Part I—Forests and Biomass

Introduction

Millions of acres of forest and woodland in the Western United States have been historically shaped by fire. (Agee 1993 and 1990; Arno and Wakimoto 1988; Covington et al. 1994). Whether induced by lighting or by humans, fire once was the primary means of recycling carbon and nutrients for many forests of all kinds. European settlement and the introduction of grazing, farming, mining, forestry, and fire suppression greatly changed forest systems accustomed to fire regimes, and they underwent a long, slow buildup of woody biomass (Clark and Sampson 1995; Covington and Moore 1994; Everett et al. 1993). In the absence of fire, undergrowth and small trees thrived. Such additional growth in the forest was welcomed for many years as a sign of management success (Langston 1995).

In recent years, however, the toll of fire suppression on the ecosystems has become evident. Plant communities too dense for the moisture and nutrient conditions of a particular site compete with each other for limited resources. When the competition becomes excessive during dry spells, major diebacks occur (USDA Forest Service 1996). Native insects and diseases are usually the agents of death, but the stress of competition is an underlying cause (Sampson et al. 1994). As mortality rates increase, so do the flammable fuels. In many Western climates—characterized by dry summers and cold winters—biological decomposition is too slow to offset the fuel buildup (Harvey 1994). As more living and dead fuels are present, both in larger landscape patches and in the vertical structure of the forest, any ignition in dry weather is likely to result in a major wildfire (Anderson and Brown 1988; Covington et al. 1997). Absent any treatment to remove the fuel buildup, fires are inevitable (Sampson 1999). The fuels have no other route for recycling, and they accumulate until they are removed or burned.

Where fuel loads are high and fuel structures continuous, the result is intense fire that often behaves so violently that suppression may be impossible. Such intense fires kill plant communities that were historically tolerant of milder fires, and their heat causes serious and often permanent soil damage (Borchers and Perry 1990; Cromack et al. 2000; Neuenschwander and Dether 1995; Sampson and DeCoster 1997).[1] Recent research suggests that areas of extreme heat and damage are becoming a larger percentage of the total affected area within a forest fire perimeter (USDA Forest Service 1996; Covington et al. 1997).[2]

In addition to their on-site impacts, these wildfires impose enormous public and private costs. In 1994, the Forest Service spent close to $1 billion on fire suppression activities; in 1996, the figure was in excess of $835 million (USDA Forest Service 1997). Because there are homes and communities scattered throughout much of this territory, it was estimated that about one-third of these expenditures went toward trying to save private property from destruction (USDI/USDA 1995). The smoke from these wildfires affected air quality for weeks, producing more $PM_{2.5}$ (fine particulate) pollution in a few weeks in 1994 than all the nation's diesel engines and smokestacks emitted for the entire year (Core 1995).

Treatment to return forests to a more fire-tolerant condition—consistent with their historical development—usually involves removing excess fuels and introducing prescribed fire when conditions allow low-intensity burns (Arno 1995; Arno and Brown 1991; Biswell 1989; Oliver et al. 1994; Thomas and Agee 1986; USDI/USDA 1995; Mutch 1994). Although treatment approaches are fairly well known for most conditions, treatment is often absent because the material to be removed has low economic value, at-risk landscapes often cover prohibitively large areas, and many areas lack road access (Sampson 1997).

Federal land managers face an additional barrier to forest treatment in the form of groups opposed to harvesting or road building on federal forests. The federal government owns and manages 70 percent of the forest and woodlands in the West (Powell et al. 1993). Many federal lands have been designated as parks, wilderness or reserves, making them off-limits to vegetative manipulation. Such legal distinctions limit preventive treatment, but they do not change the wildfire hazards or the risks a system faces if it burns too severely. The problem of fuel buildup in Western federal forests poses an enormous policy dilemma to the federal government.

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[1] While soil damage has seldom been featured as a long-term fire effect, the increasing amount of fuels involved has brought attention to the fact that some areas may be damaged in ways that will affect long-term ecological functioning. See also Giovannini 1994, McNabb and Cromack 1990 and Sampson 1997.

[2] In their findings, the Assessment Team for the Interior Columbia Basin Study said, "The threat of severe fire has increased; 18% more of the fires that burn are in the lethal fire severity class now than historically." (Quigley et al., p. 181)

The fact that Western forests face a health[3] problem from fuel buildup is extensively documented. No universal agreement exists about a cure, but it is reasonably clear that a broad consensus supports identifying and treating high-priority areas. Some experts propose using hazard-risk models to identify high-priority areas and guide public debate. Several such models are in development across the West.[4]

In general, hazard-risk models help identify areas where there is high probability of ignition and where vegetative conditions will support high-intensity wildfires that put people, property and environmental values at risk. In many cases, the highest priority areas identified for treatment using hazard-risk models are those associated with the wildland-urban interface (Davis 1989), key watersheds that serve municipal water supplies or critical stretches of habitat for endangered fish such as Pacific salmon. Often, these are areas where existing roads and access combine to produce less public controversy about treatment than would occur for less-accessible areas.

What remains to be addressed, however, is the enormous problem of what to do with all of the material that results from reducing an area's fuel load (Nijhuis 1999). Although some of the material to be removed in a forest-health project may be saleable on local markets, much of it is not.[5]

On private forestlands, the value of biomass for energy is too low to cover the cost of gathering and hauling it to market. Owners may be willing to produce biomass fuel as part of a timber harvest where sawlogs, pulp and biomass can be combined. This allows some of the costs of biomass disposal to be written off and offers a least-cost way to achieve forest health goals. Private forest owners' decisions will be shaped by the technical information they receive from professional foresters and by the cost involved. If these owners can receive anything close to break-even prices, the result seems likely to be a significant biomass supply from the private lands of the region.

Forest treatments on federal lands raise political problems as well as economic, but the political climate of inertia regarding such treatment may be changing. In October 2000, the Forest Service released a major study that notes the agency will follow an executive order by collaborating with others to analyze the economic feasibility of increasing the use of biomass (USDA 2000). In addition, the Biomass Research and Development Act of 2000 (PL 106-224) provides a legislative mandate for the USDA and the U.S. Department of Energy to cooperate on policies and procedures that promote research and development leading to the production of biobased industrial products, such as fuels and chemicals. Under the act, applicants can earn grants, contracts and financial assistance for conducting research to improve the conversion of biomass into biobased products, for developing technologies that would result in cost-effective and sustainable industrial products, and for promoting the development and use of agricultural and energy crops for conversion into biobased fuels and chemicals. Also important is the inclusion within the FY 2001 appropriations bill of a major new $250 million fund for the Forest Service and the Department of the Interior to carry out fuel management activities.

This new federal emphasis on forest health and biomass development could have a significant impact in the West. For example, forest health problems in Eastern Oregon led Oregon Governor John Kitzhaber and Mike Dombeck, Chief of the Forest Service, to create the Blue Mountains Demonstration Area (BMDA) on June 30, 1999. The goal is to coordinate efforts so that ecosystem restoration will be accelerated in the Blue Mountains in a manner that benefits local communities and unites land managers and scientists in a cooperative effort across the landscape.

In both of the Oregon cases outlined in Part III, but particularly in Grant County, it appears that the single most important factor is not the amount of fuel physically available but the ability of the USDA Forest Service to carry out the kinds of forest treatments that would make that fuel available to a biomass energy facility. The fuels are there; the ability to deliver them in necessary quantities, over a long enough period of time, is not. Whether the new emphasis created by the 106th Congress and the creation of the BMDA

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[3] In this report, "forest health" means a sustainable, more fire-tolerant forest condition and the elimination of unnatural woody biomass accumulations that have resulted from fire suppression in the past.

[4] One of the early efforts to develop wildfire hazard-risk models was completed on the Boise National Forest, as described in Boise National Forest 1996. The approach was expanded in Sampson et al. 2000.

[5] In Arizona, Wallace Covington and his co-workers removed 58 tons of non-merchantable biomass per acre before they felt the ponderosa pine site had been properly prepared for a prescribed fire that would mimic historic fires; that is, the prescribed fire would not kill the large trees they were trying to save (Covington et al. 1997). They hand-raked around big trees and disposed of the excess material by open burning in a nearby pit. That is possible for a research project, but not feasible for large-area treatment. Moving to field scale treatments will demand methods that can be done with available labor and machines at reasonably low costs, and that can dispose of the material in some way other than open burning so that air pollution does not become the limiting factor.

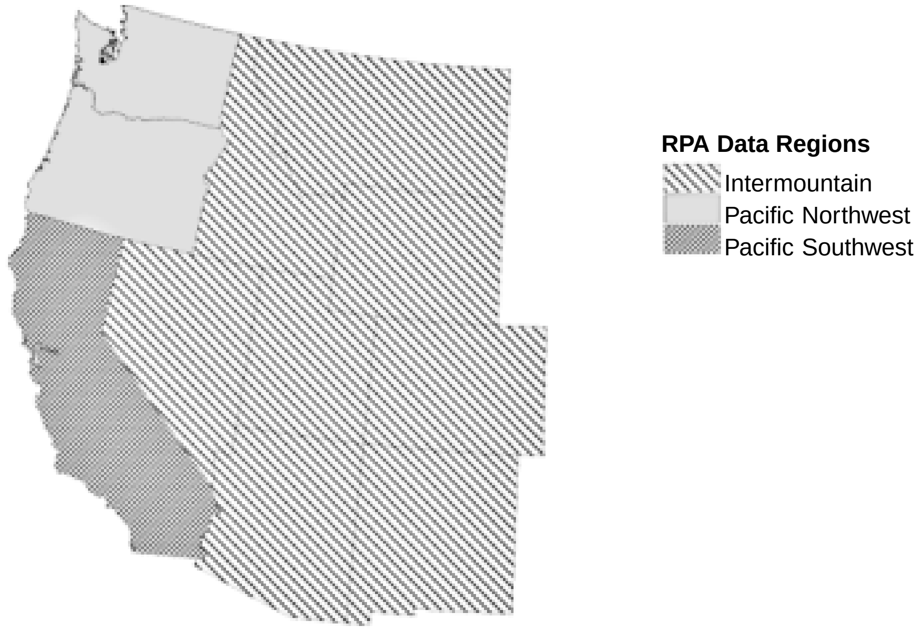

Figure 1.1 Forest resource data provided by the USDA Forest Service is often summarized by regions, as shown above for the Western United States. Note: These are not the same as Forest Service Administrative Regions.

will change that picture significantly is not yet clear, but the signs are at least hopeful.

This report is not a comprehensive attempt to establish the case for increased attention to the health of forests in the West. It is, at best, an overview that touches on a few situations and areas. A comprehensive dataset that ties current forest condition to locations does not exist. As a result, the report demonstrates that some forest types often exhibit a certain set of conditions, but no local conclusions can be drawn until local situations are assessed.

Nor does the report try to develop a prescription for treating specific forests as a means of returning them to a more fire-tolerant and sustainable condition. Those prescriptions must be adapted to the particular circumstances in each forest situation and can only be developed locally by people who understand those places and who must live with the consequences of their actions on the land.

This report will, instead, focus on what can be done to help provide a policy basis for environmentally, socially and economically positive approaches to forest-health problems and biomass development. The major focus is on the various methods through which biomass unsuited for current industrial uses can become a feedstock in energy production. Because much of the available feedstock in the West is on federal lands, the report places some emphasis on USDA Forest Service policy opportunities. However, regardless of land ownership, having excess biomass burned directly as a feedstock for electric power generation or used in the chemical production of biofuels would represent a more positive use than leaving it on the land to fuel a wildfire of destructive intensity.

With the limited data available, it is not possible to say with assurance how much land is in any particular condition, how much of that land should be considered high priority in a hazard-risk analysis, or whether public opinion would support treatment. What we can say with assurance is that hundreds of thousands—if not millions—of forested acres in the West need attention soon. As land managers and local communities struggle with how to respond, it is our hope that reasonable options will be found and that new approaches to biomass energy production have an opportunity to provide some of those options

General Forest Conditions and Wildfire Hazards

In the following discussion, references to "the West" refer to the 11 conterminous Western United States (Figure 1.1). Although generally well understood by the public, this area does not lend itself to easy analysis in terms of forest conditions. Data from the USDA Forest Service, which provides virtually all of the large-area information on forest conditions, are normally broken down by regions (Figure 1.1) (Powell et al. 1993). Those data presentation regions, it should be noted, are not the same as Forest Service Administrative Regions.

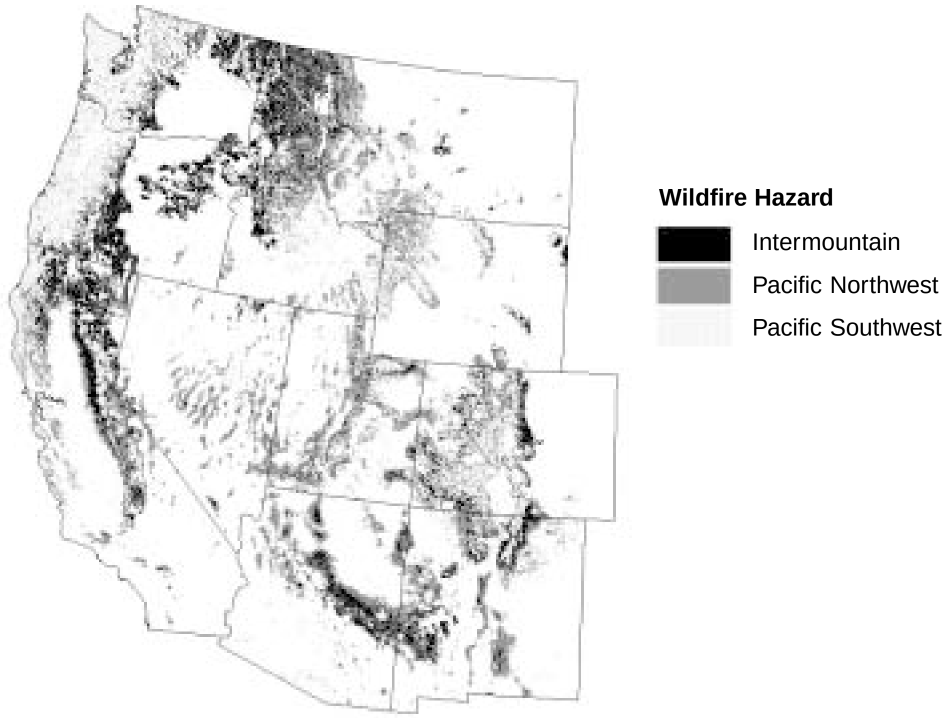

Figure 1.2 General wildfire hazard map of the Western U.S., indicating forest types that are at risk of suffering wildfires outside the historical range of severity (Sampson and DeCoster 1998).

There are anomalies that make these data interpretations a bit difficult at times. One is that Hawaii is included with California in the Pacific Southwest Region (PSW). The inclusion can, at times, influence regional dataset comparisons. We try to minimize that confusion by subtracting the Hawaii data out of the PSW regional data wherever possible.

The second difficulty arises in the Pacific Northwest dataset, which classifies the Douglas-fir forest type as one single type on both sides of the Cascade and Coastal mountain ranges. However, the west-side Douglas-fir is different from that of the east-side Douglas-fir. On the east-side, drier conditions and more frequent fires led to forests of mature ponderosa pine that were maintained as a seral-stage forest on Douglas-fir sites.[6] When those east-side forests are protected from fire, stands of small pines and Douglas-firs create a fuel ladder that supports lethal crown fires. A similar condition does not develop on the warmer, wetter Douglas-fir sites of the west-side forests. There, fire return intervals are much longer, and pure or nearly pure stands of mature Douglas-fir make up many of the highly valued old-growth forests. The fact that both types of forest are shown as "Douglas-fir" on the general forest type maps and in the datasets for Oregon and Washington creates some difficulty in sorting out exactly how much of each type of condition exists.

The final complexity is the enormous range of different conditions under which ponderosa pine occurs in the West. Because it tends to occupy many of the landscapes where settlement impacts have been most prevalent, and because it is one of the forest types that were most dramatically affected by fire suppression, ponderosa pine is one of the forest types of greatest importance in today's forest health concern.

The most pervasive influence on Western forest conditions has been fire exclusion. Its effect has been most pronounced on the forest types that historically experienced a regime of frequent, low-intensity wildfires (Agee 1998). Low-intensity fires once ranged over large areas of Western forests, leaving a patchy forest pattern that altered the effect of subsequent fires. A century of fire exclusion has often resulted in a convergence, or filling-in, of patch structure, so that now large areas are very similar. When wildfire strikes those areas, much larger areas may suffer more uniformly severe effects (Neuenschwander et al. 2000).

[6] A seral-stage forest is one that is held in a temporary or intermediate stage of succession (Helms 1998).

Table 1.1 Unreserved forest land (in thousands of acres), all productivity classes and all owners, Western United States (1997 RPA, Table 9).

	Pacific Northwest		Pacific Southwest[a]		Intermountain		Western States	
Forest Type	*Area*	*Percent*	*Area*	*Percent*	*Area*	*Percent*	*Area*	*Percent*
Pinyon-juniper	2,552	5.59	1,461	4.28	42,927	35.67	46,940	23.46
Douglas-fir	17,237	37.77	1,979	5.80	17,860	14.84	37,076	18.53
Ponderosa pine[b]	7,095	15.55	7,327	21.47	15,245	12.67	29,667	14.83
Western hardwoods	5,210	11.42	9,186	26.91	13,303	11.05	27,699	13.84
Fir-spruce	4,294	9.41	2,946	8.63	14,617	12.15	21,857	10.92
Lodgepole pine	2,426	5.32	166	0.49	10,499	8.72	13,091	6.54
Chaparral	235	0.51	4,386	12.85	126	0.10	4,747	2.37
Other softwoods	232	0.51	5,136	15.05	2,619	2.18	7,987	3.99
Non-stocked	900	1.97	697	2.04	617	0.51	2,214	1.11
Hemlock-Sitka spruce	5,108	11.19	11	0.03	1,510	1.25	6,629	3.31
Larch	287	0.63	0	0.00	889	0.74	1,176	0.59
Redwood	6	0.01	732	2.14	0	0.00	738	0.37
Western white pine	54	0.12	105	0.31	131	0.11	290	0.14
All Forest Types	45,636	100.00	34,132	100.00	120,343	100.00	200,111	100.00

[a] Hawaii is included in this regional summary. Its 1.6 million acres of unreserved forests are classified mainly as western hardwoods.
[b] Jeffrey pine is included with ponderosa pine, mainly in the Pacific Southwest.

Forest Types of the West

About 55 recognized forest types exist in the West, but more than 80 percent of the total unreserved forest area is described by the top five forest types in Table 1.1.

One of the challenges in looking at forest data and trends is to understand what they might indicate about forest condition. Most estimates of forest condition have historically been made on the ground, stand-by-stand, as foresters made decisions about forest management and treatment. There, the signs of an oncoming health problem may be fairly clear. Overstocked, stressed stands or symptoms of insect or disease outbreaks may signal the need to take management action.

Making broad-area estimates has been much more difficult, however. Until recently, the forest inventory data available from the Forest Service did not provide information about the stand condition of the forests. If the inventory showed the amount of acres and timber in the region that was in the 5-to-7-inch size category, there was little indication whether these were young forests that were growing freely and would soon become larger trees, or whether they were older, stagnated stands that would not grow another inch before insects or fire killed them. As the area of stagnated stands grows larger, this distinction becomes more important. Current studies are trying to provide better data, particularly on federal lands.

There are other methods, however, that may offer useful insights. An estimate developed by the Forest Service, and cited by the General Accounting Office, shows 39 million acres—almost one-third—of the National Forest system in the Interior West to be at high risk of catastrophic wildfire (GAO 1999). A more recent study conducted by Forest Service researchers developed improved data on historical vegetative conditions and coupled those with current conditions to arrive at estimates of the forest types that are significantly outside their historical range of variability (Hardy et al. 1999). These studies were then used to identify forested areas that were so far outside their historical ranges that a wildfire posed significant risks of altering their ecosystems through the destruction of one or more critical ecosystem components or processes. In plain terms, these were forests where the current fuel conditions pose a hazard so great that the ecosystems could be diminished in a long-term or permanent way if a wildfire is allowed to burn.

The study classified forest condition in three categories:

Condition 1—The ecosystem is largely intact and functioning in historical patterns. It may be subject to wildfire, but the disturbance patterns and severity should be fairly normal.

Condition 2—The ecosystem has undergone moderate changes, and conditions have shifted toward a less resilient system. A wildfire disturbance may or may not cause the loss of ecosystem components or processes.

Condition 3—The natural, historical disturbance regime of the ecosystem has been significantly altered, and the

Table 1.2. Estimated amount of forestland in the National Forest System, Western States, by historical fire regime and current condition, 2000.

Historical Fire Regime	*Condition 1*	*Condition 2*	*Condition 3*	*Fire Regime Totals*
0–35 years; low severity	4,846,406	23,719,091	24,158,447	52,723,944
0–35 years; stand replacement	762,311	621,459	284,168	1,667,938
35–100+ years; mixed severity	14,242,726	23,535,004	6,177,545	43,955,275
35–100+ years; stand replacement	3,689,236	830,755	7,561,081	12,081,072
200+ years; stand replacement	14,829,079	1,030,166	1,132,111	16,991,356
Class Totals	38,369,758	49,736,475	39,313,352	127,419,585

Source: Hardy and Bunnell 1999.

current condition predisposes the system to major changes, including the possible loss of key components or processes.

Obviously, Condition 3 describes places that should be of concern from an economic, environmental or national policy point of view. The best estimate is 39 million acres of Condition 3 forest exist in the National Forests of the Western United States. Table 1.2 highlights the fact that by far the largest category of Condition 3 forests includes land with historical fire regimes of 0 to 35 years and low-severity fires. That category is dominated by the ponderosa pine and dry Douglas-fir forests, reinforcing the conclusion that it is these forest types that are at greatest ecological risk.

The current condition of these high-risk forest areas is a result of past and current management—of that there is little controversy. The problem, of course, is how best to achieve the goal of returning them to a more ecologically stable state (Sampson 1992a). In the eyes of some people, it is best to allow nature to take its course. Because past management was part of the problem, they reason, it is important to prevent future management from continuing to meddle in the situation. Others see it differently. Past management has provided lessons upon which future management can be based, they say. Because of the enormous amount of fuel involved, the inevitable result of "letting nature take its course" is to see a fire that is likely to damage soils and watersheds, set forest ecosystems back into much degraded conditions and perhaps preclude forest recovery for generations. From this perspective, the future is best served by taking management actions that will improve the chances for the forest to become more tolerant of future wildfire conditions, preferably to the point where the forest ecosystem is sustainable long into the future (USDA Forest Service 2000).

Condition and Treatment Approaches to Forest Types of the West

Pinyon-Juniper—This most extensive forest type in the West is also one of the least well documented in terms of condition and change. Because it produces less than 20 cubic feet of wood per acre per year, this type of land has not been classified as timberland. As a result, the Forest Service has little information about its growth rates or area change. However, studies in Oregon demonstrate that juniper woodlands began increasing both in density and in area during the latter part of the nineteenth century (Miller and Rose 1995; Miller and Wigand 1994). Traditional Native American burning practices stopped when tribes were forced from their lands, cattle and sheep grazing reduced the grasses and shrubs that formerly carried fires through the landscape, and cowhands and settlers suppressed every grass fire they could handle.[7] Without periodic fire to hold its advance in check, juniper began to expand, aided, it appears, by the warmer and wetter conditions that followed the end of the "Little Ice Age" around 1850. As fire suppression and grazing have continued, juniper's expansion has proceeded virtually unchecked (Laroe et al. 1995).

Managing juniper forests is a major challenge, particularly for the Bureau of Land Management, which has extensive areas of remote pinyon-juniper woodlands. Few of

[7] When viewed from the standpoint of today's conflagrations, it is hard to imagine people with only shovels and wet blankets putting out a range or forest fire. When those fires were burning regularly, however, the fuel accumulations were much smaller, making the fires more manageable. Pyne (1982) tells the story of "beef drags," where a bull or cow would be slaughtered, split open, then dragged behind two horsemen; one riding on the fire side and on one the unburned side of an advancing grass fire. One chronicler says that such tactics could put out more range fire than 50 men working only with wet blankets and sacks.

the trees are cut for commercial purposes because no market exists except for occasional firewood or fence posts, and those trees that are cut generally are not part of a planned area-management scheme. Efforts such as chaining and grass planting have been largely unsuccessful in restoring desired grassland conditions. Prescribed fire is difficult to use successfully because tree densities are often too low to carry a fire from tree to tree, and the juniper's aggressive root systems keep grass and shrubs from growing profusely enough to help maintain the fire.

Another complicating factor for managers of juniper forests is that adjoining grasslands and sagebrush steppes often have been taken over by cheat grass (*Bromus tectorum),* an exotic annual that builds up large thatches of highly flammable dead material and that reseeds aggressively following a fire. Dry cheat grass ranges burn with dangerous speed, and the fire effect may be to favor the exotic weed rather than reduce it. It is estimated today that cheat grass has become a dominant species on 17 million acres of the Western sagebrush steppe, with the potential to infest another 62 million acres (Ferry et al. 1995). It is possible that some of the intensive-grazing schemes being tested in the West can help bring back perennial grass cover, but the invading junipers would need to be killed or removed first (Daggett 1995).[8]

Ponderosa pine—In addition to being the dominant forest type on some 28 million acres of Western forests (Table 1.1), ponderosa pine is an important component of the Douglas-fir forest types in the Intermountain Region (16 million acres) and the mixed conifer forests. In California, a similar species called Jeffrey pine is included with ponderosa in both the datasets and the management schemes. Historically, these species were often found in park-like stands with grassy understories maintained by frequent, low-intensity surface fires that burned through the dry summer grass, killing young trees and consuming fallen bark flakes, needles and branches. Because of the location of ponderosa pine forests on the lower and gentler portions of the Western landscape, it is likely that Native Americans ignited many of the historical fires in these forests as part of their land management strategy. Fire exclusion efforts that began with

[8] The use of intensive short-term grazing is increasingly promoted as a way in which native grass re-establishment may be possible.

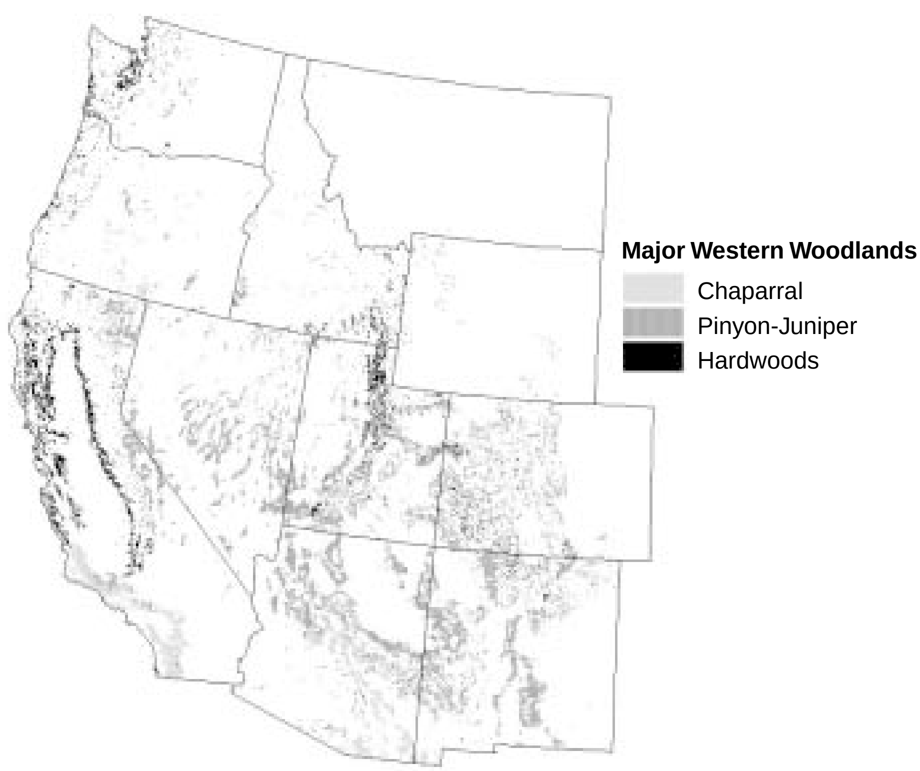

Figure 1.3 Woodlands with little or no commercial forest product output cover extensive areas of the West.

Figure 1.4 Ponderosa pine occurs under a wide range of conditions in the Western United States.

the earliest European settlers began to change these stands, and brush and small trees were able to grow around the larger pines (Mutch 1994). Because these forests were often found on the more-accessible foothills and steppes of the region, the valuable large trees were heavily harvested, often in the process of converting the land to crop or pasture.

Extending from Mexico to Canada, from elevations of less than 1,000 feet in the valleys of the North to more than 12,000 feet in the Southwest, and from annual precipitation zones of less than 16 inches per year up to more than 40 inches per year, ponderosa pine is a species of great diversity. This diversity contributes to significant differences in the conditions that affect the pines and their treatment.

In the drier, colder portions of the pine's range, for example, fire suppression coupled with slow decomposition rates can result in a large pile of bark flakes, needles and other debris on the forest floor and around the base of large trees. Where this has occurred, even careful use of prescribed fire may achieve such high heats around the base of trees that they are killed. A similar period of fire exclusion in the warmer, moister northern Idaho forests will not result in any significant buildup of ground fuels, and a prescribed fire that is properly handled poses no threat to trees.[9] With such a wide range of conditions possible, it is important that generalizations about this species be used with caution.

The impact of fire exclusion on ponderosa stands was noted as early as the 1940s by observers such as Harold Weaver, but conventional thinking for many years rejected the idea that fire might be necessary to protect the integrity of the ecosystem.[10] Today, prescribed fire is much more widely accepted, but the underlying situation has changed so dramatically that, as one study of the Blue Mountains of

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[9] From discussions with Leon Neuenschwander, University of Idaho Forest Ecologist, about widely varying situations. For example, in Heyburn State Park in northern Idaho ground fuel buildup around large trees is virtually non-existent. In contrast, on the Boise National Forest, large piles of bark flakes make prescribed fire much more difficult to achieve without killing the large trees (See also Covington et al. 1997).

[10] This history is well reviewed in the works of Agee, Biswell, Covington and Pyne. Harold Weaver published his classic "Fire as an ecological factor in the ponderosa pine region of the Pacific slope" in 1943. The outcry raised by the article resulted in several subsequent articles being published with the following caveat imposed by his employer: "This article represents the author's views only, and is not to be regarded as an official expression of the Indian Service on the subject discussed." Being ahead of one's time is no easy task, as many of the fire researchers in the West discovered.

Oregon noted, removal of unnatural fuel accumulations and manipulation through mechanical harvest will be needed to modify current stand conditions before fire can play its historical role (Mutch et al. 1993).

A variety of estimates have been made concerning the amount of Western ponderosa pine forest that will need some form of fuel reduction or modification before it can be safely returned to an historical fire regime. However, until remote imaging technology can be improved to the point where fuel amounts and structures below the forest canopy can be estimated, there are no broad-area estimates that can accurately pinpoint either the amount or location of those areas.

What we can safely say is that millions of acres of ponderosa and inland Douglas-fir forests are at serious risk of lethal wildfires if they are not treated within the next decade or two (Covington et al. 1994). The best available estimates suggest that the area involved covers approximately 39 million acres (USGAO 1999; Hardy et al. 1998). These forests have reached conditions that are increasingly unstable and vulnerable (Scott 1998). Reaching such a precarious position has taken, in many cases, 100 years or longer. In contrast, devastation by wildfire would take only minutes or hours if trees are ignited during a dry, windy summer period, given their current condition.

If such wildfire occurs—as it has since 1989 on nearly one-third of the ponderosa pine forests of the Boise National Forest in Idaho—the recovery of many areas would be questionable. For the most part, these wildfires are uniformly lethal. In the 1992 Foothills fire, for example, even scattered lone trees and isolated north-slope pockets of pine were killed. Trees that had survived dozens of prior fires could not tolerate the heats generated by the amount of fuel being consumed (Sampson et al. 1994). The risk is that, for the foreseeable future, these lands will become brush fields rather than forests.

Ponderosa pine is also the Western forest type that is most often used and altered for residential and recreational development because it generally occupies the lower and more accessible areas. Treating ponderosa pine forests becomes even more urgent in the places where they form part of the "wildland-urban interface" made up of mixed forestland and development.

Treatment approaches to ponderosa pine generally focus on returning the stands to a condition that is likely to survive a future wildfire. This is generally done by removing or reducing understory fuels, thinning the stand to reduce the likelihood of a crown fire and pruning dead or low branches so that a ground fire is less likely to burn into the crowns (Scott 1998). Treatment guidelines are generally agreed upon by land managers, but they are still likely to be controversial with people who view timber harvest or tree cutting as the problem, not the solution.

Mixed conifer—Mixed conifer is the most complex of the forest types in the West, and one that is not broken out as a forest type in Forest Service data (see Table 1.1). These forests have a wide variety of coniferous and, in some places, hardwood species. They differ in their position on the landscape, their fire regimes, and the manner in which they respond to disturbances such as fire (Agee 1993). They are generally found at somewhat higher elevations or on cooler, moister landscape than ponderosa pine forests, and they may adjoin true fir, spruce-fir, sub-alpine or alpine areas at their upper limits. Ponderosa pine can be found as a seral species in most of these forest types, although it will not be present in every site. Douglas-fir also can be found in most of these forests, either as a seral or climax species. Much of the area listed as Douglas-fir (Table 1.1) in the Pacific Southwest and Intermountain data regions is included within this general category.

Although highly variable, most of these mixed conifer forests developed in connection with fairly short fire-return intervals, subject both to lightning-caused fires and Native American ignitions (Agee 1993). This resulted in many areas being dominated by the more fire-resistant seral species such as ponderosa and Jeffrey pine, western larch and Douglas-fir. Fire suppression has resulted in a significant increase of the less fire-resistant species such as white fir, grand fir, lodgepole pine, incense cedar and hardwoods. The result is a fuel build-up similar to that of the ponderosa pine forests and the likelihood that an ignition during dry conditions will turn into a lethal crown fire. Another ecological result is the decrease in biological diversity in the understory and on the forest floor because dense shade and competition from the trees has diminished herb and shrub components (Agee 1993).

Restoring these forests to a more fire-tolerant condition will be both complex and costly. The amount of biomass on many sites will make restoration with prescribed fire extremely difficult, often requiring two or three treatments. Fuel reduction treatments prior to prescribed fire may be needed in many places, and, in areas intermixed with housing and other development, some form of non-fire treatment regime may be necessary. These treatments often look like the "thinning from below" described for ponderosa pine, but the presence of additional species makes the design of a treatment project somewhat more complex. In general, thinning tries to not only reduce fuels and break up fuel ladders but also to shift species composition toward the more fire-tolerant species such as ponderosa and Jeffrey pine, western larch and Douglas-fir. In the process, most of the small and medium-sized white and grand firs, incense cedars, and Douglas-firs would be removed.

Lodgepole pine—Like all Western forests, lodgepole pine forests have been affected significantly by the fire-exclusion efforts of the last century. Although normally found at higher elevations and colder climates than ponderosa pine, this species is found in areas where remnant trees suggest that it has replaced ponderosa pine or mixed conifer forests (Figure 1.5). Because they are found in higher, colder areas, almost 82 percent of Western lodgepole pine forests are found in the Intermountain Region, and almost 90 percent of the lodgepole pine in that region occurs on the National Forests, National Parks and other public lands. Treatment for lodgepole pine would take place on some of the most wild and remote lands in the nation.

Lodgepole pine forests have developed under a combination of low-, moderate- and high-severity fires that have occurred in complex and not-very-well documented frequency and locational patterns. Estimated fire-free intervals have historically ranged from 50 to as long as 350 years (Agee 1993). The species is often found in fairly pure stands that have resulted from regeneration following a stand-replacing fire. The forests are susceptible to insect attack by mountain pine beetles, which can combine with small, patchy fires to create a mosaic of age and size patterns. Where fire suppression has been successful in preventing the small events, the result can be larger areas of more uniform stands that, when attacked by the mountain pine beetle, can produce epidemic conditions and wide-area mortality. This, in turn, can create conditions for larger-than-historical fires such as those at Yellowstone Park in 1988.

Because the dense stands that often characterize lodgepole forests allow room for few or no understory plants, and because surface fuels are limited, it is difficult or impossible to use a cool ground fire as the prescribed fire in many situations (Agee 1993). In the backcountry, what is called a "prescribed natural fire" may be a lethal crown fire, mimicking the natural regime. Often these would be lightning-ignited rather than management-ignited, but they would be allowed to run their course because they would burn in

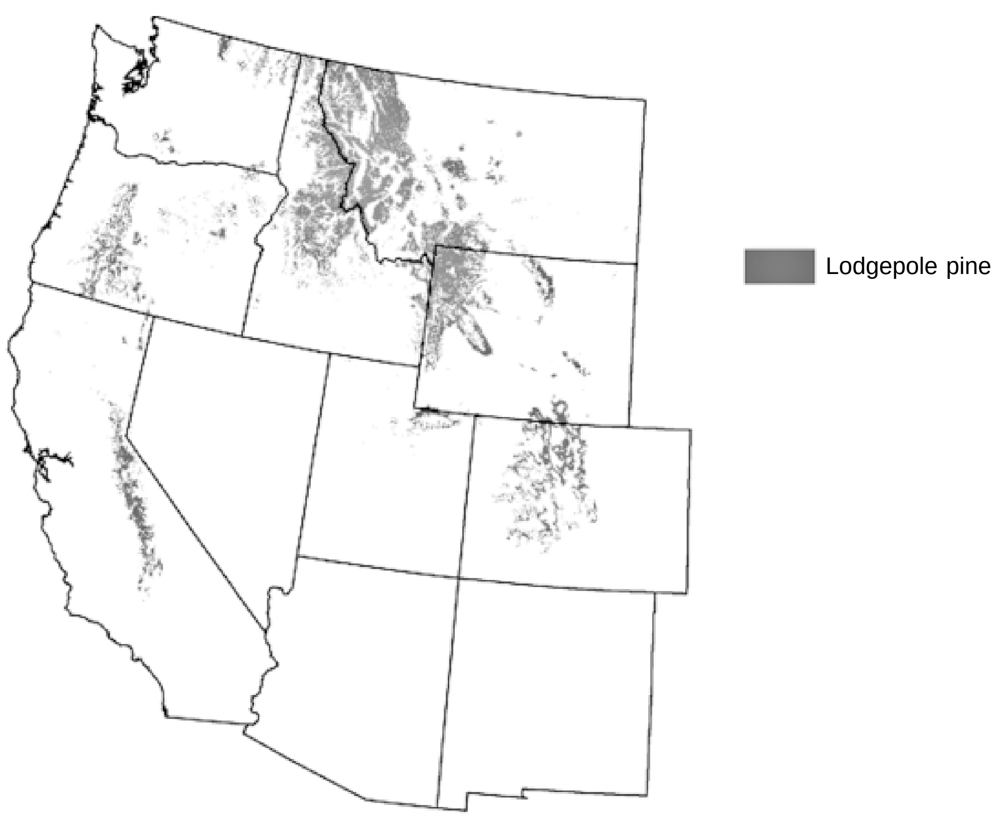

Figure 1.5 Lodgepole pine inhabits higher and colder areas, and its normal wildfire is usually stand replacing.

an ecologically acceptable manner. Where that is not feasible, however, fires would be suppressed, and any treatment would probably involve some form of fuel removal, either through biomass harvesting or thinning accompanied by pile burning of slash.

Environmental Issues and Forest Health Treatment

Soil Damage and Ecosystem Recovery

The most important impact of the large, intense wildfires of the recent decades is likely the damage to the soils. This soil damage is a much more serious and enduring setback than killed trees. Fire can reduce thin mountain soils to little more than bedrock. The most extreme fire and post-fire effects are long-term ecological setbacks that may permanently convert forest sites to shrub or desert sites. Where that happens, the price of allowing such high fuel loads to burn will be paid far into the future.

One problem with evaluating soil damage is that, in many cases, the extent of the damage will not be clear for years or even decades. In some places, on the other hand, the degree of damage is fairly apparent. In the 1992 and 1994 fires on the Boise National Forest, for example, some soils were so badly heat-damaged that the topsoil literally slid off the slope, in a process called *dry raveling*. Similar impacts were seen on the Buffalo Creek fire in Colorado (see Water Quality and Stream Flow below). In both cases, summer rainstorms caused additional soil erosion before protective ground cover could be re-established in some areas. Those sites were damaged to the extent that forest vegetation may not be able to regenerate on what is left behind. In other areas, the damage was less evident, although reduced organic matter and nutrient losses were clearly involved, particularly in the soil layer above the hydrophobic layer that was formed a few inches down (Agee 1993; Cromack et al. 2000; Giovannini 1994; McNabb and Cromack 1990; Sampson 1997).

What can be safely assumed is that the most marginal soils will suffer the greatest damage (Cromack et al. 2000). Soils with low organic matter and nutrient content in their pre-fire condition will experience far slower recovery following an intense wildfire. The extent of soil damage is determined largely by the degree and duration of heating that takes place. Degree and duration, of course, can be highly variable within the boundaries of any wildfire, so it defies easy generalizations.

In addition to changing soil quality, fire may significantly affect a burned site's micro-climate. The loss of shade, coupled with the loss of local seed sources, may prevent reforestation for decades or longer. Follow-up monitoring is being performed on many of the recent wildfire areas, but it is too soon to draw many conclusions about the degree to which the burned sites have been degraded. It is not yet clear whether these sites will simply be delayed a few years before they begin to recover, or whether they instead have begun a downward spiral of deterioration that could eventually turn into the desertification process. The marginal forest sites in the West are often very close to nearby deserts, and any significant soil change, particularly if it is followed by a period of adverse weather or permanent climate change, could significantly move the forest edge.

Air Quality

With prescribed fire being an important part of many forest treatment strategies, the issue of air quality must be addressed for many parts of the West. Wildfire emissions are often violent, difficult to monitor and impossible to regulate. Prescribed fire emissions, on the other hand, are relatively easy to monitor, and, because they occur as the result of human-caused events, are susceptible to regulation.

Under the Clean Air Act, the U.S. Environmental Protection Agency is responsible for this regulation. It established national standards in the 1980s for particulate matter (PM) smaller than 10 microns in size (commonly referred to as PM_{10}). After a lengthy review, the agency determined that there were important health effects associated with even very small particles and, as a result, established new standards for particulate matter smaller than 2.5 microns ($PM_{2.5}$). This change focused additional attention on the PM emissions from combustion smoke. More than 90 percent of wood smoke particulates are small enough to enter the human lung, and their average size is near enough to the wave length of visible light that they can scatter sunlight and dramatically reduce visibility (Pyne et al. 1996).

The amount of PM emitted by a fire event depends on the amount of fuel burned and the type of fire—flaming or smoldering. Total PM emission estimates range from around 25 to 40 pounds per ton of fuel burned, with the higher estimates associated with smoldering fires (Hardy et al. 1992). Precise modeling should be used to consider the type of fire in each event, but a rough estimate of about 30 pounds of PM for each ton of fuel burned in either prescribed or wild fires can be used (Hardy et al. 1992; Ward et al. 1989).[11]

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[11] Colin Hardy, in a personal communication, cites estimates of 25 pounds of PM_{10} per ton of fuel consumed in a ponderosa pine broadcast-burn slash fire and 30 pounds of PM_{10} per ton of fuel burned in forest wildfires.

Table 1.3. Fuel consumption estimates for fire events in ponderosa pine on Douglas-fir site, Boise National Forest.

Type of fire event	*Tons of fuel consumed per acre*
Prescribed fire in low-density stand	11.0
Low intensity wildfire in low-density stand	20.0
Moderate intensity wildfire in high-density stand	74.0
High-intensity wildfire in high-density stand	79.5

Fuel consumption likewise can vary considerably among forest types and fire events, making a general estimate difficult to derive (Fahrenstock and Agee 1983). In the ponderosa pine research described below (see Designing Forest Health Treatments), fuel consumed per acre ranged from 11 tons for a prescribed fire in a low-density stand to 79.5 tons for a high-intensity fire in a high-density stand. (Table 1.3).[12]

We used the Forest Service's post-fire estimates of the amount of low-, moderate- and high-intensity fire contained within the boundaries of two large 1994 wildfire events on the Boise National Forest to derive an estimate of 47.2 tons of fuel per acre as an average over the 119,400 acres burned in the events (Neuenschwander and Sampson 2000). Those estimates indicate that the Boise National Forest could conduct a significant prescribed fire program, which, if successful in breaking up large areas of heavy fuels and reducing wildfire intensity, could result in a lower average annual emission of air pollutants.

Other factors to be considered in the impact of forest fires on air quality are meteorological conditions, which affect the direction, dispersion and impact of smoke plumes, and land use and population characteristics of the areas likely to be affected as a result. A smoke transport model for the West has been developed to test the likely outcome of large wildfire events (Rigg et al. 2000). A map produced by the model for two simulated 25,000-acre wildfires in Colorado is shown in Figure 1.6.

In planning for prescribed fires today, forest managers must consider fuel moisture conditions on the site along with current and forecast weather conditions. Managers must

[12] The plot data and FIRESUM results were input into the CONSUME (Ver 1.0) model to estimate the fuel consumption of the modeled fire events. This required that crown fuel consumption be added to the CONSUME outputs because that version of the model was limited to surface fuel consumption (Neuenschwander and Sampson 2000).

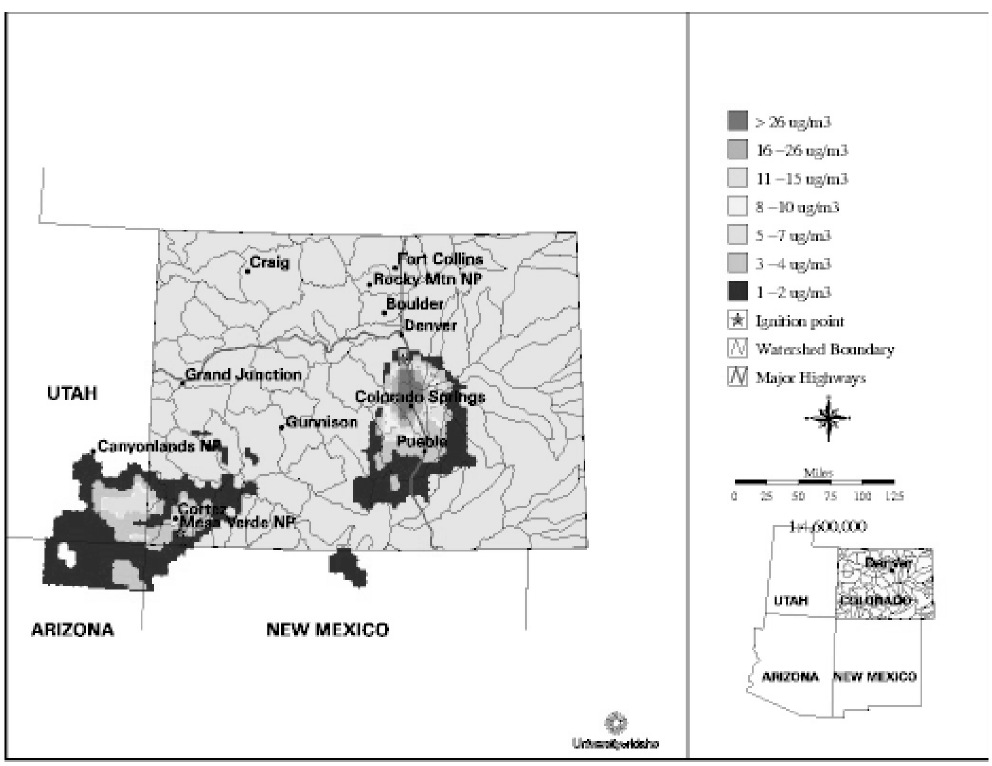

Figure 1.6 Maximum 24-hour PM_{10} concentration from two simulated wildfires.

make certain that the fire will burn with the intended effect on the forest vegetation and that the fire can be managed safely (Biswell 1989). They must assess wind conditions, which limit how burns may be undertaken and can affect air-quality considerations. Wind conditions will also determine the total amount of burning (and therefore smoke production) that will be allowed on a given day. In many areas, this combination of fuel moisture, wind and other weather conditions can amount to a fairly narrow window of opportunity between the time when the fuels are too dry and dangerous to burn and when they are too wet to burn at all.

The Western states have developed State Implementation Plans (SIPs) that establish procedures for regulating prescribed fire based on weather conditions, location, and the amount of fire proposed at any one time. Those SIPs are being reviewed in light of the new EPA guidance in regard to $PM_{2.5}$. Whether the new regulations will result in a further narrowing of the opportunity to use prescribed fire is not yet known.

What is apparent, however, is the enormous dilemma posed by the amount of fuels present, the increasing danger of large wildfires and the need to protect public health from the smoke of both prescribed and wild fire. With around 60 million people living in the West, concentrated mostly in large urban areas and inter-mountain valleys, the amount of smoke associated with a land management regime that depends entirely on fire is not acceptable. In a study that graphically illustrates the dilemma, Leenhouts estimated the presettlement levels of biomass burning in the conterminous 48 states. These estimates were then compared to current levels of burning as well as those levels of burning needed if wildland (non-urban or non-agricultural) ecosystems were returned to their historical fire regimes (Leenhouts 1998). Air pollution from biomass burning today, he estimated, is about one-seventh of what it was in the pre-industrial era, and a return to ecologically based fire regimes would result in a four-fold increase in $PM_{2.5}$ emissions. Such an increase, in light of the already worrisome levels of pollution from fossil fuel burning and other modern industrial activities, would clearly be unacceptable.

On the other hand, leaving Western forests untreated is not a workable solution for maintaining air quality either. The untreated forests will burn at some point. If they burn in their current fuel condition during the hot, dry season when most wildfires of any consequence occur, the pollution emissions will not only be enormous, they also will be concentrated into a few days or weeks during and after the event. The high smoke concentrations likely in some areas will pose serious health hazards, particularly to children, the elderly and people with respiratory disease.

The need to return Western forests to a more fire-tolerant condition seems, therefore, to hinge on avoiding such enormous impacts on air quality by mechanically removing excess fuel from much of the affected areas. Because the areas involved are huge and often remote, such an effort poses an enormous challenge. Because much of the material involved is economically marginal or sub-marginal, the costs will be enormous as well. The health of 60 million Westerners, however, may dictate that mechanical removal be considered as solutions for forest health are debated.

Water Quality and Streamflow

The National Forest System was established in 1905 in part to protect the nation's watersheds, and watershed protection was a major justification given for the purchase of the Eastern National Forests authorized by the Weeks Act of 1911 (Steen 1976). Today, it can be argued that the most valuable product of many forested areas is the clean water and reasonably stable flow regimes that result from forested watersheds. It also can be demonstrated that hydrophobic (water-repellant) soil conditions can result when lethal wildfires leave soil without vegetative protection. A subsequent rainstorm can do extraordinary damage to the watershed.

In one example, the 1996 Buffalo Creek fire southwest of Denver, Colorado, burned almost 12,000 acres of ponderosa pine forest. Because of the fuel and fire conditions, about two-thirds of those acres burned in a high-severity, lethal crown fire. The rest burned largely as a low-severity ground fire. Shortly after the fire, a thunderstorm dumped 2.5 inches of rain over the burned area, causing erosion losses that averaged 1.4 inches of soil removed across the 8,000-acre area of severe wildfire (Agnew et al. 1997; Cromack et al. 2000). The results downstream were catastrophic and continue today. At the Strontia Springs Reservoir operated by Denver Water, it was estimated that more sediment (200,000 cubic yards) was deposited in the first flush of Buffalo Creek flooding than had been deposited in the prior 13 years of the reservoir's life (Weir 1998). The cost of lost power generation revenues during the time when debris prevented turbine operation, of cleaning the wood debris out of the reservoir and of restoring drinking water quality and electrical generation service was estimated at nearly $1 million in the first clean-up, and the work continues today with each subsequent runoff event. Another continuing cost is created by the ongoing turbidity in the water, which raises treatment costs as well as the cost of sludge disposal created by the clean-up process.

In July 1998, a summer rainstorm in the Buffalo Creek watershed washed an estimated 50,000 cubic yards of new sediment into the reservoir. The utility estimates that sedimentation from continued runoff and movement of stream channel deposits above Strontia Springs could be in the range of 150,000 tons per year and could continue for many years, if not decades. Removing that sediment costs between $6 and $7 a yard. One storm alone created a liability of almost $350,000 for the water customers in Denver, and the annual figure could average $1 million or more.

The increased severity of today's wildfires can alter the amount of nutrients, sediment and organic debris delivered to streams and can increase runoff because of less water absorption by the soil and lower vegetative uptake of soil water (Wissmar et al. 1993; MacDonald et al. 2000). However, past forest management practices—particularly those practices tied to forest road construction, skid trials, landings, and clearcut harvests—have been linked to increased rates of soil erosion, water pollution and watershed damage as well. Evidence for the cause of watershed quality problems is not clearly tilted in one direction. Forest management may be essential for protecting water quality and watershed conditions, but, if done improperly, it also can be linked to undesirable effects. The stage is set for controversy over whether, and how, to treat forests for positive watershed benefits. That controversy finds its most fertile ground in dealing with the national forests that make up virtually all of the upper-elevation headwaters of the West.

Roads

Forest treatment projects to reduce wildfire hazards in Western forests are unlikely to raise serious water quality questions except where new forest roads are involved. The "thinning from below" techniques that would characterize most of the treatments in the ponderosa pine and mixed conifer forests will seldom expose the soil to additional erosion damage. Coupled with the application of "best management practices" (BMPs) designed to protect riparian areas and assure the proper installation and maintenance of landings, skid trails and stream crossings, these projects should have little, if any, effect on water quality or flow dynamics.

Roads are another issue. Opposition to installing new forest roads in roadless areas is extremely strong among environmental organizations, and any forest treatment project that involves new roads is certain to run into stiff controversy. The impact of roads on forest ecosystem integrity is serious enough that road density (miles of road per square mile of forest) was used as a specific proxy for forestland integrity in the Interior Columbia Basin Ecosystem Management Project (Quigley et al. 1996).

Many early logging roads throughout the West were poorly constructed. Many were built directly up stream bottoms, doing maximum damage to stream integrity. Others were built by cut-and-fill methods along steep hillsides to access lands with road grades that could accommodate the equipment involved. Many of those roads subsequently collapsed, and road failures have been identified as one of the biggest causes of erosion and stream damage associated with timber harvesting activities (Oliver et al. 1994). Cutting into steep hillsides meant intercepting sub-surface water transport zones, which, in some wet periods, could result in converting sub-surface flow to surface flow that concentrated in borrow pits, culverts and, ultimately, the stream itself. Intercepting sub-surface flow changes the hydrology of the watershed, increasing peak flows and reducing groundwater recharge. Under extreme conditions, these roads contributed to major hillslope failure, leading to mass soil movement into stream channels.

Another issue with roads on the National Forest System has been the fact that many of them were built to serve timber harvest needs because timber receipts often were the source of financing road building. Once built, however, these roads have served a variety of purposes, including recreation, forest management and protection access. But whether they are located and maintained to best serve these long-term purposes has been controversial.

So the question from a watershed integrity point of view seems to be: Can modern engineering techniques and equipment build access roads for forest treatments in a way that creates less watershed and water quality damage than will be experienced when untreated areas burn? This will be a vexing and controversial question in many areas because the trade-offs are not always clear, and the result in either case may be deterioration in water quality. What people often want, however—a nice stable watershed with pristine, untouched forests in the headwaters—may not be a realistic future for most of these lands, given their current condition and the hazards that they face.

Water Partitioning

A less controversial and less well-documented issue than roadbuilding is altering water partitioning on these watersheds. As rain and snow fall on a forested watershed, the forest vegetation affects how water is partitioned: the amount that soaks into the soil and ultimately feeds subsurface groundwater supplies, the amount that runs off the surface to affect streamflow, and the amount that evaporates back into the atmosphere to affect cloud formation and subsequent precipitation in adjoining regions.

Many Western watersheds, particularly those with high-elevation headwaters, depend heavily upon snowmelt for their summer streamflow. That fact has been the basis for a Snow Survey and Watershed Forecasting program that has been active since the 1940s in the Soil Conservation Service (now Natural Resources Conservation Service), in cooperation with the U.S. Weather Bureau and the state water resource agencies in the West.[13] Because many of the mountain snowpack monitoring sites were on National Forest lands, the Forest Service has been a long-time cooperator in the program, and Forest Service research has been extensive on the effects of different forest management regimes on snow accumulation, melting and runoff. Scientists have documented the effect of changing forest canopies through different forest harvest regimes. One British Columbia experiment showed that clearcut harvests can result in as much as 42 percent more snow water on the ground (Toews and Gluns 1986; Troendle and Kaufmann 1987).

Less research is available about how past fire suppression efforts have altered the forest canopy and the subsequent effect on snow accumulation, snowmelt and water partitioning. What is known is that canopy densities have increased significantly as young trees crowd around the large, widely spaced trees common in the West's stands of ponderosa pine, larch and Douglas-fir. Where canopy density increases, more snow is intercepted before it reaches the ground. Much of that snow (estimates are as high as 50 percent in arid, windy, alpine conditions) evaporates directly back to the atmosphere in a process called sublimation, where water changes from ice to vapor without going through a liquid phase (Schmidt 1991). Thus, to the extent that more snow is intercepted, less reaches the ground to ultimately contribute to surface or sub-surface flow.

In addition, as trees and other vegetation on the site increase, they demand a larger share of soil water, leaving less available to soak below the root zone and recharge groundwater supplies. To the extent this occurs, late-summer streamflow, which is largely provided by sub-surface flows, may be reduced. Year-around streams may then turn into intermittent streams, affecting both watershed flows and stream ecology.

The implications of this situation, if substantiated for particular watersheds or forest conditions, would be that restoring a more-historical forest structure through thinning from below and removing excess biomass would likely increase subsurface water supplies, recharge groundwater and improve dry-season streamflows. For many watersheds, changes such as these are essential to achieve sustainability for streams and their aquatic systems.

Greenhouse Gas Emissions

Forests have been identified as a major contributor to stabilizing the levels of atmospheric carbon dioxide, a major greenhouse gas (GHG) (Watson et al. 2000). By using carbon dioxide in the process of photosynthesis and converting much of it into stable wood, trees have the effect of storing carbon in a fixed state for many years (Sampson 1992b). Even beyond the life of the tree itself, the wood may remain intact as part of a piece of furniture, a book, a house, or other structure (Sampson and Hair 1992, 1996; Brown et al. 1996; Sedjo et al. 1998). National and international programs directed at mitigating climate change have included strategies to expand forest area, improve forest growth, extend long-term use of forest products and generally take advantage of the positive effects good forest management can have on the global climate (Clinton and Gore 1993).

But forests can also become a source of GHG buildups (Harmon et al. 1990). It has been estimated that up to 20 percent of recent global increases in atmospheric carbon dioxide is a result of the clearing and burning of tropical rainforests (Dixon et al. 1994). Similarly, it can be demonstrated that the forests of the Inland West are, in many places, carrying levels of biomass that are significantly higher than the historical range and are increasingly unstable as a result (Sampson 1997). When those forests burn in wildfires, the carbon releases will be significant. In the 1994 wildfires on the Boise National Forest, for example, it was estimated that more than 2.5 million tons of carbon were emitted as a result of the fuel consumed as 119,400 acres burned (Neuenschwander and Sampson 2000).

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[13] There is a full literature documenting the snow survey program. Current products, in the form of monthly snow and watershed conditions in the winter and runoff conditions in the major watersheds of the West, can be found on the World Wide Web at *http://www.nwrfc.noaa.gov*

It was estimated that for the year 2000 (as of Sept. 27, 2000), the 11 Western states had experienced some 4.95 million acres of wildland fire (NIFC 2000). We estimated that 15 percent of the area burned had grass cover, 23 percent was shrub land, 21 percent was open forest and 41 percent was dense forest.[14] Based on those assumptions, the estimated emissions of carbon monoxide, carbon dioxide and methane from the fires that year would be in the range of 73 million metric tons of carbon equivalent.

That is not the end of the carbon cycle story, however. In those places where the forest burned at low severity, soils remained undamaged, and forest regrowth may begin again fairly rapidly. There, the new forest will begin to recapture carbon dioxide, and the result in a few years may be a forest where the amount of new carbon captured in the wood is similar to that which was emitted in the wildfire. In terms of global emissions, the site will be back in reasonable balance (Keane et al. 1997).

At the other end of the spectrum, however, are those areas that experienced a high-severity fire that killed trees and altered soils. These soils suffered severe loss of nutrients and organic matter and altered structure, and they may have become coarser in texture and may have lower future water-holding capacity (Cromack et al. 2000; Giovannini 1994). Under these conditions, the forest is likely to take decades, if not centuries, to become re-established and begin to recapture the lost carbon. Where this is the case, the land becomes a negative contributor in terms of the global dynamic: Having lost the carbon stock it was storing before the fire, it also suffers a diminution of its previous capacity to replace that carbon. In the 1994 Boise National Forest fires cited above, it was estimated that 25 percent of the burned area suffered high-intensity wildfire conditions (Neuenschwander and Sampson 2000). While high-intensity fire is not always correlated with high-severity soil impacts, the two are closely linked.

The effect of these wildfires in terms of greenhouse gas emissions is, therefore, doubled. Stored carbon is lost, and future carbon sequestration potential may be reduced. This is a political problem for a nation with an official goal of reducing carbon emissions and increasing carbon sinks in its forests.

Forest treatments before the fires occurred could have reduced excess fuels but would not have eliminated the lighting-caused ignitions, nor would they have altered the topography. However, they would have altered the intensity of the fire event. If an effective thinning-from-below had been conducted in the area prior to the ignitions, the resulting wildfire would have almost certainly been largely a ground fire, burning through the ponderosa pine forest with minimal damage to mature trees. This speculation is supportable on the basis that treated areas within these fires were documented to act in exactly that manner (see the Cottonwood Creek example in Wildfire Costs below).

A pre-wildfire forest health treatment alters wildfire intensity and reduces forest mortality and carbon emissions. However, the forest treatment itself affects the carbon dynamics of a site. The effect on greenhouse gas emissions of harvesting biomass depends on both harvest methods and post-harvest use of the wood.

Harvest methods that cause minimal soil disturbance result in less soil carbon loss during the post-harvest recovery period. Treatment that does not burn slash or ground debris following thinning may have fewer GHG emissions directly connected with the treatment. However, that advantage could be more than offset if a stand were subject to a wildfire that then burned more of the available fuels or killed the overstory trees because it struck in drier or windier weather than would have been tolerated under prescribed fire conditions.

Biomass harvested in forest treatment tends to be used for short-term purposes. Wood used for structural purposes keeps carbon in storage longer than does wood used for pulp and paper, but how forest treatment wood is used is largely a function of the size and quality of the material removed. Thinnings tend to be smaller material, so a higher percentage is likely to go into end uses that are more short-term in nature.[15]

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[14] These estimates were generated state-by-state, but there were no on-the-ground estimates from individual fire reports available. We used average biomass consumption estimates of 5 tons per acre for grass, 10 tons per acre for shrubs, 20 tons per acre for open forest and 50 tons per acre for dense forest. The numbers could change after the final fire reports are released, if individual fire reports are comprehensive.

[15] Row and Phelps suggest that for the total 1986 U.S. timber harvest, about 33 percent of the harvested material will be either still in use, residing in landfills or dumps, or burned to replace fossil energy by 100 years after harvest (Row and Phelps 1996). Most of the wood-to-energy captured by the Row and Phelps model represents the burning of mill and factory wastes in co-generation plants at the site. Cogeneration has become a common practice in the industry, spurred by air and water pollution regulations that stopped the former practices of open burning and dumping of waste materials. It is also possible to direct all of the excess biomass (not just mill wastes) into energy production, as will be discussed later, and in that case, the biomass can be assumed to replace fossil fuel sources. The substitution leaves fossil fuels in the ground, replacing a net transfer from fossil sources to the atmosphere with a recycling source that represents a closed loop between the atmosphere and the forest. The carbon involved thus becomes credited as an "offset" of fossil emissions.

The amount of biomass involved in forest treatment can be considerable, as will be discussed below. For example, Jolley estimated that 31 dry tons of material per acre were removed in mechanical thinning on a mixed conifer stand in northern California (Jolley 1995). Of that, 30 percent went into saw timber, 50 percent into pulp chips and 20 percent into boiler fuel to produce electricity. That amounts to about 15 tons of carbon per acre, of which about one-third to one-half could be estimated to go into long-term storage (in the form of wood products or building materials) or be used to offset fossil fuel consumption. These numbers suggest that a forest health program based on thinning alone could be credited with five-to-seven tons of carbon per acre of long-term storage and offset and about eight-to-10 tons of carbon per acre of near-term emissions.

Untreated, a stand of that type might emit up to 10-to-20 tons of carbon per acre in a high-intensity wildfire and continue to emit carbon for several years of post-fire recovery. Because of the high probability that untreated stands will face a fire event in the coming decades, the GHG emission tradeoff favors treatment.

Wildlife Habitat and Biodiversity Conservation

The role of forests as wildlife habitat and conservers of biodiversity is one of the most important public benefits associated with sustainable forest management. Thus, any management action that removes biomass to reduce wildfire damages in Western forests will also need to demonstrate a positive effect on biodiversity if it is to receive public support.

There is ample evidence that changes in the forest structure that accompanied the fire exclusion efforts of the past have suppressed understory plants, converted meadows to pine and fir thickets and eliminated both old-growth and savannah forest structures (Despain et al. 2000; Oliver et al. 1997). The question that arises is whether forest management activities can reverse this trend and begin to restore the patterns of landscape and species diversity that characterized the historical range on these forests. That is a question that must be answered specifically in the context of actual forest situations by people expert in those situations.

In general, however, restoring high-priority portions of these forests would improve both landscape structural diversity and biodiversity. This is achieved through the effect of thinning on releasing forest understory, forest floor species and wildlife habitat. Some management activities that remove biomass might also restore habitat for the species that need savannah and open structures to thrive. What is not clear is whether the potential adverse effects of management activities would offset these benefits.

One such adverse effect would be the impact of roads for treatment access on previously roadless areas. Roads are often cited as one of the major hazards facing wildlife and endangered species. The problem usually is not the roads themselves but the increased human access they allow into remote forest areas. Once constructed, the roads provide access that land management agencies find difficult to control. Increased road density in forest ecosystems can lead to disturbances that affect wildlife reproduction success, increased hunting of game species and increased human contact with species such as grizzly bear that often results in death of the animal.

Road building raises difficult management choices for agencies facing the possible loss of remote forest ecosystems to high-severity wildfire. Treatment may require roads for access, but roads themselves create difficult trade-offs, primarily in the area of people management. Unless roads can first be engineered for minimal environmental impact and then be closed or controlled to minimize human access, the disadvantages of road building are significant. Nevertheless, the disadvantages of leaving stands untreated may be equally or even more troublesome. For example, in their current dense stands, problem forests lack the savannah and open structures some species need. And if they burn, these forests are likely to convert to large-area uniformity that is not beneficial for biological diversity.

Designing Forest Health Treatments

In developing silvicultural standards to create more fire-tolerant stands, O’Hara and Keyes used fire behavior models to determine critical factors and tolerance levels (O’Hara and Keyes 1995). The critical factors they propose are minimum crown base heights (CBH) and minimum crown bulk densities (CBD). In using these critical factors to evaluate the five major forest types that dominate the forest landscape in the Northern Rocky Mountain region, they found little difference in the critical minimum crown base height among forest types. They found that critical heights ranged from 7 to 8.7 feet, meaning that less than 10 percent of the trees had crown bases below that height. Using the work of O’Hara and Keyes, it seems reasonable to propose that, in a forest where the bottom crown height is somewhere in the 8-to-10-foot range, a ground fire is unlikely to crown out and become stand-replacing unless localized stocks of fuel or fuel ladders provide the fire with the energy or access to reach crowns.

Once a fire crowns out, O'Hara and Keyes found the bulk density of the crown to be the most important factor in determining whether the fire stayed there or dropped back to the ground. Critical bulk densities were found to range from 0.02 lbs/ft^3 to 0.036 lbs/ft^3 among the different forest types. While most forest stands outgrow the critical density minimums by age 70, all cover types were found to go through higher-risk densities at younger ages.

Improving fire-tolerance through silvicultural methods generally requires thinning the stands from below and pruning the lower (often dead) limbs. O'Hara and Keyes' results indicate that effective treatment would leave fewer than 10 percent of the remaining trees with crown base heights below 9 feet. Effective treatment often can be accomplished with mechanical biomass harvesting operations, which are becoming more common. Mechanical harvesting removes small- and medium-sized trees, and it offers the added benefit that the movements of the mechanical harvester through the treatment area breaks off many dead lower branches of the "leave-alone" trees as it works. This can effectively achieve the pruning effect without added work or expense. Treatment would become significantly more expensive if hand pruning would be required.

University of Idaho forest ecologist Leon Neuenschwander took another approach to testing the effects of forest treatment on fire tolerance (Neuenschwander and Sampson 2000). He used plot data from several ponderosa pine stands on Douglas-fir sites in the Boise National Forest in the FIRESUM model, which produces estimates of fire behavior and tree mortality under different weather conditions (Keane et al. 1989). The plot data selected represent the three common untreated conditions: low density with a basal area of 250 ft^2/acre, medium density with a basal area of 150 ft^2/acre and high density with a basal area of 80 ft^2/acre. Also he used plot data taken from two of the commonly used silivicultural thinning treatments: thinned to 100 trees/acre—basal area 175 ft^2/acre and thinned to 50 trees/acre—basal area 144 ft^2/acre.

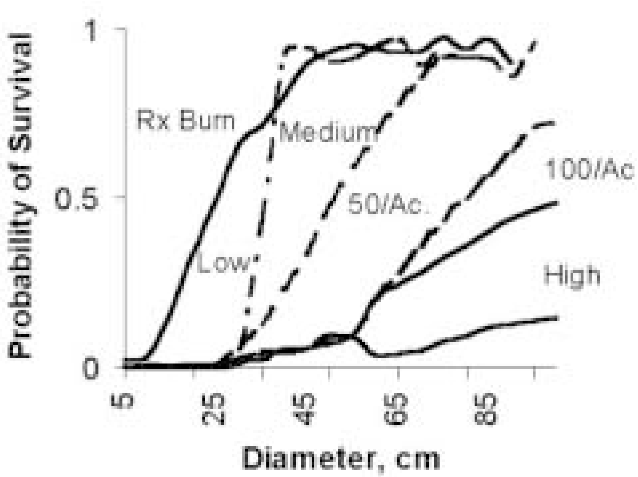

Figure 1.7 Average probability of survival by tree size (DBH), for five selected ponderosa pine stands under a range of wildfire regimes, compared to a prescribed fire (Rx Burn).

Fire weather variables were taken from the long-term records of a nearby weather station, and values between the 25th and 75th percentile of these data were used to calculate probable fire weather conditions.[16]

Neuenschwander's modeling showed that, under virtually all of the study's weather conditions, most wildfires could be predicted to kill the largest trees in the plots, even after the stands had been thinned. The results, shown in Figure 1.7, indicate that trees up to 50 centimeters (about 20 inches) in diameter breast height (dbh) have a very low probability of survival in most fire events. Untreated stands of low and medium density showed better survival probabilities than treated stands, probably because of the slash left behind in the thinning process. The best survival was achieved with the prescribed fire treatment, where trees larger than 25 centimeters (10 inches) in diameter had better than a 50 percent chance of surviving the range of fire conditions.

The conclusions to be drawn from both of these approaches are fairly similar. Most wildfires will turn into lethal crown fires in places where ponderosa pines (even very large ones) are surrounded by smaller trees and where surface fuels can provide a ladder from the ground into the canopy. Removing the ladder fuels in a thin-from-below strategy can reduce this risk, but leaving large amounts of dry untreated fuels on the ground may increase both fire risk and intensity. To assure that a thinning operation reduces fire risk for the remaining forest, the operation must either remove ladder fuels completely through biomass harvesting or carefully burn the fuels under moist conditions following mechanical treatment. Once restoration treatment is complete, future maintenance may depend more com-

[16] A random number generator chose 2,000 sets of wildfire weather data and 500 sets of prescribed fire data from these weather ranges. Five wildfire scenarios (ranging from one-to-seven fire events over 100 years) and one prescribed fire scenario (prescribed fire every 16 years) were then tested in 2,500 FIRESUM runs. An additional fire scenario with a mixture of prescribed and wildfire was tested on plot 3.

pletely on prescribed fire. Restoration may require several treatments, rather than trying to achieve a sustainable condition in the first effort (Agee 1993; Mutch et al. 1993).

The idea of using prescribed fire to return ponderosa pine forests to a more "natural" condition appeals to some ecologists, but the land-use complexity that surrounds many forests today often seems to preclude such an approach.[17] In areas where homes, farms and communities are intermixed throughout the forest, it is difficult to imagine any kind of forest management regime that relies exclusively on prescribed fire. The combination of smoke pollution and hazard to people and property likely means that a different treatment approach—one depending much more heavily on mechanical fuel treatments—will be required.

Two questions arise, however. First is the technical question: Can a "no-fire" treatment adequately mimic the essential ecological functions of fire? These functions include the periodic effect of a fire's heat on the germination or sprouting of some species, the selective impacts on species survival, and nutrient cycling. The answer is far from clear and the question deserves a great deal more research attention. The second question is: What will be done with the surplus biomass once it is removed from the forest in a "no-fire" treatment regime? Much of this material is non-merchantable in the usual wood markets and would not be removed from the forest in a traditional timber harvest. This is a problem facing managers searching for treatment options on all forest types, made even more difficult by the political opposition to removing trees from these forests.

In the last few years, a vigorous public campaign has been launched that seeks to convince Americans that all timber harvests on National Forests should be eliminated. The effort has been successful in that legislation to accomplish the goal has been introduced in Congress. For many of the overcrowded ponderosa pine forests in the West, a prohibition of future tree cutting may effectively eliminate the chance for ecosystem-based, restorative management projects and almost guarantees a lethal wildfire. Whether the eventual fires will permanently damage sites is unknown, but the risks of destructive heat levels rise as fuel levels continue to build. In a frightening paradox, a modern political movement that purports to "save" the public forests may contribute directly to their demise.

[17] The 1995 Federal Wildland Fire Management Policy and Program Review (USDI/USDA 1995) said, "Wildland fire, as a critical natural process, must be reintroduced into the ecosystem," then recognized "where wildland fire cannot be safely reintroduced because of hazardous fuel buildups, some form of pretreatment must be considered, particularly in wildland/urban interface areas." (page iii)

Economic Issues and Forest Health Treatment

The Economics of Harvesting and Hauling Forest Biomass

The basic problem with producing and delivering biomass fuels from forest treatments is tied to the fact that it is more expensive to handle a lot of little pieces in the woods than it is to handle fewer and larger ones. Thus, inevitably, a forest treatment that removes small stems will increase costs. Another problem is that the ability to cut, skid and handle small trees is nearly always tied to a field operation that is most economical when it can handle all of the trees in an area at one time. This is not only the least-cost method but also the least likely to cause environmental damage from machinery operations. Thus, a biomass harvest will, in most cases, be part of a larger forest treatment that removes the particular combination of sawlogs, pulpwood and biomass that needs to come out of the stand to restore healthy forest conditions (Lynch et al. 2000). If only the sawlogs and pulpwood are removed from the land at the time of the main operation, the collection and processing of the biomass left behind becomes so inefficient and expensive that the material will generally be burned on-site or ignored.

This can lead to significant technical and political controversy in designing forest health treatments, particularly on the federal forests. Foresters design the treatments to shape the stand for future growth. This means taking out excess trees, favoring certain species in many cases, thinning the stand to a density that allows for healthy tree growth and leaving trees that will maintain the type and quality of stand that is best suited to the site. If the process is done without other constraints, the result is often a harvest of a mixture of sizes and species that can contain enough high-quality logs to pay for the entire operation.

To some people, however, a treatment that includes the removal of large high-value logs looks like an old-fashioned logging operation, not a treatment to improve forest health. The result is criticism that the forest health treatment is simply a "cover" for removing valuable trees. One outcome of the controversy may be politically imposed rules that limit the harvest to trees below a certain size or diameter or that prohibit cutting certain types of trees. While those rules may sound logical in the abstract, they can be-ill suited to the situation on the land in many places. Where rules are in place, foresters are constrained from shaping the stand for best future health results based on their first-hand knowledge of the local ecosystem. Moreover, the resulting timber sale may be so uneconomic that no private contractor will undertake it. It is a very difficult political argument in today's high-controversy atmosphere, but allowing skilled field people to shape each stand for the best combination of environmental and economic results (both now and for the future) often can produce far superior results than will uniform rules. The magic phrase, of course, is "skilled field people," and the general level of trust these days between the environmental community and professional foresters is often very low.

Treatment Costs

Most of the mechanical treatment options for Western forests involve thinning from below, with the objective being a stand that more closely represents the historical stand in terms of species composition, spacing and structural arrangement. Historical replication is not always immediately possible because a treated stand is constrained by the same site possibilities inherent before treatment. It is almost always possible, however, to move a stand toward a better situation, even where fully desirable results may not be achievable.

Some treatments will be designed to get a site ready for a safe prescribed fire that can provide ecological processes such as soil heating, nutrient cycling and species selection. In areas where land use conflicts with prescribed fire or air quality limits its use, the treatment will be designed to reduce fuels, including slash and other fuels created during thinnings, to the point where they will not support an undesirably hot fire in the event of an ignition. In most cases, one goal is to return the forest to a condition where a subsequent wildfire will remain primarily on the ground, burning in a non-lethal manner so that it can be managed or suppressed if other factors dictate such action.

In contrast to traditional timber harvests, which were designed to remove the merchantable material that was profitable under existing market conditions, these treatments are designed to remove all the material needed to achieve the desired forest condition. That goal creates significant economic issues because much of the material that needs to be removed is either low-value or has no local market except where a wood-fired energy industry exists. Development of a disposal strategy is necessary, and while available options may not be profitable, they should be least-cost for the forest manager.

Costs are, however, not comparable across different land ownerships. In one study of timber sale and administration costs (which included surveying, prescription writing, environmental analysis and documentation, appeals and litigation, sale preparation and administration), the costs were estimated to be $52 per thousand board feet (MBF) for the Forest Service, but only $13/MBF for private industry lands (Keegan et al. 1996). Forest health treatment projects may widen that gap somewhat, depending on future policies. If, for example, Forest Service policy required that all trees to be removed must be marked before cutting, the fact that these treatments often remove hundreds of small stems per acre might impose such exorbitant costs that no project would survive. Even where the trees are not marked, the fact that more small stems must be handled will make the costs significantly higher than would have been experienced in a traditional timber harvest. In spite of these factors, Scott (1998) found that any one of the commonly used local forest health treatment approaches could produce net revenues given the timber prices available in 1996.

Creating a least-cost forest health treatment project begins with deciding what material needs to be removed from a forest stand in the restoration process. Often, managers can create a combination of sale units that can achieve both ecosystem restoration goals and maintain financial viability, even in difficult situations (Lynch et al. 2000). Some of the material may be suitable for commercial sawlogs for which local markets generally exist although prices may vary. Much of the material may be suited for pulp and paper production, but markets for wood chips tend to be more uncertain and, in many areas, unavailable. Where markets are available, suitable material can either be sorted and chipped in the woods or at a central mill or wood yard. If wood chips can be sold at a profit, the economics of a specific treatment project will be much more favorable than if the only outlets available for material of this quality is energy production or disposal.

The material that has neither sawlog or pulp chip values, but that needs to be removed to restore fire-tolerant conditions, provides either a huge cost obstacle to the land manager or a significant resource opportunity for the production of biomass energy, depending on the local situation. The amounts available from specific forest projects vary. The case studies in Part III illustrate the opportunities.

Biomass harvesting operations have been documented in connection with the wood-fired power plants operated by Wheelabrator Technologies Inc. near Redding, California. The forests involved are typically located on slopes of less than 30 percent, allowing the removal of trees with rubber-tired feller-bunchers that can remove the thickets crowded around the "leave" trees without damaging them. Past management, including decades of fire exclusion and selective harvest, have converted stands that were once composed primarily of ponderosa pine, sugar pine and Douglas-fir into stands dominated by white fir and incense cedar.

Stands to be treated are selectively marked to remove much of the white fir and incense cedar while leaving the most dominant, healthy, well-formed trees and favoring the pines and Douglas-fir. Small trees that crowd around selected leave trees are removed because they provide ladder fuels that can carry fire into the forest canopy.

No prescribed fire is used following treatment because there is an almost-total absence of activity fuels or ladder fuels remaining in the stand, though in most situations ground fires could be safely used. In cases where these treatments were conducted in a heavily urban-intermix area, it appeared that fire would pose unacceptable risks under virtually any prescription scenario.

In the harvest process, the trees removed are separated for highest economic return. In a "typical" stand, 2,000-to-5,000 board-feet of sawlog production per acre are anticipated, and biomass fuel yield averages 17 bone-dry tons (BDT) per acre. Where a market for pulp chips is available, the economics are significantly improved, and clean pulp chips are produced by portable field chipper-blowers and loaded directly into vans for transport. In an economic analysis of one such operation, Jolley estimated that 31 BDT of material were removed per acre, consisting of 10 BDT sawlogs (3 MBF), 15 BDT of pulp chips, and 6 BDT of hog fuel. Profits created largely by the sawlogs and pulp chips were estimated at $700 per acre (Jolley 1995).

We inspected these stands visually in 1995. After treatment, they were open, well-spaced stands dominated by pines and Douglas-fir. Of the pre-treatment merchantable material (more than 10 inches dbh in this example), from one-half to two-thirds remained as healthy, well-formed leave trees. Most of the suppressed and deformed trees in the understory were removed. Soil disturbance was minimal and restricted primarily to roads, landings and primary skid trails. The majority of the forest floor was intact, and it appeared that a fire, if ignited, would most likely be a cool ground fire that could do little or no damage to the residual stand.

It is also clear that the off-site environmental impacts of the biomass operation are significant. The 50 megawatt (MW) generating plant operated by Wheelabrator at Anderson, California, uses forest-produced biomass, mill waste, agricultural residues and woody wastes from local land development as well as other materials. In addition to the effects on forest health at the treated forest stands, the mill takes in wood wastes that would either be open-burned—creating regional air quality impacts—or land-filled. Rather than adding to the area's air pollution load, the power plant's virtually pollution-free exhausts represent a significant pollution reduction in a region that is currently struggling to achieve air quality goals.[18]

Wildfire Costs

Normally, the potential costs of a wildfire on the land in question are not factored into the economics of forest treatment options. Basically, the treatment program is funded directly out of the land manager's profits or budgets, while the costs of a wildfire are borne elsewhere, either by the public as a whole or by another aspect of the agency's budget. In the event the forest burns in a high-severity wildfire, the resource losses (usually timber values) are borne by the landowner or manager, sometimes being partially offset by a salvage harvest. In addition, any site degradation due to soil or watershed damage accrues to the landowner over time, even though it may not be recognized because it is lost opportunity rather than cash outlay.

Forest fires, particularly large, intense fires, can quickly run up enormous suppression costs. Fire fighting costs for the USDA Forest Service have averaged $216 per acre of fire in the West for the past 15 years (USDA Forest Service 1995). Estimates for the Boise National Forest in 1994 ran to $408 per acre (Table 1.4). Nationally, the total costs of bad fire years like 1994 or 1996 amount to close to $1 billion for the Forest Service alone. Expenditures by other state, local and federal firefighting agencies add to that total. Forest fires are major drains on the public treasury, particularly those fires that are large, intense and dangerous—a description that becomes more common every year as fuel conditions worsen and land use conflicts created by urban-type development in forested areas continue to grow.

Wildfire costs should be factored into the "no-treatment" option on many Western forests because, without treatment, a large, intense wildfire is virtually certain. The date and time of the event is unpredictable, making any attempt at a discounted cost estimate difficult. However, if the event seems reasonably certain within a decade or two, as many scientists argue, those costs are imminent enough

[18] One should note, however, that the near-complete combustion returns virtually all of the carbon in the biomass to the atmosphere because there is no charcoal and only very small amounts of ash produced in the process.

to warrant inclusion in the land management budget (Covington et al. 1994).

The amount of resource value lost depends on the timber stands killed, the market opportunity at the moment and the amount of salvage that can be recovered to offset the loss. It is also necessary to develop an estimate of the growing potential lost in the immediate future, particularly in the case of a mid-aged forest stand that would have produced some of its most rapid growth in the years ahead. Methods developed and used in the Forest Service since the early 1980s calculate least cost plus net value change (LC + NVC) as a means of evaluating the indirect benefits of fire protection.[19]

The spring 1994 prescribed burn of the Cottonwood Creek area on the Boise National Forest can be used as an example for these calculations. In August of 1994, the area was hit by the Star Gulch wildfire, one of the fires in the Idaho City Complex. Trees at the edge were killed as the high-intensity wildfire came in from adjoining untreated areas. Upon hitting the prescribed fire area, however, the fire dropped to the ground, where it burned through the thinned stand with little or no further damage. Table 1.4 gives the economic estimates of the resulting damage to the Cottonwood area from both the treatment and the subsequent wildfire and compares them to the damages suffered across the Idaho City Complex. In the Cottonwood area, every dollar spent on forest health treatment returned an estimated $6.58 in reduced wildfire losses and suppression costs.

Table 1.4 illustrates the implications of experiencing a high-intensity wildfire versus a mixed-intensity event. The 1992 Foothills fire was almost entirely high-intensity, while the 1994 Idaho City Complex was mixed. The estimated resource losses were more than double on the Foothills fire. Suppression costs remained the same because these were the averaged costs for the Boise National Forest over this period of high fire activity (Dether 1996).

There weren't enough trees killed in the Cottonwood area to ignite a political battle over whether to conduct a timber salvage sale. In all the untreated areas surrounding it, though, a sea of dead trees became the battleground over where, how much, and with what methods, salvage should be attempted. Today, the Cottonwood area is a living forest, 100-to-200 years ahead of its surrounding landscape in terms of forest successional growth. By those measures, the treatment of the Cottonwood area represented a political, economic and ecological success for the Forest Service, and it provides an example of how strategic treatment might affect a larger area.

Disposal of Excess Biomass

Estimating the Resource

As described so far in this report, there are forest situations in the West where a century or more of fire suppression has created biomass buildups that are contributing to forest health problems, that pose high risks of supporting a wildfire of such severity that site damage is likely, and that will not go away until they burn unless they are intentionally removed from the site by forest managers. We also have shown that a major obstacle to removal of this biomass, even in roaded, non-controversial forest situations, is the lack of a least-cost, environmentally safe disposal method or, in the best case, a profitable market.

A primary concern for potential investors considering a processing plant for forest biomass is knowing how much resource is available. Estimating the amount of biomass resource available requires taking two phases into consideration: forest restoration and sustainable forest production.

Forest restoration efforts will result in a large amount of biomass as fuel built up over decades through fire suppression is removed. Sustainable forest management on restored sites will most likely produce far less biomass. Even in those areas where prescribed fire is limited or not feasible, re-entry for biomass harvest is likely to be on a 15- or 20-year cycle, and the amounts of biomass available on a sustainable basis are likely to be smaller than the amounts removed in the initial restoration.

The re-entry cycle is a challenge for regional planners, who need to attract a biomass industry for the initial restoration period. They need to understand that biomass plants require a continuing, reliable biomass feedstock supply. These plants will need other sources of biomass once restoration is complete. In Table 1.5, we used currently available forest inventory data for the West to provide some indication of the amount of biomass that might be considered available during the restoration phase. It is based on the latest Forest Service data, which estimated forest volumes in 1996 (Smith 2000).

Several caveats must be observed in this analysis. The data have been published as regional summaries, which are useful in considering total resource supplies but mean little or

[19] In a personal communication, Charles McKetta, University of Idaho forest economist, calls LC+NVC a "very old and straight forward way of evaluating the indirect benefits of protection." He cites Gorte and Gorte (1979), *Application of Economic Techniques to Fire Management* (Gen Tech Ref INT-53), and Hirsh et al. (1981), *The Activity Fuel Appraisal Process: Instructions and Examples* (Gen Tech Ref RM-83), as original sources.

Table 1.4. Comparative net value change and management costs of wildfire versus forest treatment, Boise National Forest, 1994.

Event for Comparison	*Acres*	*NVC or cost*	*Cost/Ac*	*Notes*
Wildfire — High Intensity	1,000			Scenario similar to Foothills wildfire, July-Aug 1992
Resource Loss		$1,400,450	$1,400	
Suppression Cost		$408,000	$408	
Total Losses/Costs	1,000	$1,808,450	$1,808	
Wildfire — Mixed Intensity				Similar to Idaho City Complex, July-Sept 1994
Low Intensity	550	$83,166	$151	
Moderate Intensity	200	$243,424	$1,217	
High Intensity	250	$350,113	$1,400	
Total	1,000	$676,703	$677	*Note: totals are average per acre costs (total acreage / total costs)*
Suppression Cost		$408,000	$408	
Total Losses/Costs		$1,084,703	$1,085	
Cottonwood Prescribed Burn Area				Treated in April 1994
Mosaic Unburned	200	$0		
Low Intensity	800	$11,200		
Total	1,000	$11,200	$11	
Implementation Cost		$12,000	$12	
Total Losses/Costs		$23,200	$23	
Wildfire Impact on Cottonwood Prescribed Burn Area				Treated area was re-burned in Star Gulch wildfire, Aug 1994
Low Intensity	900	$12,609	$14	
Moderate Intensity	60	$73,033	$1,217	
High Intensity	40	$56,018	$1,400	
Total Losses	1,000	$141,660	$142	
Total 1994 Losses/Costs		$164,860	$165	
Benefit: Cost Ratio — Cottonwood Treatment			$6.58	Benefits per dollar of treatment

nothing in terms of any localized situation. Biomass is a low bulk-density, low-value product, so feasible handling may be limited to a range of 25-to-50 miles. Whether an available resource supply of sufficient size exists for any localized development will require study conducted at that level.

The other caveat is that there are no indications of land ownership or land use designation on the Forest Service timber data except that the data come from timberland inventory plots. This means that the information does not include parks, wilderness or other areas where timber harvest is not allowed and does not come from the vast woodlands such as pinyon-juniper where growth rates are less than 20 ft^3 per acre per year. There may be areas included, however, where roadless considerations or other values may preclude consideration of biomass harvests or forest health treatments.

To minimize the likelihood of encountering those obstacles, we have considered only three forest types—ponderosa pine, Douglas-fir, and true firs—from the forest dataset. Then, to account for the fact that much of the volume of Douglas-fir in the Pacific Northwest is on the moist west-side forests of the Cascade and Coastal Range, we eliminated PNW Douglas-fir from our calculations and from the graphic display in Figure 1.8.[20]

[20] Our elimination of PNW Douglas-fir means we fail to count some Douglas-fir forests in Eastern Washington and Oregon where the historical forest included far more seral species like ponderosa pine and larch and where forest health treatments are badly needed today. However, such a consequence is unavoidable at this level of data consideration. The inventory data is shown in Table 1.5. See Part III of the report for consideration of Eastern Oregon conditions.

Table 1.5. Forest inventory data for three species in three Western regions, by size class in inches dbh (USDA Forest Service 2000)

Species/Region	*5-7”*	*7-9”*	*9-11”*	*11-13”*	*13-15”*	*15-17”*	*17-19”*	*19-21”*	*21-29”*	*29”+*	*Total*
						(million cubic feet)					
Ponderosa Pine											
INT	658	1,254	1,736	1,947	2,005	1,718	1,479	1,176	3,042	1,410	16,426
PNW	334	599	868	994	1,026	950	865	826	2,728	2,374	11,564
PSW	120	270	394	529	605	696	711	746	2,546	3,105	9,722
Western	**1,113**	**2,123**	**2,998**	**3,470**	**3,636**	**3,364**	**3,055**	**2,748**	**8,316**	**6,889**	**37,713**
True Firs											
INT	1,993	2,554	2,702	2,619	2,182	1,785	1,378	1,014	1,849	836	18,912
PNW	547	959	1,182	1,378	1,323	1,294	1,276	1,213	3,580	3,580	16,332
PSW	234	399	587	780	809	920	964	923	3,112	4,619	13,346
Western	**2,774**	**3,911**	**4,471**	**4,777**	**4,313**	**3,999**	**3,619**	**3,149**	**8,541**	**9,035**	**48,590**
Douglas-fir											
INT	1,419	2,502	3,243	3,672	3,615	3,336	2,792	2,171	4,612	1,689	29,052
PNW	1,225	2,558	3,621	4,427	4,825	5,011	4,744	4,517	13,741	24,889	69,559
PSW	314	499	670	656	680	748	723	703	2,644	6,261	13,898
Western	**2,957**	**5,559**	**7,534**	**8,755**	**9,120**	**9,096**	**8,260**	**7,391**	**20,996**	**32,840**	**112,509**

It is often stated that decades of timber harvest activities have left Western forests with few or no large trees left. These data, particularly as illustrated in Figure 1.8, tend to disagree. To the extent these forests are lacking in size and age cohorts, it would appear that concern should instead focus on the lack of trees in the 13-inch-to-21-inch range. Two possible explanations seem reasonable for the lack of trees this size in the age-size distribution. One explanation could rest on shortened harvest rotations and the harvest of smaller trees. Another could rest on the fact that overcrowded forest stands tend to stagnate. When the trees reach a certain size, growth slows dramatically as the competition for nutrients and water affect them.[21]

Based on this size distribution, one could argue that forest health treatments on these forests should remove trees under 12 inches in diameter where they are overcrowded or are creating ladder fuel problems around larger trees and that saving trees 12 inches or larger in well-spaced patterns could begin to help rebalance the resource distribution. Such a strategy would limit the amount of sawlogs that would result from the project, making the economics of the operation very difficult. Using these thinning assumptions, we can quickly estimate roughly how much biomass volume regional inventory data indicate are available. To do so, we developed Table 1.6 that includes the species and regions listed in Table 1.5 and adds incense cedar where it exists. Managers seek to remove incense cedar as they try to convert mixed conifer forests back toward a higher proportion of seral species like pine and Douglas fir.

We assumed in Table 1.6 that a higher percentage of the small material would be removed in the treatment process and that treatment would result in an average of around 15 tons per acre of biomass removed. The Intermountain Region would end up with the most acres treated—39 percent of the area covered by the three forest types—and the Pacific Northwest would end up with the least. In total, biomass harvest in the West could run in the range of 40 million tons per year for the 10 years of restoration treatment work. The total would almost certainly drop to somewhere in the range of 15-to-20 million tons per year as more prescribed fire became feasible.

The point here is not to provide an accurate biomass inventory. That should be accomplished on a location-by-location basis where biomass energy production or other

[21] We have inspected stands on the Boise National Forest containing trees ranging in size from 5 inches to 25 inches. After a forest treatment project, we counted rings on the stumps and discovered that all the trees removed were roughly the same age, even though they varied greatly in size. The stand had been established soon after one of the pioneer timber harvests, and the 5 inch trees were 120 years old. They were never going to grow into the next size cohort, and if they had been part of an inventory plot, they might have been measured in the same size range for the past 50 years.

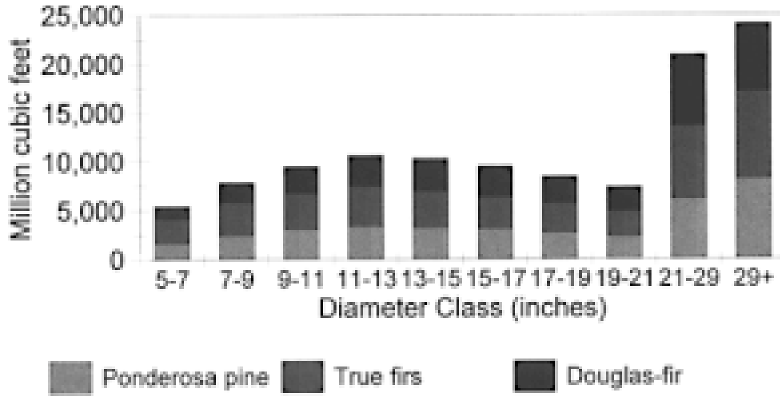

Figure 1.8 Merchantable wood in selected species, as shown in Table 1.5. (Douglas-fir in the PNW is omitted from this figure.)

processing is being contemplated. Rather, the point is to illustrate that a lot of biomass exists in these regions in just a limited number of forest types. This is true even assuming that almost all of the harvesting for 10 years would be limited to small-diameter trees removed in thinning and forest health treatment operations.

The heating value of Western softwoods is about 17 million Btu per bone-dry ton of biomass. Assuming one ton of bone-dry biomass converts to one megawatt-hour (MWH) of electricity under current technologies, that's 40 million MWH of power per year from a biomass resource that otherwise is a liability and hazard to the ecosystems where it exists today.

Table 1.6. Estimated biomass harvests possible in a 10-year accelerated forest-health treatment program for the Western United States, based on forest inventory of small-diameter timber in four target species.

Species	*Intermountain*				*Pacific Northwest*				*Pacific Southwest*			
	1000 ac	*Diameter Class*			*1000 ac*	*Diameter Class*			*1000 ac*	*Diameter Class*		
		5-7"	*7-9"*	*9-11"*		*5-7"*	*7-9"*	*9-11"*		*5-7"*	*7-9"*	*9-11"*
Douglas-fir	17,645	1,419	2,502	3,243	16,912				1,977	314	499	670
Ponderosa pine	14,482	658	1,254	1,736	6,286	334	599	868	7,267	120	270	394
True fir	14,213	1,993	2,554	2,702	4,278	547	959	1,182	2,936	234	399	587
Incense cedar						19	30	33		76	116	134
Saw timber (mcf)*		4,070	6,310	7,681		900	1,588	2,083		744	1,284	1,785
Non-merchantable		2,035	3,155	3,841		450	794	1,042		372	642	893
Total biomass (mcf)		6,105	9,465	11,522		1,350	2,382	3,125		1,116	1,926	2,678
Removal percentage		80%	70%	60%		80%	70%	60%		80%	70%	60%
Total biomass (MBDT)*		78	106	111		17	27	30		14	22	26
Annual removal (10 yrs)		7.81	10.60	11.06		1.73	2.67	3.00		1.43	2.16	2.57
Total acres	46,340				27,476				12,180			
Total annual harvest/region		29.48				7.40				6.16		
Annual harvest (1000 ac)		1,965				493				410		
Percent of area harvested in 10 years		42.41%				17.94%				33.69%		

* mcf = million cubic feet; MBDT = million bone dry tons