THE WATERSHED IMPACTS OF FOREST TREATMENTS TO REDUCE FUELS AND MODIFY FIRE BEHAVIOR

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prepared for

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PREFACE

This report is based on information from diverse fields, such as fire ecology and watershed hydrology. While it uses terms and concepts from those disciplines that may not be completely familiar to some readers, the literature cited in the report provides more detail on these terms and concepts.

The report repeats key concepts and findings in different sections. This was done to increase the stand-alone utility of the individual sections for readers with diverse backgrounds who may not wish to search the entire report in order to access information and findings in context.

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I. INTRODUCTION

Purpose and Scope

This report examines the effects on watersheds and aquatic resources from forest fuel reduction treatments aimed at modifying wildland fire behavior on public lands. Such treatments have been promoted in some scientific assessments (e.g., Graham et al., 1999; Allen et al., 2002; Graham et al., 2004; Stephens and Ruth, 2005) and recent public forest policy and legislation (Associated Press, 2004) for extensive implementation on Western public lands in an attempt to reduce fire severity and size by altering fuel levels, character, and continuity. For instance, the U.S. National Fire Plan (U.S. Forest Service (USFS), 2002) and the Healthy Forests Restoration Act of 2003 encourage these treatments on a grand scale. Proponents assert that these treatments, when effective, benefit watersheds because higher-severity fire can sometimes trigger severe soil erosion and elevated peakflows (Allen et al., 2002; Graham et al., 2004).

However, fuel treatments will not always provide these benefits to watersheds, because they are not universally effective in reducing fire severity, restoring fire regimes, or reducing the ecological effects of higher-severity fire. As this paper discusses, in most forest systems such treatment benefits are unlikely, due to the transience of treatment effects on fuels, combined with the patchy and poorly predictable nature of fire behavior and occurrence.

Mechanized fuel treatments also incur ecological costs by damaging soils, vegetation and hydrologic processes, as proponents of fuel reduction treatments have acknowledged (e.g., Allen et al., 2002; Graham et al., 1999; 2004; Agee and Skinner, 2005). Mechanical fuel reduction treatments typically involve the same suite of activities as logging, with the same set of impacts to soils, runoff, erosion, sedimentation, water quality, and stream structure and function. These effects, their mechanisms, and their aquatic impacts have been extensively and repeatedly documented across the West (e.g., Geppert et al., 1984; Meehan, 1991; USFS et al., 1993; Rhodes et al., 1994; CWWR, 1996, USFS and USBLM, 1997a; c; Beschta et al., 2004). Watershed damage ultimately translates into aquatic damage.

The collateral impacts of fuel treatments are of considerable concern due to the existing aquatic context. Across the West, aquatic systems are significantly and pervasively degraded (Rieman et al., 2003; Beschta et al., 2004). As a result, many populations of aquatic species, including most native trout and salmonids, have undergone severe contractions in their range and number and remaining populations are now imperiled and highly fragmented (Frissell, 1993; USFS and USBLM, 1997a; Kessler et al., 2001; Behnke, 2002; Bradford, 2005). Additional damage to watersheds and aquatic systems reduces the prospects for the protection and restoration of imperiled aquatic species (USFS and USBLM, 1997c; USFWS, 1998; Karr et al., 2004).

Previous work on this front has not adequately characterized the likely outcomes of mechanized fuel reduction treatments and the resulting tradeoffs for watersheds and aquatic systems. For example, some have viewed the tradeoffs involved with mechanized fuel treatments (MFT)[1] on the basis of unwarranted assumptions, including:

- the assumption that MFT consistently reduce the effects of fire on watershed and aquatic resources (e.g. Allen et al., 2002; Elliot and Miller, 2002; Agee and Skinner, 2005), without any consideration of the distinct and significant probability that they are ineffective;
- the assumption that best management practices and other treatment techniques render the watershed and aquatic impacts negligible (Allen et al., 2002; Graham et al., 2004);
- the binary comparison of treatment impacts with those from high-severity fire based on the implicit or explicit assumption that the former persistently eliminates the latter and, conversely, that high-severity fire is guaranteed to occur in the absence of MFT (Elliot and Miller, 2002; Istanbulluoglu et al., 2004; O'Laughlin, 2005);
- the assumption that trade-offs between the ecological costs and assumed fuel treatment benefits are positive overall, without thorough examination of the likelihood, magnitude, and persistence of the ecological costs or benefits (e.g. Allen et al., 2002; O'Laughlin, 2005);
- the narrow consideration of only the isolated effects of tree removal without consideration of the combined effects of associated activities, including the elevated use, construction, reconstruction and maintenance of roads and landings (Allen et al., 2002; Nez Perce National Forest, 2002; 2004; Santa Fe National Forest, 2004a; Graham et al., 2004; Istanbulluoglu et al., 2004; Agee and Skinner, 2005), the removal of surface fuels via broadcast burning and/or piling and burning, and/or other associated follow-up or repeated treatments.

There are several reasons why the above assumptions are not warranted. First, it cannot be assumed that MFT will be generally effective. The transient effects of treatments on forest fuels (Kauffman, 2004; Graham et al., 2004), coupled with the relatively low probability of higher-severity fire, makes it unlikely that fire will affect treated areas while fuel levels are reduced. Obviously, when treatments do not encounter fire while fuels are reduced, they cannot reduce fire severity and size.

Fuel treatments do not always reduce fire severity and size when they encounter fire. Fuel treatments have been documented to be ineffective at reducing fire severity under some weather conditions (Martinson et al., 2003; Graham et al., 2003; Romme et al., 2003a). In some prevalent forest types, fuel treatments are highly unlikely to reduce fire severity or size (Veblen, 2003; Schoennagel et

[1] In this report, the term "mechanized fuel treatment" (MFT) is used to denote the spectrum of mechanized treatments that remove vegetation as part of efforts to reduce fuels and fire severity. This term is used in lieu of "thinning" because some commonly proposed fuel treatments, such as lineal fuel breaks, do not meet the criteria for "thinning." Graham et al. (1999) noted "there are many stand treatments similar to thinnings that may or may not be thinnings," many of which have been proposed or implemented to reduce fuels (e.g., SFNF, 2004a; RSNF, 2004). The term MFT, as used in this report, includes prescribed fire use when used in combination with mechanical treatments. It does not include prescribed fire when used in isolation or wildland fire use.

al., 2004a; Noss et al., 2006b). Some MFT practices can exacerbate fire severity (Agee, 2003), as documented in Southwest Oregon (Raymond and Petersen, 2005) and the Sierra Nevada, California (Hanson and Odion, 2006). Increases in fire severity *add* to the collateral damage to watersheds and aquatic resources caused by the treatments.

Second, MFT cannot be assumed to eliminate higher-severity fire, nor can it be assumed that untreated areas will burn at high severity, if left untreated. In contrast, there is complete certainty that a single iteration of MFT cannot persistently reduce fuels and future fire severity (Kauffman, 2004; Graham et al., 2004; Agee and Skinner, 2005).

Third, there are no reliable data indicating that "Best Management Practices" (BMPs) consistently reduce the adverse effects of significant soil and vegetation disturbance on aquatic resources to ecologically negligible levels, especially within the context of currently pervasive watershed and aquatic degradation (Ziemer and Lisle, 1993; ISG, 1999; Espinosa et al., 1997; Beschta et al., 2004). BMPs are often not implemented to the degree promised in environmental analyses, and their implementation may be slipshod and/or ineffective (Espinosa et al., 1997: Rhodes, 2002). Activities implemented with somewhat effective BMPs still often contribute to negative cumulative effects on aquatic systems (see Photograph 1 on pg. 18).

Fourth, it cannot be assumed that MFT will always be applied consistent with the best available information on how to reduce fire severity and where such treatments might be needed. Although much of the literature on MFT has largely ignored the issue of implementation, it is a key concern because how and where MFT are implemented affects the treatments' potential effectiveness and their effects on aquatic resources.

Fifth, road construction, reconstruction, use, and maintenance are inexorably linked to MFT and are known to be among the primary sources of aquatic damage on public lands. Similarly, the construction, reconstruction, and use of landings, which have impacts similar to roads, are also inextricably intertwined with MFT.

With some rare exceptions (e.g., Gresswell, 1999; Rieman et al., 2003), most of the literature assessing the aquatic tradeoffs inherent with MFT has not examined the consistency of proposed MFT with known watershed and aquatic protection and restoration priorities. This is significant because MFT have negative or chilling effects on some priority restoration needs, such as the need to reduce the extent and negative impacts of roads, which has been consistently identified in numerous scientific assessments as a vital step to watershed restoration (e.g., USFS et al., 1993; Rhodes et al., 1994; Beschta et al., 2004).

This report aims to plug some of these gaps by taking a harder look at the likely direct, indirect, and cumulative effects of MFT on watersheds and aquatics. While it makes a somewhat detailed examination of the issues, the scope is not exhaustive due to length considerations. Complete books can, and have, been written about some of the topics involved, such as the effects of forest management on salmonids (Meehan et al., 1991).

This report does not focus solely on the effects of thinning to reduce fire severity for several reasons. Under the aegis of fuel reduction, MFT include methods spanning the spectrum from those akin to clearcutting to significant thinning. Such treatments have been proposed or implemented as part of efforts to treat forest fuels across the West (USFS, 1999 (California); Graham et al., 1999 (nationwide); Bitterroot National Forest

(BNF), 2001 (Montana); Clearwater National Forest (CNF), 2002 (Idaho); Umatilla National Forest (UNF), 2001; 2003 (Oregon); Ochoco National Forest (ONF), 2002 (Oregon); Santa Fe National Forest (SFNF), 2004a, 2004 (New Mexico); Graham et al., 2004 (nationwide); Apache-Sitgreaves National Forest (ASNF), 2004 (Arizona)).

This report does not discretely focus on treatments aimed at protecting infrastructure in the "wildland urban interface" on public lands. Treatments in these areas are sometimes primarily predicated on infrastructure protection rather than ecological restoration, while the focus of this report is on treatments aimed at forest and fire regime restoration.

Because the overall effectiveness of MFT on fire behavior is integral to evaluating the net effects on watersheds and aquatic systems, this report also factors in their likely effectiveness, based on a synthesis of available literature and case histories. Since treatments always involve some ecological costs due to the impacts of associated watershed disturbance, this report also examines some of the likely fire-related consequences to aquatic systems with and without MFT.

The evaluation of potential effectiveness of MFT in this report is based on six important contexts, some of which are intertwined.

1. Forest types and their associated natural fire regimes strongly influence the potential effectiveness of MFT in reducing fire severity and restoring natural fire regimes (Veblen, 2003; Schoennagel et al., 2004a).
2. MFT that do not work towards restoring natural fire regimes are likely to ultimately fail (Veblen, 2003; Schoennagel et al., 2004a) and cause damage to forests and watersheds without conferring any of the compensatory ecological benefits of restoring natural fire regimes.
3. In forests where the natural fire regime has not been altered, fuel treatments do not aid in restoring natural fire behavior (Noss et al., 2006b; Baker et al., 2006).
4. The occurrence of high-severity fires that are characteristic of the natural fire regime are not a restoration concern but rather a restoration need (Veblen, 2003; Baker et al., 2006; Odion and Hanson, 2006).
5. If fire that would be higher severity in the absence of treatment does not affect treated areas during the limited time period when fuels have been reduced, the treatments *cannot* reduce fire severity. Therefore, the upper bound of the potential treatment effectiveness is determined by whether or not higher-severity fire affects treated areas while fuel levels are reduced. The location of future fires cannot be predicted with accuracy, but their likelihood can be estimated. This report provides some discrete estimates of the likelihood of fire affecting treated areas while fuels have been reduced at regional and West-wide scales.
6. In order to be ultimately effective at helping to restore natural fire regimes, fuel treatments must be part of wider efforts to address the root causes of the alteration in fire behavior. At best, MFT can only address symptoms of fire regime alteration. Evidence indicates that primary causes of altered fire regimes in some forests include changes in fuel character caused by the on-going effects and legacy of land management activities. These activities include logging, post-disturbance tree planting, livestock grazing, and fire suppression (Veblen, 2003; Noss et al., 2006a; b; Baker et al., 2006). Many of these activities remain in operation over large areas. Therefore, unless treatments are accompanied by the elimination of or sharp reduction in these activities and their

impacts in forests where the fire regime has been altered, MFT alone will not restore fire regimes (Baker et al., 2006)

This report's evaluation of the likely combined effects of MFT on watershed and aquatic resources is also based on an explicit consideration of the level of certainty based on available scientific evidence. For each set of propositions analyzed, this report explicitly categorizes the level of associated certainty in one of the following three categories:

- high degree of certainty = robust field and/or applicable laboratory data;
- medium degree of certainty = effects that can be reasonably inferred from known linkages and available evidence; and
- low degree of certainty = limited data and information on known linkages.

Existing Management Context: Aquatic and Watershed Conditions

The existing condition of watersheds and aquatic systems is key to assessing the significance of additional damage that might be caused by MFT. Any further damage is superimposed on watersheds and aquatic systems that are already pervasively degraded biologically and physically, as independent assessments of watershed and aquatic conditions and trends throughout the West have repeatedly concluded for more than six decades (Leopold, 1937; Nehlsen et al., 1991; Henjum et al., 1994; CWWR, 1996; Hirt, 1996; USFS and USBLM, 1997a; Kessler et al., 2001; Beschta et al., 2004; Karr et al., 2004). In its assessment of the condition of public land in the Sierra Nevada, CWWR (1996) noted that aquatic and riparian systems are "the most altered and impaired habitats." This is also likely true for most other regions in the West.

This pandemic aquatic damage has rendered many aquatic species wholly imperiled due to enormous extirpations throughout their historic range. This has resulted in severe population fragmentation, which further threatens their persistence (Frissell, 1993; Propst and Stefferud, 1997; Shepard et al., 1997; ISG, 1999; USFS and USBLM, 1997a; USFWS, 1998; Kessler et al., 2001; Bradford, 2005).

Freshwater ecosystems have lost a greater proportion of their species and habitat than any other ecosystems (Revenga and Mock, 2000). About 40% of North American freshwater species are extinct or at-risk, with the extinction of at least 123 species of aquatic and amphibian species in the past century (Postel, 2005). North American freshwater species extinctions are estimated to be occurring at roughly five times the rate of terrestrial animals (Postel, 2005). A high percentage of the freshwater fish species native to Western states are imperiled, as summarized in Table 1.

Native inland trout are particularly imperiled. Relatively healthy populations of bull trout and several species of native cutthroat trout now occupy less than 5% of their historic ranges in areas spanning the interior Southwest to the interior Pacific Northwest to the Northern Rockies (USFS and USBLM, 1997a; Kessler et al., 2001; Young and Harig, 2001). Kessler et al. (2001) documented that only one of the eight species of native trout analyzed had relatively healthy populations that occupied more than 6% of their historic range; none of the these eight native trout species had relatively healthy populations in more than 16% of their historic range.

The situation is similar for native salmonids and other native fishes in other areas, such as the Sierra Nevada and western Washington (Moyle et al., 1996; WDFW, 2000). At least 214 individual stocks of anadromous Pacific salmonids in California and the Pacific Northwest are at risk of extinction or of special concern; at least another 106 stocks are already extinct (Nehlsen et al., 1991). Extinct or at-risk stocks of salmon and steelhead in the Pacific Northwest outnumber those considered healthy by more than four-to-one (Huntington et al., 1996). Habitat damage is a major cause of the loss of native salmonids in the West (Henjum et al., 1994; Moyle et al., 1996a; Shepard et al., 1997; USFS et al., 1997a; WDFW, 2000; Behnke, 2002).

Many native trout species are not likely to persist or recover without considerable improvement in habitat conditions and connectivity of habitats and populations (Nehlsen et al., 1991; Henjum et al., 1994; Propst and Stefferud, 1997; Shepard et al., 1997; USFS et al., 1997a; c; USFWS, 1998; Kessler et al., 2001; Rieman et al., 2003). Any additional habitat damage increases the likelihood of local extirpations and ultimate extinction due to increased population fragmentation (USFS and USBLM, 1997c). The impacts of current watershed conditions limit the capacity for recovery of aquatic habitats, by constraining the restoration potential of watershed systems (Beschta et al., 2004). Additional watershed damage is inimical to the restoration of native salmonids (Karr et al., 2004).

Habitat damage is one of the principal reasons that so many species of aquatic invertebrates, fish and amphibians in the Sierra Nevada are in decline (Moyle, 1996a; ECONorthwest and Pacific Rivers Council, 2002). Amphibians are in widespread decline in the West and the degradation of aquatic habitats appears to be the major cause (Willson and Dorcas, 2002; Bradford, 2005).

Despite considerable geographic differences, many aquatic systems throughout the West are now beset with the same set of problems. These include elevated sedimentation, reduced levels of large wood, simplified stream structure, elevated peak flows, reduced baseflows, increased seasonal water temperature extremes, damaged riparian areas, and vastly reduced numbers and diversity of native aquatic and riparian flora and fauna. Scientific assessments in areas ranging from the arid Southwest to the Pacific Northwest have consistently noted these problems and ascribed their causes to a common set of water and land uses: grazing, logging, roads, and water diversions (Leopold et al., 1937; Sublette et al., 1990; USFS et al., 1993; Henjum et al., 1994; CWWR, 1996; Moyle et al., 1996a; Shepard et al., 1997; USFS and USBLM, 1997a; Gresswell, 1999; USFWS,

Table 1. Number of imperiled native freshwater species in several Western states, based on Johnson (1995).

State	Number of Imperiled Native Freshwater Fish Species	Total Number of Native Freshwater Fish Species	Percent of Native Freshwater Fish Species That Are Imperiled
Nevada	39	43	91%
Arizona	22	26	85%
California	42	58	72%
Oregon	25	57	44%
Utah	20	26	77%
New Mexico	20	66	30%

1998; WDFW, 2000; Kessler et al., 2001; Behnke, 2002; Rieman et al, 2003; Dunham et al., 2003b).

In particular, roads have been consistently singled out as a primary cause of the reduced range and abundance of many aquatic species, not only in the West but also across the continent (CWWR, 1996; USFS and USBLM, 1997a; Trombulak and Frissell, 2000; Kessler et al., 2001; Angermeier et al., 2004). Czech et al. (2000) estimated that roads in the U.S. contribute to the endangerment of some 94 aquatic species.

Due to differences in biophysical attributes (e.g., climate, topography, soils, and vegetation), different watersheds and aquatic systems respond somewhat differently to anthropogenic disturbances. However, these responses differ in degree, not in type. For instance, the loss of groundcover inevitably leads to increased erosion whether it occurs in the subalpine terrain of Utah or the coastal lowlands of Washington. Regardless of the stream and setting, elevated sediment delivery contributes to increased turbidity, sedimentation, and the amount of fine sediment in channel substrate. Irrespective of soil type, the use of ground-based machinery causes soil compaction and decreased available soil water storage, soil productivity, and infiltration rates. The removal of trees from along streams ultimately reduces the amount of large woody debris (LWD) in channels, and consequently reduces channel diversity, regardless of geographic setting.

Similarly, all native salmonid species across the West are adversely affected by elevated summer water temperatures, reduced low flows, loss of stream cover, and the consequences of increased sediment delivery, including elevated levels of fine sediment in channel substrate and pool loss (Meehan, 1991; Rhodes et al., 1994; USFS and USBLM, 1997a; USFWS, 1998; McCullough, 1999; WDFW, 2000; Behnke, 2002; Rieman et al., 2003).

These broad similarities in the general direction of both physical and biological responses allow reasonable generalization of the direct, indirect, and cumulative impacts of MFT-related disturbance on watershed and aquatic systems, though the magnitude and persistence of the response often varies with geographic setting and related factors. Correspondingly, the broadly degraded state of aquatic systems across the West also allows one to reasonably interpret that additional watershed degradation has adverse repercussions.

The effects of MFT on aquatic systems are strongly influenced by their scale and location within watersheds (Rhodes et al., 1994). The cumulative scale of watershed disturbance from MFT is a key concern because, other factors remaining equal, adverse cumulative effects on aquatics tend to increase with increasing area of watershed disturbance (Rhodes et al., 1994; Murphy, 1995; Fore and Karr, 1996; Willson and Dorcas, 2002). Similarly, repeated treatments contribute to adverse cumulative effects on aquatic communities over time (Ziemer, 1991; Ziemer et al., 1991; Abbruzzese and Leibowitz, 1997). Tree removal and associated activities in sensitive areas and watersheds increase their negative impacts on aquatic systems (USFS et al., 1993; Rhodes et al., 1994; Murphy, 1995; Beschta et al., 2004). Therefore, the likely spatial and temporal scales of MFT are also evaluated in this report.

II. EVALUATION

The Likely Extent and Frequency of Mechanized Fuel Treatments

There is high degree of certainty that MFT will be proposed over extensive areas, involving significant numbers of watersheds and proportions of watershed areas. Graham et al. (2004) stated that extensive areas needed to be treated to provide "fire safe" landscapes in order to attempt to modify fire behavior, because "[t]reating small or isolated stands without assessing the broader landscape will most likely be ineffective in reducing wildfire extent and severity."

There appears to be general agreement that MFT must be applied extensively in order to alter fire behavior, both at the scale of individual areas and at the broader landscape scale (Finney, 2001; Rummer et al., 2003; Graham et al., 2004; Agee and Skinner, 2005; Stephens and Ruth, 2005). Stephens and Ruth (2005) suggested treating fuels on more than 24.2 million acres of USFS land in the Pacific Northwest and California, or approximately 53% of all such USFS lands.

Graham et al. (2004) suggest treating between 11 and 100 million acres on public lands in the West to reduce fire risk. USGAO (2003) cites a range of 90 to 200 million acres of public lands nationwide that have been considered[2] to have high risk of uncharacteristic wildfire due to fuel conditions, which might be treated to reduce fuel loads. Even if only a fraction of these areas are treated to reduce fuels, this clearly translates into a considerable scale of disturbance on Western public lands.

With the passage of components of the so-called Healthy Forests Initiative in 2003, it is likely that the rate of attempted MFT in the West will increase significantly. A USFS official has stated that the USFS aims to treat about 8 million acres of lands annually to reduce fuels (Associated Press, 2004), equivalent to more than 54% of all such lands in the West over the course of a decade. It is apparent that if the trend continues and/or stated intentions are carried out, a considerable amount of forested public lands in the West will be disturbed and altered by attempts to reduce fire severity by reducing fuels.

These levels of watershed disturbance would be ecologically significant and cause negative cumulative watershed effects under any circumstances. However, they are especially significant due to the already considerably disturbed state of watersheds on public lands and pandemic aquatic degradation.

Data on recent rates of treatment implementation and planning also indicate that treatments will be proposed on a significant scale. Based on data in USGAO (2001), about 1,280 fuel reduction projects on USFS lands in the West were readied for implementation in fiscal year 2001, although the total size (acreage) and amount (board feet) comprised of MFT were not given. In 2001 and 2002, USGAO (2003) estimated that the USBLM and USFS treated an average of about 1.6 million acres for fuel reductions nationwide, with a goal of treating more than 2.3 million acres in 2003. Although the amount of MFT used in these fuel treatments or the amount occurring in the West was not provided, it is safe to assume that many of these projects involved mechanical treatments.

[2] One of the prime points of USGAO (2003) is that the current estimates of the area with adversely high fuel loads have questionable accuracy. The Fire Regime Condition Class method (Hann and Bunnell, 2001) is widely used to assess the potential for uncharacteristic fire posed by elevated fuel loads, if fire occurs. This method likely has very limited accuracy and tends to overestimate the risk of higher-severity fire posed by fuel loads, as documented by studies of recent fires (Odion and Hanson, 2006).

Current broad scale fuel reduction proposals corroborate the high degree of certainty that MFT will affect significant amounts of watersheds. In the Sierra Nevada, the USFS approved an aggressive program of fuels reduction in 11 national forests comprising more than 11 million acres (the program includes 10 national forests in their entirety and a portion of an eleventh national forest) (USFS, 2004). This program aims to mechanically treat about 72,000 acres per decade (USFS, 2004), which translates to more than 12% of the entire planning area over the course of twenty years.

One of the projects revived in USFS (2004) is the Herger-Feinstein Quincy Library Group fuel treatment project (USFS, 1999), which proposed MFT on a significant scale on several forests in the Sierra Nevada of California. The project proposed the construction of 100 miles of permanent road and 300 miles of "temporary" road, while reconstructing 1,000 miles of road and greatly increasing haul traffic on thousands more miles of road. It also proposed to construct hundreds of miles of logged "fuel breaks," together with hundreds of thousands of acres of thinning and other types of forest removal that are more akin to clearcutting. The road reconstruction component alone, under this single massive project, would not only reverse recovery, but also increase road impacts on about 4% of the entire road network on all USFS lands in the Sierra Nevada. If implemented, it would also increase the length of the total road network on these lands by about 1.6%. The project, as proposed, would increase existing disturbance levels by an average of about 30% over the entire project area, which included hundreds of smaller watersheds (USFS, 1999). At the scale of individual affected watersheds, disturbance levels would be more than doubled by the project (USFS, 1999).

Extensive postfire MFT have been increasingly proposed since 2000, at scales without antecedents. For instance, ASNF (2004) proposed to log more than 40,000 acres in Arizona with postfire fuel reductions as one of the objectives. Before settlement of litigation by citizens, BNF (2001) proposed to log about 44,000 acres in Montana, citing postfire fuel reduction as a primary aim. In the area burned by the 2002 Biscuit fire, RSNF (2004) proposed to log more timber volume than logged in the previous year on all national forests in all of Oregon and Washington, with fuel reduction cited as an objective.

At the scale of individual large watersheds, there is a high degree of certainty that there will be attempts to subject a significant amount of watersheds to disturbance by MFT to reduce fuels. For instance, SFNF (2004a; b) proposed to disturb soils and remove trees and other vegetation on 28% of the area under public ownership in a municipal watershed in New Mexico, equating to about 16% of the total municipal watershed area.

At an intermediate scale, which may affect multiple watersheds, there is also a high degree of certainty that fuel treatments will disturb a significant amount of a given area. Graham et al. (2004) cite modeling that indicates that about 50-60% of the landscape area may need to be treated for fuel reductions in order to modify fire behavior if the treatments are not strategically placed. However, even with strategic placement of fuel treatments, models indicate that at least 20% of the landscape area must treated to modify fire behavior (Finney, 2003; Graham et al., 2004).

Most of the foregoing estimates of areas disturbed by fuel treatments do not include the area of associated severe disturbances from the increased use, construction, reconstruction and maintenance of roads and landings, which are typical elements of MFT. These additional disturbances are far from trivial due to their persistence and severity.

For instance, landing construction has impacts that are similar in magnitude and persistence to those from road construction on a per unit basis (Menning et al., 1996; Beschta et al., 2004). Landings are typically constructed on about 2% of the area of fuel treatments by tree removal (ENF, 2004a; b; c). Assuming that landings occupy 2% of the area treated by MFT, the level of fuel treatments proposed under USFS (2004) would result in the construction of about 14,400 acres of landings over the course of a decade. This is roughly equivalent to building more than 4,700 miles of road with an average width of 25 ft. This plainly corroborates the high degree of certainty that fuel treatments will affect a significant amount of an area, in a manner that persistently and significantly contributes to cumulative effects on soils, watersheds, and aquatic systems. It also indicates that the actual area intensively disturbed by fuel treatments is likely to be about 2% higher than the area solely subjected to tree removal.

There is a high degree of certainty that fuel treatments involve repeated entries for repeated treatments on the same area or sequential treatments of different areas. Allen et al. (2002) recommended repeated entries for staggered, piecemeal implementation of fuel treatments, which intrinsically involves repeated entries into landscapes.

Repeated treatments are clearly required to maintain reduced fuels, due to the transience of treatment effects (Graham et al., 2004; Kauffman, 2004). For this reason, Agee and Skinner (2005) suggested repeating treatments at intervals of 10-20 years. Fuel treatments that open forest canopies, such as significant thinning or “fuel breaks,” can create a self-perpetuating need for repeated treatment due to their effect on vegetation regrowth (Baker et al., 2006). Without repeated entries, post-treatment vegetation and fuel conditions can be more conducive to increasing fire spread and severity than before treatment (Keeley, 2001; 2002; Kauffman, 2004). Fuel breaks are estimated to require repeated entries, on the order of every 10-20 years (USFS, 1999; Rogue River-Siskiyou National Forest (RSNF, 2004).

There is high degree of certainty that repeated entries for fuel treatments as part of MFT increase the scale of cumulative effects and effective level of disturbance. This is because the effects of initial treatments often do not completely subside before the effects of subsequent treatments are superimposed on watershed systems, resulting in increased chronic cumulative disturbance that deleteriously affects aquatic communities (Ziemer, 1991; Ziemer et al., 1991; Abbruzzese and Leibowitz, 1997). For instance, if 20% of a watershed is subjected to repeated fuel treatments every 20 years for 100 years, this equates to the level of disturbance that is akin to that from completely treating an entire watershed over 100 years. This level of disturbance is generally acknowledged to cause significant adverse cumulative effects on watershed and aquatic resources over time (e.g. Ziemer, 1991; Ziemer et al., 1991; Murphy, 1995).

Persistent and chronic aquatic impacts from persistent, repeated watershed disturbances may be more deleterious for native fish than infrequent, but acute, impacts, such as fire and its watershed effects (Rieman et al., 2003; Dunham et al., 2003b). Therefore, the persistent effects generated by the watershed impacts of repeated treatments likely increase the cumulative biotic effects on aquatic ecosystems.

Mechanized Fuel Treatment in Areas Important to the Protection and Restoration of Watershed and Aquatic Resources

While there are no unimportant, expendable parts of landscapes and watersheds, there has been a growing agreement over the past 20 years that some watershed areas with particular physical or biological features are essential for the protection and restoration of aquatic systems. For instance, numerous assessments and studies have concluded that the following are critically important to fully protect and restore in order to aid in the restoration of aquatic systems:

- riparian areas (e.g. USFS et al., 1993; CWWR, 1996; National Research Council (NRC), 1996; 2002; Beschta et al., 2004);
- roadless areas, including those greater than 1000 acres in area (e.g., Henjum et al., 1994; Rhodes et al., 1994; May, 2000; Kessler et al., 2001; Beschta et al., 2004; Karr et al., 2004);
- watersheds with imperiled aquatic species, including fringe populations in fringe habitats (e.g., Rhodes et al., 1994; USFS and USBLM, 1997a; b; Kessler et al., 2001; Beschta et al., 2004; Karr et al., 2004);
- relatively undamaged watersheds that have relatively higher quality habitat and/or water quality, the most potential for restoration, relatively healthy populations of aquatic biota, and/or high aquatic biodiversity (USFS et al., 1993; Henjum et al., 1994; USFS and USBLM, 1997a; b; Pacific Rivers Council, 1996; ISG, 1999; ECONorthwest and Pacific Rivers Council, 2002);
- the most historically productive stream habitats (e.g., Rhodes et al., 1994; Propst and Stefferud, 1997; ISG, 1999);
- stream corridors that, if restored, can ultimately provide connectivity between fragmented aquatic populations (e.g., Henjum et al., 1994; Propst and Stefferud, 1997; Gresswell, 1999; ISG, 1999; Kessler et al., 2001; Rieman et al., 2003; Dunham et al., 2003b).

Overall, there is a high degree of certainty that fuel treatments will deleteriously disturb these areas that are critical to aquatic restoration efforts. A primary reason for this determination is the empirical evidence from recent proposals for MFT. The lack of enforceable provisions protecting these areas bolsters the level of certainty in this regard (Espinosa et al., 1997).

Riparian Areas

There is considerable empirical evidence that MFT will be implemented in riparian areas in close proximity to streams. While a full catalog of planned and implemented MFT projects on public lands in the West is well beyond the scope of this report, recent examples of MFT projects that would occur in close proximity to riparian and stream systems are USFS (1999) and SNF (2001) in California; CNF (2002) and NPNF (2002) in Idaho; ONF (2002), UNF (2001), and RSNF (2004) in Oregon; and, SFNF (2004a) in New Mexico. RSNF (2004) proposed to create copious amounts of fuel breaks within 50 feet of streams. The empirical evidence provided by these numerous proposed projects confers a high degree of certainty that MFT will be located in riparian areas, damaging aquatic resources in an enduring manner.

Forest plans for areas with imperiled native trout habitat in the inland West that are outside of the range of bull trout and anadromous fish lack adequate protection of riparian areas, as the USFS's own assessments have noted (May, 2000). Due to the impetus for MFT, recent national forest land management plans specifically allow riparian areas to be damaged by fuel treatment activities. For instance, the recently adopted forest plans for national forests in southern California allow complete deforestation and removal of all large downed wood in riparian areas in zones prioritized for fuel reduction measures (USFS, 2005). Such impacts are assured to cause severe and persistent riparian, watershed, and aquatic damage.

While riparian protection measures are less inadequate within the area of public lands managed under the aegis of the Northwest Forest Plan (NWFP) (USFS et al., 1993), PACFISH (USFS and USBLM, 1995a) and INFISH (USFS and USBLM, 1995b), they are still inadequate. The riparian protections in these three land management schema have numerous deficiencies, but the primary one is inadequate protection of smaller perennial and non-perennial streams. These streams are extremely sensitive to disturbance, comprise the bulk of the stream network, and cumulatively exert an extremely strong control on downstream aquatic conditions, which makes them crucial to protect if downstream conditions are to be protected (Rhodes et al., 1994; Moyle et al., 1996b; Allen and Dietrich, 2005).

For instance, USFS et al. (1993), USFS and USBLM (1995a; b) indicate that a protected area with a width of at least about 300 feet from each side of a stream is needed to protect aquatic resources from the impacts of upslope disturbance, but both provide less than half this protected width to non-perennial streams. Because of their importance and sensitivity, smaller non-perennial and headwater streams need to receive as much *or more* protection than larger streams if aquatic resources are to be protected (Rhodes et al., 1994; Moyle et al., 1996b; USFS and USBLM, 1997a). The lack of adequate riparian protection contributes to the high degree of certainty that fuel treatments will occur in riparian areas, even those that are offered some protection from forest management activities.

Damage to headwater streams and riparian areas not only degrades habitats in headwater streams, but downstream habitats as well, because headwater streams provide most of the water and sediment for downstream reaches (Rhodes et al., 1994; Moyle et al., 1996b; Erman et al., 1996). They also exert a strong control on downstream water temperature in salmonid habitats (Allen and Dietrich, 2005). Based on the data from Jackson et al. (2002), inadequate riparian protection damages amphibian habitat in headwater streams.

Roadless Areas

There is a high degree of certainty that MFT will be proposed in roadless areas. BNF (2001), SNF (2001) and RSNF (2004) all proposed to reduce postfire fuels in inventoried and uninventoried roadless areas in Montana, California, and Oregon, respectively. SFNF (2004a) aimed to treat fuels in inventoried roadless areas in New Mexico. The scale of roadless area entry is considerable in all of these proposals. RSNF (2004) proposed to reduce fuels by logging more than 12,000 acres of inventoried roadless areas. BNF (2001) originally aimed to do the same in almost 17,000 acres of uninventoried roadless areas, before citizen participation scaled that back by about 88%. SFNF (2004b) proposed to reduce fuels via a variety of treatments in about 4,100 acres of inventoried roadless areas; about 47% of the total project was proposed to occur in inventoried roadless areas (SFNF, 2004a).

As of October 2006, the so-called “Roadless Rule” (USFS, 2000b) had been re-instated (Harden and Eilperin, 2006). However, the rule may not protect roadless areas from damage by MFT, because it did not protect uninventoried roadless areas less than 5,000 acres from roads and logging associated with fuel treatments (USFS, 2000b). Scientific assessments have repeatedly concluded uninventoried roadless areas less than 5,000 acres from roads and logging associated with fuel treatments (USFS, 2000b). Scientific assessments have repeatedly concluded roadless areas greater than 1,000 acres in size are critical to protect in order to protect and restore native salmonids (e.g., Henjum et al., 1994; Rhodes et al., 1994).

The Roadless Rule (USFS, 2000b) also provided explicit allowances to conduct MFT in inventoried roadless areas. Although USFS et al. (1993), USFS and USBLM (1995a), and USFS (2001) clearly noted the importance of remaining roadless areas to aquatic protection and restoration efforts, none of these public land management schemes protected roadless areas, inventoried or not, from damage by MFT. Together with the proposals for MFT in roadless areas, the lack of roadless area protection contributes to the likelihood that they will be entered for MFT.

The "Roadless Rule" has also been in limbo and flux (Frick, 2004). It may again be in flux, depending on the outcomes of legal actions challenging the rule (Associated Press, 2006).

Areas With Imperiled Aquatic Species or High Restoration Potential

There is a high degree of certainty that MFT projects will be implemented or attempted in watersheds with imperiled aquatic biota, high restoration potential, or high aquatic biodiversity. A few recent examples of planned or attempted MFT projects in such areas are SNF (2001), UNF (2001), BNF (2001), CNF (2002), ONF (2002), NPNF (2002), and RSNF (2004). All of these projects proposed MFT for fuels reductions in watersheds that provide habitat for imperiled salmonids, including several listed under the Endangered Species Act (ESA).

UNF (2001) proposed mechanized fuel reduction, including the construction of fuel breaks, in Oregon in one of the most important watersheds for the production of wild anadromous salmonids remaining in the entire Columbia River system. The watershed provides habitat for steelhead and bull trout listed under the ESA.

CNF (2002) targeted watersheds that have some of the most productive remaining populations of wild steelhead in the Snake River portion of the upper Columbia River system. These watersheds also provide habitat for imperiled westslope cutthroat trout, threatened bull trout, and threatened chinook salmon. NPNF (2002) targeted watersheds that provide habitat for imperiled cutthroat trout and listed bull trout, chinook salmon, and steelhead.

RSNF (2004) proposed to reduce fuels via logging in watersheds with imperiled steelhead and chinook salmon and listed coho salmon. These watersheds also currently provide an important source of relatively high water quality and have extremely high potential for restoration, if fully protected from additional damage.

These projects are but a few out of many of a similar ilk. They amply demonstrate the high degree of certainty that MFT will be proposed for implementation in ecologically important areas that are vital to the protection and restoration of aquatic ecosystems.

Currently there is no adopted land management scheme that assures that such areas are fully protected, even though there have been calls to do so for more than a decade (e.g., Rhodes et al., 1994; Henjum et al., 1994; ISG, 1996). This bolsters the high degree of certainty that MFT will continue to be proposed in areas with imperiled aquatic species.

High-Hazard Areas

There is general scientific agreement that areas with certain sets of attributes are likely to lead to disproportionately significant degradation of aquatic systems if subjected to the suite of disturbances involved in MFT. These areas include:

- steep slopes (USFS et al., 1993; USFS and USBLM, 1997a);
- thin soils (Rhodes et al., 1994; CWWR, 1996);
- soils with high erosion hazards, including landslide prone areas (USFS et al., 1993)riparian areas (USFS et al., 1993; Beschta et al., 2004);
- zones subject to rain-on-snow events (MacDonald and Ritland, 1989; USFS et al., 1993);
- areas recently burned at moderate to high severity (Beschta et al., 2004; Karr et al., 2004);
- watersheds where cumulative effects are already pronounced under existing levels of disturbance (Rhodes et al., 1994; Henjum et al., 1994).

There is a high degree of certainty that MFT will disturb such areas. This level of certainty is due to the empirical evidence from recent proposals and bolstered by lack of protective measures for these areas in land management plans.

USFS (1999), SNF (2001), BNF (2001), CNF (2002), NPNF (2002), ONF (2002), and UNF (2003) all targeted watersheds where negative cumulative effects on aquatic resources were already extremely significant. All of these projects proposed fuel treatments that would increase erosion and sediment delivery in watersheds where sedimentation was already documented to be a severe problem.

Ridgelines are often targeted for fuel breaks to modify fire behavior as part of MFT (USFS, 1999; SNF, 2000; UNF, 2001; RSNF, 2001). Ridgelines typically have relatively thin soils, increasing severity of damage from soil loss and other soil impacts.

CNF (2002) proposed to conduct MFT on areas known to be prone to mass wasting. BNF (2001), UNF (2003), SFNF (2004a), ASNF (2004) and RSNF (2004) all proposed fuel reduction via logging on soils known to have high erosion hazards, if disturbed. BNF (2001), SNF (2001), and RSNF (2004) proposed extensive logging to reduce fuels on areas that had recently been burned at high severity. ASNF (2004) proposed ground-based logging to reduce fuels on more than 40,000 acres that primarily had been burned at high or moderate severity.

USFS (1999) proposed extensive logging to reduce fuels in watersheds subject to relatively frequent rain-on-snow events in California. RSNF (2004) proposed logging to reduce fuels in areas with exceedingly thin soils subject to rain-on-snow events. It also proposed logging to reduce fuels on steep slopes: more than 21,000 acres on slopes greater than 30% and more than 6,700 acres on slopes over 60% (Pers. comm., E. Fernandez, GIS specialist, ONRC, January, 2004).

The Ecological Costs of Mechanized Fuel Treatments: Damage to Watershed/Aquatic Attributes and Processes

Mechanized Fuel Treatment Effects on Soil Productivity

Soils are a fundamental element of forested ecosystems. Soil conditions strongly influence long-term forest productivity, the composition and condition of vegetation, rates of vegetative recovery after disturbance, and the quantity, timing, and quality of water produced by watersheds (Beschta et al., 2004). Soil and vegetation conditions profoundly affect sediment flux to streams, which, in turn, affects aquatic ecosystems.

There is a high degree of certainty that MFT activities will reduce soil productivity in an enduring fashion through several mechanisms, including: reductions in sources of organic matter and nutrient capital; soil compaction and consequent effects; soil displacement and disruption; increased erosion; and effects on soil structure.

MFT removes trees, branches, and needles that are the prime sources of organic matter and nutrients vital to long-term maintenance and protection of soil productivity (USFS and USBLM, 1997a; Graham et al., 2004; Beschta et al., 2004; Karr et al., 2004). The removal of this material ultimately leads to persistent losses of soil productivity (Amaranthus and Perry, 1987; USFS and USBLM, 1997a; b; Beschta et al., 2004).

The loss of organic matter from vegetation removal cumulatively reduces the ability of soils to absorb and store water. Soils with higher levels of soil organic matter typically have higher infiltration rates and are able to store more soil moisture (Rawls et al., 1993). Amaranthus et al., (1989) documented that large, decaying and downed logs contain 25 times more moisture than the surrounding soil after fire. Reductions in infiltration rates and the loss of soil water storage capacity both contribute to increased surface runoff and reduced subsurface flow to streams.

Soil nutrient levels are also reduced by the removal of branches and needles, which provide as much as 45% of potassium and 25% of nitrogen stores at the site scale (Graham et al., 1999). This removal is likely to exacerbate nutrient shortages on sites that are short on potassium and nitrogen, a common condition in many forests (Graham et al., 1999).

The removal of whole trees has more intense and persistent negative effects on soils than fire (USFS and USBLM, 1997b). The USFS and USBLM (1997b) concluded that wildfire usually has fewer and less persistent negative impacts on soil productivity than the removal of whole trees, due to the patchy nature of wildfire, the residual wood left on site, and the lack of soil compaction.

Many MFT include the removal of native shrubs (sometimes referred to as “brush”) as part of the effort to reduce surface fuels. This removal has been recommended as part of MFT, in order to reduce surface fuels (e.g., Agee and Skinner, 2005). The removal of shrubs reduces sources of organic matter. It also reduces nutrient levels by reducing levels of nitrogen fixation, because several types of native shrubs in the West are nitrogen-fixers (Rhodes, 1985). A 35-year study (Busse et al., 1996) of the effect of brush control on ponderosa pine showed that complete brush removal did not increase the growth of ponderosa pines older than 20 years old, while soils with retained brush had higher soil productivity due to much higher levels of soil nitrogen and carbon in the upper several inches of the soil than under soils where brush had been removed. Busse et al. (1996) concluded, “A long-term benefit to upper soil horizons is associated with maintaining understory vegetation.” Conversely, the study indicates that brush removal, as will occur with MFT aimed at reducing surface fuels, will have long-term costs to soil productivity in upper soil horizons that are most important to overall soil productivity.

MFT also impedes the recovery of degraded soil productivity. One of the most effective steps to restoring soil productivity is to retain all sources of wood recruitment to soils and to leave areas undisturbed until they have recovered (Kattleman, 1996; USFS and USBLM, 1997a; Beschta et al., 2004). MFT and associated activities conflict with this approach because they remove trees, shrubs, and groundcover, while disturbing and compacting soils.

Numerous activities associated with MFT increase the extent and intensity of soil compaction. The effects of compaction on soils are more persistent than the baring of soil by fire (USFS and USBLM 1997b). Soil compaction persists for 50-80 years or longer and persistently reduces soil productivity (USFS and USBLM, 1997a; Beschta et al., 2004). Regional assessments of conditions on large tracts of public lands have concluded that soil compaction is a significant concern on many national forests due to thin soils, the longevity of the impact, the existing extent of soils already compacted by logging, grazing and roads, and the impacts from on-going and likely future ground-disturbing activities (e.g., CWWR, 1996; USFS and USBLM, 1997a).

Research in northern Idaho (Page-Dumroese et al., 1998) indicates that significantly compacted soils have poor prospects for recovery. Ground-based logging compacts soils to levels that significantly reduce tree growth (Page-Dumroese et al., 1998). Soil compaction also reduces the ability of soils to absorb and store water (Rawls et al., 1993). Many types of MFT are likely to compact soils as severely as logging, because the same types of practices often will be used, although fuel treatment may sometimes target different trees than conventional logging.

It is extremely likely a significant amount of MFT will be done with ground-based machinery. Yarding methods that cause less on-site soil damage, such as helicopter yarding, do not meet fuel reduction objectives, because they generate relatively high levels of flammable slash (Halpern and McKenzie, 2001; Agee and Skinner, 2005; Donato et al., 2006). For these reasons, Agee and Skinner (2005) explicitly recommended using ground-based machinery to reduce fuels, despite the well-known ecological costs of this practice (Beschta et al., 2004).

Economic considerations also contribute to the propensity to use ground-based machinery to accomplish MFT. Ground-based yarding methods are the least costly (Rummer et al., 2003).

Ground-based machinery is typically proposed as the primary means of implementing MFT. Examples of fuel reduction projects employing ground-based machinery include USFS, (1999), UNF (2001), ONF (2002), SFNF (2004a), ENF (2004a; b; c), and ASNF (2004). The latter proposed to conduct logging on more than 40,000 acres, solely using ground-based machinery, with postfire fuel reduction as one of the stated objectives. This empirical evidence amply demonstrates that MFT will employ ground-based machinery. There is a high degree of certainty that soil compaction and disruption are inevitable with ground-based machinery (Geppert et al., 1984; Kattleman, 1996; USFS and USBLM, 1997a; b; c; Beschta et al., 2004).

There is a high degree of certainty that MFT will accelerate topsoil erosion through the combined impacts of soil compaction and removal of soil cover. Topsoil loss causes the most persistent and serious loss of soil productivity (USFS and USBLM, 1997a; Beschta et al., 2004). This is especially serious in areas where topsoil layers are thin and rates of soil formation are exceedingly slow, as is the case in most forested areas in the interior West (CWWR, 1996; USFS and USBLM, 1997a). The loss of topsoil is irreversible within human timescales; associated reductions in soil productivity are essentially permanent (Beschta et al., 2004; Karr et al., 2004).

Topsoil loss contributes to reductions in the capacity of watersheds to absorb, store, and slowly release water to streams. For instance, the loss of only one inch of topsoil over one square mile of watershed translates into the loss of over one million cubic feet of water storage capacity in watershed soils. Topsoil loss can also reduce infiltration rates, since surface soils typically have the highest infiltration rates (Rawls et al., 1993). Both impacts typically translate into increased surface runoff and contribute to reductions in low flows (Hancock, 2002).

Many aspects of MFT and associated activities increase topsoil erosion. SFNF (2004a) estimated that 8,800 acres of fuel treatments in a municipal watershed would increase rates of soil erosion more than four-fold. Notably, SFNF (2004a) ignored accelerated erosion caused by broadcast burning, piling and burning, and elevated road use, all of which significantly increase soil erosion (Foltz, 1996; Megahan et al., 1995; Kauffman, 2004). A more than four-fold increase in rates of surface erosion would have significant and persistent negative effects on soil productivity, watershed hydrologic processes, water quality, and stream conditions (Rhodes et al., 1994).

Intensive thinning, which is a likely component of MFT to reduce fuels, can involve the cutting and yarding of more trees per unit area than conventional logging. This increases the area of soil disturbance from yarding and felling per unit area affected. Therefore, soil damage from intensive thinning is likely to be as great as or greater than that from conventional logging. Based on similar logistical considerations, Geppert et al. (1984) concluded that intensive thinning with ground-based machinery likely causes greater soil damage per unit treatment area than the conventional clearcutting of large trees.

Compaction, topsoil loss, accelerated erosion, and loss of organic matter sources are especially severe with the construction of roads and landings, because vegetation and groundcover are completely removed, erosion is dramatically and persistently increased, and compaction is severe (Geppert et al., 1984; Beschta et al., 2004). MFT often involves the construction of roads and/or landings (e.g., USFS, 1999; BNF, 2001; SNF, 2001; UNF, 2003; CNF, 2002; RSNF, 2004; ASNF, 2004; ENF, 2004a; b; c).

Roads are typically the single largest source of elevated erosion in forested watersheds. Landing construction and use involves impacts to soils and vegetation that are as severe and persistent as those from roads, resulting in similar effects on watershed hydrology, erosion and sediment delivery (Geppert et al., 1984, Ketcheson and Megahan, 1996). In their study in Idaho forests, Megahan and Ketcheson (1996) found that the longest travel distance of sediment from forest disturbances originated from a landing. Cumulative effects methods indicate that landings contribute to adverse watershed cumulative effects as persistently and significantly as roads (Menning et al., 1996).

Erosion on roads is especially high during the first year after construction (Rhodes et al., 1994). However, it remains dramatically elevated as long the roads exist, and even well after abandonment or decommissioning (Potyondy et al., 1991; Rhodes et al., 1994; USFS, 2000b).

Roads and landings essentially zero out soil productivity for some time and reduce it for long periods thereafter (Geppert et al., 1984; Menning et al., 1996). This is the case even with “temporary” roads and landings. Due to the persistence of their impacts, “temporary” landings and roads do not have temporary impacts (Beschta et al, 2004). The negative effects of road and landing construction are large, enduring, and immediate, while recovery is relatively minor and protracted, even with obliteration, all of which belie any application of the term "temporary" (Beschta et al., 2004). The USFS has conceded that the loss of soil productivity on temporary landings and roads is not reversible, because such areas never completely regain their productivity or function naturally even with remediation or abandonment (BNF, 2001; RSNF, 2003).

The degree of soil compaction on roads and landings retards vegetative recovery and vastly elevates surface erosion for decades after abandonment (Rhodes et al., 1994). It also significantly reduces the ability of affected soils to absorb and store water. Roads reduce infiltration rates by about 97% relative

<u>Photograph 1</u>. An example of an ineffective Best Management Practice (BMP) that failed to significantly reduce major impacts of land use on aquatic systems on the Bitterroot National Forest, MT. High levels of fine sediment from road impacts continue to be funneled directly to this stream via road drainage features, despite the recent addition of rock to the stream crossing in 2005. The stream is in the East Fork of the Bitterroot River watershed, which is designated as having impaired water quality due to sedimentation. This watershed provides habitat for bull trout, listed as threatened under the Endangered Species Act, and imperiled western cutthroat trout. Increases in fine sediment significantly reduce the survival of both these native trout, as discussed in the text. *Photograph: G. Carnefix*.

to undisturbed areas (Luce, 1997), causing surface runoff to be generated by even minor rain and snowmelt events.

The area likely to be affected by landings constructed as part of MFT is far from trivial if MFT are extensively implemented. Assuming landing construction occurs on 2% of treated areas and the levels of fuel reduction recommended by Stephens and Ruth (2005) for USFS lands in the Pacific Northwest and California were conducted mechanically, the likely level of landing construction would be more than 480,000 acres. This affected area is roughly equivalent to building almost 160,000 miles of road with an average width of 25 ft. As noted previously, the area of landings for the proposed magnitude of mechanized fuel treatments on 11 Sierra Nevada nationalforests (USFS, 2004) would be roughly equivalent to building more than 4,700 miles of road with an average width of 25 ft. Due to the severity and persistence of the impacts of landings, these levels of landing construction would contribute significantly to the degradation of watersheds and aquatic systems.

There is a high degree of certainty that MFT will involve the construction of roads, including "temporary" ones. Examples of projects that proposed road construction as part of MFT include USFS (1999), BNF (2001), SNF (2001), NPNF, (2001); CNF, 2002; ONF (2002), SFNF (2004a); and RSNF (2004). Agee and Skinner (2005) recommended the construction of temporary roads to facilitate MFT in roadless tracts.

The reconstruction of roads and/or landings is a typical aspect of MFT (e.g. BNF, 2001; SNF, 2001; CNF, 2002; NPNF, 2002; SFNF, 2004, RSNF, 2004). There also is a high degree of certainty that road reconstruction significantly increases erosion, especially when the roads have previously undergone some degree of hydrologic recovery through non-use (Potyondy et al., 1991; Beschta et al., 2004; Karr et al., 2004). Reconstruction of unused landings and roads also effectively reverses recovery of soils and soil processes that have occurred in the absence of use (Beschta et al., 2004; Karr et al., 2004). (See Photographs 2a and 2b on pg. 20.)

Elevated road use is a typical part of MFT that significantly increases surface erosion on unpaved roads (Wald, 1975; Reid et al., 1981; Reid and Dunne, 1984; Foltz, 1996; Luce and Black, 2000; Gucinski et al., 2001; Ziegler, 2001; Luce and Black, 2001). The USFS's summary of scientific information on roads (Gucinski et al., 2001) concluded that "rates of sediment delivery from unpaved roads are…closely correlated to traffic volume." Reid et al. (1981) documented that roads used by more than four logging trucks per day generated more than seven times the sediment generated from roads with less use and more than 100 times the sediment from abandoned roads.

Even with a road surface of crushed rock aggregate, which is often used with the intent of reducing sediment production on road surfaces, Foltz (1996) documented that elevated truck traffic increased sediment production by 2 to 25 times that on unused roads in western Oregon. Foltz (1996) noted that since the processes are the same across regions, a similar range of increases was likely. Primary mechanisms for increased erosion and sediment production from road use are the production of highly mobile fine sediment on road surfaces, road prism damage, disruption of gravel or aggregate surfaces, and rutting.

The effect of road use on surface erosion is magnified by use during wet periods. Wet weather haul causes rutting, documented by USFS research to increase sediment delivery from surface erosion on roads by about 2-5

Photographs 2a & 2b. An example from Karr et al. (2004) of some of the typical impacts of the reconstruction of unused roads on vegetation, soils, road runoff, and sediment production. The left photo (2a) shows an unused road in 1996 on the Malheur National Forest, OR, which had undergone some recovery of vegetation and soil conditions through non-use, resulting in revegetation and reduced erosion and runoff. The right photo (2b) shows how reconstruction of the same road for MFT had reversed this recovery in 1999, increasing soil erosion and sediment delivery by surface runoff to a tributary of a stream inhabited by steelhead trout (listed as threatened under the Endangered Species Act at the time of photo). Such increases in sediment delivery lower the survival rates of steelhead and other aquatic species and degrade water quality, as discussed in the text. ***Photographs: J. Rhodes.***

times that occurring on unrutted roads (Burroughs, 1990; Foltz and Burroughs, 1990). Gucinski et al. (2001) noted, "As storms become larger or soil becomes wetter, more of the road system contributes water directly to streams."

Burroughs (1990) concluded that road closure during wet weather is one of the most important measures to reduce sediment production from roads and damage to roads. However, MFT will very likely include elevated road use during wet weather, as evidenced by the allowances to do so in several projects (USFS, 1999; ENF, 2000a; b; c; RSNF, 2004).

Increased road use typically requires increased road maintenance. Road maintenance to facilitate haul is a common component of MFT proposals (e.g., USFS, 1999; BNF, 2001; ENF, 2000a; b; c; CNF, 2002; RSNF, 2004). It increases soil loss by removing vegetation and disturbing road prisms and ditches (RSNF, 2004). Black and Luce (1999) found grading of roads elevated sediment production for at least a year. Luce and Black (2001) documented that ditch maintenance also elevated erosion.

The several-fold increases in erosion and sediment delivery caused by road and landing reconstruction and elevated use are significant. Even without these increases in sediment delivery, roads are usually the primary source of management-induced sediment delivery in managed watersheds (Furniss et al., 1991; USFS et al., 1993; CWWR, 1996; Gucinski et al., 2001).

There is a high degree of certainty that road impacts from MFT will be extensive and involve significant road mileage. MFT is likely to involve repeated entries for dispersed and extensive treatments. Such activities require the perpetuation, use, and maintenance of an extensive road network. Road impacts increase with increased frequency of entry for MFT. As previously noted, it is extremely likely that MFT will involve repeated entries over relatively short time spans.

MFT typically involves broadcast burning and machine piling and burning in order to reduce activity fuels, commonly referred to as slash, and other surface fuels, such as native shrubs. Projects that have included these practices include SNF (2001); UNF (2002); SFNF, (2004a); RSNF (2004); ENF (2004a; b; c); ASNF (2004).

There is a high degree of certainty that broadcast and pile burning reduces soil productivity. It does so by heating and denuding soils, removing sources of organic matter, increasing erosion, and sometimes causing hydrophobic soils to develop.

In a study area with soils prone to erosion, Megahan et al., (1995) documented that areas broadcast burned after helicopter logging in Idaho had surface erosion rates that were 66 times those on undisturbed slopes. The broadcast burned areas also had copious amounts of bare soils, even 10 years after initial treatment. The helicopter logging and broadcast burning nearly doubled sediment yield from a small watershed for a period of 10 years, indicating that combined impacts significantly elevated soil export from the watershed in an enduring fashion. Megahan et al., (1995) concluded that the broadcast burning "…had potentially serious implications for on-site productivity" due to the magnitude and duration of the impacts. For these reasons, Megahan et al. recommended avoidance of broadcast burning in areas where soil loss was a significant concern.

Although broadcast burning is often assumed to have nominal effects on soil conditions, available information indicates that it involves high severity burns and sometimes causes reduced infiltration rates and the development of hydrophobic soils. Robichaud (2000) documented that broadcast burning in Montana created hydrophobic soils that reduced infiltration rates by about 10-40%. About 28% of the sampled area showed signs of hydrophobicity. High severity burns occurred on 5% of one site and 15% of another. Hydrophobic soils temporarily increased runoff and soil erosion for one to two years (Robichaud, 2000). Debyle (1973) also documented the development of hydrophobic soils from post-logging broadcast burning.

Notably, broadcast burning of materials felled by MFT may result in higher fire severity than occurs from post-logging broadcast burning. This is because MFT may provide even more downed, fine fuels for broadcast burning, if felled material is left in place. These felled materials would ostensibly be comprised of the most flammable fuels, since they were targeted for thinning and, hence, could contribute significantly to elevated fire severity if fire occurs. Further, the burn would occur under a canopy that had been opened up, contributing to increased wind speeds and reduced fuel moisture, both of which are conducive to elevated fire severity. Thinning with untreated surface fuels has been shown to increase wildland fire severity (Raymond and Peterson, 2005). Therefore, broadcast burning as a follow-up to MFT with felled materials in place may result in negative effects on soils and surface erosion that are as great as or greater than those from broadcast burning after conventional logging.

Prescribed fire does not usually increase topsoil loss significantly or persistently because it often burns primarily at lower severity. However, prescribed burns that are often used as part of MFT occasionally burn hotter and over greater areas than expected. The Cerro Grande fire in New Mexico started as a prescribed burn; in parts of the area burned at high severity by the fire, postfire erosion was vastly accelerated (Allen et al., 2002).

Machine piling of slash and other fuel materials generated by MFT has severe impacts on soil productivity via compaction, soil disruption, and removal of vegetation and groundcover. On a per unit area basis, the soil disturbance caused by impacts of machine piling is only rivaled by the construction of roads and landings (Geppert et al., 1984; Menning et al., 1996). Data from a study of the impacts of several logging treatments with significant tree retention in the Pacific

Photograph 3. Machine piled area, Malheur National Forest Oregon, approximately 2.5 years after being burned as part of mechanized fuel treatment. Soil has been severely damaged by prolonged exposure to high temperatures under the pile. Note that piled topsoil ringing the scar was moved there by the machine piling. The only significant revegetation occurring on the burn scar is by invasive weeds at the outer edge of the burn. Compare and contrast this severe soil damage and lack of revegetation approximately 2.5 yrs after pile burning to the rapid revegetation after high-severity wildland fire about one year after fire on the Eldorado and Sequoia National Forests, CA, in Photos 4-6. ***Photograph: J. Rhodes.***

Northwest indicate that machine piling treatments had the highest level of disturbed soil of any treatments studied (Halpern and McKenzie, 2001). Machine piling and burning left little ecologically important coarse woody debris onsite and significantly reduced groundcover (Halpern and McKenzie, 2001).

Machine piling also reduces soil productivity by extensively displacing topsoil. Much of this displaced topsoil ends up underneath the piles, where it is exposed to high temperatures when piles are burned. This displacement of topsoil to areas where it is subjected to prolonged burning at high temperatures greatly increases the impact of piling and burning on soil productivity (See Photograph 3 on pg. 22).

Pile burning removes organic matter and nutrients and sterilizing soils beneath the piles (Kauffman 2004; Korb et al., 2004). The soil damage under burned piles is so intense and enduring that burn scars often remain persistently unvegetated or occupied only by exotic, and often invasive, weeds (Korb et al., 2004), as shown in Photograph 3. Slash disposal by any means removes the sources of organic matter critical to soil productivity (Amaranthus and Perry, 1987; USFS et al., 1997a).

Project proposals demonstrate that MFT often includes significant amounts of machine piling and burning (SNF, 2001; UNF, 2002; SFNF, 2004a; RSNF, 2004; ENF, 2004a; b; c; ASNF, 2004). For instance, MFT in the Sierra Nevada have called for about 54% to 100% of the areas treated for fuel reduction to be machine piled and burned.

Postfire Mechanized Fuel Treatments

Postfire fuel reduction treatments have multiple negative impacts on soils, which are of heightened concern due to the inherent sensitivity of soils after fire (Beschta et al., 2004; Karr et al., 2004: Noss et al., 2006a; b). These treatments reduce large woody debris recruitment critical to soil productivity. They also directly damage soils via displacement, compaction, groundcover removal, and elevated erosion.

Although it is outside of the scope of this report, it is worth noting that there is a high degree of certainty that postfire fuel reduction treatments have significant negative ecological impacts on a wide variety of non-aquatic processes and resources (Lindenmayer et al., 2004; Beschta et al., 2004; Franklin, 2005; Hutto, 2006). Some imperiled avian species are almost entirely dependent on untreated, burned forests (Hutto, 2005; 2006). Due to their ecological importance, Franklin (2005) concluded that the removal of burned and dead trees often has greater ecological costs than the logging of live trees. A burgeoning body of scientific work has noted that postfire tree removal has negative impacts that undermine the recovery of burned forest ecosystems (Beschta et al., 2004; Karr et al., 2004: Noss et al., 2006a; b; DellaSala et al., 2006).

In aggregate, this information clearly indicates that there is a high degree of certainty that MFT is likely to involve road and landing impacts, anthropogenic burning, and vegetation removal with ground-based machinery. There is also a high degree of certainty that these impacts will cumulatively reduce soil productivity by reducing sources of soil organic matter and soil nutrients, compacting soils, burning soils, and increasing topsoil erosion.

Mechanized Fuel Treatment Effects on Erosion and Sediment Delivery to Aquatic Systems

There is a high degree of certainty that MFT will increase erosion and sediment delivery to stream systems with consequent negative impacts on water quality. This is due to the activities involved, their likely extent and frequency, and their likely placement within the watershed context. As concluded by Megahan et al. (1992) and USFS and USBLM (1997c) it is not possible to log areas without increasing erosion and sediment delivery, regardless of BMPs involved or care in implementation, especially when roads are involved. MFT involves the same suite of impacts as logging. There is ample evidence that MFT will almost always involve roads and landings.

MFT and associated cumulative impacts will contribute to elevated erosion and sediment delivery via both surface erosion and mass erosion. However, since these mechanisms have differing associated degrees of certainty, they are evaluated separately.

There is a very high degree of certainty that road and landing activities related to MFT will increase surface erosion and sediment delivery, as discussed in the previous section. Landings are extensions of roads and are treated as such in the following discussion.

Much of the erosion from roads is delivered to streams due to direct hydrologic connection via ditches, drainage, and gullies below drainage relief features (Wemple et al., 1996; Rhodes and Huntington, 2000). Both Wemple (1996) and Rhodes and Huntington (2000) found that a sizable proportion of the road network effectively served as extensions of the stream network, effectively increasing drainage density.

The delivery of sediment from roads is particularly acute at stream crossings where road segments drain directly into streams (Kattleman, 1996; Rieman et al., 2003). Road impacts associated with MFT inevitably increase sediment delivery at stream crossings due to their frequency on roads (Rieman et al., 2003). For example, in the forests of the Sierra Nevada, on average, there are about 3.8 stream crossings per mile of road or about one stream crossing per every 0.26 miles of road (USFS, 2001; ECONorthwest and Pacific Rivers Council, 2002). In Wyoming, Eaglin and Hubert (1993) showed that the amount of fine sediment in trout habitat was correlated to the number of stream crossings by roads in a statistically significant manner. In the same study, the amount of trout per stream area was negatively correlated with the number of stream crossings in a statistically significant fashion.

Much of the road network on public land is relatively proximate to streams. For instance, about 43% of all classified roads on the Clearwater National Forest in Idaho are estimated to be within 300 feet of stream channels (CNF, 2003). Under such conditions, there is usually a high degree of direct hydrologic connectivity between roads and streams, regardless of the region and climate (Wemple et al., 1996; Rhodes and Huntington, 2000; CNF, 2003). For example, Rhodes and Huntington (2000) documented that roads occupying the lower one third of slopes in a watershed in Idaho had about 59% of their length hydrologically connected to streams; while roads in similar slope positions in eastern Washington had about 52% of the road length directly connected to streams. Wemple et al. (1996) also found that a high percentage of valley bottom roads in watersheds in western Oregon were directly connected to streams by relief culverts, gullies, ditches, or stream crossings.

The hydrologic connectivity between roads and streams and consequent high levels of sediment delivery is not restricted to stream crossings, nor to valley-bottom roads. In a watershed in eastern Washington, Rhodes and Huntington (2000) found that about 23% of the ridgetop roads directly connected to streams were connected by downslope gullies rather than stream crossings or ditches leading to stream crossings. In a watershed in Idaho, 10% of the midslope roads directly connected to streams were connected by downslope gullies rather than stream crossings (Rhodes and Huntington, 2000).

Both Wemple et al. (1996) and Rhodes and Huntington (2000) found a significant amount of connectivity between streams and roads, even when roads were located on ridgetops. Montgomery (1994) also documented a high level of connectivity between road drainage from ridgetop roads and headwater streams in western Washington.

Studies throughout the West corroborate that elevated erosion from roads triggered by MFT-associated road use will increase sediment delivery and subsequent negative effects on aquatic resources. A large number of studies and reviews have repeatedly documented significant increases in sedimentation, sediment yield, turbidity, and suspended sediment in response to the existence, construction, reconstruction, and use of roads (e.g., reviews and results in: Geppert et al., 1984; Eaglin and Hubert, 1993; Meehan, 1991; MacDonald and Ritland, 1989; Rhodes et al., 1994; Kattleman, 1996; Espinosa et al., 1997; USFS et al., 1997a; USFS 2000; McIntosh et al., 2000).

Based on review of available data, MacDonald and Ritland (1989) concluded that roads typically double suspended sediment yield even with state of the art construction and erosion control. MacDonald and Ritland (1989) also concluded that suspended sediment contributions from roads alone, even in the absence of mass failure, are typically in the range of 5 to 20 percent above background and remain at elevated levels for as long as roads are in use. Kattleman (1996) concluded that BMPs could do little to reduce sediment delivery from roads at stream crossings. Therefore, there is a high degree of certainty that MFT effects on roads will translate into increased sediment delivery and consequent negative effects on fish habitat, water quality, and channel form.

Vegetation removal activities as part of MFT will increase surface erosion, sediment delivery, and resulting negative impacts on aquatic resources. While most studies of the effect of vegetation removal on sediment delivery have focused on traditional logging practices, these findings are relevant because MFT involves largely the same suite of activities, whether as a matter of thinning, creating fuel breaks, or other types of cutting (e.g. partial removal, group selection, etc.). These activities include: felling, ground-based yarding or piling, and, in many circumstances, broadcast burning or piling and burning, as demonstrated by many proposed or implemented MFT projects (e.g., USFS, 1999; SNF, 2001; UNF, 2001; ONF, 2002; SFNF, 2004; ASNF, 2004). Further, logging is often proposed as a primary method of fuel reduction (e.g., USFS, 1999; BNF, 2001; SNF, 2001; UNF, 2001; ONF, 2002; SFNF, 2004; ENF 2004a: b; c; ASNF, 2004).

A multitude of studies across the West have documented that forest removal significantly increases sediment delivery to streams, even when helicopter yarding is used (Megahan et al., 1995). Streams draining watersheds with extensive vegetation removal have higher levels of sedimentation, suspended sediment, and turbidity (Geppert et al., 1984; Eaglin and Hubert, 1993; Meehan, 1991; MacDonald and Ritland, 1989; Rhodes et al., 1994; Kattleman, 1996; Espinosa et al., 1997). In Wyoming, Eaglin and Hubert (1993) found statistically significant evidence that the amount of watershed area logged was correlated to stream sedimentation. Rhodes et al. (1994) found that fish habitat conditions affected by sedimentation were generally poor in watersheds in interior Oregon where more than 15% of the area had been affected by logging.

Empirical evidence indicates that MFT will be proposed in close proximity to streams and in areas with extremely high soil erosion hazards. The hazards posed by these areas are often compounded by several factors, including steep slopes, severely burned soils, *and* erosion-prone soils. Examples of where forest removal for MFT has been proposed on areas with surface erosion hazards include USFS (1999), BNF (2001), SNF (2001), UNF (2002), NPNF (2002), SFNF (2004a), RSNF (2004) and ASNF (2004). The extensive and repeated nature of MFT greatly increases the likelihood that MFT will elevate soil erosion and sediment delivery to aquatic systems.

While vegetation removal does not increase surface erosion and sediment delivery per unit area to the degree that roads do, vegetation removal is typically far more extensive. In many systems, the area affected by vegetation removal is on the order of 30-60 times that affected by roads and landings, significantly contributing to cumulative sediment delivery and resulting aquatic impacts (Rhodes et al., 1994).

Burning piled woody material also contributes to elevated sediment delivery by damaging and baring soils, increasing surface runoff, and severely retarding the recovery of native vegetation (Kauffman, 2004; Korb et al., 2004). The piling itself, when done by ground-based machinery, negatively affects a much larger area than the piles, contributing to persistent increases in surface erosion and sediment delivery due to soil baring, displacement, compaction, and elevated surface runoff.

MFT may also indirectly increase erosion and sediment delivery by aiding in the spread and establishment of noxious weeds. Extensive repeated MFT will likely aid in the spread of these weeds, via effects on road traffic, disturbance of soils and vegetation by machinery, prescribed fire, and pile burning (CWWR, 1996; USFS, 1999; USFS, 2000b; USFS, 2001; Korb et al., 2004; Dodson and Fiedler, 2006). Weed establishment appears to be most likely when the impacts of mechanical treatments are coupled with prescribed burning (Dodson and Fiedler, 2006). The establishment of noxious weeds can increase erosion, sediment delivery, and runoff (CWWR, 1996; USFS, 2001), which reduces soil productivity and degrade aquatic systems.

In aggregate, there is a high degree of certainty that sediment delivery and consequent negative impacts on aquatic resources will be increased by elevated surface erosion from MFT. This is due to the combined activities involved, their likely extent, frequency of disturbance, and locations involved.

Overall, there is a medium degree of certainty that MFT will increase erosion and sediment delivery from mass wasting in areas susceptible to this type of erosion. However, the level of certainty varies among types of activities involved in MFT.

In areas prone to mass erosion in response to disturbance, there is a medium degree of certainty that MFT will contribute to increased rates of mass wasting from road networks. This is partially based on the high degree of certainty that MFT requires that extensive road networks remain in place.

Roads are a primary cause of elevated rates of mass wasting in forested landscapes. Although the relationship varies with location, mass failures from roads are generally larger, more frequent, travel farther, and transport less wood than mass failures from undisturbed areas (Dunne and Leopold, 1978; Ziemer et al., 1991a; b; MacDonald and Ritland, 1989; Furniss et al., 1991; USFS et al., 1993; Montgomery, 1994; Rhodes et al., 1994;

Rhodes and Huntington, 2002; May, 2002; Montgomery et al., 2004).

Roads have been found to increase mass erosion volumes by about 30 to 350 times the amount generated in undisturbed forested areas on a per unit area basis, depending on the area under investigation. In a study on the Idaho batholith, roads were found to have increased mass erosion by about 188 times the rate found in forested areas (Furniss et al., 1991).

Dunne and Leopold (1978) concluded that road construction in most mountainous terrain contributes to increased mass wasting, regardless of how much care was taken in planning and construction. Geppert et al. (1984) echoed this conclusion, stating, "The association of roads with debris avalanches is not specifically related to the construction phase or road use, but the fact that roads exist…Unlike failures within harvest units, the potential for debris avalanches from roads does not appear to decline with time except as the more susceptible areas fail." Therefore, there is a medium degree of certainty that MFT will increase mass failures by maintaining or increasing the extent of road networks.

There is a low degree of certainty that forest removal activities associated with MFT will increase erosion by mass wasting and subsequent effects on aquatic resources. Forest removal clearly increases the frequency and volume of mass wasting relative to undisturbed areas (Dunne and Leopold, 1978; MacDonald and Ritland, 1989; Ziemer et al., 1991a; b; USFS et al., 1993; Rhodes et al., 1994; Rhodes and Huntington, 2002; May, 2002; Montgomery et al., 2004). The mass erosion volumes originating in clearcuts range from about 1 to about 9 times that found in undisturbed areas in the coastal Northwest (Furniss et al., 1991).

Forest removal increases the propensity for mass failures by decreasing root strength while increasing saturation in soils (Rhodes et al., 1994; Montgomery et al., 2004). Most types of forest removal, including thinning or partial harvest, will decrease root strength for at least a period of time, contributing to mass failure risks during major rain or rain-on-snow events. This is also made more likely by the continued attempt to remove larger trees, as part of MFT, in areas prone to mass failure, as exemplified by CNF (2002).

Because most of the field research on mass failure and forest removal has involved areas where trees were completely cleared, there is some uncertainty in the mass failures response to partial harvests with tree retention or thinning. However, complete clearing is often proposed for the creation of fuel breaks in terrain susceptible to mass failures (USFS, 1999; UNF, 2001; RSNF, 2004). Therefore, in aggregate, there is a low degree of certainty that MFT will increase sediment delivery and erosion from mass failures in susceptible landscapes.

The potential effects of MFT on mass wasting are ecologically significant. Sediment delivery by mass wasting in mountainous terrain can dominate long-term sediment budgets.

MFT may also increase sediment delivery by increasing peakflows. Increases in channel erosion and downstream sediment delivery are inevitable with persistent increases in peakflows (Dunne and Leopold, 1978; Richards, 1982). King (1989) expressly noted that increases in peakflows in headwater systems were likely to increase delivery of sediment downstream.

In most of the West, snowmelt is the primary source of peakflow. In such areas, research across the West has consistently shown that forest removal and roads significantly elevate peakflows, with the greatest increases occurring in the wettest years (Troendle and King, 1985; 1987; King, 1989; MacDonald and Ritland, 1989; Rhodes et al., 1994; Gottfried, 1991; Burton, 1997; MacDonald and Stednick, 2003). Road construction alone has been shown to increase peakflows generated by snowmelt (King and Tennyson, 1984).

Research on the hydrologic alteration by logging and roads has conclusively demonstrated at the site scale that vegetation removal in the rain-on-snow zone increases snow accumulation, snowmelt, and runoff during rain-on-snow events (MacDonald and Ritland, 1989; Harr and Coffin, 1992; Bowling et al, 2000: La Marche and Lettenmaier, 2001). The effect of forest removal and roads on the largest and most infrequent peakflows in rain-dominated systems is a matter of some contention. However, there is little dispute that all available evidence clearly indicates that the most frequently occurring peakflows (e.g. with a recurrence interval of 1-5 years) are increased in a statistically significant fashion by forest removal and roads (Jones and Grant, 1996; Thomas and Megahan, 1998; Beschta et al., 2000; Bowling et al., 2000).

The Effects of Elevated Sediment Delivery From Mechanized Fuel Treatments on Aquatic Resources and Populations

There is a high degree of certainty that increased sediment delivery from the cumulative effects of MFT will degrade water quality and aquatic habitats, and reduce the survival and production of sensitive aquatic biota. Elevated sediment delivery to streams increases suspended sediment and turbidity levels, fine sediments in streams, and degrades channel form. Elevated sediment delivery and its impacts are some of the most pervasive aquatic problems in streams draining Western watersheds that have been disturbed by land management (Sublette et al., 1990; Rhodes et al., 1994; CWWR, 1996; USFS and USBLM, 1997a; WDFW, 2000).

Elevated sediment delivery from the cumulative impacts of MFT increases suspended sediment and turbidity. Although there is tremendous variability across the West in the relationship between sediment delivery and suspended sediment or turbidity, at the scale of individual watersheds, turbidity and suspended sediment generally correlate with sediment delivery. Elevated turbidity and suspended sediment levels can impair water uses and increase treatment costs for water supplies (Reid, 1999; ECONorthwest and Pacific Rivers Council, 2002). It also impairs sight feeding by fish and, at higher levels, causes gill damage in fish (Rhodes et al., 1994).

There is a high degree of certainty that the effects of increased sediment delivery from MFT will increase damage to water quality and imperiled aquatic populations. This is due to the strong empirical evidence that MFT has been and will be proposed for implementation in watersheds important to the protection and restoration of aquatic populations. Such proposals have been made across the West (USFS, 1999; BNF, 2001; SNF, 2001; UNF, 2001; 2003; CNF, 2002; NPNF, 2002, and RSNF, 2004). All of these proposed MFT projects, save RSNF (2004), were proposed in areas that had already documented severe problems with sedimentation generated by forest removal and roads that were adversely affecting imperiled salmonids.

There is a high degree of certainty that increased sediment delivery to streams increases levels of fine sediment in streams, as laboratory and field experiments have repeatedly demonstrated, especially when the increased sediment supply primarily consists of fine sediment (Lisle et al., 1993; Rhodes et al., 1994; Hassan and Church, 2000; Kappesser, 2002). Increased surface erosion from MFT-related disturbances is comprised almost solely of fine sediment.

Increases in fine sediments in streams negatively affect salmonids and other aquatic biota (e.g., see reviews in Meehan, 1991; Rhodes et al., 1994; Waters, 1995). Increases in fine sediment in streams sharply reduce the survival and production of all salmonid species. Bull trout and cutthroat trout undergo especially sharp drops in survival with increased levels of fine sediment (Weaver and Fraley, 1991). Increased levels of fine sediment also negatively affect salamanders, which require relatively coarse channel substrate (Jackson et al., 2001).

Streams that have the following characteristics are the most sensitive to increases in fine sediment: snowmelt-dominated hydrology, relatively arid climates, significant mass erosion, granitic geology, low gradient streams, steep terrain, and low frequency of large woody debris (Everest et al., 1987). Notably, many habitats for many inland fish populations, including imperiled trout and salmon, have these characteristics and, therefore, are highly susceptible to increases in fine sediment (Rhodes et al., 1994).

Increases in fine sediment in stream substrate serve to decrease the exchange of water between subsurface near-stream flows and streams, by reducing the permeability of substrates, which strongly influences rates of water movement through soils (Freeze and Cherry, 1979; Hancock, 2002). This is not an esoteric or trivial issue. These exchanges are important for thermal regulation and the provision of baseflows to streams draining forest watersheds (Dunne and Leopold, 1978; Kirkby, 1978; Rhodes et al., 1994).

Thermal regulation and the supply of relatively cool water to streams are critically important to salmonids and amphibians (McCullough, 1999; Jackson et al., 2001). Thermal regulation is important in headwater reaches, because they provide amphibian habitat (Jackson et al., 2001) and are critical in a systemic fashion to downstream thermal regulation (Rhodes et al., 1994; Allen and Dietrich, 2005).

Elevated sediment delivery reduces the quality and volume of pools and impedes pool development via several mechanisms. Fine sediment tends to be deposited and sequestered in pools during lower flows, reducing pool volume and quality (Kappesser, 2002; Buffington et al., 2002). The loss of pool depth from sedimentation has been shown to be correlated with increased levels of fine sediment in streams caused by increased sediment delivery (Kappesser, 2002). Increased sediment delivery increases stream width and decreases stream depth in depositional reaches (Richards, 1982; Dose and Roper, 1994), which is also associated with reduced pool dimensions (Buffington et al., 2002).

USFS et al. (1993) concluded that increased sediment delivery was one of the primary causes of the extensive pool loss within the NWFP area. The regional analysis of McIntosh et al. (2000) documented the consistent loss of large pools over a 50-year period in streams in Columbia River Basin and noted that elevated sediment delivery from land management activities was a major cause of this loss. Lisle and Hilton (1992) documented that fine sediments occupied a larger proportion of pools in streams subjected to elevated sediment loads than streams with lower levels of sediment supply.

The loss of pool volume and quality negatively affects native salmonids (Meehan, 1991; Rhodes et al., 1994; USFS, 1997a; McIntosh et al., 2000). Pools provide multiple habitat functions and are an essential habitat feature of native salmonids at a variety of lifestages (McIntosh et al., 2000). Studies have repeatedly shown that salmonid production is positively correlated with pool quality, volume, and frequency (Meehan, 1991; USFS and USBLM, 1997a; McIntosh et al., 2000).

Elevated rates of mass wasting from roads and logging significantly degrade aquatic habitats, causing long-term reductions in salmonid survival as both field studies and modeling exercises have shown (Platts et al., 1989; Ziemer et al., 1991a; b; Espinosa et al., 1997; Rhodes and Huntington, 2001). For instance, it is extremely well documented that mass failures from logging and roads in the South Fork of the Salmon River, Idaho decimated salmonid populations (Platts et al., 1989; Rhodes et al., 1994). More than 30 years after the initial degradation, aquatic habitats had not fully recovered and habitat productivity remained significantly depressed (Rhodes et al., 1994).

Dose and Roper (1994) identified increased sedimentation from roads and logging as one of the primary causes of the statistically significant increase in channel width in watersheds subjected to forest removal and roads in southwestern Oregon. Increases in width/depth ratio increase summer water temperatures (Beschta et al., 1987; Rhodes et al., 1994). Bartholow (2000) estimated that the increases in channel width documented by Dose and Roper (1994) significantly increased summer water temperatures, even in the absence of any reduction in stream shading.

Elevated summer water temperature has numerous negative effects on native salmonids and stenothermic amphibians, at scales ranging from a stream reach to regions (Beschta et al., 1987; Meehan, 1991; USFS et al., 1993; McCullough, 1999, USFS and USBLM, 1997a; Jackson et al., 2001; Dunham et al., 2003a). Elevated summer water temperature is a pandemic water quality problem afflicting salmonids and other aquatic stenotherms in streams draining public lands with a history of management disturbance (USFS et al., 1993; Rhodes et al., 1994; CWWR, 1996; USFS and USBLM, 1997a; USFS, 2001; USFWS, 1998; Dunham et al., 2003a).

Increases in channel width/depth also increase the rate of heat loss from streams during winter periods, rendering streams more susceptible to freezing (Platts, 1991; Rhodes et al., 1994). Anchor ice can cause complete mortality of most aquatic life within the stream substrate (Platts, 1984). In many parts of the interior West, anchor ice can cause of significant levels of overwinter mortality of native trout (Platts, 1984).

Mechanized Fuel Treatment Effects on Riparian Areas and Functions

Riparian areas provide a variety of functions essential to protecting water quality, channel form, aquatic habitat conditions, and the survival and production of salmonids and other sensitive aquatic biota (USFS et al., 1993; NRC, 1996; 2002). Among the most vital riparian functions are the recruitment of LWD to streams, thermal regulation, bank stability, hydrologic regulation, and sediment detention and storage (Meehan, 1991; USFS et al., 1993; Rhodes et al., 1994; Henjum et al., 1994; CWWR, 1996; NRC, 1996; USFS and USBLM, 1997a; b; NRC, 2002).

There is a high degree of certainty that MFT, if implemented as proposed, will damage riparian area and functions. This level of certainty is due to the plethora of evidence that forest removal damages riparian functions together with the considerable empirical evidence that MFT will be proposed in relatively close proximity to streams and the lack of adequate riparian protections in current public land management plans. It is also partially due to the fact that existing conditions in much of the riparian areas across the West that are outside of roadless areas are already in a damaged condition caused by a variety of cumulative impacts from past and on-going activities including grazing, logging, and roads (Leopold, 1937; USFS et al., 1993; Henjum et al., 1994; CWWR, 1996; Moyle, 1996a; b; Espinosa et al., 1997; NRC, 1996; 2002; WDFW, 2000; Beschta et al., 2004). This degradation has sharply reduced the ability of riparian areas to absorb some incremental damage without it translating into aquatic damage. It has also compromised the ability of riparian areas to buffer the effects of upslope disturbance (Rhodes et al., 1994).

The removal of vegetation within about one tree height or 100 feet of streams reduces stream shading (USFS et al., 1993). The loss of stream shading contributes to increases in summer water temperature that are deleterious to salmonids, amphibians, and other stenothermic aquatic biota (Beschta et al., 1987; USFS et al., 1993; Rhodes et al., 1994; McCullough, 1999).

The removal of vegetation within a few hundred feet of streams reduces microclimate regulation of the near stream environment. This has been shown to be the case in Pacific Northwest riparian zones (USFS et al., 1993) and likely holds for other areas, although the microclimate regulation from vegetation as a function of distance from streams has not been well documented in other forest types and regions. However, it is clear that the removal of vegetation close to streams results in some loss of microclimate regulation within the stream environment (Hewlett and Fortson 1982; Platts, 1984; Rhodes et al., 1994; Brosofske et al., 1997). Vegetation loss near the channel can increase the propensity of streams to freeze during winter periods, increasing the overwinter mortality of fish and other aquatic life (Platts, 1984; Rhodes et al, 1994).

The removal of trees within one tree height of streams reduces the recruitment of large woody debris to streams (USFS et al., 1993; Rhodes et al., 1994). However, retention of all trees within one site potential tree height of streams does not insure that LWD recruitment rates are not reduced. In northwest California, Reid and Hilton (1998) found that about 30% of trees falling from within a tree's height of the channel are triggered by trees falling from upslope of the contributing tree. Since this process also likely holds for other areas, but has not been documented by field studies in other regions, there is a low degree of certainty that maintaining natural rates of LWD recruitment requires retaining trees within more than one tree-height from streams.

The loss of LWD recruitment to streams ultimately depletes LWD levels, contributing to loss of stream cover and pool volume, quality, and frequency (Meehan, 1991; USFS et al., 1993; USFS and USBLM, 1997a; McIntosh et al., 2000; Buffington et al., 2002). These impacts adversely affect salmonid populations and their habitats (Meehan, 1991; USFS et al., 1993; Rhodes et al., 1994; USFS and USBLM, 1997a).

Vegetation removal within several hundred feet of streams increases the probability of sediment delivery in two ways. First, it reduces the ability of vegetation and downed wood to detain and store sediment supplied from upslope sources. Second, it increases the proximity of sediment-generating activities to streams, which generally increases the probability and efficiency of sediment delivery to streams (USFS et al., 1993; Rhodes et al., 1994; USFS and USBLM, 1997a). Vegetation removal within about one half of a tree height reduces bank stability, which can increase bank erosion and sediment delivery (USFS et al., 1993; Rhodes et al., 1994).

MFT in riparian areas increases the propensity for the spread and establishment of noxious weeds in riparian areas, due to the effects of soil and vegetation disturbance, coupled with seed dispersal by equipment. In some parts of the West, riparian areas are especially susceptible to the establishment of some noxious weeds or already afflicted with significant infestations of non-native noxious weeds (CWWR, 1996; USFS, 2005).

The effects of MFT on existing road networks in riparian zones increase aquatic damage. This is primarily because MFT increases road traffic, especially on main haul roads, which are typically in fairly close proximity to streams (e.g., CNF, 2003). This increase in traffic increases erosion and sediment delivery from the road network, a significant percentage of which acts as extensions of the stream network (Wemple et al., 1996; Rhodes and Huntington, 2000; CNF, 2003).

MFT perpetuates damage from riparian roads in riparian areas by creating a perceived "need" for such roads for repeated treatments. This is extremely significant ecologically, because the existing impacts of the road network are known to be a major cause of damage to soils, water quality, native plants, and aquatic populations throughout the West, especially when roads are in close proximity to streams (e.g., Meehan, 1991; USFS et al., 1993; Henjum et al., 1994; Rhodes et al., 1994; CWWR, 1996; Kattleman, 1996; USFS and USBLM, 1997a; USFS, 2000b; Kessler et al., 2001). As long as roads remain in place, watersheds and aquatic resources are consigned to significant degradation accruing on an annual basis.

Based on the foregoing, there is a high degree of certainty that MFT will cause additional damage to riparian areas resulting in the loss of large wood, channel complexity, thermal regulation, stream substrate quality and water quality. There is a high degree of certainty that such impacts individually, but especially in combination, will further degrade aquatic resources.

The Potential Effectiveness of Mechanized Fuel Treatments

Proponents of fuel treatments have claimed that if treatments effectively reduce future fire severity, they will yield net benefits to soils, watersheds, and aquatic systems (e.g., Allen et al., 2002; MacDonald and Stednick, 2003; Graham et al., 2004; SFNF, 2004; ASNF, 2004; O'Laughlin, 2005). The primary basis for this view is that severe fire often increases soil erosion and runoff, sometimes dramatically, as will be discussed in greater detail. However, any potential benefits of reduced fire severity by MFT clearly come at an ecological price. Therefore, examining the potential effectiveness of MFT is a crucial step in assessing net impacts to aquatic systems from wildfire versus treatments to alter its behavior. This requires consideration of several key contexts:

- MFT is unlikely to be effective unless it aids in the restoration of the natural fire regime. Therefore, forest type and natural fire regime affected by MFT is critical to determine and consider. Because MFT is aimed at reducing fire severity and size, it is unlikely to help restore fire regimes unless wildfires are burning more severely and extensively than under the natural fire regime.
- In forest types that are experiencing uncharacteristically severe wildfire, MFT, if effective, only addresses symptoms of altered fire regimes. Unless it is part of wider efforts to restore natural fire regimes, MFT alone is unlikely to reduce fire severity and effectively restore natural fire regimes in a self-sustaining manner (Noss, 2006b).
- MFT has only transient effects on fuel conditions. Some MFT practices increase short-term and/or long-term levels of fuels that may contribute to higher-severity fire. MFT can open stands and increase wind

speeds, while reducing moisture levels, which can contribute to higher-severity fire (Martinson and Omi, 2003, Raymond and Peterson 2005).

- MFT cannot reduce fire severity unless fires that would otherwise be high severity affect treated areas during the window when fuels have been reduced. Therefore, the likelihood of this confluence of events *must* be considered in evaluating the potential effectiveness of MFT.

Forest Types, Natural Fire Regimes, and Mechanized Fuel Treatments

Despite the variability and uncertainty, and for the sake of simplicity, this report follows the route taken by other researchers (Romme et al., 2003a; b; Schoennagel et al., 2004a; Noss et al., 2006b) of grouping forests and their fire regimes into three very broad types. These are:

1) <u>Forest types with natural fire regimes characterized by relatively infrequent, high-severity fires.</u> These forest types include subalpine forests comprised of spruce, subalpine fir, and lodgepole pine in the northern Rockies, forests in the wetter maritime climates of coastal California and the Pacific Northwest, and pinyon pine-juniper woodlands (Romme et al., 2003a; b; Schoennagel et al., 2004a; Noss, 2006b). Hydric riparian and wetland forests in much of the West also likely have such a natural fire regime. Weather is the dominant control on fire frequency, severity, and extent in forests with this natural fire regime (Romme et al., 2003a; b; Schoennagel et al., 2004a; Noss et al., 2006b).

2) <u>Forests types with a natural fire regime of mixed severity and frequency, where both low-severity fires and high-severity fires occur naturally at varying frequencies.</u> Infrequent high-severity fire and frequent low-severity fire are both characteristic of the natural fire regime in these forest types. These forests are often comprised of mixed conifers, including ponderosa pine, Douglas fir, grand fir, and western larch in the northern Rocky Mountains (Romme et al., 2003a; b; Schoennagel et al., 2004a; Noss et al., 2006b). Field studies indicate that some drier forests comprised primarily of ponderosa pines at lower to mid-elevations throughout the Rocky Mountain region also have this fire regime (Ehle and Baker, 2003; Romme et al., 2003a; b; Baker et al., 2006), as do many of the mixed ponderosa pine forests in the interior Northwest (Hessburg et al., 2005). Many mixed-conifer forests of the Sierra Nevada and the Pacific Northwest also likely have this natural fire regime (Odion and Hanson, 2006; Noss et al., 2006b).

Although these forest types are among the most prevalent in the West, this fire regime is the least thoroughly understood in terms of the extent, severity, and frequency of wildfire under natural conditions (Romme et al., 2003a; b; Schoennagel et al., 2004a; Noss et al., 2006b). However, there is general agreement that both weather and fuel conditions influence fire frequency, severity, and extent in forests with this natural fire regime (Romme et al., 2003a; b; Schoennagel et al., 2004a; Noss et al., 2006b; Baker et al., 2006).

3) <u>Forest types with a fire regime characterized by relatively frequent, low-severity fire.</u> This forest type appears to be relegated mostly to some of the relatively arid ponderosa pine forests of New Mexico and Arizona, although some other ponderosa pine systems may also have this natural fire regime (Romme et al., 2003a; b; Schoennagel et al., 2004a; Noss et al., 2006b). Notably, in the Rocky Mountain region, most forests with this fire regime occur primarily on private, rather than public lands (Baker et al. 2006). Although fire frequency in these forest types

may have been overestimated due to limitations in current sampling methods (Baker and Ehle, 2001; 2003), there is currently a lack of evidence that frequent high severity crown fires occurred naturally in the forest types with this fire regime (Ehle and Baker, 2003; Schoennagel et al., 2004a). There is general agreement that fuel conditions are a primary influence on fire frequency, severity, and extent in forests with this natural fire regime (Romme et al., 2003a; b; Schoennagel et al., 2004a; Noss et al., 2006b).

It is worth stressing that not all drier lower-elevation forests with ponderosa pine have fire regimes of high-frequency, low-severity fire, as Ehle and Baker (2003) and Baker et al. (2006) have shown for these forest types in the Rocky Mountain region. Based on analysis of forest structure and fire scars, Hessburg et al. (2005) determined that many drier, mixed ponderosa forests of the interior Northwest have a natural fire regime of mixed-severity fire.

In all forest types, fire size and severity is ultimately influenced by longer-term climatic patterns. There is good evidence that the historic extent and severity of fires increased during periods of protracted drought (Whitlock et al., 2003; Pierce et al., 2004). Historical fires appear to have been most severe and extensive after a multi-annual period of drought preceded by a relatively wet period (Veblen, 2003; Romme et al., 2003a: b). Low-elevation ponderosa pine forests historically experienced extensive natural fires that burned at high severity in response to protracted historic drought (Pierce et al., 2004).

Due to changing climate, the area annually burned by fires may increase (Whitlock et al., 2003; Pierce et al., 2004; Westerling et al., 2006). Westerling et al. (2006) found that increase in the annual area burned by wildfire since 1980 has been most pronounced in forests with fire regime of low frequency, high-severity fire, where fire behavior is primarily controlled by weather.

While the three general categories of forest types/fire regimes are used for the sake of tractable analysis, the heterogeneity within and among fire regimes and forest types should both be kept in mind during the following discussion. Similarly, uncertainties and biases in fire regime estimation should temper the interpretation of the following evaluation.

The natural fire regimes in forest types are a critical context for interpreting the potential effectiveness of MFT in reducing uncharacteristically high-severity fire and restoring natural fire regimes (Romme et al., 2003a; b; Bebi et al., 2003; Schoennagel et al., 2004a; Noss et al., 2006b). It cannot be overemphasized that if MFT are not tailored to be consistent with natural forest regime restoration, they are unlikely to be successful in altering fire behavior and reducing fire severity (Schoennagel et al., 2004a; Baker et al., 2006; Noss, 2006b). Such efforts likely damage forest ecosystems without yielding any ecological benefits from the restoration of the fire regime (Veblen; 2003; Ehle and Baker, 2003; Schoennagel, 2004a; Kauffman, 2004; Baker et al., 2006).

There are some obstacles to accurately identifying natural fire regimes and potential departures from it (Baker and Ehle, 2001; Veblen, 2003; Baker and Ehle, 2003; Ehle and Baker, 2003; Romme et al., 2003 a; b). Some current methods of assessing natural fire regimes and their alteration are fraught with uncertainty, potential for error and/or inherent bias (Baker and Ehle, 2001; Veblen, 2003; Baker and Ehle, 2003; Ehle and Baker, 2003; Romme et al., 2003 a; b; Baker et al., 2006).

Due to high temporal variability in natural fire behavior from synergies among climate, forests, and fire, accurate identification of natural fire regimes may requires several centuries worth of information on fire behavior, including its extent, severity, and frequency (Baker and Ehle, 2003; Romme et al., 2003a; b; Veblen, 2003; Ehle and Baker, 2003). Even with this information, there is still some uncertainty and potentially significant bias in some common estimation methods, such as those that rely only on fire scars (Baker and Ehle; 2001; 2003; Veblen, 2003; Baker et al., 2006; Baker, 2006).

Baker and Ehle (2003) examined some of the uncertainties of estimating fire occurrence in ponderosa pine ecosystems, and concluded that past studies have overestimated fire occurrence in these forest types. They noted that sources of uncertainty and bias in some estimation methods include a lack of modern calibration, inappropriately targeted sampling, absence of fire severity evidence, and insufficient treatment of variability and uncertainty (Baker and Ehle, 2003). For instance, fire scars are widely used to estimate fire frequency, but not all fires leave scars. Conversely, stand-replacing fires do not leave trees to be scarred.

Veblen (2003) recommended that fire scar evidence should be considered only as an index of past fire occurrence rather than a complete record of past fires. Veblen (2003) noted that "Absence of the evidence (the fire scar) is not necessarily evidence of absence of the event (the fire)." Romme et al. (2003b) came to similar conclusions regarding whether recent large, stand-replacing fires were within the range of natural variation in some forest types in the northern Rocky Mountains.

Hessburg et al. (2005) documented that measurement of fire scars on individual trees tends to underestimate fire severity and overestimate fire frequency. Minnich et al. (2000) and Lentile et al. (2005) documented similar results. Data on forest structure and age at the broader scale of forest patches in conjunction with fire scar analysis appear to provide far more reliable estimates of fire severity and frequency, and hence, determination of the natural fire regime, than fire scar analysis alone (Minnich et al., 2000; Ehle and Baker, 2003; Hessburg et al., 2005; Lentile et al., 2005; Baker et al., 2006). Notably, these types of data are not typically considered in assessments of natural fire regimes and decisions to implement fuel treatments (e.g., NPNF, 2002; RSNF, 2004; SFNF; 2004a) or, for that matter, in widely used methods aimed at assessing current departures from natural fire regimes, such as the Fire Regime Condition Class approach (Hann and Bunnell, 2001).

The differing results from different methods of estimating the occurrence of fire have significant ramifications. The work of Hessburg et al. (2005) indicates that a majority of the dry forest types sampled on public lands in the interior Columbia basin were typified by a fire regime of mixed severity. However, fire scar analysis alone indicated a fire regime of frequent low-severity fire in these same areas, an assessment that is likely an incorrect artifact of sampling (Hessburg et al., 2005). Lentile et al. (2005), documented similar results in South Dakota.

These differences in the assessment of natural fire regimes have considerable ramifications for assessing the likely effectiveness of fuel treatments. Misidentification of the natural fire regime and its potential alteration can lead to implementing fuel treatments where they are unlikely to be effective and where fire regimes have not been altered. There is not a sound basis for intrusive attempts to restore fire regimes unless multiple lines of site-specific evidence convergently indicate that the fire regime has been altered (Veblen, 2003; Schoennagel et al., 2004a; Baker et al., 2006). Without such information, MFT aimed at fuel reduction and/or alteration of current fire behavior has the potential to cause ecological damage without providing the ecological benefits that can accrue from the restoration of natural fire regimes (Ehle and Baker, 2003; Schoennagel et al., 2004a; Baker et al., 2006; Baker, 2006).

Veblen (2003) noted that there are considerable differences of scientific opinion about the importance of biases and limitations of methods of estimating fire regimes, and how they should be treated or improved upon. Veblen (2003) concluded that improvements in methods of estimating natural fire regimes are important to pursue, but "…are unlikely to completely remove the uncertainties in reconstructions of historic fire regimes and their effects on forest conditions."

For these combined reasons, the degree of certainty associated with the assessment of natural fire regimes varies with the type and amount of evidence used to make the assessments. There is a low degree of certainty that natural fire regimes have been accurately assessed, *unless* there is considerable long-term data on fire frequency and severity, together with data on forest structure and age at the larger scale of forest patches (Baker et al., 2006). In areas with some natural component of high-severity fire data, dating downed wood and growth release on surviving trees in order to assess synchronous tree deaths can help date past high-severity fires (Baker et al., 2006). If all such data have been properly collected and analyzed, there is still only a medium degree of certainty that current and natural fire regimes have been correctly identified due to inherent uncertainties in fire regime reconstruction.

Even where there is convergent evidence that fire regimes have been modified, there is uncertainty as to how to treat these areas to restore natural fire regimes. For instance, although there is convergent evidence that higher-severity fire was a rarity in low elevation ponderosa pine systems in Arizona and New Mexico (Schoennagel et al., 2004a), there is still considerable potential that fire frequency in some of these systems has been overestimated (Baker and Ehle, 2003; Baker, 2006). Attempts to restore these fire regimes through repeated MFT or burnings, based on overestimated fire frequency, may be unsound and inconsistent with ecological restoration (Baker and Ehle, 2003; Baker, 2006).

Assessments of the potential alteration of fire regimes often ignore a critical context regarding fire occurrence. Forests may be within their range of natural variability with respect to fire occurrence if the current fire-free interval is shorter than the longest fire-free period in the historical record, regardless of how many mean estimated fire return intervals have been skipped. Many MFT proposals have only considered the latter while ignoring the former (e.g., NPNF, 2002; RSNF, 2004; SFNF; 2004a). Veblen (2003) cautioned that "Researchers should not overemphasize summary statistics such as mean fire intervals or fire rotation. Mean fire intervals (both composite and individual tree intervals) have *an uncertain ecological meaning*." (Emph. added.)

Several of the biases identified in the foregoing are embodied in the Fire Regime Condition Class (FRCC) approach (Hann and Bunell, 2001), which is widely used to provide an index of the potential for uncharacteristically severe fire and fire regime alteration. The FRCC relies on of estimates of mean fire intervals, but does not require that they be estimated on the basis of site-specific historical data. It emphasizes fire scar data, but does not require its collection and analysis on a site-specific basis. The FRCC's analysis of departure from natural fire regimes also relies on estimates of how many estimated mean fire intervals may have been skipped. The method does not require identification and consideration of fire-free intervals in site-specific historic record.

Notably, a recent study that examined the correlation of FRCC estimates of likely fire behavior with actual fire behavior in several large fires recently burning the Sierra Nevada in California concluded: "[Fire Regime] Condition Class was not able to predict patterns of high-severity fire. . . . Condition Class identified nearly all forests as being at high risk of burning with a dramatic increase in fire severity compared to past fires. Instead, we found that the forests under investigation were at low risk for burning at high-severity, especially when both spatial and temporal patterns of fire are considered." (Odion and Hanson, 2006.) These results corroborate that FRCC is biased toward overestimating the alteration of fire regimes and the likelihood of areas burning at uncharacteristically high severity if affected by fire. Therefore, in aggregate there is medium degree of certainty that the FRCC is biased toward overestimating departures from natural fire regimes and the propensity of forests to burn at higher severity when affected by fire.

The Consistency of Mechanized Fuel Treatments With Efforts to Restore Natural Fire Regimes by Changing Fire Behavior and Reducing Fire Severity

Forest Types With a Natural Fire Regime Typified by High-Severity, Low-Frequency Fires

There is a high degree of certainty that MFT aimed at reducing fuels and altering fire behavior in forests with a natural fire regime of relatively infrequent, high-severity fire will *not* effectively aid in restoring natural fire regimes or reducing fire severity based on convergent evidence, including:

- There is considerable evidence that fire suppression and other factors have not altered fire behavior or fire regimes in these forest types (Romme et al., 2003a; b; Schoennagel et al., 2004a; Noss et al., 2006b). Therefore, there is no sound basis for attempting to restore these forests by reducing fire severity via MFT.
- There is also considerable evidence that fire behavior and frequency in these forest types is primarily controlled by weather and *not* by fuel levels (Bessie and Johnson, 1995; Veblen, 2003; Romme et al., 2003a; b; Schoennagel et al., 2004a; Noss et al., 2006b).
- There is a high degree of certainty that if MFT were carried out to a degree that reduced fire severity in these forest types, it would alter, rather than restore, the natural fire regime (Romme et al., 2003a; b; Schoennagel et al., 2004a). Reductions in fire severity and fire regime restoration are not convergent goals in forests with this fire regime (Veblen, 2003). Fire regime restoration in these forests *requires* high-severity fire (Veblen, 2003).
- There is a high degree of certainty that MFT has been proposed in forests with these fire regimes, despite its ineffectiveness and inconsistency with the

restoration of natural fire regimes. Examples of large projects proposed in these forests include BNF (2001) and NPNF (2002).

Forest Types With a Natural Fire Regime Typified by Mixed-Severity[3]

There is a medium degree of certainty that MFT in forests with this natural fire regime will *not* be effective in most cases, due to some pieces of convergent evidence, coupled with some uncertainties. These include:

- There is a medium degree of certainty that there is no clear and convergent evidence that fire behavior in these forest types is operating outside of historic natural fire regimes (Romme et al., 2003a; b; Schoennagel et al., 2004a; Noss et al., 2006; Baker et al., 2006).
- Conversely, there is convergent evidence that large areas of forests with this fire regime have not had major alterations in the fire regime (Baker et al., 2006). This evidence includes data on downed wood levels, growth release on surviving trees to assess synchronous tree deaths, and forest structure and age at the larger scale of forest patches (Baker et al., 2006).
- Recent large fires with a considerable fraction of the area burning at high severity, such as the 2002 Hayman Fire in Colorado, may be characteristic of the natural fire regime (Romme et al., 2003a; b; Schoennagel et al., 2004a). While some more densely forested sites within these forest types might be experiencing larger and more severe fires than historically, this is highly uncertain (Romme et al., 2003b). Analyses of fire severity in some recent large fires in this forest type indicate a high degree of heterogeneity in fire severity, and belie that these forests are burning uncharacteristically or primarily at high severity (Odion et al., 2004; Odion and Hanson, 2006).
- Calls for MFT in these forests have not marshaled convergent evidence that fire severity has increased in these forests, such as that from assessment of historic fire behavior from analysis of data on fire scars and downed wood levels, together with data on forest structure and age at the larger scale of forest patches, as needed to reasonably ascertain the natural fire regime and current deviation from it.
- Because it is unclear that fire behavior has been altered in many forests with this fire regime, there is not a sound basis for MFT in these forests (Schoennagel et al., 2004a; Noss et al., 2006b; Baker et al., 2006).
- There is good evidence that fire behavior in these forest types is controlled by both weather and fuel conditions, which, at times, causes weather to trump fuel conditions in affecting fire behavior (Veblen, 2003; Romme et al., 2003a; b; Schoennagel et al., 2004a). For these forest types, Schoennagel et al. (2004a) concluded, “Extreme climate and weather conditions can override the influence of stand structure and fuels on fire behavior.”
- There is a medium degree of certainty that fuel treatments in these forests do not significantly reduce fire size or severity during weather conducive to rapidly spreading fire burning at higher severity. This has been documented in recent fires (Martinson et al., 2003; Romme et al., 2003a; b; Schoennagel et al., 2004a; Hanson and Odion, 2006). In the 2002 Hayman fire in Colorado, a wide variety of fuel treatments less than 14 years old did not burn less severely than untreated areas during the extreme fire weather (Romme et al., 2003a), as summarized in Table 2.

[3] Baker et al. (2006) suggest the term “variable severity” to describe this fire regime, however this report uses the more widely used term, “mixed-severity.”

Wildfires often burn through or breach most fuel treatments during weather conducive to rapid fire spread (Graham et al., 2003). Available data and the current understanding of fire behavior in mixed-severity fire regimes both indicate that fuel treatments are often ineffective during extreme fire weather, however, there are not plentiful field data from treatments in numerous fires burning under these conditions to corroborate this conclusion, hence, the medium degree of certainty.

- The intertwined effects of weather on treatment effectiveness, fire size, and fire severity, also make it unlikely that MFT can significantly reduce fire severity. Weather that is conducive to rapid fire spread often exerts a stronger control on fire severity than fuel levels (Graham et al., 2004; Schoennagel et al., 2004a). A very few large fires burn the vast majority of area burned annually (Gresswell, 1999; Romme et al., 2003a; b), including that burning at higher severity. These larger fires predominantly occur during more extreme fire weather (Romme et al., 2003a; b), when MFT effectiveness is limited or eliminated. Therefore, the MFT are unlikely to reduce fire severity because the majority of higher-severity fire is caused by the largest fires that burn during weather that reduces or eliminates the ability of MFT to reduce fire severity. This limitation on the ability of MFT to reduce fire severity logically follows from the current understanding of fire behavior and treatment effectiveness in mixed fire regimes, and is bolstered by available data. However, there are not plentiful field data from a large number of fires burning under different weather conditions to corroborate this conclusion. Hence, there is a medium degree of certainty that the effects of weather on fire size and severity and MFT effectiveness makes it unlikely that MFT will be effective in these forest types.
- There is a medium degree of certainty that some fuel treatments in these forests reduce the extent and severity of smaller fires during relatively moderate fire weather. There is some field evidence that some fuel treatments can decrease fire severity under some fuel moisture and burning conditions (Omi and Martinson, 2002; Martinson et al., 2003, Graham et al. 2004, Raymond, 2005; Agee and Skinner, 2005).
- There is a high degree of certainty that MFT can *increase* fire severity in these forests, exacerbating fire impacts. Raymond (2004) documented that thinning in the absence of surface fuel treatments increased fire severity in a statistically significant fashion in mixed conifer forests in SW Oregon burned by the 2002 Biscuit fire. Hanson and Odion (2006) found that MFT increased fire severity in the majority of treated areas burned by four major fires in forests with a fire regime of mixed severity in the Sierra Nevada, CA.
- Increased fire severity in treated areas may be due to the overriding influence of weather on fire behavior during large fires (Romme et al., 2003a; b; Hanson and Odion, 2006), the rapid regrowth of vegetation after treatment, reduced fuel moisture levels from increased solar heating, increased wind speeds from tree removal, treatment methods, and/or the generation of activity fuels (Hanson and Odion, 2006). Donato et al. (2006) demonstrated that postfire logging increases fine activity fuels that are highly likely to increase fire severity should fire occur. The data of Martinson et al. (2003) and Odion et al. (2004) showed that plantations burned more severely than untreated areas, indicating that MFT that give rise to dense even-aged stands are likely to increase fire severity, if fire

occurs. Based on modeling, Agee (2003) concluded that fuel reduction via group selection does not aid in reducing fire severity, and removal of the largest trees is a "disaster" with respect to fire severity. All of these results are consistent with the body of scientific information on how forest and fuel conditions affect fire severity, conferring a high degree of certainty that some treatments increase fire severity in these forests.

- There is a medium degree of certainty, based on field evidence and modeling, that MFT in these forest types must include surface fuel treatments in order to reduce fire severity (Agee, 2003; Raymond, 2004).
- While MFT may reduce fire severity during weather that is only moderately conducive to fire, there is a low degree of certainty that this will aid in restoring the spatial and temporal complexity of the mixed severity natural fire regime (Veblen, 2003; Baker et al., 2006; Odion and Hansen, 2006).

Table 2. Fire severity in areas with unmodified fuels and fuels modified less than 14 years prior, on slopes less than 30% in the 2002 Hayman Fire, during extreme fire weather on June 6 (Martinson et al., 2003). The treatments shown below were in forest types with a natural fire regime of mixed severity (Romme et al., 2003a; Schoennagel et al., 2004a). No associated statistical information (e.g. variance, confidence intervals, sample numbers) was supplied by Martinson et al. (2003). However, it is reasonable to assume that the 90% confidence interval is at least +/- 10% from mean values. Treatment types with less than 100 acres affected are likely too small to be statistically significant. Based on these data, Martinson et al. (2003) and Schoennagel et al. (2004) concluded that these fuel modifications did not reduce fire severity. Martinson et al. (2003) concluded that plantations burned more severely than untreated areas.

Treatment	Area (ac)	Area Unburned (%)	Area Low-Severity (%)	Area Moderate-Severity (%)	Area High-Severity (%)
Unmodified	22,546	4	18	8	70
Wildfire	12	0	0	25	75
Prescribed fire	719	8	20	11	63
Improvements + treatments	395	0	19	7	74
Improvements but no treatments	625	3	12	9	76
Harvest + treatments	1622	5	14	10	71
Harvest but no treatments	583	0	1	22	66
Plantation	136	0	8	5	87

- There is a medium degree of certainty that undisturbed, mature forests with this fire regime are not in need of MFT to restore fire regimes; a proactive approach of allowing wildland fire and other sources of natural tree mortality to operate is likely to be effective at gradually restoring fuels and fire behavior in these forests (Baker et al., 2006).
- There is high degree of certainty that retention and protection of all large trees that pre-date Euro-American settlement is essential to restore old-growth structure (Baker et al., 2006). There is a medium degree of certainty that the wood that might be taken via MFT is generally needed to replenish wood previously lost to logging or burning (Baker et al., 2006).
- In some forests with this fire regime, grazing has likely contributed to alteration of fire regimes (Belsky and Blumental, 1997). In such areas, there is high degree of certainty that eliminating or very significantly curbing such grazing will be needed in order to restore fire regimes (Baker et al., 2006).
- Logging has also altered fire regimes in some of these forests (Veblen, 2003; Romme et al., 2003a; b; Baker et al., 2006). There is a high degree of certainty that natural fire regime restoration in such forests requires ensuring that logging does not continue to alter the fire regime and cause additional degradation (Baker et al., 2006).
- There is a medium degree of certainty that MFT in forests with this fire regime will likely lead to increased tree regeneration, creating a "need" for additional, future thinning. Baker et al. (2006) note that such effects from thinning can initiate "… a potentially endless, costly, and futile cycle that does not restore the forest."
- There is a medium degree of certainty that fire, which would be higher severity in the absence of treatment, will *not* affect treated areas during the 10-20 year time period when fuels are reduced (Rhodes and Baker, in review; see expanded analysis that follows below). Since this is the necessary, but not sufficient, condition for MFT to potentially reduce fire impacts, it confers a medium degree of certainty that MFT in forests with this fire regime cannot reduce fire severity in the majority of areas treated.

Forest Types With Frequent, Low-Severity Natural Fire Regimes

There is a medium degree of certainty that MFT that reduce fire severity could aid in restoring these natural forest regimes in areas that currently are subject to *uncharacteristically* higher-severity fire. However, it cannot be assumed that most treatments will reduce fire severity in these forest types. Fire must affect treated areas during their window of treatment effectiveness in order to reduce fire severity. In these forest types, there is a low likelihood that fire will affect treated areas during the period when fuels have been reduced (Rhodes and Baker, *in review*). Therefore, there is a medium degree of certainty that the majority of fuel treatments in these forest types will not aid in restoring the natural fire regime, because they will not reduce fire severity. These levels of certainty are based on available evidence, coupled with some key uncertainties.

- There is convergent evidence that high severity or stand-replacing fires did not occur with significant frequency in forests with these fire regimes within relatively recent history and climate regimes (Allen et al., 2002; Veblen, 2003). However, there is some uncertainty in the frequency of fire in these forest types due to the limitations of current fire regime reconstruction methods (Baker and Ehle, 2001; 2003; Veblen, 2003; Hessburg et al, 2005). In aggregate, this confers a medium degree of certainty that natural fire regimes of some lower elevation ponderosa pine forests in the arid Southwest and some other areas are naturally dominated by frequent, low-severity fire.

- There is a low degree of certainty regarding the distribution and extent of forests with this fire regime in the Western U.S. Some ponderosa pine stands do not have this fire regime (Veblen, 2003; Baker and Ehle, 2003). There is a medium degree of certainty that many Western forest types previously believed to have this fire regime, including drier ponderosa pine-Douglas fir forests, instead, have a fire regime of mixed severity (e.g., Veblen, 2003; Hessburg et al., 2005; Baker et al., 2006). In the Rocky Mountain region, the majority of forests with a natural fire regime of frequent, low-severity fire are not on public lands (Baker et al., 2006).
- There is convergent evidence that fire suppression, grazing (Belsky and Blumenthal, 1997), and other land management activities have contributed to altered fire behavior forests with this natural fire regime. However, fire suppression has probably had limited effects on wildland fire and resulting fuel conditions in larger roadless and wilderness areas (Allen et al., 2002; Noss et al., 2006a). In such areas, MFT to reduce fire severity are not needed to restore fire regimes.
- Where fire regimes have been altered by grazing, logging, post-disturbance planting, and/or fire suppression, it is critical to curb these actions in order to avoid continued degradation and departure from natural fire regimes.
- The combined evidence indicates there is a medium degree of certainty that treatments that reduce fire severity in these forest types can aid in restoring the natural fire regime in areas where it has been altered (Allen et al., 2002; Schoennagel et al., 2002). It is worth stressing that restoration of the fire regime in these systems requires that wildfire frequency be restored. Otherwise, attempts to restore the fire regime by reducing fire severity are may be futile (Allen et al., 2002; Kauffman, 2004).
- There is a high degree of certainty that MFT alone cannot increase fire frequency. This is a key issue, because, as Kauffman (2004) noted: "A basic tenet of ecological restoration is that creation of form without function does not constitute ecological restoration…" MFT alone plainly do little to restore fire as a process.
- As in forests with mixed-severity fire regimes, there is a high degree of certainty that any MFT should retain all larger, older trees as part of efforts to restore fire regimes (Allen et al., 2002; Noss et al., 2006a; b).
- There is a medium degree of certainty that MFT can reduce fire severity in forests with this fire regime, if the treated areas encounter fire while fuels have been transiently reduced (Schoennagel et al., 2004a; Noss et al., 2006a; Cram, 2006). However, reliable, robust, corroborative field evidence of such effectiveness is lacking (Carey and Schumann, 2003; Rhodes and Odion, 2004; Schoennagel et al., 2004b). As Graham et al. (2004) noted, available studies have failed to consistently demonstrate that fuel treatments significantly altered the behavior, spread, or severity of wildfire.
- There is a medium degree of certainty that the majority of MFT in forests with this natural fire regime will *not* reduce fire severity under current management. This is because there is a small probability that higher-severity fire will affect these areas during the 10-20-year period when fuels

have been reduced, based on the analysis of data on more than 40,000 fires that occurred over a 23-year period (Rhodes and Baker, *in review*). This analysis indicates that, on average, about 92-96% of fuel treatments in these forests will not encounter higher-severity fire within 20 years of treatment (Rhodes and Baker, *in review*). Therefore, on average, 92-96% of treated areas will not reduce fire severity and aid in restoring the natural fire regime.

Due to its strong control on the potential effectiveness of fuel treatments in reducing fire, the probability of fire affecting treated areas is discussed in more detail in a subsequent section evaluating the potential effectiveness of MFT in reducing fire severity.

The foregoing overview of the potential for MFT to aid in restoring fire regimes by forest type and natural fire regime is summarized in Table 3.

Table 3. Forest types, natural fire regimes and potential efficacy of MFT in restoring the natural fire regime.

Natural Fire Regime	Example Forest Types	Primary Control(s) on Fire Severity	Likely Effectiveness of MFT at Restoring the Natural Fire Regime	Net Impacts of MFT on Watersheds and Aquatic Systems
Infrequent, High-Severity	Subalpine, lodgepole, coastal temperate, riparian forests and pinyon pine–juniper woodlands	Weather	Wholly ineffective; reduction of fire severity is not consistent with fire regime restoration.	Negative, without positive benefits from reduced fire severity.
Mixed Frequency, Mixed-Severity	Mixed conifer forests, most dry Douglas fir-ponderosa pine forests in the Rocky Mountain region	Weather and fuels	Low, due to limited alteration of natural fire regimes, weather effects on fire size and MFT effectiveness, as well as the low probability of fire affecting treated areas during the window of reduced fuels.	In the vast majority of treated areas effects are negative without compensatory benefits from reduced fire severity; some potential benefits in the rare cases where fire severity is reduced.
Frequent, Low-Severity	Low elevation ponderosa pine forests in the Southwest	Fuels	Potentially high, *if* treatments encounter higher-severity fire during window of reduced fuels (10-20 years), but the likelihood of this is low due to low probability of higher-severity fire affecting treated areas within 10-20 years.	In the vast majority of treated areas, effects are negative without benefits from reduced fire severity; some potential benefits in the rare cases where fire severity is reduced.

Mechanized Fuel Treatments and Fire Regime Restoration Within the Context of Overall Public Land Management

In forests and fire regimes where MFT might be effective in helping to restore natural fire regimes, it must be part of integrated efforts to address the root causes of altered fire behavior in order to restore natural fire regimes. It is well-established that in forests where natural fire regimes have been altered, some of the primary causes of altered fire behavior are changes in fuel conditions caused by the legacy and on-going effects of fire suppression, post-disturbance planting resulting in dense early seral forests, logging, and grazing. These activities continue over large areas. Therefore, unless these activities are sharply curbed or eliminated, there is a medium degree of certainty that MFT will ultimately fail to restore natural fire regimes.

Continued suppression of wildfires impedes, rather than aids, the restoration of fire regimes (Kauffman, 2004). Continuing to implement activities that cause departure from historic natural processes is inconsistent with attempts to restore altered ecosystems, in general (Kauffman et al., 1997; Beschta et al., 2004), and for the specific case of forests with altered fire regimes (Kauffman, 2004; Baker et al., 2006; Odion and Hansen, 2006). Wildland fire helps to restore altered fire regimes (Kauffman, 2004; Baker et al., 2006; Odion and Hansen, 2006; Noss et al., 2006b); in some forests, it may be all that is needed to restore their fire regimes (Baker et al., 2006; Noss et al., 2006b). Therefore, there is a high degree of certainty that continued aggressive fire suppression of all fires undermines efforts to restore altered fire regimes and contributes to further fire regime alteration.

Fuel breaks and other treatments aimed at reducing the extent of wildland fires by providing areas where fire can be more easily suppressed are probably not consistent with efforts to restore altered fire regimes. These approaches and treatments perpetuate fire suppression, which is a key cause of altered fire regimes in some forests. Therefore, fuel breaks and other approaches that abet fire suppression methods are unlikely to help restore natural fire regimes.

Mechanized Fuel Treatment Effects on Fuels

An important consideration in evaluating the overall potential effectiveness of fuel treatments in reducing fire severity is their effectiveness at reducing fuels. Fundamental issues that affect the effectiveness of MFT in reducing fuels are:

- There is a high degree of certainty that fuel reductions from MFT are relatively fleeting. Vegetation and fuels begin to re-accumulate as soon as fuel reduction treatments are completed (Kauffman, 2004; Graham et al., 2004). Although this varies with site factors that affect vegetative regrowth, it is unlikely that reduced fuel levels persist for longer than 20 years (Martinson et al., 2003; Graham et al., 2004). In some areas, it is considerably more fleeting. In a study of fuel treatments in the Sierra Nevada, van Wagtendonk and Sydoriak (1987) estimated that fuels returned to pre-treatment levels within 11 years. Therefore, there is a high degree of certainty that the effectiveness of fuel reduction by MFT declines over time and becomes non-existent after about 20 years or less (Kauffman, 2004; Graham et al., 2004; Agee and Skinner, 2005; Rhodes and Baker, *in review*).
- There is a high degree of certainty that some types of MFT can increase levels of the most flammable fuels. For instance, if treatments are not repeated, clearcut fuel breaks are likely to give rise to highly flammable, even-aged, early seral forests, similar to plantations. Such forests have

been repeatedly shown to be prone to burning at high severity (Martinson et al., 2003; Odion et al., 2004). The medium degree of certainty that such treatments increase shrub growth and the likelihood of invasion by fire-prone noxious weeds (Keeley 2001, 2002) increases this likelihood. Thinning and other forms of MFT cause increased tree regeneration, which can create a self-perpetuating cycle of repeated treatments without restoring natural fire regimes (Baker et al., 2006; Noss et al., 2006b).

- There is a high degree of certainty that MFT generates activity fuels (Donato et al., 2006) that are particularly flammable (Brown et al., 2003). These flammable fuels persist for some period of time, unless and until the generated fuels are treated via off-site removal or burning (broadcast or pile) or they decay over time.
- There is a high degree of certainty that MFT for fuel reductions will *increase* fire severity if fire occurs while treatments are in progress, e.g. activity fuels and felled material remain on the ground prior to being treated by off-site removal or burning. Assuming that such a risk persists for only a year, the probability is on the order of 0.4-1.0%.[4] Although this probability is low, the ecological costs are likely to be high, because the negative watershed impacts include those from MFT *and* significantly increased fire severity.
- MFT can facilitate the spread and establishment of noxious weeds (Dodson and Fiedler, 2006), which can alter the trajectory of post-MFT plant succession, potentially increasing fuels and propensity for higheseverity fire (Veblen, 2003; Brooks et al., 2004). Due to the activities involved and their effects, MFT greatly increase the likelihood of accelerated spread and establishment of noxious weeds, especially in areas where infestations are already present (USFS, 1999). Increases in noxious weeds appear to be most likely when MFT is employed in tandem with prescribed burns (Dodson and Fiedler, 2006).
- There is a high degree of certainty that the removal of large trees does not appreciably reduce the types of fuels that significantly affect uncharacteristically high fire severity. Evaluations have consistently concluded that MFT which remove the largest trees and/or leave the smallest diameter trees are unlikely to reduce uncharacteristic fire severity or restore natural fire regimes (Allen et al., 2002; Agee, 2003; Carey and Schumann, 2003; Graham et al., 2004; Stephens and Moghaddas, 2005; Noss et al., 2006a; b; Baker et al., 2006).
- There is high degree of certainty that many proposed MFT proposals aim to remove the largest trees and/or propose to leave the smallest trees via lower limits on tree diameters to be removed or treated. Such proposals include USFS (1999); BNF (2001), SRNF (2001), CNF (2001), NPNF (2002), RSNF (2004), and ASNF (2004). Therefore, it is clear that many MFT projects will not be consistent with the aim of retaining the largest trees, reducing fire severity, or restoring fire regimes.

[4] Slash/surface fuels treatments frequently lag multiple years behind tree removal, increasing the probability of fire affecting treated areas while such fuels from felling remain elevated. Even when surface fuel treatments are prescribed it is not uncommon for them to run well behind schedule (e.g., USGAO, 2006) or to not be implemented.

Fire Occurrence and the Potential Efficacy of Mechanized Fuel Treatment

An overarching control on the potential effectiveness of MFT to reduce fire severity is whether treated areas are affected by fire that would burn at higher severity in the absence of treatment. If higher-severity fire[5] does not affect treated areas during the time that fuels are reduced, treatments cannot reduce fire severity. In contrast, if fire affects treated areas while fuels are reduced, there is some potential for MFT to reduce fire severity and restore the natural fire regimes in forests with natural fire regimes of mixed-severity fire or infrequent low-severity fire. This might potentially confer benefits to aquatic systems, by reducing the adverse effects of uncharacteristically severe fire. Therefore, more detailed examination of the likely effectiveness of MFT in these forest types is merited.

There are two reasons why only higher-severity fire is of primary interest in this analysis. First, only higher-severity fire has significant impacts on watershed processes. Lower-severity fire has minimal and transient watershed impacts (Minshall et al.?, 1997; Gresswell, 1999), many of which are beneficial. Second, fuel treatments are highly unlikely to entirely reduce the occurrence of low-severity fires, because they do not "fireproof" areas.

Although some low-severity fires can become high-severity, this need not be assumed to uniformly occur in a probabilistic analysis, for several reasons. First, aggressive fire suppression continues as a routine part of fire management (e.g., USFS, 2005). Low-severity fires are easily and often extinguished under current management. Second, the potential for higher-severity fire to occur is captured by its probability based on past records, making consideration of the low-severity fire becoming higher-severity fire unnecessary in a probabilistic analysis of the potential for higher-severity fire to affect treated areas. Last, the future likelihood of lower-severity fire becoming higher severity cannot be estimated deterministically with any precision.

Models cannot accurately predict future fire occurrence or behavior at a given location (Graham et al., 2004). A large array of conditions that interactively control fire behavior and occurrence (e.g. ignitions, weather, fuel moisture, etc.) are not possible to accurately predict spatially or temporally. However, the likelihood of occurrence over larger areas can be estimated from past fire behavior and occurrence (Minnich et al., 2004; Baker, 2006; Rhodes and Baker, *in review*). Such methods are commonly used for natural phenomena that occur episodically at variable frequencies that are not completely predictable (e.g., Dunne and Leopold, 1978).

The probability of higher-severity fire affecting treated areas during the period when fuels have been reduced can be estimated using the standard formula for the probability of an event occurring during a specific time frame, as is often employed in hydrologic analysis of the probability of a flooding event:

$$q = 1 - (1 - p)^n \qquad (1)$$

where q is the probability that a fire that would be high severity in the absence of treatment occurs within n years, p is the annual probability of fire of high severity at the treatment location, and n is the duration, in years, that treatments decrease fuels to a degree that can reduce fire severity. Both n and p can be estimated from available data and information on fire severity and extent.

[5] "Higher-severity fire" is used to denote fires that are high to moderate in severity.

The estimated probability of fire occurrence within a given timeframe, q, in Equation 1, provides an estimate of the mean fraction of treated areas that can potentially reduce fire severity during the period when fuels have been transiently reduced. It also provides an estimate of the mean fraction of areas that are expected to burn at higher severity over the time period, n, in the absence of fuel treatments. Analysis of the probability of these outcomes provides a framework essential to estimating the risks and potential consequences of treatment versus non-treatment of forest fuels, as recommended in risk assessments of action versus non-action (NRC, 1996).

Equation 1 can also be used to estimate the number of repeated treatments, on average, that are needed to achieve a specified value of q. This abets the estimation of cumulative effects on ecosystems from repeated treatments (Ziemer, 1991; Ziemer et al. 1991; Abbruzzese and Leibowitz 1997).

Rhodes and Baker (*in review*), estimated the annual probability of higher-severity fire, p, based on the analysis of fire data for Western USFS lands at three scales: 1) for all vegetation types from 1960 to 2003; 2) in more-frequently burning ponderosa pine forests by USFS administrative region from 1980-2003; and, 3) for all ponderosa pine forests from 1980-2003. For the latter two scales, GIS data from more than 40,000 fires were analyzed (Rhodes and Baker, *in review*). To estimate, p, in Equation 1, data on fire area were used in conjunction with assessments of the spatial extent of fire severity in USFS burned area emergency rehabilitation (BAER) reports conducted for 470 fires in the 11 Western states from 1973-1998 in six Western USFS regions (Robichaud et al. 2000), as in the following equation:

$$p = (F/(A*D))*r \qquad (2)$$

where p is the annual probability of fire of a specific severity, F is total area burned at any severity within the analysis area over the duration of the data record, A is the total area of the analysis area, D is the total duration of the data record in years, and r is the estimated fraction of the total fire area that burned at the specified severity over the analysis area, as estimated from the extensive fire severity assessments in Robichaud et al. (2000).

Rhodes and Baker (*in review*) used post-1960 fire data rather than estimates of natural fire return intervals for three primary reasons. First, evidence indicates that natural fire regimes no longer operate in some forests. Fire behavior has been altered by a number of factors, including the changes in fuel character caused by livestock grazing (Belsky and Blumenthal, 1997), logging and fire suppression (Veblen, 2003). These activities remain in operation over large areas. Data for recent fires ostensibly integrate this alteration, providing some reflection of how fires are likely to burn in the near future under current conditions and management. Natural fire return intervals do not capture this alteration. Second, there is considerable uncertainty regarding the accuracy of estimated natural fire intervals (Baker and Ehle, 2001; Veblen, 2003).

Third, the amount of area burned by fire may be increasing due to climatic warming. Westerling et al. (2006) found that annual fire area in the Western U.S. increased during the mid-1980s, relative to the annual area burned from 1970-1986. By using data from all fires from 1980-2003 in ponderosa pine forests on Western USFS lands, the analysis of Rhodes and Baker (*in review*) incorporates the recent increases in fire area found by Westerling et al. (2006) and its effects on the annual probability of fire in the Western U.S.

Table 4 summarizes the estimated annual probability of fire of various severities at the three scales analyzed by Rhodes and Baker (*in review*). Using Equation 1, these results were used to estimate the probability that fire of various severities affects treated areas at the three scales analyzed (Table 4). At all scales analyzed, the duration of treatment effectiveness, *n*, was assumed to be 20 years in order to estimate the upper bound of potential treatment effectiveness.

At the scale of all vegetation types, Rhodes and Baker (*in review*) estimated that a mean of about 0.35% of USFS Western lands burn annually at any severity. Despite the approximations involved, this estimate compares reasonably with other independent estimates. It falls between the approximately 0.14% mean estimated annual fire probability on Deschutes National Forest in Oregon, based on data from 1910-2001 (Finney 2003; 2005), and the 0.46% estimated for 11 national forests in the Sierra Nevada, California, based on data from 1970-2003 (USFS, 2004).

Based on these results, at the scale of all vegetation types, on average, it is expected that about 3.9% of fuel treatments would encounter high-to-moderate-severity fire, and about 1.9% would encounter high-severity fire within 20 years of treatment (Rhodes and Baker, *in review*). Substituting space for time, this indicates that if treatments were randomly placed on the landscape, on average, they only have the potential to reduce fire severity about 2-4% of the time. In the remaining area (96-98%), treatments have negative effects on watershed and aquatic resources that are not counterbalanced by reduced fire severity.

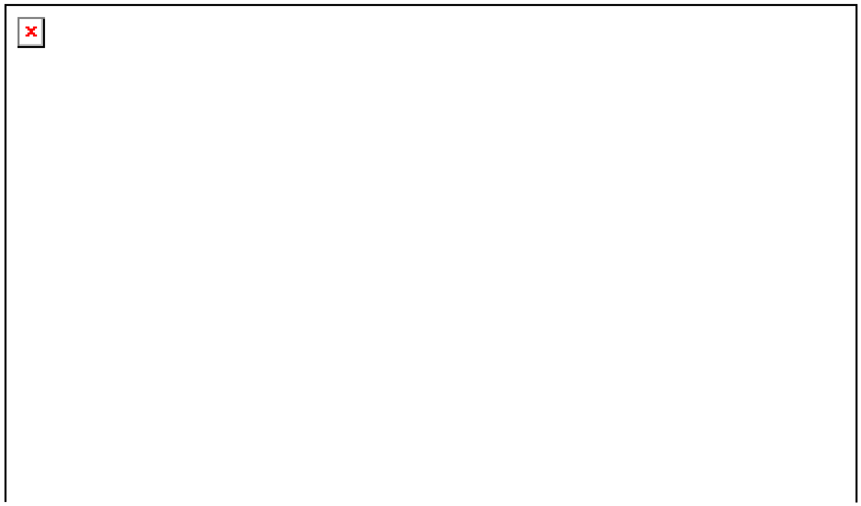
The results of this analysis at the scale of all vegetation types includes forest types that burn infrequently. These results are unlikely to be applicable to more-frequently burning forest types, such as some ponderosa pine forests.

In ponderosa pine types, it is still likely that the majority of treatments will not be affected by higher-severity fire, based on the results in Table 4. The probability, *q*, of treated areas being affected within their window of effectiveness varies regionally from 2.0 to 4.0% for high-severity fires and from 4.2 to 7.9% for high-to-moderate-severity fires (Table 4). As expected, *q* in these forests is higher than for the West-wide analysis of all vegetation types. Substituting space for time, the analysis indicates that of 92-98% of fuel treatments in ponderosa pine forest types would have negative effects on watershed processes and aquatic systems without any compensatory reduction in fire severity.

As mentioned, *q* is also an estimate of the mean fraction of the analysis area affected by higher-severity fire in the absence of treatment. The results in Table 4 indicate that high-severity fire is likely to affect about a mean of 2.0 to 4.0% of the ponderosa pine forest area across these regions over a 20-year period, if fuel treatments are foregone. Similarly, it is expected that high-to-moderate-severity fire would affect about 4.2% to 7.9% of forest area across these regions in the absence of fuel treatments.

These results and the analytical framework indicate that watersheds and aquatic ecosystems in these treatments would have to be repeated many times over long time spans or extensive areas before a majority of treated areas are expected to be affected by higher-severity fire. Across the six regions analyzed, treatments in ponderosa pine would have to be repeated every 20 years for 340 to 700 years (17 to 35 times), depending on the region, before it is expected that high-severity fire affects more than 50% of treated areas during periods when fuels have been reduced. Treatments would have to be repeated for 180 to 340 years (9 to 17 times) before more than 50% of treated areas are expected to be

Figure 1. Relationship between *q*, the probability of higher-severity fire affecting treatments, and *n*, the duration of reduced fuels after treatment, in Equation 1 at *p* = 0.3%.



affected by higher-severity fire. On average, these repeated treatments would cumulatively impact watersheds and aquatic systems without providing any reduction in fire severity on almost 50% of the treated areas. Although repeat treatments may involve prescribed burns, rather than MFT, the former still contributes to cumulative watershed impacts, sometimes severely.

The probability that higher-severity fire affects treated areas in forest types with a fire regime of mixed severity was not estimated by Rhodes and Baker (*in review*). However, these forests generally burn less frequently than ponderosa pine systems. Hence, it is likely that q in forests with fire regimes of mixed severity ranges between that estimated for all vegetation types and ponderosa pine forests (Table 4).

For several reasons, the analysis summarized in Table 4 likely provides an estimate of the upper bound of potential fuel treatment effectiveness. First, the duration of fuel treatment effectiveness, n, is typically less than the 20 years assumed by Rhodes and Baker (*in review*). In Equation 1, q decreases with decreasing values of n (Figure 1). In the range of values of p summarized in Table 1, reducing n from 20 to 11 years in Equation 1 reduces the probability that higher-severity fire affects treatments by about 40%. In other words, as the duration of post-treatment fuel reduction decreases, so, too, does the likelihood that treated areas will encounter higher-severity fire while fuels are reduced by the treatments.

Second, fuel levels steadily rebound after treatment, eventually negating potential treatment effectiveness. This makes it likely that ability of treatments to reduce fire severity also declines over time. This is not factored into the results in Table 4. If the reduction in effectiveness over time is such that mean effectiveness over the duration, n, is half the initial value, then the probability that fuel treatments reduce higher-severity fire is approximately half the value of q for any value of p and n calculated using Equation 1.

Finally, fuel treatments do not always reduce fire severity when they encounter higher-severity fire while fuels are reduced (Schoennagel et al., 2004a; Agee and Skinner

2005; Hanson and Odion, 2006; Table 2). For these combined reasons, the results in Table 4 likely provide an upper bound of the potential effectiveness of fuel treatments in reducing fire impacts on aquatic systems and native fish.

Table 4. Estimated *p* and *q* for three classes of fire severity in ponderosa pine (PIPO) forests and all vegetation types on Western USFS lands, based on the analysis of Rhodes and Baker (*in review*).

USFS Administrative Region	Forest Type	Fire of All Severity		High-to-Moderate-Severity Fire		High-Severity Fire	
		p[a]	*q*[b]	*p*	*q*	*P*	*q*
1 No. Rockies	PIPO	0.0070	0.1311	0.0036	0.0693	0.0020	0.0402
2 Cen. & So. Rockies	PIPO	0.0059	0.1116	0.0041	0.0786	0.0014	0.0269
3 Southwest	PIPO	0.0053	0.1008	0.0025	0.0487	0.0016	0.0307
4 Gt. Basin	PIPO	0.0090	0.1654	0.0037	0.0715	0.0013	0.0257
5 Calif.	PIPO	0.0046	0.0881	0.0031	0.0603	0.0017	0.0338
6 Northwest	PIPO	0.0037	0.0715	0.0022	0.0421	0.0010	0.0198
West-Wide	PIPO	0.0054	0.1026	0.0031	0.0602	0.0015	0.0295
West-Wide	All Vegetation Types	0.0035	0.0677	0.0020	0.0394	0.0010	0.0192

[a] *p*, mean annual fire probability, for ponderosa pine forests, is derived from GIS analysis of historical fire occurrence data (National Interagency Fire Center 2004) and fire severity data (Robichaud et al. 2000).

[b] *q*, probability that a fire occurs within a 20-year window of treatment effectiveness, based on Equation 1.

The results of Rhodes and Baker (*in review*) are based on extensive data from more than 40,000 fires over more than two decades. The results for ponderosa pine forests in Table 4 are based on fire data from 1980-2003 and, thus, incorporate relatively recent increases in the area annually burned by fire as found by Westerling et al. (2006) for fires from 1987-2003. Hence, the results in Table 4 provide a fairly reliable indication that a majority of treatments in forest types with relatively frequent fire will not reduce fire severity. However, they may not be completely applicable to specific areas that might burn at a frequency and severity that is different than the mean. Climatic warming may increase the amount of area burned annually in the future.[6] Therefore, the results in Table 4, together with key uncertainties, confer a medium degree of certainty that most MFT in these forest types will not encounter higher-severity fire while fuels are reduced, and, hence, will not be effective at reducing fire severity.

These results do not mean that MFT might not sometimes reduce fire severity considerably in some areas. However, they do indicate that, on average, most treatments will not affect fire severity, even when they are properly implemented in forest types where they are most likely to be effective.

The Combined Effects of Mechanized Fuel Treatments and Fire on Watersheds and Aquatic Systems

The foregoing provides a framework for assessing five discrete possibilities regarding fire occurrence, MFT, its effects on fire, and consequent watershed impacts. These are summarized in Table 5, together with some estimates of their likelihood.

The first discrete possibility is that fire does not affect treated areas during the time when fuels have been reduced. Because fire severity cannot be reduced if fire does not occur, with this outcome MFT has negative impacts on watersheds that are not counterbalanced by any reduction in fire severity and consequent fire effects. Rhodes and Baker (*in review*) found that this outcome is by far the most likely (> 92%), even in systems that burn the most frequently and are often cited as most in need of MFT to reduce fire severity (Table 4).

The second discrete outcome is that fire affects treated areas during their window of effectiveness, but does not reduce fire severity, due to weather, fire regime, or implementation practices. This outcome will almost always be the case in subalpine systems, and likely the most widespread case in mixed-conifer systems, due to the influence of weather on fire size, fire severity, and MFT effectiveness. Under this outcome, MFT has negative impacts on watersheds, which are not counterbalanced by fire severity reduction. In this outcome, the net effects on watersheds are the negative aquatic effects of MFT plus those from fire.

A third discrete outcome is that fire occurs in treated areas, but treatment increases fire severity. Such an outcome is possible where implementation practices are not consistent with the reduction of fire severity. For instance, MFT that remove large wood and generate large amounts of untreated activity fuels, while increasing wind speed and reducing fuel moisture through canopy removal, can increase fire severity (Raymond, 2004). Even with ideal implementation, MFT may also increase fire severity if fire occurs while treatments are in progress, with

[6] Equation 1 can easily be used to analyze areas with different annual probabilities of fire, allowing more spatially refined estimates of the likelihood of higher-severity fire affecting treated areas. This flexibility also allows consideration of potential increases in the annual probability of fire due to climatic warming.

<u>Table 5</u>. Five discrete outcomes of MFT with respect to fire and net effects on watershed and aquatic systems; the estimated likelihood is based on the probability of occurrence during a 20 year time period.

Discrete Outcome: Fire Occurrence and Fuel Treatments	Effect of MFT on Fire Severity	Likelihood of Outcome (Per Unit Area Basis)	Forest Type/Natural Fire Regimes Where Outcome Is Most Likely	Net Impacts of MFT on Watersheds and Aquatics
Fire Does Not Affect Treated Areas Within Window of Reduced Fuels	No potential effect	> 92%	Subalpine and maritime forests/infrequent, high-severity fire	All of the negative impacts of MFT without any compensatory benefits from reduced fire severity
Fire Affects Treated Areas Within Window of Reduced Fuels	None	< 8%	Subalpine and maritime forests/infrequent, high-severity fire; and mixed-conifer/mixed severity during extreme fire weather	All of the negative impacts of MFT, *plus* fire impacts without any compensatory benefits from reduced fire severity
Fire Affects Treated Areas	Fire severity increased	Variable, depending on persistence of increased fuels after MFT	Mixed conifer/mixed-severity fire; ponderosa pine/frequent low-severity fire, if activity fuels remain present and untreated	All of the negative impacts of MFT, *plus* increased fire impacts from increased severity
Fire Occurs During Window of Reduced Fuels, but Does Not Affect Treated Areas or Fire Affects Treated Areas After Fuel Levels Have Rebounded	No potential effect	Variable; on a per unit area basis, <8%	Ponderosa pine/frequent, low-severity fire (for fire affecting untreated areas)	All of the negative impacts of MFT, *plus* fire impacts without any compensatory benefits from reduced fire severity
Higher-Severity Fire Affects Treated Areas During Window of Reduced Fuels	Reduced	< 8%	Ponderosa pine/frequent, low-severity fire	All of the negative impacts of MFT, *plus* the benefits from reductions in fire severity

abundant downed, untreated activity fuels present. Although the chance of this occurring is relatively small – on the order 0.4 – 1.0% over the first year after treatment, based on the results in Table 4 – is still about a sixth as likely as higher-severity fire affecting completed fuel treatments within the window of effective fuel reduction. Therefore, this outcome should be explicitly considered, via the analysis in Equation 1. Under this outcome, MFT have negative watershed impacts from the combined collateral treatment effects *plus* those from *increased* fire severity.

A fourth discrete outcome is that fire affects a treated watershed, but not treated areas, or affects treated areas after fuels have returned to pre-treatment levels, resulting in no reduction in fire severity. This outcome is possible in any forest type and forest regime, but fire affecting untreated areas is probably most likely in forests with frequent fire, such as ponderosa pine in the Southwest. The likelihood of this potential outcome can also be estimated via Equation 1, coupled with consideration of treated and non-treated areas at the watershed scale. Within a given analysis area (e.g., watershed) and timeframe, the probability of high-severity fire affecting 1,000 untreated acres in a specific watershed is the same as the probability of it affecting 1,000 treated acres in the same watershed, other factors remaining equal. Under this outcome, MFT have negative effects on watersheds without any compensatory benefit from reductions in fire severity. The net watershed impacts are those from MFT plus fire.

A fifth, and the most ecologically desirable, discrete outcome is that fire affects treatments during the period that fuels are reduced and effectively reduces fire severity. This outcome is most likely in forests with frequent, low-severity fire regimes, such as some ponderosa pine forests in the Southwest. However, even in these forests, there is a low probability of higher severity affecting treated areas while fuels have been reduced (Rhodes and Baker, *in review*). This outcome is even less likely in mixed conifer systems due to lower fire frequency and the effect of weather on treatment effectiveness. Under this outcome, the net effects of MFT on watersheds are the benefits from reduced fire severity ?minus the aquatic and watershed costs incurred by treatment.

The likelihood associated with these discrete outcomes indicates that in most cases, MFT impacts will *not* be counterbalanced by reductions in the aquatic effects of higher-severity fire within treated areas. The most likely outcome of MFT for aquatic systems is that treatment impacts are added to impacts from fires that have not been reduced in severity by MFT.

The results in Tables 4 and 5 also provide some perspective on the assessment of the risks and effects of treatment of forest fuels versus those from non-treatment. Some previous assessments of these risks have inherently assumed that a single cycle of fuel treatment always reduces fire impacts in a treated watershed, subsequently reducing consequent watershed impacts from fire (Elliot and Miller, 2002; O'Laughlin, 2005). For instance, Elliot and Miller (2002) compared the erosional effects of fuel treatments with those from high-severity fire under the explicit assumption that high-severity fire was inevitable in the absence of treatment and the implicit assumption that treatments always reduce or eliminate the potential for high-severity fire.

These assumptions employed by Elliot and Miller (2002) and O'Laughlin (2005) mischaracterize likely treatment outcomes and associated impacts by greatly overestimating the potential effectiveness of treatments and the likelihood of higher-severity fire in the absence of treatments. There is general agreement that a single cycle of fuel treatments does not persistently reduce fuels. It is also unlikely that higher-severity fire affects treated areas while fuels are reduced. Even in ponderosa pine forests that burn relatively frequently, the results of Rhodes and Baker (*in review*) indicate that even after 17 cycles of treatments that contribute to cumulative watershed and aquatic impacts, only a scant majority of treated areas could potentially have fire severity reduced, on average. The results in Table 4 also indicate that high-severity fire is far from inevitable in areas left untreated. For instance, in forest types and regions where it is most likely to occur, high-severity fire is expected to affect a small fraction (<4%) of untreated areas, on average, over a 20-year period (Table 4).

Finally, available evidence indicates that when MFT are effective, they do not eliminate higher-severity fire, but rather incrementally reduce its occurrence (Schoennagel et al., 2004a). If risks of MFT versus non-treatment are to be credibly assessed, they must factor in the probability of divergent outcomes of treatments together with reasonable assessment of treatment effectiveness, as well as the effects of non-treatment (NRC, 1996). For these reasons, there is a high degree of certainty that binary comparison of the impacts of MFT versus high-severity fire greatly mischaracterize the risks posed to native fish and aquatic systems by treating or not treating forest fuels.

The Effects of Wildland Fire on Watersheds and Aquatic Systems

Fire has numerous negative and positive effects on aquatic systems. Wildfire impacts on aquatic resources often vary with the extent and severity of fire, although many other factors influence these impacts, including topography, soils, and climate. Fires that burn a relatively small portion of a watershed have relatively minor impacts on watershed functions and aquatic conditions (Minshall et al., 1989; Minshall et al., 1997; Gresswell, 1999; Beschta et al., 2004). Other factors remaining equal, the higher the burn severity, the greater the impacts on aquatic systems (Gresswell, 1999; Beschta et al., 2004).

In the following discussion of fire impacts, a few considerations should be kept in mind. First, the pattern of burn severity is typically patchy and discontinuous in many fires (e.g., Odion and Hanson, 2006), limiting their negative impacts, especially those from high-severity burns, on watershed and aquatic resources (Beschta et al., 2004). The impacts of fire on watershed conditions and processes are also transient. This temporal and spatial heterogeneity and discontinuity also causes high variability in watershed and aquatic system response to fire, complicating accurate prediction of the impacts of moderate- and high-severity fire on soils, runoff, erosion, and sediment delivery (Shakesby et al., 2000; Letey, 2001; Beschta et al., 2004).

Although burn patterns exhibit high variability, there are some general patterns in fire severity that also limit the negative effect of fire on aquatic systems. Riparian zones tend to burn at lower severity than uplands, as documented by USFS research (Fisk et al., 2004). This reduced level of fire severity in riparian zones is entirely consistent with their topography, moisture levels, and microclimate and the well-known effects of these attributes on the fuel moisture conditions and site-level weather conditions, and fire behavior.

Second, much available information on watershed response to fire is from burned watersheds that have also been affected by roads, logging, and grazing. All of these land uses can significantly affect postfire runoff and erosion, as well as postfire recovery of soils, vegetation, watershed processes, and aquatic systems (Rhodes et al., 1994; Kattleman, 1996; Beschta et al., 2004). The effects of these land uses on postfire watershed responses confound the identification the effects of fire alone on watershed and aquatic systems (Wondzell and King, 2003).

Third, much of the scientific literature on the effects of fire on watershed processes has focused on dramatic, episodic postfire hydrologic events triggered by large extreme storms on areas burned extensively at higher severity (Wondzell and King, 2003). Such events may not be representative of general watershed responses to fire. It also appears that there has been limited study of fires that produced muted watershed responses (Wondzell and King, 2003). Wondzell and King (2003) note that for these reasons, available scientific information may not provide a balanced perspective of the effects of fire on watersheds and aquatic systems.

Methods of assessing fire severity may also introduce some inaccuracy and bias in assessment of potential effects on watershed and aquatic resources. Watershed functions are most influenced by burn severity at the soil surface (USFS and USBLM, 1997a; Beschta et al., 2004). However, in many fires, fire severity assessments, including USFS Burned Area Emergency Recovery (BAER) inventories, are based on canopy conditions (e.g., Martinson et al., 2003; RSNF, 2004; Robichaud et al., 2003). Fire severity based on canopy conditions may not accurately reflect fire severity at the soil surface, because fire that burns a forest canopy at high severity can be low severity at the soil surface, due to differential burning (Romme et al., 2003a). Although the reverse situation is possible, it is rare (Romme et al., 2003a).

BAER assessments also tend to be confined to larger fires that burn at higher severity. Fire severity sampling is also typically limited spatially and based on remote sensing methods that are often not ground-truthed, which may limit their accuracy. For these reasons, it is likely that fire severity assessments tend to overestimate the extent of soils affected by higher-severity fire and underestimate the extent of soils affected by low-severity fire.

These potential sources of inaccuracy should be kept in mind because the following discussion of the extent of fire severity relies on assessments that are often based on canopy conditions, while the watershed impacts of fire are discussed in terms of burn severity at the soil surface. Table 6 describes some of the different aspects of fire by severity class in the forest canopy versus that at the soil surface, as summarized by Romme et al. (2003a).

Watershed Effects of Low-Severity Fire

Low-severity fire has minimal impacts on watershed and aquatic systems that usually persist for less than a year (e.g., Minshall et al., 1989; Minshall et al., 1997; Gresswell, 1999; Benavides-Solorio and MacDonald, 2001; Kershner et al., 2003; Rieman et al., 2003; Beschta et al., 2004). These impacts include nominal and transient increases in soil erosion and sediment delivery, mainly from some loss of soil cover. Low-severity fire has little effect on soil properties and conditions, resulting in little change in runoff. Vegetation usually recovers rapidly after low-severity fire, muting postfire erosional effects.

Low-severity fires consume some of the most flammable fuels, reducing fuel loads that can contribute to larger and more extensive fires. Therefore, low-severity fire reduces the potential for higher-severity fire until fuel loads rebound. It also aids in creating and maintaining an open low-density forest structure (Romme et al., 2003a; b).

Low-severity fire accounts for a significant amount of the area burned annually on public

<u>Table 6</u>. Aspects of fire severity in the forest canopy and at the soil surface (after Romme et al., 2003a).

Focus of Description	Term(s)	Definition
Effects On Forest Canopy And Understory Vegetation	High-Severity = Lethal = Stand-Replacing	Fire kills all or most canopy and understory canopy and understory trees and initiates a succession process that involves recruitment of a new cohort of canopy trees.
	Mixed-Severity = Intermediate Severity	Used in two different ways, depending on scale: <u>Within-stand</u> – fire kills an intermediate number of canopy trees (less than high-severity but more than low-severity), and may or may not lead to recruitment of a new canopy cohort; <u>Among-stand</u> – fire burns at high severity in some stands but at low or intermediate severity in others, creating a mosaic of heterogeneous fire severity across the landscape.
	Low-Severity = Non-Lethal = Non-Stand-Replacing	Fire kills only a few or none of the canopy trees but may kill many of the understory trees and does not result in recruitment of a new canopy cohort but creates or maintains an open, low-density forest structure.
Effects On Soil And Soil Organisms	High-Severity	Fire consumes all or nearly all organic matter on the soil surface, as well as soil organic matter in the upper soil layer, and kills all or nearly of the plant structures (for example, roots and rhizomes) in the upper soil layer; results in possible water repellency and slow vegetative recovery.
	Low-severity	Fire consumes little or no organic matter on the soil surface or in the upper soil layer, and kills few or no belowground plant parts; results in limited or no water repellency, and rapid vegetative recovery via re-sprouting.
BAER Definitions	High-Severity	Fire consumes areas of crown (i.e., leaves and small twigs); always stand-replacing.
	Moderate-Severity	Fire burns areas where the forest canopy was scorched by an intense surface fire but the leaves and twigs were not consumed by the fire; may be stand-replacing or not, depending on how many canopy trees survive the scorching.
	Low-Severity	Fire burns areas on the surface at such low intensity that little or no crown scorching occurred (may include small areas that did not burn at all); never or rarely stand-replacing.

lands over the past few decades (Table 7). Low-severity fire is estimated to have comprised slightly more than 40% of the area burned on national forests in the West from 1973-1998, based on the data of Robichaud et al. (2000). This likely varies considerably among fires, forest types, regions and years.

Low-severity fire typically accounts for almost half or more of the area burned, even in large fires burning under weather conditions extremely conducive to rapidly spreading fire burning at high severity fire (Table 7). For instance, in the 2002 Hayman Fire in Colorado, much of which burned during extreme fire weather, about half the area burned at low severity (Robichaud et al., 2003). The 2002 Rodeo-Chediski in Arizona burned during extreme fire weather in 2002 after a period of significant drought; about 45% burned at low severity or not at all (ASNF).

Although much of the 2002 Biscuit Fire in Oregon burned during extreme fire weather, about 45% of the area within the fire perimeter burned at low severity or not at all, based on the analysis of Harma and Morrison (2003). Based on surveyed fires, about 70% of the area burned in 2002 on USFS lands in the Pacific Northwest burned at low severity (Associated Press, 2002). In the two largest fires since 1999 on USFS lands in the Sierra Nevada, California, about 59-66% of the area within the fire perimeters was unburned or burned at low severity (Odion and Hansen, 2006).

Based on these data, there is a high degree of certainty that low-severity fire comprises a significant portion of most fires, including some large fires that burn during extreme fire weather in areas where altered fuel conditions may have increased fire severity. There is also a high degree of certainty that low-severity fire has minimal negative impacts on watersheds and aquatic systems.

Watershed Effects of Moderate-Severity Fire

Moderate-severity fires have impacts on watersheds and aquatic resources that are more pronounced and less fleeting than low-severity fire (Minshall et al., 1997; Robichaud et al., 2000; Benavides-Solorio and MacDonald, 2001; Beschta et al., 2004). However, this has a low degree of certainty due to the relative paucity of data on the effects of moderate-severity fire (Wondzell and King, 2003). The effects of moderate fire severity on watersheds appear to be less well studied than low- or high-severity fire with relatively few studies, such as Benavides-Solorio and MacDonald (2001) that discretely assessed effects of moderate-severity fire.

Moderate-severity fire increases erosion through loss of groundcover. It may also increase runoff and erosion through the loss of the evapotranspiration caused by tree mortality and the development of hydrophobic soils. However, hydrophobic soils do not always develop in response to fires of moderate, or even high, severity (Benavides-Solorio and MacDonald, 2001). When hydrophobic soils develop in response to fire, they typically persist for less than three years. During the period that they exist, hydrophobic soils do not have uniformly low infiltration rates, because the level of hydrophobicity decreases over time and is also reduced as soils are wetted (Shakesby et al., 2000; Letey, 2001; Huffman et al., 2001; Wondzell and King, 2003). Wetting sometimes completely eliminates hydrophobicity in soils, at least temporarily, while soils are wet (Letey, 2001; Shakesby et al., 2000; Huffman et al., 2001; Benavides-Solorio and MacDonald, 2001).

Elevated erosion of topsoil from moderate-severity fire reduces soil productivity. However, moderate-severity fire also provides benefits to soil productivity. Moderate-severity fire increases the recruitment of woody debris, needles and leaves that provide sources of organic matter vital to soil productivity. It plays an important role in providing soil nutrients in a form readily usable by plants (Brown et al., 2003).

Moderate-severity fire does not typically consume all leaves and twigs on trees. Postfire needlecast and recruitment of dead limbs and twigs from scorched trees helps to provide postfire soil cover, effectively reducing postfire soil erosion (Pannkuk and Robichaud, 2003).

Moderate-severity fire in riparian areas is likely to have both positive and negative effects on stream conditions. Tree mortality is likely to increase LWD recruitment to streams, which is beneficial to aquatic systems (Burton, 2003; Beschta et al., 2004; Karr et al., 2004). LWD is essential to the development of pools (Buffington et al. 2003) and habitat complexity that are vital to the production and survival of native salmonids (Meehan, 1991). However, depending on the degree of the loss of stream shade, moderate-severity fire in riparian areas likely contributes to elevated water temperatures that adversely affect native salmonids.

Although riparian areas tend to burn at lower severities than uplands, there is currently limited information on the extent and frequency of moderate-severity fire in riparian areas, because burn severity in these areas is not typically assessed discretely. However, based on the innate characteristics of riparian areas (e.g. topography, microclimate, etc.), and what is known about their effect on fire severity, there is a medium degree of certainty that riparian areas burn with a lower frequency and extent of moderate severity burns than occurs in uplands.

Moderate-severity fire is estimated to have comprised roughly 30% of the area burned on national forests in the West from 1973-1998, based on the data of Robichaud et al. (2000). This clearly varies among fires, regions, forests and years (Table 7). Based on surveyed fires, about 19% of the area burned on USFS in the Pacific Northwest in 2002 burned at moderate severity (Associated Press, 2002). Moderate-severity fire was estimated to have affected about 16-30% of the areas burned in five large fires in different forest types from 2000 to 2004 in California, Colorado, Arizona, and Oregon (Table 7).

There is a high degree of certainty that a sizable fraction of moderate-severity fire occurring in many forests is characteristic of those forests and their natural fire regimes, and, hence not a restoration concern. However, there is a low degree of certainty regarding how much of it is characteristic. This is due to the lack of comprehensive data on the burn severity in various forest types and the uncertainty regarding what is characteristic for the forests affected by the fires.

Watershed Effects of High-Severity Fire

High-severity fire can have pronounced impacts on watersheds, especially with respect to soils, runoff and sediment delivery (Minshall et al., 1997; Gresswell, 1999; Robichaud et al. 2000; 2003; Beschta et al., 2004; Burton, 2005). High-severity fire increases erosion through several mechanisms. It severely reduces soil cover, thus greatly increasing surface erosion. This can be further exacerbated by the development of hydrophobic soils that increase surface runoff, sometimes dramatically (Martin and Moody, 2001; Moody and Martin, 2001; Robichaud et al., 2003).

Increased runoff from areas extensively burned at high severity can greatly increase fluvial erosion in stream channels (Moody and Martin, 2001; Robichaud et al., 2003). The tree mortality caused by high-severity fire can trigger mass failures due to the loss of root

<u>Table 7</u>. Estimated fire severity on public lands at various temporal and geographic scales; the data may overestimate the amount of higher-severity fire at the soil surface, as discussed in the text.

Geographic and Temporal Scale	Fire Weather and Fire Size	Affected Forest Types	Unburned and Low Severity (% fire area)	Moderate Severity (% fire area)	High Severity (% fire area)	Data Source
Western USFS lands, 1973-1998	Unknown, but likely highly variable	Many	42%	30%	28%	Robichaud et al., 2000
USFS Region 6 (OR and WA), 2002	Unknown, but likely highly variable	Many	69%	19%	12%	AP, 2002
Hayman Fire, CO, 2002	Extreme, large	Ponderosa pine, mixed conifer, and subalpine	49%	16%	35%	Robichaud et al., 2003
Rodeo-Chediski, AZ, 2002[a]	Extreme, large	Ponderosa pine, chaparral, mixed conifer, and subalpine	45%	26%	27%	ASNF, 2003
Biscuit Fire, OR, 2002	Extreme, large	Ponderosa pine, mixed conifer, maritime, and subalpine	45%	25%	31%	Harma and Morrison, 2003
McNally Fire, CA, 2002	Variable, large	Jeffrey pine, ponderosa pine, mixed conifer, and subalpine	59%	30%	11%	Odion and Hanson, 2006
Storrie Fire, CA, 2000	Variable, large	Jeffrey pine, ponderosa pine, mixed conifer, and lodgepole	66%	19%	15%	Odion and Hanson, 2006

[a] Approximately 2% of the fire area did not have burn severity determined (ASNF, 2003).

strength combined with increased soil saturation (Burton, 2005).

In some cases, the postfire erosion from areas burned extensively at high severity can approach that from roads on a per unit basis (Robichaud et al., 2003; Moody and Martin, 2001), although these results may be extreme cases that are broadly representative because they occurred on soils highly prone to erosion. However, unlike erosion from roads, increased erosion in response to fire is relatively transient.

Erosion triggered by high-severity fire declines over time (Moody and Martin, 2001; Wondzell and King, 2003). This is likely due to several mechanisms, including postfire revegetation, the recovery of soil properties and infiltration rates (Shakesby et al., 2000; Huffman et al., 2001; Wondzell and King, 2003), and the recruitment of needles, branches, and other woody material from dead trees to the soil surface (Pannkuk and Robichaud, 2003). Elevated surface erosion from high-severity fire typically persists for about three years in most systems (Robichaud, 2000; Moody and Martin, 2001; Robichaud et al., 2003). Mass failures in response to fire may lag several years after fire (Istanbulluoglu et al., 2004).

Natural rates of postfire recovery of groundcover are sometimes rapid, triggering rapid reductions in surface erosion rates after fire. Rhodes (2003) documented that groundcover was >80% in unlogged areas a little more than one year after burning at high severity in the 2002 McNally fire in the Sierra Nevada, California (See Photograph 4 on pg. 62). In the areas burned at high severity in the 2004 Power fire in the Sierra Nevada, groundcover was as high as 91% approximately one year after fire (Rhodes, 2005) (See Photographs 5 and 6 on pg. 63).

High-severity fire can sometimes dramatically increase runoff. Postfire increases in runoff generally decline with time and appear to persist for less than three years, even in areas that have been extensively burned at high severity (Moody and Martin, 2001; Robichaud et al., 2003). Similar to postfire erosional responses, this is likely due to revegetation and the recovery of soil properties.

There is a high degree of certainty that high-severity fire increases erosion and runoff. However, there is a low degree of certainty regarding the frequency of severe increases in response to fire. Some fires that have burned large areas at high severity have not triggered extreme levels of postfire erosion and runoff at the watershed scale, including the 1988 Yellowstone fire (Minshall et al., 1989; 1997) and the Biscuit fire (RSNF, 2004). Wondzell and King (2003) noted that major runoff and erosion events in response to high severity burns are relatively rare in the Pacific Northwest. This indicates that extreme increases in erosion, sediment delivery, and runoff in response to high-severity fire are not a certainty. Documented examples of severe elevated erosion and runoff from severely burned areas (Moody and Martin, 2001; Allen et al., 2002; Robichaud et al., 2003; Burton, 2005), appear to be largely relegated to the Southwest and Intermountain West.

The apparent regional variation in postfire response is probably due to the interaction of climate with postfire conditions. Major runoff and erosion events in response to higher-severity fire appear to be contingent on the extensive development of hydrophobic soils in conjunction with the occurrence of higher-intensity storms, during a time when soils are relatively dry, within 1-2 years after fire (Wondzell and King, 2003). Although this confluence of postfire conditions and events is not a certainty in any region, it is most likely in the climates of Southwest and

Intermountain West and is least likely in the Pacific Northwest, which has few intense rain events during drier periods (Wondzell and King, 2003).

In the Pacific Northwest, frequent low intensity rains probably reduce hydrophobicity in soils through progressive wetting (Wondzell and King, 2003). Rapid postfire revegetation is another potential contributing factor that limits the propensity for elevated surface erosion after fire. Soils in the Pacific Northwest may also have relatively high moisture content that limits the development of hydrophobic soils in response to higher-severity fire.

There is a high degree of certainty that higher-severity fire can cause hydrophobic soils to develop. However, there is low degree of certainty regarding the frequency and extent at which this occurs. With some notable exceptions (e.g., Martin and Moody, 2001), the existence and extent of hydrophobic soils in burned areas are often not verified via direct measurement (Beschta et al., 2004). It can be difficult to ascertain the cause of hydrophobic soils where they exist, because such soils can occur naturally on unburned areas (Benavides-Solorio and MacDonald, 2001; Beschta et al., 2004). High-severity fire does not always cause hydrophobic soils (Benavides-Solorio and MacDonald, 2001) or increase the hydrophobicity of soils that are already hydrophobic prior to fire (Shakesby et al., 2000).

The development of hydrophobic soils in response to fire is dependent on a number of factors besides fire severity, including vegetation, soil texture, and soil moisture conditions during fire (Robichaud, 2000; Letey, 2001), all of which can vary considerably in burned terrain. All of these factors, together with a lack of extensive and complete inventory data on soil conditions after fires, contribute to the low degree of certainty regarding the frequency and extent of the development of hydrophobic soils in response to higher-severity fire.

Hydrophobic soils reduce infiltration rates in forest soils, but not to a degree that causes elevated surface runoff from *all* snowmelt and rainfall events. Undisturbed forest soils typically have relatively high infiltration rates, and, hydrophobicity transiently reduces these by about 50%, on average, based on the data in Wondzell and King (2003). Because these infiltration rates tend to increase over time and as soils are wetted (Letey, 2001), the intensity of rainfall or snowmelt needed to exceed infiltration rates in hydrophobic soils is lower during times when soils are dry and/or soon after fire. These effects also likely explain why such events are not triggered by snowmelt, because it usually wets soils progressively and is of relatively low intensity (Wondzell and King, 2003).

High-severity fire can also lead to increased rates of erosion by mass failure in susceptible terrain. The occurrence of mass failures depends on the magnitude of snowmelt, rain, or rain-on-snow events, but high-severity fire has several effects that increase the likelihood of mass failures (Wondzell and King, 2003). These include increases in soil moisture due to the loss of evapotranspiration from tree mortality, loss of root strength as the roots of dead trees decay, and increased input of precipitation and snowmelt due to the loss of the forest canopy (Wondzell and King, 2003). Burned areas tend to be most susceptible to mass wasting during a window of about 5-10 years after fire, which is likely due to the time needed for roots to decay and for the roots of trees regenerating after fire to begin to stabilize burned areas.

<u>Photograph 4</u>. Approximately 1.3 years after this area burned at high severity in the 2002 McNally Fire on the Sequoia National Forest, CA, measured groundcover was more than 85%, with more than half of supplied by live vegetation. If undisturbed, groundcover will continue to increase via re-growth of vegetation and recruitment of wood and needles from trees. Compare and contrast this rapid revegetation after high-severity wildland fire to the severe soil damage and lack of revegetation about 2.5 years after pile burning in Photo 3. *Photograph: J. Rhodes.*

<u>Photographs 5 & 6</u>. These photos were taken in two different areas several miles apart approximately one year after they burned at high severity in the 2004 Power Fire on the Eldorado National Forest, CA. Photo 5 shows an area where measured groundcover was about 87% approximately one year after burning at high severity. Photo 6 shows a broader-scale view of an area where measured groundcover was over 90% approximately one year after burning at high severity. Groundcover in both areas will continue to increase due to vegetative regrowth and recruitment of needles and wood from burned trees. Compare and contrast this rapid revegetation approximately one year after wildland fire burned the two areas at high severity to the severe soil damage and lack of revegetation about 2.5 years after pile burning in Photo 3 (pg. 22). *Photographs: J. Rhodes*.

Based on the foregoing, there is a high degree of certainty that high-severity fire sometimes generates dramatic increases in runoff and erosion. Conversely, there is high degree of certainty that high-severity fire does not always result in extreme increases in postfire erosion and runoff. This is because fire severity is often patchy, with transient impacts, and the degree of impacts are partially dependent on postfire climatic events (Wondzell and King, 2003), as well as the severity, contiguity, and persistence of soil impacts (Shakesby et al., 2000). For these reasons, there is a low degree of certainty regarding the predictability and frequency of the occurrence of severe postfire hydrologic events triggered by higher-severity fire. Other factors contribute to the low degree of certainty regarding the magnitude and frequency of postfire erosion and runoff responses to high-severity fire:

- limited information about the frequency, contiguity, extent of hydrophobic soils that develop in response to high-severity fire;
- the lack of systematic data on postfire erosion and runoff responses from a large number of high-severity wildfires across several regions.

There is a high degree of certainty that high-severity fire reduces soil productivity by increasing erosion and consuming soil organic matter that is essential to soil productivity. However, high-severity fire also has effects that improve soil productivity over time. High-severity fire typically consumes less than 10-15% of the total organic matter in a forest stand (Franklin and Agee, 2003). After high-severity fire, much of this material (whole trees, limbs, needles) ultimately falls to the forest floor, providing sources of organic matter critical to soil productivity. While high-severity fire can volatilize nutrients, it also makes nutrients available in a form that is more readily usable by vegetation. Therefore, there is a high degree of certainty that high-severity fire also provides benefits to soil productivity.

There is a high degree of certainty that pulsed erosion from higher-severity fire increases turbidity and sedimentation. There is also a high degree of certainty that high-severity fire in riparian areas contributes to elevated summer water temperatures via the loss of shade. However, there is medium degree of certainty that riparian areas tend to burn at a lower severity and more infrequently than uplands, due to their topographic attributes, moisture levels, and microclimate.

High-severity fire often comprises the smallest fraction of burned area by severity class, even in large fires burning during extreme fire weather (Table 7). Based on Robichaud et al. (2000), about 27% of areas burned on USFS land in the West from 1973-1998 burned at high severity. In the 2002 Hayman Fire in Colorado, much of which burned during extreme fire weather, about 35% the area burned at high severity (Robichaud et al., 2003). Despite initial characterizations of the severity of the 2002 Rodeo-Chediski Fire in Arizona, about 27% of the area burned at high severity (ASNF, 2004).

In the 2002 Biscuit Fire, much of which burned during extreme fire weather, about 31% of the area burned at high-severity (Harma and Morrison, 2003). About 12% of the area burned on USFS lands in the Pacific Northwest in 2002 burned at high severity, based on surveyed fires (Associated Press, 2002). In the two largest fires in the Sierra Nevada since 1999, areas burned by high-severity fire comprised about 11-15% of the fire area (Odion and Hansen, 2006). Therefore, there is a high degree of certainty that high-severity fire affects the minority of area burned by wildfire, as evidenced by several large fires that have recently burned during extreme fire weather in areas estimated to have elevated fuel levels that increase the propensity for high-severity fire.

There is a high degree of certainty that a sizable fraction of the high-severity fire occurring in many forests is characteristic of those forests and their fire regimes. For instance, all of the high-severity fire in the Hayman Fire may have been within the bounds of the natural fire regime of the affected forest types, although the homogeneity and scale of mortality from fire in some large patches may have been unprecedented in the available historical record (Romme et al., 2003a; b). In the Sierra Nevada, Odion and Hansen (2006) found that area burned at higher severity in several larger, recent fires was within the bounds of the range of natural variation.

However, there is a low degree of certainty regarding how much of the areas burned at high severity in recent fires is characteristic of affected fire regimes. This is due to the lack of comprehensive data on the burn severity by forest types and the uncertainty regarding the fire regimes of some of the affected forests.

Effects of Higher-Severity Fire on Native Salmonids and Aquatic Ecosystems

Higher-severity fires are important agents of disturbance that have positive and negative effects on aquatic systems in both the short and long terms (Gresswell, 1999; Rieman et al., 2003; Burton, 2003; Lindenmayer et al., 2003; Beschta et al., 2004; Karr et al., 2004; Burton, 2005; Rieman et al., 2005; DellaSala et al., 2006). Much of the following discussion focuses on the effects of fire on native salmonid populations, due to the level of information available. Native fish also provide an indication of aquatic ecosystem conditions because they integrate a wide variety of biophysical stream conditions.

Increases in fine sediments and channel width are typical responses to higher-magnitude postfire runoff and erosion events (Gresswell, 1999; Burton, 2005). Increased postfire erosion also likely contributes to the loss of pool volume. Based on available information, the magnitude of these habitat changes in response to postfire sediment delivery and runoff is expected to increase with increased levels of postfire erosion and runoff, other factors remaining equal (Rhodes et al., 1994).

These impacts of postfire sedimentation and runoff are typically transitory (Gresswell, 1999). Burton (2005) suggested sedimentation from higher-severity fire affects fine sediment levels in streams for less than 5-10 years on the Boise National Forest. In some cases, fine sediment levels after postfire runoff and erosion events were ultimately lower than before fire, a condition Burton (2005) ascribed to winnowing of fine sediment by postfire runoff concurrent with gravel recruitment from postfire erosion. The transient nature of postfire impacts on erosion likely provides an explanation of why postfire channel responses are also transient.

The transitory effects of fire on aquatic habitat conditions are in strong contrast to anthropogenic impacts, such as roads and grazing, which persistently elevate erosion and sediment delivery, resulting in consistently degraded habitat conditions in many areas (Rhodes et al., 1994; Espinosa et al., 1997). In a regional assessment of changes in large pools in streams in the Columbia River basin, McIntosh et al. (2000) found that some streams in roadless areas subjected to fairly recent wildland fire did not lose large pools over a period of approximately 50 years, while, over the same time period, comparable streams with roads and grazing, but no fire, lost a large amount of pools in a statistically significant fashion. Persistently elevated sediment delivery from management activities was identified as the prime cause of pool loss in streams with roads and grazing (McIntosh et al., 2000).

Changes in aquatic habitat conditions affected by runoff and sedimentation do not always occur in response to fire. Spina and Tormey (2000) documented that a fire in Southern California had little effect on aquatic habitat features relative to pre-fire conditions; there were no statistically significant differences between pre- and postfire conditions. The large 2002 Biscuit Fire in Oregon appeared to have negligible effects on an array of aquatic habitat conditions affected by runoff and sedimentation, according to RSNF (2004).

The pulsed sediment recruitment from higher-severity fire rejuvenates aquatic habitats (Minshall et al., 1997; Rieman et al., 2003; Karr et al., 2004; Burton, 2005; Rieman et al., 2005). Higher-severity fire also increases LWD recruitment, which increases habitat complexity and fulfills a number of roles vital to productive aquatic habitats (Meehan, 1991). These combined positive impacts likely contribute to the transience of postfire degradation of aquatic habitat.

For these reasons, there is a high degree of certainty that postfire watershed response often transiently degrades fine sediment and channel conditions. However, there is a low degree of certainty regarding the frequency and magnitude of this habitat degradation, due to limited data. There is also a low degree of certainty regarding the frequency and magnitude of postfire runoff and erosion events. There is a medium degree of certainty that higher-severity fire also has positive impacts on aquatic habitat conditions via increased LWD recruitment and habitat rejuvenation.

The impact of higher-severity fire on native fish populations is strongly influenced by habitat and population connectivity. Fish populations with habitat connectivity have access to distributed habitats in stream networks that can serve as refugia from short-term, severe events, limiting the impacts on affected populations (e.g., Gresswell, 1999; Rieman et al., 2003; Dunham et al., 2003a; Beschta et al., 2004; Burton, 2005).

High population connectivity also allows fish populations to recolonize habitats after severe postfire events (Gresswell, 1999; Burton, 2005). However, many populations of native salmonids are now isolated in fragmented habitats with limited habitat connectivity due to habitat degradation and/or impassable barriers, such as road culverts (USFS and USBLM, 1997a; USFWS, 1998; Kessler et al., 2001; Dunham et al., 2003; Burton, 2005).

Higher-severity fire sometimes triggers flow events with high turbidity and low levels of dissolved oxygen that can cause direct mortality of native fish in affected streams (Gresswell, 1999; Burton, 2005). Severe postfire erosion events have been documented to eliminate native trout in affected streams. However, this is a transient condition in areas with habitat or population connectivity (Gresswell, 1999; Rieman et al., 2003; Burton, 2005). Even where fire impacts cause significant levels of fish mortality, it is not always complete or uniform, with some native fish surviving, possibly due to refugia within affected streams (Gresswell, 1999; Burton, 2005).

Studies have documented that native fish populations with habitat connectivity often rapidly recolonize streams after native fish have been temporarily eliminated by postfire impacts (Gresswell, 1999; Rieman et al., 2003; Burton, 2005). Burton (2005) found that native fish populations in burned streams rebounded within a year after fire. The recolonization of fish habitats sometimes results in greater abundance of native fish after fire than before fire (Gresswell, 1999; Rieman et al., 2003; Burton, 2005). Rieman et al. (1997) documented that native redband trout and bull trout recolonized habitats in tributaries of the Boise River in Idaho in less than three years after being extirpated by fire impacts; the density of both species was greater than in areas unaffected by fire. Burton (2005) documented that five years after extirpation, redband trout numbers in recolonized habitats were higher than before apparent extirpation by postfire flood and erosion events in two streams in Idaho.

Several factors probably contribute to the transience of fire impacts on native fish. The first is the pulsed nature of fire impacts and the transience of fire's impacts on habitat conditions. The second is the positive effects of fire on habitat conditions. The third is the mobility of native fish, which are able to find refugia in areas with high habitat connectivity. The fourth is that chronic, widespread and persistent habitat degradation appears to have greater negative effects on fish populations than acute, patchy, transient impacts produced by fire (Gresswell, 1999; Rieman et al., 2003; Dunham et al., 2003b).

Based on available information, there is a medium degree of certainty that high-severity fire has no persistent negative impacts on fish populations that have population and habitat connectivity (Gresswell, 1999; Rieman et al., 2003; Burton, 2005). In such situations, the reduction of high-severity fire and consequent impacts may not significantly benefit such aquatic populations (Rieman et al., 2003; Burton, 2005), although this has a low degree of certainty.

There is a medium degree of certainty that, when they occur, major adverse impacts of fires are deleterious to isolated fish populations that are incapable of finding refugia from postfire hydrologic events and/or recolonizing habitats after fire impacts abate. However, this appears to be a relatively rare occurrence. In a review of studies on the effects of fire on aquatic ecosystems, Gresswell (1999) noted that out of the many cases evaluated, permanent extirpation of isolated fish populations by fire has only been documented in one case in response to an extreme postfire runoff and erosion event (Rinne, 1996).

Population and habitat fragmentation are caused by physical barriers, habitat degradation, and population extirpations (USFS and USBLM, 1997a; Dunham et al., 2003b; Rieman et al., 2003). Although isolated and fragmented populations of native fish might benefit from reductions in high-severity fire and its watershed effects, it is unlikely to help restore isolated populations unless the causes of fragmentation are effectively addressed and connectivity is restored (Dunham et al., 2003b; Rieman et al., 2003; Burton, 2005; Rieman et al., 2005). Based on available information, there is a high degree of certainty that reductions in fire severity and its impacts will *not* provide long-term benefits for imperiled fish populations if the major causes of population decline, fragmentation, and habitat degradation continue unabated or are intensified.

Perspective: A Comparison of the Magnitude and Persistence of the Aquatic Impacts of Wildfire to Those From Land Management Activities

There is a high degree of certainty that livestock grazing and roads have impacts that are more numerous, enduring, intense, and pervasive than fire. Roads and grazing compact soils, alter hydrologic processes, elevate erosion, reduce soil productivity, and spread noxious weeds (Fleischner, 1994; Belsky et al. 1999; Beschta et al., 2004).

Due to their extent and effects, grazing and roads are typically the greatest management-induced sources of sediment delivery over extensive areas of public lands (Rhodes et al., 1994; CWWR, 1996). For the same reasons, roads and grazing are also primary causes of the decline in range and abundance of many aquatic species on public lands, including imperiled amphibians and fish (Sublette et al., 1990; Rhodes, 1994; Henjum et al., 1994; Duff, 1996; CWWR, 1996; USFS and USBLM, 1997a; USFWS, 1998; USFS, 1999).

Road impacts on soil conditions, vegetation, and hydrologic processes are particularly enduring. Grazing impacts on watersheds are also persistent and typically do not undergo significant recovery unless grazing is eliminated or sharply curbed (Belsky et al., 1999). Riparian vegetation may respond quickly to grazing cessation, but recovery of soils and channel form occurs slowly after grazing cessation (Rhodes et al., 1994).

Although higher-severity fire can cause severe soil erosion and topsoil loss, its effects on runoff and soil hydrology are transient, seldom lasting more than three years. The effects of grazing and roads on soil hydrology are more persistent and often more negative than those from fire.

For instance, severe fire can temporarily reduce infiltration rates by about 50% if hydrophobic soils develop, based on the data of Wondzell and King (2003). When it occurs, hydrophobicity declines with time and moisture content, seldom persisting for more than three years. In contrast, grazing and roads persistently reduce infiltration rates by about 85% and 95-99%, respectively (Figure 2). Due to the extremely low infiltration rates on roads, they generate surface erosion and runoff in response to frequent, low-intensity rainfall and snowmelt events, for as long as the road exists, resulting in persistent and chronic degradation of water quality and aquatic habitats. This is not the case when fire causes hydrophobic soils to develop temporarily (Wondzell and King, 2003), and fire does not always cause hydrophobic soils. There is a high degree of certainty that compaction from grazing and roads always reduce infiltration rates. For instance, Cowley (2002) calculated that the hooves of a 1,000 pound cow exert more than five times the pressure per square inch on soils and streambanks than that from a bulldozer.

Fire does not compact soils and thereby reduce the ability of soils to store water as grazing (Kauffman et al., 2004) and roads do. Full recovery from soil compaction typically requires 50-80 years after the complete cessation of impacts (USFS and USBLM, 1997a; Beschta et al., 2004).

The locations of roads and grazing elevate their impacts. Grazing impacts are commonly most concentrated and intense in riparian areas, causing bank damage, elevated sedimentation, reduced stream shading, increased water temperature, and reduced habitat complexity and quality (Rhodes et al., 1994; Belsky et al., 1999).

Due to historic development patterns, significant portions of road networks are in riparian areas (USFS and USBLM1997a; 1997; USFS, 2001; CNF, 2003), and in some watersheds they are concentrated there (Rhodes et al., 1994). Roads in riparian areas vastly elevate sedimentation and surface runoff and eliminate stream shading and the recruitment of large woody debris (LWD), while interrupting groundwater flow to streams in an enduring fashion.

In contrast to roads and grazing, higher-severity fire does not target riparian areas which likely burn at a lower severity than uplands (Fisk et al., 2004) and likely at a lower frequency. Fire provides important aquatic benefits, including a bonanza of LWD recruitment to streams. Grazing and roads provide no ecological benefits to aquatic systems.

Figure 2. Measured mean reductions in infiltration rates due to high-severity fire in CO, NM, OR, and ID (Wondzell and King, 2003); grazing in OR (Kauffman et al., 2004); and roads (Luce, 1997). The losses in infiltration rates caused by grazing and roads are vastly more enduring, less patchy, and less temporally variable than the 1-3 year span of reduced infiltration capacity sometimes caused by higher-severity fire.

Data on conditions in 11 national forests in the Sierra Nevada indicate that grazing and roads affect a much greater area on an annual basis than high-severity fire does (Table 8). In these national forests, higher-severity fire affects an average about 15,500 acres annually, based on data for fire area from 1970-2003 (USFS, 2004) and fire severity from 1973-1998 (Robichaud et al., 2000). Importantly, much of this fire area is characteristic of the natural fire regimes and, hence, not an ecological aberration.

In contrast, roads occupy almost 106,000 acres in the Sierra Nevada, based on data from USFS (2000a) and an assumed mean road width of 30 feet. Therefore, roads annually affect about seven times the area annually affected by high-severity fire (Table 8). In these same forests, grazing is allowed on active allotments that have a total area of about 7.1 million acres (USFS, 2000a). While grazing impacts are not uniform on active allotments, they are extensive. The area of active allotments on these 11 national forests is more than 460 times the mean area annually affected by high-severity fire (Table 8).

The USFS model for "Equivalent Roaded Area" (ERA) used on Sierra Nevada national forests provides another perspective on the differences in the magnitude of the aquatic impacts conferred by roads, grazing, and high-severity fire. This model uses coefficients to convert areas affected by different activities to a common impact "currency" based on the estimated intensity of the activities' impacts on per unit area basis (Menning et al., 1996). The use of the USFS ERA method together with data on the amount of roads in these forests indicates that the annual impacts of existing roads are more than *38 times* those from high-severity fire (Table 8). Using the ERA factor for grazing suggested by Menning et al. (1996), annual grazing impacts are about *34 times* those from high-severity fire on these 11 national forests (Table 8). Based on these data, there is a very high degree of certainty that roads and grazing, both of which cause significant and enduring damage to soils, watersheds, and aquatic resources on an annual basis, annually affect an area that is many times greater than that affected annually by high-severity fire in the Sierra Nevada. Due to the nature of these impacts, there high degree of certainty that roads and grazing negatively affect watersheds within this analysis area to a far greater degree than does high-severity fire.

Table 8. Area of annual watershed impacts in the planning area for the Sierra Nevada Forest Planning Amendment (SNFPA), spanning 11 national forests in the Sierra Nevada, CA. ERA acres for roads and high-severity fire were calculated from coefficients from the USFS ERA model as excerpted in Menning et al. (1996). ERA acres for grazing were calculated from coefficients for grazing as suggested by Menning et al. (1996)

Activity or Impact	Area Annually Affected (acres)	Percent of Total SNFPA Analysis Area Annually Affected	Ratio of Affected Area to Area of High-Severity Fire	ERA (acres)	Ratio of ERA Area to High-Severity Fire ERA Area
Roads	105,455	0.9	7	105,455	38
Grazing	7,165,085	62.1	462	95,296	34
Mean Annual Estimated High-Severity Fire	15,500	0.1	--	2,790	--

This general result likely applies to other regions, where it may even be more pronounced. For instance, road density on many national forests in the interior Northwest is considerably higher than it is on national forests in the Sierra Nevada (Karr et al., 2004). Karr et al. (2004) noted that in 1994, road densities on three national forests in the US Northwest averaged 2.5 miles per square mile and attained 11.9 miles per square mile in some watersheds (Henjum et al. 1994), while the national forests of California's Sierra Nevada have a mean road density of about 1.7 miles per square mile at the regional scale.

Wuerthner (2002) estimated that about 69% of the area of Western national forests is subject to grazing. Several national forests in the Interior Northwest have more than 80% of their area subjected to grazing (J. Rhodes, unpublished data). The mean annual area burned by high-severity fire is lower in many national forests than it is in the Sierra Nevada, based on regional analysis of the occurrence of high-severity fire (Table 4). Therefore, there is a medium degree of certainty that this general pattern holds for many other regions and national forests.

The Consistency of Mechanized Fuel Treatments With Aquatic Restoration Needs and Priorities

Assessments have noted that reductions in fire severity alone, if realized, are unlikely to help restore native fish and aquatic ecosystems, because other stressors are greater threats to aquatic ecosystems and constrain their improvement. As discussed, there is general agreement that the primary restoration measures needed to improve water quality, channel form, and aquatic habitats include:

- Full protection of roadless and riparian areas from degradation due to anthropogenic disturbances (Henjum et al., 1994; Karr et al., 2004);
- Reductions in the extent and impact of road systems (USFS et al., 1993; Rhodes et al., 1994; ECONorthwest and Pacific Rivers Council, 2002);
- Reduction in impacts from domestic livestock grazing (Rhodes et al., 1994; CWWR, 1996; Duff, 1996);
- Reduction in the impacts of water withdrawals (CWWR, 1996; Rieman et al., 2003);
- Re-establishment of habitat and population connectivity by removing migration barriers caused by habitat degradation or physical obstructions (Rieman et al., 2003; Beschta et al., 2004);
- Reductions in sediment delivery from management activities (Rhodes et al., 1994; CWWR, 1996).

These are primary restoration measures because they address existing impacts that are extensively damaging to aquatic systems.

Even if they effectively reduced fire severity, MFT do not advance most of these protection and restoration needs. MFT do not reduce grazing impacts, water withdrawals, or the causes of the fragmentation of aquatic populations.

MFT have negative impacts on restoration needs related to roads. They increase road impacts by elevating road use and maintenance. Because they are likely to be extensive and repeated, MFT has a chilling effect on efforts to reduce the extent and impacts of roads. These same aspects of MFT, coupled with the current lack of the protection of roadless and riparian areas, increase the propensity for damage to these areas.

There is some limited potential for MFT to aid in reducing the negative effect of sediment delivery on imperiled aquatic biota by reducing fire severity. In some cases, this might significantly reduce sediment delivery that would otherwise be triggered by postfire events. In such situations, the benefits of MFT are likely to greatly outweigh the negative impact of MFT on disturbance levels and sediment delivery. However, these situations are likely rare, even in fire regimes where MFT are most likely to encounter high-severity fire and effectively reduce it (Rhodes and Baker, *in review*; Table 4). In the majority of situations, the effects of MFT are likely to be negative and additive to those impacts of fire, if fire occurs. Higher fire severity does not pose an imminent and extensive threat to aquatic ecosystems, based on its frequency and extent of occurrence (Table 4).

MFT will often conflict with other restoration needs. One of the most effective measures for restoring soil productivity is to leave affected areas undisturbed to prevent additional compaction, erosion, and loss of organic matter sources and coarse woody debris (Kattleman, 1996, USFS and USBLM, 1997; Beschta et al., 2004). MFT conflict with this approach. Ground-based machinery, road use, and maintenance of existing road networks also conflict with efforts to stem the spread of noxious weeds. MFT coupled with repeated treatments likely aids in the establishment of exotic vegetation (Dodson and Fielder, 2006).

As noted by ECONorthwest and Pacific Rivers Council (2002), high priority restoration measures for aquatic systems have the following attributes:

1. They address the root causes of pressing problems;
2. They address impacts that are extensive and imminent;
3. Available data, case histories, and other scientific information indicate the restoration measures will be generally effective at treating the cause of degradation;
4. They do *not* convey a high risk of additional damage that is likely to outweigh potential benefits.

Based on the foregoing, there is a high degree of certainty that efforts to reduce fire severity and its aquatic effects via MFT do not complement aquatic restoration needs and are not among major restoration priorities for aquatic systems. MFT are unlikely to be effective in most cases and do not address the most pressing restoration needs for aquatic ecosystems. It is likely that MFT will commonly and extensively incur ecological costs that are not outweighed by their benefits. They address neither the root causes of aquatic degradation nor the root causes of uncharacteristic fire behavior in forests with altered fire regimes.

Other restoration approaches are highly likely to effectively help restore aquatic systems while incurring negligible or limited ecological costs (ECONorthwest and Pacific Rivers Council, 2002). For instance, the elimination of grazing in damaged riparian areas is known to reliably provide numerous aquatic benefits, without conferring any ecological costs (Meehan, 1991; Rhodes et al., 1994; Belsky et al., 1999; ECONorthwest and Pacific Rivers Council, 2002).

Similarly, reductions in water withdrawals are highly unlikely to confer ecological costs to aquatic systems. Reduction in the extent of road networks and removal of physical barriers associated with roads do confer some temporary ecological costs, though these are highly likely to be outweighed by the benefits (ECONorthwest and Pacific Rivers Council, 2002). These activities also address the root causes of extensive and major threats to aquatic ecosystems.

Potential for Adaptive Management to Limit Aquatic Damage from Mechanized Fuel Treatments

Adaptive management involves attempts to "learn by doing," through iterative monitoring of outcomes with use of the monitoring information to guide future activities. It has been suggested as a means to limit ecological damage from MFT and improve their effectiveness (Allen et al., 2002; Rieman et al., 2003; Graham et al, 2004). However, there are several formidable obstacles to doing so for aquatic resources (Ziemer et al., 1991a; Ziemer, 1994; Rhodes et al., 1994; Rhodes, 1998). As Ziemer (1994) noted, the notion of effective use of adaptive management to fine tune activities, while protecting watersheds and aquatics "…is an attractive, but ecologically naive idea."

Aquatic degradation is often lagged in time, occurring well after on-site causes have been fully implemented. This leaves no possibility for fine-tuning or rapidly reversing on- and off-site impacts within the affected watershed (Ziemer et al., 1991a; Ziemer, 1994; Rhodes et al., 1994). For instance, if cutting and burning in a municipal watershed (e.g., SFNF; 2004) significantly elevate suspended sediment levels in a protracted fashion, little can be done to rapidly reverse the cumulative effects accruing at the watershed scale. The elevated suspended sediment levels in a municipal watershed might require building expensive water treatment facilities, the cost of which cannot be reversed.

Second, adaptive management requires both adequate monitoring and detection of change. Because of the high variability in aquatic systems, only dramatic and persistent changes are typically detectable, even with first-rate monitoring (Rhodes et al., 1994; Ziemer, 1994). Adverse impacts that are not detectable with conventional monitoring can still have considerable ecological and societal impacts. For instance, consider the case of a municipal watershed, where variability in suspended sediment levels is such that only a persistent change of greater than 20% over three years is detectable. If the actual change is only 17% over this time period, it would not be detectable by monitoring, but it would have significantly degraded drinking water, possibly to the point of requiring additional treatment facilities. Notably, adequate monitoring of aquatic impacts is seldom done comprehensively.

Third, aquatic impacts are usually caused by cumulative effects. This makes it difficult to unequivocally link the monitored effects to specific activities (Ziemer, 1994; Rhodes, 1998). Nonetheless, it is fairly common for management entities to insist on a fairly clear cause and effect relationship before considering changes in on-going practices (Rhodes et al., 1994; Hirt, 1996; Rhodes, 1998).

Fourth, activities may have irreversible impacts, such as extirpation of imperiled species or loss of irreplaceable topsoil. This is in direct conflict with one of the prime guidelines for responsible use of adaptive management: the impacts of the activities should be reversible (Ludwig et al., 1993).

There are major institutional barriers to effective adaptive management. Bureaucracies are resistant to change, often more committed to maintaining status quo direction than dealing with information indicating that management corrections are needed (Worster, 1985; Hirt, 1996; Wilkinson, 1998). There is considerable empirical evidence that such information is often suppressed and that those who collect such information often have their careers truncated (Wilkinson, 1998). Since adaptive management requires rapid response to information, these are formidable obstacles to its use.

Learning by doing requires learning by what has been done. Public land management has a consistent track record of failing to do so. To provide but one of many possible examples, current levels of riparian protection are inadequate on many national forests with habitats for imperiled native trout (May, 2000). This remains the case despite legions of studies spanning more than 20 years demonstrating the importance of riparian areas to a multitude of critical aquatic processes and conditions, and despite binding legal mandates to protect water quality and aquatic species. This is clear empirical evidence of the failure to learn from what has been done.

For these combined reasons, there is a high degree of certainty that adaptive management cannot be effectively used to prevent or avoid the impacts of MFT on watershed and aquatic resources. There is also a high degree of certainty that it is not an effective substitute for avoiding impacts known to cause enduring and significant damage to watersheds and aquatic resources (Espinosa et al., 1997).

III. RECOMMENDATIONS TO LIMIT OR REDUCE THE NEGATIVE IMPACTS OF MECHANIZED FUEL TREATMENTS ON PUBLIC LANDS

There are several measures that should be taken to reduce some of the negative impacts to aquatic systems from MFT on public lands. Some of these are likely to improve the effectiveness of MFT, some are likely to have neutral effect, and some may not complement efforts to reduce fire severity via MFT.

Restrict mechanized fuel treatments to areas of forests where they are most likely to encounter uncharacteristically severe fire and reduce its severity.

Although this is unlikely to reduce the ecological costs of MFT on aquatic systems, it should aid in limiting the *net* costs, by increasing the probability of some benefits accruing from MFT. However, it must still be acknowledged that implementing MFT in such areas will still most often have net negative impacts on aquatic systems without compensatory positive effects from reduced fire severity (See: Tables 4 and 5). This makes it more imperative to reduce MFT impacts.

Focusing on areas where the potential for success is greatest is likely to complement efforts to restore fire regimes. Treatments should only be considered in forests where site-specific evidence convergently indicates the forests have a natural fire regime of high-frequency/low-severity fire that has been altered. Notably, this requires site-specific examination of multiple lines of evidence (Veblen, 2003; Baker et al., 2006). As Baker et al. (2006) note, “It is impossible to determine the correct restoration model for a particular place without some collection of information on the site to be restored…”

The site-specific probability of higher-severity fire based on adequate site-specific data can be used in conjunction with Equation 1 to identify areas in forests with altered fire regimes where the probability of MFT affecting such fires is greatest.

Conversely, fuel treatments should not be implemented in systems with a natural fire regime of low-frequency, high-severity fire, such as subalpine and coastal forests. Where natural fire regimes and departures are unknown, fuel treatments should not be considered until there is ample convergent evidence indicating that fire regimes have been altered. Based on available data (Table 4), this does not incur a high degree of risk with respect to uncharacteristic fire.

Most forests with natural fire regimes of mixed severity are not in need of MFT to restore their natural fire regimes; this appears to be the prevalent case in undisturbed, mature forests with this fire regime (Baker et al., 2006). In these forests, a proactive approach of allowing wildland fire to restore fire regimes, together with efforts to curb fire regime alteration by grazing and logging, is likely to be effective in restoring fire regimes (Baker et al., 2006).

Riparian areas likely burn less frequently and at lower severity than uplands due to the effects of topography, microclimate, fuel moisture, and, in some cases, forest type. Hence, treatments in riparian areas tend to be less likely to encounter higher-severity fire. Prohibiting MFT in riparian areas also prevents damage from tree removal and associated activities to a host of riparian functions and aquatic resources.

Limit the scale of mechanized fuel treatments.

There are many key uncertainties associated with efforts to reduce fire severity. These include those related to the level of fire regime alteration in many forests and the potential effectiveness of MFT. Benefits from MFT remain uncertain, while there is a high degree of certainty regarding their costs. Given the uncertainty, MFT should not be aggressively pursued unless and until more robust information is available regarding their effectiveness, their overall costs and benefits, and the overall costs and benefits of fire. Given current knowledge, MFT must be considered experimental. MFT projects should be of limited scope and treated as ecological experiments, and include credible monitoring of their effectiveness and impacts.

Although it has been repeatedly suggested that the threat of uncharacteristically high-severity fire is high, analysis of available data on fire occurrence at several scales (USFS, 2004; Finney, 2005; Rhodes and Baker, *in review*) indicate that it is not. Recent large fires have not burned at uniformly high severity; they may well have burned at severities that are within natural ranges. Therefore, limiting the application of MFT to a conservative scale does not appear to confer a risk of ignoring a pressing and extensive threat to aquatic or terrestrial systems.

Retain large trees.

There is general agreement that the removal of large trees does not help reduce fire severity and is not consistent with the restoration of natural fire regimes. Retention of larger trees is vital to restoring forest structure and function (CWWR, 1996; Baker et al., 2006).

Larger trees provide numerous critically important ecological and watershed functions, whether live, dead, or downed (CWWR, 1996; Brown et al., 2003; Karr et al., 2004). Diameter limits on trees removed by MFT should also be adopted in order to ensure that larger trees are retained.

Although it has been argued that placing limits on the diameter of trees that can be removed might stymie efforts to reduce fire severity (Franklin and Agee, 2003; Noss et al., 2006a), there are two countervailing considerations that indicate that diameter limits are critical in order to protect watersheds and aquatic systems and restore forests. The first is that most efforts to reduce fire severity will not be effective regardless of implementation (Table 4). The removal of larger trees has adverse impacts on watershed functions. Second, the failure to adopt strict standards leads to the irretrievable loss of large trees as recent MFT proposals amply indicate. These large, ecologically important trees are now relatively rare in many forests (Henjum et al., 1994; Baker et al., 2006).

About 90% of trees in Western forests are less than one foot in diameter, based on data from the 2002 USFS Resource Planning Act Assessment Report (Center for Biological Diversity, undated). Smaller trees represent the greatest fuel hazard. Since the logistics of MFT are such that the removal of all of these smaller trees is not possible, nor desirable in many areas, adopting a diameter limit of one foot is highly unlikely to seriously impede MFT in areas where they might aid in fire regime restoration. Therefore, this likely represents a reasonable starting point for a diameter cap for MFT.

Restrict or eliminate grazing.

In some forests with altered forest regimes, grazing has contributed to the situation (Belsky and Blumenthal, 1997), thereby contributing to potential increases in fire severity. Restricting or eliminating grazing is likely to aid in restoring natural fire regimes in some forests (Baker et al., 2006). Conversely, continued grazing in such forests is likely to hobble efforts to restore natural fire regimes (Belsky and Blumenthal, 1997) and reduce fire severity.

Grazing clearly has numerous negative impacts on watersheds, soils, riparian areas, water quality, and aquatic systems (Platts, 1991; Fleischner 1994; Belsky et al. 1999; Beschta et al., 2004). Grazing elimination, especially in riparian areas, clearly has numerous positive benefits for aquatic systems (Platts, 1991; Fleischner 1994; Rhodes et al., 1994; Belsky et al. 1999). Although the curtailment of grazing does not reduce the impacts of MFT, it does reduce the cumulative impacts on aquatic systems that may be affected by fire and/or MFT.

Forego mechanized fuel treatments when proactive re-establishment of forest processes can restore altered fire regimes, and implement mechanized fuel treatments only as part of wider efforts to restore fire regimes, including the use of prescribed and wildland fire.

No matter how well they are implemented, MFT alone will not restore fire regimes in a self-sustaining manner. The latter requires wider efforts. Taking an integrated approach that addresses the sources of fire regime alteration and restores natural processes will likely increase the effectiveness of efforts to restore fire regimes.

One of the most critical steps in undertaking effective ecological restoration is to forgo those activities and land uses that either cause additional damage or prevent the recovery of degraded systems (Kauffman et al. 1997). This is because the avoidance of degradation is far more effective and tractable than trying to rehabilitate degraded conditions (Beschta et al., 2004). Additionally, restoration approaches that do not address the root sources of degradation are unlikely to effectively restore systems.

Effectively addressing the sources of fire regime alteration can also aid in establishing forests that do not need to be repeatedly treated to reduce fuels in order to restore fire regimes. As Noss et al. (2006b) note, "Although many forests will require continued management, a common sense conservation goal is to achieve forests that are low maintenance and require minimal repeated treatment." The latter can aid in limiting the aquatic costs of MFT by reducing the need to repeatedly implement MFT, while still restoring natural fire regimes (See Figure 3 next page), because repeated treatments contribute to adverse cumulative effects on watershed and aquatic resources. Sole reliance on MFT, alone, requires repeated cycles of treatments (Baker et al., 2006; Noss et al. 2006b; Figure 3).

In many prevalent forest types, such as those with a fire regime of mixed severity, proactive approaches that curtail the causes of altered forest conditions and fire behavior are likely to be all that is needed to restore natural fire regimes over time (Baker et al., 2006).

Logging, grazing, post-disturbance planting, and/or fire suppression have likely contributed to the alteration of natural fire regimes, fuel characteristics, and fire severity in some forests. Assessments of the potential alteration of fire regimes in forests should specifically examine the causes of fire regime alteration (Veblen et al., 2003). Where ongoing activities have contributed or are likely to contribute to fire regime alteration, measures should be taken to restrict or eliminate these sources of alteration in fire behavior. Such measures likely increase the effectiveness of other efforts to restore fire regimes in a self-sustaining manner that does not require continual repetition of treatments with ecological costs.

The ultimate and primary goal of fire regime restoration should be re-establishment of wildfire frequency and severity that is characteristic of the natural fire regime (Kauffman, 2004). Nature can be an inexpensive and effective agent of restoration. Allowing more wildland fire to burn without suppression is likely to help prevent continued

Figure 3. A conceptual illustration of how the level of need for periodic retreatment varies among fire regime restoration approaches (After Noss et al., 2006b). Addressing the root causes of altered fire regimes by restoring processes may be self-sustaining. In contrast, mechanical fuel treatments, which at best, can only restore forest structure, require relatively frequent repeated treatments.

alteration of fire regimes. It is also likely to aid considerably in restoring forests with altered fire regimes (Kauffman, 2004; Baker et al., 2006; Odion and Hanson, 2006). As Odion and Hanson (2006) note regarding wildland fire:

> There may be no other effective strategy for restoring and maintaining ecological integrity and for fostering the natural diversity of species dependent on effects specific to fire. The structural modifications of forests cannot mimic the heterogeneous effects of fire. Instituting a policy that allows more fire to burn would require considerable planning and additional efforts to improve human safety, but such efforts are needed under any management scenario.

Both wildland and prescribed fires can have watershed and aquatic impacts that are sometimes significant, but they are typically less persistent than those from MFT, and, therefore, should be favored over MFT as a means to reduce fuels and restore forest structure and natural fire regimes.

The use of prescribed and wildland fire should also complement efforts to restore altered fire regimes by increasing fiscal efficiency. MFT are costly. While prescribed fire has fiscal costs, on a per unit area basis, it is typically far less costly than MFT (Lynch and Mackes, 2003; Rummer et al., 2003). Fire suppression is also costly. Therefore, the use of wildland fire can also help reduce fiscal costs, if zealous and costly fire suppression is reduced in places where it can be foregone at reasonable risk to ecological systems and human infrastructure.

Reductions in fire suppression may also produce the added benefit of reducing damage to watershed and aquatic systems. This is because some fire suppression methods can have significant impacts that persistently contribute to watershed and aquatic degradation (Beschta et al., 2004).

Avoid mechanized fuel treatments in areas and watersheds where adverse impacts are likely to be significant and enduring.

Due to limited potential for fuel treatments to benefit aquatic systems, high hazard areas should be avoided consistently. Such areas include those with topographic or soil hazards, roadless and riparian areas and watersheds with pronounced cumulative effects, high potential for restoration, high biodiversity, or imperiled aquatic populations.

Although higher-severity fire may pose a threat to some isolated populations of fish in some degraded systems, MFT is unlikely to significantly reduce this threat in most cases, while incurring aquatic costs and causing additional degradation. A more effective approach to protecting such fish populations is to focus efforts on effective measures to reduce the causes of fragmentation and habitat degradation, such as water withdrawals, road networks, impassable barriers, and grazing.

MFT should also be avoided in watersheds where sedimentation and/or other MFT impacts are already a concern for fish, amphibians, and other beneficial uses. Due to its extent and associated road impacts, MFT is likely to increase sedimentation, especially in already damaged systems. Although MFT has the potential to reduce sedimentation from higher-severity fire, sometimes quite considerably, this is likely to be a relatively rare occurrence, even in forest types where it is most likely to be effective. A more reliable approach to reducing sediment delivery is to address current management sources through restoration while foregoing the implementation or continuation of activities that increase sediment delivery.

There is no compelling evidence that riparian areas are a priority for MFT. Available information on fire behavior amply indicates that riparian areas are less prone to higher-severity fire than uplands, making them a very low priority for efforts to reduce fire severity. Treatments in these areas are likely to cause manifold aquatic impacts that are highly unlikely to be outweighed by benefits from MFT, even in the unlikely case that they are effective.

Due to their sensitivity, land disturbance in remaining roadless areas is likely to cause significant degradation (USFS et al., 1993; Rhodes et al., 1994). Roadless areas are limited in extent and critically important for the protection and recovery of aquatic resources (Henjum et al., 1994; Kessler et al., 2001). Additionally, roadless areas typically have the least altered forest structure, fuels, and fire regimes, reducing the need for interventionist approaches, such as MFT (Franklin et al., 2000; DellaSala and Frost, 2001; Baker et al., 2006). For these reasons, the prohibition of MFT in roadless areas will not only help protect these areas, but also ensure that purposeless fuel treatments are not introduced.

Constrain or prohibit the most damaging activities.

Avoid practices that consistently cause severe and persistent watershed damage, including machine piling and burning and the construction of roads and landings, including “temporary” ones. The numerous negative effects of roads are one of the primary sources of aquatic and watershed damage on a continental scale. Additional road construction is inimical to reducing road effects. It also inexorably adds to the currently insurmountable backlog in needed, but deferred, road maintenance on existing roads (USFS et al., 1993; USFS, 2000b; Beschta et al., 2004). Even "temporary" roads and landings that are subsequently obliterated have impacts on forests and soils that last for decades. For these reasons, it is essential to ensure that MFT do not involve road or landing construction.

Undertake effective watershed restoration.

Effective watershed restoration can help make aquatic systems, aquatic populations, and watersheds more resilient to fire impacts (Beschta et al., 2004). Prime examples include road obliteration or decommissioning, attempting to hydrologically decouple roads from stream networks, removal of impassable barriers in streams, reduction of water withdrawals, and curtailing or eliminating livestock grazing.

In areas where MFT are pursued, they should be always accompanied by effective watershed restoration. Although this will not reduce the aquatic costs or increase the effectiveness of MFT, it should help ameliorate the cumulative negative impacts on aquatic systems at the watershed scale.

Credibly analyze and disclose likely cumulative effects of treatment versus non-treatment.

Treatments should be carefully analyzed for their cumulative effects, including all related disturbances and impacts. These should be evaluated on the basis of the likely outcomes of treatment and non-treatment on fire, taking into account fire probability and the transience and limited effectiveness of treatments. To be credible, cumulative effects analysis related to MFT must include the following in all assessments:

- Critical review of the evidence regarding the alteration of fire regimes. Data gaps and uncertainties and potential biases in methods must be scrutinized, disclosed, and accounted for in analyses, including those inherent in ascertaining the natural fire regime and potential departures from it. Some current methods of estimating the departure from natural fire regimes are likely misleading. For instance, Fire Regime Condition Class approach (FRCC), is likely misleading because: a) it overestimates the occurrence of high-severity fire when fire occurs (Odion and Hanson, 2006); b) it is not based on site-specific data on fire occurrence; and c) it is based on guesswork regarding the number of mean fire intervals that have been skipped without consideration of longest fire-free intervals in the historic record. Mean fire return intervals have questionable meaning. Unless forests have been free of fire for longer than the longest fire-free interval in the historic record, they may not be outside of the historical range of fire frequency. If the historic record is unknown, there is no sound basis for assuming that the natural fire regime has been altered and is in need of restoration.
- The likelihood of higher-severity fire affecting untreated areas, which can be estimated via data on the annual probability of such fires in conjunction with Equation 1. This can provide a quantitative estimate of the amount of higher-severity fire that may occur in the absence of treatment. Most fire does not burn at high severity. Typically, even in large fires burning during extreme fire weather in forests with high fuel loads, only about a third or less of the area burns at high severity (Table 7).

™ The likelihood of higher-severity fire affecting treated areas during the period that fuels have been transiently reduced. This can be estimated from data on the annual probability of such fires in conjunction with Equation 1. This can provide a quantitative estimate of the fraction of treatments that might reduce fire severity.

™ Congruent analytical timeframes. Both probabilities above should be estimated for congruent timeframes. For instance, if a single treatment cycle is anticipated and the duration of reduced fuels is expected to be 12 years, then the probability of higher-severity fire encountering treated or untreated areas should both be estimated for a 12-year timeframe.

™ Incremental effects. It is not tenable to assume that all untreated areas burn at high severity in the absence of treatment during a given timeframe or that fuel treatments eliminate higher-severity fire at the scale of an analysis area, even if several cycles of treatment are anticipated. Available evidence does not indicate that treatments that encounter higher-severity fire eliminate it. Instead, when treatments are effective, they incrementally reduce high-severity fire (e.g., Schoennagel et al., 2004a). Some amount of high-severity fire still occurs at the analysis scale. Therefore, it is not valid to compare the effects of higher-severity fire at the scale of the analysis area to those solely from treatment at the scale of the analysis area. Analyses of the impacts of MFT need to consider the likely effects of treatments *combined* with those of fire.

™ Complete cumulative effects. The analysis should include all impacts from all anticipated treatments over the entire period that they are likely to be applied. For instance, if fuels generated by MFT are expected to be machine piled and burned, with treated stands subsequently reburned every 12 years for 84 years, cumulative effects analysis must consider the impacts of these activities on watersheds and aquatic systems over the entire 84-year period. Similarly, if piecemeal MFT of different areas of the same watersheds are planned over a 50 year period, cumulative effects analysis must consider these impacts.

™ Duration of effectiveness. Treatment effects on fuels decline with time. Hence, it is likely that the potential effectiveness of treatments in reducing fire severity also declines with time. Absent better information, it is probably reasonable to assume that mean effectiveness of treatments over the time period that fuels are reduced is roughly half the initial effectiveness.

™ Limits of effectiveness. There are limits to the effectiveness of MFT when they do encounter fire. Some practices have limited effects on fire severity. In some forest types, weather limits the effectiveness of MFT in reducing fire severity.

™ Associated road impacts. MFT often involve elevated use, reconstruction, and/or construction of roads and landings. Analyses must include the impacts of all such activities on watershed and aquatic systems.

™ Persistence of impacts. MFT and associated activities have effects on soils, water quality, and watersheds that may be more persistent than those from fire.

™ The status and connectivity of affected aquatic populations. Fire likely has less enduring impacts on fish populations with a high level of habitat and population connectivity.

- Not all higher-severity fire causes hydrophobic soils, nor does it always trigger extreme postfire erosion and runoff events.
- Fire provides important benefits to aquatic systems (e.g., LWD recruitment). Some impacts of MFT, such as soil compaction and elevated surface erosion on roads, provide no benefits to watersheds and aquatic systems.
- Assessments should identify how MFT might affect current impacts that constrain or prevent the restoration of aquatic systems.

The foregoing is not an exhaustive list of issues, because some are likely specific to analysis areas. But it does provide a reasonable framework for assessing likely outcomes of treatment versus non-treatment on aquatic systems.

Assessment alone cannot improve decisions about how to deal with the risks involved in MFT versus non-treatment of fuels. But if done correctly, it can help bring the issues and potential outcomes of treatment versus non-treatment into focus.

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