# Recent Forest Insect Outbreaks and Fire Risk in Colorado Forests: A Brief Synthesis of Relevant Research

Photo: Jeff Hicke


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## Introduction

Extensive outbreaks of tree-killing insects are occurring in many parts of the West, including Colorado. In combination with recent high-intensity forest fires, these insect outbreaks are raising concerns about the health of our forests and our ability to deal with these issues. The visual impact of a high-severity bark beetle outbreak or fire may give the impression that we are in a crisis situation and that we must take dramatic steps to deal with this "emergency". However, recent scientific research on the ecology of forest disturbances, by scientists in Colorado and elsewhere, leads us to interpret these recent events in a much more nuanced manner.

We believe that the responses to insect outbreaks and fires will not produce beneficial results unless those responses are consistent with the basic ecology of the affected forest ecosystems. Hence, we have written this brief synthesis of the current state of knowledge about forest insects and fires in Colorado to help inform effective management options. Our emphasis is on the ecological aspects of the insect outbreaks now affecting thousands of acres in the state. We do not deal extensively with other dimensions of insect outbreaks and fires, although we acknowledge that aesthetics, economics, wildlife management, recreation, watersheds, and fuels are all important considerations in making decisions about forest policy and management.

This report is organized into two sections. The first section addresses nine key questions about the basic ecology of insect outbreaks in Colorado forests; the second section evaluates six possible treatment options. We do not advocate any particular policy or management treatment, but instead describe the likely ecological effects of each potential option. We also provide a very brief synopsis of each answer or treatment option in italics at the beginning of each section. Our hope is that the information summarized here will aid managers and policy-makers in making decisions about how to deal (or not deal) with different kinds of insect outbreaks occurring in different contexts. As will become clear below, not all forests and not all insects are alike. The authors all have training and research experience in forest ecology or hydrology, both in Colorado and elsewhere.

## Questions about the Basic Ecology of Forest Insects

**Question #1: Which insects are killing trees across large areas in Colorado?**

Summary: *The major insects killing trees in Colorado today include bark beetles (mountain pine beetles, spruce beetles, and piñon ips beetles) and defoliators (notably western spruce budworm). All of these insects are native to Colorado and have co-existed with their host tree species for thousands of years (Figure 1).*

Two major groups of insects have been responsible for killing large numbers of trees over extensive areas under outbreak conditions in Colorado: bark beetles and defoliators (Schmid and Mata 1996). Adult bark beetles bore through a tree trunk and lay eggs within the inner bark. The eggs hatch and the beetle larvae eat the inner bark, killing the tree. After the larvae mature, the new adults fly to new trees, bore through the bark, and continue the cycle. There are several species of bark beetles, each of which feeds on one or several species of trees. For example, the mountain pine beetle feeds on ponderosa, lodgepole, and limber pine; the spruce beetle feeds on Engelmann spruce; and the piñon Ips beetle feeds on piñon pine.

Defoliators are a group of insects having a life cycle very different from the bark beetles. The adult defoliators are tiny moths that lay their eggs in the buds of trees. The eggs hatch into caterpillars that feed on the emerging new leaves in spring and early summer. When numerous, the caterpillars may eliminate essentially all of a tree's annual production of leaves or needles. Small trees, or trees that are stressed by other factors, may die after a few years of defoliation, though usually most of the trees in a stand survive the outbreak of defoliators. The most important defoliator in Colorado forests is the western spruce budworm which feeds on Douglas-fir, white fir, subalpine fir, and spruce. Douglas-fir tussock moth is a less frequent but locally significant defoliator of Douglas-fir, white fir, and spruce. Aspen trees may be defoliated by tent caterpillars and large aspen tortrix.

These insects are usually present in a forest in very low numbers, killing only the occasional weak tree. Such low numbers are referred to as

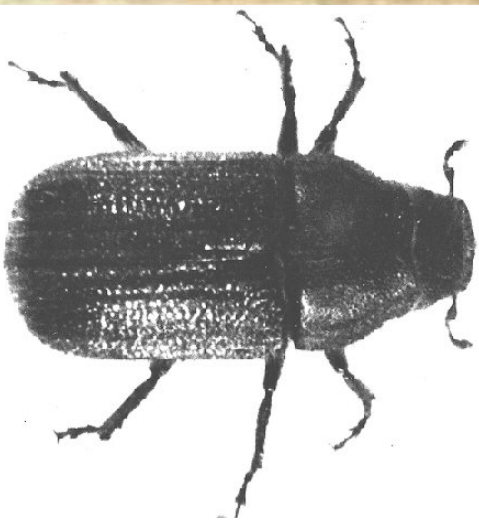

Figure 1. The major insects killing trees in Colorado today include several species of bark beetles (such as the mountain pine beetle above and the spruce beetle below) as well as various species of defoliators. All are native to Colorado and have co-existed with their host tree species for thousands of years. Mountain pine beetle photo from Colorado State Educational Extension Service. Spruce beetle photo from USDA National Agricultural Library.

"endemic" populations. Periodically, however, insect populations grow rapidly and kill large numbers of trees over large areas. This is referred to as an "outbreak" or "epidemic" population. Outbreaks of all of the insect species described above have occurred recently, and have caused extensive mortality events in their respective tree hosts. It is important to note, however, that the trees of Colorado and the Rocky Mountains have coexisted with these native bark beetles and defoliators for thousands of years.

**Question #2: Are the insect outbreaks now occurring in Colorado unprecedented in the ecological history of this region, or are they "natural" events similar to outbreaks that occurred in the past?**

<u>Summary</u>: *There is no evidence to support the idea that current levels of bark beetle or defoliator activity are unnaturally high. Similar outbreaks have occurred in the past (Figure 2)*.

There is no evidence to support the idea that current levels of bark beetle or defoliator activity in Colorado's lodgepole pine and spruce-fir forests are unnaturally high. The outbreaks now taking place in Colorado are similar in intensity and ecological effects to previously documented outbreaks in the Rocky Mountains. For example, mountain pine beetle outbreaks killed millions of lodgepole pine trees over thousands of square miles in the Cascade and Rocky Mountains during the 1960s, 1970s, and early 1980s (Lynch 2006; chapter 4); and a spruce beetle outbreak in the 1940s killed spruce trees over much of the White River Plateau in western Colorado. Historic photos and tree-ring evidence also document extensive insect outbreaks prior to the 20th century (Baker and Veblen 1990, Veblen et al. 1991, Veblen et al. 1994, Swetnam and Lynch 1998, Eisenhart and Veblen 2000, Veblen and Donnegan 2006). Thus, insect outbreaks are a natural occurrence in almost all of the different kinds of forests in Colorado. Outbreaks do not occur very frequently; the time interval between successive outbreaks in any given area is usually measured in decades. Nevertheless, outbreaks can be expected periodically in almost any place in the state where forests are found.

It is true that bark beetle outbreaks are now

Figure 2. The insect outbreaks now occurring in Colorado are similar in extent and severity to outbreaks of the past. For example, spruce beetles killed millions of trees over thousands of acres in the White River National Forest in the late 1940s and early 1950s. The dead trees (above) are still visible. (Photo by T. T. Veblen).

occurring in parts of Colorado where such extensive insect activity had not been seen at any time during the previous hundred years (e.g., in the Fraser Valley). However, in the absence of tree-ring reconstructions or other spatially detailed information on historical mountain pine beetle outbreaks in Colorado, it is not known if similar outbreaks occurred in the same locations or habitats in the past several centuries. Given the naturally long intervals between recurrent bark beetle outbreaks in Rocky Mountain forests, there is nothing unusual about a hundred-year period of low activity followed by an extensive outbreak. It also is true that mountain pine beetles now are killing trees at unusually high elevations (Wayne Shepperd, personal communication). This may be a significant departure from previous outbreaks. However, it is difficult to know if the current insect activity at high elevations is truly unprecedented, given the lack of data on precise spatial patterns of prehistoric outbreaks. The occurrence of outbreaks today at high elevations, where the insects ordinarily are limited by cold temperatures, is not surprising considering the warm temperatures we have experienced during the past decade, as we discuss in the next question.

**Question #3: Why are the insect outbreaks so severe and so widespread at this time?**

Summary: *The ecological factors that control insect populations are complex. Recent bark beetle outbreaks in Colorado probably are a result of four interacting factors: (i) long-term drought, which stresses trees and makes them more vulnerable to insects, (ii) warm summers, which further stress the trees and may accelerate growth of the insects, (iii) warm winters, which enhance survival of insect larvae, and (iv) abundant food (trees) for the insects in Colorado's extensive and often dense forests (Figure 3).*

The factors that control the initiation, spread, and termination of insect outbreaks are complex, and involve a combination of climatic conditions and characteristics of forest stand structure. The relative importance of climate vs. stand structure in any given outbreak is not fully worked out, and in fact may vary from place to place and among the various insect and tree species. Nevertheless, the following is what we know about the interacting influences of drought, temperature, and stand conditions on insect outbreaks.

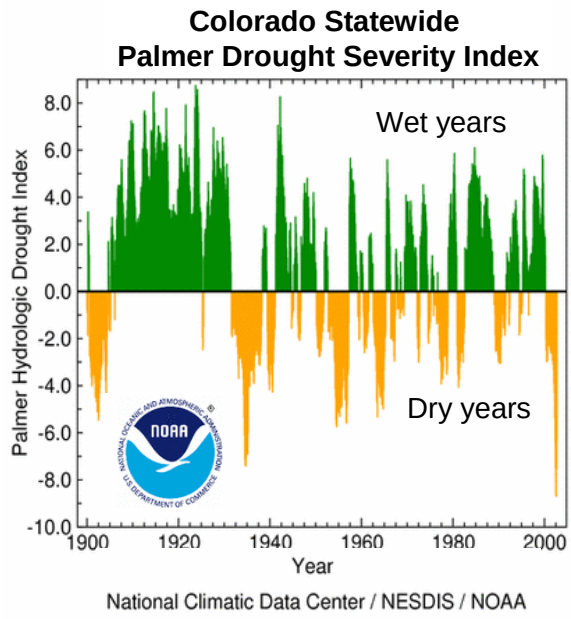


Figure 3. The reason why bark beetle outbreaks are so extensive and severe in Colorado today is because of four interacting ecological factors: (i) long-term drought, as shown above, that stresses trees; (ii) warm summers and (iii) warm winters, which enhance beetle growth and survival; and (iv) abundant food sources (trees) for beetles.

Evidence from observational, laboratory, and modeling studies indicates that climate is a major controlling factor of bark beetle outbreaks (Bentz et al. 1991, Logan et al. 2003, Carroll et al. 2004, Breshears et al. 2005). The initiation of a bark beetle outbreak is often associated with drought. It is thought that the dry conditions stress the trees and make them less able to defend themselves against the beetles (Carroll et al. 2004). For some insects, the end of the drought usually means the end of the outbreak. However, with mountain pine beetles and spruce beetles, once the beetles have killed a large number of trees and produced abundant offspring, their numbers may become so great that they can overwhelm even healthy trees. If this point is reached, continued drought is not so important: the beetle population continues to grow until it is checked either by a prolonged period of bitter cold weather or until they exhaust their food supply. Low temperatures (around –40 degrees F for about a week), especially in late fall or early spring, may kill the beetle larvae in

the trunks of the trees, and thereby terminate the outbreak at any stage in its development.

A warming climate during the last 100 years, particularly in the last few decades, also appears to have played a role in driving recent insect outbreaks. Higher temperatures and a longer frost-free period subject the trees to additional water stress, and may accelerate the growth and development of the beetle larvae. The warming trend of the past few decades (Westerling et al. 2006) may have contributed to the current outbreak of mountain pine beetle in Colorado, as well as recent outbreaks that have occurred outside of Colorado in historically marginal environments for bark beetles, such as at the northern extent of their range in Canada (Carroll et al. 2004) or in high elevations of the northern Rockies (Logan and Powell 2001, Hicke et al. 2006). Furthermore, changing climate conditions are thought to have been responsible for a very severe mortality event in the piñon trees of southern Colorado and adjacent states. Between 2002 and 2004, extensive piñon mortality occurred during a severe drought and an accompanying outbreak of Ips bark beetle (Breshears et al. 2005). Although a more intense drought actually occurred in the 1950s, piñon mortality was far more severe and widespread in 2002 - 2004, apparently because the unusually warm conditions that accompanied the recent drought put additional stress on the trees and allowed more extensive outbreaks of the piñon Ips beetle. Breshears et al. (2005) documented elevated maximum and minimum temperatures at numerous weather stations throughout the Four Corners region during the past decade.

Stand structure also is important in bark beetle outbreaks. The inner bark of very small trees usually is not thick enough to support beetle larvae, and consequently the adult beetles tend to select larger trees to lay their eggs. The minimum tree size for the mountain pine beetle is around four to five inches diameter, but is different for other beetle species (Furniss and Carolin 1977). Thus, stands with large trees are more susceptible to bark beetle outbreaks than are stands with smaller trees. In addition, trees in old or dense stands may be less vigorous and therefore more susceptible to beetles than trees in young or less dense stands, because of competition among trees for limited water and nutrients (Shore and Safranyik 1992). At the landscape scale, if most of the forest is of similar age and has a structure conducive to bark beetle outbreaks, it is likely that outbreaks will be widespread -- if climate conditions are also appropriate. Although fire suppression in the lodgepole pine zone probably reduced opportunities for establishment of young stands since about 1940, young stands have established after timber harvests during this period. The main influence on lodgepole pine age structure in Colorado, however, is widespread burning in the late 1800s that resulted in extensive cohorts of relatively similar age that now are entering a stage that is susceptible to bark beetle outbreaks.

So, why have recent insect outbreaks been so extensive and severe in Colorado? We believe the answer is as follows. The past decade has brought severe drought to many parts of the state (Pielke et al. 2005, Figure 3), accompanied by relatively warm temperatures in both summer and winter (Westerling et al. 2006). The combination of drought and hot summers probably stressed the trees and made them more susceptible to bark beetles; the warm summers may have accelerated the growth and reproduction of some bark beetle species (e.g., spruce beetles and piñon Ips); and the mild winters produced very little mortality of beetle larvae. These climatic conditions probably are the major reason why insect outbreaks have gotten started in many different regions of the state. Once the outbreaks began, the beetles found an abundant food supply (trees) in most of Colorado's forests. Many stands are densely stocked with trees because they have not been disturbed for a very long time by fire, insects, or harvest. All of these factors have combined to create a "perfect storm" of bark beetle outbreaks across much of Colorado.

**Question #4: Are the dense forest stands that we see in Colorado today the unnatural consequence of past fire suppression and lack of timber harvesting?**

<u>Summary</u>: *The answer to this question depends on the type of forest and its geographic location, as explained below. For example, high density in lodgepole pine and spruce-fir forests is not related to fire suppression; it is simply a natural*

*ecological feature of these subalpine forests. It is important to note that not all forests have been affected in the same way by past fire suppression and other human activities (Figure 4).*

Many Colorado forests are very dense, but not all dense forest stands are the unnatural consequence of past fire suppression and lack of timber harvesting. For example, high tree density is a natural condition of most high-elevation forests, including lodgepole pine and spruce-fir. On the other hand, some ponderosa pine forests (but not all) do have unnaturally high tree densities -- higher than would have been seen prior to Euro-American settlement of the region. Thus, it is necessary to distinguish among different forest types in Colorado and elsewhere in the West when considering the effects of past fire suppression and timber harvest (or lack thereof) on current stand density.

**Ponderosa Pine Forests** *Summary*: *Tree densities have increased significantly in dry ponderosa pine forests in parts of Arizona, New Mexico, and southern Colorado, largely as a result of fire suppression and other human activities. Ponderosa pine in northern Colorado has been affected to a lesser extent, because fires were historically less frequent in this region than farther south, and the historical landscape was a mosaic of dense and open stands. The proportion of dense vs. open stands is greater in some areas of the Front Range today than historically, in part because of fire suppression, but also because of recovery from 19th century disturbances and because 20th century climate was generally favorable for tree growth.*

Dry ponderosa pine forests in the Southwest were formerly characterized by frequent, low-intensity surface fires, and it is primarily in these forests where fire suppression has contributed to unnaturally dense stands and increased fire severity today (Covington and Moore 1994, Mast et al. 1999, Moore et al. 1999, Allen et al. 2002). Although fire suppression is part of the reason for very dense stands of ponderosa pine in the Southwest, previous grazing, logging, and climate have also contributed to this change in forest structure (Allen et al. 2002). For example, abundant recruitment of pine seedlings typically occurs during moist climatic periods, and the twentieth century has been characterized by several such periods (Savage et al. 1996, Brown and Wu 2005). In the Colorado Rockies, a

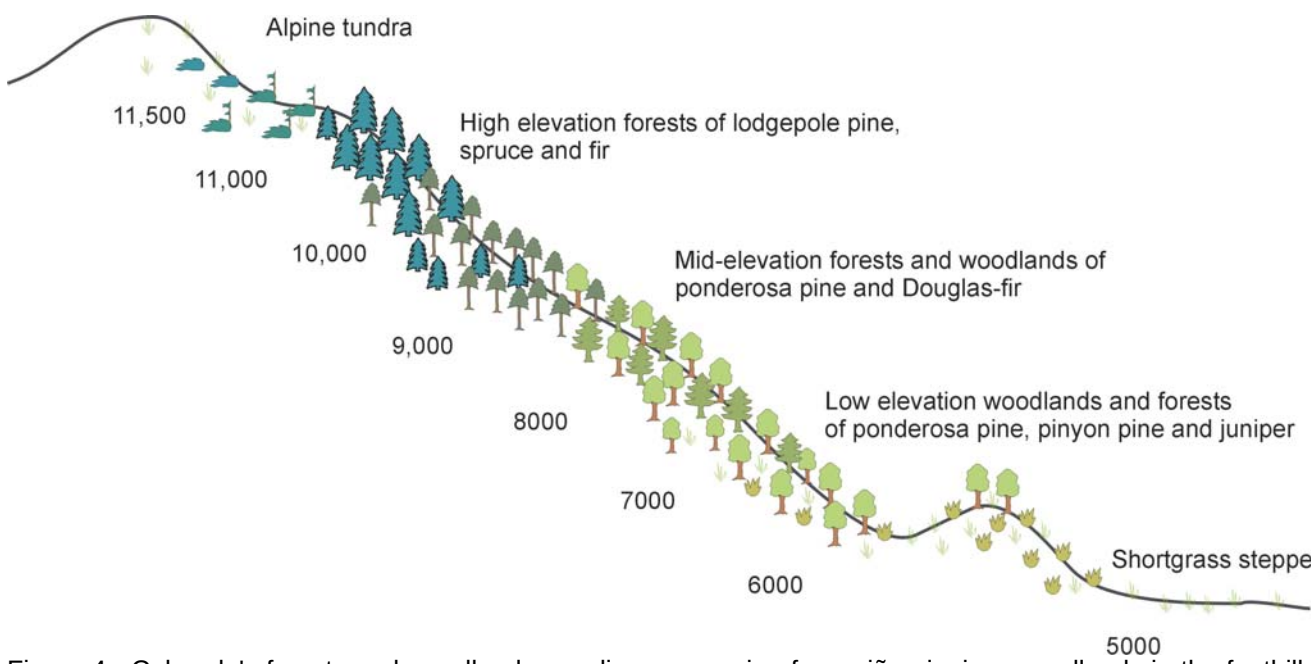


Figure 4. Colorado's forests and woodlands are diverse, ranging from piñon-juniper woodlands in the foothills and basins, to ponderosa pine and Douglas-fir forests at middle elevations, to lodgepole pine and spruce-fir forests at the highest elevations. The natural frequency and effects of forest fires are equally diverse. Tree density in some ponderosa pine forests is greater today than historically because of fire suppression, grazing, and logging during the past century. In contrast, dense stands in high-elevation forests are not related to 20th century fire suppression or land use history; they are simply natural features of these forests where fires have always occurred infrequently. (Figure prepared by L. Huckaby)

model similar to that in the Southwest -- suppression of formerly frequent low-severity fires followed by increased tree density -- applies to some but not all ponderosa pine forests. This "Southwestern ponderosa pine" model appears to be most applicable towards lower elevations and more southerly portions of Colorado.

The Southwestern ponderosa pine model generally does not apply throughout the moister, cooler forests in northern Colorado and at higher elevations, even though ponderosa pine may still dominate (Kaufmann et al. 2006, Baker et al. 2006). For example, in ponderosa pine forests of the Colorado Front Range, tree-ring and other evidence demonstrates that the historical fire regime included both low-severity fire (i.e., surface fires that thin the forests) and high-severity fires (i.e., fires that kill canopy trees and often result in dense regeneration) (Mast et al. 1998, Brown et al. 1999, Kaufmann et al. 2000, Veblen et al. 2000, Huckaby et al. 2001, Ehle and Baker 2003, Sherriff 2004, Kaufmann et al. 2006). In fact, less than 20% of the ponderosa pine zone in the northern Colorado Front Range appears to have been characterized mainly by frequent, low-severity fires. Instead, most of the ponderosa pine zone was characterized by a variable–severity fire regime that included a significant component of high-severity fires (Sherriff, 2004).

The high-severity fires of Front Range ponderosa pine forests tend to occur less frequently than the low-severity fires, and forests naturally grow dense during the long intervals between successive fires. These dense stands are interspersed with more open stands, creating a complex mix of forests. Thus, we conclude that the dense ponderosa pine forests seen in some parts of Colorado's northern Front Range are only partly due to 20$^{th}$ century fire suppression and low rates of timber harvest in recent decades. In contrast to some forests in the Southwest, dense stands of ponderosa pine have always been a component of the Front Range landscape. The proportion of dense vs. more open pine stands has shifted towards more dense stands during the past half-century in many areas, in part because of fire suppression, but also because of climatic conditions conducive to tree growth and natural recovery of forests that were burned or logged in the late 19th century.

**Lodgepole Pine Forests** <u>*Summary*</u>: *Dense lodgepole pine stands are not an artifact of fire suppression. These forests have always burned infrequently (intervals of many decades or centuries between fires) and at high intensity, and these fires are naturally followed by development of a dense young stand. Fire suppression has not significantly altered the natural frequency or ecological effects of fire in most lodgepole pine forests.*

Dense stands historically were the norm in lodgepole pine and other high-elevation forests throughout the Rocky Mountain region (Parker and Parker 1994, Kashian et al. 2005, Schoennagel et al. 2004). In these forests in Colorado, fires occur infrequently (on the order of many decades or a century or more between successive fires in any given stand) and naturally tend to be high-intensity fires, usually crown fires, that kill the majority of the trees (Buechling and Baker 2004, Sibold et al. 2006, Veblen and Donnegan 2006). This type of natural fire behavior contrasts strikingly with the frequent surface fires of dry, low-elevation ponderosa pine forests: rather than thinning forests by killing primarily small, fire-intolerant individuals, the naturally severe fires of high-elevation forests typically kill all of the forest canopy and stimulate regeneration of the stand. Post-fire regeneration of lodgepole pine often results in a dense stand, especially where a large proportion of the trees have serotinous cones. Serotinous cones remain sealed by resin until the heat of a fire melts the resin and releases the seeds; thus, even though the adult trees are killed by the fire, they have stored huge numbers of seeds in their cones and those seeds are released into an optimal seed bed created by the fire.

The effect of fire suppression on the structure of individual stands and on the characteristics of stands across the landscape has been relatively minimal in lodgepole pine and other high-elevation forests in Colorado and throughout the Rocky Mountains (Schoennagel et al. 2004). The remote mountainous areas where these forests grow were generally difficult to access for fire-fighting, especially prior to the 1950s. Furthermore, the length of time that fire has been effectively excluded (~50 to 80 years) is short relative to the natural fire return interval

(measured in centuries). As a consequence, fire exclusion has not significantly lengthened fire intervals in lodgepole pine forests. Note that this is in marked contrast to frequent, low-severity fire regimes such as Southwestern ponderosa pine.

It is true that a large proportion of the lodgepole pine stands in Colorado are more than 100 years old today (e.g. as reflected in stand age data from USDA Forest Service). However, this pulse of tree establishment was mainly due to widespread severe fires during the second half of the 19th century when climate was conducive to fires in the subalpine zone (Sibold and Veblen 2006). Tree-ring data show that similar pulses of establishment of lodgepole pine followed similar episodes of widespread fire in the 17th and 18th centuries across the subalpine zone of northern Colorado (Kulakowski and Veblen 2002, Kulakowski et al. 2003, Sibold et al. 2006). Thus, the tree-ring record of fire and tree establishment in subalpine forests indicates a high degree of variability in fire extent and stand initiation at time scales of 100 years. This variability included periods of extremely rare fires over 100-year periods of climate unfavorable to fire spread, so that long fire-free intervals such as in the 20th century are not outside the historical range of variability for these forests. Thus, age structures similar to the current dominance of the 100+ year old age class are typical of the historical conditions of lodgepole pine forests.

Because of the natural disturbance regime in lodgepole pine forests, characterized by infrequent but periodically large severe fires and insect outbreaks, these high-elevation forests do not exhibit a static or consistent average age class over time. We know that fires before 1900 in this forest type were infrequent but could grow to very large size under very dry weather conditions (Schoennagel et al. 2004, Sibold et al. 2006). It follows that we should expect large fires in lodgepole pine in the future, and that these future large fires should not be viewed as abnormal from an ecological standpoint. The key point about lodgepole pine forests is that they were dense and burned infrequently historically, and they are dense and burn infrequently today. High density in lodgepole pine forests is not related to fire suppression in any way; on the contrary, it is a natural feature of their ecology.

**Spruce-Fir Forests** <u>Summary</u>: *Dense spruce-fir stands are not artifacts of fire suppression either. Spruce-fir forests have always burned infrequently (intervals of centuries between fires) and at high intensity, and these fires are naturally followed by development of a dense young stand. Fire suppression has not significantly altered the natural frequency or ecological effects of fire in most spruce-fir forests.*

As in lodgepole pine forests, dense stands are also normal in spruce-fir forests (Veblen and Donnegan 2006). Prior to the beginning of fire suppression efforts in the 20th century, these forests were primarily shaped by large and severe fires that occurred in a given stand, on average, only once per several hundred years (Kulakowski et al. 2003, Buechling and Baker 2004, Sibold et al. 2006). Natural patterns of post-fire stand development resulted in high tree densities. Since long fire-free periods were normal in these forests prior to fire suppression efforts, it is very unlikely that several decades of fire suppression have fundamentally changed the natural fire regime or have resulted in forest structures that could be considered unnaturally dense. Instead, the dense spruce-fir forests today are very much like they have been in past centuries.

**Question #5: Are recent wildfires in some of Colorado's dense forest stands unusually severe compared to pre-20th century fire severity?**

<u>Summary</u>: *Recent fires have been more severe than historically in some forests, notably dry ponderosa pine forests in parts of Arizona, New Mexico, and southern Colorado. However, recent fires have behaved just as they did historically in most of Colorado's high-elevation forests, such as lodgepole pine and spruce-fir. Large intense fires are the normal fire behavior in these latter kinds of forests, and 20th century fire suppression has not caused them to be unnaturally severe (Figure 5).*

Again we stress the importance of distinguishing among forest types. Recent fires have been more severe, for example, in dry ponderosa pine forests in the Southwest, including some of the forests in southwestern Colorado. However, recent fires clearly are not

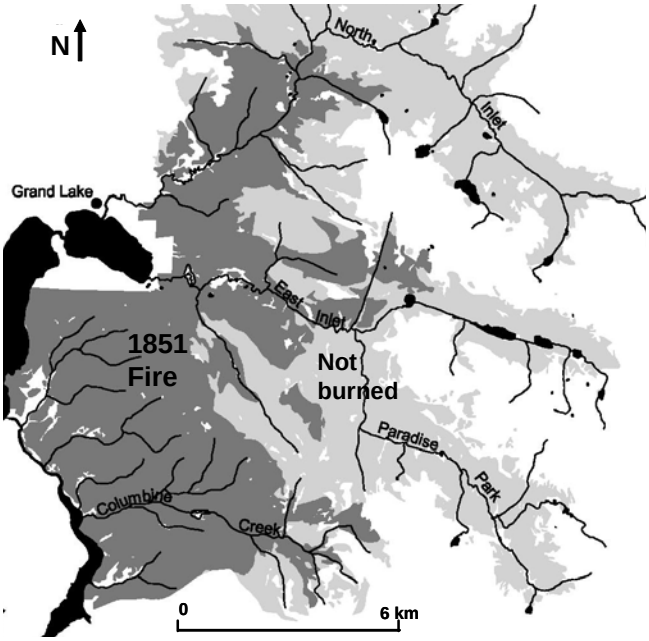


Figure 5. Large, intense forest fires are a natural feature of high-elevation forests in Colorado. For example, much of the country around Grand Lake, Colorado, burned in 1851. Most of the burned area now is covered by 150-year old lodgepole pine forests. (Figure from J. Sibold, 2005 Ph.D. Dissertation, CU Boulder). Some recent fires in ponderosa pine forests have been more severe than would have occurred historically, because of fuels changes associated with fire suppression, grazing, and logging during the past century. However, recent fires in lodgepole pine and spruce-fir forests also have been intense -- but no more intense than occurred historically.

more severe in lodgepole pine or in spruce-fir than fires that occurred in previous centuries. Even in the case of ponderosa pine forests in Colorado, not all areas follow the Southwestern pattern of increased stand densities following the near elimination of fires by grazing and fire suppression in the late 19th and 20th centuries. As noted above, in the Colorado Front Range the ponderosa pine zone was characterized by an historical mixed-severity fire regime in which some areas burned at low severity (as in the Southwest) but other areas, often large, were burned severely and regenerated to dense stands (Mast et al. 1998, Brown et al. 1999, Kaufmann et al. 2000, Veblen et al. 2000, Huckaby et al. 2001, Ehle and Baker 2003, Sherriff 2004, Kaufmann et al. 2006).

**Question #6: Do outbreaks of mountain pine beetles and other forest insects increase the risk of severe wildfires?**

<u>Summary</u>: *Although it is widely believed that insect outbreaks set the stage for severe forest fires, the few scientific studies that support this idea report a very small effect, and other studies have found no relationship between insect outbreaks and subsequent fire activity. Theoretical considerations suggest that bark beetle outbreaks actually may reduce fire risk in some lodgepole pine forests once the dead needles fall from the trees. It is true that severe fires have occurred recently in some forests following insect outbreaks (e.g., in spruce-fir forests of western Colorado). However, these fires burned under very dry weather conditions, and severe fires are the norm for these kinds of forests even without insect activity. Based on current knowledge, the assumed link between insect outbreaks and subsequent forest fire is not well supported, and may in fact be incorrect or so small an effect as to be inconsequential for many or most of the forests in Colorado (Figure 6).*

Our focus here is on active crown fires, i.e., fires that move from tree crown to tree crown under dry windy conditions. Surface fires also are significant; they can affect soils and understory plants, cause major damage to homes and other structures, and can be difficult to control, especially when burning in heavy fuels. However, in this discussion we emphasize crown fires because these often are the most fast-moving fires, they are the fires that typically cause the most damage to homes and other vulnerable structures, and they are almost impossible to control even with modern fire-fighting technology. It is important to realize that active crown fires do not burn only the dead fuels. On the contrary, crown fires are propagated through both live fuels (needles and small twigs) and dead fuels. Tree-killing insects do not really increase the amount of fuels in a forest stand; what they do is shift some of the live fuels into the dead fuel category. Both live and dead fuels can carry fire under very dry weather conditions.

Although more research is needed to confidently predict the effects of insect outbreaks on subsequent fires in Colorado forests, we offer the following interpretation based on theoretical considerations. Whether beetle-caused mortality enhances fire risk and severity compared to an

Figure 6. Lodgepole pine and spruce-fir forests typically burn at high intensity even without previous insect activity. It is widely believed that insect outbreaks set the stage for intense forest fires, but there is little scientific evidence for such a connection. Some recent Colorado fires have burned intensely in lodgepole pine and spruce-fir forests where insect outbreaks had occurred from a few to 50 years previously (e.g., in the Routt and White River National Forests). However, these fires occurred during extremely dry weather conditions, and forests unaffected by bark beetle outbreaks burned in similar fashion. (Photo by W. H. Romme)

unaffected stand very likely depends on time since outbreak. Post-outbreak stand development and associated fire risk may proceed through three stages. (i) Immediately following an outbreak, when trees are dead and dry needles remain on the trees, the chance of a crown fire getting started may be greater than for live trees. However, the dead needles may not significantly change the likelihood of a crown fire spreading from tree to tree, because crown fire spread is controlled not just by dead fuel quantity, but also by live fuel moisture, wind speed, and canopy bulk density (total amount of live and dead fuels in the canopy). This first stage lasts a relatively short time, because the dead needles usually fall within about two years of a tree's death. (ii) Once the needles fall off the dead trees, the likelihood of both crown fire initiation and spread actually may be reduced in comparison to an unaffected stand, since the dead trees create gaps in the canopy and reduce canopy bulk density. It is known that reducing canopy continuity and bulk density through mechanical thinning or harvesting can reduce crown fire risk (Graham et al. 2004), and it is likely that reductions in canopy continuity and bulk density resulting from insect-caused mortality would have a similar effect. (iii) After the dead snags fall, typically one to several decades after the insect outbreak, it is expected that the risk of crown fire initiation and spread may increase once again through two mechanisms. First, the fallen snags may fuel an intense surface fire, with heat and flame lengths that reach into the crowns of the trees. Second, small trees, which generally survived the outbreak and grew more rapidly in the more open conditions resulting from death of canopy trees, create “ladder fuels” that can carry a surface fire into the canopy. In sum, crown fire risk may be elevated for a brief time during and immediately after the peak of the outbreak, while the trees retain their dead needles; then fall to lower levels for the next few decades while the bare snags remain standing; and finally return to pre-outbreak levels some 20 – 50 years after the outbreak when the snags have fallen and a fast-growing understory has created ladder fuels between the heavy surface fuels and the canopy.

We emphasize again that the interpretation just presented is primarily theoretical and requires further study before definitive conclusions can be drawn. We also stress that this analysis focuses on effects of insect-caused mortality within a single stand. The impact on subsequent fire behavior will be different depending on the proportion of the trees killed in the stand. Moreover, it is important to recognize that a large forest landscape is composed of many individual stands. Substantial changes in stand structure and fire behavior within just one or a few stands may have little influence on fire spread and fire severity across the entire landscape.

A few empirical studies have evaluated subsequent fire activity in areas across the West that have been affected by major insect outbreaks, as summarized below. The general conclusion of these studies has been that the outbreak had no effect or only a small effect on subsequent fire occurrence or severity. However, more research of this kind is needed before we can make definitive statements about insects and fire.

**Spruce beetle in subalpine spruce-fir forests***.* It is well established that in spruce-fir forests, extensive fires are highly dependent on infrequent, severe droughts (Buechling and Baker 2004, Sibold and Veblen 2006). Under those extreme drought conditions, dead fuels from insect outbreaks or other causes appear to play only a minor role, if any, in increasing fire risk. For example, following the 1940s spruce beetle outbreak that resulted in dead-standing trees over most of the subalpine zone of White River National Forest of western Colorado, there was no increase in the numbers of fires compared to unaffected subalpine forests (Bebi et al. 2003). Large fires did not occur in these forests until the drought of 1980, when 10,000 acres burned in the Emerald Lake Fire, and in the very severe drought year of 2002 (Pielke et al. 2005) when 31,000 acres burned in the Big Fish and Spring Creek fires. The 2002 fires in western Colorado affected extensive areas of spruce-fir and lodgepole pine forests that were previously affected by outbreaks of spruce beetle and of mountain pine beetle. Yet despite the expectation that these outbreaks (both the 1940s and an ongoing post-1998 outbreak) would have led to an increased risk of severe fires, the forests that were affected by the outbreaks generally did not burn more extensively or more severely than forests that were not affected (Bigler et al. 2005; Kulakowski and Veblen 2006).

**Mountain pine beetle outbreaks in lodgepole pine forests***.* Turner et al. (1999) evaluated the influence of beetle outbreaks that had occurred 5-15 years previously on the behavior of the 1988 Yellowstone fires in lodgepole pine forests. They found that the likelihood of crown fire was increased somewhat where beetle-caused tree mortality had been high (perhaps because the fallen trees created heavy fuel loads), but was reduced where beetle-caused mortality was only moderate (perhaps because the dead trees interrupted the horizontal continuity of the canopy). Lynch (2006; chapter 3) also examined the influence of previous beetle activity on the 1988 Yellowstone fires by testing whether fire was more likely where the beetles had killed trees than in areas unaffected by the beetles. She found that beetle-affected areas did have a higher probability of burning, but that the increase was only about 11% compared with areas unaffected by beetles.

**Spruce budworm defoliation***.* Massive outbreaks of western spruce budworm affected the Douglas-fir forests of the northern Colorado Front Range in the late 1970s and 1980s, but there is no evidence that they resulted in increased fire occurrence. Widespread fires have occurred recently in these forests, but these fires were associated with the extreme drought of 1998-2002. Therefore, if there was any potential increase in fire risk associated with the spruce budworm outbreaks, that potential was not realized until at least 25 years later when weather conditions were conducive to extreme fire behavior even in the absence of insect effects. In Ontario, Canada, Fleming et al. (2002) found a significant increase in probability of fire 3-9 years after an outbreak (perhaps because of increased vertical fuel continuity between fuels on the forest floor and fuels in the canopy), but probability of fire was not continuously elevated after the outbreak. However, in British Columbia, Canada, Lynch (2006; chapter 2) reported a significant decrease in risk of forest fire for nine years following a spruce budworm outbreak.

The upshot of these few studies of insect effects on subsequent fire risk is that the relationships are complex, and that no simple statements can be made about how outbreaks do or do not increase the risk of fire. One reason for the lack of clear-cut patterns is that spruce-fir and lodgepole pine forests naturally burn very infrequently, and only under very dry weather conditions. When the weather conditions are right for a big fire in spruce-fir or lodgepole pine, fire behavior is naturally intense, whether affected by previous insect activity or not. If insect outbreaks do in fact increase the likelihood

of fires getting started or burning intensely through these kinds of forests, the magnitude of increase probably is small and difficult to detect, because fire is so strongly controlled by weather in these forests, and because they naturally burn at high intensity.

**Question #7: Are forests with large amounts of insects and dead trees "unhealthy?"**

Summary: *"Forest health" is an ambiguous concept, one that is not well defined scientifically. The presence of dead or dying trees does not necessarily mean that the forest ecosystem as a whole is not functioning appropriately, even when such trees are numerous. In fact, dead trees and fallen logs perform some important ecological functions in forests, such as providing wildlife habitat and returning nutrients and organic matter to the soil. Nevertheless, dead trees are unattractive and unappealing to many people, and it can be quite painful to lose trees that have special meaning to an individual, such as large pines surrounding one's home (Figure 7).*

Although it may be relatively easy to ascertain whether an individual tree is healthy or not, the concept of "forest health" is very ambiguous. The presence of unhealthy trees does not necessarily imply that the forest as a whole is unhealthy. On the contrary, standing dead trees and fallen logs (coarse wood) play important roles in wildlife habitat, soil development, and nutrient cycling, and are a defining characteristic of old-growth forests. Bark beetle outbreaks rarely kill all of the trees in a stand, because they preferentially attack the larger trees and generally ignore the smaller trees. These smaller trees may be hidden by the red needles of the large killed trees during the peak of the outbreak, such that one often has an impression of total tree mortality. However, once those needles fall it usually becomes apparent that many small and moderate sized trees survived the outbreak. These smaller trees may grow two to four times more rapidly after the outbreak than they did before, because they are no longer competing with the big trees for light, water, and nutrients (Romme et al. 1986). In mixed forests of lodgepole pine and aspen, the aspen may grow more vigorously after beetles kill the dominant pine trees. Even when all of the trees are killed, as in a severe forest fire, the result usually is stand regeneration, as described above for lodgepole pine. Thus, from a purely ecological standpoint, dead and dying trees do not necessarily represent poor "forest health." They may instead reflect a natural process of forest renewal.

Figure 7. "Forest health" is an ambiguous concept. The presence of dead and dying trees does not necessarily mean that the forest ecosystem as a whole is not functioning appropriately. Dead trees and fallen logs perform important ecological functions, such as providing wildlife habitat and returning nutrients and organic matter to the soil. (photo by W. H. Romme)

Nevertheless, dead trees are unattractive and unappealing to many people, especially when those dead trees are abundant, and it can be quite painful to lose trees that have special meaning to an individual, such as large pines surrounding one's home. The change in the appearance of the forest after an insect outbreak also can have negative economic consequences for a community. Over time, the visual impacts are lessened as aspen and small pines grow larger and more abundant, and the gray trunks of the beetle-killed trees gradually fall to the ground. Nevertheless, the visual evidence of an

insect outbreak may persist for a decade or more after the outbreak subsides.

**Question #8: Does a large insect outbreak constitute an "emergency?"**

Summary: *Forests naturally change slowly, almost imperceptibly, over long periods of time. But periodically this slow process of change is punctuated by rapid change via insect outbreak, fire, or other natural disturbance. The sudden death of thousands of trees may be an emergency for people and communities whose amenities, economic activities, and management plans were based on the slowly changing forest that used to occupy the area. From an ecological perspective, however, insect outbreaks are part of the natural rhythm of change in forest ecosystems, and are followed by a gradual re-development of the forest through natural ecological processes. Where aspen was present before the outbreak, the death of the pines may lead to an increase in the aspen component of the forest (Figure 8).*

The normal development of forests involves very slow changes that continue over decades or centuries. A large-scale insect outbreak or forest fire changes a forest rapidly, over a period of a few weeks or years. Such a rapid change often generates great concern about the health and future of the forest and landscape. Is this an emergency? The sudden death of thousands of trees may be an emergency for people and communities that are accustomed to the slowly changing forest that used to occupy the area. Recreational opportunities and values suddenly change, and long-term plans that relied on only slow changes in the forest (such as estimations of annual wood yield) no longer apply. Thus, these may be emergencies from certain standpoints.

From an ecological perspective, we recognize that the forest will slowly re-develop through natural processes. Many montane landscapes in central Colorado are well suited for both conifers (lodgepole pine, spruce, and fir) and aspen, and several of these species commonly occur in the same forest. A century of forest development without any major disturbance typically leads to decreasing abundance of aspen as the conifers increase in dominance. A bark beetle outbreak that kills many of the conifers may be beneficial to the aspen. Old aspen trees will likely grow faster, and new aspen will become established. An increase in aspen will occur only where aspen clones were present before the beetle outbreak. If there was not aspen already present, then composition of the forest will not change; the surviving conifers (mostly smaller individuals and non-susceptible species) will increase their growth rates and replace the large conifer trees that were killed by beetles.

The terms "ecological emergency" and "insect emergency" suggest that insect outbreaks are unforeseen events. However, insect outbreaks, even extensive ones that kill canopy trees over hundreds of thousands of acres, are natural events in forest ecosystems throughout the Rocky Mountains, and have been occurring for thousands of years (e.g., Swetnam and Lynch 1998, Lavoie 2001). The insects have long been natural components of these forest ecosystems. Therefore, from a purely ecological perspective, an insect outbreak generally would not be regarded as an "emergency," but as an infrequent but normal episode of rapid change within an ecosystem that most of the time is changing only slowly.

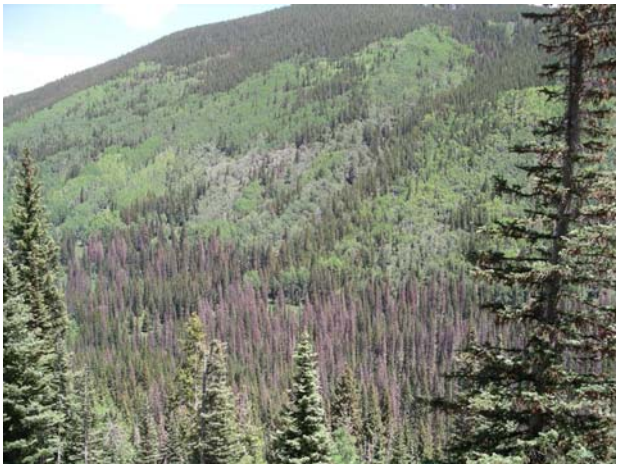

Figure 8. Forests naturally change slowly, almost imperceptibly, over long periods of time. But period-ically this slow process of change is punctuated by rapid change via insect outbreak, fire, or other natural disturbance. From an ecological perspective, insect outbreaks are part of the natural rhythm of change in forest ecosystems, and are followed by a gradual re-development of the forest through natural ecological processes. (photo by Dominik Kulakowski)

**Question #9: How do insect outbreaks affect streamflow and water quality?**

Summary: *An insect outbreak, or any disturbance that reduces the total area of leaf surface in a forest, can potentially increase streamflow by reducing the amount of interception and transpiration. No increase in streamflow is likely when the total annual precipitation is less than 18-20 inches. In areas with more than 18-20 inches of annual precipitation, an increase in streamflow generally will not be detectable unless at least 15-20% of the forest canopy is killed. By themselves, insect outbreaks are unlikely to cause erosion or degrade water quality because they do not disturb the forest soil. Unpaved roads and high-severity wildfires can cause much greater effects on runoff, erosion, and water quality (Figure 9).*

The hydrologic effects of insect infestations vary with the type of forest, the number and size of trees that are killed, and the amount and type of precipitation. The likely effects of a given change in forest density and structure can be predicted with a relatively high degree of confidence because of the long history of plot, process, and watershed scale studies in Colorado and elsewhere (MacDonald and Stednick, 2003). Over the last decade there has been a sharp increase in our understanding of how wildfires, prescribed fires, and thinning affect runoff and erosion rates in Colorado (e.g., Moody and Martin, 2001; Benavides-Solorio and MacDonad, 2005; Kunze and Stednick, 2006).

Removal of all or a part of the forest canopy may potentially increase streamflow via two mechanisms. First, the forest canopy intercepts a portion of incoming precipitation, and this intercepted rain or snow simply evaporates or sublimates back into the atmosphere without ever reaching the soil. A reduction in the forest canopy generally reduces the amount of water that is intercepted and thereby increases net precipitation (but see below for other complicating factors). Second, live trees take up water from the soil and transpire that water into the atmosphere.

Several principles determine whether a particular insect infestation or management action will significantly alter the amount and timing of runoff. First, removing the forest cover from areas that receive less than about 18-20 inches of annual precipitation will have little effect on the amount and timing of runoff as long as there are no significant changes to the infiltration rate of the soil. The primary reason for this lack of change is that any reductions in interception and transpiration are negated by an increase in soil evaporation and transpiration by any remaining vegetation (Bosch and Hewlett, 1982). Once annual precipitation exceeds about 18-20 inches, the reduction in interception and transpiration due to forest harvest or dieback will increase annual runoff, and this increase generally will be proportional to the amount of annual precipitation. Second, at least 15-20% of

Figure 9. An insect outbreak can potentially increase streamflow by reducing the amount of water transpired by trees. However, the increase probably will not be detectable unless total annual precipitation is greater than 18-20 inches and at least 15-20% of the forest canopy is killed. By themselves, insect outbreaks generally do not cause erosion or degrade water quality, because they usually do not disturb the soil. (photo by J. A. Hicke)

the forest canopy has to be killed or removed before there will be any measurable increase in annual runoff. Removing a smaller proportion of the forest cover may still increase the amount of runoff, but this increase probably will not be statistically detectable. Third, the increase in annual runoff due to forest harvest or tree death is roughly proportional to the amount of the forest canopy that is removed or killed. Fourth, the absolute changes in streamflow will be much smaller in dry years than wet years, and become harder to detect as spatial scale increases (MacDonald and Stednick, 2003).

Extrapolation of paired-watershed studies in snow-dominated areas of Colorado and Wyoming indicates that removing the forest canopy from 100% of a watershed will increase mean annual water yields as follows: by a little over 1 inch or about 18% of the mean annual runoff when the mean annual precipitation is 21 inches (Bates and Henry, 1928); by 8 inches or roughly 90% when the mean annual precipitation is 30 inches (Troendle and King, 1985); and by over 12 inches or about 70% when the mean annual precipitation is 34 inches (Troendle et al., 2001). Nearly all of this increase in water yield will come on the rising limb of the snowmelt hydrograph in May-June. Complete removal of the forest canopy can be expected to increase the size of the mean annual peak daily flow by about 40% while having minimal effect on the timing of the annual peak flow (MacDonald and Stednick, 2003).

The hydrologic effects of insect outbreaks are similar in many respects to the effects of forest harvest, but there also are some important differences (MacDonald and Stednick, 2003; Uunila et al., 2006). One difference is that under natural conditions the insect-killed trees remain in place, and this residual canopy will still intercept a portion of the incoming rain and snow, especially while the needles and fine twigs are still in place. This means that the water yield increase due to bug-killed trees will be smaller than the water yield increase due to a comparable amount of forest harvest. A second important difference is that although the insects may kill most or all of the trees within small patches of a few acres, outbreaks never kill all of the trees across a large watershed or landscape; thus, the increases in water yield following insect outbreaks will be smaller than the values listed in the previous paragraph for complete tree harvest (Schmid et al., 1991). Finally, any increase in runoff will decay over time with forest re-growth, and the time to hydrologic recovery may be shorter for an insect outbreak as compared to forest harvest. Studies in Colorado indicate that the time needed for hydrologic recovery after a clearcut varies from about 60 years in the spruce-fir and lodgepole pine zones to around half this time in aspen stands (MacDonald and Stednick, 2003). Insect outbreaks usually kill a portion of the trees, and the surviving trees may grow two to four times faster than they did before the outbreak. Therefore, canopy basal area may return to pre-outbreak levels within a shorter period of time, and this will reduce the potential increase in water yields relative to timber harvest.

Several studies have attempted to evaluate or predict the hydrologic effects of insect outbreaks in Colorado and elsewhere, but most of these studies were been hampered by the lack of a well-controlled design and the available statistical tools. After the 1939-1946 spruce beetle epidemic in the White and Yampa River basins, Love (1955) claimed that annual streamflow in the White River increased by about 2.3 inches or 22%, but this was refuted by Bue et al. (1955). Bethlahmy (1974, 1975) conducted more extensive analyses using different techniques and claimed that the beetle epidemic increased annual water yields by up to 2.0 inches in the White River basin and 2.4 inches in the Yampa River basin, and that the water yield increases were still present after 25 years. A more recent modeling study predicted that water yields would increase in the North Platte River basin by 2.2 inches if 30-50% of the trees were killed by insects (Troendle and Nankervis, 2000). While none of these studies can be considered definitive, the general results are consistent with the principles and values outlined in this section.

In terms of water quality, forested areas typically have very high infiltration rates and rarely generate surface runoff. The death of trees by insects should not compact the soil or cause a loss of the protective litter layer. In the absence of any compaction or ground disturbance, there should be minimal change in soil infiltration rates or the soil moisture storage

capacity. Hence an insect outbreak should not induce overland flow or increase erosion rates, even on steep slopes. On the other hand, the increased duration of high flows due to forest harvest or dieback can increase watershed-scale sediment yields by increasing the stream's sediment transport capacity (Troendle and Olsen, 1994). In practical terms this is of little significance because the sediment yields from forested areas are typically very low (MacDonald and Stednick, 2003). In many forested areas, unpaved roads are a primary source of sediment (Libohova, 2004), and the number, location, and design of forest roads is a key control on whether thinning or harvest activities will affect water quality and aquatic ecosystems (MacDonald and Stednick, 2003; Libohova, 2004). Forest harvest and bug kill can reduce slope stability as a result of the decay in root strength (Sidle et al., 1985), but the increased susceptibility to landslides and debris flows is rarely an issue in Colorado.

Although insect outbreaks usually produce little or no soil erosion, and may have minimal impact on runoff, other disturbances may have significant impacts on soils and runoff. The effects of wild and prescribed fires on runoff and erosion depend primarily on fire severity as well as the timing and cause of peak flows. Low severity fires have minimal effects on runoff and erosion rates because these do not remove the protective litter layer and generally do not kill the larger and more mature trees. In contrast, high severity fires consume all of the protective organic layer, kill most or all of the vegetation, and can induce a water repellent layer at or near the soil surface (Huffman et al., 2001; Benavides-Solorio and MacDonald, 2005; Pietraszek, 2006). In areas with summer convective storms, peak flows and erosion rates can increase by several orders of magnitude after a high-severity fire (Moody and Martin, 2001; Libohova, 2004; Benavides-Solorio and MacDonald, 2005), and the combination of ash and sediment can severely degrade water quality (Moody and Martin, 2001; Kunze and Stednick, 2006). A series of studies in the ponderosa pine zone in the Colorado Front Range suggests that long-term sediment delivery rates from unpaved roads may be similar in magnitude to rates from periodic high-severity fires, while forest thinning has no detectable effect on runoff or erosion rates (MacDonald and Larsen, in press). In snowmelt-dominated areas high-severity fires may have a much smaller effect because soils are not water repellent under wet conditions (MacDonald and Huffman, 2004), and the number and intensity of summer thunderstorms may be lower than in mid-elevation forests. Hence the hydrologic effects of fires in the higher-elevation forests may be more similar to the effects of forest harvest, but there are few data from these higher-elevation sites.

## Potential Treatment Options

Even though the insect outbreaks now occurring in Colorado generally cannot be regarded as *ecological* emergencies, there is no denying that the extensive stands of dead and dying trees do affect the aesthetic and economic attributes of many forests. Moreover, forest fires may cause serious damage to property and may even threaten human lives – whether or not previous insect activity has caused those fires to be more severe than they would be otherwise. Therefore, efforts to reduce the impacts of insects and fires are warranted in many areas. The following sections describe and evaluate the likely effects of a range of treatments that have been used or proposed to ameliorate the effects of insect outbreaks and fires.

**Option #1: Spraying with Insecticide**

Summary: *This can be an effective means of saving high-value trees in localized areas, but is not feasible over large landscapes (Figure 10).*

Spraying trees with an appropriate insecticide can be an effective means of preventing bark beetle attack or reducing defoliator damage. County extension agents and personnel of the Colorado State Forest Service and USDA Forest Service can recommend the best products to use against a particular insect in a particular area.

This may be the best means available for protecting high-value trees around homes, in town parks, or other localized places. However, there are limits to what can be accomplished by spraying insecticides. Annual spraying, or even spraying several times in a single year, is required to prevent attacks by each successive

Figure 10. Spraying with insecticide can be an effective way to preserve high-value trees, such as around a home. However, spraying is not feasible or effective in stopping insect outbreaks over large landscapes. (photo by W. H. Romme)

generation of insects. Spraying is not feasible at the scale of an entire forest landscape because of cost and difficulty of hitting all of the places where insects may be present. In addition, insecticides are not entirely species-specific: a broad-scale spraying of insecticides will kill many harmless and beneficial insects, such as pollinators and butterflies, in addition to the target bark beetles and defoliators. In general, bark beetle preventive sprays have less impact on non-target insects than do insecticide sprays used to control defoliatiors, because the former sprays are targeted to the trunk of the tree whereas the latter sprays need to cover entire tree canopies.

**Option #2: Preventing or controlling outbreaks through forest management**

Summary: *Removing stressed or unhealthy trees, and thinning to prevent crowding and competition among trees, can effectively reduce the risk of an insect outbreak getting started in a forest stand. Forest management is unlikely to prevent all outbreaks, however, because (i) it will never be feasible to intensively manage all of the forests of Colorado, and (ii) drought and warm temperatures are also important causes of outbreaks. Once an outbreak has begun, management generally cannot stop it, because the insects are numerous enough to overcome even healthy trees (Figure 11).*

Because outbreaks may initiate in stressed or unhealthy trees, intensive forest management focused on regular removal of old or unhealthy trees may reduce the likelihood of an insect outbreak getting started in a stand. Thinning may reduce tree-to-tree competition, increase tree vigor, and thus provide an enhanced ability of trees to defend against an attack (Amman and Logan 1998, Schmid and Mata 2005). If periodic harvest removes large trees and maintains a preponderance of small-diameter trees, this too may help prevent the start of a bark beetle outbreak, since bark beetles (but not defoliators) prefer larger trees. Thus, careful forest management, including appropriate timber harvest, may help locally to prevent the onset of an outbreak (Cole et al. 1976).

By itself, however, forest management probably cannot prevent all insect outbreaks -- for two reasons. First, it is unlikely that all stands in Colorado landscapes will be managed intensively enough to remove all of the stressed trees in which an outbreak can get started; in fact, the public values “unmanaged” forests that contain large and old trees. Second, drought and warm temperatures are major causes of bark beetle outbreaks, and forest management by itself cannot entirely overcome these climatic effects. And it is important to recognize that once an extensive bark beetle outbreak has started, it is unlikely that timber management can stop it. Under outbreak conditions, the beetles can overwhelm even the healthiest trees, so selective removal of weak or stressed trees will

Figure 11. Removing stressed or unhealthy trees, and thinning to prevent crowding and competition among trees, can effectively reduce the risk of an insect outbreak getting started in a forest stand. Forest management is unlikely to prevent all outbreaks, however, because it will never be feasible to intensively manage all of the forests of Colorado, and drought and warm temperatures are also important causes of outbreaks. (photo by W. H. Romme)

likely have little impact. Most entomological evidence indicates that once an outbreak has started, there is nothing that can be done to stop it. The outbreak ends when there are no more suitable trees for the beetles, or when unusually cold conditions kill beetle populations. Intensive even-aged management was applied to lodgepole pine forests in the Targhee National Forest, along the western boundary of Yellowstone National Park, from the 1960s through 1980s; yet a mountain pine beetle outbreak that swept through the region in the 1970s and early 1980s appeared to affect the managed Targhee stands as severely as the unmanaged stands in Yellowstone Park (Romme et al. 1986). Similarly, the lodgepole pine forests of British Columbia, Canada, are now being affected by a very extensive and severe mountain pine beetle outbreak, despite a long history of intensive forest management in this province.

Figure 12. Removing dead trees and other fuels can effectively reduce the risk of fire damage at a local scale, e.g., in the immediate vicinity of a home or community. However, the effectiveness of harvest in reducing fire risk to homes and communities over larger areas, e.g., a forest landscape, is less clear. (photo by W. H. Romme)

**Option #3: Harvesting insect-killed trees to reduce wildfire risk**

Summary: *Removing dead trees and other fuels can effectively reduce the risk of fire damage at a local scale, e.g., in the immediate vicinity of a home or community. However, the effectiveness of harvest in reducing fire risk over larger areas, e.g., a forest landscape, is less clear. Conventional timber harvest may do little to reduce fire risk at any scale if it removes primarily large trees, because smaller trees, brush, and dead fuels often are the major carriers of a spreading fire. Harvesting smaller trees and removing small fuels may more effectively reduce fire risk (Figure 12).*

As with the spraying and forest management options, the effectiveness of this option varies with the scale at which it is applied. Removing dead trees – plus other flammable material (including wood roofs and decks, woodpiles and burnable vegetation) from the immediate vicinity of a home or other vulnerable structure -- has been shown to be effective in protecting the structure from wildfire (Cohen 2000). The local characteristics of a home's external materials and adjacent fuels are the primary determinant of home ignitability -- not spatially extensive wildland fuel conditions. For example, the heat released even from intense crown fires will not ignite wooden walls at distances greater than 40 meters away (Cohen 2000). Fuel reduction around a home needs to focus not just on the dead fuels (e.g., the insect-killed trees), but often needs to include some of the live fuels (living trees and shrubs) which also carry fire under severe fire weather conditions. Specific guidelines for reducing fire risk around a home can be found at the Firewise website (Firewise.org) or from extension agents or the Colorado State Forest Service.

Moving up to a broader scale, however, the effectiveness of harvesting insect-killed trees to reduce fire risk across an entire forest landscape is far less certain than the effectiveness of Firewise techniques to protect an individual home. This is especially true in high-elevation forests such as lodgepole pine and spruce-fir. Commercial tree harvest typically involves the removal of large fuels (tree trunks) rather than smaller fuels (branches and needles) due to economic and logistical constraints. These smaller fuels contribute to ignition and spread of fire (e.g., to start a campfire one begins with tinder and kindling). Smaller surface and ladder fuels are important precursors to crown fire initiation (Agee and Skinner 2005). Hence, harvesting tree trunks has little effect on the risk of fire ignition or spread. It is true that if tree harvest also results in reduced canopy bulk density, this may make it more difficult for crown fires to spread. Nevertheless, it is the fine fuels (on the ground or in the canopy) that have the greatest influence on fire initiation and spread,

not the large pieces of wood. Thus, management of fine surface or ladder fuels (which is usually time-consuming and expensive) would have the greatest impact on fire spread and potential high-severity crown fire.

It is important to acknowledge that traditional timber management usually is not designed or intended to reduce crown fire risk, but to produce wood fiber in an economically sustainable manner. Although anything that thins the canopy without greatly increasing the amount of fine fuels can reduce fire spread and intensity during moderate weather conditions (Graham et al. 2004), the most damaging wildfires typically occur under extreme conditions of wind and drought. Most traditional harvesting techniques (including overstory removal and individual tree selection) do not effectively reduce fire severity under extreme fire weather conditions (Stephens and Moghaddas 2005). In the 2002 Hayman fire, pre-fire harvesting where residual fuels (small, non-merchantable material) had not yet been removed, actually contributed to higher severity fire compared to unmodified areas (Omi and Martinson, 2002). If the goal is to reduce fire risk, removal of small trees either via mechanical thinning or prescribed fire (or a combination of both), plus retention of large, old-growth trees, can lower expected fire severity (Stephens and Moghaddas 2005, Agee and Skinner 2005). For example, portions of the 2002 Rodeo-Chediski fire in Arizona experienced lower fire severity where prescribed burning and other management activities during the previous decade had reduced fine fuels and small trees, but had left larger trees intact (Finney et al. 2005). Much of the research on thinning and underburning effects on subsequent wildfire severity has primarily been conducted in low-elevation, dry-forest types: similar effects cannot be assumed in high-elevation forests.

A single thinning treatment cannot maintain lowered wildfire risk over the long-term, because thinning typically stimulates rapid growth of the vegetation that is not taken (Graham et al. 2004). Research shows, for example, that past timber harvesting in ponderosa pine forests is responsible in part for the high densities we witness today (Kaufmann et al. 2000, Gruell et al. 1982, Baker et al. 2006). Although low-intensity prescribed burns reduce fine fuels in the short-term, they also contribute to subsequent dead fuels by killing understory trees, which can result in fuel levels that exceed pre-burn levels within a decade (Agee 2003). Therefore, repeated or staged prescribed fire or mechanical thinning treatments are essential for maintaining lower forest densities; otherwise, a one-time thinning may facilitate dense tree establishment.

Thus, it may be possible to reduce fire intensity and to obtain some control of fire spread patterns across a forest landscape by strategic placement of appropriate timber harvest activities, which may need to focus more on removal of small trees than of commercially valuable sawtimber (Finney 2001, Stratton 2004, Graham et al. 2004). Research is underway to develop specific prescriptions for effective use of vegetation management to alter wildfire intensity and spread at the scale of an entire forest landscape, e.g., at the U.S. Forest Service's fire laboratory in Missoula, MT (Mark Finney, personal communication). Another recently developed tool is the Fuel Treatment Evaluator, a web-based program that uses standard U.S. Forest Service inventory data to identify locations offering the greatest opportunities for hazardous fuel reduction activities (Wayne Shepperd, personal communication). However, this research is still in the early stages, and most has been conducted in only a few forest types (notably drier, lower-elevation forests like ponderosa pine). Thus, it is difficult at this time to make confident predictions of how a specific forest treatment will affect fire behavior under a range of forest types and fire weather conditions.

A major source of uncertainty about the effectiveness of landscape-level fuel treatments in altering fire behavior, is the fact that extreme fire weather can over-ride fuel effects (as seen, for example, in Hayman 2002, Routt National Forest 2002, and Yellowstone 1988). In the Hayman fire, most of the vegetation treatments that had been implemented prior to the fire had very little impact on the severity or direction of the fire during the extreme weather conditions of June 9$^{th}$ and 18$^{th}$, which were the two days when the majority of the area burned (Finney et al. 2003). It should be noted that not all previous vegetation treatments in the Hayman area had been designed to mitigate fire behavior, but were

implemented for other objectives such as timber stand improvement -- further illustrating the point that not all timber harvest activities can be assumed to reduce fire hazard. In the 1988 Yellowstone fires, once fuels reached critical moisture levels, the spatial pattern of burning was largely controlled by weather (wind direction and velocity), rather than by fuels (Minshall et al. 1989, Turner et al. 1994). A study of the 2002 fires in Routt National Forest in Colorado found that previous salvage logging had no detectable influence on fire extent or severity during the extreme drought conditions (Kulakowski and Veblen 2006).

In sum, there is no doubt that Firewise activities in the immediate vicinity of vulnerable structures can increase their survivability in a forest fire (though it must be recognized that the risk of fire damage can never be reduced to zero). However, it is far less certain how effective fuel reduction treatments at greater distances from homes will be in protecting those homes. We also note that timber harvest may be conducted for more purely ecological objectives rather than or in addition to protection of homes. In some types of forests, notably Southwestern ponderosa pine, thinning of overly dense small trees can reduce the risk of stand-replacing wildfire and also contributes to a larger goal of forest restoration (Friederici 2003, Schoennagel et al. 2004). But in other forest types, notably lodgepole pine and spruce-fir, thinning of small trees does not represent restoration of more natural conditions, because these kinds of forests are naturally dense and naturally burn at high intensities; fuel management also has less influence on fire behavior in these ecosystems where climate so strongly controls fire occurrence and severity (Schoennagel et al. 2004). We emphasize again the importance of distinguishing among forest types in evaluating the opportunities and impacts of forest management for wildfire mitigation and ecological restoration.

**Option #4: Salvaging insect-killed trees to improve overall forest health**

Summary: *From a purely ecological standpoint there usually is little or no need to remove insect-killed trees. However, many people do not like to see great numbers of dead trees surrounding their communities or places they like to visit. If the dead trees have a negative impact on aesthetic preferences or local economics, then it may be desirable to remove them (Figure 13).*

As discussed above, “forest health” is an ambiguous concept. From a purely ecological standpoint there usually is little or no need to remove insect-killed trees. In fact, standing snags and fallen logs actually contribute to a number of ecological and aesthetic values in forests, including maintenance of "natural" forest structures and processes, protection of soils and water quality, and preservation of species at risk from the effects of roads, exotic species, and habitat alteration. For example, the three-toed woodpecker feeds on bark beetles in dead and dying trees, and nests most successfully in areas of recent fire or beetle outbreak. Withdrawing all or most of the large dead trees after a fire or insect outbreak will reduce habitat quality for this and other species.

At the same time, there is a widespread public perception that a forest filled with dead or

Figure 13. From a purely ecological standpoint there usually is little or no need to salvage insect-killed trees in the interest of improving forest health. However, if the dead trees have a negative impact on aesthetic preferences or local economics, or if timber production is an important goal in an area, then it may be desirable to remove the dead trees. (photo by J. A. Hicke)

Figure 14. Salvage of insect-killed trees may be a preferred option in some areas because of the economic value of the timber product that can be obtained. In these situations, especially where lodgepole pine trees have been killed by mountain pine beetles, the dead trees must be harvested as soon as possible, because the wood quality deteriorates rapidly after the trees die. (photo by D. Binkley)

dying trees is “unhealthy,” and many people do not like to see great numbers of dead trees surrounding their communities or in places that they like to visit. Whether or not this perception is consistent with what we know about forest ecology, it nevertheless has an impact on aesthetic preferences and local economics. Visitors may choose not to come to a resort surrounded by dead trees; home buyers may avoid locations where the view is one of sick and dying trees. For these and other reasons, efforts to reduce tree mortality (options 1 and 2) and to remove the unsightly results of that mortality (this option), will be the preferred response to insect outbreaks in some locations.

**Option #5: Salvaging insect-killed trees for economically valuable products**

Summary: *Salvage of insect-killed trees may be a preferred option in some areas because of the economic value of the timber product that can be obtained. In these situations, the trees usually must be harvested as soon as possible, because the wood deteriorates rapidly after the trees die (Figure 14)*.

Although salvage of insect-killed trees usually is not necessary for the normal development of the forest, it may be a preferred option in some areas because of the economic value of the timber product that can be obtained. Harvest of large trees for economic reasons can be done in ways that minimize adverse ecological impacts, e.g., by laying out harvest units in spatial patterns that mimic the patterns created by natural disturbances such as fire (e.g., Kohm and Franklin 1997, Friederici 2003, Romme et al. 2003, Perera et al. 2004). If ponderosa pine or lodgepole pine killed by mountain pine beetles are to be salvaged for their timber value, they must be harvested as soon as possible, because the wood deteriorates rapidly after the trees die. However, spruce trees killed by spruce beetles may remain merchantable for decades (Wayne Shepperd, personal communication).

**Option #6 -- No treatment**

Summary: *Natural ecological processes generally lead to the development of new forests after insect outbreaks, so a "no treatment" option can be a form of responsible forest management (Figure 15).*

Natural ecological processes generally lead to the development of new forests after insect outbreaks and fires, without salvage logging or other operations, so post-outbreak or post-fire treatment usually is unnecessary from a purely ecological perspective. Other choices may be made for other reasons, such as including a logging program to salvage economic value from dead trees or to create more desirable visual conditions (options 4 and 5 above).

Figure 15. Natural ecological processes generally lead to the development of new forests after insect outbreaks, as in this lodgepole pine forest 30 years after a bark beetle outbreak killed more than 50% of the canopy. Thus, a "no treatment" option can be a form of responsible forest management. (photo by W. H. Romme)

Nevertheless, a "no treatment" option can be a form of responsible forest management.

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