 25th and 75th percentiles, and bar caps are 10th and 90th percentiles). Existing conditions for late successional stages on over 90% of the BNF watersheds do not fall within the range of means during three 50-year periods in YNP (HRV = 29-63% across time, EC of 11 watersheds = 1 to 21% across space). Similarly, existing conditions of middle successional stages on 90% of the BNF watersheds do not fall within the range of means during the historic period (HRV = 2 to 66% across time, EC of 11 watersheds = 67 to 98% across space). This suggests that percent of area in old-growth forests is below the HRV and for middle successional stages is above the HRV on the BNF. Note that the 1988 fires reduced old-growth forest substantially (to 7%) in the YNP watershed. If the size of these 1988 fires would have occurred even without European-American influences on the landscape, one might conclude that the HRV should include conditions after the 1988 fires.

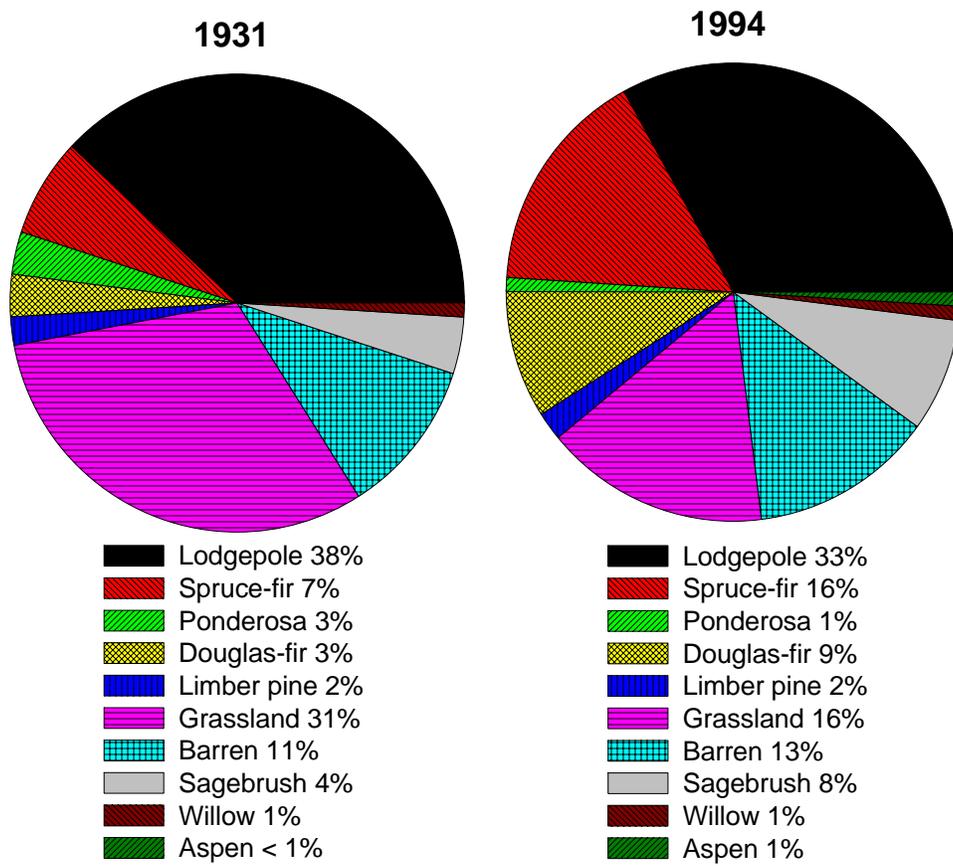


Figure 40. Change in proportions of land cover types on the Bighorn National Forest between 1931 and 1994 (from Dayharsh 1931 and Bighorn RIS database).

Bighorn NF Successional Stages

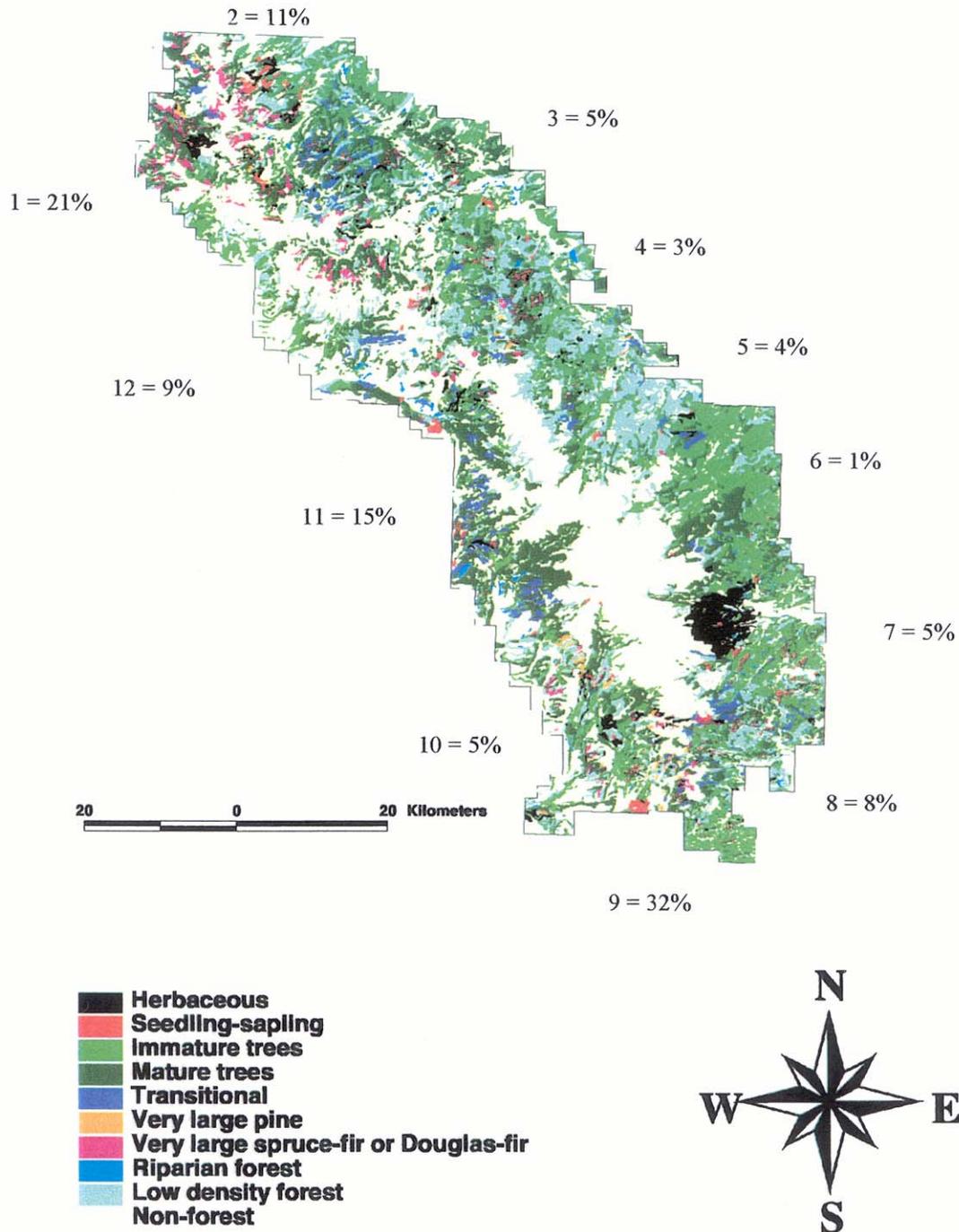


Figure 41. Forest successional stages on the Bighorn Forest (from RIS database). Percentage in late successional stages are shown for each watershed, numbered 1 to 12 (see Fig. 7 for watershed boundaries). Late successional stages include stands with large lodgepole pine, Engelmann spruce, subalpine fir and Douglas-fir.

Disturbance Patterns on the Bighorn National Forest Landscape

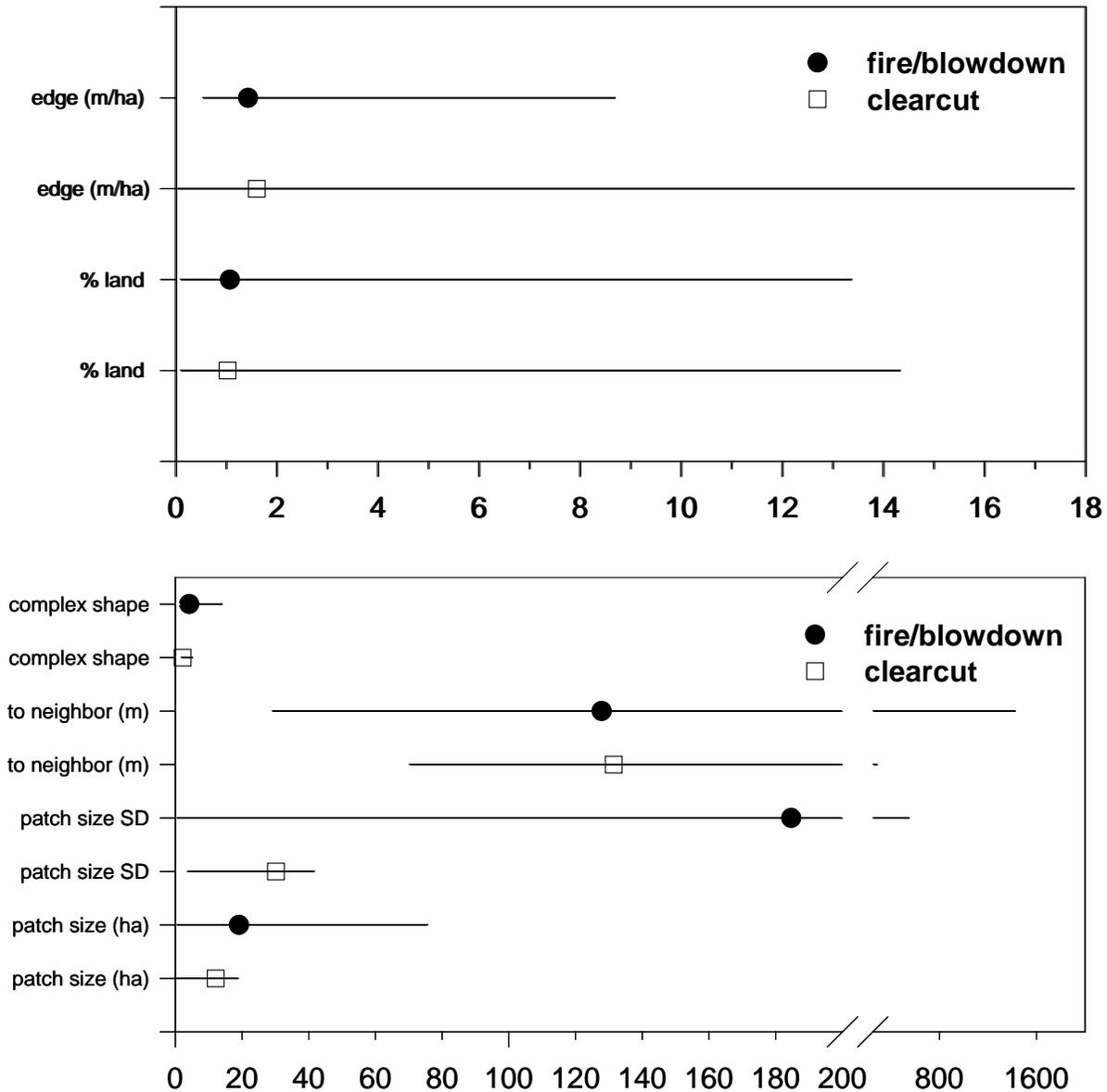


Figure 42. Mean (box or circle) averaged over 12 watersheds and range (line) of variables describing landscape patterns of disturbance patches that were created on the Bighorn National Forest by fires, blowdowns, and clearcuts between 1955 and 1998. SD = standard deviation (measure of variability), “complex shape” = area-weighted shape index, and “to neighbor” = distance to nearest neighboring disturbance patch. Watershed size ranges from 4,162 to 57,067 ha. Means were similar for amount of land and edge in disturbance patches and distance between neighboring patches, but patches formed by fire or blowdowns had more complex shapes and larger and more variable patch sizes. However, the range extended higher for amount of land and edge in clearcut patches than patches formed by fire or blowdowns, an indication that some watersheds had a much higher amount of disturbance than caused by fires or blowdowns.

Disturbance Patterns on Bighorn National Forest

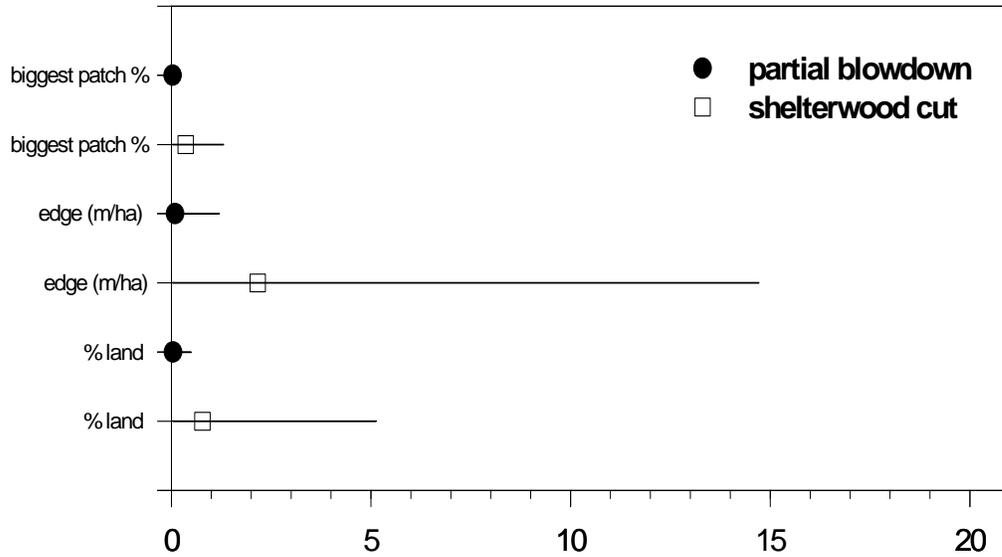


Figure 43. Mean (circle or square) and range (line) of characteristics of “partial” disturbance patches created by blowdowns or shelterwood cuts between 1955 and 1998, averaged over the 12 watersheds on the Bighorn National Forest. The “biggest patch %” value is the percent of watershed composed of the largest partial disturbance patch. Much more land has been affected by partial cutting (shelterwood cuts) than by partial blowdowns.