INSECTS AND ROADLESS FORESTS

A Scientific Review of Causes, Consequences and Management Alternatives

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Suggested citation: Black, S. H., D. Kulakowski, B.R. Noon, and D. DellaSala. 2010. *Insects and Roadless Forests: A Scientific Review of Causes, Consequences and Management Alternatives.* National Center for Conservation Science & Policy, Ashland OR.

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EXECUTIVE SUMMARY

IN RESPONSE TO RECENT BARK BEETLE EPIDEMICS, decision-makers are calling for landscape-level mechanical treatments to prevent the spread of these native insects and to reduce the perceived threat of increased fire risk that is believed to be associated with insect-killed trees. The best available science indicates that such treatments are not likely to reduce forest susceptibility to outbreaks or reduce the risk of fires, especially the risk of fires to communities. Furthermore, such silvicultural treatments could have substantial short- and significant long-term ecological costs when carried out in national forest roadless areas.

The findings in this report regarding the causes and appropriate treatment of insect outbreaks generally apply to both roaded and roadless forest areas. However, this report specifically addresses a recent proposal to exempt national forest roadless areas in Colorado from protections under the 2001 Roadless Area Conservation Rule, in part to address insect outbreaks and perceived fire risk in these forests.

Colorado's roadless areas have recognized ecological and social importance. The 2001 Roadless Area Conservation Rule provides a consistent and scientifically based national standard that safeguards more than 4 million acres of Colorado's inventoried roadless areas from development. The state has submitted a proposal under the Administrative Procedure Act to change the 2001 rule to allow road construction and tree-cutting in inventoried roadless areas in part to address recent outbreaks of bark beetles and related perceived fire risks. In this paper we outline key aspects of and review research related to bark beetle outbreaks, their relationship to fire risk and the efficacy of silvicultural practices to control outbreaks and reduce the risk of fire. We also summarize the importance of roadless areas for wildlife and water quality. The key findings of this paper are presented below.

FINDING 1

Insect outbreaks and fires have been part of the ecology of these forests for millennia.

Large outbreaks of native forest insects and large forest fires have played an important role in the development and maintenance of many forest types. In Colorado, as elsewhere, lodgepole pine and spruce-fir forests are characterized by large, infrequent, high-severity fires and occasional large-scale bark beetle outbreaks.

FINDING 2

Ongoing outbreaks of insects are probably caused primarily by climate.

Climate, specifically drought and high temperature, may be the most important factor behind the current bark beetle epidemic in the western United States. Because silvicultural treatments cannot effectively alleviate the overriding effects of climate, their application in roadless areas is likely to do little to mitigate ongoing or future outbreaks.

FINDING 3

Insect outbreaks in roadless areas are not likely to heighten fire risk in adjacent communities.

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 in lodgepole pine and spruce-fir forests. The best available scientific knowledge, derived from empirical and modeling studies from numerous independent researchers, indicates that the assumed link between insect outbreaks and subsequent forest fire is not well supported, especially for the forest types that are currently most affected by outbreaks. Furthermore, the risk of fires in remote roadless areas is not likely to influence the risk of fires adjacent to communities. Instead, the presence of fuels (both living and dead) near homes and other structures is likely to be most important in determining the risk of fire in these areas.

FINDING 4

Tree-cutting is not likely to control ongoing bark beetle outbreaks or other insect species common to Colorado.

Although individual trees may be saved by spraying with insecticides, efforts aimed at stopping outbreaks are unlikely to be effective once bark beetles reach epidemic levels and cause extensive tree mortality.

FINDING 5

Thinning in roadless areas is not likely to alleviate future large-scale epidemics of bark beetle.

Thinning is often recommended to control outbreaks of bark beetles, but the evidence is mixed as to its effectiveness at the stand level, and it is unlikely to be effective in controlling or alleviating large-scale outbreaks. Experimenting with thinning in roadless areas also can cause short- and long-term ecological effects.

FINDING 6

Tree-cutting in roadless areas will not keep communities safe from wildfire.

If the goal is to protect communities from wildfire, then management would be most effective if it focused in and around communities and involved creating defensible space immediately adjacent to homes. Fire-hazard reduction efforts around communities are more effective, much less expensive and less ecologically damaging than trying to make wholesale modification of forest structure. The limited funds available for community protection would be most effective if used to create defensible space around homes and communities, rather than to build roads and implement fuel treatments in remote roadless areas.

FINDING 7

Building the roads necessary to enter roadless areas affects their ecological values.

The construction of temporary or permanent roads in roadless areas can have substantial short- and long-term ecological costs. The presence and use of roads has been linked to increased wildfire ignitions, increased wildlife mortality and fragmentation of habitat, changes in the physical and chemical environment, diminished water quality, introduction of invasive species and increased likelihood of landslides.

FINDING 8

Green and familiar forests will eventually return following insect outbreaks in most locations.

Forests have continued to develop following past insect outbreaks. Although the current outbreaks are very large and may even be unprecedented in extent and severity in recent history, there is no evidence that affected forests cannot regenerate following these disturbances. The forests that are now losing many trees to insect attack will not look the same in our lifetimes, but healthy trees and familiar forest structures will eventually return in most locations. Although beetle-affected forests may look different to the human eye, they are still functioning ecosystems that provide food and shelter for animals and water for fish and people.

FINDING 9

The 2001 Roadless Area Conservation Rule allows sufficient flexibility to manage Colorado's roadless areas.

Anticipating the need for limited active management, the 2001 Roadless Area Conservation Rule allows sufficient flexibility locally to address public health and safety, fire and undesirable insects, while maintaining the qualities and character of national forest roadless areas. Under the state's proposal, Colorado's national forest roadless areas would be subjected to numerous exceptions to the protections that are provided under the national rule, thereby degrading roadless qualities and providing fewer protections to these areas than any state in the nation.

INTRODUCTION

Johnny N. Dell/bugwood.org

THERE IS STRONG SCIENTIFIC EVIDENCE that roadless areas provide high-quality habitat for threatened species; contain important concentrations of mature and old-growth forests, aquatic strongholds and other sensitive and ecologically important habitat; and act as a buffer against invasive species (Strittholt and DellaSala, 2001; DeVelice and Martin, 2001; Loucks *et al.,* 2003; Peterson, 2005). Because more than half the nation's roadless areas are at elevations above 7,000 feet (U.S. Forest Service, 2000), they are vital for wildlife seeking cooler, moister conditions in the face of a warming climate. Colorado's roadless areas also provide recreational opportunities of all kinds, including hunting and fishing (Peterson, 2005) and world-class backcountry skiing and trekking. The old trees in roadless areas also sequester and store carbon for centuries, playing a pivotal role in absorbing greenhouse gas pollutants, and many of the nation's roadless areas are at the source of downstream drinking water supplies (USFS, 2000).

Due to the ecological and social importance of roadless areas, the 2001 Roadless Rule applies a consistent and scientifically based national standard that safeguards the numerous ecological benefits that roadless areas provide (USFS, 2000). The state of Colorado has proposed a change to the 2001 rule and has proposed exemptions that would remove at least 246,000 acres of roadless area from the national inventory or by allowing logging, road construction, oil and gas development, additional coal mining and exploration, and expansion of ski areas (scientists' letter to Governor Bill Ritter Jr., 2009).

One key reason given for the proposed exemptions is to address recent outbreaks of bark beetles that have killed millions of trees across Colorado and other Rocky Mountain states. The state's 2009 proposal would allow the following activities:

- ***Tree-cutting anywhere in a roadless area where the regional forester determined it was needed to prevent or suppress an insect or disease epidemic.***

- ***Tree-cutting, with accompanying road construction, for fire and insect management within a 1.5-mile radius of any at-risk community*** (a term not defined in the state's proposal).

Modifying the Roadless Rule in Colorado is intended to help managers mitigate the spread of ongoing insect outbreaks, reduce susceptibility to future outbreaks and reduce the risk of

forest fires that is believed to increase due to insect infestations.

The widespread concern about recent outbreaks is understandable, because they may be unprecedented in recent history. Furthermore, it is reasonable to be concerned about the health of affected forests and the potential for the elevated risk of forest fires due to outbreaks of bark beetles. However, those concerns need to be informed by the best available science to ensure that our responses do not have unintended ecological consequences with undesirable effects now and in the future.

In this paper we outline key aspects of bark beetle outbreaks as well as their relationship to fire risk. We also discuss the effects that the 2009 Colorado proposal to modify the 2001 Roadless Rule would probably have on bark beetle outbreaks, fire risk and the ecological values of roadless areas.

COLORADO'S FOREST ECOSYSTEMS

Wilderness Workshop

NATIONAL FORESTS IN COLORADO provide a diverse array of vegetation types, ranging from warm, dry pinyon-juniper woodlands at lower elevations to cold, moist subalpine forests at upper elevations. While large expanses of grasslands, shrublands, croplands, rangelands and other vegetation types dominate Colorado, its roadless areas are predominantly coniferous forests occupying mountainous terrain. Approximately 72 percent of the state's roadless areas contain various forest communities. The largest types of forests in Colorado's roadless areas are spruce-fir (24 percent), followed by aspen (20 percent), lodgepole pine (13 percent), Douglas fir (8 percent), ponderosa pine (3 percent), pinyon juniper (2 percent) and other tree species (2 percent) (USFS, 2008).

Mature and old-growth forest stands are likely to be more prevalent within roadless areas than in areas where higher levels of logging and other human activities have occurred (USFS, 2008). These forests, in particular, are highly valuable as part of Colorado's wildlife heritage, providing habitat for countless species dependent on older forests. They also play an important role in carbon storage and carbon cycling. Finally they offer scenic value to local residents and tourists. Of particular importance to the people of Colorado is the unique role that roadless areas play in supplying clean water to facilities that treat and distribute drinking water.

Insects and Colorado's Forest Ecosystems—A Co-Adapted System

Bark beetles are native to the vast conifer forests of temperate North America and the tree species have co-adapted with these organisms. In fact, these small insects play a vital role in ecosystem function, and their absence could have a profound effect on the function of forest ecosystems (Black, 2005).

There are more than 6,000 species of bark beetles worldwide. Most species cause little or no economic damage, normally infesting branches, stumps and stems of standing dead or severely weakened trees or downed woody material. A few species, including members of the *Dendroctonus* genus, (which includes the mountain pine beetle) normally exist as small endemic populations that feed mainly on trees that recently died, but when conditions are right, the populations can grow rapidly to epidemic levels, overwhelming the defenses of live trees and resulting in widespread mortality.

Native insects, including those that attack and sometimes kill large patches or stands of trees, have been part of Rocky Mountain forests for millennia and have played an important role in

Edward S. Ross

Bark beetles feed on the inner bark of trees, cutting off the flow of nutrients from the leaves to other parts of the tree.

the development of forest ecosystems. For example, there is abundant scientific data that document epidemics of spruce beetles (*Dendroctonus rufipennis*) having killed trees over extensive areas of forest, even long before Colorado was a state (Veblen *et al.*, 1991; Veblen *et al.*, 1994; Eisenhart and Veblen, 2001; Kulakowski and Veblen, 2006). These and other native insects evolved with their host trees over thousands of years; they function as nutrient recyclers, agents of disturbance, members of food webs and regulators of forest productivity, diversity and tree density (Clancy, 1993). Low-severity outbreaks that kill select susceptible trees may increase growth rates of surviving trees (Haack and Byler, 1993). High-severity outbreaks that reach epidemic levels can contribute to the development of complex, multi-aged stands (Axelson *et al.*, 2009) and may ultimately contribute to increased biodiversity and resilience to future disturbances (Alfaro *et al.*, 1982).

Source of Food Web Dynamics

Bark beetles are important parts of many forest food webs. A salient feature of bark beetle communities is the staggering number of organisms associated with them (Dahlsten, 1982). These insects are hosts for parasites and are prey for a variety of animals, including spiders, birds and other beetles.

Bark beetle outbreaks can help create habitat and resources for a variety of species. By feeding on dead or dying trees, bark beetles provide food to insect-eating birds such as woodpeckers (Koplin and Baldwin, 1970) and create snags that may be used by woodpeckers, owls, hawks, wrens and warblers, as well as such mammals as bats, squirrels, American marten, Pacific fishers and lynx. Epidemics of bark beetles increase the availability of plant material for foraging, browsing and nesting for wildlife such as small mammals and birds (Stone and Wolfe, 1996). In a study of ponderosa pine forest on the Front Range of Colorado, herbaceous biomass was 50 to 100 times greater in stands five years after an infestation of mountain pine beetles than in uninfested stands (Kovacic *et al.*, 1985). Kovacic *et al.* (1985) estimated that it is possible that levels of wildlife habitat will remain elevated above pre-infestation levels for 10 to 15 years following beetle infestation. Forest insects ultimately contribute to recruitment of large coarse woody debris into riparian areas and stream systems, which is essential for building pools that provide habitat for trout. Compared to historic conditions, these large, deep pools are lacking in many streams today (Williams and Williams, 2004).

Maintaining Forest Heterogeneity and Diversity

In many forest types, low-severity outbreaks of bark beetles and defoliators reduce the density of trees and cull weak trees, relieving the stress on the survivors, increasing the diversity of stands and creating multi-aged stands (Schowalter, 1994). In many cases, the prime beneficiaries of low-severity outbreaks are the surviving trees (Schowalter and Withgott, 2001). For instance, the Douglas-fir beetle (*Dendroctonus pseudotsugae*) may help maintain ponderosa pine by contributing to a shift in dominance from Douglas fir to ponderosa pine (Schowalter and Withgott, 2001). At the plant community level, insect outbreaks can also increase the diversity of tree species over the long term (Schowalter, 1994), which can promote functional stability and regeneration of forest ecosystems following subsequent disturbances. Trees that are killed by insects and remain on site decompose and ultimately contribute to improved soil fertility (Schowalter, 1994).

IMPORTANT INSECTS IN COLORADO'S FORESTS

Whitney Cranshaw/www.bugwood.org

Native mountain pine beetle.

MOUNTAIN PINE BEETLE IN LODGEPOLE PINE STANDS

Lodgepole pines of the inland West often form extensive, even-aged stands across vast landscapes. Stands of lodgepole pine are the main hosts for mountain pine beetles (*Dendroctonus ponderosae*) and are most affected by the ongoing outbreak. The mountain pine beetle is a native species that has played an important role in affecting the structure and dynamics of lodgepole pine ecosystems for millennia (Fuchs, 1999). It is considered one of the most important insects of western forests and can cause substantial mortality to lodgepole and ponderosa pine as part of the process of forest succession.

Mountain pine beetles typically exist as small, endemic populations in many stands of lodgepole pine. These populations feed on the innermost bark layer of trees (the phloem that transports soluble organic material made during photosynthesis) that have been weakened by disease or injury. At low population levels, mountain pine beetles do not usually successfully attack healthy trees. However, populations of mountain pine beetle can sometimes erupt to epidemic levels if stand structure and climatic conditions are appropriate. During outbreaks, large numbers of beetles launch pheromone-mediated attacks (beetles produce a chemical signal that attracts other beetles to the area) on healthy trees and can overcome the defenses of even healthy trees, sometimes leading to widespread mortality of host species. Such outbreaks of mountain pine beetle are part of a normal boom-and-bust cycle (Amman, 1977). These cycles are likely to have occurred in lodgepole pine forests for thousands of years. Epidemics lasting five to 20 years occur at irregular intervals, affecting large areas by sometimes killing more than 80 percent of trees that are 10 centimeters in diameter (about 4 inches) or greater (Safranyik, 1989).

There is considerable variation in the degree of mortality in lodgepole pine forests affected even by severe outbreaks; and even in stands in which all trees appear to have been killed, live seedlings, saplings or trees of lodgepole pine or other species are often present (Rocca and Romme, 2009; Axelson *et al.,* 2009). In dense stands, these seedlings, saplings or surviving trees will be released from competition for light, water or other resources and will be able to grow rapidly to re-establish the canopy. In areas where seedlings or saplings are not present in stands severely affected by beetles, forest development following outbreak will be via the establishment of new seedlings from seeds in the soil.

Generally speaking, outbreaks of beetles can facilitate the development of a forest that is structurally, genetically and compositionally more diverse (Axelson *et al.,* 2009) and therefore perhaps less prone to subsequent beetle attack (Amman, 1977). Thus, despite causing mortality of many individual trees, outbreaks can also play a critical role in ecosystem processes (Berryman, 1982).

Insects in Non-Lodgepole Colorado Forest Types

Most western pines are susceptible to mountain pine beetle, but most of the mortality associated with the ongoing outbreak is in lodgepole pine and ponderosa pine forests. Recently, mountain pine beetle has caused mortality of whitebark pine at higher elevations. Although ponderosa pine has been less affected than lodgepole pine, there is a potential for increased mortality of ponderosa pine along Colorado's Front Range as the outbreak progresses (Negrón and Popp, 2004). High levels of mountain pine beetle were observed along the Colorado Front Range in the mid-1970s (Schmid and Mata, 1992). In several regions of the western United States, past logging and fire suppression have led to overly dense and simplified ponderosa pine stands that may be more vulnerable to insect outbreaks (Hessburg *et al.,* 1994; Ferrell, 1996; Filip *et al.,* 1996; Maloney and Rizzo, 2002).

The Douglas-fir beetle is often a secondary agent that attacks low-vigor or damaged Douglas firs. Outbreaks usually occur in areas of wind-thrown trees, at sites damaged by fire or during periods of extreme drought (Furniss and Carolin, 1977). The beetle often attacks Douglas firs that have been infected with root disease or defoliated by western spruce budworm (*Choristoneura occidentalis*) or Douglas fir tussock moth (*Orgyia pseudotsugata*). In some areas, the Douglas-fir beetle may help maintain dominance of ponderosa pine by thinning out individual Douglas firs (Schowalter and Withgott, 2001).

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Bark beetle infestation of a stand of lodgepole pines.

The most important insect of mixed forests of Engelmann spruce and subalpine fir is the spruce beetle. Usually these beetles are restricted to recently wind-thrown trees or trees weakened by root disease, but they can reach epidemic levels if the right stand structure and climatic conditions are present (Romme *et al.,* 2006). As mentioned above, there is abundant scientific evidence that epidemics of spruce beetles have killed trees over extensive areas of forest over the past centuries (Veblen *et al.,* 1991; Veblen *et al.,* 1994; Eisenhart and Veblen, 2001; Kulakowski and Veblen, 2006).

ROOT CAUSES OF INSECT INFESTATIONS IN COLORADO'S FORESTS

THERE IS NO SINGLE, SIMPLE REASON that a population of bark beetles reaches epidemic levels. However, warm and dry conditions have been shown to be important to outbreaks in the Rocky Mountains (Logan and Powell, 2001). Such conditions favor the reproduction and survival of beetles and also stress host trees. Appropriate stand structure, including forests of a susceptible age, tree size and density conducive to supporting large numbers of the beetle, also are important (Shore and Safranyik, 1992; Shore *et al.,* 2000; Safranyik and Wilson, 2006; Bentz, 2008; Kaufmann *et al.,* 2008; Bentz *et al.,* 2009).

A warming climate during the last 100 years, particularly in the last few decades, appears to have played a major role in recent insect outbreaks. Numerous scientific studies indicate that climate is an important factor for outbreaks of bark beetles of various species (Bentz *et al.,* 1991; Logan *et al.,* 2003; Carroll *et al.,* 2004; Breshears *et al.,* 2005). Dry and warm conditions can stress host trees and make them less able to defend against the beetles; these same climatic conditions can also accelerate the growth of beetle populations and reduce winter mortality (Carroll *et al.,* 2004).

In addition to climate, stand structure is important in outbreaks of bark beetles. Bark beetles attack trees by boring through their bark (Safranyik and Carroll, 2006). They then lay their eggs in the phloem (the inner bark), the larvae kill the tree by girdling it and by the introduction of blue stain fungus, which can block water transport. Finally, new beetles emerge and repeat the cycle. Beetles prefer larger trees because the thicker phloem is better suited to supporting beetle larvae. During low- to moderate-severity outbreaks, trees in old or dense stands may be more susceptible to beetles than trees in young or less-dense stands because of competition for limited water and nutrients (Shore and Safranyik, 1992). For example, spruce-fir forests that became established after severe forest fires in the late 19th century were less susceptible to a severe spruce beetle outbreak in the 1940s (Veblen *et al.,* 1994; Kulakowski *et al.,* 2003; Bebi *et al.,* 2003). If this relationship between fires and outbreaks holds, it would imply that the forests that are now affected by severe fires may be less susceptible to outbreaks in subsequent decades.

The current epidemic of mountain pine beetle is possible, in part, because vast areas of lodgepole pine provide suitable habitat for the mountain pine beetle (Hicke and Jenkins, 2008; Raffa *et al.,* 2008). This expanse of mature lodgepole pine forests in the central Rocky Mountains is in large part the result of widespread severe wildfires during several years of extreme drought in the 19th century (Veblen *et al.,* 1994; Kulakowski and Veblen, 2002; Kulakowski *et al.,* 2003; Veblen and Donnegan, 2006; Sibold *et al.,* 2006; Sibold and Veblen, 2006). Numerous scientific studies have determined that these fires were largely responsible for current landscape structure and are typical of the fires that have shaped lodgepole pine forests in the Rocky Mountains for centuries. Thus, the presence of suitable habitat for mountain pine beetle is a factor that is characteristic of lodgepole pine-dominated landscapes in the Rocky Mountains.

Steve Nix

Southern Colorado hardwood fire.

FOREST INSECTS AND FIRE: ESSENTIAL ELEMENTS OF ECOSYSTEMS

FIRE INTERACTS WITH BARK BEETLES IN MANY WAYS. Occurrence and severity of fire following an insect infestation will depend on the forest type, intensity of the outbreak and time since the outbreak. Despite the long-standing belief that insect outbreaks lead to increased risk of fire, this assumed link is not well supported by the best available science for most of the forests in Colorado currently affected by outbreaks (Romme *et al.,* 2006; Jenkins *et al.,* 2008; Simard *et al.,* 2008). Rather, the best available science indicates that the occurrence of large, severe fires in lodgepole pine and spruce-fir forests is primarily influenced by climatic conditions rather than fuels.

Although it is widely believed that insect outbreaks set the stage for severe forest fires, the few scientific studies that support this idea report only a small effect, while other studies have found no increase in fire following outbreaks of spruce beetle and mountain pine beetle (Kulakowski *et al.,* 2003; Bebi *et al.,* 2003; Kulakowski and Veblen, 2007; Simard *et al.,* 2008; Jenkins *et al.,* 2008; Bond *et al.,* 2009; Tinker *et al.,* 2009). Theoretically, the effect of outbreaks on subsequent fires will vary with the time since the outbreak occurred (Romme *et al.,* 2006). For example, it is reasonable to expect that foliar moisture in trees killed by beetles will decrease and canopy density will be reduced during and immediately after an outbreak; in subsequent years, canopy density may be further reduced as dead needles and small branches fall from killed trees, which may be associated with an increase in volume of large fallen fuel; and finally increased growth of smaller trees may lead to greater structural heterogeneity and fuel ladders (Romme *et al.,* 2006; Bentz *et al.,* 2009). Although such a model is theoretically possible, studies on the influence of outbreaks on subsequent stand-replacing fires over a range of years since outbreak have found little or no increase in fire occurrence, extent or severity.

Fire and Mountain Pine Beetle Outbreaks In Lodgepole Pine Forests

Although outbreaks of mountain pine beetle can alter fuel structure (Page and Jenkins, 2007a; Tinker *et al.,* 2009; Klutsch *et al.,* 2009), the actual effects of these changes in fuels on subsequent fire risk are complex and may be counterintuitive. Lodgepole stands that experienced high mortality from beetles (more than 50 percent of susceptible trees) in the five to 15 years preceding the 1988 Yellowstone fires had a higher incidence of crown fire than stands that did not have high beetle mortality (Turner *et al.,* 1999). Stands with low to moderate beetle mortality had a lower incidence of high-severity crown fires than stands with no beetle mortality. However, because beetle mortality occurs preferentially in older stands, it is not clear whether the changes in fire behavior were the result of outbreak or pre-outbreak stand structure (Simard *et al.,* 2008)—that is,

that stands with higher beetle-caused mortality may have been generally older stands, which are inherently more likely to burn at high severity than are younger stands because of different fuel structure even in the absence of beetle activity (Renkin and Despain, 1992). Beetle-killed lodgepole pine stands, which were characterized by lower density, experienced significantly lower fire severity compared to adjacent burned areas that had not been affected by beetles in the 3,400-hectare (8,398-acre) Robinson Fire that burned in Yellowstone National Park in 1994 (Pollet and Omi, 2002). A possible explanation is that beetle kill may actually decrease the hazard of high-severity crown fire by reducing the continuity of the canopy.

Lynch et al. (2006) also examined the influence of previous beetle activity on the 1988 Yellowstone fires by testing whether fire was more likely where beetles had killed trees than in areas unaffected by the beetles. These researchers found that stands affected by beetles in 1972-5 had a higher probability of burning but that the increase was only about 11 percent compared to areas unaffected by beetles. In contrast, stands that were affected by beetles in 1980-3 did not increase the likelihood of fire in comparison to uninfested stands (Lynch *et al.,* 2006).

It has been hypothesized that the risk of fire may increase only during and immediately after outbreaks of bark beetles when the dry red needles are still on the trees (Romme *et al.,* 2006). However, Kulakowski and Veblen (2007) found that ongoing outbreaks of mountain pine beetle and spruce beetle did not affect the extent and severity of fire and suggested that changes in fuels brought about by outbreaks may be overridden by climatic conditions. Tinker et al. (2009) examined fuel conditions for 35 years following outbreaks of mountain pine beetle in Yellowstone National Park. They documented reduced canopy moisture content after an outbreak, which was coupled with reduced canopy bulk density. In simulation models of fire behavior, under intermediate wind conditions (40 to 60 kilometers per hour, or 12.5 to 37 miles per hour), the probability of active crown fire in stands recently affected by beetles was significantly lower than in stands not affected by beetles (Tinker *et al.,* 2009). If winds were below 40 kph, (12.5 mph) or above 60 kph (37 mph), stand structure had little effect on fire behavior. Thus, although the canopy was drier immediately after an outbreak, no increase in fire risk was observed likely because of the more important effect of reductions in canopy bulk density. Other independent modeling studies have also predicted a reduced risk of active crown fire five to 60 years after outbreaks, due to decreased canopy bulk density (Page and Jenkins, 2007b; Jenkins *et al.,* 2008).

The best available science indicates that outbreaks of bark beetles in lodgepole pine may have little or no effect on subsequent fires and may in some cases actually reduce the risk of fire. In contrast, there is strong scientific evidence linking severe forest fires in lodgepole pine to drought conditions (Bessie and Johnson, 1995; Sibold and Veblen, 2006; Schoennagel *et al.,* 2004). Thus, the occurrence of severe fires in lodgepole pine forests is primarily influenced by climatic conditions rather than changes in fuels caused by bark beetle outbreaks.

Whitney Cranshaw, Colorado State University/bugwood.org

Trees can sometimes fight off bark beetle attacks by blocking their way with flows of sap, called pitch tubes.

Fire and Spruce Beetle In Subalpine Spruce-Fir Forests

There is increasing evidence that spruce beetle outbreaks have little or no effect on the occurrence or severity of fires capable of replacing stands of spruce-fir forests (Simard *et al.,* 2008). 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; Schoennagel *et al.,* 2004). Under such extreme drought conditions, increased dead fuels from bark beetle outbreaks appear to play only a minor role, if any, in increasing fire risk (Romme *et al.,* 2006). After a 1940s spruce beetle outbreak that resulted in dead-standing trees over thousands of acres of subalpine forests in the 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). Likewise, beetle-affected stands were not more susceptible to a low-severity fire that

spread through adjacent forest several years after the outbreak subsided (Kulakowski *et al.,* 2003). During the extreme drought of 2002, large fires affected extensive areas of Colorado, including some spruce-fir stands that were previously affected by this outbreak of spruce beetle (Bigler *et al.,* 2005). Despite the expectation that these outbreaks would have led to an increased risk of severe fires, the outbreak had only a minor influence on fire severity (Bigler *et al.*, 2005). Likewise, ongoing outbreaks of spruce beetle (and mountain pine beetle) had no detectable effect on the extent or severity of fires in 2002 (Kulakowski and Veblen, 2007). These empirical findings support modeling studies that predict likely reductions in the probability of active crown fire for one to two decades after high-severity bark beetle outbreaks in pure stands of Engelmann spruce (Derose and Long, 2009). Other independent modeling studies have also predicted a reduced risk of active crown fire five to 60 years after outbreaks due to decreased canopy bulk density (Page and Jenkins, 2007b; Jenkins *et al.,* 2008).

As with lodgepole pine forests, empirical and modeling studies from numerous independent researchers indicate that increased risk of wildfire is not an inevitable consequence of bark beetle outbreaks. Instead, climatic conditions appear to have an overriding effect on fire regimes in spruce-fir forests—so much so that changes in fuels brought about by outbreaks of spruce beetle have little or no effect on fire occurrence, extent or severity.

DOES TREE-CUTTING TO ADDRESS BARK BEETLE INFESTATIONS REDUCE STAND SUSCEPTIBILITY TO OUTBREAKS?

William M. Ciesla/bugwood.org

Adult pine beetles in egg gallery under bark.

TWO IMPORTANT QUESTIONS need to be considered when making forest management decisions in response to bark beetle outbreaks:

- Considering specific goals, what are the efficacies of various management strategies?
- What ecological and economic costs do these management strategies impose?

There is very little reliable empirical evidence to suggest that silvicultural treatments can effectively stop outbreaks once a large-scale insect infestation has started. Despite nearly 100 years of active forest management to control the mountain pine beetle, evidence for the efficacy of this approach is scant and contradictory. Citing multiple sources, Hughes and Drever (2001) found that most control efforts have had little effect on the final size of outbreaks, although they may have slowed beetle progress in some cases and prolonged outbreaks in others. They also suggest that management interventions have never controlled a large-scale outbreak. Although control of such outbreaks is theoretically possible, it would require treatment of almost all of the infected trees (Hughes and Drever, 2001), which may be possible only for a small infestation.

In some situations, removing infested trees prior to the emergence of broods is recommended to protect remaining trees. However, the overall effectiveness of this strategy over a large area is unproved (Wilson and Celaya, 1998). Further, in most situations, it is probably not logistically feasible to locate and remove all trees before the emergence of adult beetles (Wilson and Celaya, 1998).

Amman and Logan (1998) point to failed attempts to use direct control measures, such as pesticides and logging, after an infestation starts. They suggest that by the early 1970s, it was apparent that controlling the extensive mountain pine beetle outbreaks that were occurring in the northern Rockies by directly killing the beetles was not working.

Wickman (1990) detailed the effort to control the mountain pine beetle at Crater Lake National Park in Oregon from 1925 to 1934. More than 48,000 trees were cut down and then burned in the last three years of the outbreak. The lesson learned was that once a mountain pine beetle population had erupted over a large area of susceptible forest, and as long as environmental conditions remained favorable, there was no effective way to stop the beetles until almost all the susceptible trees were either killed or removed by logging or until climatic conditions became unfavorable for sustaining an outbreak (Wickman, 1990).

The Crater Lake experience is not an isolated one, as control efforts have been standard practice across the West. Klein (1978) traced several mountain pine beetle epidemics from beginning to end and detailed the control efforts. More

than 30,000 infested ponderosa pine trees and 20,000 infested lodgepole pine trees were treated in 1910 and 1911 in the Wallowa-Whitman National Forest in Oregon. The treatments included felling and peeling, felling and scoring the top, and felling and burning. Chemical methods were employed in the 1940s and '50s. DDT and other pesticides, such as lindane, were sprayed on thousands of acres across the intermountain West. In Operation Pushover, more than 1,800 acres of lodgepole pine in the Wasatch National Forest in Utah were mowed down by heavy tractors linked together, and the surrounding stands were sprayed with pesticides. In spite of these control attempts, mountain pine beetle outbreaks continued (Klein, 1978). Klein (1978) ultimately suggests that letting infestations run their course may be a viable option.

Pine beetle suppression projects often fail because the basic underlying causes (e.g., stand structure, age of trees, drought) of the outbreak have not changed (DeMars and Roettgering, 1982). Wickman (1990) pointed out that once bark beetles reach epidemic levels and cause extensive tree mortality, treatments aimed at stopping the outbreak are futile because it is logistically impossible to eliminate all suitable habitat or to mitigate the overriding effect of climate.

Large-scale efforts to control beetles are also expensive and ecologically harmful. The uncertain benefits of control efforts should be weighed carefully against costs (Hughes and Drever, 2001). In fact, much of the logging in stands infested with bark beetles has been to log merchantable timber. In 1994, then-U.S. Forest Service Chief Jack Ward Thomas, in testimony before the Senate Subcommittee on Agricultural Research, Conservation, Forestry and General Legislation, acknowledged that "the Forest Service logs in insect-infested stands not to protect the ecology of the area, but to remove trees before their timber commodity value is reduced by the insects." There is no doubt that timber extraction is a viable and legitimate use of national forest lands. However, it is important that ecosystem management be driven by clear and explicit goals (Christiansen *et al.,* 1996).

The Case for Thinning: Mixed Reviews

Because stressed and unhealthy trees may be more susceptible to bark beetles, another management approach is to modify stand structure by thinning forests before an outbreak starts. Some thinning studies show success in ameliorating mountain pine beetle infestations in lodgepole and ponderosa pine forests (Amman and Logan, 1998). But the overall evidence of the effectiveness of thinning is mixed.

The evidence for thinning

Most evidence supporting thinning as a control for bark beetles is based on tree vigor, not on directly measured insect activity in the stand. Thinning may increase tree vigor, which in turn may make trees less susceptible to insect infestation. The premise is that if the trees are healthy and highly vigorous, they may be able to "pitch out" the attacking beetles, essentially flooding the entrance site with resin that can push out or drown the beetle.

Some studies suggest that thinning forest stands to reduce competition for light and water may increase tree vigor, leaving what appear to be the best trees and resulting in less successful bark beetle attacks (Sartwell, 1971; Schmid and Mata, 2005; Fettig *et al.,* 2006). Larsson *et al.* (1983) examined the relationship between tree vigor and susceptibility to mountain pine beetle in ponderosa pine in central Oregon. Overall, low-vigor trees were more often attacked by beetles than high-vigor trees in early stages of outbreaks.

Perhaps the studies by Negrón most conclusively show that beetle activity is associated with high densities of stocking (Negrón, 1997; Negrón, 1998; Negrón *et al.,* 2000; Negrón *et al.,* 2001; Negrón and Popp, 2004). These studies show a positive correlation between attacked trees and poor growth. Research in Arizona, Utah and New Mexico showed roundheaded pine beetles (*Dendroctonus adjunctus*) prefer stands and trees exhibiting poor growth, and poor growth rates were positively correlated to dense stands (Negrón, 1997; Negrón *et al.,* 2000). Similarly, research in Colorado's Front Range showed Douglas-fir beetles attacked stands containing a high percentage of basal area represented by Douglas-fir, high tree densities and poor growth during the five years prior to attack (Negrón, 1998; Negrón *et al.,* 2001). Negrón and Popp (2004) reported that ponderosa pine plots in Colorado's Front Range infested by mountain pine beetle had significantly higher tree basal area and density.

Several studies in areas across the west have shown that thinning reduces the amount of mortality caused by mountain pine beetle in ponderosa pine stands (McCambridge and Stevens, 1982; Fiddler *et al.,* 1989, Schmid and Mata, 2005) and lodgepole pine (Cole *et al.,* 1983; Whitehead, 2005), and some scientists

and managers recommend thinning as a viable management strategy for managing bark beetles in these forest ecosystems (Fettig *et al.*, 2007).

Evidence against thinning

Other research has found bark beetles do not preferentially infest trees with declining growth (Santoro *et al.,* 2001). Sánchez-Martínez and Wagner (2002) studied bark beetles in ponderosa pine forests of northern Arizona to see if differences in species assemblages and relative abundance were apparent for managed and unmanaged stands. They found no evidence to support the hypothesis that trees growing in dense stands are more colonized by bark beetles.

Some scientists have suggested caution in using thinning to control bark beetles because geographic and climatic variables may alter the effect (Hindmarch and Reid, 2001). Hindmarch and Reid (2001) found that pine engravers had longer egg galleries, more eggs per gallery and higher egg densities in thinned stands. Warmer temperatures in thinned stands also contributed to a higher reproduction rate (Hindmarch and Reid, 2001). However, pine engravers in Arizona had the opposite reaction to a similar thinning experiment (Villa-Castillo and Wagner, 1996).

There is also evidence to suggest that thinning can exacerbate pest problems. Outbreaks of pine engravers have been shown to be initiated by stand management activities such as thinning (Goyer *et al.,* 1998). The process of thinning can wound remaining trees and injure roots, providing entry points for pathogens and ultimately reducing the trees' resistance to other organisms (Paine and Baker, 1993). Hagle and Schmitz (1993) suggest that thinning can be effective in maintaining adequate growing space and resources to disrupt the spread of bark beetles; but note that there is accumulating evidence to suggest that physical injury, soil compaction and temporary stress due to changed environmental conditions caused by thinning may increase susceptibility of trees to bark beetles and pathogens.

Even if thinning does alleviate tree stress at the stand level it is unlikely to be effective against large-scale infestations (Safranyik and Carroll, 2006). Preisler and Mitchell (1993) used autologistic regression models to analyze mountain pine beetle colonization in thinned and unthinned lodgepole pine in Oregon. Thinned plots were initially reported to be unattractive to beetles; but when large numbers of attacks occurred, colonization rates were similar to those in unthinned plots (Preisler and Mitchell, 1993). Similarly, Amman *et al*. (1988) studied the effects of spacing and diameter of trees and concluded that tree mortality was reduced as basal area was lowered. However, if the stand was in the path of an ongoing mountain pine beetle epidemic, spacing and density of trees had little effect (Amman *et al.,* 1988).

Although thinning may be effective in certain circumstances, it must significantly reduce water stress to be effective, which is unlikely during severe droughts associated with many outbreaks. Thus, forest management, either in the form of searching for and removing infested trees or thinning forest stands before outbreaks, is unlikely to prevent major outbreaks due to the inherent difficulties of manipulating stand structure over large enough areas of Colorado and the overriding influence of climatic stress in driving outbreaks.

In conclusion, if a bark beetle infestation is relatively small and concentrated in a limited area, it may be feasible to reduce the population growth by removing infested trees from a forest stand or by thinning a stand to reduce stress on trees competing for limited nutrients, sunlight and moisture. For example, if a small stand of spruce is blown down by a windstorm and populations of bark beetles begin growing in fallen logs, then it may be feasible to remove all fallen, infested trees over a small area. However, given the climatic requirements for beetle population levels to reach epidemic levels, it is not known whether such a situation would lead to an outbreak. In other words, a small population of beetles is not sufficient for an outbreak to occur. Conversely, under climatic conditions favorable for an outbreak, such as those of the past decade, outbreaks of bark beetles can erupt simultaneously in numerous dispersed stands across the landscape. Unfortunately, even if a growing population of beetles is successfully removed from one stand, under outbreak conditions beetles from other stands are likely to spread over a landscape. Given that climate typically favors beetle populations and stresses trees over very large areas, it is unlikely that management could successfully identify and remove all populations of beetles over an extensive region. Thinning and associated roads can also have a negative impact on wildlife and water quality. Experimenting with thinning for bark beetle control should be done only on a limited scale in areas that are already roaded—not in ecologically sensitive roadless areas.

After the Fact: Post-Disturbance Logging After Outbreaks

Timber production may be an appropriate activity in the right places at the right times and with the right methods, and there may be economic benefits from utilizing some of the dead trees that are now abundant on the landscape, but from an ecological standpoint there is little or no need to remove trees killed by insects (Romme *et al.,* 2006), and tree removal may cause ecological harm and exacerbate insect outbreaks. Standing snags and fallen logs contribute to a number of ecological 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 (Romme *et al.,* 2006).

Depending on how it is done, logging after a natural disturbance (so-called salvage or post-disturbance logging) can also inadvertently lead to heightened insect activity. Specifically, logging after insect outbreaks can reduce parasites and insect predators by effectively eliminating their habitat of standing and downed trees (Nebeker, 1989). Therefore, outbreaks could be prolonged because of a reduction in the effectiveness of the beetle's natural enemies (Nebeker, 1989). Standing dead trees are important for several birds that feed on mountain pine beetles (Steeger *et al.,* 1998), and the widespread removal of dead and dying trees eliminates the habitat required by insectivorous birds and other species with the result that outbreaks of pests may increase in size or frequency (Torgerson *et al.,* 1990). Post-disturbance logging differs from natural disturbance as it tends to decrease habitat complexity and diversity by removing large legacies (e.g., standing dead and downed logs), which can lead to an increase in insect activity (Hughes and Drever, 2001).

Furthermore, logging following insect outbreaks can seriously damage soil and roots by compacting them (see Lindenmayer *et al.,* 2008, for synthesis), leading to greater water stress. Soil damage resulting from logging with heavy equipment can increase the susceptibility of future forests to insects and disease (Hagle and Schmitz, 1993; Hughes and Drever, 2001), reduce conifer regeneration by increasing sapling mortality (Donato *et al.,* 2006) and, in general, cause more damage to forests than that caused by natural disturbance events (DellaSala *et al.,* 2006).

COLORADO'S ROADLESS FORESTS

Aron Ralston/Wilderness Workshop

There are 14.5 million acres of National Forest within Colorado.

ENVIRONMENTAL CONSEQUENCES OF MANAGEMENT IN ROADLESS AREAS

Forman (2000) estimated that 20 percent of the land surface in the United States is directly affected by roads. In addition, in a recent analysis of road impacts, Ritters and Wickham (2003) report that 60 percent of the total land area in the United States is within 382 meters (about 1,250 feet) of the nearest road. Closer to home, national forest lands within the southern Rocky Mountain ecosystem have about 45,000 kilometers (28,000 miles) of roads for an average density of about 0.5 km/km2 (Baker and Knight, 2000). This density is relevant to deer and elk populations since large vertebrates may show a negative threshold response to road density if it is approximately 0.6 km/km2 or more (Forman *et al.,* 2003). Over the next 20 years, the projected increase in population and road density for private lands along the Front Range in Colorado is pronounced (Theobald, 2005). As private land areas are developed and transformed, they will no longer provide the environmental benefits that they did prior to development. One consequence is that the value of roadless areas, and public lands in general, for providing key ecological services increases as private lands are developed for other uses.

A broad scale program to treat stands of ponderosa and lodgepole pine and spruce-fir forests that have been affected by the bark beetle will require an extensive road system. The effects of roads on ecological systems have been thoroughly studied and summarized in several recent publications (Trombulak and Frissell, 2000; Forman *et al.,* 2003; Coffin, 2007). Significant unintended consequences can result from individual roads and from the cumulative effects of extensive road networks whether they are temporary, long-term temporary (as in the case of the Colorado proposal) or permanent. Notably, temporary roads may cause even more short-term damage than permanent roads as they are seldom engineered to permanent road standards.

In general, researchers have found roads to have mostly adverse effects on ecological and physical processes and on fish and wildlife populations (Forman *et al.,* 2003). These effects can be either direct or indirect and chronic or acute as summarized below.

Effects on Aquatic Habitats and Fishes

By creating generally impermeable surfaces, roads disrupt the natural infiltration of water into the

Russell Snitzer, Colorado Trout Unlimited

Ecosystems benefit from healthy forests.

soil and increase runoff to streams. Roads effectively increase the area of the drainage resulting in increased peak flows, erosion of channel banks and increased deposition of debris (Forman and Alexander, 1998). These effects are particularly pronounced in mountainous regions, especially on high gradient streams and headwaters (Ziegler *et al.,* 2001). Increased sediment input to streams can result in changes to channel morphology and channel substrate, and the creation of shallow pools (Beschta, 1978). These changes to stream structure, an indirect effect of road construction, often adversely affect native fish habitat. Logging of trees within the riparian zone, coupled with sediment deposition and increased light penetration, often result in increases in stream temperature that will adversely affect cold-water game fishes (Myrick, 2002). In Colorado, trout that occupy headwater streams are particularly vulnerable to logging (Peterson, 1995). Any road network constructed to thin or harvest insect-infested stands will have to be carefully engineered to prevent increased sedimentation rates or alteration of hill slope processes (Reid and Dunne, 1984; Beschta, 1978). While proper engineering can help mitigate some negative effects, it does not mitigate the overall impact of roads on hydrologic processes, water flow and fragmentation of habitat.

The effects of roads on watersheds and fish populations are often greater than the effects of wildfires (Neville *et al.,* 2009). In general, disturbances such as wildfires are known as "pulse" disturbances, because their effects amount to short-lived bursts, while roads, with their more permanent footprint, contribute to ongoing and cumulative "press" disturbances. The type of press disturbance from roads occurs because roads are permanent sources of sediment and altered water flow, whereas fire effects may be relatively short-lived (Burton 2005). In fact, some wildfire events may positively affect the natural function of stream ecosystems over the long-term (Bisson *et al.,* 2003; Minshall 2003). In contrast, roads and stream culverts in particular often act as barriers to fish passage and may reduce the genetic variability of native trout populations (Neville *et al.,* 2009).

Importantly, there is high value to leaving dead and dying trees on steep slopes and near streams and rivers. For example, the presence of such trees for eventual recruitment into Colorado's stream channels may actually improve stream habitat for fishes over the long-term (Riley and Fausch, 1995; Richmond and Fausch, 1995). Trees in stream channels provide a source of large wood to create complex habitat structures (e.g., deep, still pools for safe spawning) beneficial to many aquatic organisms (Gregory *et al.,* 1991).

Effects on Terrestrial Landscapes

The major physical results of roads on the terrestrial environment are increases in forest fragmentation and disruption of the movement of organisms and flow of ecological processes across the landscape (Lindenmayer and Fisher, 2006). For the most part, roads are permanent transformations of the landscape; as such, their effects accumulate over time and space. As fragmentation increases, so will edge effects with accompanying decreases in habitat quality (Fahrig, 1997). In addition to the direct loss of habitat from the imposition of roads, changes to landscape connectivity arise because roads act as barriers (or facilitators) to the transport of matter, nutrients and organisms. For example, invasive, exotic plants are often found along the edges of roads because seed dispersal is facilitated by vehicle or human transport (Forman *et al.,* 2003).

The impact of roads at a landscape scale requires a consideration of cumulative effects. Even if individual road segments are constructed to minimize local environmental effects, they can still have significant adverse effects when considered collectively and as part of a larger network (or matrix) of roads and related disturbances (Forman *et al.,* 2003). In the context of cumulative effects, of most concern are nonlinear changes to abiotic and biotic processes such as nutrient transport and wildlife dispersal that may show threshold effects—that is, sudden changes

to ecological processes as a result of small changes in the environment (Coffin, 2007; Frair *et al.,* 2008).

Direct and Indirect Effects on Wildlife

Effects on wildlife can be categorized as direct or indirect consequences, and population responses may be numerical or behavioral. Direct effects leading to a numerical response by wildlife populations are those related to the immediate loss of habitat due to road creation and any wildlife mortality that may result from subsequent vehicle collisions or by increases in harvest. Direct mortality following road construction can be significant. For example, data for white-tailed deer and mule deer suggest that an estimated 720,000 animals are killed on U.S. roads each year (Conover *et al.,* 1995). However, direct effects beyond habitat loss may be relatively minor for forest roads that receive little traffic.

In the context of possible extensive silvicultural treatment of insect-infected forests in Colorado, however, it may be that indirect effects are most relevant. For example, it is important to recognize that the total area affected by a road may be considerably larger than its immediate footprint. For example, animals may avoid roads because they associate them with human disturbance or with increased mortality risk. As a result, habitat quality is effectively reduced over some boundary zone that may extend a considerable distance from the road (Rost and Bailey, 1979; Rowland *et al.,* 2000; Trombulak and Frissell, 2000; Fahrig and Rytwinski, 2009). A threshold effect is possible because the overlapping effects of neighboring roads cause a nonlinear accumulation of road effects in the landscape (Frair *et al.,* 2008). These effects can lead to reduced and isolated wildlife populations that are vulnerable to continuing decline because of small population size, loss of landscape connectivity and reduced recolonization of vacant habitat patches.

In a recent meta-analysis, Fahrig and Rytwinski (2009) summarized the effects of roads on the abundance of wildlife. Based on 79 studies, with results for 131 species, they found that the number of negative effects of roads on animal abundance outnumbered positive effects by a factor of five. Particularly vulnerable were species such as deer, elk and bear that move over large distances and are thus more likely to encounter roads. Highly mobile animals with large body size often occur at lower densities and have lower reproductive rates and, as a result, are less able to rebound from small population size. For example, Rost and Bailey (1979) documented a negative effect of roads on mule deer and elk in Colorado. More recent studies on mule deer response to roads in areas of oil and gas development in Colorado also have reported avoidance and reductions in habitat quality (Sawyer *et al.,* 2006). Studies conducted to date in the western United States suggest that ungulate populations may be at risk from increasing road densities.

In summary, there are at least three population-level consequences of road construction on wildlife (Bissonette, 2002). First, behavior and movement of animals are often altered significantly by the presence of roads. Second, roads lead to forest fragmentation, which in turn leads to smaller and more isolated populations vulnerable to further loss. Third, declines in landscape connectivity cause isolated populations to lose genetic variation and thus the ability to adapt to changing environmental conditions (Reh and Seiz, 1990; Frankham, 2006).

Benefits of Maintaining Roadless Areas

Roadless areas often serve as refuges for wildlife, providing a source of dispersing animals and enabling them to recolonize depleted populations. In Colorado, wilderness areas and national parks also serve this purpose, but most are at high elevations with extensive areas of rock and ice. As a result, mid- and low-elevation forest communities are poorly represented in protected areas except for designated roadless areas. In the context of climate change, persistence of native plant and animal communities may require the unimpeded opportunity to migrate to higher elevations that unroaded landscapes uniquely provide. Therefore, protecting native plant communities along entire elevational gradients may be required for successful adaptation.

Roadless areas may have their greatest value in terms of protecting watersheds that can maintain high water quality and predictable flows throughout the year. As an example of the increasing value of water in the western United States, consider that a gallon of water in the grocery store may cost more than a gallon of gas. As water becomes increasingly scarce in the southern Rocky Mountains, the hydrological significance of roadless areas for providing clean water to downstream users and offering other ecosystem services to local communities will increase. Roadless areas also serve as refuges for aquatic populations and for replenishing source populations of degraded biota in downstream aquatic ecosystems (Peterson, 2005).

In addition to the role of roadless areas for watershed

protection and for maintaining landscape connectivity, roadless areas are also important as ecological benchmarks for the management and restoration of the roaded landscape. These areas provide reference conditions—"the condition in the absence of human disturbance which is used to describe the standard, or benchmark, against which the current condition is compared" (Sánchez-Montoya *et al.,* 2009). Reference conditions from roadless areas are needed to assess the effects of management in the roaded landscape.

In combination with wilderness areas and parks, roadless areas act as natural laboratories for studying effects of various environmental stressors—for example, global climate change, invasive species, air pollution impacts and fire—in areas not confounded by management or direct human impacts.

In sum, Colorado's 4.3 million acres of inventoried roadless areas provide unique benefits to Coloradoans. Managing these areas consistent with the 2001 Roadless Area Conservation Rule is the highest and best use of these natural resources and the ecosystem services they provide in an increasingly human-dominated landscape. A rapidly changing global and regional climate makes this even more imperative.

MANAGING FIRE RISK NEAR COMMUNITIES VERSUS ROADLESS FORESTS

NFPA Firewise Communities Prgram

Houses among trees and vegetation.

LARGE FIRES AND EXTENSIVE GROWTH of residential communities adjacent to fire-prone forests has raised public awareness of the overall risks of forest fires. As discussed previously, fires, even severe and extensive ones, are essential and normal components of Colorado's forests. Furthermore, rigorous scientific research has found that the influence of outbreaks on subsequent fire risk in Colorado's spruce-fir and lodgepole pine forests is minimal (Romme *et al.*, 2006).

Though outbreaks do not necessarily increase the risk of fires in Colorado's forests, drought conditions can lead to a high risk of fires with or without outbreaks. Although a major goal of fire hazard mitigation is to protect communities, a recent analysis of the location of fuel treatments found that only 11 percent of federal fuel treatments in the western United States have been within 2.5 km (1.5 mi) of the wildland-urban interface, where they would be most effective at reducing fire risk to homes and settlements (Schoennagel *et al.*, 2009). Schoennagel *et al.* (2009), as well as other scientists (Cohen, 2000), conclude that greater priority should be given to treating fuels in and near the wildland-urban interface.

Defensible Space—First Line of Defense

Perhaps the most important component of the wildland-urban interface is the space immediately surrounding homes—also known as defensible space. Building roads and cutting down trees far from communities is not likely to reduce fire risk to homes and neighborhoods. Forest Service fire expert Jack Cohen states categorically that logging away from communities "does not effectively change home ignitability" (Cohen, 2000). Cohen points to a "40-meter zone" (about 130 feet) around the home that determines a home's ignitability (Cohen, 1999). Reducing flammable material from the immediate vicinity of structures and replacing flammable building materials such as wooden decks with nonflammable alternatives has been shown to effectively protect structures against fire damage (Cohen, 1999).

Materials explaining how to protect communities are available from several sources, most notably from the national Firewise Communities, a multi-agency effort designed to involve homeowners, community leaders and planners to protect people and property. Its Web site (www.firewise.org) offers brochures, videos and articles (Firewise, 2009).

As discussed above, silvicultural treatments (often mechanical thinning of merchantable trees) are not a viable option for controlling bark beetle epidemics. Similarly, conventional timber harvests will 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 (Romme *et al.*, 2006). Thus, to be effective at reducing fire hazard to communities, tree-cutting must be executed in a way that removes all flammable material (not just economically valuable timber) and must be located in the immediate vicinity of homes and settlements.

Overall, it is going to be much less expensive, more effective and less damaging to focus fire-hazard reduction efforts around communities and homes than it would be to try to make a wholesale modification of forest structure over large landscapes.

CONCLUSION

Johnny N. Dell/bugwood.org

CLIMATE CHANGE and other factors are leading to unprecedented changes in Colorado's forest ecosystems. One likely consequence of a changed climate is increased bark beetle activity leading to tree mortality over large areas of the state, including its roadless areas. Although ongoing outbreaks have led to widespread public concern about increased fire risk, the best available science indicates that outbreaks of mountain pine beetle and spruce beetle do not lead to increased risk of fire. Nevertheless, forests of lodgepole pine and spruce-fir are naturally prone to high-severity fires during drought conditions, regardless of the influence of bark beetle outbreaks. Thus, there is a need to take effective steps to protect homes and communities from fire risk that is associated with climatic conditions.

There is substantial evidence that logging in roadless areas will not stop bark beetle outbreaks and would be ecologically damaging because such mechanical treatments would necessitate an expansive and potentially damaging roads network. The best available science indicates that once a large infestation has started, it is not possible to stop the outbreak.

Even forest thinning, which is widely promoted as a solution by reducing tree susceptibility to outbreaks, has had mixed results and is unlikely to stem bark beetle epidemics on a large landscape scale, especially during drought cycles. Further, this type of thinning would not be a one-time treatment, but would require regular thinning of all treated stands every decade or so because thinning tends to promote rapid growth of understory vegetation, making it a potential fuel ladder. Moreover, too much thinning can moderate stand climates, which may be favorable to some beetles, and increase wind speeds adding to crown fire spread.

The 2001 Roadless Area Conservation Rule provides a consistent and scientifically based national standard that safeguards roadless areas from development. Colorado's petition to address recent outbreaks of bark beetles by proposing logging and road building in inventoried roadless areas is inconsistent with the best available science and would degrade roadless area values. Colorado's 2009 proposed rule would potentially put hundreds of thousands of acres of roadless areas at risk. Furthermore, it is unlikely to be effective in mitigating the risk of insect outbreaks or severe fire. If there are cases in which it is necessary to remove trees in a roadless area, the 2001 rule allows sufficient flexibility while maintaining needed rigorous protections for roadless qualities.

Scientists, land managers and residents of Colorado are concerned about how wildfire might affect our forests and communities. If the goal is to protect communities, fire-mitigation efforts should be focused around those communities and homes, not in remote and ecologically valu-

able roadless areas. Prioritizing fire-hazard reduction around communities is much less expensive, more effective and less damaging to roadless values than trying to make a wholesale modification of forest structure across the landscape.

Although the current insect outbreaks are very large and their scope and size may even be unprecedented in recent history, there is strong evidence that affected forests will regenerate in time. The forests that are losing many trees to insect attack will not look the same in our lifetimes, but current patterns of regrowth indicate that green forests will eventually return in most locations (Kaufmann *et al.*, 2008; Rocca and Romme, 2009; Axelson *et al.*, 2009). These forests may look different to us, but beetle-affected forests are still functioning ecosystems that provide food and shelter for animals, cool clear water for fish and humans, and irreplaceable refuges for wildlife from the effects of logging, road building and climate change.

ACKNOWLEDGMENTS

For helpful comments on earlier drafts of this report, we thank J. Hicke, D. Jarvis, W.H. Romme, T. Schoennagel, M. Simard, R. Spivak, T. Veblen and C. D. Williams. The authors of this report take sole responsibility for its content.

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