

**The High Costs and Low Benefits
of Attempting to Increase Water Yield by
Forest Removal in the Sierra Nevada**



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Executive Summary

Although there has been renewed interest in attempting to boost runoff from Sierra Nevada watersheds by removing copious amounts of forest cover, recent assessments promoting the approach have not given ample attention to well-known factors that sharply limit its utility for augmenting water supplies. These assessments have also largely ignored the considerable and enduring environmental costs of pursuing such an approach.

This report provides a more thorough assessment of the environmental costs and limited utility for water supply from attempts to increase water yield via forest removal in the Sierra Nevada. Although data are limited from the Sierra Nevada, there is considerable body of information from applicable studies throughout the western U.S. that provides a context for assessing the limited benefits and significant costs of pursuing a forest removal or thinning management approach.

This information indicates that the following limits the utility of any potential increase in water yield from forest removal:

- Water yield increases are highly variable and not amenable to accurate prediction solely as a function of the amount of forest removed. However, aggregate data indicate that, on average, only very modest increases in water yield can be expected.
- At the scale of major watersheds which supply water, any actual water yield increase from forest removal is likely to be too small to verify via field flow measurement.
- Increases are very strongly affected by seasonal precipitation. Flow increases are most unlikely and smallest during dry years and during dry seasons. Thus, the approach has very nominal potential to improve water yield during droughts. For the same reasons, the approach is unlikely to provide additional water during dry seasons when demand is high relative to supply.
- Increases are typically greatest during the period of highest runoff and during the wettest years. Due to this timing, any realized increases may have negligible benefits for water supply, while contributing to increased flooding.
- Any increases in water yield from forest removal are diminished by transmission losses and storage losses, reducing any increase in downstream water supply.
- Increased water yield in response to forest removal is transient. Any increases are erased by vegetative regrowth within several years after forest removal. In effect, forest removal promotes regrowth that exacerbates water demand by second-growth vegetation.
- In the absence of continued removal, forest removal contributes to net reductions in low flows in subsequent decades, exacerbating water supply problems when demand is typically highest.
- The maintenance of potential increases in water yield would require clearing of large percentage of forests at high frequency, on the order of 25% of watershed area every 10 years. This frequency and magnitude of forest removal would incur significant fiscal, logistical, and environmental costs.

Due to these well-established limitations, previous assessments of this forest management approach, including those of the US Forest Service and National Academy of Sciences, have consistently noted that it is not likely to be practical due to the innate limitations identified above. The National Academy of Sciences consensus panel report on forest hydrology (2008) concluded:

“...water yield increases from vegetation removal are often small and unsustainable, and timber harvest of areas sufficiently large to augment water yield can reduce water quality...There is little evidence that timber harvest can produce sustained increases in water yield over large areas...the potential for augmenting water yield on a sustainable basis in western forests and rangelands is very low.”

Forest removal associated with attempts to increase water yield is unlikely to significantly alter fire behavior. There is a low probability that wildfire would affect treated areas, during the time when fuel levels are reduced, even with extensive forest removal. Weather, rather than the fuel conditions altered by fuels treatments, often exerts the dominant control on fire behavior, especially during large wildfires, further limiting the effectiveness of fuel treatments. Moreover, it is not ecologically desirable to reduce the extent and severity of wildfire in most Sierra Nevada forest, because there is currently a deficit of wildfire relative to historical levels. Wildfire is a natural keystone forest process which provides many critical medium- and long-term ecological functions and benefits for aquatic and terrestrial ecosystems.

Intensive forest management aimed at elevating water yield would incur major and enduring environmental costs, due to the frequency and magnitude of forest removal that would be needed to maintain increases in water yield. Together with associated forest removal activities, including roads, landings, and skid trails, frequent and extensive forest removal would permanently degrade soils, riparian areas, aquatic systems, and water quality. The latter would incur significant water supply costs, including increased costs of treatment for elevated sediment and nutrient levels, as well as the likelihood of increased flood damage. Thus, the at-best modest benefits for water yield would come at the expense of high environmental and economic costs.

Alternative forest and rangeland management measures can benefit water supplies by improving low flow and water quality conditions at a relatively low fiscal cost, while conveying a host of additional ecological benefits. These measures include sharply curtailing livestock grazing, reducing the extent and impacts of road networks, and re-establishment of beaver. These measures also likely increase the resiliency of watersheds in the face of drought, floods, and climate change.

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FOREWORD by Douglas Bevington, Forest Program Director, Environment Now

There is a long history of claims being made by logging proponents that cutting more trees would lead to more water for downstream users, even though forest preservation is widely understood to be an effective way to protect water supplies. Time and again, these logging-for-water claims have turned out to be ill-founded. For example, the National Research Council concluded that “There is little evidence that timber harvest can produce sustained increases in water yield over large areas” (quoted on p. 1 of this report).

Despite these findings, the logging-for-water claim has a way of popping up again, especially during times of drought. It is not surprising then that this claim has gained renewed attention in California during the recent intense drought. It is easy to understand the temptation of this claim. In desperate times, one may be more likely to believe in promises of easy water, rather than looking closely at the shortcomings and drawbacks. During such times, diligent fact-checking is especially important, and that is why Environment Now commissioned this new report.

Environment Now is a family foundation that supports water and forest protection in California. At this critical time, we wanted to understand the basis for the latest claims about using logging to increase water supply, particularly in light of the recent support by the Nature Conservancy. We were interested in seeing if something new had been discovered, or if this is simply old wine in a new bottle. Prior to the recent drought, previous attempts to apply this claim to the forests in California’s Sierra Nevada mountains were examined in a report titled “Thinning for Increased Water Yield in the Sierra Nevada: Free Lunch or Pie in the Sky?” We commissioned the co-author of that report, hydrologist Jonathan Rhodes, to reexamine his research in light of the recent versions of the logging-for-water claims. This new study is co-authored by fisheries scientist Chris Frissell, who was the co-author of “Conservation of Freshwater Ecosystems on Sierra Nevada National Forests.” Together they have now produced a remarkably thorough investigation of this topic—“The High Costs and Low Benefits of Attempting to Increase Water Yield by Forest Removal in the Sierra Nevada”—which builds on the findings of more than 230 studies and reports.

The findings in the Rhodes and Frissell report should be of keen interest to anyone who cares about water and forests in California. Overall, they show that the effects of logging on water flows are often negligible, nonexistent, or negative, and even in the more optimistic scenarios, the potential effects are small, transient, and ill-timed. In contrast to the false hopes for significant water flow benefits, these logging schemes are far more likely to produce many harmful side effects for aquatic ecosystems and downstream water users. In short, using logging to increase water flows is still a bad idea whose time has not come.

This report is particularly helpful for addressing four ways that the current version of the logging-for-water claim has been presented as somehow new or different from earlier versions. One way that proponents have repackaged this claim is by presenting water increases as being a secondary effect from extensive logging (“thinning”) proposed ostensibly to reduce fire in Sierra Nevada forests. However, fire-related logging justifications are contradicted by a growing body of scientific research showing that big, intense wildfires are a

natural and beneficial component of California's forest ecosystems, and there is currently less fire (including less high-severity fire) in our forests than there naturally should be. (A full review of this fire science literature is beyond the scope of this study, which focuses on water-related aspects, but readers wanting to learn more about the latest research can find it compiled in *The Ecological Importance of Mixed-Severity Fires: Nature's Phoenix*, edited by Dominick DellaSala and Chad Hanson.) A second, related feature of current logging-for-water claims is a focus on snow accumulation related to logging, as seen in the work of Roger Bales, a professor of engineering at UC Merced. Yet, Rhodes and Frissell show that these two new points are contradictory because "forest removal aimed at modifying fire behavior is more likely to accelerate than delay snowmelt." (p. 21).

A third new feature of the current logging-for-water claim is the involvement of The Nature Conservancy in promoting this idea, including a report by Podolak et al. titled "Estimating the Water Supply Benefits from Forest Restoration in the Northern Sierra Nevada." Yet Rhodes and Frissell find that "projected increases in water yield in response to proposed levels of forest removal in Podolak et al. (2015) are likely considerably overestimated." (p. 10) Indeed, findings from other studies indicate that for the majority of the watersheds in which Podolak et al. claimed there could be water increases, the proposed logging would actually produce no increase in water yield (see Table 1 and Figure 2, p. 11).

A fourth new feature is the idea that downstream water users should be involved in paying for the cost of logging projects in the Sierra Nevada, based on the notion that the logging will result in more water. However, downstream water agencies and their customers should be very cautious of these claims. As Rhodes and Frissell note, "Assessments of attempts to increase water yield on public lands have consistently noted that it is very unlikely that any potential changes in water yield would be measurable at the scale of larger watersheds... Absent verifiability, there is no way to reliably determine if investments in such an approach might yield any returns." (p. 25).

Beyond these four facets, here are some other key points that stood out for me after reading the report. In the first section of their report ("Forest Removal Effects on Water Yield, Peakflows, and Low Flows"), Rhodes and Frissell analyze eleven ways that claims of increased water flows from logging are problematic or overstated. One example is the often overlooked issue of water flow timing. The intrinsic appeal of logging-for-water schemes lies in the notion that they would provide additional water during dry times when water is scarce. However, Rhodes and Frissell found that "even high levels of forest removal are unlikely to provide any additional water during dry seasons, especially during the driest years, when it would be most beneficial for downstream uses." (p. 18).

This timing aspect illustrates that logging-for-water schemes are particularly ill-suited for California's climate, which is characterized by periods of drought and periods of heavy precipitation and the effects of La Niña and El Niño. Small, transient upticks in water flows after logging would be least likely to manifest in the summer and times of drought, and instead would occur during already wet times when additional water is not needed. Moreover, increased flows during wet times can actually be a problem. As Rhodes and Frissell explain, "Due to the magnitude and timing of these effects on seasonally high flows, forest removal is

likely to contribute to increases in downstream flooding and associated flood damage when these elevated high flows coincide with downstream flooding.” (p. 28)

In other words, in addition to having questionable benefits, logging-for-water schemes can also cause very real harms, though these downsides are often overlooked or downplayed by logging proponents. In the second section of their report (“The Enduring Environmental Costs of Forest Removal”), Rhodes and Frissell examine nine types of ecological damage resulting from logging-for-water schemes: increased peakflows, flooding, and flood damage; increased erosion and sedimentation (i.e. dirty water); effects from intensified prescribed burning associated with logging; channel erosion; soil compaction and soil degradation; interactions of forest removal with wildfire; harms to fish habitats and populations; harms to downstream water supplies; and increased invasive vegetation and noxious weeds.

In addition to harming to aquatic ecosystems, the effects of logging-for-water can also be costly to downstream water users. As Rhodes and Frissell note, “Forest removal would have several impacts that would incur significant costs for downstream water supplies and associated infrastructure and activities. These costs would be pervasive and enduring.” (p. 57)

With so many problems associated with logging-focused approaches, it was good to see that, in the final section of their report (“Land Management Approaches that Benefit Water Supplies and Watersheds without Incurring Significant Environmental Costs”), Rhodes and Frissell identify alternate forms of land management that can increase water flows without the downsides of logging. Three alternatives include: the reduction or cessation of livestock grazing near streams and meadows in the headwaters of the Sierra Nevada; reductions in the extensive and expensive network of logging roads in Sierra Nevada national forests; and restoration of beaver populations in the Sierra Nevada. The advantages of these alternate approaches are numerous. As Rhodes and Frissell explain, these steps can reliably contribute to improved flows during drier times when additional water is most beneficial (in contrast to logging approaches that have uncertain flow effects which are least likely to occur during drier times); they are self-sustaining (in contrast with logging-based approaches that must be done over and over again, with the resulting increases in environmental and economic costs); they do not incur high or enduring environmental costs; they provide an array of ecosystem benefits; they provide benefits for downstream water use via improved water quality; they address pressing forest restoration needs; and they contribute to watershed resiliency in the face of climate change.

Especially in light of the uncertainty, expense, and harms from logging-based approaches, readers of this report will likely be left wondering why those who claim to be interested in improving water flows have focused so much on logging and so little on the three compelling alternative approaches to Sierra Nevada land management presented by Rhodes and Frissell. If one genuinely seeks to improve water flows in the Sierra Nevada, rather than simply trying to find a new justification for logging, these alternatives offer a better way for us to direct our resources.

1. Introduction

There has been renewed interest in attempts to increase annual water yield¹ from watersheds in the Sierra Nevada by removing forest cover on public lands. This approach is based on the simplified conceptualization of complex hydrologic processes: tree removal temporarily decreases evapotranspiration and increases snow accumulation, which can result in increased runoff, under some conditions.

Attempts to increase water supply via forest removal are not new. Although there are few data from the Sierra Nevada, the effects of forest removal on water yield have been studied in many other regions over many decades. The relevant information indicates that there are many known factors that limit the tractability and ability of the approach to reliably and significantly increase downstream water supply, particularly during periods when additional water would be most beneficial. Available information also indicates that such approaches would incur significant environmental, fiscal, and societal costs. Past assessments have repeatedly concluded that such approaches are inherently impractical and unpromising due to their innate limitations and associated costs. These assessments include those of USFS research (Ziemer, 1986; Sedell et al., 2000) and the National Research Council (NRC) consensus panel report on forest hydrology (NRC, 2008) which concluded:

“...water yield increases from vegetation removal are often small and unsustainable, and timber harvest of areas sufficiently large to augment water yield can reduce water quality... There is little evidence that timber harvest can produce sustained increases in water yield over large areas... the potential for augmenting water yield on a sustainable basis in western forests and rangelands is very low.”

Despite this information, recent assessments of the potential to increase water yield via forest removal in the Sierra Nevada (Bales et al., 2011; Podolak et al., 2015) have given sparse attention to practical aspects of the approach that significantly limit potential water supply benefits and have also generally ignored associated environmental, fiscal, and social costs. This report attempts to provide a more thorough and unbiased assessment of available scientific information on critical factors that influence the practicality and utility of attempts to increase water yield by forest removal and the associated costs of such a forest management approach.

Recent assessments of the potential to increase water yield via forest removal in the Sierra Nevada (Bales et al., 2011; Podolak et al., 2015) have primarily focused on areas with coniferous forest cover where snow is the dominant form of precipitation. Therefore, this report's discussion of the effects of forest removal on water yield, peakflows, and low flows, primarily centers on the effects of the removal of conifer forests where snowmelt supplies the bulk of annual runoff.

¹ Annual water yield is the total surface runoff in streamflow from a watershed over a water year (October 1 through September 30) or calendar year.



Photo 1. This type of naturally regenerated, mature overstory, multi-tiered, mixed-species stand on the west slope of the Sierra Nevada is often targeted for thinning and other forest removal treatments. Plumas National Forest. Photo by C. Frissell.

There is a paucity of studies on water yield in response to forest removal in the Sierra Nevada (Kattelman, 1987; Stednick, 1996; Bales et al., 2011), with only two in snow-dominated areas (Kattelman, 1987). Therefore, this report uses the results from studies in relatively small, paired watersheds in other areas, particularly those from the western US, to elucidate forest removal effects on runoff.

This report examines several factors that influence the tractability and utility of attempts to increase water yield via forest removal. First, increases in water yields from forest removal have often been modest, even in some cases where small watersheds have been extensively deforested.

Second, it is not possible to accurately predict the magnitude of changes in water yield in response to forest removal. Available data indicate that the relationship between forest removal and increased annual water yield is extremely variable.

Third, increases in water yield from forest removal typically are lowest and least likely in the driest years. In years with adequate precipitation, almost all increased runoff from forest removal occurs during the period of highest runoff, during annual peakflows, rather than late in the low flow period. Thus, the approach has little promise for supplying water during years

or seasons with the greatest downstream demand relative to supply. Similarly, water yield increases tend to be greatest in wet years when the additional runoff has relatively little value and may exacerbate flooding.

Fourth, a large fraction of forest must be removed to increase water yield transiently at the scale of smaller watersheds.

Fifth, forest removal provides only transient increases in water yield, which begins to decrease as vegetation re-grows after treatment. Due to the temporary nature of water yield effects in response to forest removal, vegetation must be continually removed over extensive watershed areas at relatively short intervals in order to attempt to maintain increases in water yield.

Sixth, due to the significant distances between headwater forest watersheds and downstream water uses, any increases in annual water from forested watersheds are likely to be significantly diminished by transmission losses, and when captured in reservoirs, storage losses. These unavoidable losses will reduce any potential increase in downstream water yield at points of use.



Photo 2. Intense soil damage on a log landing for a forest thinning project in Giant Sequoia National Monument in the Sierra Nevada, California. The severe damage from mechanized skidding and other operations at landings is akin to that from roads in intensity and persistence. Photo by A. Maradosian.

The ecological, societal, and fiscal costs of forest removal aimed at increasing water yield are likely to be numerous, enduring, and considerable. The magnitude and frequency of forest removal sufficient to potentially increase water yield, together with associated activities and extensive necessary infrastructure, such as roads, would cause widespread, lasting damage to soils, watershed processes, water quality, stream structure, fish populations and habitats, reservoir capacity, and other aquatic resources. Forest removal would elevate soil erosion, and consequent sediment delivery to streams and downstream reservoirs. The road network required to implement extensive and relatively frequent vegetation removal increases the flashiness of runoff and harmful sediment and nutrient loads.

This report also evaluates the potential for forest removal projects to modify fire behavior and potentially reduce associated watershed and aquatic impacts. Regardless of the potential effectiveness of forest removal treatments in affecting fire behavior, if fire does not affect treated areas while fuels are transiently reduced, such treatments cannot affect fire behavior. Studies have repeatedly shown that this probability is relatively low. Therefore, it is likely that most areas treated to reduce fuels will not affect fire behavior, and instead only convey watershed costs from treatment impacts.

Further, the greatest potential for augmenting water yield via forest removal is in areas with higher annual precipitation where fire is relatively infrequent. In such areas, weather exerts a significant control on fire behavior, which further reduces the potential for fuel treatments to significantly modify fire behavior even if fire affects treated areas. Thus, forest removal in areas with some potential for increasing water yield has relatively low potential to affect fire behavior. Conversely, while forest removal in drier forests has a greater, although still limited, likelihood to affect fire behavior, such treatments are unlikely to augment water yield substantially.

This paper also examines some low-risk alternatives to forest removal that can help increase low flows and decrease peakflow, while providing important additional environmental benefits without incurring significant environmental costs. These actions would have the added benefits of helping to restore water quality, a variety of watershed conditions and processes, and aquatic habitats and populations. These alternative actions include greatly reducing or suspending livestock grazing, road network reduction, and re-establishment of beaver populations.

2. Forest Removal Effects on Water Yield, Peakflow, and Low Flows

2.1 Forest removal effects on annual water yield are highly variable

Data from studies in small, paired watersheds have shown that the response of annual water yield to various levels of forest removal² is extremely variable, as shown in Figure 1, which

² Many studies of relationship between water yield and forest removal have involved complete deforestation of portions of watersheds (Marvin, 1996) and only reported the percent of watershed area deforested. Many assessments of the relationship between water yield and forest removal have assumed that the effect of the magnitude of basal area removed in a watershed is equivalent to the area of deforested, e.g., that the removal of 40% of basal area over an entire watershed is equivalent to the effect of deforestation of 40% of a watershed

includes data for 54 results from studies in relatively small, experimental watersheds in conifer forests. Importantly, more than 90% of the data shown in Figure 1 is for the maximum annual increase in the first five years after forest removal.³ This significantly overestimates the mean measured annual water yield response after deforestation, because increases are typically greatest the first few years after logging, other factors being equal, and also vary with precipitation (Bosch and Hewlett, 1982; Troendle and King, 1985; 1987; Gottfried, 1991; Stednick, 1996; Brown et al., 2005; NRC, 2008). Mean annual water yield increases after forest removal are considerably lower than the measured maximum annual increases. For instance, the removal of 30-40% of the forest in a Colorado watershed resulted in increases in water yield during three wet years, with no measurable increase in other years (Troendle, 1985). In Arizona, the removal of ca. 34% of conifer cover resulted in annual maximum of 70 mm increase in water yield (Stednick, 1996), while the mean annual change over seven years was considerably less, at 44 mm (Gottfried, 1991; Marvin, 1996).

Regression analysis of the data in Figure 1 demonstrates that the level of forest removal explains very little of the variation in maximum annual water yield, as indicated by the low R^2 of 0.25, a standard error (SE) of 129.3, and the wide scatter about the regression line. This high variability in the relationship between percent forest removal and water yield has been consistently noted in past assessments, whether the results were assessed by region (Marvin, 1996; Stednick, 1996; MacDonald and Stednick, 2003) or vegetation cover type (Bosch and Hewlett, 1982). Therefore, available data indicate that even for small, experimental watersheds it is not possible to accurately predict changes in water yield solely based on the level of forest removal.

While the aggregate data in Figure 1 indicate that increases in annual water yield tend to increase with increasing levels of forest removal in relatively small experimental watersheds, it does so in a manner that cannot be reliably or accurately predicted solely on the basis of the amount of forest removed. As shown in Figure 1, there are two instances where greater than 20% of small watersheds were deforested with no increase in annual water yield over five years. Notably, one of the two studies of forest removal on water yield in the Sierra Nevada snow zone found no measurable increase in water yield from the removal of 25% of conifer forest in the Kern Plateau, while sediment levels increased appreciably post-logging (Kattelman, 1987; Marvin, 1996). There are nine instances in Figure 1 where 50% or more of forest cover was removed, but the maximum annual increase over five years over the watershed area was 100 mm or less. Further, as previously discussed, most of the data in Figure 1 are for maximum annual water yield, which significantly overestimates multi-annual water yield response to forest removal.

(e.g., Stednick, 1996; Podolak et al., 2015). Although the effect of dispersed basal area removal (e.g., via thinning) on water yield may be considerably different than the level of basal area removal from deforestation (Marvin, 1996), this report generally uses the analytically expedient assumption that percent basal area removal is equivalent to the percent of area deforested in a watershed. However, this assumption may overestimate water yield increases from thinning. This because most studies have found that changes are greatest in watersheds with deforested openings (Kattelman, 1987).

³ Five of the 54 data points in Figure 1 are for mean increases in water yield in the first five years after forest removal, as reported by Sahin and Hall (1996) (Brown et al, 2005).

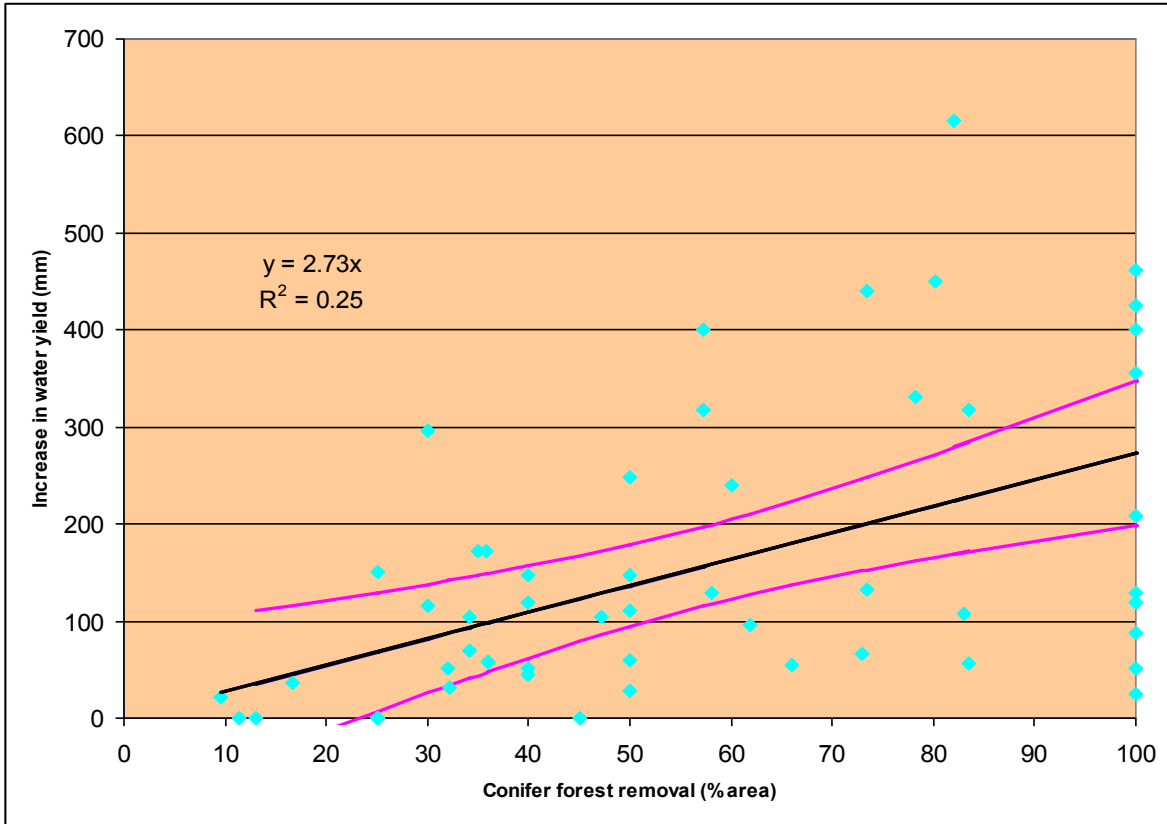


Figure 1. Maximum annual water yield response after conifer forest removal. All but 5 points are for the single year maximum water yield measured after forest removal. Data are from Stednick (1996), Marvin et al. (1996), and Brown et al. (2005). Regression intercept set at zero because zero forest removal has zero effect on water yield (Stednick, 1996). Black line is linear regression line. Purple lines demarcate the 95% confidence interval for the regression line.

Increased snowpack sublimation and abiotic evaporation caused by forest canopy loss may explain the sometimes limited response of streamflow to forest removal in snow-dominated areas. Biederman et al. (2014) found that high levels of tree mortality did not increase streamflow in small watersheds in Colorado, despite considerable reductions in transpiration and canopy interception of snow. Multiple lines of field evidence indicated that this lack of streamflow response was due to increases in both snowpack sublimation and abiotic evaporation caused by forest canopy loss (Biederman et al., 2014). These results underscore that the notion that reductions in evapotranspiration and canopy interception from forest removal consistently translate into increased runoff is an oversimplification of complex ecohydrologic interactions that is in considerable error in some watershed settings.

Past research has suggested that several factors contribute to the documented variability in the response of annual water yield to forest removal. These include many of the watershed attributes that influence streamflow, including evapotranspiration by remaining vegetation, soil conditions, the size and location of openings created by forest removal, and precipitation (Bosch and Hewlett, 1982; Kattelmann, 1987; Troendle and Olsen, 1994; Stednick, 1996; Marvin, 1996; Brown et al., 2005). However, as discussed in greater detail in a following

section, precipitation clearly has a major effect on the response of water yield to forest removal, but is outside of management control.

2.2 Forest removal effects on annual water yield are transient

Water yield increases from forest removal are transient. Other factors being equal, annual water yields decline over time after forest removal, as vegetation grows back (Harr, 1983; Ziemer, 1986; Brown et al., 2005; NRC, 2008). Recent assessments of water yield augmentation from forest removal in the Sierra Nevada have estimated that increases in annual water yield from forest removal levels are eliminated in about seven to less than 20 years after forest removal (Bales et al., 2011; Podolak et al., 2015).



Photo 3. Sierra Nevada forests have naturally heterogeneous and patchy structure due to natural disturbance processes including wind, mass erosion, fire, and other agents of mortality, such as insects. Tree establishment and successional recovery after disturbance is relatively rapid in many forest types. Behind the mature stand in the foreground of this view on the Plumas National Forest are stands vigorously regenerating from past disturbance. Photo by C. Frissell.

The re-growth of vegetation not only ultimately eliminates the transient increases in annual water yield from forest removal, but can cause annual water yields to decrease relative to yields prior to forest removal. As discussed in greater detail in a later section, several studies

have documented significant and persistent reductions in low flows after vegetative recovery from forest removal (Harr, 1983; Hicks et al., 1991; Moore et al., 2004; Jones and Post, 2004; Perry, 2007; Reid, 2012). These longer-term reductions during the low-flow season when water supply is most critical, adversely affect societal water uses and aquatic resources.

2.3 Estimated magnitude of water yield response to forest removal

Although the slope of the linear regression line in Figure 1 indicates an increase of approximately 27.3 mm in annual water yield per 10% of conifer forest removed in small watersheds, this likely significantly overestimates the incremental change in annual water yield with forest removal. This is because, as mentioned, most of the data in Figure 1 is for the maximum annual response over years when water yield was measured, while mean response over multiple years is considerably lower due to variations in precipitation and the decline in water yield over time.

Bosch and Hewlett's (1982) estimate of a *ca.* 40 mm increase in annual yield per 10% removal of conifer cover can be reasonably dismissed as unreliable for two primary reasons. First, it was based on the more limited data available at the time, in comparison with that in Figure 1. Second, it likely overestimates changes in water yield because it is based on the analysis of the maximum annual increase in water yield measured in the first few years after conifer forest removal, which considerably overestimates multi-year water yield response, as previously discussed.

Marvin's (1996) assessment of potential changes in mean annual water yield in the Sierra Nevada in response to forest removal demonstrates the bias resulting from using maximum single-year changes in water yield in response to forest removal rather than mean annual change over multiple years. Regression analysis of five-year mean annual yields and corresponding forest removal levels for 31 results from watershed studies in the western U.S., indicates an increase in mean annual water yield of *ca.* 13 mm (0.51 in.) per 10% reduction in forest cover in the range of mean annual precipitation levels estimated to occur in the Sierra Nevada (Marvin, 1996). At the 90% confidence interval, the response of mean annual water yield ranged from 0 to 26 mm per 10% of forest cover removed (Marvin, 1996). The R^2 was 0.14, corroborating the high variability in water yield response found in other studies. Marvin (1996) noted that the use of estimated reductions in evapotranspiration is likely to consistently overestimate increases in water yield.

Although Marvin's (1996) analysis of the results of 9 studies with relatively high levels of mean annual precipitation (960-2400 mm) indicated an increase of *ca.* 31 mm in water yield per 10% of forest removal, this level of response is unlikely to be valid for estimating water yield from forest removal at the scale of the Sierra Nevada for four reasons. First, these relatively high levels of precipitation are not applicable at the scale of the entire Sierra Nevada. Instead, they are applicable to only a smaller, wetter portion of the Sierra Nevada.

Second, a considerable amount of area in the Sierra Nevada with precipitation at the higher end of the 960-2400 mm range is in protected areas (wilderness, national parks) that are not available for forest removal aimed at increasing water yields.

Third, much of the area precipitation at the higher end of the 960-2400 mm range has subalpine or alpine vegetation. There is no potential to increase water yield via forest removal in areas with alpine vegetation (Ziemer, 1986). Forest removal in subalpine areas is unlikely to increase water yields, because forest density is already low (Kattelmann, 1987).

Fourth, it is unlikely that most forest removal on Sierra Nevada national forests will be primarily located in wetter areas with relatively high mean annual high precipitation, >960 mm, if such treatments are located in areas where they have the greatest, albeit still limited, potential to affect fire behavior. A considerable amount of recent, on-going, and likely future forest removal is aimed at attempting to modify fire behavior or restore fire regimes (USFS, 2004). For instance, the estimates of potential water yield effects in Podolak et al. (2015) are primarily based on forest removal under the rubric of such aims. Ostensibly, these fire-related treatments would tend to be concentrated in drier rather than wetter forests because the former burn more frequently, potentially have more altered fire regimes, and have a greater likelihood of treatments affecting fire behavior (Schoennagel et al., 2004; Rhodes and Baker, 2008). For these combined reasons, Marvin's (1996) estimate of 13 mm in mean annual water yield per 10% reduction in forest cover in the range of mean annual precipitation levels estimated to occur throughout the Sierra Nevada is likely a better estimator than that derived for areas with relatively high levels of mean annual precipitation.

Sahin and Hall (1996) analyzed the relationship between the five-year mean change in annual water yield versus levels of vegetation removal from many studies and estimated that annual water yield increased by *ca.* 23 mm for every 10% of conifer forest cover removed. This result also corroborates that analyses based on the maximum annual change in water yield measured after forest removal are biased, overestimating multi-year water yield changes in response to forest removal.

The range of 13-23 mm mean annual increase water yield per 10% conifer forest removal from Marvin (1996) and Sahin and Hall (1996) likely provide a more reasonable estimates of multi-year water yield changes after forest removal, as a first approximation, than those based on maximum water yield change in a single year. However, both estimates are still prone to the same problems previously discussed: changes in annual water yield show high levels of variability and are strongly influenced by levels of precipitation that occur during the measurement period.

Studies and assessments have consistently noted that more than *ca.* 20% of conifer cover must be removed in order to measurably increase in water yields in small experimental watersheds (Bosch and Hewlett, 1982; Stednick, 1996; Marvin, 1996), although there is some difference among regions in the U.S. (Stednick, 1996). In four of the five studies reviewed by Marvin (1996) where 25-28% of forest was removed, there were no increases in annual water yield. The data in Figure 1 also support the generalization that measurable increases in water yield in small watersheds are unlikely unless 20% or more of forest cover is removed. As shown in Figure 1, the lower end of the 95% confidence interval from the regression analysis only reaches a positive water yield response value at a conifer removal level at or above *ca.* 25%. Notably, this is based on maximum annual rather mean water yield response, which overestimates water yield response to forest removal. For these reasons, it can be reasonably

accepted that the removal of less than 20% of forest cover is unlikely to provide any measurable increase in annual water yield from small watersheds.

Although there is considerable variation in the response of water yield to forest removal, the foregoing indicates that projected increases in water yield in response to proposed levels of forest removal in Podolak et al. (2015) are likely considerably overestimated. Podolak et al. (2015) used an estimate of 40 mm per 10% of forest removal derived from Bosch and Hewlett (1982) for the high end of water yield response and Sahin and Hall's (1996) estimate of *ca.* 22 mm per 10% of forest removal as the low end. Podolak et al. (2015) assumed that these increases in water yield would be maintained for seven years, which is not tenable for the high end estimate because it is based on single year maximum responses. In contrast, as previously discussed, water yield effects from forest removal decrease over time and the effect averaged across years is, therefore, always significantly lower than the maximum annual response.

Podolak et al. (2015) also overestimates potential increases in water yield by assuming that it would increase in watersheds with less than 20% forest removal. Available scientific information indicates that this is extremely unlikely. Figure 4 in Podolak et al. (2015) indicates that forest removal in the Truckee, Mokelumne, Yuba, Lassen Creeks (including Deer, Mill, Butte, and Battle Creeks) watersheds would have less than 20% forest removal under the scenarios examined. Yet, Podolak et al. (2015) estimates increased water yield in all of these watersheds, contrary to available scientific information.

Table 1 and Figure 2 provide an indication of the overestimation of increased water yields in Podolak et al. (2015), by providing a comparison of the water yield estimates from forest removal in Sierra Nevada watersheds from Podolak (2015) to those from using more reasonable estimates from the literature. The latter include: a) a 13 mm increase in mean annual water yield per 10% of forest removal from Marvin (1996) as a reasonable, but liberal estimate⁴ of the low end in water yield response; b) a 26 mm increase in mean annual water yield per 10% of forest removal from Marvin (1996) as a reasonable, but liberal estimate of the high end in water yield response; and, c) no increase in water yield in watersheds with less than 20% forest removal, consistent with available scientific information. In both Table 1 and Figure 2, the levels of forest removal used to estimate water yield are intrinsically assumed to be the same as that in Podolak et al. (2015). The estimates in Table 1 and Figure 2 also use the assumption in Podolak et al. (2015) that forest removal in these Sierra Nevada watersheds has the same effect on water yield regardless of location and watershed scale, although this assumption is not likely valid, for several reasons, as discussed in greater detail later in this paper.

⁴ At the 90% confidence interval, the estimated change in mean annual water yield per 10% forest removal from the analysis of Marvin (1996) is 0 to 26 mm.

Table 1. Comparison of estimated percent change in mean annual streamflow (water yield) in response to forest removal in Sierra Nevada watersheds from Podolak et al. (2015) versus those from Marvin (1996) and other literature. Watersheds with an “*” would have <20% forest removal based on information in Podolak (2015).

Watershed	Estimated increase in water yield from forest removal reported in Podolak et al., 2015 (% of mean annual streamflow)		Estimated increase in water yield from forest removal ^a based on scientific information described in the text (% of mean annual streamflow)	
	Low estimate	High estimate	Low estimate	High estimate
Feather	2%	6%	1.2%	3.9%
American	1%	3%	0.6%	2.0%
Yuba*	0%	1%	0.0%	0.0%
Lassen Creeks*	1%	2%	0.0%	0.0%
Mokelumne*	1%	3%	0.0%	0.0%
Truckee*	2%	5%	0.0%	0.0%
Cosumnes	2%	6%	1.2%	3.9%
Bear*	0%	1%	0.0%	0.0%

^a Estimates intrinsically assume forest removal levels are the same as those in Podolak et al. (2015). Estimates are prorated based on data from Podolak et al. (2015) versus those from Marvin (1996) for mean annual water yield response per 10% forest removal in watersheds with forest removal >20%. For watersheds with forest removal levels <20%, water yield increase is zero, consistent with the literature.

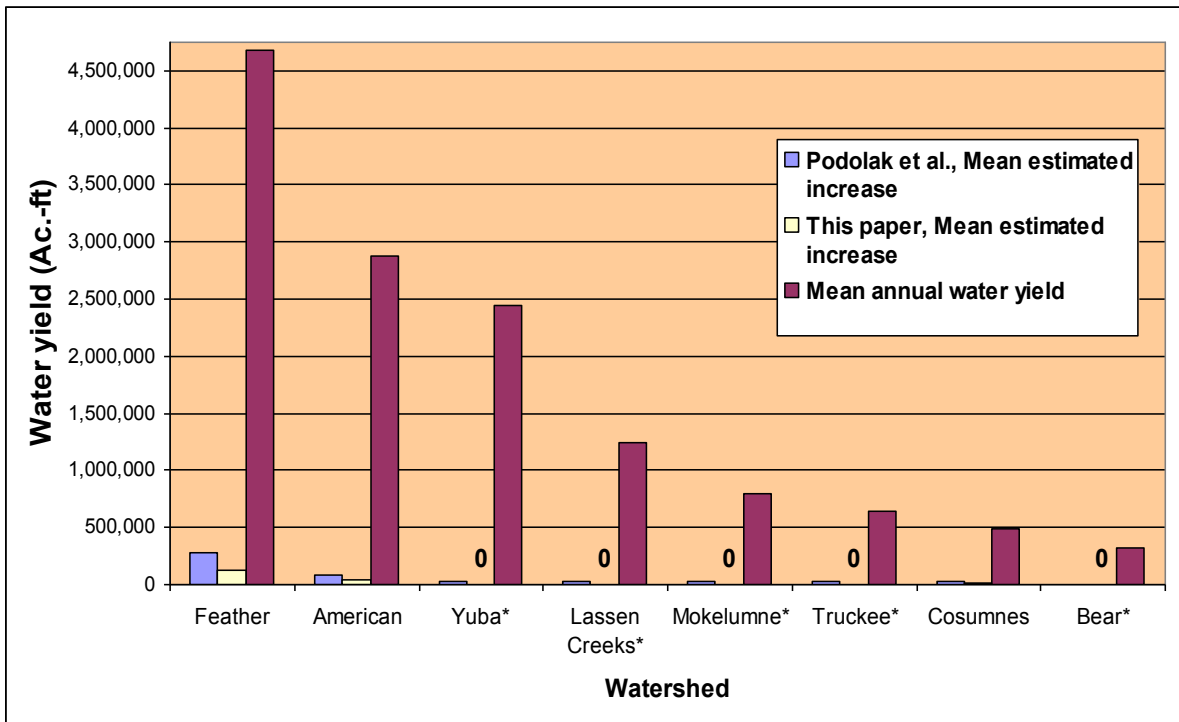


Figure 2. Mean water yield and comparison of estimated increase in water yield forest removal in Sierra Nevada watersheds from Podolak et al. (2015) versus those from Marvin (1996) and other literature. The mean estimates shown are the mean of the high and low estimates, as shown in Table 1. Watersheds with an “*” would have <20% forest removal based on information in Podolak (2015), and, hence, zero estimated increase in water yield based on available literature, as denoted by the bold “0”.

2.4 Precipitation strongly affects water yield response

Studies and assessments have consistently found that water yield response to forest removal declines with declining mean annual precipitation, as shown in Figure 3. This relation applies over time within a given watershed, as well as across space between forests in wetter and drier climates. For instance, Marvin (1996) noted that mean annual precipitation explained so much of the variability in water yield change after forest removal that levels of forest removal were almost insignificant in comparison. Importantly, the aggregate data in Figure 3 overestimate water yield response to forest removal because all but a few data points are for the single-year maximum water yield measured after forest removal.

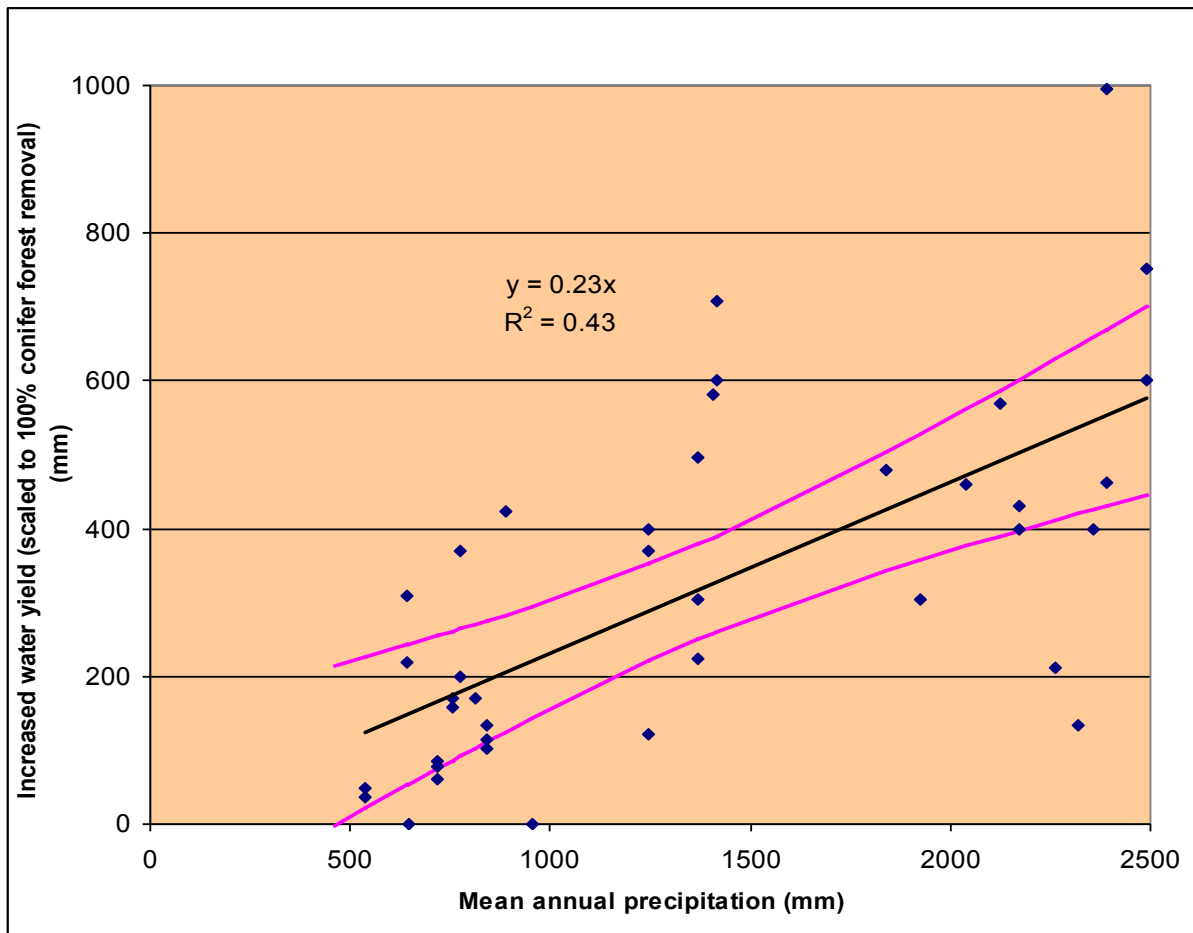


Figure 3. Maximum annual water yield response scaled to 100% removal versus mean annual precipitation (After Brown et al., 2005). All but a few data points are for the single year maximum water yield measured after forest removal, hence, the aggregate data inflate multi-annual water yield response. Data are from Troendle and King (1987), Marvin (1996), and Brown et al. (2005). Regression intercept set at zero because at zero mean annual precipitation forest removal cannot affect water yield. Black line is linear regression line. Purple lines demarcate the 95% confidence interval for the regression line.

Numerous assessments have concluded that it is unlikely forest removal can measurably increase in water yields in watersheds where mean annual precipitation is less than ca. 460

mm (18.1 in.), even when watersheds are nearly deforested (Ziemer, 1986; Marvin, 1996; MacDonald and Stednick, 2003; Brown et al., 2005; NRC, 2008). This is generally corroborated by the aggregate data in Figure 3, especially because these data overestimate multi-annual water yield response to forest removal, as discussed in a previous section. Kattelmann (1987) noted that there is low potential for increasing water yield in areas with less than ca. 680 mm of mean annual precipitation, which is also corroborated by the data in Figure 3.

Although changes in water yield that occur with forest removal plainly correlate with mean annual precipitation, it is unlikely that long-term mean annual precipitation exerts a direct influence on water yield response. Rather, mean annual precipitation provides a robust indicator of the annual precipitation levels experimental watersheds tend to receive during the period of measurement in studies of water yield and forest removal. For a given watershed, it is the distribution and quantity of precipitation among water years that influences and constrains the effect of forest removal on stream flow and water yield.

It is well-established that at given level of forest removal, annual changes in water yield decrease with decreasing precipitation, other factors remaining equal. Thus, water yield increases per increment of forest removal are the lower and less likely in drier years and higher and more likely in wetter years (Bosch and Hewlett, 1982; Harr, 1983; Troendle, 1985; Troendle and King, 1985; 1987; Gottfried, 1991; Stednick, 1996; Marvin, 1996; MacDonald and Stednick, 2003; Brown et al., 2005; NRC, 2008). This has been demonstrated repeatedly in experimental watersheds (e.g., Troendle and King, 1985; 1987; Gottfried, 1991). Because annual precipitation can vary considerably, this variability is likely a primary reason for the variability in water yield in response to forest removal.

The effect of precipitation on the water yield-forest removal response is a critical issue from perspective of the potential water supply benefits. Due to this effect, forest removal has very limited promise for significantly increasing water yields during drought cycles, when it would be most beneficial for downstream uses. This significant limitation on the utility for improving water supply has been repeatedly noted in assessments of the approach (Harr, 1983; Ziemer, 1986; Kattelmann, 1987; Sedell et al., 2000; MacDonald and Stednick, 2003; NRC, 2008).

Because water yields undergo the greatest increases during the wettest years, they may be of limited or no benefit for downstream water uses during these periods. For instance, in wet years during cycles of higher precipitation, reservoirs that are required to operate for flood control may reach a capacity that requires that they simply spill any additional water coming from upstream forests in order to meet flood control mandates. Water supply reservoirs also commonly fill to capacity in wetter years, requiring that they spill additional flows that might have been created by forest removal during a period of high runoff.

2.5 Temporal distribution of increases in water yield from forest removal

The annual distribution of changes in water yield in response to forest removal further limits their benefits for downstream uses. Studies have repeatedly documented that the overwhelming majority of increased runoff caused by forest removal occurs during the

wettest part of the year when runoff is relatively high (Harr et al., 1983; Troendle and King, 1985; 1987; Kattelman, 1987, Stednick, 1996; Marvin, 1996; MacDonald and Stednick, 2003; NRC, 2008, Bales et al., 2011, Podolak et al., 2015). In areas where snowmelt is the dominant runoff source, increases in water yield from forest removal is almost solely confined to the period of snowmelt runoff (Troendle and King, 1985; 1987; Ziemer, 1986; Kattelman, 1987, Gottfried, 1991; Stednick, 1996; Marvin, 1996; MacDonald and Stednick, 2003; NRC, 2008, Bales et al., 2011).

Most water yield increases from forest removal in small watersheds occur in years and seasons when the additional water is least needed and can seldom be effectively used. Increased water yield is primarily restricted to seasonal periods when downstream seasonal water demands are relatively low, as are the immediate benefits, while supply is relatively high. As others have noted (Harr, 1983; Ziemer, 1986; MacDonald and Stednick, 2003), due to this timing, any additional water yield from forest removal must be stored in reservoirs for a minimum of several months in order to be useful when water demand is high, especially relative to supply: in drier months during periods of seasonally low runoff. Such long-term storage in reservoirs may not be possible during the annual high runoff period, especially during wet years, due to capacity and flood control mandates, as previously discussed. Hence, the timing of increased runoff from forest removal compounds the limitations on downstream benefits caused by precipitation effects: most of additional water is not only supplied during the wettest time of the year, but to the greatest extent during the wettest years.

2.6 Forest removal increases peakflows

Many studies have found that forest removal of a magnitude (> 20%) sufficient to increase water yields also increases peakflow. This is corroborated by the data in Table 2 which summarizes peakflow and annual water yield data from readily available literature for 11 results from studies that reported the responses of peakflow and/or annual water yield in response to conifer removal in watersheds with runoff dominated by snowmelt in the western North America. The data on peakflow and forest removal in areas dominated by snowmelt in Grant et al. (2008) also corroborate that peak flows are significantly elevated by levels of forest removal greater than 20% over a watershed. Studies in snowmelt-dominated areas across the West indicate that when forest removal is extensive enough to increase annual water yields, the greatest increase occurs during the peak snowmelt period (MacDonald and Ritland, 1989), including during rain-on-snow events that contribute to peakflows (Berris and Harr, 1987; Harr and Coffin, 1992). The elevation of peakflow by forest removal sufficient enough to increase water yields is consistent with the well-documented finding that most of the increased water yield triggered by forest removal occurs during the period of highest runoff.

Forest removal increases peakflow via several mechanisms. These include reduction in canopy interception and evapotranspiration and increased snow accumulation, as well as, greatly increased rates of snowmelt due to canopy reduction effects (Kattelman, 1991; Alila et al., 2009; Reid, 2010; Varhola et al., 2010). Roads, which inexorably occur in tandem with forest removal, also contribute to peakflow elevation by intercepting subsurface runoff, increasing surface runoff on compacted surfaces, and accelerating the delivery of elevated surface runoff to streams (Wemple et al., 1996; La Marche and Lettenmaier, 2001; Alila et al.,

2009). The effect of roads on peakflows are additive to those caused by forest removal (La Marche and Lettenmaier, 2001).

MacDonald and Stednick (2003) suggested that complete deforestation of watersheds in the Colorado Rockies would increase annual maximum peakflows by *ca.* 40-50%. However, the data assembled in Table 2 indicates that increases in peakflows of 50% or greater occur at levels of forest removal well below complete watershed deforestation (e.g., Gottfried, 1991; Troendle and King, 1997; Burton, 1997).

Table 2. Summary results from some studies of peakflow and annual water yield (AWY) responses to conifer removal.

Study area	Forest removal (%)	Mean annual peakflow increase (%)	Mean AWY increase (%)	Source data	Analysis period (yrs)
Thomas Cr. Az	34	65	45	Gottfried, 1991	8
Fool creek, CO	40	23	- ^a	Troendle and King, 1985	28
Brownie Cr, UT	25	66	52	Burton, 1997	19
NF Deadhorse Creek, CO	36	50	25	Troendle and King, 1987	7
Wagon Wheel, CO	100	50	25	Van Haveren, 1981, as cited in Gottfried, 1991	Not reported
Camp Creek, Interior BC, Canada	30	21	21	Cheng, 1989	6
Horse Creek, ws-12, ID	36.6	15	29	King, 1989	5
Horse Creek ws-14, ID	29.2	35	23	King, 1989	5
Horse Creek ws-16, ID	25	36	13	King, 1989	4
Horse Creek ws-18, ID	33.4	34	17	King, 1989	4
Main Deadhorse Creek, CO	11.4	0	0	Troendle and King, 1987	7

^a Change in water yield only reported for a portion of each year in the study.

The data on mean multi-annual peakflow increases in Table 2 do not represent the maximum peakflow response to forest removal, because these effects tends to diminish over time, with greatest effects in the first several years after logging, although precipitation plays a strong role (Troendle and King, 1985; 1987; King, 1989; Gottfried, 1991; Alila et al., 2009). For instance, forest removal increased peakflow by mean of *ca.* 23% over 28 years in Fool Creek, Colorado (Table 2), but peakflow increased by an average of *ca.* 32% during the first 10 years

after forest removal and was increased by more than 50% in five of those 10 years, based on the data in Troendle and King (1985).

Although some assessments have posited that larger, more infrequent peak flows are not affected by forest removal (e.g., Troendle and Olsen, 1994; MacDonald and Stednick, 2003; Grant et al., 2008), the findings of Brown et al. (1974) conflict with this appraisal. The estimated flow increases for large peakflows with estimated recurrence intervals of 100 to 200 years in northern Arizona (Brown et al., 1974), indicate that three watersheds with 100%, 75%, and 32% forest removal, respectively, had peakflows increased by 167%, 88%, and 20%.

Alila et al. (2009) noted that the notion that relatively infrequent, larger peakflows are not affected by forest removal is partially premised on the following: a) soil and canopy interactions with precipitation are a major way that forest removal affects peakflow generation; b) the relative importance of these interactions with precipitation with respect to peakflows decline with increasing magnitudes of peakflow-generating events (MacDonald and Stednick, 1996), and c) the lack of statistical significance of increases in larger peakflows with longer return intervals. However, increases in peakflow from snowmelt are influenced by other factors besides soil and canopy interactions, including increased rates of snowmelt and the effects of roads, both of which may not decline in importance as peakflow event size increases (Alila et al., 2009).

The lack of statistical significance for increases in larger peakflows with greater return intervals may be a matter of sample number, length of streamflow record, and associated statistical power, rather than actual differences in flow magnitudes (Bowling et al., 2000; Alila et al., 2009). Larger flows with longer return intervals intrinsically occur less frequently than smaller peakflows with lower return intervals,⁵ resulting in steadily dwindling sample number for larger flows. Statistical power decreases dramatically with decreases in sample number. As a result, the size of the minimum effect that can be statistically detected increases dramatically with decreasing sample number. Therefore, the lack of statistically significant differences from forest removal for larger flows may be primarily due to the statistical effects rather than the actual magnitude of change in these peakflows (Bowling et al., 2000; Alila et al., 2009).

Forest removal not only increases annual peakflows, but also increases the frequency of peakflows of a given magnitude (Alila et al., 2009). For instance, re-analysis of flow data from Fool Creek, Colorado indicated that forest removal shifted peakflows of a magnitude that prior to forest removal had a recurrence interval of *ca.* 30 years, to a recurrence interval of *ca.* 14 years after forest removal (Alila et al., 2009).

Research indicates that the magnitude of peakflow increases from forest removal increases with increasing precipitation and runoff (Troendle and King, 1985; 1987; King, 1989; Cheng, 1989; Gottfried, 1991; Troendle and Olsen, 1994; Burton, 1997). Troendle and King (1985; 1987) found that precipitation amounts accounted for much of the inter-annual variation in peak snowmelt caused by forest removal. King (1989) also found that the level of peakflow

⁵ This is irrefutably occurs, because the frequency of occurrence is the factor used to estimate recurrence interval of peakflows of different sizes (e.g. Dunne and Leopold, 1978).

elevation from forest removal was positively correlated with the magnitude of peakflow in four watersheds in Idaho. Thus, the absolute (rather than relative) magnitude of peakflow increases are greatest during the wettest years when flood potential is highest and increased flows are least desirable. These increases are least during droughts (when additional runoff is most desirable), which partially explains why water yield increases decline with declining annual precipitation.

Many studies in snowmelt-dominated systems indicate that relative increases in peakflows are greater than the relative increase in annual water yield, although the magnitude of response exhibits some variability (Table 2). One significant cause for this effect is that, as previously discussed, most of the increase in water yield from forest removal occurs during the peakflow period, with little or nominal effects on flows during the low flow period.

Forest removal also increases the duration of peakflows (Troendle and King, 1985; 1987; Troendle and Olsen, 1994; Burton, 1997). Therefore, it is likely that forest removal would contribute to increasing the duration of flooding during flood events.

2.7 Forest removal effects on low flows

It is highly unlikely that forest removal in the snow zone, which includes most of the public lands of the Sierra Nevada, increases summer low flows. If they occurred, such increases could be beneficial to both aquatic ecosystems and water supplies, provided they were fairly damped and were sustained across years.

Many studies of the effects on conifer forest removal on streamflow in watersheds dominated by snowmelt have found no or negligible effects on low flows (Troendle and King, 1985; 1987; Troendle, 1987; Cheng, 1989; King, 1989; Gottfried, 1991; Troendle and Olsen, 1994; MacDonald and Stednick, 2003; Biederman et al., 2014). In contrast to these results, one study in Utah found an increase in low flows (Burton, 1997), which is one of the reasons the results were disputed by Troendle and Stednick (1999). The anomalous increase in low flow documented by Burton (1997) may be due to site characteristics (Burton, 1999). However, the consistency of the lack of low flow response to forest removal in most studies indicates that low flow augmentation is not a likely or reliable outcome in snowmelt-dominated watersheds.

One of the cited reasons for the lack of increased low flows in snowmelt-dominated areas is the compensatory water uptake by remaining vegetation the dry summer and early fall, which precludes increases in low flows, except during the wettest years (Gottfried, 1991; Stednick, 1996; Troendle, 1987; Troendle and Olsen, 1994; MacDonald and Stednick, 2003). Boggs et al. (2015) found that riparian vegetation greatly increased water uptake in response to increased runoff caused by forest removal, using *ca.* 43% more water after forest removal than prior to removal.

Any increase in low flows that might result from forest removal is likely to be more transient than effects on annual water yield, based on consideration of watershed processes (MacDonald and Stednick, 2003). The vegetative communities in Sierra Nevada are adapted to use available soil moisture in order to persist during drought periods. For these combined

reasons, even high levels of forest removal are unlikely to provide any additional water during dry seasons, especially during the driest years, when it would be most beneficial for downstream uses.

Increased snowpack sublimation and abiotic evaporation caused by forest canopy loss may also contribute to the consistent lack of low flow response to forest removal in snow dominated areas. Biederman et al. (2014) found that high levels of tree mortality did not increase streamflow in Colorado, despite considerable reductions in transpiration and canopy interception of snow. Confluent evidence indicated that this lack of streamflow response was due to increased snowpack sublimation and abiotic evaporation caused by forest canopy loss (Biederman et al., 2014).

Studies have documented increases in low flows in response to forest removal in areas with relatively high levels of precipitation, a considerable amount of precipitation comprised by rain, and/or at high levels of forest removal (Harr, 1983; Reid, 2012). But, even under these conditions, increases in low flows are transient, relatively insignificant in terms of effects on water yield, and are often followed by decreases in low flows relative to pre-treatment levels, in the absence of additional forest removal (Harr, 1983; NRC, 2008; Reid, 2012).

For instance, Harr (1983) documented transient increases in low flows in a watershed in Oregon with mean annual precipitation of *ca.* 2390 mm that underwent 100% forest removal followed by burning (Figure 4). The deforestation of the watershed precluded compensatory uptake of water by remaining vegetation, which likely contributed to the observed low flow response. Increases in low flows were greater in wetter years, but declined rapidly after the cessation of forest removal (Harr, 1983). Reduced low flows occurred four years after forest removal ceased (Figure 4). Low flows were consistently reduced relative to pre-treatment levels eight years after forest removal ceased and the magnitude of low flow reductions increased over time (Figure 4). These reductions in low flows from the watershed continued to persist more than thirty years after the initial deforestation (Jones and Post, 2004).

This reduction in low flows is likely because stands that grew after forest removal had higher rates of evapotranspiration and higher basal areas than the older stands they succeeded (Moore et al., 2004). The analysis of Stubblefield et al. (2012) of low flows in the Mattole River, California also indicated that young stands regenerating after forest removal are likely to reduce low flows. Reductions in low flows after a period of forest regrowth following forest removal have also been documented by other studies in Oregon (Hicks et al., 1991; Jones and Post, 2004; Perry, 2007) and California (Reid, 2012).

Reid (2012) found that after forest removal, initial increases in low flows lasted *ca.* 7-16 years in two watersheds with mean annual precipitation of 1170 mm comprised primarily of rain in coastal California. These increases were followed by decreases in low flows relative to pre-treatment levels as vegetation recovered. In one of the two watersheds, decreased low flows persisted for *ca.* 20 years and may continue (Reid, 2012).

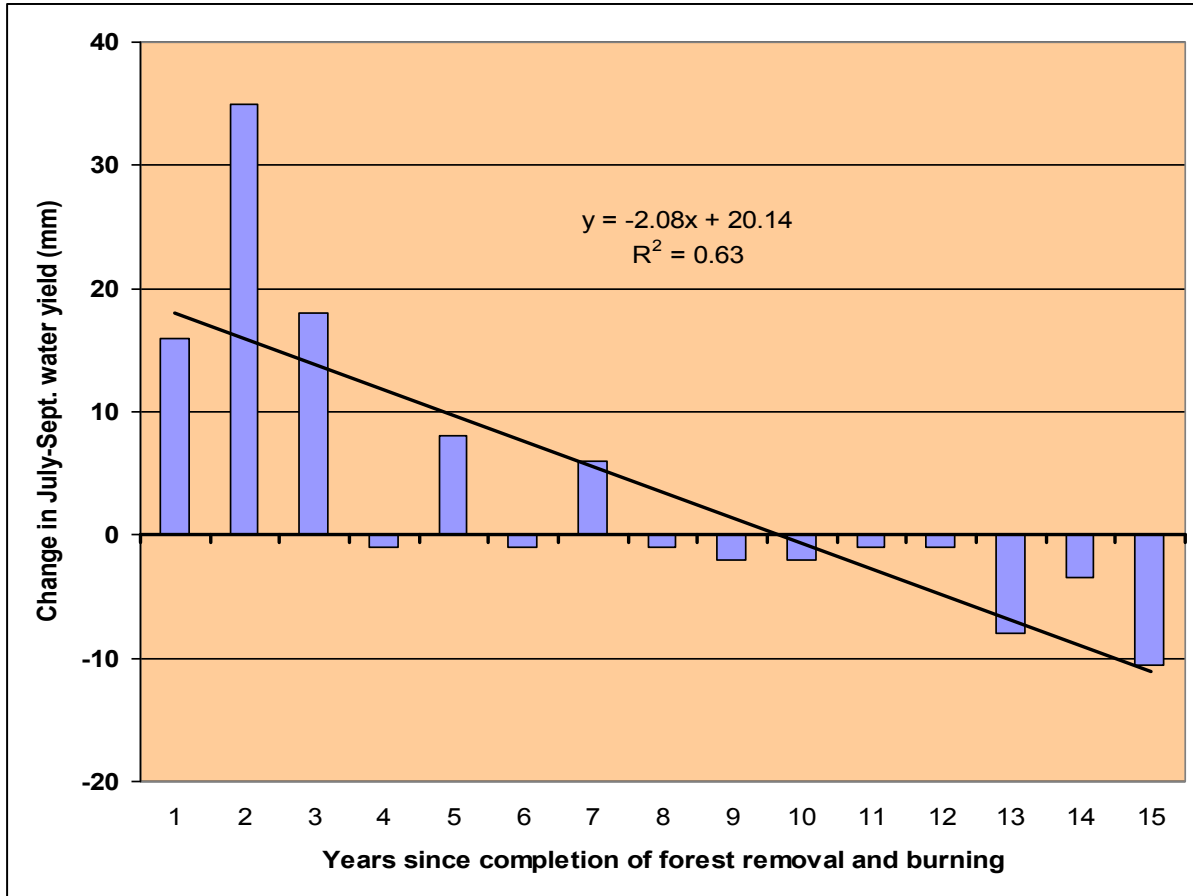


Figure 4. Changes in low flow (July through September) water yield (mm) annually relative to estimated pre-treatment flows over 15 years after the cessation of forest removal in a watershed in Oregon (from Harr 1983). The watershed was completely logged and then burned, starting four years prior to the cessation of logging and burning.

Although Harr (1983) documented transient low flow increases in response to deforestation in an area with relatively high precipitation, the magnitude of the increase was relatively insignificant in terms of annual water yield. The mean increase in low flow water yield over the first seven years after forest removal only comprised *ca.* 0.8% of the pre-treatment mean annual water yield. Over the 15 years of data after forest removal ceased, the mean increase in water yield from complete deforestation of the watershed only comprised *ca.* 0.3% of the pre-treatment mean annual water yield. This underscores that even large relative (e.g., percent) increases in low flows have small effects on water yield, because low flows comprise a small fraction on annual water yield. Further, as previously discussed, the reductions in low flows from the watershed continue to persist more than 30 years after deforestation.

The magnitude of transient increases in low flows found by Reid (2012) and Harr (1983) may not be applicable to forest removal over much of the Sierra Nevada. The studies occurred in areas with relatively high precipitation and increases in low flows tend to increase with increasing precipitation (Harr, 1983; MacDonald and Stednick, 2003). The study of Harr

(1983) involved complete deforestation of an experimental watershed, which is unlikely to occur on public lands in the Sierra Nevada.

Based on the foregoing, it is unlikely that forest removal in the snow zone of the Sierra Nevada will increase low flows. If low flow increases do occur, they are likely to be quite transient, relegated to wetter years, and relatively insignificant for downstream water supply. Reductions in low flows are likely to occur sometime after initial forest removal in the absence of relatively frequent, extensive forest removal.

2.8 Forest removal effects on the timing and rate of snowmelt

It appears unlikely that forest removal can delay or extend snowmelt, because it typically accelerates snowpack ablation (loss) (Varhola et al, 2010), especially in openings, resulting in the earlier disappearance of snow cover. In the Sierra Nevada melt rates are typically much higher in openings than under forest cover (Kattelmann, 1991; Rittger, 2012). This is consistent with the results from analysis of data from 65 sites in North America, which found that reductions in forest cover significantly and consistently increase rates of snowpack loss (Varhola et al., 2010).

Snowpack ablation includes sublimation, the direct evaporative depletion of snowpack. Because the moisture is lost directly to the atmosphere, sublimation does not contribute to snowmelt runoff. This mechanism of snowpack loss can also increase with reductions in canopy density (Varhola et al. 2010). Significant tree mortality can increase snowpack sublimation and abiotic evaporation to a degree that offsets reduced transpiration from tree mortality, precluding increases in streamflow, as documented by the detailed study of hydrologic responses in forests with high levels of beetle-induced mortality in Colorado (Biederman et al., 2014).

Studies in the Sierra Nevada have repeatedly found that snow cover lasts longer under forest cover than in openings. This occurs despite increases in snow accumulation with decreasing forest cover, as widely documented in western North America (Varhola et al., 2010). In the Sierra Nevada, Kattelmann (1991) found that snowpack under forest cover persisted for an average of 18 days longer than in an adjacent opening over the course of 10 years of monitoring. Although the average snowpack water equivalent in the opening averaged 200 mm greater than under forest cover, melt rates in the opening were significantly higher than under forest cover, which caused the snowpack to persist longer under forest cover than in the opening (Kattelmann, 1991). Anderson (1956, as cited in Raleigh et al., 2013) found that snow cover in forests persisted for *ca.* 16 days longer than in forest openings in the Sierra Nevada. Raleigh et al., (2013) found that snow cover disappeared many days earlier in several openings than in adjacent forested areas in the Sierra Nevada. This earlier loss of snow cover in areas with lower levels of forest canopy cover in the Sierra Nevada is consistent with the results of Varhola (2010) which indicate that forest removal increases snowpack loss to a greater degree than it increases snow accumulation.

It is theoretically possible to delay snowmelt via forest removal if openings were very carefully designed, located, and implemented to minimize snow ablation in the openings (Kattelmann, 1987). However, this may not be consistently tractable with the large-scale

removal of vegetation that would be necessary to boost water yield at the scale of river basins. Additionally, the location and design of such openings would render them highly unlikely to significantly affect fire behavior. This is because such openings would need to be located in areas that minimize snow ablation, primarily on northerly aspects. In contrast, forest removal aimed at modifying fire behavior is likely most effective when it is primarily located on southerly slopes, with higher evapotranspiration and where fires tend to occur more frequently (Hessburg et al., 2015), but where snowmelt is typically accelerated by openings created by forest removal. Thus, forest removal aimed at modifying fire behavior is more likely to accelerate than delay snowmelt.



Photo 4. Many Sierra forests are relatively low in stem density. Forest removal is likely to increase wind speed and solar insolation at the ground, speeding snowmelt and increasing sublimation losses from snowpacks. Photo by C. Frissell.

The results in Varhola et al. (2010) indicate that reductions in forest cover (including loss of cover from forest removal) have overarching, consistent effects that, on balance, increase rates of snowpack loss. Although numerous factors likely affect the magnitude of the increase in the loss of snow cover caused by forest removal, including elevation, aspect, local meteorology, and, the arrangement and geometry of created openings, aggregate data from numerous sites in western North America indicate that changes in forest removal explains much of the observed variability in rates of snowpack loss (Varhola et al., 2010). This

indicates that forest removal exerts a stronger influence on the rate of snowpack loss and its persistence than do other factors.

In contrast to the foregoing, Lundquist et al. (2013) found that in areas with warmer winters on several continents, snow cover persisted longer in openings than in forested sites, although several forested sites exhibited the opposite response. Notably, the analysis in Lundquist et al. (2013) did not include data from Kattelmann (1991) or Raleigh et al. (2013), which documented earlier loss of snow cover in openings in the Sierra Nevada.

Based on simulation modeling of estimated climatic changes on snowmelt and runoff, Cristea et al. (2013) estimated that forest removal could delay snowmelt runoff in the Tuolumne Basin of the Sierra Nevada. However, given the preponderance of field-verified evidence from the Sierra Nevada and geographically extensive results reported from across western North America (Varhola et al., 2010), it appears unlikely that forest removal can delay snowmelt or extend the snowmelt period to later in the year by increasing snow cover persistence. Instead, available evidence indicates it is most likely that forest removal will cause snowmelt to end earlier in the year. Thus, forest removal in the Sierra Nevada is unlikely to provide water supply benefits hypothesized to accrue from a delay in the annual loss of snow cover.

2.9 Transmission and storage losses

Any increases in annual water yields that might occur in watersheds affected by forest removal are likely to be diminished by transmission losses along streams before reaching points of downstream use (Ziemer, 1986; Gottfried, 1991). Transmission loss sources include evapotranspiration by riparian vegetation and seepage through channel and the near-channel environment, such as floodplains (Gottfried, 1991). Water lost through these vectors is generally not available for consumptive human uses. Similarly, it is well-documented that water stored in reservoirs is subject to storage losses via evaporation and seepage (Lund, 2006). This is likely to diminish the magnitude of any water yield increases prior to use, because, as discussed, the majority of increased water yield occurs during periods of high runoff. Hence, any potential increase in water yield from forest removal must be stored in reservoirs for several months to be useable when water demand is relatively high (Harr, 1983; Ziemer, 1986; NRC, 2008).

It is likely that the magnitude of transmission losses vary with factors that affect seepage and evapotranspiration, including soil conditions, hydrologic conditions near streams, riparian vegetation type and conditions, and meteorological conditions. While a comprehensive review of these influences on the magnitude of transmission losses in streams is beyond the scope of this report, it is likely that the magnitude of transmission losses vary among years and seasons.

Transmission losses are likely highest during the relatively warm, low flow period due to relatively high evapotranspiration rates by riparian vegetation. These seasonal losses may be compounded by lower riparian water table elevations and increased levels of available soil moisture storage during the low flow season, which are conducive to seepage losses. Thus, any small increases in low flows that might accrue from forest removal in forested watersheds

are likely subject to the greatest seasonal level of transmission losses. It is also likely that transmission losses from seepage are relatively high during low flow periods during drought due to relatively depleted soil moisture levels and reduced water table elevations in riparian areas, both of which are conducive to seepage. This further diminishes the already remote prospects that forest removal can increase water supply during the drought periods.

In contrast, transmission losses are likely lowest annually during winter and peak runoff periods, especially during wetter years. Thus, elevated peak flows from forest may not undergo significant transmission losses and, thereby contribute to downstream high flows, including during floods.

Other factors remaining equal, increased transit distance between the mouth of affected watersheds and downstream points of use likely increases the magnitude of transmission losses affecting water yield (Ziemer, 1986), because increased transmission distance allows the accrual of multiple sources contributing to longitudinal transmission losses. Ziemer (1986) also noted that increases in water yield may itself foster increased growth of riparian vegetation which then resulting in increased transmission losses via evapotranspiration, creating an inexorable negative feedback loop from the standpoint of increasing water supply.

While an assessment of the magnitude of storage losses are beyond the scope of this report, it likely varies with factors that affect seepage, evaporation, and vegetative uptake, including soil conditions, hydrologic conditions near reservoirs, riparian vegetation type and conditions, and meteorological conditions. Rates of storage losses from evaporation are generally highest during drier and warmer summer months. Thus, storage losses are likely to be greatest for water that is stored during summer months. Other factors remaining equal, the magnitude of storage loss is likely increases with increasing time in storage due to seepage and evaporation rates. Thus, increased water yield that enters reservoirs during the high flow period and is stored for months before use during the low flow period is subject to relatively high levels of storage losses. Because this is likely to be the case for the majority of any increased water yield in order be useable during periods of higher demand (Ziemer, 1986; NRC, 2008), such stored water is subject to relatively high levels of storage losses.

While estimating the potential magnitude of these losses is beyond the scope of this report, these losses will diminish any potential increases in water yield from forested watersheds before water is ultimately used downstream. Hence, estimates of water yield increases in response to forest removal that fail to account for transmission and storage losses overestimate the magnitude of water delivered downstream, as well as their associated benefits.

2.10 Scale matters: Related uncertainties in the amount of water yield delivered downstream

The assessment of forest removal effects on downstream water supply is greatly complicated by the major differences in scale between the large scale of watersheds that supply water and the small watersheds that have provided data on water yield response to forest removal. It is not currently possible to reliably and accurately extrapolate findings from small-scale studies to much larger scales with the current state of hydrologic information (NRC, 2008)

Most of the data on the effects of conifer forest removal on water yield come from studies of relatively small, paired watersheds. Most of these watersheds have areas less than 600 ha (e.g., Harr, 1983; Troendle and King, 1985; 1987; King, 1989; Gottfried, 1991; Troendle and Olsen, 1994; Stednick, 1996; Marvin, 1996; King, 1989; Brown et al., 2005). For instance, in the studies examined by Marvin (1996), the maximum watershed area was 563 ha; only four of the studies were from watersheds with an area greater than 400 ha. Constraints on area is an inherent part of paired watershed studies because these studies rely on comparisons of responses between watersheds that are similar in terms of soils, vegetation, precipitation, geology, and other factors that influence hydrologic processes. Some small watersheds generally have greater homogeneity in these factors than do larger watersheds. Hence, it is exceedingly difficult, if not impossible, to conduct valid paired watershed studies at the scale of relatively large watersheds (NRC, 2008).

The area of watersheds that supply water for downstream uses and reservoirs that drain the Sierra Nevada are orders of magnitude greater than the area of most paired watershed studies. For instance, the water supply watersheds examined in Podolak et al. (2015) ranged in area from *ca.* 730 to 9,334 km², or about 500 to 1900 times greater in area than a 500 ha experimental watershed. Thus, the use of estimates from paired watershed studies to the scale of major water supply watersheds involves the extrapolation from relatively small, relatively homogenous watersheds to vastly larger and far more heterogeneous watersheds.

Such extrapolation is tenuous for two primary interrelated reasons. First, it is not currently possible to reliably and accurately extrapolate data from relatively small watersheds to much larger watersheds due to current limitations in hydrologic knowledge. As stated by the NRC (2008), “A key unresolved issue in forest hydrology is how to “scale up” findings from one part of a watershed to larger areas or the whole watershed.”

Second, as watershed size increases, so does the spatial and temporal heterogeneity in a host of factors that affect runoff generation and streamflow, including precipitation, land use, forest cover, vegetation types and cover, evapotranspiration rates, geomorphology, geology, soil types and conditions, hydrologic conditions, meteorological conditions, water withdrawals, and transmission losses. Collection of data on these and other factors that influence streamflow at the scales of large watersheds presents a formidable and likely insurmountable obstacle, especially because many of the aforementioned natural and anthropogenically-affected factors also have significant temporal variability. This variability at the scale of larger watersheds compounds the limitations on reliable extrapolation posed solely by scale.

While the scale-related uncertainties associated with how to accurately extrapolate information on water yield response from numerous small watersheds to larger watersheds are not immediately resolvable, it is not likely that data from small experimental watersheds can be simply extrapolated to larger watersheds under the assumption that they are the same regardless of scale, contrary to the operational assumptions in Podolak et al. (2015).

For these combined reasons, issues related to scale compound the uncertainties associated with estimating potential effects on water yield from forest removal that stem from the variability in responses from studies in small experimental watersheds. As a result, it is likely

not possible to accurately forecast the magnitude of water yield that might accrue from extensive forest removal at the scale of larger watersheds that provide water supplies. This renders forecasts of potential fiscal benefits associated with potential increases in water yields from forest removal highly uncertain and prone to considerable error.

2.11 Water yield response to forest removal is likely to be undetectable at larger scales

Assessments of attempts to increase water yield on public lands have consistently noted that it is very unlikely that any potential changes in water yield would be measurable at the scale of larger watersheds (Ziemer, 1986; MacDonald and Stednick, 2003; NRC, 2008). This likely holds for the water supply watersheds of Sierra Nevada, as well.

For instance, the highest estimate of increased water yield from forest removal in a major watershed is 6% in Podolak et al. (2015) (Table1). The resolution of streamflow measurements is commonly cited as plus or minus 10% (Ziemer, 1986; Grant et al., 2008). Thus, it is highly likely that even if the maximum forecast of increased water yield in Podolak et al. (2015), which considerably overestimates water yield response, were realized, these changes in water yield would not be measurable.

Models are not a surrogate for measurement. The accuracy of forest hydrology models is limited, especially at larger scales (NAS, 2008). Grayson et al. (1993) suggested that there are formidable obstacles to accurate prediction of changes in runoff caused by disturbance even at relatively small scales, because even physically-based hydrologic models have a limited physical basis due to the limited understanding of actual watershed functions, aggravated by the difficulty of collecting adequate data and the lack of methods for collecting and analyzing data at physically-meaningful scales.

The inability to reasonably detect the causes of changes in runoff at larger scales compounds the uncertainty caused by the lack of measurability. As Ziemer (1987) noted, “The technical problem of documenting or proving ... that water yield from any parcel of land has actually been increased is overwhelming... By the time the increased flows combine with unmeasured flows from untreated watersheds, there is virtually no chance of observing or proving that any increase occurred.”

Measurability has practical ramifications. If increased use is justified on the basis of unverified, but inaccurate estimates of increased water yield, it risks exacerbation of water supply problems and conflicts among competing water uses.

The likely immeasurability of water yields does not mean that water yields may not increase. However, it does make it impossible to verify changes in water yield from forest removal or verify that any change in water yield at the scale of water supply watersheds is due to forest removal. Absent verifiability, there is no way to reliably determine if investments in such an approach might yield any returns.

2.12 Summary: Forest removal effects on water yield, low flows, and peakflows

It is not possible to accurately predict the magnitude of changes in water yield in response to forest removal, especially at the scale of larger watersheds, due to scale-related uncertainties and considerable variability in data from numerous studies in small, paired watersheds. Nevertheless, as a gross estimate, there might be transient increases in water yield in small watersheds of about 13mm per 10% forest removed, although this estimate also has associated uncertainty, due to scale issues, storage and transmission losses, and variability associated with precipitation. This estimate is also unlikely applicable to larger watersheds, due to several confounding scale-related factors.

These problems make it impossible to reliably forecast water yield returns from investment in forest removal, which include the associated environmental costs of the latter. As past assessments have consistently noted, it is very likely that any potential changes in water yield will not be measurable at the scale of water supply watersheds. Hence, it will not possible to verify any firm return in water supply from forest removal, even if it extensively applied.

Available information indicates that less than 20-25% forest removal in watersheds is unlikely to increase water yields. Similarly, forest removal in watersheds that receive less than *ca.* 460-500 mm/yr of precipitation is unlikely to increase water yields.

The utility for water supply of efforts to augment water yield via forest removal is further limited by the timing and transience of water yield response, and the effect of precipitation on the magnitude of water yields. Increases in water yields in response to forest removal are transient and decline over time, other factors remaining equal. Thus, they are not self-sustaining. Due to the transience of forest removal effects on water yield, vegetation must be continually removed over large areas of watersheds at relatively short intervals in order to maintain increases in water yield.

Increases in water yields are lowest and least likely during drier years. Further, when they occur, the bulk of increased water yields occur during the wettest period of the year. Forest removal also increases peak flows. These increases in peak flow and wet-season water yield are also largest during the wettest years. The increases in streamflows during the peak runoff period in wetter years may have marginal or no benefits for downstream water uses. These increases during wetter periods may incur significant societal costs by increasing flooding and flood damage.

Forest removal is unlikely to significantly increase low flows or extend snowmelt runoff. Transient increases in low flows are especially unlikely in drier years. Forest removal also ultimately results in reductions in low flows as vegetation recovers. Therefore, forest removal is highly unlikely to provide additional water during the period of lowest runoff and highest demand driest times of the year, and especially during the driest years, when it could likely provide the greatest benefits. It is highly unlikely that forest removal could substantially augment water supplies during periods of severe drought.

Any potential increase in water yield will be subject to transmission losses, and, if stored in reservoirs, storage losses, prior to availability for use. Transmission losses are likely greatest

during the low flow period in the driest years. To be usable during periods of relatively high demand and low water supply, the majority of the water from potential water yield increases would need to be stored in reservoirs for several months during the summer, which likely has the highest rate of storage losses. These aspects of the approach also considerably limit the utility of attempts to augment water yield via forest removal.

The foregoing is consistent with prior assessments of the potential for augmenting water supplies via forest removal, which have repeatedly concluded that it is not promising (Ziemer, 1986; Sedell et al., 2000; NRC, 2008). Ziemer (1986) stated, “There is every indication that management of vegetation for increased water yield will continue to be impractical.” As Sedell et al., (2000) noted, “Producing substantial and extensive increases in water yields from the national forests does not appear to be practical...Legal constraints, land allocations, technological limits, as well as societal values and environmental, ecological, and biological concerns all favor not committing national forest lands to the management regimes that would be needed to increase water yields.” The latter part of the assessment of Sedell et al. (2000) is partially based on the substantial and enduring environmental costs of forest removal, which are discussed in the following sections.

3. The Enduring Environmental Costs of Attempts to Increase Water Yield via Forest Removal

As is the case with any extensive disturbance in watersheds, forest removal at levels (>20%) sufficient to potentially increase water yield would incur a range of environmental costs, many of which are enduring. Forest vegetation removal aimed at increasing water yields and/or attempting to alter fire behavior would also need to be repeated with some frequency to meet these aims. The effects of forest removal treatments often do not completely subside before the effects of subsequent treatments are superimposed on watershed systems, resulting in increased chronic cumulative impacts (Ziemer et al., 1991; Reid, 1993) that deleteriously affect aquatic communities. Thus, repeated entries to remove vegetation at a scale sufficient to increase water yields are likely to cause watershed cumulative effects that increase over time. For instance, if 25% of a watershed is subjected to repeated fuel treatments every 10 years for 40 years, this equates to a level of disturbance that is akin to treating 100% of a watershed over 40 years. This level of disturbance is generally acknowledged to cause significant adverse cumulative effects on watershed and aquatic resources over time (e.g. Ziemer et al., 1991; Murphy, 1995). Thus, the environmental impacts of repeated forest removal treatments are likely to be cumulative due to the frequency of forest removal treatments, their spatial extent, and the duration of adverse effects.

Forest removal, including that aimed at modifying fire behavior, inexorably involves roads, road activities (Reid, 2010; Robichaud et al., 2010) and often involves skid trails associated with ground-based forest removal. Roads and transportation-related activities negatively affect a host of aquatic and watershed conditions and processes in a persistent fashion (Trombulak and Frissell, 2000; Gucinski et al, 2001). Skid trails compact soils and contribute to elevated erosion, sediment delivery, and runoff (NRC, 2008; Reid, 2010). Therefore, the effects of road activities and skid trails associated with forest removal are included in the

following assessment of the environmental costs of attempting to increase water yields via forest removal on public lands in the Sierra Nevada.

3.1 The frequency and extent of forest removal necessary to potentially maintain elevated water yields: A context for evaluating cumulative effects and environmental costs

Because forest removal has transient effects on water yield that decline with time, increased water yield cannot be maintained without repeated removal of forest vegetation at levels sufficient to increase water yields. This approach would also require periodic additional watershed entries for re-treatment to kill vegetation in areas where it has been previously removed (Bales et al., 2011; Podolak et al., 2015) or additional removal of forests. Due to regrowth of vegetation, periodic re-treatment or additional forest removal is also required to maintain fuel reductions from initial forest removal (Finney et al., 2007; Rhodes and Baker, 2008). Forest removal that opens forest canopies, such as significant thinning, creates a self-perpetuating need for repeated treatment because they stimulate vegetation regrowth (Noss et al., 2006; Baker, 2009).

Recent assessments of water yield effects from forest removal have estimated that increases in water yields become insignificant *ca.* 6-7 years after forest removal (Robles et al., 2014; Podolak et al., 2015). Based on the foregoing, the following assessments of associated environmental costs are based on the estimate that at least 25% of watershed area will undergo forest removal involving mechanical forest removal or prescribed burning that would occur at a frequency of *ca.* 10 years in order to initiate or maintain increases in water yields. This is within the range of the estimated 7-20 year frequency of vegetation removal necessary to maintain reduced fuel levels in treatments aimed at modifying fire behavior (Finney et al., 2007; Rhodes and Baker, 2008; Robichaud et al., 2010).

3.2 Peakflows, flooding, and flood damage

As previously discussed, available information indicates that forest removal at levels sufficient to increase annual water yield will increase peakflows and other high flows from affected watersheds during periods of higher runoff. These flow increases tend to be most pronounced during the wettest years, when downstream flooding is also most likely. Due to the magnitude and timing of these effects on seasonally high flows, forest removal is likely to contribute to increases in downstream flooding and associated flood damage when these elevated high flows coincide with downstream flooding. Burton (1997) noted that the increases in peakflows from forest removal in his study could increase downstream flooding and flood damage. It has long been recognized that flood protection is maximized by a forest management approach that retains all forest canopy (Anderson and West, 1966; Brown et al., 1974; Anderson et al., 1976).

The absolute increases in peakflows from forest removal are greatest during the wettest years, although the relative (e.g., percent) increase in larger peakflows may be smaller than for lower peakflows (NRC, 2008). However, the absolute increases in larger peakflows can cause physically significant downstream effects due to the streamflow volume involved. NRC

(2008) noted, regarding forest removal effects on larger peakflows with longer recurrence intervals, "...small percentage increases in very large floods as a result of forest harvest may be quite large in absolute terms; a 10% increase in a 10-year flood is much more water than a 50% increase in a 1-yr flood."

Transmission losses are unlikely to significantly abate high flows from affected watersheds in wet seasons. During high runoff events, stream channels in forested watersheds often gain, rather than lose, water in the downstream direction. Evapotranspiration rates by riparian vegetation are relatively low during the peak snowmelt period. Thus, in most cases, the elevated high flows caused by forest removal are likely translated downstream without substantial attenuation via transmission losses. Where there are large-pool reservoirs managed specifically for flood control, smaller and intermediate magnitude flood flows might be moderated downstream, but the largest flood flows can overwhelm the ability of such reservoirs to significantly moderate flooding.

The effects of forest removal on high flows and downstream flooding would likely be greatest in areas immediately downstream of watersheds with significant forest removal, and increase with increasing levels of watershed-level forest removal. While the effect of increased high flows on downstream flooding is likely to be relatively incremental, and may not be measurable, it may still be physically significant, because relatively small changes in flood magnitudes still trigger increased flooding and associated damage (e.g., area flooded and duration of inundation can increase disproportionately relative to incremental flow increase).

Although forest removal effects on higher flows decline with time, repeated forest removal or re-treatment of areas subject to prior forest removal can largely maintain increases in high flows, including peak events, especially during the wettest years, which is when downstream flooding is most likely. As previously discussed, this is because the removal of forest vegetation on a scale sufficient to increase water yields is also sufficient to elevate high flows, especially during the wettest years. Roads and compacted soils on skid trails associated with forest removal also contribute to increased high flows (La Marche and Lettenmaier, 2001; NRC, 2008), but in ways that do not strongly decline with time (e.g., rates of hydrologic recovery on roads are nominal in the absence of major rehabilitation efforts). Repeated forest removal treatments at intervals of 10-15 years over extensive portions of watersheds will require a permanent, relatively high-density road network.

For these reasons, extensive forest removal and re-treatment or repeated forest removal will likely contribute to increased flooding at the scale of affected watersheds. These effects may be relatively small, but not discountable, at the scale of larger watersheds.

3.3 Forest removal increases erosion and sedimentation, degrading water quality and aquatic habitats

3.3.1 Mechanical forest removal requires an extensive network of roads and landings

Logging and associated activities significantly increase erosion and sedimentation, as documented by a legion of studies. Both Megahan et al. (1992) and USFS and USBLM (1997c) concluded that it is not possible to conduct logging activities without adding sediment to streams.

Roads and related activities are an integral aspect of extensive forest removal treatments (Kattelmann, 1987; Robichaud et al., 2010; Reid et al., 2010). Roads are one of the most significant and enduring causes of watershed and aquatic resource degradation on public lands, especially when located within riparian zones (USFS et al., 1993; Rhodes et al., 1994; CWWR, 1996; Espinosa et al., 1997; USFS and USBLM, 1997a; Trombulak and Frissell, 2000; USFS, 2000b). Roads are one of the primary causes of the severe reduction in the abundance and range of trout and aquatic species in the western US (USFS and USBLM, 1997a; Kessler et al., 2001).

Logging landings are typically associated with mechanical forest removal that includes the hauling of wood products for off-site commercial purposes. Landings have soil and vegetation impacts that are similar to those of roads, and, thus, erosion, in terms of longevity and severity (Geppert et al., 1984; Beschta et al., 2004). Ketcheson and Megahan (1996) found that the single largest sediment plume in a study of sediment transport in national forests with granitic soils in Idaho came from a landing. Cumulative effects assessment methods used on USFS lands in the Sierra Nevada indicate that landings contribute to adverse watershed cumulative effects as persistently and significantly as roads (Menning et al., 1996).

The area affected by landings as part of mechanical forest removal is far from trivial if forest removal is implemented at a scale that might increase water yields in the Sierra Nevada. Landings typically occupy *ca.* 1-2% of the area affected by forest removal with an aim of the sale of the removed wood products. Assuming landings only occupy 1% of treated areas and the levels of fuel reduction proposed for the high end of forest removal scenarios examined by Podolak et al. (2015) of 694,593 acres on USFS lands in the Feather River watershed, the likely level of landing area would be almost 7,000 acres. This area is equivalent to *ca.* 2,865 miles of road with an average width of 20 feet. Because landings have impacts on erosion that are akin to those of roads on a per unit area, the erosional impacts of these landings would be similar to those from 2,865 miles of road with an average width of 20 feet. If only half of this landing area associated with this level of forest removal in the Feather River required landing construction, the erosional impacts of these landings would be akin to the construction of *ca.* 1,430 mi. of road. Plainly, landings associated with forest removal are likely to significantly add to the elevation of erosion and sediment delivery caused by roads and forest removal.

Kattelmann (1987) noted that extensive and dispersed forest removal associated with efforts to increase water yield from forests in the Sierra Nevada would require the existence and use of an extensive road network in affected watersheds. Forest removal projects at least partially premised on fuel reduction aims typically involve the construction or reconstruction of roads and landings (e.g., Eldorado National Forest (ENF), 2004a; b; c; Plumas National Forest (PNF), 2012; Lassen National Forest (LNF), 2012; ENF, 2013). Forest removal also increases road use and requires increased road maintenance activities to facilitate this use

(Robichaud et al., 2010; Reid et al., 2010). All of these activities related to roads and landings, significantly increase erosion and sediment delivery to streams.

Roads vastly increase erosion, soil loss, and stream sedimentation for as long as they exist. But, increases in surface erosion on roads are usually greatest during construction and the first year after, and remain greatly elevated over natural levels for the life of the road (Furniss et al., 1991; King, 1993; Kattelman, 1996; USFS, 2000b).

Even after costly decommissioning or obliteration of roads, reductions in erosion can be slow to accrue because infiltration capacity and vegetation recover slowly (Foltz et al., 2007). Thus, erosion remains significantly elevated for a considerable period of time, even after decommissioning or obliteration (Potyondy et al. 1991; Beschta et al., 2004). Kolka and Smidt (2004) documented that runoff and sediment production from recently obliterated roads was vastly higher than on undisturbed soils, indicating that road obliteration did not rapidly eliminate elevated sediment production from roads. Foltz (2007) noted that it is uncertain if hydrologic conditions that influence erosion ever fully recover, even after road obliteration. For these reasons, it is unlikely that road obliteration or decommissioning of a limited portion of extant roads in a watershed can offset the erosional impacts of significant levels of new road construction, reconstruction, and extensive road use in a near-term timeframe, as noted by Beschta et al. (2004). Nevertheless, this is the tradeoff often proposed in forest thinning and fuels reduction projects on public lands (e.g., PNF, 2012; LNF, 2012; ENF, 2013).

3.3.2 Erosion and sedimentation from roads, landings, and related activities

Forest removal increases road use (Reid, 2010; Robichaud et al., 2010), which significantly elevates surface erosion on unpaved roads (Reid et al., 1981; Reid and Dunne, 1984, Gucinski et al., 2001). 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. Foltz (1996) documented that elevated truck traffic on roads surfaced with crushed rock aggregate with the intent to reduce sediment production, 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 in other regions.

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 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."

Road reconstruction, which often accompanies forest removal, significantly increases surface erosion on roads for several years, especially when it occurs on roads that have been largely unused and have undergone some soil and vegetation recovery. Such reconstruction also reverses whatever hydrologic and erosional recovery might have occurred prior to reconstruction (Beschta et al., 2004).

Road maintenance to facilitate haul is a common component of forest removal projects aimed at reducing fuel levels (e.g., ENF, 2000a; b; c; PNF, 2012; LNF, 2012; ENF, 2013). Road maintenance activities, including ditch maintenance and grading, increase sediment production by removing vegetation and disturbing road surfaces (Black and Luce, 1999; Luce and Black, 2001; Coe, 2006; Robichaud et al., 2010). In the Sierra Nevada, grading greatly increased sediment production from roads for several years (Coe, 2006).

In steep mountainous terrain, roads increase the frequency of landslides, debris flows, and other types of mass soil erosion (Furniss et al. 1991, Gucinski et al., 2001). Even ridgetop roads contribute to gullying and downslope instability (Montgomery, 1994).

Much of the erosion from roads is efficiently delivered to streams due to direct hydrologic connection via ditches, drainage, gullies below drainage relief features, and other points of stream-road connectivity (Wemple et al., 1996; Robichaud et al., 2010). Roads commonly drain directly to streams at road crossings (Kattelman, 1996; Rieman et al., 2003), which are frequent on most road systems. This is the case for USFS lands in the Sierra Nevada, which are estimated to have *ca.* 95,983 road crossings of streams (USFS, 2000a). Based on road mileage on these lands (USFS, 2000a), there are about 3.8 crossings per mile of road or about one crossing per *ca.* 0.26 miles of road, on average. These crossings and portions of roads that are connected to streams at these crossings are so numerous and widespread that they will inevitably increase sediment delivery to streams from elevated road use and maintenance associated with forest removal activities. Efforts to disconnect roads near streams are often ineffective, because as a thorough review of road BMP effectiveness (Great Lakes Environmental Center, 2008) concluded, location often trumps technique with respect to roads near streams.

Besides crossings, roads typically have other points of connectivity to streams, *where* efficient delivery of sediment occurs. While Coe (2006) found that the majority (*ca.* 59%) of stream-road connected segments sampled in and near the Eldorado National Forest in the Sierra Nevada were associated with stream crossings, a considerable fraction (35%) of the road segments were connected by hillslope gullies. These gullies also acted as additional sources of sediment delivered to streams, which was of comparable magnitude to that delivered from the road segments (Coe, 2006). About 25% of the lengths of road segments examined were connected to streams (Coe, 2006). Notably, data on road-stream connectivity as a function of precipitation (MacDonald and Coe, 2007) indicates that this connectivity is likely higher in areas of the Sierra Nevada with higher levels of mean annual precipitation than in the study of Coe (2006). This, again, indicates that a considerable amount of sediment generated from roads on USFS lands in the Sierra Nevada caused by forest removal activities is likely to be efficiently delivered to streams.

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



Photo 5. Extensive soil disturbance associated with log skidding, yarding and landings associated with forest removal activities with mechanized equipment. This type of soil damage elevates erosion and cumulatively contributes to increased sediment delivery, degrading aquatic habitats and water quality. Sierra National Forest. Photo by C. Hanson.

The repeated and extensive forest removal that would be required to maintain elevated water yields would also require the existence of an extensive road network. The continued presence of an extensive road network would work against reductions in road density, which has consistently been noted as a widespread and pressing aquatic restoration need in the Sierra Nevada.

The several-fold increases in erosion and sediment delivery caused by road and landing construction, reconstruction, maintenance and elevated use associated with frequent and extensive forest removal 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). Further, these sediment inputs will combine with increased sediment delivery from other extensive land-disturbing activities, such as livestock grazing, on public lands.

3.4 Riparian management requirements are inadequate protect streams from damage from forest removal practices

It cannot be assumed that the Riparian Habitat Conservation Areas (RHCA) established for streams under the current land management direction for USFS lands in the Sierra Nevada

(USFS, 2004) are adequate to reduce aquatic degradation, including that from increased sediment delivery, from forest removal and associated landing and road activities to negligible levels, for four primary reasons. First, a considerable amount of the road network, which would be affected by forest removal activities, is connected to the stream network, as previously discussed. In such situations, sediment is delivered directly to streams. Portions of RHCAs that are upslope of roads near streams have no effect on arresting sediment generated at such locations.

Second, riparian areas on USFS lands in the Sierra Nevada are widely degraded by roads, past logging, and past and on-going grazing. The 1996 assessment of conditions in the Sierra Nevada (CWWR, 1996) concluded that riparian areas are the most altered and impaired systems in the Sierra Nevada. This degradation reduces the ability of RHCAs to arrest and capture sediment from upslope disturbances.

Third, recent forest removal projects aimed at fuel reductions indicate that a physically-significant amount of forest removal typically occurs within RHCA widths (ENF, 2004a: b; c; PNF, 2012; LNF, 2012; ENF, 2013). Under such conditions, the full width of RHCAs is not available to arrest sediment delivery from such activities. Instead, sediment from forest removal has a considerably shorter pathway to streams than the entire width of RHCAs. Decreases in the distance between land disturbances and streams generally increase the likelihood and efficiency of sediment delivery from the disturbance to streams (USFS et al., 1993; Rhodes et al., 1994; Kattelman, 1996; USFS and USBLM, 1997a).

Forest removal within RHCAs also removes vegetation and ultimately reduces downed wood in streams and riparian areas (Pollock and Beechie, 2014) in the RHCAs. These effects degrade aquatic systems and cause long-term reduction in the ability of RHCAs to detain and store sediment supplied from upslope sources (Rhodes et al., 1994).

Fourth, the default RHCA widths of 150 feet from each side of the non-perennial streams under USFS (2004) are inadequate to consistently prevent sediment delivery from upstream disturbances. USFS et al. (1993), USFS and USBLM (1995a; b) indicate that a protected area of undisturbed vegetation 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. O'Laughlin and Belt's (1995) evaluation of available data arrived at a similar conclusion, but noted that channelized sediment flows can transport sediment from anthropogenic sources for thousands of feet (O'Laughlin and Belt, 1995). Channelized sediment delivery from roads is not uncommon and can occur in response to significant land disturbance, especially when runoff has been increased or concentrated (Wemple, 1996).

USFS and USBLM (1997a) estimated that the width of riparian reserves protection needed to reduce the risk of damage to non-perennial stream channels from elevated sedimentation from upslope impacts was a function of slope adjacent to the stream channel. Based on an analysis of sediment movement data, USFS and USBLM (1997a) estimated that a distance of about 550 feet from each side of an ephemeral stream was needed to reduce the risk of accelerated sediment delivery to intermittent streams from upslope sources where a 50% slope abutted an ephemeral channel. Notably, USFS and USBLM (1997a) concluded that such a reserve width did not

ensure protection against increased sedimentation, but only reduced its probability. Based on similar considerations, Erman et al. (1996) came to similar conclusions regarding the riparian protection widths needed to protect intermittent streams with steep side slopes from degradation in the Sierra Nevada. Based on this combined information, riparian reserve widths of more than 300 ft are needed to fully protect against elevated sediment delivery to streams and consequent negative effects on aquatic resources.

The inadequate RHCA widths on non-perennial streams are significant due for several reasons. These streams comprise the majority of the stream network, are sensitive to degradation, convey the majority of sediment downstream to perennial streams, and cumulatively exert an extremely strong control on downstream aquatic conditions (USFS and USBLM, 1997a). Due to their frequency, position, and setting, non-perennial streams are most prone to be affected by upslope disturbances from extensive forest removal. Non-perennial headwater streams are typically flanked by slopes that are steeper than those flanking larger downstream perennial stream segments. These non-perennial streams also have smaller floodplains to buffer impacts from upslope increases in erosion and runoff (Rosgen, 1996). Channel types in headwaters are highly vulnerable to increased channel erosion in response to upslope increases in runoff (Rosgen, 1996; Montgomery and Buffington, 1998). Headwater streams and adjacent areas are often the areas most prone to slope instability within a watershed (USFS et al., 1993; May, 2002). Once degraded, many high-gradient headwater streams in have very poor prospects for recovery, even after the cause of degradation has been eliminated (Rosgen, 1996). Assessments have consistently noted that for these reasons, smaller headwater streams are critical to protect in order to protect downstream conditions (Rhodes et al., 1994; Moyle et al., 1996b; USFS and USBLM, 1997a). Impacts to headwater streams are translated downstream where they cumulatively affect fish habitat (Montgomery and Buffington, 1998). 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; Erman et al., 1996; USFS and USBLM, 1997a).

The extensive analysis of turbidity data from rivers in northern coastal California by Klein et al. (2012) indicates that riparian protections and other BMPs are likely inadequate to eliminate sediment impacts from forest removal. The analysis indicated that the amount of recent logging exerted a strong control on turbidity levels, which is an index of suspended sediment levels, despite modern logging-related BMPs (Klein et al., 2012). Turbidity levels in watersheds with relatively high levels of recent logging were consistently elevated relative to undisturbed areas and at levels that adversely affect salmonids and exceed water quality standards (Klein et al., 2012). While these results were for areas dominated by rainfall, rather than snowmelt, they strongly indicate that BMPs, such as RHCAs, do not eliminate sediment-related impacts of forest removal.



Photo 6. Present RHCA practices on USFS lands in the Sierra Nevada include an equipment exclusion zone that is too narrow to fully buffer streams from erosion and sediment generated by adjacent equipment operations and roads. Removing riparian vegetation and downed woody debris through thinning and fuels reduction compromises sediment retention functions of slopes and floodplains, further increasing sediment delivery to streams. This also degrades aquatic habitat via the loss of in-stream wood recruitment. Plumas National Forest. *Photo by Chris Frissell.*

For these combined reasons, it cannot be assumed that RHCAs can consistently reduce sediment impacts to negligible levels from forest removal extensive enough to increase water yields. Therefore, it is highly likely that forest removal of at least 25% of the area of affected watersheds repeated at 10 year cycles would significantly elevate sediment delivery and downstream sediment transport, affecting a host of aquatic resources and downstream uses.

Although some of the sediment-related impacts of road, landing, and forest removal abate with time, repeating forest removal activities every 10 years will maintain frequent elevation of sediment delivery to streams. This will cumulatively ratchet up stream sedimentation and sediment transport to downstream areas.

3.5 Erosion and sediment effects from forest removal via prescribed fire

Prescribed fire, which might be used to initiate or maintain forest removal, would also contribute to cumulatively elevated erosion and sediment delivery, although usually to a lesser degree than mechanical treatments and associated activities. The erosional impacts of prescribed fire largely depend on burn severity, other factors being equal. Erosion from prescribed fire increases typically with increased burn severity, because the soil cover loss increases with burn severity. Other factors being equal, bare soils erode at more than 100 times the rate on vegetated surfaces (Dunne and Leopold, 1978; Rhodes et al., 1994). Loss of soil cover also can increase surface runoff, which also increases surface erosion. In contrast, undisturbed soil cover on forest floor helps maintain relatively high infiltration rates, which limit the frequency and magnitude of surface runoff (Reid, 2010). Higher burn severities can sometimes also transiently reduce infiltration rates in some areas via the development of hydrophobic soil conditions, contributing to increased runoff and soil erosion.

Although prescribed fire burning at low severity, has nominal and fleeting erosional impacts, the forest removal to initiate or maintain water yield increases would require killing a sizable fraction of trees. In the Sierra Nevada, this appears to require prescribed fire burning at relatively high severity (van Mantgem et al., 2011). Prescribed fire would also need to be applied about every 10 years and be severe enough to kill enough vegetation to be equivalent to complete mortality over 25% of a watershed, to maintain or initiate potential increases in water yields during non-drought years. Importantly, such a prescribed fire regime would burn far more area at higher severity and much more frequently than wildfire currently does in the Sierra Nevada. Estimates of recent high-severity fire rotation⁶ associated with wildfire ranges from about 460 - 700 years in the Sierra Nevada, depending on forest type and geographic location (Hanson and Odion, 2014; Baker, 2015). In contrast, the application of prescribed fire needed to maintain the potential for increased water yields would result in a high-severity fire rotation of only *ca.* 40 years. This level of management-imposed high-severity fire would contribute to increased erosion and sediment delivery, due to its effects on bare ground, runoff, hydrophobic soil development, far in excess of that generated by wildfire alone.

Even when prescribed fire burns at low to moderate severity, it can significantly increase the area of bare soils. In Central Oregon, bare ground area nearly doubled after two cycles of low severity prescribed burns conducted at five year intervals (Hatten et al., 2012). Although some of these effects on soil cover were not statistically significant (Hatten et al., 2012), they are physically significant, because erosion rates are far higher on bare ground than soil with cover.

Prescribed fire conducted with the aim of burning at lower severity still often burns with a significant component of high-severity fire. In the Sierra Nevada, Knapp and Keeley (2006) found that burn severity from prescribed fire was patchy and variable. Robichaud (2000) documented that prescribed fire in Montana burned 5-15% of the treated area at high severity.

⁶ High-severity fire rotation is the expected time for high-severity fire to burn an area of interest, based on the rate of high-severity fire occurrence. Thus, at the scale of watersheds, burning at least 25% at high severity every 10 years would result in a high-severity fire rotation of only 40 years.

The burning created hydrophobic soils over about 28% of the burned area and reduced infiltration rates by about 10-40%. The hydrophobic soils temporarily increased runoff and soil erosion for one to two years (Robichaud, 2000). Reid (2010) noted that prescribed fire was likely to contribute towards increasing hydrophobicity in affected soils.



Photo 7. Prescribed fire verging on riparian area, Plumas National Forest. While prescribed fire elevates erosion, watershed damage is greatly multiplied if logging or mechanical fuels treatment is conducted prior to (or after) fire. Photo by Chris Frissell.

Prescribed burns occasionally burn hotter and over greater areas than intended. As the extent and frequency of burning increase, the likelihood of unintended increases in burn severity and area increases.

Thus, the frequency, severity, and scale of prescribed fire that would be needed to periodically initiate or maintain elevated water yields would add to the erosion and sediment delivery caused by mechanical forest removal and associated road and landing impacts from attempts to increase water yield via forest removal. The frequency and magnitude of these impacts from such a prescribed fire regime would likely be significantly greater than that from wildfire, because the former would involve burning at greater frequency, over more area, at a higher severity than occurs under the natural-historical wildfire regimes. Moreover, the erosional effects of wildfire and prescribed fire would be additive, because frequent

prescribed fire would not eliminate wildfire occurrence, as discussed in greater detail in a following section.

3.6 Effects of elevated peakflows from forest removal on channel erosion and downstream sediment transport

Forest removal at a scale sufficient to increase water yield will elevate channel erosion and consequent downstream sediment delivery by increasing the magnitude and frequency of peakflows, as previously discussed. Increases in channel erosion and downstream sediment delivery are inevitable with persistent increases in peakflows (Dunne and Leopold, 1978; Richards, 1982). Peakflow elevation by forest removal and roads is an important concern because even minor changes in peakflow magnitude and frequency can trigger significant changes in channel erosion and sediment transport (Dunne et al., 2001).

There is little dispute 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). Significant forest removal also increases the frequency of these flow magnitudes (Alila et al., 2009). Because flows in this range of recurrence intervals exert a strong influence on channel cross-sections in streams with erodible banks, increases in these flows and their frequency will increase bank and channel erosion, and consequent downstream sediment transport.

King (1989) concluded that the increased peakflows caused by forest removal in his study would likely modify channel form and increase sediment transport to downstream reaches, because the majority of bedload sediment is transported in headwater channels during the 7-8 days of highest flows, which were the flows most increased by forest removal in the study. Troendle and Olsen (1994) measured significant increases in erosion and sediment transport from watersheds in which forest removal had increased peakflows and water yield. This increase was primarily ascribed to increased channel erosion and transport from increased flows (Troendle and Olsen, 1994).

Studies have found that increased peakflow increases headward channel erosion and/or expansion of cross-sectional channel area, even when riparian vegetation has not been removed (Megahan and Bohn, 1989; Heede, 1991). Both impacts increase downstream sedimentation. Based on Heede's (1991) data, the increases in channel cross section, excluding headward channel erosion, caused by increased peakflows contributed approximately 550 yd³ of sediment downstream over eight years in ephemeral reaches that totaled less than 1.25 miles in length. These impacts cannot be viewed in isolation. Due to the scale of deforestation required to increase annual water yield, increases in channel erosion would occur at the scale of extensive channel networks, not just in a few, isolated, short reaches of ephemeral streams. At the scale of the channel network in larger watersheds, this magnitude of sediment delivery (440 yd³/mi of affected stream channel) would have significant long-term effects on downstream sedimentation and the resources affected by it. Further, these sediment sources would be combined with elevated sediment delivery caused by logging and associated impacts, from extensive and frequent forest removal.

It is well-established that increases in streamflows increase downstream sediment transport, including the total sediment yield delivered to downstream points. Therefore, the effects of forest removal and related activities on streamflows, even in the absence of other sediment impacts, will significantly increase downstream sediment yield and deposition, especially since these impacts will also increase sediment delivery to affected streams with considerable frequency under management aimed at maintaining increases in water yield.

3.7 Forest removal effects on soils and related watershed processes

Forest removal activities at the scale and frequency that could maintain elevated water yields would have numerous adverse impacts on soils, most of which would be extremely persistent. Notably, many of these impacts would contribute to cumulative losses in the ability of watersheds to absorb, store, and release water, which is likely to exacerbate the hydrologic impacts of climate change on watershed conditions and processes.

Mechanical ground-based forest removal and the construction of landings and roads would elevate soil compaction. It is extremely well-documented that forest removal using ground-based machinery inevitably compacts soils (e.g., Ampoorter et al., 2012; Busse et al., 2014). Although a wide variety of soil conditions affect the degree of compaction from ground-based forest removal, it tends to increase with soil moisture and clay content, other factors remaining equal. Landings and roads severely compact soils.

Soil compaction has numerous adverse hydrologic impacts. Soil compaction decreases infiltration rates and reduces soil water storage capacity, both of which increase surface runoff. Much of the surface runoff in western forests is the result of exceedance of soil water storage capacity. Hence, the reduced water storage capacity in the soil profile increases the magnitude extent, and duration of surface runoff. Increased surface runoff contributes to elevated peakflows and surface erosion.

The loss of soil water storage capacity also reduces the ability of watersheds to absorb, store, and release water. This capacity sets the upper limit on the amount of water from snowmelt and rain that soils can absorb. Thus, soil compaction contributes to reductions in the amount of water that soils store at the watershed scale. This is a significant ecosystem impact because water that is stored in soils provides the water available for vegetation and is the primary source of late-season streamflows (Kirkby, 1978). Thus, compaction-related loss of water storage capacity in soils likely contributes to reductions in late season low flows and water available for vegetation after snowmelt. Both are concerns due to the effects of climate-change (Beschta et al., 2013).

The effects on soil moisture storage capacity from soil compaction caused by forest removal activities are not trivial. In a 25 km² watershed subject to ground-based forest removal on 25% of the watershed area, soil compaction would reduce available water storage in soils by about 15,000,000 liters, based on: a) the mean soil compaction levels found in Ampoorter et al. (2012); b) the compaction of 15% of the area subjected to forest removal; and, c) one percent of forest removal area occupied by in landings, compacted to the degree found by Rab (2004). This indicates that levels of forest removal sufficient to affect water yield would

result in significant reductions in watershed-scale water holding capacity which would affect both low flows and water availability for vegetation.

Soil compaction also reduces soil productivity (USFS and USBLM, 1997a; Beschta et al., 2004; Ampoorter et al., 2012). However, compaction may not universally reduce soil productivity, depending on soil type, texture, and drainage (Busse et al., 2014).

These impacts on soil compaction are significant due to their duration. Compaction, and associated effects, typically persists for 40-80 years in the absence of additional impacts before there is complete recovery (USFS and USBLM 1997a; Beschta et al., 2004). Compaction on roads and landings persists for even longer due to the level of compaction and, in the case of roads, continued impacts. Repeated, frequent, and extensive entries to remove forests, as would be needed to maintain the potential for water yield increases in wetter years, would cumulatively increase the extent of soil compaction and its adverse impacts, due to the persistence of compaction.

Road obliteration does not rapidly restore or eliminate the persistence of compaction-related soil impacts from roads and landings. Foltz et al. (2007) found the recovery of infiltration rates was nominal on roads four years after obliteration. Kolka and Smidt (2004) documented that runoff and sediment production from recently obliterated roads remained vastly greater than from undisturbed soils, indicating that infiltration rates on the obliterated roads remained low. Foltz et al. (2007) noted that the amount of time needed for full recovery of hydrologic properties to occur on roads is unknown. Hence, it is not certain that full recovery of hydrologic soil properties is ever reached on obliterated or abandoned roads (Foltz et al., 2007).

Roads and landings associated with forest removal essentially zero out soil productivity for some time and reduce it for long periods thereafter (Geppert et al., 1984; Beschta et al., 2004). 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 on soil processes (Beschta et al., 2004). The findings of Foltz et al. (2007) corroborate that the soil processes critical to soil productivity may not ever be fully restored, even with attempted remediation, on roads and landings.

Soil compaction will be cumulatively elevated by the repeated removal of vegetation at extensive scales, due to the persistence of compaction impacts and the frequency and scale of continuing periodic treatment required for attempts at periodic boosting of water yields. This will lead to increasing cumulative long-term impacts on watershed hydrology. Soil compaction and its impacts have been identified as a major concern over large areas in the Sierra Nevada (CWWR, 1996).

The scale and frequency of forest removal that would be needed to increase water yields would also degrade soils by greatly increasing soil erosion in an extensive fashion. Topsoil loss causes serious and enduring reductions in soil productivity (USFS and USBLM, 1997a; Beschta et al., 2004). The loss of topsoil is irreversible and associated reductions in soil productivity are essentially permanent (Beschta et al., 2004; Karr et al., 2004).

Loss of topsoil reduces the ability of soils to infiltrate and store water. The upper layers of soils typically have the highest infiltration rates and water storage capacity. Thus, topsoil loss can contribute to reductions in infiltration and water storage. Because topsoil typically has relatively high water storage capacity, the loss of topsoil and associated thinning of the soil profile reduces the water storage capacity in soils.

Forest removal can also contribute to long-term degradation of soils by removing vegetation and forest floor litter that are the primary source of organic matter in soils (USFS and USBLM, 1997a). Repeated prescribed fire has been found to reduce the thickness of the organic-rich layer (O horizon) at the forest soil surface (Hatten et al., 2012).

Losses of organic matter in soils are of concern for several reasons. Organic matter is a critical element of soil productivity. It strongly influences hydrologic soil properties and processes. It is well-established that organic matter contributes to infiltration and the ability of soils to store water (Bodner et al., 2015). Loss of organic matter in forest systems has persistent effects that cannot be rapidly reversed, because they depend on the decay of plant material, which is relatively slow (USFS and USBLM, 1997a).

The prevention of soil damage and loss of productivity is far more tractable and effective than attempts to restore productivity (Kattelman, 1996; USFS and USBLM, 1997a). One of the primary approaches to restoring soil productivity is to restore organic matter and coarse woody debris levels, accomplished by leaving areas undisturbed until organic matter levels have recovered, and controlling surface erosion (Kattelman, 1996, USFS and USBLM 1997a). The extensive and frequent forest removal necessary to affect water yield conflicts with the prevention of soil damage and the restoration of degraded soils.

3.8 Forest removal and wildfire interactions

Wildfire affects a wide array of watershed conditions and processes that are affected by extensive and frequent forest removal, such as water yield, peakflows, erosion, and soil conditions. Thus, wildfire and forest removal have interacting, combined effects on watershed conditions and processes.

3.8.1 *Wildfire effects on watersheds and aquatic systems*

As is the case with prescribed fire, wildfire effects on watershed processes generally vary with fire severity, especially that at the soil surface. High-severity fire can have pronounced, but transient, impacts on watersheds, especially with respect to soils, runoff and sediment delivery (Minshall et al., 1997; Gresswell, 1999; Robichaud et al. 2000; 2003; 2010; Beschta et al., 2004). In contrast, fires that burn at lower severity have more muted and transient impacts on watershed processes.

The majority of the area annually burned in the western Sierra Nevada burns at low to moderate severity (Hanson and Odion, 2014; 2015). High-severity fire comprises the smallest fraction of area burned annually in the western Sierra Nevada, even in large fires burning during extreme fire weather (Rhodes, 2007; Hanson and Odion, 2014; 2015; Harris et al., 2015). For instance, in the 2013 Rim Fire in the Sierra Nevada, which burned during

relatively extreme fire weather, 63% of area burned at low-moderate severity, with 13% unburned within the fire perimeter, and 20% at high severity (MTBS 2014).⁷

Importantly, some forests have natural fire regimes that include a significant component of high-severity fire. Many common wetter and colder forest types, including subalpine and maritime forests, burn infrequently and usually at high severity (Schoennagel et al., 2004). Efforts to reduce fuel levels in such areas are unlikely to reduce fire severity or restore natural fire regimes (Schoennagel et al., 2004; Noss et al., 2006).

Recent assessments have provided confluent evidence that high-severity fire naturally occurred with considerable frequency in most Sierra Nevada forests (Baker, 2014; Odion et al., 2014; Baker 2015). Analyses of recent fire data and historical estimates have indicated that fire severity has not increased on forested lands in the Sierra Nevada (Hanson and Odion, 2014; 2015; Baker, 2015). While Safford et al., (2015) presented analyses indicating that an increased proportion of area burned is burning at higher severity on public lands in the Sierra Nevada, this finding has been refuted by the data and analyses of Hanson and Odion (2015) and Baker (2015).

Extreme increases in erosion, sediment delivery, and runoff in response to high-severity fire are far from a certainty. Fires that have burned large areas at high-severity have not triggered extreme levels of postfire erosion and runoff, including the 1988 Yellowstone fire (Minshall et al., 1989; 1997) and the Biscuit fire in Oregon (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. Severe postfire erosion and runoff appear to be largely relegated to the Southwest and Intermountain West.

While hydrophobic soils reduce infiltration rates in forest soils, they do not do so to a degree that causes elevated surface runoff from all snowmelt and rainfall events. Hydrophobicity, when it occurs, transiently reduces naturally high infiltration rates in forests 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 interactions also likely explain why such events are not triggered by snowmelt, because it progressively wets soils and is of relatively low intensity (Wondzell and King, 2003).

An important context is that the reduction in infiltration rates on hydrophobic soils is far less than that caused by livestock grazing, roads, and landings. In contrast to the transient effects of hydrophobic soils on infiltration, soil compaction and other impacts of grazing and roads persistently reduce infiltration rates by about 85% and 95-99%, respectively (Figure 5). These compaction-related infiltration effects are persistent, unlike the quite transient, patchy, and inconsistent effects of fire on hydrophobicity and infiltration.

⁷ Harris et al. (2015) used different, lower resolution methods and estimated 24% of the Rim Fire burned at high severity.



Photo 8. Rapid recovery of live vegetation approximately one year after this area burned at moderate-to--high severity on the Plumas National Forest in the Sierra Nevada. This rapid postfire revegetation quickly mutes postfire erosion and resulting aquatic effects. In the absence of the ecological insults from postfire logging, ground cover in this area will continue to increase due to revegetation and recruitment of needles and wood from burned trees. *Photo by J. Rhodes.*

Box 1. Effects of Wildfire on Watersheds

- High-severity fire increases erosion by reducing soil cover. The impacts of high severity fire on erosion and runoff can be exacerbated by the development of hydrophobic soils that increase surface runoff, sometimes dramatically (Wondzell and King, 2003; Robichaud et al., 2003). However, while areas burned at high severity can sometimes develop hydrophobic soils, this is not a consistent effect of high-severity fire (Beschta et al., 2004; Wondzell and King, 2003).
- The development of hydrophobic soils in response to fire depends on several factors besides fire severity, including vegetation, soil texture, and soil moisture conditions (Robichaud, 2000; Letey, 2001), all of which can vary considerably in burned terrain. Hydrophobic soils also occur in forested areas unaffected by recent fire (Doerr et al., 2009), which makes it difficult to determine if they have developed in response to fire. Doerr et al. (2009) noted that the notion that long unburnt conifer forest soils of the northwestern USA consistently have low levels of hydrophobicity "...is therefore incorrect." These findings also indicate that fire severity is not a strong predictor of the development of hydrophobic soils.
- Increased runoff from areas extensively burned at high severity can greatly increase fluvial erosion in stream channels (Wondzell and King, 2003; Robichaud et al., 2003). Tree mortality caused by high-severity fire can trigger mass failures due to the loss of root strength combined with increased soil saturation (Wondzell and King, 2005).
- Erosion and runoff triggered by high-severity fire declines over time, due to several mechanisms, including postfire revegetation, the recovery of soil properties and infiltration rates (Wondzell and King, 2003; Robichaud et al., 2010), and the recruitment of needles, branches, and other woody material from dead trees to that provides soil cover that reduces erosion (Pannkuk and Robichaud, 2003).
- Elevated surface erosion and runoff from wildfire typically persists for about three years or less, even in systems that have been extensively burned at high severity (Wondzell and King, 2003; Robichaud et al., 2010). Mass failures in response to fire may lag several years after fire (Wondzell and King, 2003). Natural rates of postfire recovery of groundcover are sometimes rapid (Rhodes, 2007), triggering rapid reductions in postfire surface erosion rates.



Photo 9. Rapid recovery of ground cover by vegetation and needle cast approximately one year after this area burned at high severity on the Eldorado National Forest in the Sierra Nevada. Measured ground cover in this area was ca. 87% at the time of the photo. Such rapid postfire recovery of ground cover quickly mutes postfire erosion. Ground cover in this area will continue to increase due to vegetative regrowth and recruitment of needles and wood from burned trees. *Photo by J. Rhodes.*

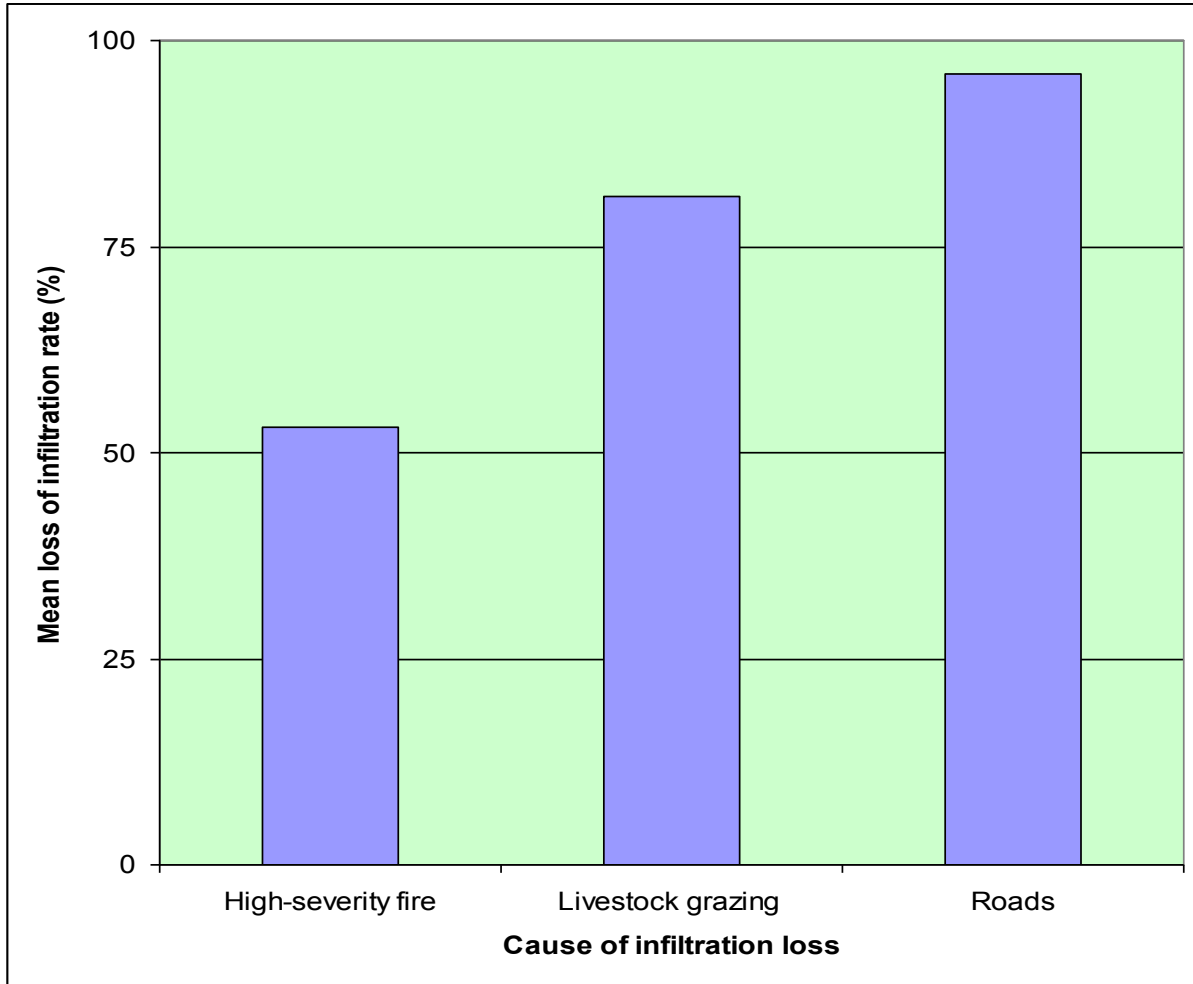


Figure 5. Measured mean reductions of infiltration rate due to fire in CO, NM, OR, and ID, USA (Wondzell and King, 2003); grazing in OR (Kauffman et al., 2004); and roads (Luce, 1997), relative to infiltration measured in comparable soils unaffected by these impacts. The losses in infiltration rates 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.

Box 2. Ecological Benefits of High-Severity Fire

Proponents of fuels reduction and logging frequently claim that removal of forest has the benefit of reducing the incidence of high-severity wildfire. This overlooks that high-severity wildfire is a natural and valuable component of the fire regime of Sierra Nevada forests (e.g., Beatty and Taylor 2001, Stephens and Collins 2004).

Scientists have identified a number of important ecosystem processes and properties that are shaped by and benefit from high-severity wildfire:

- While high-severity fire can reduce soil productivity by increasing erosion and consuming sources of soil organic matter that are essential to soil productivity, it 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). In the absence of postfire logging, 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 (Busse et al., 2014).
- High severity fire can provide significant benefits for aquatic systems (Jackson et al. 2015). It is an important agent for the pulsed recruitment of large woody debris (LWD) to streams, which provides numerous long-lasting benefits to streams and aquatic biota, including salmonids (Karr et al. 2004; Beschta et al., 2004; Ratliff et al., 2015). LWD is vital to stream complexity and pool development, which are critical for the production and survival of salmonids (USFS et al., 1993; Meehan, 1991; Rhodes et al., 1994) and many other aquatic species. Woody debris recruited to channels and floodplains also provides stream cover, sediment storage, and stream shade. As Karr et al. (2004) noted: "...there is no debate about the key role that large trees play in aquatic systems and many ecological processes..."
- The pulsed sediment supply from high severity fire may also benefit aquatic systems by rejuvenating certain habitats and fluvial processes (Rieman et al., 2003; Karr et al., 2004; Rhodes and Baker, 2008).
- Post-fire trophic pulses driven by light and nutrients can benefit stream ecosystem productivity and speed natural biological recovery from fire events (Jackson et al. 2015; Ratliff et al., 2015).
- High-severity fire creates and maintains forest heterogeneity which is critical to biodiversity (Odion and Hansen, 2006; Noss et al., 2006; Rhodes and Baker, 2008; DellaSalla and Hanson, 2015). The early seral habitats created by high severity fire provide numerous benefits to wildlife species (DellaSalla and Hanson, 2015, Hutto 2008).

3.8.2 *The limited effectiveness of forest removal to modify fire behavior*

One of the aims associated with extensive forest removal in the Sierra Nevada is to alter fire behavior by reducing fuel levels (Bales, 2011; Podolak et al., 2015). However, the effectiveness of forest removal to substantially reduce fire severity is significantly limited by three critical contexts.

First, natural fire regimes influence the likelihood that fuel treatments can affect fire severity and help restore natural fire regimes (Romme et al., 2003a; b; Schoennagel et al., 2004; Noss et al., 2006). Treatments that are not consistent with natural fire regimes are likely to damage forest ecosystems without yielding any ecological benefits from fire regime restoration (Veblen, 2003; Schoennagel et al., 2004; Kauffman, 2004; Noss et al., 2006).

There are some obstacles to accurately identifying natural fire regimes and potential departures from them (Veblen, 2003; Romme et al., 2003 a; b). Due to high temporal variability in natural fire behavior, accurate identification of natural fire regimes may require several centuries-worth of information on the fire extent, severity, and frequency (Romme et al., 2003a; b; Veblen, 2003; Baker, 2009). Nonetheless, for the sake of simplicity, this report follows the route taken by other researchers (Romme et al., 2003a; b; Schoennagel et al., 2004; Noss et al., 2006) of grouping forests and their fire regimes into three broad categories. These are:

1) *Forest types with natural fire regimes characterized by relatively infrequent, high-severity fires.* These forest types include subalpine forests comprised of spruce, subalpine fir, and lodgepole pine, forests in the wetter climates, and some woodlands (Romme et al., 2003a; b; Schoennagel et al., 2004; Noss et al., 2006). Hydric riparian and wetland forests in much of the West also likely have such a natural fire regime. Weather, rather than fuels, 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., 2004; Noss et al., 2006).

2) *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.* 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 species (Romme et al., 2003a; b; Schoennagel et al., 2004a; Noss et al., 2006), including many mixed-conifer forests of the Sierra Nevada (Odion and Hanson, 2006; Odion et al., 2014). 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., 2004; Noss et al., 2006). 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., 2004; Noss et al., 2006; Baker et al., 2006).

3) *Forest types with a fire regime primarily characterized by relatively frequent, low-severity fire.* This forest type appears to be relegated mostly to some of the relatively arid ponderosa pine forests of New Mexico and Arizona, though some other ponderosa pine systems may

also have this natural fire regime (Romme et al., 2003a; b; Schoennagel et al., 2004; Noss et al., 2006). There is evidence that high-severity fire also naturally occurred in forests with this fire regime (Odion et al, 2014). There is general agreement that fuel conditions usually exert a stronger control than weather on fire severity in forests with this natural fire regime (Schoennagel et al 2004; Noss et al., 2006).

Due to the relative effect of weather and fuels on fire severity, as well as fire frequency, fuel reductions are unlikely to reduce fire severity in wetter forests with a natural fire regime of infrequent, severe fires (Schoennagel et al., 2004). Fuel treatments in forests that have a natural forest regime of mixed severity and frequency of fire, have a somewhat greater potential to affect fire behavior, although this is mediated by both the effect of weather (Romme et al., 2003a; b) and fire frequency. Relatively dry forests characterized by relatively frequent, lower severity fire have the greatest potential for fuel treatments to reduce fire severity (Schoennagel et al., 2004), however, this is limited by the low probability of fire affecting treated areas while fuels are reduced (Rhodes and Baker, 2008; Moritz et al., 2014; Price et al., 2015; Boer et al., 2015).

The foregoing points to the major trade-offs between the efficacy of forest removal in increasing water yields versus that for affecting fire behavior. The potential for forest removal to increase water yields generally increases with precipitation. However, due to effects on fire frequency, fire regimes, and the relative influence of weather and fuels on fire behavior, the potential effectiveness of fuel removal to affect fire behavior likely declines with increasing precipitation. Fire behavior in wetter forests is more strongly controlled by weather than fuel levels, making it unlikely that fuel reduction treatments can effectively modify fire behavior. Additionally, fire is more infrequent in wetter forests, making it more unlikely that fuel treatments encounter fire while fuels are reduced. Thus, forest removal in areas with the greatest potential to increase water yields has a very low likelihood of reducing fire severity or restoring natural fire regimes. Forest removal in areas with the greatest potential to affect fire severity has the lowest prospects to increase water yield.

A second overarching control on the potential effectiveness of fuel treatments to alter fire behavior is the occurrence of fire and the transience of fuel reduction. Areas with fuel reduction cannot affect fire behavior if fire does not affect those areas during the temporal window of reduced fuels. Due to fraction of forested area burned annually,⁸ the probability of fire affecting areas where fuels have been transiently reduced is generally low on public lands in western forests, as numerous studies have repeatedly shown or noted (Rhodes and Baker, 2008; Law and Harmon, 2011; Campbell et al., 2011; Price et al., 2012; Restaino and Peterson, 2013; Moritz et al., 2014, Price et al., 2015; Meigs et al., 2015; Boer et al., 2015). In Sequoia Kings Canyon National Park in the Sierra Nevada, Price et al. (2015) found that the potential for areas with reduced fuels to reduce fire area was nil. This was partially due to

⁸ Estimated natural rates of burning cannot be used to estimate the probability that fire affects a given area over a given time frame, because current rates of the area burned by wildfire on public lands in the Sierra Nevada depart significantly from natural rates of burning. For instance, Ager et al. (2014) estimated that the area burned annually under current conditions on several national forests in the Sierra Nevada is vastly lower than estimated historical rates of annual area burned by wildfire. Thus, natural rates of wildfire occurrence do not reflect the probability of wildfire occurrence under current conditions and management.

the low percentage of the area annually burned by wildfire (Price et al., 2015), and, hence, the low probability of wildfire affecting areas with fuels reduced by previous fires (Boer et al., 2015).

Due to the low probability of fire affecting areas of fuel reduction while fuels are reduced, most areas with fuels reduced by forest removal will not affect fire behavior during the window of transiently reduced fuels. Thus, over most time frames in most areas subject to forest removal, the ecological costs of forest removal will not be offset by reduced watershed impacts from reductions in fire severity.

As shown by Rhodes and Baker (2008), under recent rates of fire occurrence, even in the most frequently burning forest type, many cycles of re-treatment would have to occur over hundreds of years before the majority of such treated areas were affected by fire, and thus had the potential to modify fire behavior. However, by the time these cycles of re-treatment are completed, the accrued impacts of repeated forest removal would also occur, without providing reduction in fire severity on almost 50% of the repeatedly retreated areas.

Notably, the probability of fire affecting areas while fuels are reduced is lower in fire regimes in wetter forests due to less frequent fire than in drier forests which tend to burn more frequently. For instance, Bales et al. (2011) proposed targeting the Onion Creek experimental watershed, which is dominated by fir forests, for forest removal to boost water yield, partially based on the potential for increases due to the watershed's mean annual precipitation. Under natural fire regimes, it is estimated that fire occurs about nine times more frequently in dry forest types than in fir forests in the Sierra Nevada (North et al., 2012), which indicates that treatments in drier forests are far more likely to encounter fire during the period that fuels are reduced than the forests in the Onion Creek watershed targeted in Bales et al. (2011). This, again, underscores that forests that have the greatest potential for increases in water yield in response to forest removal have the lowest potential to modify fire behavior via forest removal.

For these reasons, the impacts of wildfire and forest removal are not mutually exclusive at the watershed and larger scales. In most cases over longer timeframes, they will have combined impacts. If areas treated by fuel reductions are affected by fire while fuels are reduced, treated areas might, at best, reduce fire severity in the immediate treatment area and its vicinity, and, thus provide some spatially restricted benefits. But in such cases, the treated areas do not extinguish fire or completely prevent its spread. Thus, even when fire affects treated areas while fuels are reduced, these treatments do not assure that untreated areas, which cover the majority of the landscape, do not burn at an unchanged severity. Therefore, in most cases, the impacts of wildfire and forest removal will have combined effects on affected watershed conditions, especially over longer timeframes. The majority of fuel treatments in time and space will have impacts that are not offset by reduced watershed effects from reductions in fire severity, and, instead will combine with the impacts of wildfire.

The foregoing is an important context, because it has been estimated that the effects on watershed erosion and sediment delivery from repeated forest removal might be roughly equal to those of severe wildfire over longer timeframes due to: a) the magnitude and frequency of

re-treatment needed to maintain fuel reductions; and, b) the relatively low frequency of higher severity fire (MacDonald and Larsen, 2009; Robichaud et al., 2010). Importantly, it must be recognized the effects of both wildfire and forest removal will have combined impacts on erosion and sediment delivery, as well as soils, hydrologic processes, and downstream affected resources.

The third limit on the potential effectiveness of forest removal to affect fire behavior is fire weather. Fuel treatments have been documented to be ineffective at reducing fire severity under weather conditions that are conducive to fire burning at high severity (Romme et al., 2003a; Lydersen et al., 2014). As Lydersen et al. (2015) noted regarding fuel reductions in the area burned by the 2013 Rim Fire, “Our results suggest that wildfire burning under extreme weather conditions, as is often the case with fires that escape initial attack, can produce large areas of high-severity fire even in fuels-reduced forests with restored fire regimes.” Notably, the overwhelming majority of the forest area burned annually is burned by a small fraction of fires that burn during relatively extreme fire weather (Rhodes, 2007).

For these combined reasons, forest removal has limited potential to reduce wildfire effects on watersheds. Confluent evidence indicates there are innate tradeoffs regarding locations that affect potential effectiveness of forest removal aimed at changing fire behavior versus water yield. Locating forest removal in areas with the greatest potential for increased water yield would have a lowest likelihood of affecting fire behavior. In the vast majority of cases, the impacts of forest removal and wildfire would have combined effects on watershed processes and aquatic resources, including downstream resources.

Importantly, even if forest removal could reduce fire severity, this may not be ecologically beneficial in the Sierra Nevada forests. Available information indicates that recent rates of wildfire are well below historic natural rates in many areas (Ager et al., 2014; Baker, 2015; DellaSalla and Hanson, 2015). This is likely to contribute to stymieing the restoration of natural fire regimes, while maintaining a deficit of biologically-important early seral habitats (DellaSalla and Hanson, 2015).

3.9 Forest removal impacts on fish habitats and populations

Forest removal and associated activities will have significant negative impacts on fish habitats and populations. These impacts are significant, due to existing habitat and population conditions. Sierra Nevada fish habitats are widely degraded (Moyle et al., 1996a). Riparian areas, which are critical to the maintenance of healthy fish habitat conditions, are also pervasively degraded by past and ongoing activities in the Sierra Nevada (Kondolf et al., 1996; CWW, 1996).

Habitat damage is major contributor to the loss in the range and abundance of native fish and amphibians in the Sierra Nevada (Moyle et al., 1996a; EcoNorthwest and PRC, 2002). Johnson (1995) estimated that about 72% of native freshwater fish species in California are imperiled. Amphibians are in widespread decline in the West and aquatic habitat degradation appears to be the major cause (Bradford, 2005).

Additional habitat damage increases the likelihood of local extirpations and ultimate extinction of native salmonids due to increased population fragmentation (USFS et al., 1997c). The impacts of current watershed conditions limit the capacity for recovery of aquatic habitats by constraining the potential for restoration (Beschta et al., 2004). Additional watershed damage is inimical to the restoration of native salmonid populations (Karr et al., 2004).

While the impacts of forest removal would affect a host of aquatic species, the following focuses on salmonids for several reasons. First, the habitat requirements for salmonids are well-documented. Second, salmonid populations have high cultural and recreational value. Third, salmonid response to habitat degradation is fairly well-understood. Fourth, considerable sums of public and private funds are annually spent on efforts to maintain or increase the range and abundance of native and non-native salmonids. Fifth, a variety of policies, laws, and regulations require limiting harm to salmonid habitats and populations from activities on public lands.

The following assessment of the impacts of forest removal on salmonid habitats and populations focuses on habitat attributes that are most likely to be affected by forest removal and strongly affect salmonid survival and production. These include sediment-related conditions, large wood debris (LWD), stream channel morphology, pools, water temperature, water quantity, and riparian conditions.

3.9.1 Large woody debris and fish habitat

Forest removal associated with efforts to increase water yield are likely to involve physically significant levels of tree removal in riparian areas, as previously discussed. This would reduce LWD recruitment in streams in two ways. First, removal of trees within a site potential tree height reduces the number of trees that can be recruited to streams. Second, in the case of thinning, it significantly reduces the rates of mortality for remaining trees, which serves to amplify the effect of tree removal (Pollock and Beechie, 2014). Leaving riparian areas undisturbed by forest removal likely produces the greatest level of LWD recruitment and the greatest benefits for salmonids (Pollock and Beechie, 2014).

These impacts of LWD via tree removal in riparian areas are persistent, due to the long recovery time for trees to grow to a recruitable size and age (USFS et al., 1993; Rhodes et al., 1994; Pollock and Beechie, 2014). The removal of trees that would have otherwise become instream LWD is an irretrievable impact.

Loss of LWD recruitment is also significant due to its numerous functions in streams and their importance to salmonids. The loss of LWD reduces stream cover and the quality and quantity of pools (USFS et al., 1993; McIntosh et al., 2000; Buffington et al., 2002). Pools have been repeatedly shown to be an essential feature of salmonid habitats. The loss of LWD in streams reduces sediment storage and contributes to increased channel erosion and downstream sediment transport. The loss of LWD in headwater systems also reduces the supply of LWD downstream, because headwater channels provide LWD to downstream river reaches (USFS et al., 1993; May and Gresswell, 2003). The widespread loss of LWD in streams has been

repeatedly assessed to have contributed to the decline in the abundance and range of salmonids. It is likely that LWD loss from forest removal would reduce salmonid production.

In contrast to forest removal impacts in riparian areas, wildfire can contribute significantly to increasing LWD recruitment from intact riparian systems (Karr et al., 2004). This recruitment contributes numerous long-term benefits to salmonid habitats, as documented in Oregon by (Ratliff et al., 2015). This benefit may be one of the reasons that salmonid populations with population connectivity rebound fairly rapidly from high-severity fire impacts (Rieman et al., 2003; Rhodes, 2007)

3.9.2 Stream substrate and sediment-related habitat conditions

Forest removal at the scale and frequency sufficient to increase water yields would greatly elevate sediment delivery to streams, as previously discussed. This would have multiple impacts on water quality and salmonid habitats that separately, but especially in concert, reduce the survival and production of salmonids.

The increases in sediment delivery from forest removal will increase fine sediment levels in streams. Lab and field studies have repeatedly documented that elevated sediment delivery increases fine sediment in channel substrate, especially when the supply of fine sediment to the stream is increased (Eaglin and Hubert, 1993; Rhodes et al., 1994; Buffington and Montgomery, 1999; Hassan and Church, 2000; Kappesser, 2002; Cover et al., 2008). Increases in fine sediment in streams are particularly likely when the increases in sediment delivery are primarily comprised of fine sediment, which is the case for sediment delivery from elevated surface erosion caused by forest removal and related impacts.

Studies have consistently shown that salmonid survival and production is significantly reduced as fine sediment in streams increases (Meehan, 1991; USFS and USBLM, 1997a). Increased fine sediment in substrate sharply reduces the survival of salmonids from egg-to-emergence (Meehan, 1991; USFS and USBLM, 1997a). It also reduces available habitat for juvenile salmonids and disrupts food webs (Meehan, 1991; USFS and USBLM, 1997a; Cover et al., 2008). There appears to be no threshold below which any increase in fine sediment levels does not reduce the production of steelhead (Suttle et al., 2004), which are closely related to rainbow trout.

Elevated sediment delivery and transport contributes to the loss of pool volume, depth, and quality through several mechanisms, including increases in the sedimentation of pools via the sequestering of fine sediments in pools during lower flows (Kappesser, 2002; Buffington et al., 2002; Cover et al., 2008). USFS et al. (1993) and McIntosh (2000) noted that increased sediment delivery was one of the primary causes of the documented extensive loss of pools over several decades. This effect of sediment delivery on pool conditions would be compounded by the long-term loss of LWD recruitment from forest removal impacts. This is likely to reduce the survival and production of salmonids, because studies have repeatedly shown that pools are essential component of productive salmonid habitat (Meehan, 1991; USFS and USBLM, 1997a; McIntosh et al., 2000).

Increased sediment delivery also increases stream width and decreases stream depth in depositional reaches (Richards, 1982). Channel widening and loss of channel depth is associated with reduced pool dimensions (Buffington et al., 2002). Increases in channel width increase summer water temperatures, even in the absence of the loss of stream shade (Rhodes et al., 1994; Bartholow, 2000).

Increased sediment delivery increases suspended sediment levels, which also elevates turbidity. Increased levels of suspended sediment and turbidity impair sight feeding by salmonids (Meehan, 1991; Rhodes et al., 1994). Significantly elevated suspended levels also have negative physiological effects on salmonids (Meehan, 1991; Rhodes et al., 1994).

The impacts of forest removal at the scale and frequency sufficient to increase water yields will chronically elevate sediment delivery in pervasive manner. Widespread, persistent, and chronic aquatic impacts from persistent, repeated watershed disturbances, as would occur under forest management aimed at increasing water yield, may be more deleterious for native fish than infrequent, but acute, impacts, from wildfire and its watershed effects (Rieman et al., 2003; Dunham et al., 2003).

3.9.3 *Water temperature*

Forest removal at a scale and frequency that can increase water yields is likely to contribute to increased water temperatures via two mechanisms. First, the combined effect of peakflow elevation and increased sediment delivery are likely to widen unconfined stream channels in depositional settings (Richards, 1982; Reid, 1993). This is especially likely because logging and roads clearly increase peakflows with return intervals of 1-5 years. Peakflows with recurrence intervals in this range exert the dominant control on channel dimensions (Dunne and Leopold, 1978; Richards, 1982). In combination with increased sediment delivery, which will also occur in response to forest removal, increases in channel width are especially likely (Richards, 1982).

Increases in channel width are likely to increase low flow stream widths. Dose and Roper (1994) found that low-flow stream widths had increased in a statistically significant fashion with increased levels of logging in watersheds in southwestern Oregon. Dose and Roper (1994) cited increases in peakflow from logging and roads as one of the primary mechanisms causing increased channel width.

Due to the nature of stream heating, increases in channel width contribute to elevated water temperature, even in the absence of shade loss. Bartholow (2000) estimated that the increases in channel width documented by Dose and Roper (1994) significantly increased summer water temperatures, in the absence of any reduction in stream shading.

Second, recent forest projects aimed at fuel reduction are likely to involve physically-significant levels of removal of stream shade due to forest removal within RHCA's, as previously discussed. Reductions in stream shade contribute to increased water temperatures (USFS et al., 1993; USFS and USBLM, 1997a; Rhodes et al., 1994; McCullough, 1999). These effects on small streams can elevate water temperatures in downstream reaches (Allen and Dietrich, 2005;

Allen et al., 2007). The increased scale of forest removal needed to increase water yields would likely increase shade removal and consequent water temperature impacts.

The full recovery stream-shading requires more than a decade after forest removal (Rhodes et al., 1994). Thus, the removal of forest vegetation at a frequency of every 10 years, as would be needed to potentially elevate water yields, would likely cumulatively increase water temperature over time.

Increased water temperature has numerous adverse impacts that significantly reduce the survival and production of salmonids (McCullough, 1999). Water temperature is already a pervasive water quality problem for salmonids, which is expected to worsen with climate change (Wade et al., 2013)

3.9.4 *Low flows*

As previously discussed, forest removal in snow-dominated areas of the Sierra Nevada are unlikely to increase low flows. However, if forest removal at a scale sufficient to affect water yields is not followed by repeated vegetation removal, decreases in low flows are likely to occur.

Decreases in low flows would have several negative effects on salmonids. These include a reduction in available habitat during the low flow period and increased water temperatures. Both effects would be significant, because climate change is likely to contribute to reductions in low flows and increased water temperatures (Viers et al., 2013; Wade et al., 2013; Beschta et al., 2013). Thus, the water temperature impacts from forest removal on salmonids are likely to be exacerbated by climate change.

3.9.5 *Summary: Effects on fish habitats and populations*

Forest removal at a scale and frequency sufficient to potentially affect water yield would have numerous adverse impacts on fisheries. The loss of pool volume and depth, increased fine sediment, and increased summer water temperature all have several significant negative effects on the survival, production, and recovery of salmonids, as documented in legions of studies (e.g., Meehan, 1991; USFS et al., 1993; Rhodes et al., 1994; USFS et al., 1997a; c). Thus, the scale and frequency of forest removal needed to increase water yields will incur pervasive and enduring costs with respect to salmonids populations and fisheries. These losses would have costs that involve not only the value of the fish, but also lost angling opportunities, which can generate significant income to many local community economies.

The most effective approach to the protection and restoration of streams, salmonid habitats, and related problems is to prevent further damage, while curtailing or eliminating activities that prevent natural recovery (USFS et al., 1993; Rhodes et al., 1994; Henjum et al., 1994; Kattelman, 1996; Espinosa et al., 1997; Kauffman et al., 1997; Beschta et al., 2013). This approach is also far more efficient and tractable than trying to rehabilitate degraded conditions and/or while additional degradation is ongoing (Rhodes et al., 1994; Henjum et al., 1994; Beschta et al., 1995; Kattelman, 1996; Espinosa et al., 1997; Kauffman et al., 1997; Beschta

et al., 2013). Forest removal at the scale and frequency sufficient to increase water yield in wetter years would conflict with the most effective and efficient approach to protect and restore salmonid and other fish populations and their habitats.

3.10 Impacts on downstream water supplies

Forest removal would have several impacts that would incur significant costs for downstream water supplies and associated infrastructure and activities. These costs would be pervasive and enduring.

3.10.1 Sediment-related impacts on water quality and water supplies

Forest removal would significantly elevate downstream sediment transport, which would have several negative impacts on downstream water uses. Increases in suspended sediment increase turbidity, thus degrading water quality.

The increased downstream delivery of sediment cumulatively caused by forest removal would increase sedimentation in downstream reservoirs, decreasing available reservoir storage. Over time, this elevated reservoir sedimentation would impair the ability of reservoirs to provide intended functions, including water supply, hydroelectric generation, and flood control. It would also greatly increase dredging costs, in cases where dredging occurs. In many cases, dredging is not cost-effective. In these cases, increased reservoir sedimentation from forest removal and related effects would decrease the useful life of the reservoirs.

Harrison (1991) documented the effects that soil erosion from dispersed land management activities in the Sierra Nevada can have on sedimentation of reservoirs. Two reservoirs on the North Fork Feather River in California accumulated 5.2 million m³ of sediment in a 36-year period from natural and accelerated erosion. This caused operational problems resulting from the loss of about half the capacity of the two reservoirs. Erosion of stream banks, road cuts, logged areas, and grazing lands were among the most significant contributors to the sediment problems. Approximately 70 percent of the area of the East Branch of the North Fork Feather River, identified by project sponsor, Pacific Gas and Electric, as the major producer of sediment is National Forest land. Reduction in erosion would have considerable economic benefits related to the reservoir. Harrison (1991) stated that “improved watershed management may enhance electric generation by increasing base stream flows and decreasing peak flood flows.”

As noted in ENW and PRC (2002) regarding this case study and other sediment-related impacts:

“Water quality also can be important to other resource users. Sediment in streams, for example, can clog channels, reducing their capacity and increasing the risk of future flooding. It also can settle in reservoirs, reducing their capacity to store water for future use, and increasing the maintenance costs for hydropower turbines and other infrastructure. At two reservoirs in the North Fork Feather River owned by Pacific Gas and Electric (PG&E), sediments obstructed the low level outlets, the stream-flow release systems, and the water inlets for operation of

the spillway drum gates. Sediment at both facilities was being drawn through the turbines, accelerating wear and increasing maintenance costs. In 1995, PG&E estimated that suspended sediment increased overhaul costs of hydroelectric turbines by \$25,000 per unit per year (Sohrakoff, 1999). PG&E identified erosion of stream banks, road cuts, logged areas, burned areas, mine tailings, and grazing lands as the most significant contributors to the stream-carried sediments. The short-term solution to the sediment-related problems was to dredge 620 acre-feet of sediment, but the restricted working area and long haul distances to suitable disposal sites made dredging costs very high. PG&E became involved in an erosion control program in the watershed, where their primary goal was to reduce the rate of sediment accumulation in its hydroelectric reservoirs downstream of the program area. The benefits included reducing future dredging requirements by as much as 50 percent, reducing turbine maintenance, improving water quality, and improving public relations (Harrison, 1991).” (© ECONorthwest and Pacific Rivers Council 2002, Used by permission)

The increased suspended sediment and turbidity from forest removal would also likely affect downstream domestic water supplies. Forest removal would also increase nutrients transported downstream, as discussed in more detail in the following section. Such increases in sediment and nutrient concentrations and loads can greatly increase water treatment costs (ENW and PRC, 2002).

Cities can avoid increased water treatment costs by protecting watersheds. It is estimated that Salem, Oregon, a city of about 100,000, would incur a cost per resident of approximately \$16-\$32 if watershed degradation caused the city to need to employ a conventional filtration system (ENW and PRC, 2002). New York City concluded that protecting the watersheds providing water supply was far more cost-effective than compensating for water quality degradation by building a new water-filtration system. The estimated savings exceed \$5 billion (ENW and PRC, 2002).

ENW and PRC (2002) estimated that costs incurred by downstream users from sediment generated from logging were equal to about \$185 per acre logged. Thus, the large scale and frequent forest removal that would be needed to increase water yields in wet years would incur persistent costs to downstream water users.

3.10.2 Nutrient-related impacts on water quality and water supplies

Available studies of stream water chemistry consistently indicate that forest removal inexorably increases nutrient leaching from watersheds and resulting nutrient loads in streams (Binkley et al., 2004). Two nutrients are of principle concern in terms of forest management, although they occur in variant forms that can affect or reflect their fate and effects in soil and water. Phosphorus (P) is generally associated with soil disturbance and erosion from forest management activities, including gully erosion and channel enlargement, landslides, and roads. Nitrogen (N) is broadly generated and freed in solution in soil water, groundwater, and thus into surface water as an inevitable consequence of vegetation removal. Logging and fire, and insect or pathogen sources of extensive tree mortality all have been shown to elevate nitrogen in runoff, for up to several years after initial forest disturbance. Commonly after four

to six years, vigorous re-growth of second growth vegetation increases N demand and loading to streams may return to background rates. However, if secondary disturbances such as windthrow or disease propagated by tree damage or soil disturbance are triggered by the forest removal, as is quite common, then elevated N loads could persist for a decade or more.

It is frequently noted that on a per unit area acre basis, forest management mobilizes less nitrogen and phosphorous than many other land uses. Moreover, many sources (e.g., Binkley, et al. 2004) observe that nutrient concentrations in streams within disturbed forest watersheds seldom reach levels that violate water quality regulations. However, when large watershed areas are subject to forest disturbance at high frequency, then cumulative nutrient loading from forestry over time can assume very large proportions, and can greatly exceed that from other nutrient sources (Byron and Goldman, 1989).

Forestry, in combination with natural forest vegetation disturbances (wildfire, windstorms, and disease or pest outbreaks), in a watershed can accelerate adverse cumulative effects on nutrient loading to streams, wetlands, rivers and lakes. These effects are additive to those from other sources including septic and sewer systems, runoff from roads, grazed lands, and urban areas, and channelization and wetland loss.

To generate measurable increase in streamflow, forest removal would have to affect large areas of treated watersheds at a frequency of about every 10 years. Hence the cumulative increase in nutrient yield from forested headwaters to downstream receiving waters would exceed those from natural forested conditions by an order of magnitude, and would be dramatically higher even than that observed for watershed managed under typical commercial timber rotations (ranging roughly from 50-250 years). Hence, cumulative nutrient loads to downstream receiving waters (rivers, lakes, reservoirs, estuaries, and the nearshore marine environment) would greatly increase compared to present-day loadings. It is important to recognize that many of these waters already display acute or incipient water quality impairment from nutrient loading under existing watershed conditions (see http://www.swrcb.ca.gov/centralvalley/water_issues/tmdl/impaired_waters_list/).

Proportional losses of nutrients into waters are dramatically higher with the initial disturbance of intact natural vegetation—as occurs with logging of even small areas of forest—than when vegetation is further altered in extensively-disturbed ecosystems such as croplands or urbanizing areas (Wickam et al., 2008). This is because undisturbed natural forest vegetation has exceedingly small baseline nutrient losses (i.e., undisturbed natural forest cover, with its dense and highly biologically integrated subsurface root and microbial systems, is highly retentive of nutrients) as documented in the Sierra Nevada (Rhodes, 1985; Rhodes et al., 1985). As a result, increased logging or other forest disturbances can dramatically increase nutrient loading to downstream waters compared to similar changes of disturbance on other land use types, where background losses are already quite high and sustained. Because ecosystems tend to equilibrate to prevailing nutrient loads, in terms of triggering progressive eutrophication of downstream lakes, wetlands, rivers, and estuaries, the magnitude of change of nutrient loading could be as important as or more important than the previously prevailing average loading. For example, clearcut logging increased nitrogen loading to an adjacent stream by about 7-fold in one Idaho study, while partial cutting caused a more than 5-fold

increase (Gravelle et al., 2009). Downstream of the cutting units, cumulative nitrogen concentrations increased from pre-logging background levels by about 450-500 percent. Extensive and frequent forest removal results in very large and sustained increases in nutrient loads.

While undisturbed streamside forest buffers exceeding 150-250 feet distance upslope from stream and floodplain margins could be effective in retaining most nutrients and keeping them from stream waters (Nieber et al., 2012; Sweeney and Newbold, 2014) current Riparian Habitat Conservation Areas (RHCAs) established for streams under the current land management direction for USFS lands in the Sierra Nevada (USFS, 2004) are not adequate to achieve high levels of nutrient retention. In part this is because slopes are often steep and soils coarse-texture, but also because the Forest Service allows extensive tree removal within the RHCA itself, and nutrients generated from forest removal within the RHCA are generated close to streams and escape the full width of riparian buffer retention.

Increased nutrient delivery to fresh and marine waters increases eutrophication. Increased nutrients, particularly when nitrogen and phosphorous are combined, can cause a host of undesirable effects where they accumulate in downstream waters (Goldman, 1988; Anderson et al., 2008; Cloern, 2003; Gilbert, 2010; Committee on Environment and Natural Resources, 2010). Increased algal growth in streams associated with nutrient inputs can result in increased oxygen consumption at night when the expanded plant community is respiring but not producing oxygen through photosynthesis. Large day-to-night swings in oxygen concentration and even pH can result, producing stressful conditions for fishes and other aquatic organisms. When these nutrients eventually work their way downstream to large pools, backwaters, wetlands, coastal lakes, and estuaries, they can produce acute eutrophic effects (Freeman et al., 2007). These effects include explosive growth of nuisance plants, including toxic algae, oxygen depletion, high concentrations of plant-derived solutes in the water that result in acidic conditions, discoloration, and unpalatable odor and flavor in drinking water. Filtration and chemical treatment of water from eutrophied lakes and rivers to make it suitable for municipal or domestic use can be very expensive and often only marginally effective (Schwarzenbach et al., 2006).

Unfavorable ecological conditions associated with eutrophication could be one reason why extensive areas of habitat in coastal rivers and lakes that are otherwise suitable for salmon, trout, delta smelt, and other fishes--and were historically productive for those species--appear to go largely unused by them today (e.g., Gilbert, 2010). Invasive species, such as carp, are relatively tolerant and favored by eutrophic conditions. Nitrogen loading can also cause eutrophication in estuaries and can contribute to large-scale hypoxia of nearshore and offshore marine habitats (Cloern, 2003; Committee on Environment and Natural Resources, 2010).

For these combined reasons, forest removal sufficient to increase water yield can be expected to greatly elevate nutrient loads to downstream waters over existing conditions, and it cannot be assumed that RHCAs can consistently reduce nutrient loading impacts to negligible levels. Therefore, it is highly likely that forest removal of at least 25% of the area of affected watersheds repeated at 10 year cycles would significantly elevate nutrient loading and expand

and aggravate eutrophication of downstream rivers, lakes, reservoirs, and estuaries, adversely affecting a host of aquatic resources and downstream uses.

3.11 Invasive vegetation/noxious weeds

Forest removal increases the dispersal and establishment of invasive vegetation, including noxious weeds, due to associated soil disturbance, road effects, and increased vehicular traffic. As Keeley (2006) noted, “Forest fuel reduction programs have the potential for greatly enhancing forest vulnerability to alien invasions.”

Forest removal over significant areas is likely to increase the spread of invasive non-native noxious weeds because logging disrupts native plants and creates the soil disturbance favorable to weed establishment (USFS and USBLM, 1997a). Soil compaction provides noxious weeds with a competitive advantage over native plants (USFS and USBLM, 1997a).

Machinery used in logging and log hauling is also a vector for weed dispersal. Roads and road use are one of the primary causes of noxious weed spread and establishment (USFS and USBLM, 1997a; USFS, 2000b). Roads spread noxious weeds by simultaneously acting as dispersal corridor, while disturbing soils and eliminating native vegetation on road prisms, cuts, and fills, all of which provides noxious weeds with a competitive advantage over native species (USFS and USBLM, 1997a). Mechanical treatments combined with prescribed fire have been found to favor invasion by non-native vegetation in areas where these treatments have been applied with the aim of reducing fuels and altering fire behavior (Schwiik et al., 2009).

Increases in the extent and intensity of noxious weed infestations are a serious negative by-product of forest removal for several reasons. It is already a major environmental problem on public forest and grasslands in the Sierra Nevada (USFS, 1999). Noxious weeds displace native plant species, degrading terrestrial habitats. Noxious weeds can also alter fire regimes and increase surface runoff and erosion, reducing grassland and forest productivity (CWWR, 1996; USFS and USBLM, 1997a; Beschta et al., 2013).

The prevention of noxious weed spread and establishment is the most essential and cost-effective aspect of weed control efforts (CWWR, 1996; USFS and USBLM, 1997a). Other treatments, particularly herbicide and mechanical treatments, have the double disadvantage of being fairly costly and relatively ineffective. Moreover, treatments to eliminate noxious weeds once they have become widely established often pose risk of introduction of sediment and toxic chemicals into waterways.

For these reasons, an extensive program of forest removal would have significant costs due to the negative long-term effects on non-native vegetation and noxious weeds and consequent ecological effects. Any program of large scale soil and vegetation disturbance, coupled with increased vehicular use, as extensive forest removal requires, is antithetical to efforts to control the extent of the serious environmental problem of non-native vegetation spread and establishment.

4. Land Management Approaches that Benefit Water Supplies and Watersheds without Incurring Significant Environmental Costs

The foregoing clearly indicates that attempting to increase water yield via forest removal has very limited benefits that incur high costs via numerous adverse environmental effects. It also indicates that any potential increase in water yield would not be self-sustaining. Instead, it would require initiating a cycle of fiscally and environmentally costly treatments that would perpetuate watershed and aquatic degradation.

Kattelman (1987) noted that due to the limited prospects for boosting water yield by removing forests, land management aimed at extending or augmenting low flows might have the most promise for contributing positively to water supplies from the Sierra Nevada. Sedell et al. (2000) also echoed this assessment.

For these reasons, the following examines several approaches that:

- can contribute to improved low flow conditions;
- are self-sustaining;
- do not incur high or enduring environmental costs;
- provide an array of ecosystem benefits;
- provide benefits for downstream water use via improved water quality;
- address pressing forest restoration needs;
- contribute to watershed resiliency in the face of climate change.

Notably, several of these approaches are complementary. If pursued concurrently, the total benefits of the approaches would be increased.

4.1 Significant reductions in livestock grazing

Reductions in livestock grazing has considerable promise for augmenting low flows (Ponce and Lindquist, 1990; Reeves et al., 1991; Rhodes et al., 1994; Beschta et al., 2013). Studies have found that the elimination of grazing along some formerly non-perennial streams resulted in the streams gaining year-round flow (Ponce and Lindquist, 1990; Reeves et al., 1991; Rhodes et al., 1994).

Grazing elimination results in the recovery of numerous watershed conditions and processes that contribute to increases in low flows. Grazing cessation allows areas compacted by grazing to recover. This results in major increases soil water storage capacity that augment low flows. This effect has been shown to be significant in riparian areas and other areas that typically store a significant amount of water and release it to streams during the low flow period. Kauffman et al. (2004) documented that soils that had been free of grazing for 6-18 years could store approximately 61,000 L/ha more water in just the upper 10 cm of soil than comparable soils that had continued to be affected by livestock grazing. Even at the relatively limited scale of the study, along a 30 km long riparian reach, Kauffman et al. (2004) estimated ungrazed soils could hold 16.6 million liters more water in the absence of grazing than grazing-impacted soils. This

is extremely significant because a considerable fraction of the additional water storage would contribute to low flows.



Photo 10. Typical example of persistent cattle grazing damage to vegetation and streambed and banks. Little Indian Meadow, Inyo National Forest. Photo by Chris Frissell.

The recovery of compacted soils in the absence of grazing can also contribute to increased low flows via the recovery of infiltration capacity that comes with the recovery of compaction. Kauffman et al. (2004) documented that soils that had not been subject to grazing for several years had vastly higher infiltration rates than soils subjected to on-going grazing. Thus, grazing cessation increases the amount of water that is absorbed by soils and can ultimately contribute to low flows, rather than being shed quickly as surface runoff (Beschta et al., 2013).

Grazing cessation also contributes towards the recovery of other grazing impacts that have contributed to existing reductions in low flows. These impacts include stream incision, which contributes to desiccation of floodplains and wet meadows and the loss of flood-water detention storage (Ponce and Lindquist 1990; Platts, 1991; Beschta et al., 2013; Viers et al., 2013).

Emmons (2013) estimated that restorable water volume in Sierra Nevada meadows was 120×10^9 liters, or approximately 97,000 acre-feet, based on the depth of incised streams, soil water holding capacity, and meadow area. This estimate did not take into account the potential

increases in available water storage due to the recovery of soil properties with grazing cessation, as documented by Kauffman et al. (2004). Emmons (2013) noted that restoring water volume storage in these Sierra meadows would provide important co-benefits to wildlife and to low flows.

These effects on low flows would benefit salmonids by providing increases in useable habitat area. They would also benefit water supplies by providing additional water during periods of relatively high downstream demand and relatively limited supply. Importantly, these positive effects on low flows from curtailing livestock grazing are self-sustaining (Beschta et al., 2013). They would also help offset the impacts of climate change, which is likely to reduce low flows and late summer moisture levels in meadow soils (Beschta et al., 2013), including those in the Sierra Nevada (Viers et al., 2013).

Curtailing grazing would also provide numerous other ecological benefits, many of which would increase the resiliency of watersheds and aquatic systems to the adverse impacts of climate change (Beschta et al., 2013; 2014), as Nusslé et al. (2015) corroborated in the Sierra Nevada. Reductions in the extent and impacts of grazing are likely to help curtail the spread of invasive vegetation, including that which contributes to fire regime alteration (Beschta et al., 2013; 2014). The suspension of grazing contributes to fire regime restoration in areas that primarily had a natural fire regime of frequent and low-severity fires (Noss et al., 2006; Beschta et al., 2014). Grazing curtailment contributes to the resiliency of watersheds and aquatic systems to fire effects and is key to unimpeded watershed recovery after fire (Beschta et al., 2004; Karr et al., 2004).

Grazing cessation contributes to decreases in water temperatures in several ways. Channels that have been widened by the combined impacts of grazing often substantially narrow after grazing is ceased (Rhodes et al., 1994; Magilligan and MacDowell, 1997; Beschta et al., 2013; Batchelor et al., 2014). Riparian vegetation typically recovers after grazing cessation providing increases in stream shade (Platts et al., 1991; Beschta et al., 2013; Batchelor et al., 2014; Nussle et al., 2015). Both of these effects contribute to significant reductions in water temperatures. Notably, Nusslé et al. (2015) documented that grazing suspension contributed significantly to water temperature reduction relative to grazed areas in the Sierra Nevada. This is a key benefit to salmonid fishes, because climate change is likely to increase water temperatures and related piscine impacts (Beschta et al., 2013; Wade et al., 2013), including in the streams of the Sierra Nevada (Viers et al., 2013; Nusslé et al., 2015).

Increases in low flows and increases in stream-floodplain hydrology connectivity can also make additional contributions to reductions in water temperature, especially because water supplied to streams from soil storage provide a source of cool water input that can moderate water temperatures (Rhodes et al., 1994). These confluent benefits on water temperature would significantly benefit salmonids and other aquatic species that benefit from cooler water during the summer.



Photos 11 and 12. Upper photo (11) shows heavily degraded stream and meadow conditions caused by livestock, which contribute to low flow reduction and water quality problems. Lower photo (12) is of the same creek in a fenced enclosure immediately upstream of the area in Photo 11. The fenced enclosure had largely eliminated livestock grazing for about 14 years at the time of the photo. Note the stable banks, healthy woody riparian vegetation, relatively narrow stream channel, and high degree of floodplain connectivity in the lower photo, in comparison with the grazed conditions in the upper photo. This type of rapid recovery can be simply and inexpensively achieved by grazing curtailment without any adverse environmental impacts. Malheur National Forest, Oregon. *Photo by J. Rhodes.*

Grazing cessation allows the causative impacts from grazing on increased sediment loading to streams to abate. This is significant because grazing elevates sediment delivery in several ways, including increasing surface erosion and sediment delivery to streams, as well as greatly increasing stream erosion (Beschta et al. 2013). Assessments have consistently identified livestock grazing as a significant source of elevated sediment delivery (Beschta et al., 2013). Grazing cessation is highly likely to decrease downstream delivery of sediment that affects fish habitats, fish populations, water quality, water supplies, and reservoirs. Hence, the reductions in sediment delivery would have significant benefits for downstream water use and fisheries.

The curtailment of grazing is likely to have other water quality benefits. These include reduction in nutrient loads and bacterial pathogens (Derlet et al., 2008; 2012; Beschta et al., 2013).

Some of these benefits are summarized in Table A-1 in Appendix A. Again, these environmental and societal benefits would be self-sustaining, provided livestock grazing, especially in riparian areas, was eliminated. Notably, the approach is unlikely to incur any adverse environmental costs, unlike attempts to increase water yield via forest removal. Further, a significant reduction in grazing on federal public lands is likely to result in net benefits in terms of the costs of administering grazing versus fees received (Beschta et al., 2013).

4.2 Road obliteration and cessation of road construction

There have been few studies of the effects of roads on low flows. However, it is extremely well-documented that roads intercept subsurface flow at road cuts, and shunts it to surface runoff (Wemple, 1996; La Marche and Lettenmaier, 2001). This interception of subsurface flow is inevitable at road cuts due to soil physics (Kirkby et al., 1978). This interception by roadcuts is likely to reduce downslope soil moisture levels and subsurface flow contributions to affected streams, contributing to reduced baseflows (Tague and Band, 2001). Hancock (2002) noted that logging and roads reduced subsurface flows to hyporheic areas by reducing subsurface percolation and baseflow contributions to streams.

Compacted road surfaces also shunt precipitation to surface runoff, often directly to streams, and thus prevent a considerable amount of water from infiltrating into soils where it can contribute to low flows. These combined, and inevitable, impacts of roads may contribute to reductions in low flows. These long-term persistent impacts of roads on streamflow generation may be a factor that contributes to eventual persistent reduction in low flows after forest removal that has been documented in several areas.

Based on these known hydrologic impacts of roads, road obliteration may be able to contribute to the recovery of low flows in affected watersheds. This is more likely if roads that significantly intercept subsurface flow or prevent its egress to streams, such as those in riparian areas, were targeted. Restoration of subsurface pathways that provide baseflow would likely require recontouring of the topography altered by roads.



Photo 13. Remediation and obliteration of existing forest roads, can alleviate road impacts on watershed functions over time, contributing to the recovery of hydrologic flow routing, erosion, sediment delivery, and water quality. Plumas National Forest. Photo by C. Frissell.

The potential benefits on low flows would likely be slow to accrue, due to the slow recovery of hydrologic properties on obliterated roads. For this reason, a complementary strategy would be to cease additional road construction, because it has immediate, long-lasting, and cumulative hydrologic impacts that cannot be rapidly reversed.

Such an approach would yield many other benefits to watershed and aquatic resources, as well as downstream water use. Reductions in road length, especially of those segments that intercept subsurface flows and convey copious amount of the converted runoff to streams, would significantly reduce sediment delivery to streams over time. This reduction in sediment delivery would help improve water quality and fish habitat conditions, including substrate and pools, in ways that would contribute to increases in the survival and production of fish. The improvement in sediment-related water quality would also help to reduce sedimentation in downstream reservoirs and contribute to reductions in water quality treatment costs.

Road removal would also help contribute to reductions in peakflows and associated adverse impacts on aquatic resources and downstream flood magnitudes. Reductions in roads would also contribute to limiting current rates of invasion by non-native vegetation. A reduction in the extent and impact of roads has been repeatedly cited as one of the more promising and pressing

priorities for restoring aquatic systems (USFS et al., 1993; Gucinski et al., 2001; Luce et al. 2001, Roni et al. 2002; Switalski et al., 2004; Steel et al., 2008; Furniss et al., 2010). Reducing road impacts contributes to the resiliency of watersheds and aquatic system to fire effects and is also key to unimpeded recovery after fire (Beschta et al., 2004; Karr et al., 2004).

Roads in riparian areas have greater and more numerous adverse impacts than roads that are farther away from the streams network. Thus, reductions in road extent in riparian areas would provide still greater aquatic benefits.

Re-establishing subsurface flows via road obliteration and recontouring, may contribute to long-term improvement in water temperatures. This is because subsurface flow interception likely increases water temperatures via a two-pronged effect. Reductions in subsurface flows to streams reduce low flow volumes, which, alone, increase summer water temperatures (Beschta et al., 1987; Rhodes et al., 1994). Subsurface flows are also typically far cooler than surface flows, aiding in the thermal regulation of streams during low flows (Beschta et al., 1987; Rhodes et al., 1994). Some of the benefits of reduced road mileage are summarized in Table A-2 in Appendix A.

Although road obliteration is relatively costly fiscally, the costs would be offset by decreases in otherwise perpetual road maintenance costs. The cost to erase the backlog in road maintenance on USFS lands is presently insurmountable; in 2000, it was estimated to exceed several billion dollars (USFS, 2000b). Road obliteration would contribute to decreasing this backlog over time, while cessation of road construction avoids further inflation in maintenance needs and associated expenses. The cessation of additional road construction would also prevent additional long-term watershed damage, while reducing the costs incurred by road construction, which are one of the highest in terms of per unit area activities on public forests. While road obliteration may be relatively costly, the benefits would likely be self-sustaining and accrue over time.

4.3 Re-establishment of beaver populations

Re-establishing beaver, which evidence indicates are native to the Sierra Nevada (Lanman et al., 2012; James et al., 2012), could contribute to augmenting low flows (Pollock et al., 2015). Besides low flow augmentation, beaver convey have numerous other aquatic benefits

Beaver re-establishment would also likely help higher water tables to develop, thereby increasing hyporheic exchange (Lowry and Beschta, 1994; Pollock et al., 2015).

It is also likely to help hydrologically reconnect floodplains to streams, which would help attenuate peakflows. It would also provide an important mechanism for converting snowmelt-driven peakflow to baseflow for summer streamflow. Beaver ponds aid in reducing water temperatures via effects on hyporheic exchange. This reduction in water temperatures would benefit salmonids. Beaver ponds have been shown to provide excellent salmonid rearing habitat, contributing to increases in salmonid production (Pollock et al., 2015). Beaver ponds also reduce downstream sediment transport by sequestering sediment. This provides improved downstream water quality.

These benefits would have negligible environmental costs and be self-sustaining. Notably, curtailment of riparian grazing is likely to complement beaver re-establishment via the recovery of riparian vegetation (Beschta et al., 2013).



Photo 14. Beaver lodges benefit flow conditions and fish habitats, especially when coupled with the suspension of livestock grazing. The beaver lodges in the photo are located in an enclosure which has largely eliminated livestock grazing for approximately 30 years on the Malheur National Forest, Oregon. *Photo by J. Rhodes.*

4.4 Extent matters: Reducing road and grazing impacts are major restoration priorities in the Sierra Nevada

Grazing cessation and road obliteration and construction cessation also addresses prime restoration needs, based on their ecologically negative impacts and extent on USFS lands in the Sierra Nevada. Readily available data on conditions in 11 national forests in the Sierra Nevada (USFS, 2000a; 2004) indicate that grazing and roads affect a much greater area on an annual basis than high-severity fire does (Table 4). In these national forests, high-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, high-severity fire is characteristic of the natural fire regimes in much of the area that burns at that severity; hence, it is not an ecological aberration. Further, high-severity fire conveys

numerous long-lasting benefits to watersheds and the impacts are relatively transient. In contrast, livestock grazing and roads do not provide any ancillary environmental benefits and their negative impacts are enduring.

Table 3. Area of annual watershed impacts in the planning area for the USFS 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	--

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 (Table 3). Therefore, roads annually affect about several times the area annually affected by high-severity fire (Table 3). 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 in Table 3. Notably, although the estimated mean annual area of high-severity fire in Table 3 is approximate and based on relatively stale data, even if high severity fire is several times higher than indicated in Table 3, it is still the case that roads and livestock grazing afflict a far greater area than wildfire does annually in the Sierra Nevada.

The impact indices from the Equivalent Roaded Area method, which is widely used as a cumulative effects assessment tool on USFS lands in the Sierra Nevada (Menning et al., 1996) also indicates that roads and grazing have greater impacts on watersheds than wildfire does. Due to the extent of road impacts and their lack of ecological benefits, unlike wildfire, abating road and grazing impacts are clearly pressing restoration needs on USFS lands in the Sierra Nevada. Effective reduction of these impacts would also convey water supply benefits.

4.5 Summary: Effective watershed restoration would convey considerable water supply benefits at relatively low cost

The foregoing alternative approaches can improve low flow water supply conditions, without incurring substantial costs, while providing numerous significant co-benefits to watersheds, in

self-sustaining manner. This contrasts substantially with forest removal approaches that have limited benefits that are not self-sustaining, while incurring substantial watershed and downstream water use costs. Table 4 provides a comparative summary of the some of the attributes of these approaches.

Table 4. Comparative summary of benefits and costs of forest management approaches to improve water supply.

Action	Water yield	Low flow	Peak flow	Negative collateral watershed / aquatic impacts?	Watershed/aquatic co-benefits	Self-sustaining?	Soil conditions/watershed water holding capacity	Erosion/aquatic sediment loads	Water temperature	Nutrient loads	Fish habitat productivity
Forest Removal	+, T	+/-, T	+, L	Y, L, R	N	N	-, L, R	+, L, R	+, L, R	+, L, R	-, L, R
Curtail grazing	=	+, L	-	N	Y, L	Y	+, L	-, L	-, L	-, L	+, L
Road obliteration	=	=/+, L	-	N	Y, L	Y	+, L	-, L	-	-, L	+, L
Re-establish beaver	=	+, L	-	N	Y, L	Y	+, L	-, L	-		+, L

KEY: + = increase, - = decrease, **Y** = yes, **N** = no, **L** = long-term effect, **T** = transient effect, **R** = repeated effect leading to increasing cumulative effects.

5. Conclusions

Forest removal, if conducted on a scale and frequency significant enough to increase water yield in wetter years would incur numerous environmental costs, including those that degrade water quality and would be borne by downstream water use. Other costs include increased flooding, increased soil loss and degradation, increased loss of reservoir storage, and contractions in the abundance of salmonid fishes and harm to other aquatic biota.

The potential benefits of attempting to increase water yields are limited due to the magnitude of treatment combined with water yield variability, transience of effects, and relationship to precipitation levels, as well as the degradation of water quality and its downstream effects. It is assured that increases annual water yield, if realized, would be relegated only to wetter years and only during wettest season, which would have low or no benefits. The approach has little to no promise for providing additional water during the driest seasons and years when it might be most useful.

Due to the transience of effects, even after extensive and expensive clearing of forests and associated activities, repeated treatments of a similar scale would frequently be necessary to potentially maintain increased water yields in ways that are subject to the same limitations. These treatments would be fiscally and environmentally costly. This assessment of the approach is not new. It has been made repeatedly in expert evaluations of the utility and tractability of the approach (Ziemer, 1986; Kattelman, 1987; Sedell et al., 2000; NRC. 2008) over decades. This repeated assessment is not surprising, because even with the availability of some new watershed studies in recent years, the basic set of data and associated scientific understanding of the limited benefits and high costs of attempting to increase water yields via forest removal have remained largely unchanged.

There is still much to be gained by improved management of forested watersheds in the Sierra Nevada. Several approaches that are known to improve the ecological conditions of watersheds and streams (road obliteration, cessation of road construction, suspension of grazing at the watershed scale, etc.) can contribute to improving the water quality and water supply during the low flow periods when demand is relatively high and supply is relatively low. These benefits may be the greatest that forest management in the Sierra Nevada can provide for downstream water supply.

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Appendix A: Tables Summarizing the Major Benefits and Low Costs of Reducing Road and Grazing Impacts

Table A-1. Grazing--Summary of scale, restoration needs, benefits and risks, and levels of certainty associated with curtailment of grazing

Current scale of Activity or Condition	Negative physical effects on watershed condition or function in the absence of restoration approach	Effects on fauna and beneficial uses in the absence of restoration approach	Restoration Attributes			
			Level of need and measures	Ecological Benefits and Level of Certainty	Ecological Risks and Level of Certainty	Economic or Logistical benefits
411 active allotments covering 7,165,085 to 7,881,593 acres. 463 total allotments covering 8,071,616 to 8,878,778 acres. AUMs: 412,734 to 464,326	<ul style="list-style-type: none"> • ↓ ≠ low flows • ↑ soil compaction • ↓ soil productivity • ↑ erosion, sediment delivery, sedimentation, seasonal water temperature extremes • ↓ bank stability, undercut banks, pool volume • ↑ peakflows • ↓ ≠ water table elevation • ↑ ≠ stream incisement • ↓ ≠ stream shading, groundcover, hydric species • ↑ noxious weeds, cheatgrass • ↑ ≠ fecal coliform in streams, lakes, and reservoirs • ↑ nutrient pollution streams, lakes and reservoirs 	<ul style="list-style-type: none"> • ↓ habitat quality and quantity for salmonids and amphibians • ↓ riparian wildlife habitat • ↑ storage loss in reservoirs • ↓ useful reservoir life, water supplies in reservoirs • ↑ accelerated eutrophication in lakes and reservoirs 	<p>VH – Suspension of grazing in damaged but resilient areas</p> <p>VH – Elimination of grazing in susceptible areas and areas with at risk aquatic species and aquatic emphasis areas</p>	<ul style="list-style-type: none"> • ↑ low flows • ↓ soil compaction • ≈ ↑ soil productivity • ↓ erosion, sediment delivery, sedimentation • ↓ seasonal water temperature extremes • ↑ bank stability, undercut banks • ↑ pool volume and quality • ↓ surface runoff • ↑ water table elevation • ↓ stream incisement, • ↑ stream shading, groundcover, hydric species • ↓ rate of spread of noxious weeds, cheatgrass • ↑ habitat quality and quantity for salmonids and amphibians • ↑ riparian wildlife habitat • ↓ fecal coliform in streams, lakes, and reservoirs • ↓ nutrient pollution in streams, lakes and reservoirs • All effects VHC 	<ul style="list-style-type: none"> • None --VHC 	<ul style="list-style-type: none"> • ↓ range admin., monitoring, fencing, and mitigation costs • ↑ effectiveness per unit effort of noxious weed control • ↓ storage loss in reservoirs • ↑ useful reservoir life, water supplies in reservoirs

Notes: Numbers in bold are estimates using reasonable assumptions. ↑ signifies an increase in a condition or trend; ↓ signifies a decrease; ≠ signifies possible impeded or thwarted recovery in some areas, with a downturn in others, ≈ signifies trend arrested; symbols in bold signify a strong trend. **VH**=very high need; **H**=high need; **L**=low need; **VL**=very low need, not recommended. **VHC**=Very high degree of certainty with respect to benefit or risk; **HC**=high certainty; **L**=low certainty; **VLC**=very low certainty or highly uncertain.

Table A-2. Roads-- Summary of scale, restoration needs, benefits and risks, and levels of certainty associated with road obliteration combined with cessation of road construction

Current scale of Activity or Condition	Negative physical effects on watershed condition or function in the absence of restoration approach	Effects on fauna and beneficial uses in the absence of restoration approach	Restoration Attributes			
			Level of need and measures	Ecological Benefits and Level of Certainty	Ecological Risks and Level of Certainty	Economic or Logistical benefits
<p>25,000 mi. of forest development roads . 4000 to 6000 miles of unclassified roads.</p> <p>95,983 to 119,023 road crossings</p>	<ul style="list-style-type: none"> • ↓ ≠ low flows • ↑ soil compaction • ↓ soil productivity • ↑ erosion, sediment delivery, sedimentation, • ↓ pool volume and quality • ↑ peakflows • ↓ ≠ water table elevation, stream incisement, baseflows • ↓ ≠ stream shading, groundcover, hydric species • ↑ noxious weeds • ↑ nutrient pollution streams, lakes and reservoirs 	<ul style="list-style-type: none"> • ↓ habitat quality and quantity for salmonids and amphibians • ↑ storage loss in reservoirs • ↓ useful reservoir life, water supplies in reservoirs • ↑ accelerated eutrophication in lakes and reservoirs 	<p>VH – Elimination of all new road construction, including “temporary” roads</p> <p>VH – Obliteration, especially that targeting roads in riparian areas and/or hydrologically connected to streams</p>	<ul style="list-style-type: none"> • ↑ low flows • ↓ soil compaction • ≈ ↑ soil productivity • ↓ erosion, sediment delivery, sedimentation • ↓ seasonal water temperature extremes • ↑ bank stability, undercut banks • ↑ pool volume and quality • ↓ surface runoff • ↑ water table elevation • ↓ stream incisement, • ↑ stream shading, groundcover, hydric species • ↓ rate of spread of noxious weeds • ↑ habitat quality and quantity for salmonids and amphibians • ↑ riparian wildlife habitat • ↓ fecal coliform in streams, lakes, and reservoirs • ↓ nutrient pollution in streams, lakes and reservoirs • All effects VHC 	<ul style="list-style-type: none"> • None --VHC 	<ul style="list-style-type: none"> • ↓ road construction and maintenance costs • ↑ effectiveness per unit effort of noxious weed control • ↓ storage loss in reservoirs • ↑ useful reservoir life, water supplies in reservoirs

Notes: Numbers in bold are estimates using reasonable assumptions. ↑ signifies an increase in a condition or trend; ↓ signifies a decrease; ≠ signifies possible impeded or thwarted recovery in some areas, with a downturn in others, ≈ signifies trend arrested; symbols in bold signify a strong trend. **VH**=very high need; **H**=high need; **L**=low need; **VL**=very low need, not recommended. **VHC**=Very high degree of certainty with respect to benefit or risk; **HC**=high certainty; **L**=low certainty; **VLC**=very low certainty or highly uncertain.