

Appendix A

We have used footnotes and embedded links to cite all of our research, Appendix A additionally includes physical copies of our cited literature.

Fire Probability, Fuel Treatment Effectiveness and Ecological Tradeoffs in Western U.S. Public Forests

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Abstract: Fuel treatment effectiveness and non-treatment risks can be estimated from the probability of fire occurrence. Using extensive fire records for western US Forest Service lands, we estimate fuel treatments have a mean probability of 2.0-7.9% of encountering moderate- or high-severity fire during an assumed 20-year period of reduced fuels.

INTRODUCTION

Fuel treatments to reduce fire impacts have been promoted as a public forest restoration priority by policy [1] and the Healthy Forests Restoration Act of 2003. It is difficult to generalize about the effectiveness of fuel treatments under all conditions [2, 3], but treatments are not universally effective when fire affects treated areas [4]. Factors influencing effectiveness include forest type, fire weather [4], and treatment method [5].

However, treatments cannot reduce fire severity and consequent impacts, if fire does not affect treated areas while fuels are reduced. Fuels rebound after treatment, eventually negating treatment effects [3, 6]. Therefore, the necessary, but not sufficient, condition for fuel treatment effectiveness is that a fire affects a treated area while the fuels that contribute to high-severity fire have been reduced. Thus, fire occurrence within the window of effective fuel reduction exerts an overarching control on the probability of fuel treatment effectiveness. The probability of this confluence of events can be estimated from fire records. Although this probability has not been rigorously analyzed, it has often been assumed to be high [7].

The probability of future fire occurrence also abets assessing the ecological risks incurred if fuels are not treated. Therefore, analysis of the likelihood of fire is central to estimating likely risks, costs and benefits incurred with the treatment or non-treatment of fuels.

Assessing fire occurrence and its effect on fuel treatment effectiveness also has merit because treatments can incur ecological costs, including negative impacts on aquatic systems [8], soils [7], and invasion by non-native plants [9, 10]. Here, we use watershed and aquatic systems as a specific context for evaluating tradeoffs involved with treatment and non-treatment of fuels on western public lands. However, the analysis applies to upland ecosystems as well.

The effects of fire on watersheds and native fish vary with several biophysical factors, including watershed and

habitat conditions, the condition of affected populations, and fire severity and extent [11]. If treatments reduce the watershed impacts of severe fire, they may provide benefits that outweigh treatment impacts because high-severity fire can sometimes trigger short-term, severe erosion and runoff [12] that can negatively affect soils, water quality, and aquatic populations. However, fuel treatments can also have impacts on aquatic systems. The magnitude and persistence of these treatment impacts vary with treatment methods, location, extent and frequency.

Although some fuel-treatment methods could have lower impacts, ground-based mechanical treatments are often employed because other methods generate activity fuels [7] and are more costly. Ground-based methods and associated machine piling, burning of activity fuels, construction and increased use of roads and landings can increase soil erosion, compact soils, and elevate surface runoff [8, 13, 14]. Although the effects of prescribed fire on watersheds are typically limited and fleeting, it can increase soil erosion and sediment delivery, sometimes significantly and persistently [15], especially if fires escape and burn larger and more severely than planned.

When impacts are extensive, proximate to streams, or in terrain with erosion hazards, treatments can increase runoff and sediment delivery to streams. Road activities that increase sediment production, such as elevated road traffic, often affect stream crossings where sediment delivery is typically efficient and difficult to control [16]. Elevated sediment delivery to streams contributes to water quality degradation that impairs aquatic ecosystems [17].

The extent and frequency of treatments may be significant. Stephens and Ruth [18] suggested treating fuels on 9.4 million ha, or ~53% of USFS lands in the Pacific Northwest and California. Agee and Skinner [7] suggested repeating treatments every 10-20 years, due to transient effects on fuels.

Repeated treatments increase the potential for cumulative effects on aquatic ecosystems due to the persistence and additive nature of watershed impacts over time [19] and may increase the establishment of non-native plants [9]. The chronic watershed impacts from repeated treatments may be more deleterious to native fish than pulsed disturbances from wildfires [8].

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Additional degradation of aquatic habitats on public lands may hamper efforts to protect and restore aquatic biodiversity. These habitats are increasingly important as cornerstones for restoring aquatic ecosystems and native fish [14].

Where fuel treatments might incur soil and watershed impacts, the risks from treatment and non-treatment should be assessed [7]. Although the respective impacts of treatments and fire are influenced by numerous factors, the occurrence of fire strongly affects the net balance between costs and benefits. If fire does not affect treated areas while fuels are reduced, treatment impacts on watersheds are not counterbalanced by benefits from reduction in fire impacts.

We provide a framework for quantitatively bounding the potential effectiveness of fuel treatments and the likelihood of fire affecting untreated watersheds, based on the probability of fire and the duration of treatment effects on fuels. This can be used to help statistically estimate the expected value associated with treatments or non-treatment based on the probability of possible outcomes and their associated costs and benefits [20]. Previous assessments of watershed trade-offs from treatment and non-treatment [21, 22] did not include these in quantifying risk to aquatic systems associated with treatment versus non-treatment of fuels.

We use geographically-explicit data on fire on public lands in the western US to estimate, at a broad-scale, the probability that fuel treatments will be affected by fire during the period when fuels have been reduced. We also estimate the risk of higher severity fire occurring in watersheds if fuel treatments are foregone. These estimates provide a broad-scale bounding of treatment effectiveness and potential return from the fiscal and environmental costs of fuel treatments.

METHODS

The Analytical Model

Our analysis is based on the simple conceptual framework that unless fire occurs while fuels are reduced, fuel treatments cannot affect fire severity. We examine the probability of discrete classes of fire severity because fire impacts on watersheds vary with severity [11]. For instance, lower-severity fire has minimal, transient watershed impacts [11].

Future fire occurrence in specific locations cannot be predicted with certainty, but its probability can be estimated from empirical data. The probability of fire of a particular severity affecting treated areas can be estimated using the standard formula for the probability of an event occurring during a specific time frame:

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

where q is the probability that a fire that would be of a specific severity in the absence of treatment occurs within n years, p is the annual probability of fire of a specific severity at the treatment location, and n is the duration, in years, that treatments decrease fuels and can reduce fire severity. In Equation 1, q provides an estimate of the mean fraction of an analysis area likely to burn at a specific severity within a given time frame in the absence of fuel treatments, which also represents the upper bound of potential effectiveness of

treatments in reducing fire, since treatments cannot lower fire severity unless a fire occurs.

Both n and p can be estimated from available data. The duration of post-treatment fuel reduction, n , likely varies regionally with factors affecting vegetation re-growth rates, but fuels in western U.S. forests generally return to pre-treatment levels in 10-20 years [3, 7]. To estimate the upper limit of treatment effectiveness, we assume $n = 20$ years. We estimated the annual probability of fire of various severities, p , for each analysis area based on standard methods [23]:

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

where p is the annual probability of fire of a specific severity, F is total area burned at any severity within the analysis area over the duration of the data record, r is the estimated fraction of F that burned at the specified severity over the analysis area, A is the total analysis area, and D is the total duration of the data record, in years.

We based our estimates of the annual probability of fire on post-1960 fire records rather than reported natural fire return intervals for two primary reasons. First, evidence indicates that natural fire regimes no longer operate in many forests, because of direct fire suppression and indirect changes in fuels from livestock grazing, logging and fire exclusion [24]. Annual burned area has also increased in some forest types, likely due to climatic warming [25]. Recent fire data ostensibly integrate these alterations, reflecting how fires are likely to burn in the near future under current conditions and management. Natural fire return intervals do not capture these alterations. Second, there is considerable uncertainty regarding the accuracy of reported natural fire intervals [23, 24]. However, we stress that our approach can easily accommodate alternate estimates of annual fire probability using more geographically-refined data or where management changes might alter future fire probability.

We confined analysis to USFS lands in 11 western states, the focus for most proposed fuel treatments on public lands. The probability of fire varies geographically with several factors, including weather, ignition, fuels, and forest types. To bracket this effect, we estimated the annual probability of high-severity fire, p , for (i) all landcover types and (ii) more frequently burning ponderosa pine (*Pinus ponderosa*) forests at the scale of U.S. Forest Service (USFS) administrative regions that are the finest scale at which extensive data allow estimation of fire severity. We focus on high-severity fire, but also analyze fires of broader severity, including (1) either high- or moderate severity and (2) any severity.

Our estimates represent an initial, broad-scale first approximation of the potential of fire to affect areas within a given time frame, based on the assumption that fire and treatments are random. Although fire is not random, data are insufficient to accurately quantify more local patterns. Our approach provides a valid mean result at our scale of analysis, based on data from more than 40,000 fires across the western U.S. Site-specific data could be used in future, local studies where the probability of fire is known to depart considerably from the regional mean. Ideally, fuel treatments may not be randomly located, but instead focused in areas where fire is most likely. However, this is not assured by current policy [26]. Widely used methods for assessing the risk of high-severity fire may have limited accuracy [27].

Therefore, our analysis assumes random treatment location, as a first approximation.

West-Wide Analysis

To provide a broad-scale perspective of potential fuel treatment efficacy, we estimated mean annual probability, p , of fire for all USFS lands in the 11 western U.S. states, excluding Alaska, for the entire duration that data on total annual fire area are available (1960-2006). Data on fire area from 1993-2003, reported by agency ownership [28], were used to estimate mean annual fraction of total fire area on USFS lands, which was extrapolated to estimate mean annual fire area on USFS lands from 1960-1993 and 2004-2006, for which fire area data were reported [29], but not by agency ownership. Annual fire area on USFS lands in the 11 western states was assumed proportional to the fraction of total USFS area in these states. Total number of fires on western USFS lands from 1960-2006 is not reported, but based on the foregoing areal partitioning, the fire area data are from several hundred thousand fires on western USFS lands. The estimated annual fire area on these western USFS lands from 1960-2006 was summed to yield F in Equation 2.

The fraction of total fire area, r , that burned at high severity and high-moderate severity was estimated from data in USFS burned area emergency rehabilitation reports (BAER) for 470 fires in the 11 western states from 1973-1998 in six western USFS regions [30].

Regional Analysis of Fire in Ponderosa Pine

Because ponderosa pine forests are a key forest with more frequent fire, we estimated the mean annual probability of fire by severity in these forests on USFS lands: 1) on a regional basis, in six western USFS regions; and 2) West-wide. We used geographical information system (GIS) data for 40,389 fires in these forests for the entire period of data availability, 1980-2003 (Fig. 1). Data were in a GIS point dataset, containing burned area for each fire, maintained by the Bureau of Land Management [31] and derived from a systematic National database [32]. We quality controlled these data for our study area, removing a few duplicate records.

A GIS map of ponderosa pine forests was obtained by selecting codes 5-7 (ponderosa pine) in the Westgap map from the GAP program, which includes national vegetation mapping from satellite imagery [33]. A GIS map of U.S. Forest Service regions is from the agency [34]. We converted all maps to Albers projection, Clarke 1866 datum, then used these to extract all fire records ($n = 40,389$) for ponderosa pine forests on USFS land in the 11 western states. We used USFS maps to subset fires by region, and then: (i) areas of individual fires were summed to yield F in Equation 2; (ii) the GIS was used to obtain A , and (iii) fire severity data by USFS region from 1973-1998 [30] were used to estimate r by severity.

RESULTS AND DISCUSSION

West-Wide Analysis

For the period 1960-2006, an estimated mean of ~220,000 ha, or a decimal fraction of 0.0037 of USFS western lands burned annually at any severity. Despite the approximations involved, our estimate of the mean annual frac-

tion of areas burning at any severity compares reasonably with independent estimates by falling between them. Fire of any severity annually burned a mean fraction of ~0.0014 of the Deschutes National Forest in Oregon, from 1910-2001 [35], and ~0.0046 of 11 national forests in the Sierra Nevada, California, based on data from 1970-2003 [36].

Together with fire severity data [30], our West-wide estimate yields an estimated mean annual probability, p , of 0.001 and 0.002 for high- and high-moderate severity fire, respectively (Table 1). Based on these estimates of p , Equation 1 yields a probability, q , of 0.020 and 0.042, respectively, for high- and high-moderate-severity fire. Substituting space for time, our results indicate that, on average, approximately 2.0 to 4.2% of areas treated to reduce fuels are likely to encounter fires that would otherwise be high or high-moderate severity without treatment. In the remaining 95.8-98.0% of treated areas, potentially adverse treatment effects on watersheds are not counterbalanced by benefits from reduced fire severity. These results also provide an estimate of the likelihood of high-severity fire affecting forests, if fuels are untreated. On average, over a 20-year period, about 2.0-4.2% of untreated areas would be expected to burn at high or high-moderate severity, respectively.

Using Equation 1, our results indicate that if treatments were repeated every 20 years across all USFS lands in the West, it would take about 720 years (36 cycles of treatments), on average, before it is expected that high-severity fire affects slightly more than 50% of treated areas while fuels are reduced. Treatments would have to be repeated at 20-year intervals for 340 years (17 cycles of treatments) before high-moderate severity fire is expected to encounter more than 50% of treated areas. Even after this duration of repeated treatments, it is likely that almost 50% of treated areas will be cumulatively affected by repeated treatments without compensatory benefits from reduced fire severity.

These West-wide estimates provide perspective, but include forest types, such as subalpine forests, typified by low-frequency, high-severity fire, where fuel treatments are unlikely to encounter fire [4]. Other forests, such as ponderosa pine, burn more often.

Regional Analysis of Ponderosa Pine

For ponderosa pine forests, the probability, q , of treated areas being affected within their window of effectiveness varies regionally from 0.020 to 0.040 for high-severity fires and from 0.042 to 0.079 for high-moderate severity fires (Table 1). As expected, q in these forests is higher than for the West-wide analysis of all cover types. The highest probabilities, as expected, are in the Southwest and in the Northern Rockies, with its dry summers (Table 1).

In these forests with more frequent fire, it is likely that fuel treatments can potentially reduce fire severity on a small fraction of treated areas. The results (Table 1) indicate that in 92.1-98.0% of treated areas, fuel treatment impacts on watershed processes are not likely to be counterbalanced by a reduction in higher-severity fire.

Across the six regions, treatments would have to be repeated every 20 years for 340 to 700 years (17 to 35 times), on average, before it is expected that high-severity fire affects more than 50% of treated areas during periods of treat-

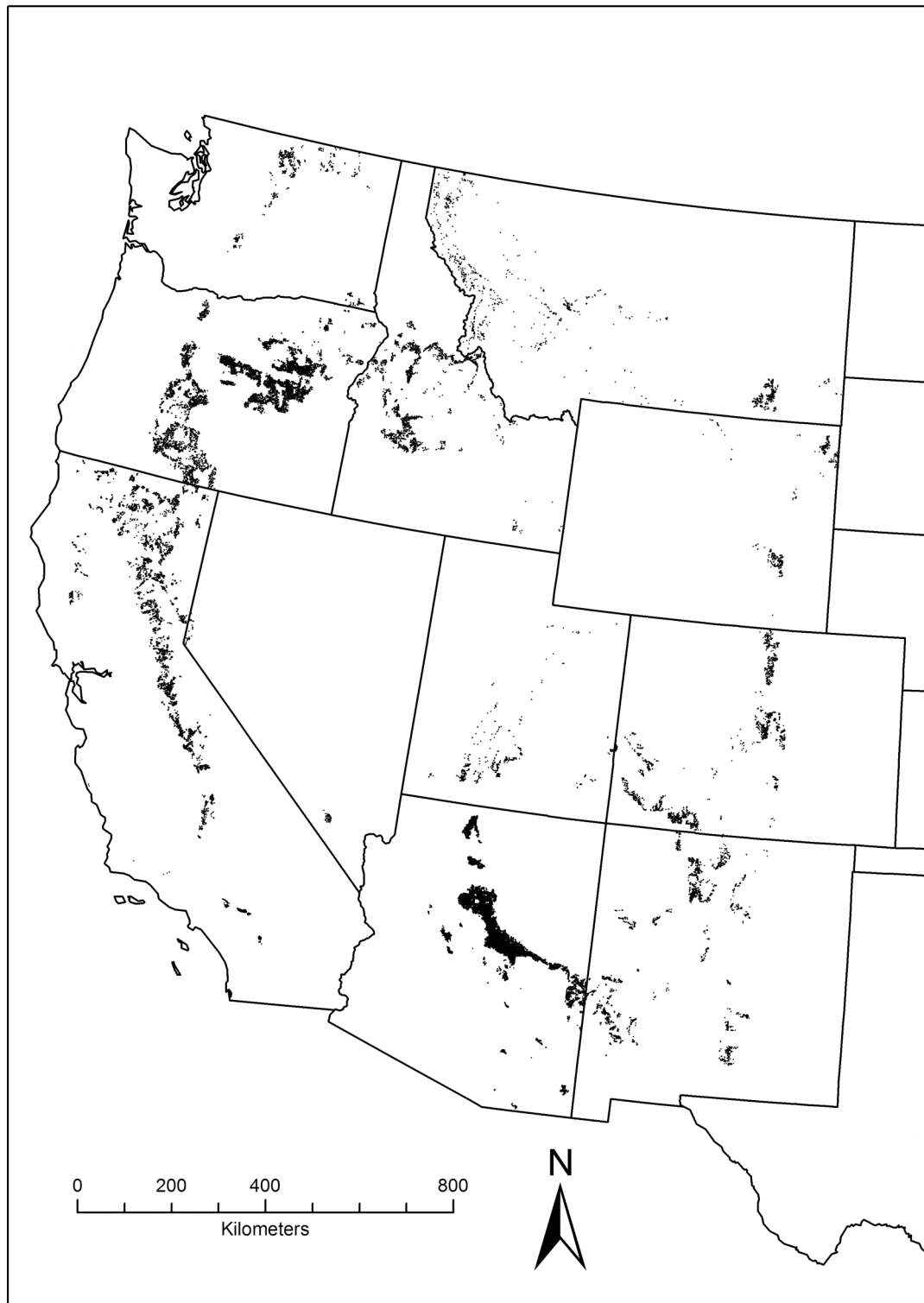


Fig. (1). Ponderosa pine forest fires ($n = 40,389$) in the western United States from 1980-2003. This is the dataset used in the regional analysis.

ment effectiveness. Treatments would have to be repeated for 180 to 340 years (9 to 17 times) before more than 50% of treated areas are expected to be affected by high-moderate severity fire. On average, these repeated treatments would affect watersheds and, potentially aquatic systems, depending on treatment practices, without providing reduction in fire severity on almost 50% of treated area.

An alternative method for estimating the risk of fire in the absence of fuel treatments is to use the fire rotation rather than mean annual probability of fire. The fire rotation indicates how long it takes, on average, for a particular area to burn one time and how often fire may return to a particular point in the landscape [23]. The fire rotation is calculated by:

$$B = 1/p \quad (3)$$

Table 1. Estimated p and q for Fires in Ponderosa Pine (PIPO) Forests. Data are Shown for Three Fire Severity Classes by USFS Region, and for All Forests on USFS Lands West-Wide

USFS Region	Any Severity		High-Moderate Severity		High Severity	
	p	q	p	q	p	q
1 N. Rockies	0.0070	0.1311	0.0036	0.0693	0.0020	0.0402
2 C&S Rockies	0.0059	0.1116	0.0041	0.0786	0.0014	0.0269
3 SW	0.0053	0.1008	0.0025	0.0487	0.0016	0.0307
4 Gt. Basin	0.0090	0.1654	0.0037	0.0715	0.0013	0.0257
5 Calif.	0.0046	0.0881	0.0031	0.0603	0.0017	0.0338
6 NW	0.0037	0.0715	0.0022	0.0421	0.0010	0.0198
West-wide: PIPO	0.0054	0.1026	0.0031	0.0602	0.0015	0.0295
West-wide: All types	0.0037	0.0715	0.0021	0.0416	0.0010	0.0203

where B is the fire rotation for fire of a specific severity and p is, again, the mean annual probability of fire of a specific severity.

Based on our analysis, the mean annual probability, p , of high-severity fire in ponderosa pine forests West-wide is 0.0015 (Table 1), implying a fire rotation, B of about 667 years, varying from 500 to 1,000 years among individual regions. Based on the results in Table 1, the fire rotation for high-moderate severity fire is about 323 years in ponderosa pine forests West-wide, varying from 244 to 454 years in individual regions, based on data in Table 1. These results suggest that western ponderosa pine forests are not currently being rapidly burned by high or high-moderate severity fire, counter to other previous work [37].

Relaxing the Assumptions and Some Caveats

In some cases, the occurrence of fire of any severity may be of interest. Such cases include areas where fire of any severity might lead to high-severity fire. In ponderosa pine forests, the probability of fire of any severity encountering treatments within 20 years is approximately 7.15-16.5% across the six regions (Table 1). Thus, if it is assumed that fuel treatments that encounter fire of any severity might be effective, the results indicate fuel treatments, on average, would not have the potential to reduce fire impacts on aquatic systems in 83.5-92.8% of the area treated. Based on Equation 1 and Table 1, treatments would have to be repeated every 20 years for 80-200 years, on average, before fire of any severity affects more than 50% of the treated areas in ponderosa forests in these USFS administrative regions.

However, the assumption that treatments that encounter low-severity fire convey benefits may not be warranted. Low-severity fires are commonly and easily extinguished under current management whether or not they encounter fuel treatments. Further, low-severity fire has minimal adverse impacts on watershed processes while conveying benefits, including maintenance of forest structure and fuel levels.

Our probabilistic approach does not explicitly address factors that can strongly influence fire area and severity, such as fuel conditions. Although spatially-explicit modeling of fire behavior can directly investigate the effects of such conditions, such models are unlikely to provide accurate

estimates of the probability of occurrence of fire of a given severity because a host of other factors that influence fire area and severity cannot be deterministically predicted, including the frequency and location of ignitions and weather conditions during fire. Methods of assessing the risk of high-severity fire that are primarily based on fuel conditions have been shown to be an ineffective predictor of the actual severity at which fires burn [38]. In contrast, extensive recent data from numerous fires, as used in our analysis, does provide a robust estimate of the mean probability of the occurrence of fire of a given severity, because it integrates the many factors that influence fire occurrence and severity.

Our estimates likely represent the upper bound for fuel treatment effectiveness at the scale of analysis. In many cases, less than 4.16-7.86% of treated area is likely to experience high-moderate severity fire during the duration of treatment effectiveness, because q decreases with decreases in n , the duration of treatment effectiveness. This duration is often less than the 20 years assumed in our analysis. In the Sierra Nevada of California, fuels returned to pre-treatment levels within 11 years [39]. At the values of p in Table 1, reducing n from 20 to 11 years (Eq. 1) reduces the probability that higher-severity fire affects treatments by ~45%.

Moreover, fuel levels rebound after treatment, eventually negating potential treatment effectiveness. If the reduction in effectiveness over time is such that mean effectiveness over the duration, n , is half the initial degree of effectiveness, the probability that fuel treatments reduce high-severity fire is approximately half the value of q for any value of p and n calculated using Equation 1.

Finally, available data indicate that fuel treatments do not always reduce fire severity when fire affects treated areas while fuels are reduced [4]. Our analysis does not address these effectiveness issues. For these combined reasons, Equation 1 likely estimates the upper bound of potential fuel treatment effectiveness in reducing fire impacts on aquatic systems.

Although our analysis focuses on higher-severity fire in bounding the effectiveness of fuel treatments and their net watershed effects, these fires do not have solely negative effects. Higher-severity fire benefits watersheds and aquatic

ecosystems in several ways, including providing a bonanza of recruitment of large wood and pulsed sediment supply that can rejuvenate aquatic habitats and increase their productivity [8, 14]. High severity fire is also a key process for the restoration of structural heterogeneity in forests, which is important for biodiversity [27, 40].

Our analysis intrinsically assumes some degree of climatic stationarity, which may not be warranted. Climatic variability influences the area annually burned in forests [25, 41]. However, the relatively recent fire data used in our regional analysis incorporates recent climatic fluctuation and possibly directional change, which would not be reflected in estimates based on natural fire return intervals. For instance, the data in our analysis of ponderosa pine forests come primarily from years in which annual fire area had increased due to climatic warming [25]. However, the analysis framework is flexible enough to accommodate projected values of the mean annual probability of fire, p , based on forecasts of climatic change or changes in fire management.

Current findings suggest treatment effects on fire severity are mostly confined to treated areas [3], but theory suggests a dense network of treatments might slow fire spread and reduce intensity, yielding a landscape-scale effect on fire severity [42]. However, empirical evidence of severity reduction was seen in the lee of only three of several dozen treatments in two Arizona wildfires [43]. Nonetheless, if dense treatment networks are shown to work in the future, our approach can aid in estimating their costs and benefits, because fire must still affect treated areas while fuels are reduced for networks to reduce fire severity.

CONCLUSIONS

Our analysis provides West-wide and regional first approximation of the likely upper bound of fuel treatment effectiveness. While valid at these two scales, they are not applicable to all smaller analysis areas, due to spatial variation in annual fire probability. However, the framework is flexible enough to allow more spatially explicit analyses of q where local estimates of n and p are available. The framework allows analysis of uncertainty, by using a range of plausible values for n and p . The analysis can also estimate the number of treatments to reach a specified q , abetting estimation of cumulative effects on ecosystems from repeated treatments.

Our approach also provides a method for quantitatively assessing the imminence of high-severity fire effects in the absence of fuel treatments and the degree of urgency of response. Based on available data, these are shown to be much lower than previously estimated in some work [37].

Our results and analyses can improve the assessment of risks to watersheds inherent in the treatment or non-treatment of forest fuels, because it accounts for the probability of fire and the transient nature of fuel treatments. For instance, previous work [22], evaluating treatment and non-treatment impacts, assessed the risks associated with fuel treatments based on the assumption that a single treatment significantly reduces fire risk on all treated areas, subsequently reducing consequent watershed impacts from fire. Other evaluations of these tradeoffs [21] compared the

erosional effects of fuel treatments with high-severity fire under the explicit assumption that high-severity fire was inevitable without treatment and the implicit assumption that treatments always reduce or eliminate the potential for high-severity fire. Our analysis indicates that these assumptions are unwarranted and likely mischaracterize the outcomes and associated impacts of treatment options.

The approach can be extended to aid in assessing the risk to other ecosystem elements and processes that may be adversely affected by either fuel treatments or high-severity fire. For instance, non-native vegetation can be influenced by high fire severity [44] and some fuel treatments [10], especially if the treatments are repeated [9].

Even in ponderosa pine forests that burn relatively frequently, our regional analysis indicates that after 17 cycles of treatments, only slightly more than 50% of treated areas could potentially have fire severity reduced, on average. Our results indicate that high-severity fire is far from inevitable in areas left untreated and is, instead, expected to affect only a relatively small fraction of such areas at the broad scale of our analysis. Factoring in the probability of fire, using our framework, can significantly improve the assessments of the risks posed to aquatic systems by treating or not treating forest fuels. Where site-specific data on fire probabilities exist, the framework can be used to help locate treatments where they are most likely to encounter higher severity fire, increasing the likelihood of treatment benefits. In fact, our results indicate that such efforts are crucial.

There are several important factors that influence the aquatic tradeoffs among fuel treatments, fire, and aquatic systems that our framework does not address. Although the probability of outcomes is critical to assessing the expected value of options, the ecological costs of the outcomes of treatment vs non-treatment are also important in assessing the expected value of these options. With respect to the aquatic context, there is an ongoing need to fully evaluate tradeoffs such as the severity and persistence of the negative and positive impacts on watersheds and aquatic populations from fuel treatments and higher severity fire [8, 45]. An additional related issue is how effective treatments are when they encounter fire under a broad array of conditions affecting fire behavior [3]. While our analysis does not address these factors, it refines evaluation of net impacts of fuel treatment vs non-treatment by providing a framework for estimating the likelihood of fire occurrence in a given time frame.

At the scales of our analysis, results indicate that even if fuel treatments were very effective when encountering fire of any severity, treatments will rarely encounter fire, and thus are unlikely to substantially reduce effects of high-severity fire.

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Adapt to more wildfire in western North American forests as climate changes

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Wildfires across western North America have increased in number and size over the past three decades, and this trend will continue in response to further warming. As a consequence, the wildland–urban interface is projected to experience substantially higher risk of climate-driven fires in the coming decades. Although many plants, animals, and ecosystem services benefit from fire, it is unknown how ecosystems will respond to increased burning and warming. Policy and management have focused primarily on specified resilience approaches aimed at resistance to wildfire and restoration of areas burned by wildfire through fire suppression and fuels management. These strategies are inadequate to address a new era of western wildfires. In contrast, policies that promote adaptive resilience to wildfire, by which people and ecosystems adjust and reorganize in response to changing fire regimes to reduce future vulnerability, are needed. Key aspects of an adaptive resilience approach are (i) recognizing that fuels reduction cannot alter regional wildfire trends; (ii) targeting fuels reduction to increase adaptation by some ecosystems and residential communities to more frequent fire; (iii) actively managing more wild and prescribed fires with a range of severities; and (iv) incentivizing and planning residential development to withstand inevitable wildfire. These strategies represent a shift in policy and management from restoring ecosystems based on historical baselines to adapting to changing fire regimes and from unsustainable defense of the wildland–urban interface to developing fire-adapted communities. We propose an approach that accepts wildfire as an inevitable catalyst of change and that promotes adaptive responses by ecosystems and residential communities to more warming and wildfire.

wildfire | resilience | forests | wildland–urban interface | policy

Wildfire is a key driver of ecosystem change that increasingly poses a significant threat and cost to society. In western North America (hereafter, the West), warming, frequent droughts, and legacies of past management combined with expansion of residential development have made social–ecological systems (SESs) more vulnerable to wildfire. As the annual area burned has increased over the past three decades, we are confronting longer fire seasons (1, 2), more large fires (3, 4), a tripling of homes burned (5), and more frequent large evacuations. In 2016, the Fort McMurray Fire in Alberta, Canada and the Blue Cut Fire in southern California prompted evacuation orders for a

combined total of more than 160,000 people. The costs of wildfire have also risen substantially since the 1990s. The US Congress appropriated \$13 billion for fire suppression and \$5 billion for fuels management in fiscal years 2006–2015 (6). Other societal costs, including real estate devaluation, emergency services, and postfire rehabilitation, total up to 30 times the direct cost of firefighting (7).

Notwithstanding these costs, many plants, animals, and ecosystem services benefit from fire, and those dependent on frequent fire have been negatively affected by the significantly reduced burning resulting from fire suppression, as compared with the period before European settlement

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(8). However the response of ecosystems to increases in wildfire activity and warming in the coming decades is not well understood. Broad heterogeneity among western forest landscapes in terms of biophysical environment, past management, human footprint, and the role of fire and future warming creates a complicated playing field. Managing ecosystems, people, and wildfire in a changing climate is a complex but critical challenge that requires effective and innovative policy strategies (9, 10).

Our key message is that wildfire policy and management require a new paradigm that hinges on the critical need to adapt to inevitably more fire in the West in the coming decades. Policy and management approaches to wildfire have focused primarily on resisting wildfire through fire suppression and on protecting forests through fuels reduction on federal lands. However, these approaches alone are inadequate to rectify past management practices or to address a new era of heightened wildfire activity in the West (11–14).

In delivering this message, we focus specifically on the distinction between specified, adaptive, and transformative resilience (15, 16). Rigorous definition and critical assessment of resilience to wildfire are needed to develop effective policy and management approaches in the context of climate change. We suggest an approach based on the concept of adaptive resilience, or adjusting to changing fire regimes (e.g., shifts in prevailing fire frequency, severity, and size) to reduce vulnerability and build resilience into SESs. Adaptive resilience to wildfire means recognizing the limited impact of past fuels management, acknowledging the important role of wildfire in maintaining many ecosystems and ecosystem services, and embracing new strategies to help human communities live with fire. Our discussion focuses on western North American forests but is relevant to fire-influenced ecosystems across the globe. We emphasize that long-term solutions must integrate relevant natural and social science into policies that successfully foster adaptation to future wildfire.

Why Has Coping with Wildfire Become Such a Challenge?

Three primary factors have produced gradual but significant change across western North American landscapes in recent decades: the warming and drying climate, the build-up of fuels, and the expansion of the wildland–urban interface (WUI; the zone where houses meet or intermingle with undeveloped wildland vegetation).

In terms of climate, wildfire activity is closely tied to temperature and drought over time scales of years to millennia (2, 17–19). Globally, the length of the fire season increased by 19% from 1979 to 2013, with significantly longer seasons in the western United States (1). Since 1985, more than 50% of the increase in the area burned by wildfire in the forests of the western United States has been attributed to anthropogenic climate change (20). Increases in the number of wildfires and area burned in most forested ecoregions of the West are a result of rising temperatures, increased drought, longer fire seasons, and earlier snowmelt (1–4, 21). Specifically, since the 1970s the frequency of large fires has increased most dramatically in the forests of the Northwest (1,000%) and Northern Rocky Mountains (889%), followed by forests in the Southwest (462%), Southern Rockies (274%), and Sierra Nevada (256%), in response to earlier snowmelt and a longer fire season (21). Based on spatial overlays in western United States forests of large wildfires since 1984 (Monitoring Trends in Burn Severity, available at www.mtbs.gov/dataaccess.html and Existing Vegetation Types, available at <https://www.landfire.gov/vegetation.php>), we found that in northern regions with dramatic increases in fire activity (the Canadian Rockies, Middle Rockies, and Idaho Batholith ecoregions) cold/wet subalpine forests predominantly burned. These forests characteristically burn at high severity and have not experienced a significant build-up of fuels. Overall, cold/wet forests account for about a quarter of total forest burning in the US West since 1984.

Fire suppression, in addition to past logging and grazing and invasive species, has led to a build-up of fuels in some ecosystems, increasing their vulnerability to wildfire. For example, drier, historically open coniferous forests in the West (“dry forests”) have experienced gradual fuels build-up in response to decades of fire suppression and other land-use practices (8, 22, 23). Historically, predominantly frequent, low-severity fires killed smaller, less fire-resistant trees and maintained low-density dry forests of larger, fire-resistant trees. Large, high-severity fires now threaten to convert denser, more structurally homogeneous dry forests to nonforest ecosystems, with attendant loss of ecosystem services (24). However, only forests in the Southwest show a clear trend of increasing fire severity over the last three decades, and only a quarter to a third of the area burned in the western United States experienced high severity during that time (25, 26). Although fuels build-up in dry forests can increase the area burned because of higher contagion, the 462% increase in the frequency of large fires in southwestern forests since the 1970s is also a result of an extension of the fire season by 3.6 mo [the average for the western United States is 2.8 mo (21)]. Overall, dry forests account for about half of the total forest burning in the western United States since 1984.

Alongside these increases in warming and fuels, the WUI has expanded tremendously in the past few decades, augmenting wildfire threats to people, homes, and infrastructure. Between 1990 and 2010, almost 2 million homes were added in the 11 states of the western United States, increasing the WUI area by 24% (27). Currently, most homes in the WUI are in California (4.5 million), Arizona (1.4 million), and Washington (1 million) (27). Since 1990, the average annual number of structures lost to wildfire has increased by 300%, with a significant step-up since 2000 (28). About 15% of the area burned in the western United States since 2000 was within the WUI, including a 2.4-km community protection zone, with the largest proportion of wildfires burning in the WUI zone in California (35%), Colorado (30%), and Washington (24%) (Fig. 1) (27). Additionally, almost 900,000 residential properties in the western United States, representing a total property value more than \$237 billion, are currently at high risk of wildfire damage (29). Because of the people and property values at risk, WUI fires fundamentally change the tactics and cost of fire suppression as compared with fighting remote fires and account for as much as 95% of suppression costs (28). Together, these gradually changing variables—climate change, fuels build-up, and residential development—interact with rapid combustion to increase wildfire risks and costs to society and some ecosystems substantially.

Potential Consequences of Future Wildfire

Wildfire activity is predicted to increase in the West over the next century (20, 30, 31). This anticipated ramp-up in burning and possible directional changes in fire regimes (e.g., increases in fire frequency, severity, and/or size) could transform the composition, structure, and function of many forest (8, 32, 33), shrubland, and grassland ecosystems (34). Changes in temperature and precipitation in semiarid shrublands and grasslands may reduce fuel availability subsequently, to the extent that fire occurrence, size, and severity in such areas will eventually decline (35). Thus, although fire activity is projected to increase in the West in the near term (i.e., in the next few decades), longer regional trends will depend on feedbacks between vegetation and fire as well as on anthropogenic alterations in vegetation and land use (36, 37).

Increased exposure of communities to wildfire is also expected with additional warming. More than 3.6 million ha, or almost 40% of the current WUI in the western United States, is predicted to experience moderate to large increases in the probability of wildfire in the next 20 y (Fig. 2). This increase is in addition to the growing wildfire risk to developed nonurban areas (e.g., energy production) and infrastructure (e.g., power lines, pipelines) that define a broader wildland–development

Wildfire and the Wildland-Urban Interface (WUI) 2000-2016

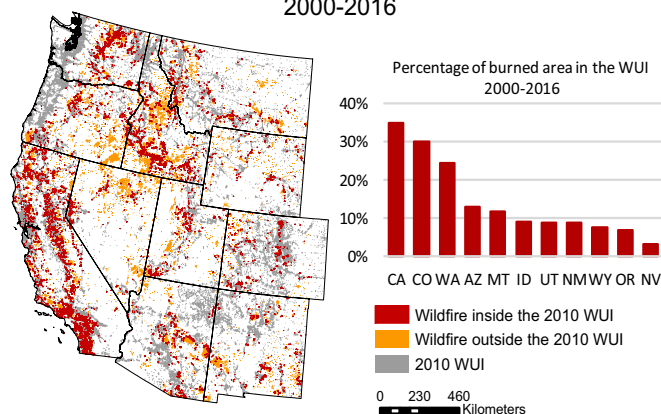


Fig. 1. (Left) Area burned by wildfires between 2000 and 2016 across the western United States inside and outside the 2010 WUI including a 2.5-km community protection zone (27). **(Right)** About 15% of the WUI burned during this period, with largest proportions of the WUI burning in California, Colorado, and Washington.

interface. Continued WUI growth will further increase human exposure to wildfires (38) and anthropogenic ignitions (37, 39). By midcentury, 82 million people in the western United States are likely to experience more and longer “smoke waves,” defined as consecutive days of high, unhealthy particulate levels from wildfires (40). Climate change and increasing exposure of existing and future development to wildfire and smoke present a dangerous and vexing problem for residents, local officials, fire fighters, and managers.

Gradual but significant changes in climate, fuels, and the WUI affect wildfire impacts on ecosystems and society but are difficult to recognize and are challenging to alter meaningfully. There often is a lack of political will to implement policies that incur short-term costs despite their long-term value or to change long-standing policies that are ineffective. For example, few jurisdictions have the will or means to restrict further residential development in the WUI, although modifying and curtailing residential growth in fire-prone lands now would reduce the costs and risks from wildfire in the long term. Furthermore, although the impacts of fire suppression on fuels build-up are now well understood, fire-suppression policies still dominate current fire management (13). Projected global warming of at least 1.1–3.1 °C in the coming century offers a unique opportunity to change policy and the course of our response to wildfires (41). A paradigm shift now in approaches to WUI development and management of fire and fuels can yield tremendous benefits to society later.

Specified, Adaptive, and Transformative Resilience to Wildfire

Resilience is increasingly invoked as a guiding principle in strategies that address the social and ecological dimensions of wildfire. The US Forest Service’s National Cohesive Wildland Fire Management Strategy (42) specifically addresses the need to bolster social and ecological resilience to increasing wildfires. Although often invoked in wildfire management and policy, resilience is defined inconsistently or neglects social or ecological contexts, despite the need for uniformity and specification in setting goals and evaluating progress (43, 44).

Defining resilience to wildfire in an SES is especially challenging in the WUI, where people, ecosystems, and wildfire interact over multiple spatial and temporal scales (12). An SES is the intersection and interdependence of biophysical units and associated people and institutions. Resilience in an SES generally has been defined as the capacity to absorb disturbance so as to retain essential structures, processes, and feedbacks and to adapt to and reorganize following disturbance (45).

These perspectives of resilience, absorbing versus adapting to disturbance, offer different guiding principles for policy and management in responding to wildfire and measuring success over different planning timelines (44). Here we outline a consistent framework that defines resilience to wildfire in coupled SESs based on the concepts of specified resilience and general resilience, the latter of which includes adaptive and transformative approaches (Table S1) (15, 16, 44).

When climate trends or disturbance regimes are relatively stable and well-characterized and planning horizons are short (years), specified resilience or restoration is an appropriate guiding principle. “Specified resilience” refers to the buffer capacity of a system to retain its identity after a well-specified disturbance (16). Specified resilience reflects the concept of ecological resilience, which refers to the capacity of a system to absorb or tolerate disturbance without shifting to a qualitatively different state controlled by a different set of processes (46). In terms of wildfire, specified resilience applies when fire characteristics are within the bounds of historical range of variability (HRV) of disturbance regimes and a burned forest recovers without converting to another state, e.g., to a nonforest state such as a persistent grassland. In a social context, specified resilience is evident when a community recovers economically and rebuilds similar structures in similar locations following a wildfire (44, 47). Management guided by specified resilience often values recent ecological and social dynamics, particularly when the goal is the conservation of particular species or landscapes. Such management is often informed by short temporal windows of HRV, or “recent HRV” (rHRV) (Fig. 3). This approach can be useful for responding to fires in the short term. However, when social and environmental conditions change rapidly, this approach may foster management goals that are unrealistic or unsustainable in the long run (48, 49).

When climate and wildfire trends are changing and planning horizons are intermediate (decades), general resilience is a more appropriate and desirable guiding principle. “General resilience” refers to the capacity of an SES to adapt or transform in response to unknown shocks or disturbances outside the rHRV (16). Adaptive resilience incorporates aspects of change, reorganization, learning, and adaptability in response to changing climate and disturbance regimes and is an on-going process achieved by harnessing adaptive capacity. In an ecological context, adaptive resilience refers to actively or passively supporting species compositions and fuel structures that are better adapted to a warming, drying climate with more wildfire. Management of specified resilience maintains ecosystems within the rHRV,

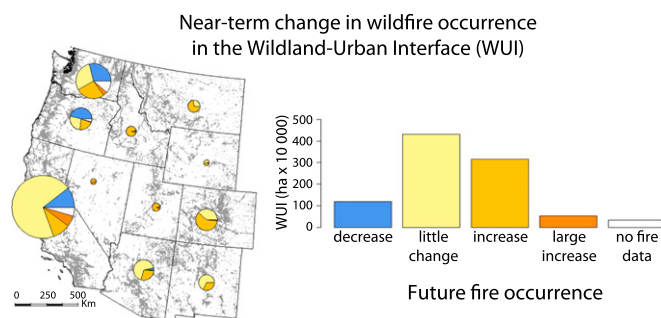


Fig. 2. (Left) Area of the WUI in the conterminous western United States, classified according to projected near-term changes in fire occurrence. The size of each pie is scaled relative to the area of the WUI (both intermix and interface) in each state, based on data from Martinuzzi, et al. (27). Within each pie, slices represent the proportion of WUI area overlapping the five categories of projected fire occurrence for the period 2010–2039, based on data from Moritz, et al. (30). **(Right)** The bar chart summarizes the area of the WUI projected to experience each level of change in fire occurrence in the western United States.

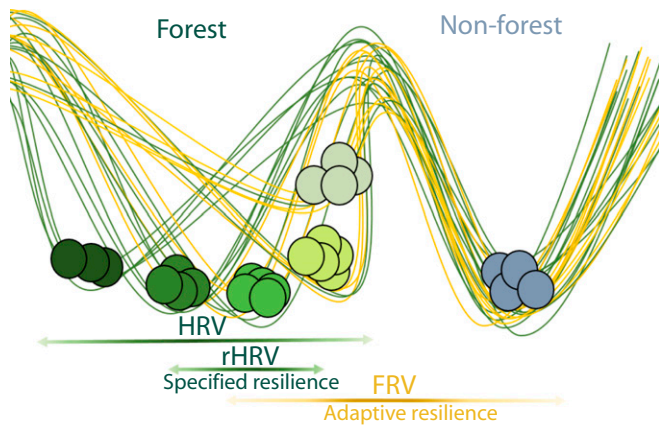


Fig. 3. Conceptual ball-and-basin representation of specified and adaptive resilience across a forested landscape. Lines defining basins depict the ranges of variation in fire regimes across forest types. Sets of green balls reflect the variation in abundance and composition within different forest types, and the set of blue balls represents nonforest ecosystems. Specified resilience of forests to wildfire is maintained within basins that fall within an rHRV of fire regimes over recent decades to centuries, typically derived from historical documents, remotely sensed data, and tree-ring data. Longer definitions of HRV reflect variation in fire regimes over the last 4,000–5,000 y, when present-day forest types were established in most regions; these data are derived from paleoecological reconstructions. Adaptive resilience to changing fire regimes is reflected within basins that fall within the FRV (yellow). Under the FRV, shifts to nonforest ecosystems remain unlikely in some cases (lower green balls) and more likely in other cases with easier transition to nonforest basin (higher green balls). Changes in the severity, frequency, and size of fire regimes and long-term regeneration following fire events reflect adaptive responses to changing fire regimes and climate conditions across broad scales.

whereas managing for adaptive resilience considers how changing disturbance regimes may favor suites of traits that are better adapted to a future range of variability (FRV) (Fig. 3) (22). Alignment of fire regimes with adaptive regeneration traits of native vegetation defines a safe operating space (50). The HRV can still play a role by providing insight into how adaptive traits align with changing disturbance regimes to confer adaptive resilience, but under the FRV the safe operating space is shifting (Fig. 3) (50, 51, 52). In a social context, communities exhibiting adaptive resilience engage in ecological, psychological, social, and policy processes that set the community on a trajectory of change to reduce future vulnerability (Fig. 4) (53). Strategies may include changing building codes to make structures more fire-resistant, planning communities to avoid or withstand future wildfire, or providing incentives, education, and resources to reduce vulnerability to future wildfire (47). Adaptive resilience also involves institutional learning, where past management approaches to wildfire evolve.

When climate and wildfire trends are significantly altered from historical trends and/or variability, and planning horizons are long (century), transformative resilience may be necessary. “Transformative resilience” refers to planned fundamental change in response to drastically altered disturbances that have the potential to create broad-scale, systemic shifts in ecological states or radical shifts in values, beliefs, social behavior, and multilevel governance. Examples might include significant regional changes in ecosystem states and associated loss of ecosystem services and/or the relocation of communities of people away from wildfire-prone areas (44, 54). Rapid, planned social-ecological transformation is rare and difficult to implement because of uncertainties about future risk, inflexible institutions and behaviors, and the high cost of transformative action (55).

Although distinct, these approaches to resilience may be nested. Promoting specified resilience may make some forests better poised for adaptive resilience as climate changes, but in some forests or conditions specified resilience may not be effective as climate changes (e.g., refs. 56, 57). Allowing postfire shifts from forest to grassland or shrubland may increase adaptive resilience to changing wildfire and climate conditions. Approaches to adaptive resilience could reduce the need for transformation if efforts keep pace with climate and wildfire trends or may help pave the way toward inevitable social-ecological change. Embracing specified resilience may be the easiest, most familiar path with the least uncertainty, but this approach is short-sighted and could come at the cost of adaptation to future wildfire as climate change continues.

Taking an adaptive resilience approach now is critical, because specified resilience, although useful in some contexts, will become a less useful guiding principle as we exceed HRVs. Adaptive resilience means adjusting to changing fire regimes and climate—in both social and ecological systems—by taking advantage of opportunities to moderate potential impacts and cope better with the consequences. Adapting to wildfire sooner rather than later provides the widest benefits to society at the least cost. If we do not adapt to wildfire now, disruptive and unintended transformations of SESs in the West may ensue.

How Policy and Management Can Promote Adaptive Resilience to Wildfire

Current approaches to managing wildfire focus primarily on controlling fire through suppression and secondarily focusing on managing fuels build-up in forests. Within the context of current and future trends in wildfire, we evaluate the following three approaches in terms of their promise for fostering adaptive resilience in ecosystems and residential communities living with more wildfire: (i) managing fire, (ii), managing fuels, and (iii) promoting adaptive capacity (Fig. 5).



Fig. 4. Wildfires are catalysts of change that promote adaptive resilience by communities and ecosystems to future wildfires. (A and B) Example of adaptation in communities. (A) A home burned in the 2010 Fourmile fire, Boulder County, CO, which at the time was the most destructive fire in Colorado history in terms of home loss. (B) A home that survived the 2016 Cold Springs fire, where many residents managed structural and vegetative fuels around their home to reduce fire hazard after the Fourmile fire through Boulder County’s Wildfire Partners program. (C and D) Heterogeneity in wildfire severity promotes diversity in postfire regeneration and fuels in the 2002 Rodeo-Chediski fire, Coconino and Navajo counties, AZ (C) and the 2016 Canyon Creek fire, Grant County, OR (D). Photographs courtesy of REUTERS/Alamy Stock Photo (A), Wildfire Partners (B), Tom Bean/Alamy Stock Photo (C), and M.A.K. (D).

Managing Wildfire

Suppressing Fewer Fires and Prescribing More Burning. Increasing the use of prescribed fires and managing rather than aggressively suppressing wildland fires can promote adaptive resilience as the climate continues to warm. Many dry forests currently experience significantly less burning than in the period just before European settlement (8, 35, 58). In recognition of the fire-dependence of many ecosystems, the 1995 Federal Wildland Fire Management policy ushered in the first federal policy aimed at reintroducing more wildfire on public lands; that policy remains in effect today. US federal agencies actively managed an average of 75,000 ha of lightning-caused fires per year under the Wildland Fire Use policy from 1998–2008 and currently burn about 1 million hectares per year with prescribed fires (58). However, prescribed fires still constitute only about 10% of the treatments implemented by the US Forest Service in the West and burn about one-third of the area burned by wildfires (National Interagency Fire Center, <https://www.nifc.gov/>). In the United States and Canada, suppression remains the primary approach to wildfire, with more than 95% of all wildfires suppressed (28). Continued aggressive fire suppression is counterproductive to building adaptive resilience to increasing wildfire in the long term (13, 14).

Using Fire to Foster Adaptive Resilience to Climate Change. In some systems, fire today attenuates future fire effects, because flames that burn dead and live fuel limit where and how severely subsequent fires burn, at least for a time (59–61). Fires often create complex patterns of burn severity that create variation in postfire regeneration and fuels (62–67). As fire regimes shift over time, individual fire events filter for species adapted to changing fire and climate conditions (68). Strategic planning for more managed and uncontrolled wild fires on the landscape today (69) may help decrease the proportion of large and severe wildfires in the coming decades and may enhance adaptive resilience to changing climate. Prescribed fires, ignited under cooler and moister conditions than are typical of most wildfires, can reduce fuels and minimize the risk of uncontrolled forest wildfire near communities. In contrast to wildfires, prescribed fire risks are relatively low, and more than 99% of prescribed fires are held within planned perimeters successfully (58).

Challenges to increasing use of managed and prescribed fires vary from the public's limited experience with smoke and wildfire to significant direct health impacts of smoke on vulnerable populations, including children, the elderly, and low-income communities (40, 70, 71). Some smoke hazards can be reduced through careful planning and management of fire, public health monitoring, and provisioning of health services for vulnerable populations. Public perceptions of fire are also an important hurdle, given the success of Smokey Bear's fire-

prevention campaign and because most urban and suburban residents have very limited experience with wildfire compared with rural residents of the early 20th century. Therefore, public education programs that demonstrate the inevitability of wildfire will be a key aspect of living with increasing fire in the West. We need to develop a new fire culture. Despite these and various legal and operational challenges (58), the benefits of prescribed fire and managed wildfires to ecosystems and communities are high (72). Promoting more wildfire away from people and prescribed fires near people and the WUI are important steps toward augmenting the adaptive resilience of ecosystems and society to increasing wildfire.

Managing Fuels

Limiting Reliance on Fuels Treatments to Alter Regional Fire Trends. Managing forest fuels is often invoked in policy discussions as a means of minimizing the growing threat of wildfire to ecosystems and WUI communities across the West. However, the effectiveness of this approach at broad scales is limited. Mechanical fuels treatments on US federal lands over the last 15 y (2001–2015) totaled almost 7 million ha (Forests and Rangelands, <https://www.forestsandrangelands.gov/>), but the annual area burned has continued to set records. Regionally, the area treated has little relationship to trends in the area burned, which is influenced primarily by patterns of drought and warming (2, 3, 20). Forested areas considerably exceed the area treated, so it is relatively rare that treatments encounter wildfire (73). For example, in agreement with other analyses (74), 10% of the total number of US Forest Service forest fuels treatments completed 2004–2013 in the western United States subsequently burned in the 2005–2014 period (Fig. 6). Therefore, roughly 1% of US Forest Service forest treatments experience wildfire each year, on average. The effectiveness of forest treatments lasts about 10–20 y (75), suggesting that most treatments have little influence on wildfire. Implementing fuels treatments is challenging and costly (7, 13, 76, 77); funding for US Forest Service hazardous fuels treatments totaled \$3.2 billion over the 2006–2015 period (6). Furthermore, forests account for only 40% of the area burned since 1984, with the majority of burning in grasslands and shrublands. As a consequence of these factors, the prospects for forest fuels treatments to promote adaptive resilience to wildfire at broad scales, by regionally reducing trends in area burned or burn severity, are fairly limited.

Targeting Fuels Treatments in Ecosystems with Fuel Build-Up and on Private Lands. Strategically targeting treatments in areas where fuels build-up has increased the expected burn severity may augment the adaptive resilience of those ecosystems to increasing wildfire. For example, treating drier forests, where the likelihood of fire is

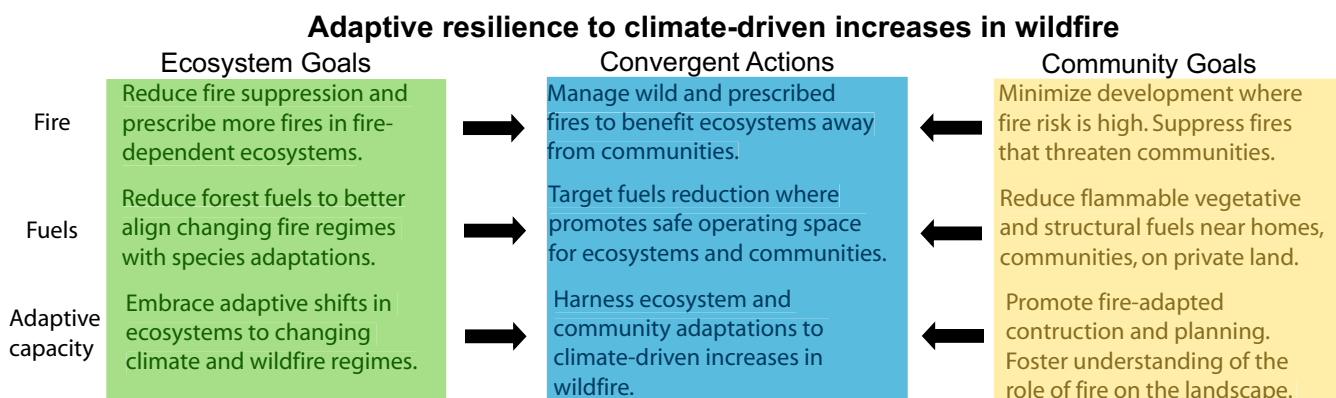


Fig. 5. Convergent actions that promote adaptive resilience to climate-driven increases in wildfire in the West by ecosystems and communities, based on goals related to management of fire, fuels, and adaptive capacity.

high, may also increase opportunities to modify wildfire behavior and postfire recovery. Burn severity has increased because of past fire suppression and fuels build-up in low-elevation dry forests adapted to predominantly frequent, low-severity surface fires (8, 11, 22, 25, 78, 79). In these forests, fuels treatments that remove midstory and understory fuels through thinning and prescribed fire can reduce fire intensity, severity, and rate of spread and may promote adaptive resilience to more frequent fire. Such forests were preferentially treated under the National Fire Plan in 2004–2008 (80). Thinning may effectively restore more frequent, low-severity fire in some dry forests, but when thinning is combined with the expected warming, unintended consequences may ensue, whereby regeneration is compromised and forested areas convert to nonforest (56, 57, 81). Strategic placement of treatments to promote low-severity fire at ecotones between dry and mesic forest distributions may help facilitate postfire migration of species better adapted to warmer, drier conditions.

Mid-elevation mixed conifer forests, or mesic forests, which typically experienced broad variance in fire frequency and severity, may also benefit from fuels treatments that reduce the likelihood of large patches of high-severity fire and facilitate the migration of species adapted to drier, warmer conditions (77). In contrast, cold/wet forests, such as high-elevation subalpine forests, are adapted to high-severity fire that historically recurred at relatively long (~100–300 y) intervals (19, 82, 83) and have not experienced unprecedented fuels build-up in recent decades. Severe wildfires have occurred for millennia across a broad range of forests and shrublands, and in many ecosystems species are adapted to severe fire (17, 19, 84, 85), although postfire regeneration may be comprised by drier, warmer conditions (86).

Fuel-reduction treatments also hold promise for locally reducing wildfire hazard around WUI communities if treatments are strategically located to protect homes and the surrounding vegetation. Fuel reduction on federal lands and in municipal watersheds is a primary management tool that has limited application in the WUI, where the majority of land is

privately owned (87). Home loss to wildfire is a local event, dependent on structural fuels (e.g., building material) and nearby vegetative fuels (88, 89). Therefore, fuels management for home and community protection will be most effective closest to homes, which usually are on private land in the WUI where ignition probabilities are likely to be high (37). Programs that facilitate the targeted removal of fuels from private land, such as community chipping programs, have been highly successful in some areas, at relatively low cost. The Wyden and Good Neighbor authorities and federal programs, such as the US Forest Service Collaborative Forest Landscape Restoration Program, take an “all-lands” approach to forest management through collaboration with landowners and communities. These policies and programs are roadmaps for augmenting fuel-management efforts across land ownerships. These and other more ambitious policies that facilitate significant fuels management on private land, on a par with fuel-reduction efforts on federal lands, are needed. New policies that facilitate private-land fuels management are critical to augment significantly the adaptive resilience of communities to increasing wildfire.

Promoting Adaptive Capacity

Fostering and Embracing Adaptive Shifts in Ecosystems.

Management of fire and fuels will help some ecosystems withstand more frequent fires and possibly may reduce the risk of larger, more severe fires that may compromise forest recovery. Such efforts will be significant in high-value ecosystems or locations, in helping slow the pace of change and providing a chance for ecosystems and species to adapt to changing fire regimes. The HRV concept can guide management in identifying ecological vulnerabilities and adaptation strategies to changing disturbance regimes (Fig. 3) (50, 51, 52). However, quantifying ecological objectives outside the HRV will be increasingly important in guiding management as fire regimes and climate continue to change (90, 91). Given such uncertainties, management must be adaptive and iterative, and monitoring will be critical to assessing progress. Given the vast area of fire-prone forests in the West, management can directly affect only a small portion of forests. In the majority of forested ecosystems beyond our effective reach, we will have to accept and even embrace changing ecological conditions. While some forests may be entering decades of significant change with high tree mortality in response to drought, wildfire, insect outbreaks, and legacies of past management (86, 92), they also are in the process of adjusting to new conditions to which they will be better adapted and that may challenge our existing philosophies of and approaches to conservation.

Creating Fire-Adapted Communities. The majority of home building on fire-prone lands occurs in large part because incentives are misaligned, where risks are taken by homeowners and communities but others bear much of the cost if things go wrong. Therefore, getting incentives right is essential, with negative financial consequences for land-management decisions that increase risk and positive financial rewards for decisions that reduce risk. For example, shifting more of the wildfire protection cost and responsibility from federal to state, local, and private jurisdictions would better align wildfire risk with responsibility and provide meaningful incentives to reduce fire hazards and vulnerability before wildfires occur. Currently, much of the responsibility and financial burden for community protection from wildfire falls on public land-management agencies. This arrangement developed at a time when few residential communities were embedded in fire-prone areas. Land-management agencies cannot continue to protect vulnerable residential communities in a densifying and expanding WUI that faces more wildfire (12). The US Government Accountability Office questioned the US Forest Service’s prioritizing protection of WUI communities that lie under private and state jurisdictions and has argued for increased financial responsibility

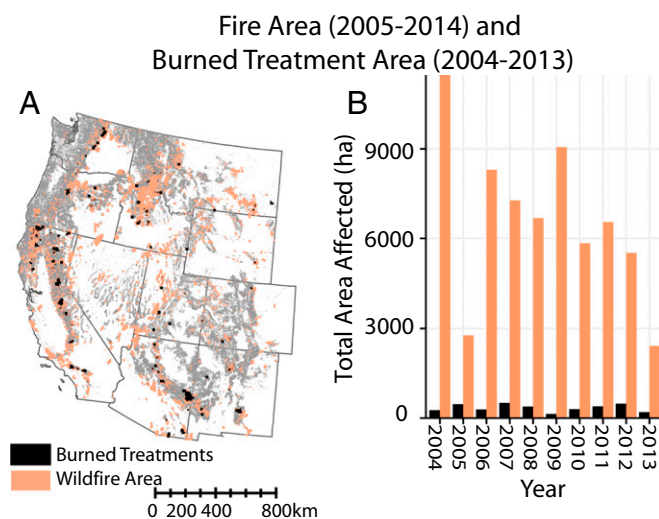


Fig. 6. (A) Spatial distribution and area of US Forest Service fuels treatments from 2004–2013 and wildfire from 2005–2014 across forests and woodlands in the western United States. About 3% of the total treated area and 10% of the total number of treatments burned in the period 2005–2014. **(B)** Annual total wildfire area and total burned treatment area. Data are from the following: (1) US Forest Service fuels treatments: Hazardous Fuel Treatment Reduction Polygon (<https://data.fs.usda.gov/geodata/edw/datasets.php>), (2) Wildfires > 1000 ac: Monitoring Trends in Burn Severity Burned Areas Boundaries (www.mtbs.gov/dataaccess.html), (3) Wildfires ≤ 1000 ac: GeoMAC Historic Fire Perimeters (https://rmgsc.cr.usgs.gov/outgoing/GeoMAC/historic_fire_data/).

for WUI wildfire risk by state and local governments (93). This shift in obligation would enhance adaptive governance and could increase the motivation to pursue adaptive resilience of WUI communities to increasing wildfire (94).

Another promising approach for increasing adaptive resilience of WUI residents to wildfire is the promotion of fire-adapted planning in communities. Providing incentives for counties, communities, and homeowners to plan fire-safe residential development for both existing and new homes and discouraging new development on fire-prone lands will make communities safer (89, 94–96). Communities can use land-use and development codes that encourage developers to set aside open space and recreational trails as fuel breaks and require ignition-resistant construction materials in fire-prone settings. For example, San Diego, California enforces strict brush management regulations; the Flagstaff, Arizona fire department uses a WUI development code to protect properties; and Santa Fe, New Mexico applies stringent fire-safe regulations on new developments to protect its watershed (97). Programs such as the Community Planning Assistance for Wildfire (CPAW; planningforwildfire.org), funded by the US Forest Service and private foundations, offer assistance to communities in the form of advice on land-use planning and detailed mapping of wildfire risk. Another example is California, which employs a statewide Fire Hazard Severity Zone map to guide development plans and building codes that reduce wildfire risk. With 84% of potential WUI lands in the West still undeveloped (98), land-use planning now has high potential to reduce the vulnerability of communities to future wildfire. Furthermore, fire-adapted planning may increase management options in terms of how, where, and when fire can be used as a tool for reducing the spread of wildfires into communities and rejuvenating fire-dependent ecosystems, thus increasing the adaptive resilience of communities and ecosystems to more wildfire.

Strengthening and expanding programs such as Fire Adapted Communities, Fire Adapted Communities Learning Network, Firewise Communities USA, and FireSmart Canada will also help communities become more fire-adapted. Capacities to assume these responsibilities will vary significantly among homeowners, communities, and local jurisdictions with markedly different risks and resources (99–101). For example, home hazard mitigation programs and community planning tools are more successful in communities at the fringe of urban areas that have more financial resources and often have a greater trust in government than in more isolated, resource-dependent WUI communities, immigrant non-English-speaking communities, or tribal and First Nations communities (101). Although some tax incentives and rebates are available for wildfire risk mitigation on and around homes, more comprehensive programs that include broader incentives and support are needed for meaningful and widespread impacts. Efforts

that combine wildfire-specific efforts with other community capacity-building efforts may leverage the networks that enable communities to act on shared notions of risk (102).

Overall, a shift in resources from the defense of the WUI from wildfire to the mitigation of wildfire hazards and risks in advance of events will build a safe operating space for fire-prone communities that increases adaptive resilience to wildfire. Encouraging development away from fire-prone areas, reducing fuels on private lands in and near communities, and retrofitting and building homes to withstand ignition will increase the adaptive capacity for managing more wildfire (89), similar to adaptive approaches for other natural hazards such as flooding and earthquakes (12). Communities and institutions are long-lived, and disruptive events such as wildfires create windows of opportunity that can shift rules, norms, and expectations to increase adaptive resilience to future wildfires.

Conclusions

Policies that foster adaptive resilience enable WUI communities and fire-prone ecosystems to adjust to increased wildfire risk and reduce future vulnerability. Adaptive resilience provides a realistic framework as the climate warms and wildfires increase, but how will we know if we are achieving adaptive resilience to future fires? On the societal front, minimizing the costs of suppression in the WUI, the number of homes lost to wildfire, the area burned in the WUI, and the number of smoke-related health problems are some metrics. Developing state- or county-wide maps of fire hazard, home survivability rating, and the adaptive capacity of communities would be useful tools in developing this framework.

Some ecosystems will survive and thrive as they adapt to novel future conditions, but not all will. Embracing rather than resisting ecological change will require a significant paradigm shift by individuals, communities, and institutions and will challenge our conservation philosophies. Wildfire is an important catalyst of responses to climate change by communities and ecosystems. Patterns of wildfire are changing with rising global temperatures, and will accelerate in the future. What we can do now is focus management efforts on the places where intervention is needed to slow the pace of change and thereby give particular species and ecosystems a chance to adapt. We also can change how we build, live, and work in fire-prone landscapes to keep our communities safe, healthy, and vibrant.

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Working from the Home Outward:

Lessons from California for Federal Wildfire Policy



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p. 7 figure 1a photo (Paradise, CA) by Trip Jennings

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Executive Summary

- The dominant current approaches to wildfire policy promoted by the Forest Service— wildlands fire suppression and forest alteration through extensive tree-cutting—are failing to keep the public safe, with record losses of lives and homes during wildfires.
- In contrast, an approach of “working from the home outward,” focusing on fire-safety home retrofits and the zone immediately around houses and communities, offers the most effective and cost-efficient tools for increasing public safety during wildfires.
- Most wildfire-related government funding subsidizes forest alteration and fire suppression rather than home-outward actions, even in “all of the above” approaches to fire policy. Less than 4% of California’s 2021 fire-related budget is for “community hardening.”
- A new task force is needed to answer this central question—Which actions will produce the greatest public safety benefits during wildfires in the most cost-efficient manner?
- This task force should include, but not be limited to: public safety experts, scientists not reliant on Forest Service funding, the insurance industry, fire managers outside of the Forest Service, indigenous representatives, and environmental groups currently challenging Forest Service fire-related projects.
- Mechanized wildfire suppression was developed during the cooler, wetter conditions in the 20th century. This approach is increasingly expensive and ineffective amid the hotter, drier conditions of the 21st century and wind-driven fires.
- Forest-altering approaches to fire policy are often based on erroneous claims that misconstrue the role of mixed-intensity wildfires in western forests.
- Tree-cutting results in significant carbon emissions, whereas wildfire is a comparatively small and short-term source of carbon emissions. Protecting forests from logging via “proforestation” can sequester five times as much carbon as tree-planting does.
- Fire policies that subsidize using trees and other forest material to fuel biomass power facilities are polluting (both for the climate and vulnerable communities), ineffective, and expensive.
- Fire policies for forest ecosystems often are not appropriate in chaparral ecosystems, which can lead to more flammable landscapes, destruction of critical habitat, and ineffective protection of nearby communities.
- While prescribed fire has a role in fire policy, the results of large-scale prescribed fire are likely to be disappointing. It can lead to ecological damage, as well as a ten-fold increase in the total amount of smoke. And prescribed fire is a relatively inefficient tool when compared to managed wildfire, cultural burning, and fire-safety home retrofits.

Video Summary

Some of the contributors to this report were interviewed for the upcoming fire documentary *Elemental*. For a short video featuring excerpts of these interviews, see: <https://vimeo.com/543710055>.

Introduction

A New Direction for Federal Wildfire Policy: Running Faster in the Wrong Direction is Not a Solution

by Douglas Bevington, PhD, Forest Program Director, Environment Now

In the wake of lives lost and communities burned, there is eagerness for more action on wildfire policy. However, *when you are running in the wrong direction, running faster is a problem, not a solution.* Yet, that is exactly what is occurring with wildfire policy in California and the West.

In the 20th century, the dominant approach to wildfire was suppression, attacking fire in remote wildlands. As discussed on pp. 9-10 of this report, *the mechanized approach to wildfire suppression that was developed in the wetter, cooler conditions of the 20th century is increasingly ineffective amid the hotter, drier conditions of the 21st century*, especially during wind-driven events such as the Camp Fire that devastated the community of Paradise in California. Yet, there is now record-level spending on outdated fire suppression approaches and 20th century technologies such as airtankers. The reality is that firefighting airplanes are not able to operate effectively in the windy conditions in which home loss is most likely to occur, and these aircraft were grounded while Paradise burned (see p. 10).

Likewise, in the past quarter century, every time there are wildfires, the Forest Service, the timber industry, and biomass power lobbyists have consistently pushed for more money to subsidize logging under benign-sounding euphemisms such “fuel reduction”, “thinning”, “restoration,” and “forest management,” and some well-intended people have joined this drumbeat. Yet, this forest-altering approach to fire policy is often based on outdated and erroneous claims about wildfire (see pp. 11-12). And, as more money has subsidized cutting trees under fire-safety justifications, we are seeing unprecedented loss of lives and homes.

The forest-altering approach has failed to keep the public safe. Instead, it mainly produces stumps, depleted budgets, and excuses. After each failure, logging proponents conveniently claim that things will be different if they get more money for more cutting. Yet, we now see the fires burning readily in places where they have already done lots of cutting. In fact, some of the fastest and most intense fire spread is occurring in these cut areas (see p. 12).

Beyond being ineffective and even counterproductive, the forest-altering approach puts more carbon into the atmosphere (see pp. 13-15) and more pollution in vulnerable communities (see p. 15). And perhaps biggest harm from *the forest-altering approach is that it diverts attention and resources away from real solutions, such as fire-safety home retrofits that genuinely protect communities during inevitable wildfires* (see pp. 7-8, p. 18) The La Tuna Fire in Los Angeles illustrated the effectiveness of home-safety approach when community-level fire-safety standards *enabled more than 99% of the houses in the path of a large wildfire to survive* amid triple-digit temperatures and high winds (Hanson 2018). In addition to being more effective, *home-safety actions can produce more jobs and better-paying jobs* than an equivalent amount of money spent subsidizing logging and other forest-altering activities (Niemi 2018).

“Working from the Home Outward” or the “All of the Above” Run-Around?

How can federal policymakers ensure that fire policies turn in a more productive direction and avoid repeating the mistakes of the past? This report builds on lessons identified in a 2019 report done in conjunction with the Leonardo DiCaprio Foundation, titled “A New Direction for California Wildfire Policy—Working from the Home Outward.” That report brought together fire and forest carbon experts from partner organizations of the Environment Now foundation—which supports environmental protection in California-- to prepare concise synopses of key points that are often misunderstood or overlooked in fire policy discussions.

It also featured a piece by Dr. Jack Cohen, a retired US Forest Service scientist. Dr. Cohen has been a pioneer in the study of the importance of home features and the zone right around them for preventing home ignitions during wildfires. Despite the significant implications of Dr. Cohen’s research, not nearly enough has been done to incorporate these findings into current fire policies. As one investigative report summarized, *“Cohen thought he had come up with a way to save houses and to let fires burn naturally — he thought it was a win-win. And so in 1999, he presented a paper about his findings at a fire conference in front of people from the Forest Service and state fire agencies. These were people who were in a position to change policies. But Cohen says they were totally uninterested. Cohen’s research implied that basically everything about how the Forest Service dealt with wildfires was wrong”* (99PI 2018). Dr. Cohen’s discoveries provide the cornerstone for a more effective new direction for wildfire policy—summarized as *“working from the home outward”*— in which the greatest public safety benefits come from prioritizing actions in or near homes and communities.

While the 2019 report was written for California’s state-level policymakers, Environment Now’s new report updates those findings with the goal of sharing lessons from California with federal policymakers. New content has also been added to address *two activities that have received increasing attention in the past two years, but that unfortunately are overstated in terms of their actual benefits— tree-planting (see p. 14) and prescribed fire (see pp. 19-22).*

Unfortunately, in California, we have not yet seen the needed change in direction from state leaders. While there have been a few steps to increase attention on home-safety retrofits, this has been buried within an approach described as “all of the above.” *“All of the above” sounds commonsensical, but in practice it is an abrogation of leadership.* The reality is that some fire policy actions are significantly more effective at increasing public safety than others. We need to actively identify and prioritize funding the most effective solutions. Yet, in “all of the above,” the well-established but failing approaches still tend to prevail in getting the biggest slices of the budgetary pie. When California’s Governor announced his billion-dollar wildfire budget for 2021, *less than 4% of those funds were designated for “community hardening”* (LAO 2021). *Even though home retrofits offer some of the most effective tools for increasing public safety during wildfires, that approach is still getting the crumbs from the actual expenditures.* Meanwhile, state leaders continue putting more money into subsidizing tree-cutting, even when these projects cause ecological damage, put more carbon into the atmosphere, increase fire intensity, and divert resources away from home-safety actions (Hanson and Hansen 2019).

A Key Lesson for Federal Policymakers

One key reason that California state leaders have not pivoted to a more effective direction for wildfire policy is that they have continued to rely heavily on the state-level fire suppression agency (CalFire) and its allies to steer policy recommendations. Unfortunately, we are unlikely to see real change advanced by the same people who promoted the failing fire policies of recent decades. Likewise, *federal policymakers should not expect to find a substantively new approach to wildfire coming from the leaders of the US Forest Service and their allies* who are invested in the current approach focused on suppression and forest-alteration.

Real progress will come from recognizing that *fire policy is fundamentally a public safety issue*, and then seeking out public safety experts rather than relying mainly on “forest management” discussions in which home-safety actions keep getting relegated to the margins. This will likely require setting up a new task force to focus specifically on *this central question—Which actions will produce the greatest public safety benefits during wildfires in the most cost-efficient manner?* While many actions may claim to have some potential benefits under particular circumstances, it is important not to fall into the “all of the above” run-around. Instead, there must be a rigorous and transparent comparative assessment of the costs and benefits of different alternatives, so that resources can then be focused primarily on the most effective actions. To produce an accurate assessment, the following guidance is recommended:

- Public safety experts should be at the forefront. This includes fire marshals, experts in home design, and community planning experts.
- As the Forest Service exerts significant influence over its scientists (Hanson 2021a), it is prudent to ask scientists to be transparent about any Forest Service funding they receive and to actively seek scientists who are not reliant on Forest Service funding.
- The insurance industry has a direct financial stake in identifying which actions are really the most effective in preventing homes from burning.
- Seek alternatives to the Forest Service for information from fire managers, including Tribes with long traditions of working with fire, municipal fire departments on the frontlines of keeping communities safe, and wildland firefighter whistleblowers (see Firefighters United for Safety, Ethics, and Ecology). Likewise, the National Park Service has explored innovative fire management policies. Now, wildfires generally burn less intensely in national parks than national forests (Bradley et al. 2016). Meanwhile, the Forest Service spends more than five times as much on fire per acre burned than the Park Service and other agencies in the Dept. of Interior (Bevington 2018, Stahl 2018).
- There has been a tendency to elevate those environmental organizations that promote the Forest Service’s approach to fire policy, which readily turns into an echo chamber. In contrast, environmental groups that are actively challenging Forest Service fire-related projects can provide due diligence in assessing Forest Service claims (Bevington 2018).

A More Effective Approach for Preventing Wildland-Urban Fire Disasters

By Jack Cohen, PhD, Research Physical Scientist, US Forest Service, retired

Summary

Communities exposed to inevitable extreme wildfire conditions do not have to incur inevitable disastrous fire destruction. Research shows that the characteristics of a home and its immediate surroundings within 100 feet (30 meters) principally determine home ignitions. This area, called the *home ignition zone* (HIZ), defines wildland-urban (WU) fires as a home ignition problem and not a problem of controlling wildfires. Communities can readily reduce home ignitability within the HIZ to prevent WU fire disasters instead of increasing wildfire suppression that fails during extreme wildfire conditions. Reducing the ignition conditions within the HIZ to produce ignition resistant homes provides an effective alternative for preventing WU fire disasters without necessarily controlling extreme wildfires.

Inevitable Wildfires and Extreme Burning Conditions

Wildfire occurrence is inevitable and thus, a small percentage of wildfires will inevitably attain uncontrollable extreme wildfire conditions. For over one-hundred years U.S. fire suppression has successfully controlled 95 to 98 percent of wildfires with initial attack (Stephens and Ruth 2005).

However, there is no historical evidence or current fire management trend to suggest that all wildfires can be excluded and if not excluded, controlled with an initial suppression response. Thus, we can assume the inevitability of wildfires and the occurrence of extreme wildfire conditions (Williams 2013). Most wildfires controlled at initial attack occur during moderate to high wildfire conditions. During severe conditions of drought, high winds, low relative humidity and multiple ignitions, 2 – 5 percent of the wildfires producing rapid growth with high burning intensities escape initial attack suppression.

The primary federal, state and local approach for protecting structures from wildfires and preventing community fire disasters is wildfire control using suppression added by pre-suppression fuel breaks and shrub and forest fuel treatments (Finney and Cohen 2003, Cohen 2010). However, disastrous community wildfire destruction (greater than 100 homes destroyed) has only occurred during extreme wildfire conditions when high wind speeds, low relative humidity and continuous flammable vegetation result in rapid fire growth rates and numerous spot ignitions from showers of burning embers (firebrands); that is, the conditions when wildfire control fails (Cohen 2010, Calkin et al. 2014).

Community fire destruction during wildfires will continue as long as wildfire suppression continues to be the primary residential protection approach. The inevitability of uncontrolled extreme wildfires suggests inevitable disastrous home destruction; however, research on how homes ignite during extreme wildfires indicates practical opportunities for effectively creating ignition resistant homes and thereby preventing community fire disasters without necessarily controlling wildfires (Cohen 2000a, Cohen 2001, Cohen 2004, Cohen and Stratton 2008, Cohen 2010, Calkin et al. 2014, Cohen 2017). We can immediately see how homes were not ignited during a wildfire from the readily observable patterns of destruction.

Figure 1.



Paradise, CA; 2018 Camp Fire



Southwest CO; 2002 Missionary Ridge Fire



S Cal; 2007 Grass Valley Fire

Patterns of Home Destruction during Wildfires

Total home destruction surrounded by green tree canopies following the Camp Fire in Paradise, CA (Figure 1, top photo) has been reported as unusual; however, unconsumed vegetation adjacent to and surrounding total home destruction is the typical WU fire pattern associated with extreme wildfire conditions (Cohen 2000b, Cohen and Stratton 2003a, Cohen 2003b, Cohen and Stratton 2008, Graham et al. 2012, Cohen 2017). The center photo (Figure 1) shows an example of a burning home that could have only ignited from lofted burning embers (firebrands) on the home and low intensity surface fire spreading to contact the home. The three photos (Figure 1) of home destruction with adjacent unconsumed shrub and tree vegetation indicate the following:

- **High intensity wildfire did not continuously spread through the residential area as a wave or flood of flame.**
- **Unconsumed shrub and tree canopies adjacent to homes did not produce high intensity flames that ignited the homes; ignitions could only be from firebrands and low intensity surface fires.**
- **The ‘big flames’ of high intensity wildfires did not cause total home destruction.**

High intensity wildfires do not spread through residential areas such as Paradise. The continuous tree and shrub canopies required to maintain high intensity wildfire spread (crown fires) are broken by fuel gaps such as streets, driveways and home sites (Cohen 2010). Figure 2 shows how a crown fire spread to but could not continue beyond the first residential street. Although the crown fire terminated at the street, firebrands showered downwind into the residential area initiating fires resulting in several blocks of total home destruction (Cohen 2010). Extreme wildfire conditions initiate ignitions within residential areas but the residential fuels, structures and vegetation continue the residential burning resulting in total home destruction. Commonly, homes ignite and burn hours after the wildfire has ceased active burning near the community (Cohen and Stratton 2008; Cohen 2010).



Figure 2.

Furthermore, the typical WU fire patterns indicate that conditions local to a home principally determine home ignitions with firebrands the principal source of ignitions within the residential area. The totally destroyed home in Figure 3 indicates firebrands as the only possible ignition source, potentially igniting the home directly and the flammable materials adjacent to the home. Firebrands are a given during extreme WU fire conditions; however, regardless of the distance firebrands were lofted, firebrand ignitions depend on the local conditions of the ignitable surfaces on or adjacent to a home.



Figure 3.

An Effective Approach for Preventing WU Fire Disasters

Research (Cohen 2004) has quantified “local ignition conditions” to be an area of a home and its immediate surroundings within 100 feet (30 meters). This area is called the *home ignition zone* (HIZ) (Cohen 2010, NFPA 2018). The relatively small area of the HIZ principally determines home ignitions during extreme wildfires and defines WU fire destruction as a home ignition problem that can be prevented by readily addressing home ignition vulnerabilities within the HIZ without necessarily controlling wildfires. For example, an ignition resistant home does not have a flammable wood roof, flammable tree debris on the roof, in the rain gutters, on decks or on the ground within 5 feet (1.5 m) of flammable siding, no open firewood within 30 feet (9 m), or unscreened vents. Clearing the HIZ of vegetation is not necessary. As indicated by the typical patterns of WU fire destruction, shrub and tree canopies are not spreading high intensity fires through communities. The inevitability of uncontrolled extreme wildfires spreading to communities does not mean WU fire disasters are inevitable if we address the problem with the readily available approach of reducing home ignitability. Ignition resistant communities increase community fire protection effectiveness, life-safety options for residents and firefighters, and decrease wildfire suppression costs while preventing WU fire disasters without attempting to protect communities by controlling wildfires.

20th Century Fire Suppression Policy is Not Appropriate for a 21st Century Climate

by Timothy Ingalsbee, PhD, Executive Director, Firefighters United for Safety, Ethics, and Ecology

Up until the mid-20th century, we had a lot more fire on the land.

Hundreds of fire history studies document that wildland fires burned significantly more area than burns now. Even in the 20th century up until the 1950s, several tens of millions of millions of acres burned in the U.S. each year (NIFC n.d.).

Then we began mechanized firefighting in the 20th century.

Federal agencies such as the U.S. Forest Service (USFS) began fighting fires in 1905, but with minimal effectiveness due to the large expanse of undeveloped wildlands, the limited size of its workforce, and primitive technology. This changed in the post-World War II period with an influx of military surplus vehicles and equipment in fire suppression (Pyne 1982). Cutting firelines with bulldozers and airtankers dropping chemical retardants brought annual burned acreage crashing. In California alone there was a 36% decline in area burned from the 1940s to the 1950s, the start of a trend of rapidly declining acres burned that continued until the 1980s (CalFire n.d.). This created a historically unprecedented shortage of fire on the landscape that is still adversely affecting fire-adapted ecosystems across the West.

But the post-war surge of suppression success accompanied a change in climate.

At the same time that mechanized firefighting was pushing deeper into backcountry wildlands and containing nearly all wildfires at a small size, the climate had changed. A prolonged cool, wet period from a natural cycle of climate variability called the Pacific Decadal Oscillation (PDO) greatly aided firefighters' efforts in stopping wildfire spread (Littell et al. 2009, Peterson et al. 2011). This created an unprecedented shortage of fire on the landscape during the 1950s and 60s. During this post-war period with its anomalously and artificially low level of wildfire activity, people developed a distorted perception of wildfires as absolutely bad along with a false sense of security that firefighters could put them all out (Murphy et al. 2018).

21st century climate change is making wildfires start easier and spread faster.

At the end of the 20th century that cool, wet PDO cycle ended and was replaced with much warmer and drier conditions that are now being amplified by global warming from fossil-fuel emissions. Prolonged droughts punctuated by frequent severe fire weather conditions (high temperatures, high winds, and low relative humidity) are making vegetation ignite much easier and fires spread more rapidly. Beginning in the 1980s but accelerating after 2000, the signal of anthropogenic climate change is now registering in greatly increased wildfire activity that is leading to longer fire seasons and increased amount of acres burned. But even this recent increase in large fires masks the fact that there still much less fire on the land than is necessary for maintenance of fire-adapted forest ecosystems across many Western landscapes.

21st century climate is ending the efficacy of conventional firefighting.

Conventional firefighting tactics of dumping retardant, cutting firelines, and lighting backfires cannot stop wind-blown flames from jumping over firelines or firebrands lofting in the sky and landing on flammable rooftops miles away from a wildfire's flaming front. Now that 21st century anthropogenic global warming is causing severe fire weather conditions to become more frequent, the efficacy of conventional suppression is further declining. Conventional firefighting strategies and tactics are unable to either prevent or suppress large wildfires that are now being driven by climatic conditions that will be with us for the far foreseeable future.

Suppression spending is soaring.

In response to increasing wildfire activity, both federal and state agencies have been dramatically escalating their suppression spending in recent decades. There had never been a billion-dollar wildfire season before 2000, but now it is nearly an annual occurrence. The most recent 5-year average indicates that firefighting costs have risen to almost \$2 billion per year (NIFC 2020). Forest Service tactics remain rooted in a suppression-based approach that is proving more and more expensive and less and less effective in a 21st century climate. In fact, the last four years have seen the highest suppression spending in the Forest Service's existence—accompanied by huge urban fire disasters and record numbers of homes destroyed.

Expanding the fleet of airtankers would be a poor investment of taxpayer dollars.

A signature example of a costly and increasingly ineffective 20th century approach to fire suppression is the emphasis on airtankers. Airtankers are one of the most expensive resources used in wildfire suppression, but several recent studies have found that airtankers are routinely deployed at times, places, and conditions where they are least useful or effective (Stonesifer et al. 2016, Stonesifer et al. 2015, Calkin et al. 2014, Thompson et al. 2012). They are particularly likely to be impaired by high winds associated with severe fire weather. In California, airtankers are required to be grounded when there is even moderate turbulence or windspeeds exceed 35 mph. Heavy smoke is another impediment to effective airtanker use. For example, while the Camp Fire raged through Paradise, a fleet of airtankers located next door in Chico was grounded by high winds and dense smoke.

Fighting fires in backcountry wildlands depletes resources needed to protect communities.

Systematic attempts to exclude or suppress all fires regardless of whether or not they are near communities is costly to taxpayers and puts communities at risk from lack of suppression crews and resources actually protecting homes. For example, in 2016 a joint USFS/CalFire effort spent over \$262 million on the Soberanes Fire that burned mostly in the Ventana Wilderness Area and became the most expensive wildfire suppression operation in U.S. history (Ingalsbee et al. 2018). A USFS internal investigation (USDA-FS 2017) concluded that the excessive spending reflected “systemic fire management issues” revolving around lack of fiscal accountability that have yet to be solved. These large expenditures on fire suppression in remote areas pull limited resources away from the actions that are most effective at preventing home loss during fires.

Recommendations:

- *Wildland fires are ecologically necessary and inevitable, but losses of life and property in urban fire disasters need not be inevitable if we adopt new fire management policies and practices suitable for 21st century climate conditions. We need to move away from 20th century mechanized fire suppression strategies, tactics, and tools (e.g., large airtankers) that are inappropriate and increasingly ineffective in the current climate.*
- *Suppression resources should be redirected away from fighting fires in remote wildlands where fire is ecologically necessary and instead focused on directly protecting communities.*
- *Invest in preparing communities to live safely and sustainably in a fire-prone environment: retrofit homes to reduce home ignitability, improve emergency communications, maintain safe evacuation routes, construct community fire shelters, bury powerlines, and implement other infrastructure projects needed to create fire-compatible communities.*

Common Myths about Forests and Fire

by Chad Hanson, PhD, Ecologist and Director, John Muir Project

Do We Currently Have an Unnatural Excess of Fire in our Forests? No. There is a broad consensus among fire ecologists that we currently have far less fire in western US forests than we did historically, prior to fire suppression (dozens of studies discussed in Hanson et al. 2015). Even a very big fire year in modern times, such as the 4-million-acre 2020 fire season in California, would have been merely average historically (Stephens et al. 2007, Baker 2017). California's forests have always burned with a mixture of intensities, including patches of high-intensity fire. We have less fire of all intensities now, including less high-intensity fire (Stephens et al. 2007, Mallek et al. 2013, Baker et al. 2018).

Do Current Fires Burn Mostly at High-Intensity Due to Past Fire Suppression? No. Current fire is mostly low/moderate-intensity in western US forests, including the largest fires (Mallek et al. 2013, Baker et al. 2018). The most long-unburned forests experience mostly low/moderate-intensity fire (Odion and Hanson 2008, Miller et al. 2012, van Wagtendonk et al. 2012).

Do Large High-Intensity Fire Patches Destroy Wildlife Habitat or Prevent Forest Regeneration? No. Hundreds of peer-reviewed scientific studies find that patches of high-intensity fire create "snag forest habitat", which is comparable to old-growth forest in terms of native biodiversity and wildlife abundance (DellaSala and Hanson 2015). In fact, more plant, animal, and insect species in the forest are associated with this habitat type than any other (Swanson et al. 2014). Forests naturally regenerate in ecologically beneficial ways in large high-intensity fire patches (DellaSala and Hanson 2015, Hanson and Chi 2021).

Is Climate Change a Factor in Recent Large Fires? Yes. Human-caused climate change increases temperatures and can exacerbate drought, which influences wildland fire. Some mistakenly assume this means we must have too much fire but, due to fire suppression, we still have a substantial fire deficit in our forests. For example, historically, snag forest habitat, from high-intensity fire and patches of snag recruitment due to drought and native bark beetles, comprised 27% to 34% of the forests in the Sierra Nevada (Baker 2014, Baker et al. 2018). Currently, based on federal Forest Inventory and Analysis data, it comprises less than 8% of Sierra Nevada forests (Hanson and Chi 2020).

Are Forests Unnaturally Dense and "Overgrown", and Do Denser Forests Necessarily Burn More Intensely? No. Our forests actually have a carbon deficit, due to decades of logging. Historical forests were variable in density, with both open and very dense forests (Baker et al. 2018). Recent studies by U.S. Forest Service scientists omitted historical data on small tree density and density of non-conifer trees. When the missing historical data were included, it was revealed that historical tree density was 7 times higher than previously reported in ponderosa pine forests, and 17 times higher than previously reported in mixed-conifer forests (Baker et al. 2018). Wildfire is driven mostly by weather/climate (Bradley et al. 2016). Forest density is a "poor predictor" (Zald and Dunn 2018).

Are Recent Large Fires Unprecedented? No. Fires similar in size to the Rim fire and Creek fire, or larger, occurred prior to modern fire suppression (Bekker and Taylor 2010, Caprio 2016).

Do Occasional Cycles of Drought and Native Bark Beetles Make Forests "Unhealthy"? Actually, it's the opposite. During droughts, native bark beetles selectively kill the weakest and least climate-adapted trees, leaving the stronger and more climate-resilient trees to survive and reproduce (Six et al. 2018). In areas with many new snags from drought and native bark beetles, most bird and small mammal species *increase* in numbers in such areas because snags provide such excellent wildlife habitat (Stone 1995).

Do Forests with More Dead Trees Burn More Intensely? Small-scale studies are mixed within 1-2 years after trees die, i.e., the “red phase” (Bond et al. 2009, Stephens et al. 2018), but the largest analysis, spanning the entire western U.S., found no effect (Hart et al. 2015). Later, after needles and twigs fall and quickly decay into soil, and after many snags have fallen, such areas have similar or *lower* fire intensity (Hart et al. 2015, Meigs et al. 2016). In the 380,000-acre Creek fire of 2020 in the Sierra Nevada, unlogged forests with the highest densities of snags from the 2014-2017 drought had the lowest fire severity, while logged areas, where live and dead trees were removed under the guise of “fuel reduction”, had high-severity levels that were 8 to 10 times higher (Hanson 2021b).

Does Reducing Environmental Protections, and Increasing Logging, Curb Forest Fires? No, based on the largest analysis ever conducted, this approach increases fire intensity (Bradley et al. 2016). Logging reduces the cooling shade of the forest canopy, creating hotter and drier conditions, leaves behind kindling-like “slash” debris, and spreads combustible invasive weeds like cheatgrass.

Do “Thinning” Logging Operations Stop Wildland Fires? No. “Thinning” is used as a euphemism for intensive commercial logging projects that kill and remove many of the trees in a stand, often including mature and old-growth trees. With fewer trees, winds, and fire, can spread faster through the forest. In fact, extensive research shows that commercial logging, conducted under the guise of “thinning”, often makes wildland fires spread *faster*, and in most cases also *increases* fire intensity, in terms of the percentage of trees killed (Cruz et al. 2008, 2014).

Did the Rim Fire Emit Carbon Equal to Over 2 Million Cars? No. Unpublished reports from the Forest Service and the California Air Resources Board regarding wildfire carbon emissions are based on a flawed model (FOFEM) that has repeatedly been shown to exaggerate carbon emissions by nearly threefold (French et al. 2011). Further, the FOFEM model falsely assumes that no post-fire regrowth occurs to pull CO₂ out of the atmosphere. Field studies of large fires find usually only about 11% of forest carbon is consumed, and only 3% of the carbon in trees (Campbell et al. 2007), and vigorous post-fire forest regrowth absorbs huge amounts of CO₂ from the atmosphere, resulting in an overall *net decrease* in atmospheric carbon within a decade after fire (Meigs et al. 2009).¹

Would Landscape-Scale Prescribed Burning Reduce Smoke? No, it’s the opposite. Prescribed fires do not stop wildland fires when they occur. Though they can briefly alter fire intensity, prescribed-burn areas can burn again in a wildfire in less than a year (Stephens et al. 2009). High-intensity fire patches produce relatively lower particulate smoke emissions (due to high efficiency of flaming combustion in higher-intensity fire patches) while low-intensity prescribed fires produce high particulate smoke emissions, due to the inefficiency of smoldering combustion. Therefore, even though high-intensity fire patches consume about three times more biomass per acre than low-intensity fire (Campbell et al. 2007), low-intensity fires produce 3-4 times more particulate smoke than high-intensity fire, for an equal tonnage of biomass consumed (Ward and Hardy 1991, Reid et al. 2005). As a result, a landscape-level program of prescribed burning would cause at least a ten-fold increase in smoke emissions relative to current fire levels if, for example, prescribed fire was conducted every 10 years in any given area, as many land managers propose (Rhodes and Baker 2008).

¹ For example, Campbell et al. (2007) found that the Biscuit fire of 2002 emitted an average of 19 tons of carbon per hectare, and Campbell et al. (2016) found that decay of fire-killed trees in the Biscuit fire emitted an average of about 0.75 tons of carbon per hectare per year over the first 10 years post-fire (there were lower emissions from decay in subsequent decades). Therefore, for the first 10 years post-fire, the total carbon emissions from the Biscuit fire (carbon emissions from the fire itself, plus subsequent emissions from decay) were approximately 26 tons of carbon per hectare. Meigs et al. (2009) (Table 5) report that, by only five years after fire, regrowth was pulling 3.1 tons of carbon per hectare per year out of the atmosphere. Therefore, by 10 years post-fire, this equates to approximately 31 tons of carbon pulled out of the atmosphere by regrowth—i.e., an overall net increase in carbon of 5 tons per hectare relative to pre-fire levels.

California Forests, Wildfires, and Carbon

By Dominick DellaSala, PhD, Chief Scientist, Wild Heritage

California's older forests (e.g., Sillett et al. 2020) and large trees (e.g., Stephenson et al. 2014) are nature's unique climate solutions readily absorbing and storing massive amounts of carbon for centuries. Protecting the carbon stored in these and other native forests from logging is key to a climate-safe future for California. However, recent misinformed policies proposed by the State seek to elevate logging in response to wildfires, which runs counter to biodiversity conservation, climate mitigation, and may actually intensify wildfires (Bradley et al. 2016). These policies are sometimes portrayed as ways to sequester and store more carbon in forests and wood products. However, there is a better way to address pressing climate issues in California. In sum, this means protecting carbon stocks present in older forests and large trees and allowing young forests time to reach their full carbon potential (referred to as "proforestation," Moomaw et al. 2019).

Do forest fires emit massive amounts of carbon dioxide?

Most of the carbon in a forest remains on site after a wildfire (Campbell et al. 2007, Meigs et al. 2009, Mitchell 2015). Total annual emissions from wildfires over large regions are generally much less (~10% in active fire seasons) than total annual emissions from logging in the same region (Meigs et al. 2009, Campbell et al. 2012, Law et al. 2018, Oregon Global Warming Commission 2018). Some prior estimates of wildfire emissions have grossly exaggerated combustion of carbon during a wildfire. In reality, however; only a small portion of a trees' biomass (mainly twigs and leaves) is actually combusted. Moreover, about half the carbon in burned forests remains bound to the soils for nearly a century, the rest of the soil carbon builds over millennia (Singh et al. 2012). After fires, growth of surviving trees and new vegetation sequester carbon, offsetting emissions within about 5-50 years (depending on site factors; Meigs et al. 2009, Mitchell 2015).

Does logging store or release carbon?

Depending on logging intensity, forest type, and forest age class, over 80% of carbon stored in a forest is released to the atmosphere within 100 years of logging due to decomposition (or burning) with additional emissions released during transport and manufacturing of wood products especially over large hauling distances (Oregon Global Warming Commission 2018, Law et al. 2018, Hudiburg et al. 2019). A small portion of the carbon is temporarily embodied in wood product pools ranging from 1 year (paper) to decades (buildings) before decomposing and emitting CO₂ in landfills (Oregon Global Warming Commission 2018, Hudiburg et al. 2019). Instead, carbon stocks in forests can be doubled if forests are protected from logging on federal lands, timber harvest rotations extended from ~35 to 70 years on industrial lands, and other forestry improvements (Law et al. 2018). Avoiding emissions from forestry is recommended by the Intergovernmental Panel on Climate Change as an effective means for preventing warming in excess of 1.5° C globally. According to NASA's Earth Observatory (2017), California already is pushing temperature increases dangerously close to unsafe levels.

Does “thinning” reduce emissions from wildfires?

Studies of landscape-scale logging (“thinning”) to reduce the probability of crown fires show that this practice will not reduce carbon emissions under current or future climate scenarios and may in fact make matters much worse especially if thinning residues are burned as biofuels (Meigs et al. 2009, Hudiburg et al. 2009, 2011, Campbell et al. 2012, Mitchell et al. 2012, Schulz et al. 2012, Law et al. 2013). This is because the amount of carbon removed by landscape-scale thinning and related activities (e.g., road building and maintenance) is larger than that potentially avoided from a fire. Importantly, fire only occurs on a fraction of the areas thinned (Rhodes and Baker 2009, Campbell et al. 2012, Schoennagel et al 2017).

What is the carbon potential from tree-planting compared to proforestation?

While some scientists (Bastin et al. 2019) estimate planting a trillion trees globally could sequester ~200 Gt C within 100 years, others (Veldman et al. 2019) have shown that the correct number is ~42Gt C. Thus, there are better ways to increase the carbon stored in forests. It is estimated that forests currently store only half their potential carbon capacity due to repeat short-interval logging (Erb et al. 2018). Thus, land managers could double the forest carbon stock over a century by simply allowing existing forests time to reach their full carbon potential. Protecting already forested areas so they can mature via proforestation would sequester 120 Gt C by century end, nearly five times that of planting trees (Moomaw et al. 2019). Thus, compared to the “trillion trees initiative,” maintaining existing forests is a much more prudent strategy.

Conclusions

California’s forests have always benefitted ecologically from periodic mixed-severity fires that create diverse wildlife habitat (DellaSala and Hanson 2015), stimulate plant growth and nutrient cycling, and carbon sequestration. Overall, they are not a major source of emissions currently as most of the carbon remains on site after natural disturbance and new vegetation restarts uptake. Much bigger emissions are produced by logging and other industrial sectors. Thus, policies that advocate for increased logging are inconsistent with California’s otherwise ground-breaking climate change efforts, and the recommendations of the Intergovernmental Panel on Climate Change. Protecting forests from logging is a natural climate solution on par with global efforts to mitigate climate change impacts (Griscom et al. 2017). California has some of the most carbon dense forests on the planet and these forests should form the backbone of a comprehensive climate change strategy that includes avoiding emissions from all sectors especially forestry while preparing for unavoidable consequences of climate impacts.

Understanding Why Forest Biomass Energy is a Bad Idea is as Easy as P-I-E

by Shaye Wolf, PhD, and Brian Nowicki, Center for Biological Diversity

Understanding why incinerating forests to generate electricity is a bad idea is as easy as P-I-E.

Forest biomass power is:

- **Polluting**, emitting greenhouse gases, worsening the climate crisis, and harming vulnerable communities
- **Ineffective** for protecting communities during wildfires
- **Expensive** and dependent on subsidies that take resources away from truly clean energy alternatives

Fire policies in California rely heavily on efforts to increase logging to alter forest fire behavior paired with burning cut forest biomass for energy production. Biomass power is often portrayed as being carbon neutral, but it is not. Instead, biomass facilities increase greenhouse gas emissions; undermine the transition to clean, renewable power; pose public health threats in already disadvantaged communities; and distort policies for forest and fire management.

Biomass energy is more climate-polluting than coal.

Forest-sourced woody biomass energy generation emits about 50% more CO₂ per megawatt-hour of electricity produced than coal-fired power and three times the CO₂ of natural gas (Booth 2014). While the baseline emission rate for California's current electricity portfolio is about 485 lbs of CO₂e per MWh (CARB 2020), biomass can emit more than 3,000 lbs of CO₂ per MWh (Booth 2014, Center for Biological Diversity 2021), and smaller-scale facilities using gasification technology are similarly carbon-intensive (Ascent Environmental 2012).

Using forest biomass as a feedstock causes a significant negative impact to the climate.

In addition to smokestack emissions, an accurate accounting of the climate harms of biomass energy must include the carbon consequences of the tree removals that generate the feedstock. Thinning operations tend to remove about three times as much carbon from the forest as would be avoided in wildfire emissions (Campbell et al. 2011), and the removal of live trees from the forest results in a loss of future growth and carbon sequestration by those trees.

The climate damage of biomass energy can persist for decades to centuries.

Incinerating biomass for energy instantaneously releases stored carbon to the atmosphere, increasing greenhouse gas emissions and creating a "carbon debt." Numerous studies show that, even if forests cut for bioenergy are allowed to regrow, it can take several decades to more than a century, if ever, to capture the carbon that was released, and to discharge the "carbon debt." This is the case even where "waste" materials like timber residues and thinning debris are used for fuel (Searchinger et al. 2009, Manomet Center for Conservation Sciences 2010; Campbell et al. 2011, Hudiburg et al. 2011, Law and Harmon 2011, Holtsmark 2012, Mitchell et al. 2012, Schulze et al. 2012, Booth 2018, Sterman et al. 2018). Meanwhile, that carbon pollution worsens the climate crisis and contributes to the probability of passing climate tipping points, causing irreversible harms.

Biomass results in significant emissions of air pollutants, often in California's most polluted communities.

In addition to producing large amounts of CO₂, biomass generation can result in significant emissions of air pollutants that harm human health (Booth 2014). Biomass emissions can exceed those of coal-fired power plants even after application of best available control technology. Many of California's biomass power plants are concentrated in vulnerable communities in the Central Valley already suffering from high pollution burdens (Center for Biological Diversity 2021). For example, the Rio Bravo biomass plant in Fresno is located less than a half-mile from the Malaga Elementary School and surrounding homes in a majority Latinx neighborhood with the state's highest pollution burden score.

Logging for biomass energy is ineffective for protecting communities during wildfires.

Biomass energy is often promoted as a tool to incentivize large-scale tree-cutting ("thinning") under the claim that this will protect communities and forests during wildfires. However, this approach is ineffective at protecting houses and communities, which is best achieved through a home-focused fire-safety strategy that helps communities safely coexist with inevitable wildfires. Although biomass energy is promoted as a means for disposing of debris piles from forest thinning projects, it is mostly lumber mill residues from commercial logging that end up being subsidized (CalRecycle 2018). Meanwhile, biomass extraction does significant ecological damage to forests.

Policies that subsidize forest biomass divert funds from zero-carbon sources like solar and wind and impede the transition to renewable energy.

Biomass power is the most expensive of California's common electricity sources. In 2018, the levelized cost of biomass power averaged \$166 per megawatt hour compared to \$49 for solar and \$57 for wind (California Energy Commission 2019). Yet, California requires that electricity suppliers collectively purchase 175 MW of forest-sourced woody biomass power annually.

Conclusion: Forest biomass energy is an expensive and highly polluting

Instead of promoting biomass energy that harms our climate, communities, and forests, policymakers should:

- Stop mandating, subsidizing, or otherwise incentivizing biomass power production, and instead direct investments toward truly clean energy production such as solar and wind.
- Fully account for the smokestack emissions from biomass power plants and stop incorrectly treating biomass power as "carbon neutral."
- Create climate-smart wildfire and forest policy that invests in proven home and community-focused approaches to wildfire safety rather than forest-cutting, while increasing forest protections that keep carbon stored in forest ecosystems as an essential climate solution.

To learn more, see "Forest Biomass is a False Solution" (Center for Biological Diversity 2021): https://www.biologicaldiversity.org/campaigns/debunking_the_biomass_myth/pdfs/Forest-Bioenergy-Briefing-Book-March-2021.pdf

Forest Fire Policies are Being Misapplied to Chaparral Ecosystems

by Richard Halsey, Executive Director, California Chaparral Institute

Chaparral is California's most extensive plant community. It is found in every county in the state. Characterized by drought-hardy shrubs, a Mediterranean-type climate, and infrequent, high-intensity fire, chaparral provides the habitat richness responsible for making California one of the most biodiverse regions on earth (Halsey and Keeley 2016). The chaparral's relationship to fire is dramatically different from that of California's forests. *Actions that are often proposed for addressing fire in forest ecosystems are not appropriate in chaparral ecosystems and can lead to more flammable landscapes, destruction of critical habitat, and are an ineffective approach to protecting human communities built in these areas.*

High-intensity fire is required.

The natural fire regime for chaparral is characterized by large, high-intensity crown fires with a return interval of 30 – 150 years (Keeley and Fotheringham 2001, Lombardo et al. 2009, Safford et al. 2014). Research has demonstrated that the higher the intensity of the fire, the better the chaparral is able to recover (Keeley et al. 2005a). Therefore, concerns over reducing fire intensity and severity are irrelevant to chaparral ecosystems; there's no such thing as a low-intensity chaparral fire except at the edges of fire perimeters or when localized conditions (e.g. boulders, wind shifts, moisture) reduce fire intensity. By the very nature of the physical structure of shrubs, high intensity fire is an inherent part of chaparral fires.

Long fire return intervals are required, and too much fire causes loss of chaparral.

When compared to most forests, chaparral has comparatively long intervals between fires (30 – 150 years or more). Long fire return intervals are vital for the chaparral's ecological health. It can take up to thirty years for the native shrubs to build up enough seed in the soil to provide adequate germination rates post fire.

However, increases in fire frequency due to human-caused ignitions and the effects of climate change cause chaparral stands to become more open and are often invaded by nonnative grasses. Fire-return intervals fewer than 10 years have been shown to be highly detrimental to the persistence of chaparral species (Haidinger and Keeley 1993, Jacobsen et al. 2004). As grasses increase, the flammability of the chaparral ecosystem also increases. As a consequence, a positive feedback loop is created whereby more grass encourages frequent ignitions. Such frequent fires not only eliminate the native shrubs, but they facilitate the further spread of invasive weeds and grasses due to the fact that grass fires are less intense than shrubland fires. The type conversion process can ultimately lead to the complete replacement of native chaparral with nonnative grasses (Halsey and Syphard 2015).

Prescribed burns and vegetation clearing are destructive to chaparral and increase fire.

When fire management policies commonly used in forests—such as prescribed fire and vegetation clearing—are misapplied to chaparral, the results are destructive to the ecosystem and can actually increase fire. Since there is too much fire in chaparral plant communities due to human-caused ignitions, adding more through prescribed burns only increases the threat to the chaparral ecosystem's continued existence and conversion to invasive grasses that bring more frequent fires. Furthermore, prescribed burns are typically conducted in the late spring when the ecosystem is the most vulnerable to damage: the plants are growing, the soil is still moist, and many animal species are breeding. Therefore, prescribed burns can cause significant damage to plant growth tissues and destroy seeds in the soil due to soil moisture turning into steam, leading to chaparral type conversion.

Similarly, large-scale vegetation clearing projects (“fuelbreaks”) also cause the loss of native chaparral and the spread of invasive grasses that leads to more frequent fires. Amid the increasing dangers to chaparral from the effects of climate change, it is imperative that land management agencies do not exacerbate the loss of chaparral through activities like prescribed burns and large-scale habitat clearance projects away from homes. Instead, fire management in chaparral should focus on reducing the unnaturally high level of fire ignitions that has accompanied human development in this ecosystem (Keeley et al. 2005b, Keeley 2006, Syphard et al. 2007).

Focus on homes and their immediate surroundings to make fire-safe communities.

While fire’s role in chaparral is different from in forests, the most effective way to keep homes from igniting during wildfires is the same in chaparral areas as in forest areas— focus on fire-safety features for homes and the zone right around them, rather than large-scale vegetation alteration in wildlands.

In a comprehensive study of the 2007 Witch Creek Fire in San Diego County, researchers found, “Wind-blown embers, which can travel one mile or more, were the biggest threat to homes in the Witch Creek Wildfire. There were few, if any, reports of homes burned as a result of direct contact with flames” from wildland fuels (IBHS 2008).

In a study examining 700,000 addresses in the Santa Monica Mountains and part of San Diego County researchers mapped the structures that had burned in those areas between 2001 and 2010, a time of devastating wildfires in the region (Syphard et al. 2012). Buildings on steep slopes, in Santa Ana wind corridors and in low-density developments intermingled with wild lands were the most likely to have burned. Nearby vegetation was not a big factor in home destruction. Looking at vegetation growing within roughly half a mile of structures, the authors concluded that the exotic grasses that often sprout in areas cleared of native habitat like chaparral could be more of a fire hazard than the shrubs. “We ironically found that homes that were surrounded mostly by grass actually ended up burning more than homes with higher fuel volumes like shrubs,” Syphard said.

Working only on defensible space is not sufficient. Many homes with adequate defensible space have still burned to the ground because embers have entered through attic vents, ignited flammable materials around the home (litter in the gutter, wood stacks, wood fencing), or found their way under roofing materials (Maranghides and Mell 2009). The solution is to reduce the flammability of the home as much as possible: install ember resistant vents, Class A roofing, exterior sprinklers operated by an independent system, and remove flammable materials 100 feet from around the structure.

Myths of Prescribed Fire: The Watering Can that Pretends to be a River

by Bryant Baker, M.S., Conservation Director, Los Padres ForestWatch
and Douglas Bevington, PhD, Forest Program Director, Environment Now

The use of prescribed fire—intentionally setting fires in forests and other ecosystems under planned circumstances—has received increased attention in California and elsewhere in recent years. On the one hand, it is good that there is growing recognition that fire is a natural and necessary part of forests and other ecosystems. On the other hand, current advocacy for large-scale prescribed fire across vast areas is often built on outdated assumptions and overstated claims, while downplaying problems stemming from how prescribed fire is actually being implemented. This factsheet identifies five key sets of myths regarding prescribed fire and shows how they can lead to misguided policies and missed opportunities to better accomplish public safety and ecological restoration goals in a more cost-effective manner. To create effective fire policies, we need to face these facts—*Prescribed fire increases fire and smoke. Prescribed fire is inefficient for public safety compared to home retrofits. Prescribed fire is inefficient for ecological restoration compared to managed wildfire. Prescribed fire can be harmful. And prescribed fire and cultural burning are not the same.*

Prescribed fire increases fire and smoke.

A central myth is that increasing prescribed fire will lead to less fire and smoke overall. Proponents of prescribed fire highlight examples where a portion of a wildfire halted when it encountered a previously burned area, but these anecdotes are the exception rather than the rule. The reality is that wildfires can burn through previously burned areas as soon as eight months after the prior fire (Stephens and Moghaddas 2005). Over 106,000 acres within the 2020 LNU Lightning Complex in California had burned within the previous five years, with 67,000 acres having burned just two years prior. As fire researchers have stated, *“fuel treatments are not intended to stop wildfires”* (Omi and Martinson 2004). Instead, the main goal of prescribed fire is to somewhat alter subsequent fire intensity in the affected area, though that may not occur under unfavorable weather conditions. In other words, *prescribed fire is additive to, rather than being a substitute for, wildfire*. Even in instances where prescribed fire has been found to limit wildfire extent, the acreage of a prescribed burn significantly exceeds the acreage of subsequent wildfire reduction, with 3-4 units of prescribed fire needed to reduce wildfire by one unit (Fernandes 2015). Furthermore, the effects of prescribed fire on wildfire behavior fades within a few years. Within as little as 2 or 3 years after prescribed fire, combustible understory vegetation can return to levels equal to or greater than levels prior to prescribed burning (Knapp et al. 2007). Thus, prescribed fires would need to be reapplied on a regular basis, repeatedly adding fire to many places that otherwise might not encounter a wildfire until many years in the future. For all these reasons, *increased use of prescribed fire will likely lead to a net increase in the total amount of fire* (Hunter and Robles 2020).

With that additional fire comes additional smoke. Proposals to implement *landscape-wide prescribed fire could result in ten times as much smoke* (Hanson 2021a; see also p. 12 in this report). In addition to increasing the total amount of smoke, *increasing prescribed fire also increases the duration of smoke*

exposure. While wildfire smoke is concentrated in the height of fire season—and landscape-scale use of prescribed fire would not preclude this—prescribed fires are typically lit in the “shoulder seasons” when wildfires are less likely, and thus prescribed fires prolong smoke exposure into times when it would not otherwise occur. There can be circumstances where it is appropriate to use prescribed fire, but it should be done knowing that the effect will be an overall increase in the amount and duration of fire and smoke.

Prescribed fire is inefficient for public safety compared to home retrofits.

Prescribed fire is an inefficient and relatively ineffective way to protect homes and communities during wildfires. As Dr. David Lindemayer recently summarized, *“The peer-reviewed evidence is that burning forest miles from houses doesn’t protect those houses”* (Foley 2021). As discussed above, prescribed fires generally do not stop subsequent wildfires, and altering fire intensity is largely irrelevant to community safety because home ignitions during wildfires are rarely caused by direct contact with high-intensity fire (Cohen and Stratton 2008, Syphard et al. 2017). Instead, home fire-safety retrofits (“home hardening”) offer the most effective ways to keep communities safe during wildfire. Yet the resources to help communities with fire-safety retrofits are currently quite limited compared to the government funding for prescribed fire and associated “fuel treatments” in wildlands. For example, in California’s 2021 proposed budget for wildfire preparedness, *less than 4% of the funding is directed to “community hardening”* (LAO 2021). While prescribed burning adjacent to communities can potentially have some benefits, proposals to use large-scale prescribed fire across vast landscapes away from communities represent a remarkably indirect and inefficient way to protect houses when compared with the direct benefits of home retrofits.

Prescribed fire is inefficient for ecological restoration compared to managed wildfire.

One positive outcome of the greater attention on prescribed fire is that it has contributed to growing recognition that fire is a necessary part of forests and other ecosystems, and that currently many forests have a shortage of fire compared to levels prior to modern fire suppression. Unfortunately, many advocates for prescribed fire rely on an *outdated “good fire/bad fire” dichotomy that is out of step with the science*. In this false dichotomy, prescribed fires are characterized as “good” because they are associated with low-intensity fire, whereas wildfires are characterized as “bad” because they are associated with mixed-intensity fire that includes some areas of high-intensity fire. However, there is a growing body of research showing that high-intensity fire has always been a part of forests and other ecosystems, and it produces ecological benefits by creating excellent wildlife habitat and stimulating nutrient cycling (DellaSala and Hanson 2015). In contrast, low-intensity fire associated with prescribed burning does not generate the habitat creation and the nutrient cycling associated with higher-intensity fire. *Each type of fire intensity has its role in a mixed-intensity fire regime, and low-intensity fire is not a substitute for the benefits from some higher-intensity effects in forests.*

Another myth repeated by some prescribed advocates is the erroneous notion that forests that previously experienced fire suppression will now “burn up” in all or mainly high-intensity fire when a wildfire occurs, unless those forests first get prescribed fire or other “fuel treatments.” But multiple studies have shown that areas that experience wildfire following long periods of fire suppression still burn mainly at low and moderate intensity, along with some high-intensity patches that provide the benefits described above (Odion and Hanson 2008, Miller et al. 2012). In fact, research has found that forests with the

longest fire exclusion actually burn at somewhat lower intensity (Odion et al. 2010). *This is exciting news for efforts at ecological restoration because it means that large-scale prescribed fire or other “treatments” are not needed as a precondition to allowing mixed-intensity wildfire back into forests.*

Instead, *managed wildfire offers a much more efficient way to restore fire to forests that currently have a shortage of fire*, and this has economic and practical benefits. Managed wildfire (also known as wildland fire use) differs from fire suppression in that, rather than trying only to extinguish a wildfire, fire managers seek to shepherd the fire away from communities and into wildland areas where the fire will provide ecological benefits. This is less intensive and costly than full suppression. Likewise, managed wildfire differs from prescribed fire because the latter has an extensive planning process and often quite restrictive parameters before a prescribed fire ignition can occur, whereas managed wildfire works with natural fire ignitions to provide more fire where it is needed in a more expeditious manner.

This is not to say that prescribed fire does not have a role in fire policy, but instead that role has been overstated. Prescribed fire is a useful tool when there are special circumstances where fine-scale control over fire is needed. In this regard, prescribed fire has a role equivalent to the role of watering can. A watering can is a good tool for tending the plants around your house, but *if you are faced with a dry field in need of irrigation, it would be absurd to propose buying thousands of watering cans. Yet this is basically equivalent to current proposals relying mainly on prescribed fire to restore fire to vast areas.* Instead, the right way to water a large field is through large-scale irrigation—a river, not a watering can—and the most efficient way to restore fire to large landscapes is through managed wildfire. Yet, despite its benefits, managed wildfire currently gets surprisingly little attention in current fire policy discussions, compared to prescribed fire.

Prescribed fire can be harmful.

Prescribed fire is sometimes called “good fire” by its proponents, but the reality is that, while it can be useful in some circumstances, *prescribed fire can also cause ecological damage.* One example is when prescribed fire policies get applied to non-forest ecosystems such as chaparral, Great Basin sagebrush, or pinyon-juniper woodlands. Research over the past few decades has established that these ecosystems naturally and historically burned infrequently, with several decades or even centuries between fires (Floyd et al. 2004, Baker 2006, Mensing 2006, Keeley and Zedler 2009, Baker and Halsey 2020). In these ecosystems, the use of prescribed fire can shorten fire-free intervals needed for slow-growing shrubs to re-establish, and such fire conducted outside of the natural fire season can inhibit seed germination for many species while favoring growth of non-native plants that can be more flammable (Parker 1987, Le Fer and Parker 2005, Baker 2006, Syphard et al. 2006).

Prescribed fire can also be harmful in forests, especially when done outside of the main fire season. Wildlife have evolved strategies to coexist with summer forest fires, but prescribed fires are often lit in the spring when bird eggs and nesting chicks cannot get away from fires (Hanson 2021a). There can also be significant ecological damage when logging (“thinning”) is treated as a prerequisite for prescribed fire. On national forests, this means that Forest Service projects involving prescribed fire can also include substantial amounts of logging that damages wildlife habitat. Furthermore, logging results in cut vegetation debris that then gets piled up and burned. *The Forest Service is now calling this pile burning*

“prescribed fire,” but it causes sustained burning in a concentrated location that can scorch and sterilize the soil, and it does not produce the beneficial post-fire wildlife habitat created during genuine forest fire restoration (Korb et al. 2004). In light of these potential harms, projects involving prescribed fire should not be exempted from proper environmental review.

Prescribed fire and cultural burning are not the same.

As attention on prescribed fire has grown in recent years, there has also been broader awareness that Native Americans have long traditions of applying fire to the land in practices known as cultural burning. Cultural burning was outlawed or severely restricted when federal and state policymakers imposed their wildfire suppression policies on Indigenous peoples. Now some Indigenous groups seek to restore their ability to do more cultural burning. Traditionally, burning has been done for a variety of purposes, including stimulating the growth of plants that are particularly useful to their communities, such as for basket-weaving or food production. In this regard, traditional cultural burning is notably different from current approaches to prescribed fire from the Forest Service and other agencies that primarily focus on trying to suppress forest fire intensity across vast areas. Yet, the Forest Service’s approach to prescribed fire often gets conflated with traditional cultural burning in discussions of fire policy. This can result in cultural appropriation that superimposes Forest Service goals on Tribal practices. The evidence is clear that, prior to modern fire suppression policies, *Native American cultural burning and mixed-intensity forest fires were both much more common than they are now* (Odion et al. 2014, 2016, Vachula et al. 2019, Wahl et al. 2019). *They coexisted, and one did not preclude the other.* Both have been suppressed and marginalized by federal and state agencies.

Conclusion—The disappointing results of prescribed fire

When all is said and done, *the actual results from broad-scale application of prescribed fire would likely be disappointing for most people.* Those who thought it would reduce fire would instead experience more fire and smoke from large-scale prescribed burning. Those who are concerned about public safety would realize that communities would have been much safer if the money used to subsidize backcountry prescribed fires and associated “fuel treatments” had instead been focused on directly assisting with fire-safety home retrofits as part of a home-outward strategy. Those who want to help ecosystems would realize that managed wildfire offers a more efficient and practical way to restore fire to forests, whereas prescribed fire is often tied to increased logging. And those who want to support Tribes’ cultural burning would find traditional practices getting appropriated by federal and state agencies. *While prescribed fire can have some benefits in special circumstances, it is important to not overstate the role of prescribed fire—a watering can should not pretend to be a river—or we risk missing more effective and cost-efficient solutions using managed wildfire, traditional cultural burning, and home fire-safety retrofits.*

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Special Section:

Fire in the Earth System

Key Points:

- Approximately 11% of the Oregon Cascades burned during September 7–9, 2020 coincident with strong offshore downslope winds
- Unprecedented compound extremes involving fuel aridity and fire meteorology facilitated the extent and spread of fires
- Very large fires in western Oregon since 1900 have generally occurred during east wind events at the end of anomalously warm-dry summers

Supporting Information:

Supporting Information may be found in the online version of this article.

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Compound Extremes Drive the Western Oregon Wildfires of September 2020

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Abstract Several very large high-impact fires burned nearly 4,000 km² of mesic forests in western Oregon during September 7–9, 2020. While infrequent, very large high-severity fires have occurred historically in western Oregon, the extreme nature of this event warrants analyses of climate and meteorological drivers. A strong blocking pattern led to an intrusion of dry air and strong downslope east winds in the Oregon Cascades following a warm-dry 60-day period that promoted widespread fuel flammability. Viewed independently, both the downslope east winds and fuel dryness were extreme, but not unprecedented. However, the concurrence of these drivers resulted in compound extremes and impacts unmatched in the observational record. We additionally find that most large wildfires in western Oregon since 1900 have similarly coincided with warm-dry summers during at least moderate east wind events. These results reinforce the importance of incorporating a multivariate lens for compound extremes in assessing wildfire hazard risk.

Plain Language Summary Several very large fires in western Oregon spread rapidly during an unusually strong offshore wind event that commenced on Labor Day in 2020. The Labor Day fires burned more area of the Oregon Cascades than had burned in the previous 36 years combined and very likely exceeded the area burned in any single year for at least the past 120 years. The fires damaged over 4,000 structures, led to several fatalities, placed over 10% of the state's residents under some level of evacuation advisory, and contributed to the hazardous air quality across the Northwestern United States. A compound set of weather-related factors leading up to and during the fires facilitated these extreme fires. Unusually warm conditions with limited precipitation in the 60-days leading up to the fires allowed for fuels to become particularly dry and combustible by early September. Downslope offshore winds materialized during September 7–9, 2020 across the Oregon Cascades bringing exceptionally strong winds and dry air that drove rapid rates of fire spread. While neither of these individual factors was unprecedented, the concurrence of these drivers created conditions unmatched in the observational record.

1. Introduction

On Labor Day, September 7, 2020, exceptionally strong east-to-northeast winds developed from Washington state progressing southward in the following days driving extreme fire activity in Washington, Oregon, and California. In western Oregon, the winds fanned several existing small, smoldering wildfires ignited by lightning several weeks prior, and contributed to numerous new human-caused fires that spread rapidly. Dry katabatic winds drove fire spread toward populated areas in the ensuing three days, hindering fire suppression efforts already strained by the numerous concurrent large fires burning across much of the western United States during a fire season of record high atmospheric aridity in much of the region (Higuera & Abatzoglou, 2020). About 40,000 people in Oregon were forced to evacuate their homes, and roughly 500,000 were under some level of evacuation alert during the event. In total, 393,315 hectares burned across 20 fires, all but a few in the Cascades of western Oregon (Figure 1; supporting information). Moreover, over 11% of the Oregon Cascades ecoregion burned in 2020, exceeding the cumulative burned area in the previous 36 years (Figure 1; supporting information) and very likely exceeding annual burned area in western Oregon since at least 1900. The direct cost of the series of fires was estimated at \$1.15B USD, which includes the destruction of over 4,000 homes (State of Oregon, 2021).

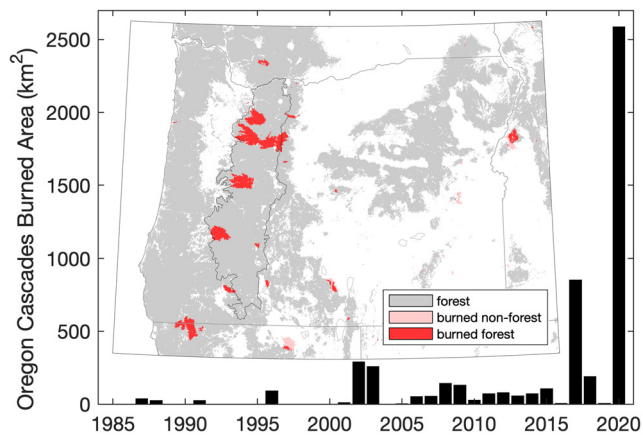


Figure 1. Time series of annual burned area for the portion of the Cascades ecoregion in the State of Oregon for 1984–2020 (black polygon in inset map). Inset map shows perimeters of fires in 2020, including much of the forested burned area in western Oregon that burned during the 7–9 September period.

Sustained strong winds are a major contributor to rapid rates of fire spread, particularly in the presence of dry fuels (Rothermel, 1972). Of particular concern are downslope mountain winds that induce adiabatic warming and entrainment of drier air leeward of mountain barriers. Such winds are a well-recognized critical fire weather pattern in various parts of the globe (Nauslar et al., 2018; Sharples et al., 2010). Offshore downslope winds are a catalyst for large autumn wildfires in California due to the co-occurrence of offshore winds that increase throughout the autumn with dry fuels prior to the arrival of winter precipitation (Guzman-Morales et al., 2016; Kolden & Abatzoglou, 2018). While downslope offshore wind events west of the Cascade Range in the Pacific Northwest are less frequent than counterparts in California (Abatzoglou et al., 2020), significant east wind events have been implicated in major historic fires in western Oregon and Washington (Beals, 1914; Cramer, 1957; Dague, 1930). The earlier arrival of cool-season precipitation and higher fuel moisture in forests along the western flank of the Oregon Cascades compared to California in late summer limit the co-occurrence of extreme fire weather conditions with critically dry fuels.

Traditional univariate hazard assessment approaches for evaluating extreme events such as fire conditioned by interacting drivers, may be ill-posed (Seneviratne et al., 2012); rather, multivariate approaches that

incorporate the geographic synchrony, concurrence, and succession of extremes are needed to adequately characterize hazards (Sadeh et al., 2018). Compound drivers have been long implicated in wildfire, including in the mesic forests of the Pacific Northwest where infrequent large high-severity fires resulted from warm-dry summers that enable fuel receptiveness and high winds that drive fire spread (Agee, 1996)—yet a systematic analysis of compound climate and weather factors in very large fires in western Oregon is lacking. Conceptual models of fire regimes consider the co-occurrence of biomass abundance, availability to burn, ignitions, and weather (Bradstock, 2010) and large wildfire events and seasons have been linked to compound drivers. For example, compound humidity and wind speed extremes are inherent features of many critical fire weather patterns (Crimmins, 2006), while the combination of critically dry fuels imparted by antecedent conditions combined with fire weather extremes have been connected with large fires and large fire seasons (e.g., Khorshidi et al., 2020; Williams et al., 2019). This study builds on previous efforts by applying different approaches for compound extremes that supported the growth of arguably the most exceptional fire outbreak in the Oregon Cascades in over a century.

In this study, we analyze the meteorological factors responsible for fire spread and antecedent climate factors that enabled fuel aridity of the wind-driven fires in the western Oregon Cascades whose explosive growth commenced on Labor Day 2020, hereafter referred to as the Labor Day fires. We first examine the synoptic and mesoscale meteorological conditions during the Labor Day fires and the rarity of individual meteorological and fuel aridity factors. We additionally use different approaches for quantifying the rarity of this event through the lens of compound extremes that combine fuel dryness and fire meteorology. Lastly, we systematically compare the compound climate and meteorological conditions during the Labor Day fires to other very large wildfire events in western Oregon since 1900.

2. Data and Methods

Meteorological and climatological data were obtained from several sources to address our research questions. First, we acquired hourly surface observations for operational automated surface observation stations (ASOS) and remote automated weather stations (RAWS) across Oregon from MesoWest (Horel et al., 2002) to examine patterns of near-surface wind and vapor pressure deficit (VPD) during the event. Second, 0Z wind speed and relative humidity observations at 1,000 hPa from version 2 of the Integrated Global Radiosonde Archive (Durre et al., 2006) for Salem, Oregon for the 64-year period 1957–2020 were used to contextualize meteorological conditions during the event against the longer-term data record. Third, daily 1/24th degree gridded meteorological data were acquired from gridMET (Abatzoglou, 2013) and used to calculate

two fire danger indices from the U.S. Fire Danger Rating System (Cohen & Deeming, 1985): Energy Release Component (ERC) and Burning Index (BI) for a commonly used fuel model (dense conifer) for 1979–2020. These data were used to contextualize fire danger during the event related to the past 42 years. Fourth, reanalysis products from the 20th Century Reanalysis v3 (Compo et al., 2011) during 1900–1947, NCEP-NCAR during 1948–present, and ERA5 (Hersbach et al., 2020) were used to examine mid-to-lower tropospheric synoptic conditions during the event, as well as to provide context relative to the longer-term record. Finally, gridded 1/24th degree monthly total precipitation and monthly mean daily maximum temperature for the period 1895–2020 were obtained from PRISM (Daly et al., 2008) for comparison of antecedent climate conditions during previous large fire events.

We use 700 and 850 hPa zonal wind velocity from reanalysis data as a simple and transparent indicator of cross-barrier downslope winds for the primarily north-south oriented Cascades rather than the more refined diagnostic approaches for downslope winds developed previously (Abatzoglou et al., 2020; Guzman-Morales et al., 2016). This was done by tabulating the maximum easterly wind velocity in the lower troposphere 700 or 850 hPa near or just above the crest of the Oregon Cascades at 6-h intervals at two points: one in the Oregon Cascades at 45°N, 122.5°W, and the other in southwestern Oregon at 42.5°N, 122.5°W.

We accompany meteorological diagnostics with two fire danger indices—ERC and BI. ERC is proxy for fuel dryness and potential energy release at the front of a fire that is a function of antecedent temperature, precipitation, and humidity from previous several weeks and is independent of wind speed. ERC has been shown to influence large fire probability and interannual variability in regional area burned (Abatzoglou & Kolden, 2013; Barbero et al., 2014). BI combines ERC and potential rate of fire spread (chiefly a function of wind speed and fuel moisture) into a metric that is a proxy for flame length and fireline intensity. We examined statistics of BI as it provides a natural single metric of compound extremes of fuel moisture, atmospheric humidity, and wind speed that are of particular relevance for wind-driven fires.

We complement the assessment of compound extremes using rank/order statistics of interrelated drivers that allow for a more flexible treatment of the resultant hazard than functional equations built into a single index like BI. Two combinations were examined: (i) the joint distribution of 0Z relative humidity and wind speed at 1,000 hPa from Salem, Oregon radiosonde observations during 1957–2020; (ii) the joint distribution of ERC averaged for the Oregon Cascades ecoregion and lower-tropospheric easterly wind speed from reanalysis during 1979–2020. The former captures purely meteorological conditions, while the latter captures antecedent climate and meteorological conditions. Order/rank statistics have 1:1 accordance with probabilistic hazard assessment scenarios (Salvadori et al., 2016). While univariate order/rank statistics correspond directly to probability $P(X < x)$, multivariate order/rank statistics can take on several definitions. Herein, we follow previous studies that have used a bivariate case that corresponds to $P(X > x \text{ AND } Y > y)$. In this case, pairs are ranked as: $Z_i = [X_i, Y_i]$ dominates $Z_j = [X_j, Y_j]$, if X_i dominates X_j AND Y_i dominates Y_j (Agha-Kouchak et al., 2014; Alizadeh et al., 2020).

Lastly, we compiled a list of very large western Oregon wildfire events (35,000 ha) since 1900 (Table S1). We additionally included the 2017 Eagle Creek fire (19,761 ha) because it was the most recent large fire in the Oregon Cascades prior to 2020. Climate summaries for August, July–August, and June–August temperature and precipitation were aggregated from the counties in which each of large wildfire events occurred (Table S2). These periods were chosen because most of the very large fires ignited in August and/or September. For each fire event, we tabulated the maximum 6-h easterly wind during the estimated duration of the fire at 700 or 850 hPa from 20th Century Reanalysis and NCEP-NCAR for events prior to 1948 and those since 1948, respectively. We note there is some uncertainty about the extent and duration of the earliest fires, as well as potential omission of large fire events in the first-half of the 20th century due to poor documentation. The Labor Day fires of 2020 were contextualized with other very large fires within the trivariate space of antecedent temperature, antecedent precipitation, and concurrent easterly wind.

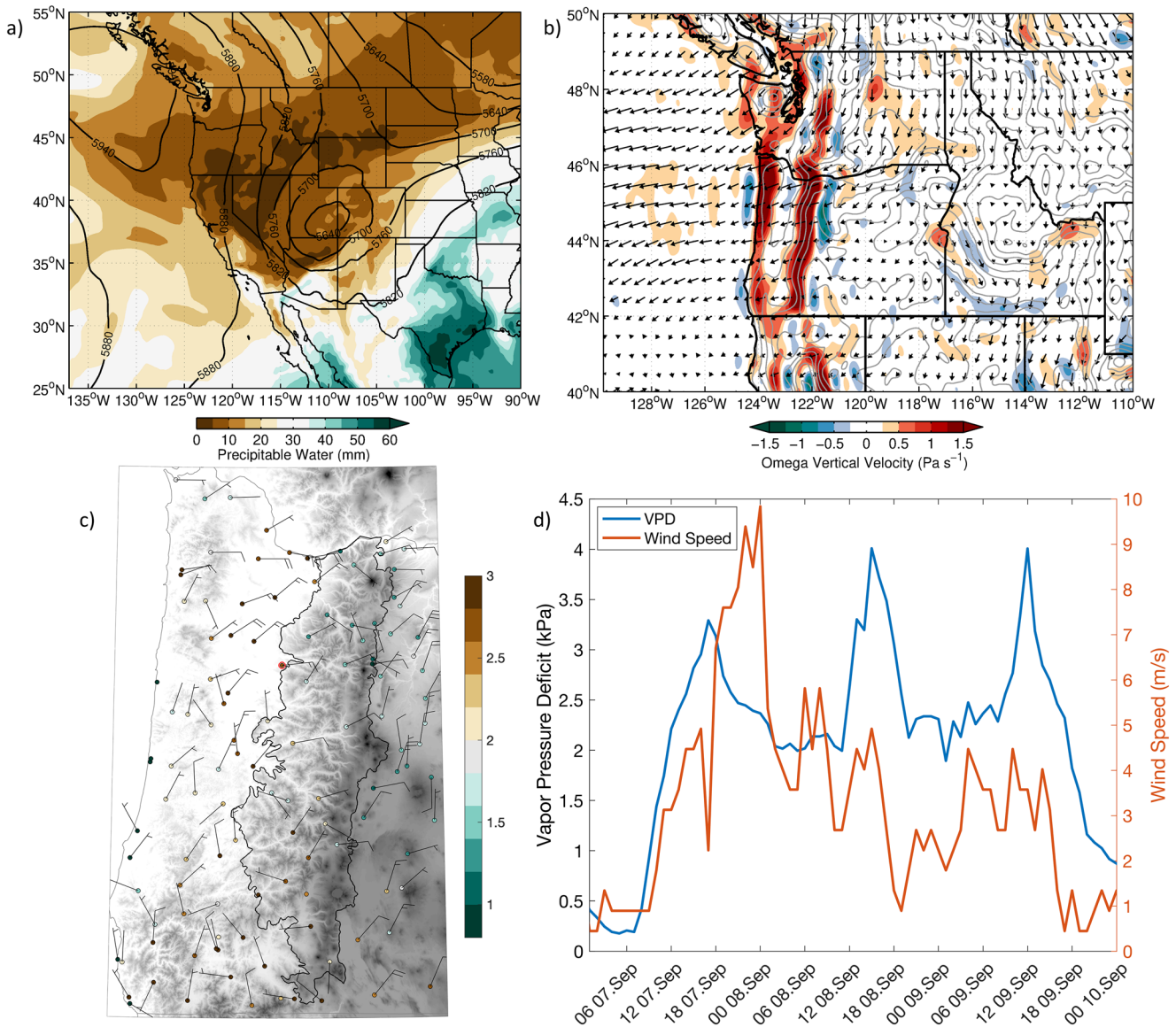


Figure 2. (a) Precipitable water (shading) and 500 hPa geopotential heights (contours, m), and (b) 850 hPa winds (vectors) and pressure vertical velocity (shading) at 0Z on 8 September 2020 (17 LST on 7 September). In (b), Gray contours show elevation, and red hues show downward motion while blue hues show upward motion. (c) Surface-based observations of wind speed (knots) and vapor pressure deficit (VPD, colors) at 20 LST on September 7, 2020. Gray shading in (c) depicts topography and black contour shows the Oregon Cascades ecoregion. (d) Time series of hourly wind speed and VPD from the Jordan RAWS station (red circle in panel c) during 7–9 September.

3. Results and Discussion

Critical fire weather conditions developed from eastern Washington state southward into western Oregon and northern California during September 7–8, 2020 resulting from an amplified wave across the Pacific-North American sector. An omega block off the coast of British Columbia and cut off low pressure near the U.S. four corners facilitated northeasterly flow and an intrusion of exceptionally dry air throughout much of the northwestern United States (Figure 2a). The onset of strong east-to-northeast (ENE) winds advanced from the northern portion of the Oregon Cascades during the afternoon of the seventh through southwestern Oregon into the morning of the eighth. Strong ENE cross-barrier lower-tropospheric winds developed across the Oregon Cascades exhibiting a classical signature of downslope winds and mountain lee wave evident by the wave pattern in ERA5 pressure vertical velocity fields and downward transport of strong easterly winds and dry air with relative humidities less than 10% in the lee of the Cascades (Figures 2a, 2b

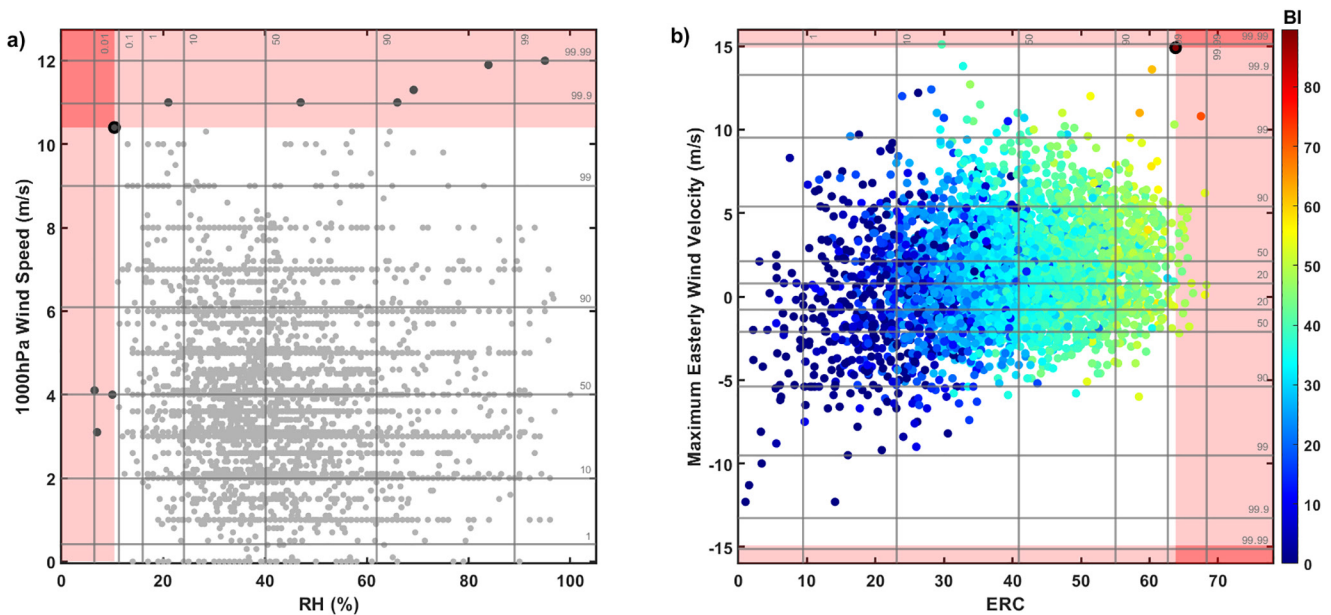


Figure 3. Scatter plots of (a) wind speed and relative humidity (RH) at 1000 hPa from the Salem, Oregon radiosonde for all July–September days during 1957–2020 at 0Z, and (b) daily maximum 6-h easterly wind velocity at 700/850 hPa from NCEP–NCAR reanalyses and daily energy release component (ERC) averaged over the Oregon Cascades for all July–September days during 1979–2020. Negative easterly wind velocity refers to winds with a westerly component. Data in (b) are color-coded with daily burning index (BI) averaged over the Oregon Cascades. Points marked with black edge show data for 0Z September 9, 2020 (17 LST September 8, 2020) in (a) and 8 September 2020 in (b). Light-red shaded areas highlight the univariate space and events that exceed this point, while the dark-red shaded areas show the space that dominates this point in a bivariate AND-scenario case. Light gray horizontal and vertical lines show percentile values. Percentiles in both panels were computed relative to the entire July–September period of record. Percentiles are shown for wind speed in (b) rather than exclusively easterly wind velocity.

and S1). ERA5 precipitable water values were less than 10 mm over all of Oregon and upstream over much of the U.S. west (Figure 2a) on the afternoon of 7 September. The mesoscale manifestation of this pattern at 20 LST on September 7, 2020 shows widespread elevated surface VPD west of the Cascades and substantial heterogeneity in 10-m wind speeds imparted by topographic station siting (Figure 2c). The strongest low-level winds were primarily constrained to the evening of 7 and 8 September. Elevated VPD was seen for much of western Oregon for a sustained 2.5 day period starting the afternoon of the seventh and with little nocturnal recovery (Figure 2d).

Several individual meteorological elements of this event qualify as extremes. Observations of 1,000 hPa relative humidity and wind speed from the Salem sounding at 0Z 9 September (17 LST 8 September) were the fourth lowest and seventh highest values, respectively, for July–September in the 64-year period of record (Figure 3a). Importantly, extreme low RH conditions co-occurred with extreme high winds yielding the most extreme combination of dry-windy conditions for any July–September day during 1957–2020 when viewed through the bivariate lens (Figure 3a). Using the NCEP–NCAR reanalysis record from 1948–2020 and our diagnostic of offshore downslope winds, we find that September 8, 2020 had the second strongest easterly winds at 700 or 850 hPa for any July–September day (Figure S2).

Fire danger indices of ERC and BI remained near their 1981–2010 average values for much of July and early August across western Oregon (Figures 4c and 4d). However, for much of August and first week of September, unusually warm temperatures, elevated VPD, and negligible precipitation facilitated escalating fuel dryness and well above normal ERC in parts of western Oregon by early September (Figures 4a and 4c). On 8 September, both ERC and 700/850 hPa easterly wind speeds exceeded their respective 99th percentile values for July–September (Figure 3b). Similar to the meteorological drivers, the co-occurrence of extremely dry fuels and strong east winds resulted in a compound extreme unprecedented in the 42-year period of record, although none of the individual drivers were the most extreme. Complementary to rank/order statistics, we find widespread record BI across much of western Oregon on September 8, 2020 (Figure 4b) with 3 of the top 10 highest days of BI for the Oregon Cascades occurring during September 7–9, 2020 (Figure 4d). Hence,

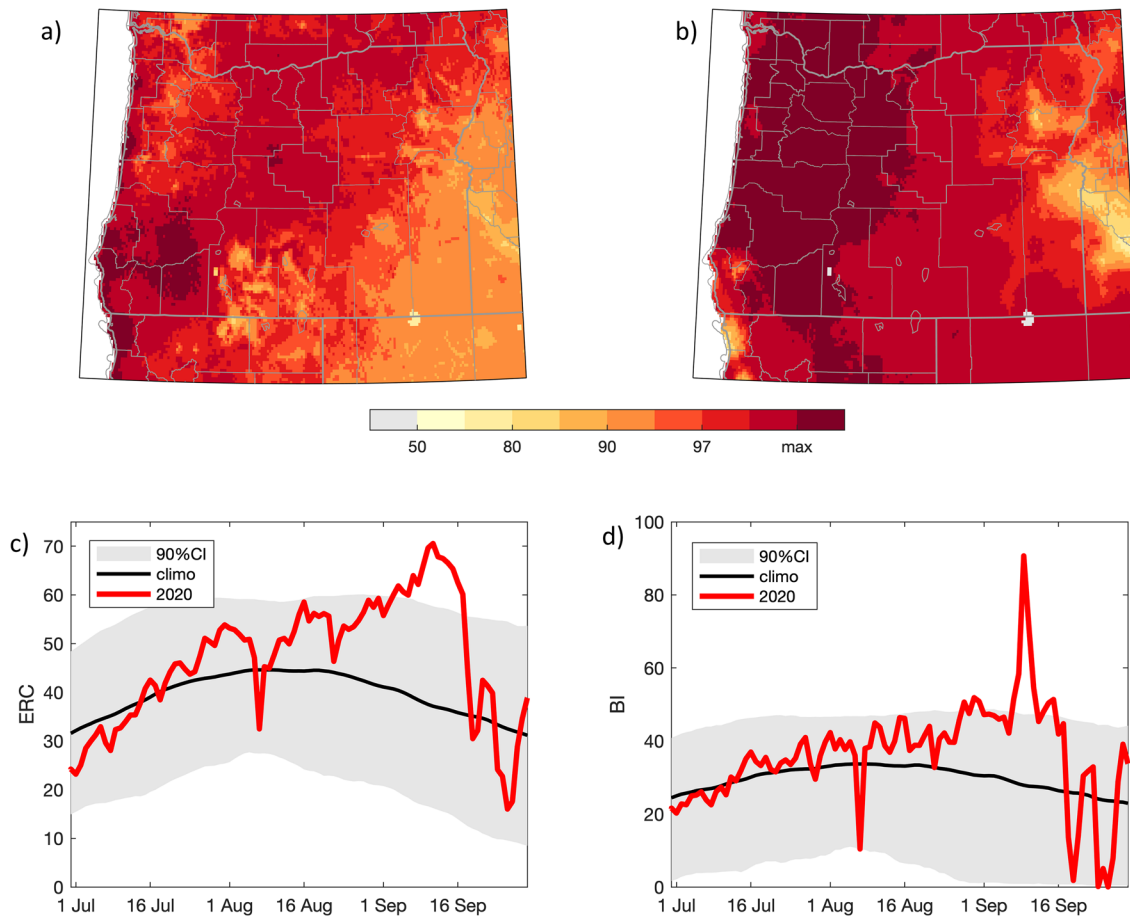


Figure 4. Percentiles of (a) Energy Release Component (ERC), and (b) Burning Index (BI) for September 8, 2020. Percentiles were tabulated based on data pooled over the entire calendar year 1979–2020. Time series of (c) ERC and (d) BI aggregated over the Oregon Cascades for 2020 (red), compared both the climatological 1981–2010 averages (black) and smoothed 5th to 95th percentile values (gray shading).

both approaches for quantifying compound extremes as a product of fuel dryness and potential rate of fire spread highlight conditions during the Labor Day fires as the most extreme in the observational record.

The majority (10 of 13) of very large fire events in western Oregon since 1900 were associated with a July–August that had below-median precipitation and above-median maximum temperatures (Figures 5, S3 and S4). Qualitatively similar results were found using June–August and August temperature and precipitation data, although with somewhat weaker results for precipitation (Figure S5). More recently, all seven very large fire events since 1987 had July–August precipitation at or below the 17th percentile and maximum temperature above the 70th percentile. Notably, July–August 2020 was neither the driest nor the warmest per these metrics.

All 13 very large fires were associated with a period of easterly lower-tropospheric wind during the fire event (Figure 5; see also Figure S6). The synoptic configuration corresponding to the days during each large fire event with the strongest easterly lower tropospheric flows showed similarity among previous events to the Labor Day fires (Figure S6) featuring a pronounced ridge off the coast, weak thermal trough along the Oregon coast, and northeastward gradient in mean sea level pressure across the region. No other very large fire had an easterly wind as strong as the 2020 Labor Day fires, nor were they associated with a 500-hPa level ridge and northeastward sea-level pressure gradient as strong. We note that potential inhomogeneities in reanalyzes products inhibited a robust comparison with fields such as wind velocity (Wohland et al., 2019; Figure S7). Notably, the 1902 Columbia fire had the second strongest easterly wind, yet the fire stands out by having neither a dry nor a warm July–August lead-up period (Figure 5), suggesting that it was more strongly a product of immediate meteorological conditions (Beals, 1914; Plummer, 1912). Previous

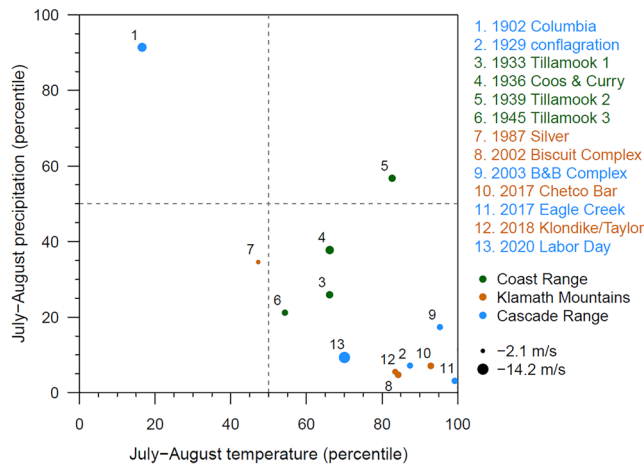


Figure 5. July–August total precipitation versus July–August mean daily maximum temperature for the years of the large wildfires in western Oregon listed in Table S1. Values are given as percentiles of the anomalies for the years 1895 through 2020 (126 years). The percentiles shown for each event are calculated with respect to the anomalies for the county/counties where the event occurred. Symbols are sized relative to the peak easterly wind component at 700 or 850 hPa at 45°N, 122.5°W from 6-hourly NCEP-NCAR reanalysis (1948–present) and 20th Century reanalysis (1900–1947).

studies have highlighted nonclimatic factors as contributors to large fires in the first half of the 20th century, primarily logging practices that increase dead fuel loading that help facilitate flammability in the absence of anomalously warm and dry summers (Dague, 1934; Kallander, 1953).

4. Conclusions

A series of factors contributed to the extent of fire impacts during the 2020 Labor Day fires in western Oregon that burned ~11% of the Oregon Cascades. A sequence of climate and weather factors enabled and drove rapid rates of fire spread following conceptual models of fire activity (Bradstock, 2010). Moreover, while individual factors such as humidity, wind speed, and fuel dryness were extreme, neither individually ranked as the most extreme on record. In contrast, compound extremes of fuel dryness and low-level wind speed, as well as wind speed and relative humidity, yielded conditions that were unprecedented in the contemporary data record. The extent of burned area across mesic forests of western Oregon during the 2020 Labor Day fires has no contemporary local analog. Very large fires (>200,000 ha) were documented in western Oregon in the 19th century coinciding with European colonization (Morris, 1934), although uncertainty of the location, timing, and contributing factors limit a more detailed comparison. The 2020 Labor Day fires do bear some resemblance to extremely large autumn wind-driven fires in California in terms of rapid rates of fire spread and materialization of compound extremes driven by offshore downslope winds at the end of the dry season

when fuel moisture is critically low (Nauslar et al., 2018). Nonclimatic contributions to these fires imparted by biomass accumulation and land management practices (Reilly et al., 2017) also likely factored into this severity of fire impacts, although we do not speculate on their specific influences.

Very large fires with rapid rates of spread and extreme fire behavior are gaining attention as environmental phenomena that impact society and Earth system processes (Bowman et al., 2009; Khaykin et al., 2020). While the details of the meteorological, fuels, and topographic drivers of such extreme fires and their impacts vary (Tedim et al., 2018), requisite conditions of flammable contiguous fuels imparted by fuel drying enable extreme fires while meteorological conditions including strong winds or convection often drive rapid rates of fire spread (Bowman et al., 2017; Rothermel, 1991). Our findings of the Oregon 2020 Labor Day fires complement previous studies on compound drivers for autumn wind-driven fires in California (Khorshidi et al., 2020; Williams et al., 2019) and the development of fire early-warning systems (Jolly et al., 2019) while showing the potential use of the approach for individual fire events. Moreover, the approach used herein to quantify this extreme wind-driven fire events through the lens of compound extremes may have value in other regions with shared biophysical and human constraints on fire activity.

While unprecedented in the modern record, extreme events of this nature with widespread impact evoke questions on causality—namely the extent to which human-caused climate change contributed to the strength or probability of the event, as well as such events will become increasingly probable in the coming decades. A formal attribution exercise is beyond the scope of this analysis, yet studies suggest that climate change has contributed to increased fuel aridity and longer fire seasons (Abatzoglou & Williams, 2016) and the probability of compound hot-dry extremes (Alizadeh et al., 2020) and climate projections suggest continued warming with slight decreases in summer precipitation in the Pacific Northwest (Rupp et al., 2017) over the 21st century. By contrast, offshore winds such as Santa Ana winds in southern California may become less frequent as a result of asymmetric continental warming with anthropogenic forcing (Guzman-Morales & Gershunov, 2019; Hughes et al., 2011). However, few studies have formally evaluated these two contrasting drivers in a compound extremes framework (Goss et al., 2020). Projected changes in the confluence of extreme fire weather patterns and fuel dryness will be critical for resolving future fire related hazards for mesic forests in western Oregon and Washington state.

Data Availability Statement

Datasets used herein were acquired from the following public data repositories: (1) ERA5: <https://cds.climate.copernicus.eu/>; (2) NCEP-NCAR reanalysis: <https://psl.noaa.gov/data/gridded/data.ncep.reanalysis.html>; (3) twentieth Century Reanalysis v3: https://psl.noaa.gov/data/gridded/data.20thC_ReanV3.html; (4) gridMET: http://thredds.northwestknowledge.net:8080/thredds/reacch_climate_MET_catalog.html; (5) ASOS and RAWS station data: <https://mesowest.utah.edu/>; (6) PRISM data: <https://prism.oregonstate.edu/>.

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Review

Countering Omitted Evidence of Variable Historical Forests and Fire Regime in Western USA Dry Forests: The Low-Severity-Fire Model Rejected

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Abstract: The structure and fire regime of pre-industrial (historical) dry forests over ~26 million ha of the western USA is of growing importance because wildfires are increasing and spilling over into communities. Management is guided by current conditions relative to the historical range of variability (HRV). Two models of HRV, with different implications, have been debated since the 1990s in a complex series of papers, replies, and rebuttals. The “low-severity” model is that dry forests were relatively uniform, low in tree density, and dominated by low- to moderate-severity fires; the “mixed-severity” model is that dry forests were heterogeneous, with both low and high tree densities and a mixture of fire severities. Here, we simply rebut evidence in the low-severity model’s latest review, including its 37 critiques of the mixed-severity model. A central finding of high-severity fire recently exceeding its historical rates was not supported by evidence in the review itself. A large body of published evidence supporting the mixed-severity model was omitted. These included numerous direct observations by early scientists, early forest atlases, early newspaper accounts, early oblique and aerial photographs, seven paleo-charcoal reconstructions, ≥18 tree-ring reconstructions, 15 land survey reconstructions, and analysis of forest inventory data. Our rebuttal shows that evidence omitted in the review left a falsification of the scientific record, with significant land management implications. The low-severity model is rejected and mixed-severity model is supported by the corrected body of scientific evidence.

Keywords: ponderosa pine; mixed conifer; dry forests; historical range of variability; fire regime; low-severity fire; mixed-severity fire; high-severity fire; forest structure



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1. Introduction

1.1. The Issue of Historical Forest Structure and Fire in Dry Forests

Wildfires are increasing in dry forests of the western USA, particularly from climate change [1], with dire concerns for the nearby built environments facing serious fire risk to lives and property [2]. The scientific basis for this difficult nature–people wildfire problem has recently been compounded by omitted and false evidence in a review by Hagmann et al. [3] of historical forest structure and fire in western USA forests, including dry forests, the subject of rebuttal in this paper. Dry forests cover ~25.5 million ha of the western USA [4] and include ponderosa pine (*Pinus ponderosa*) forests (Figure 1) and dry mixed-conifer forests (ponderosa with several associated trees). Dry forests often begin at the lower limit of tall trees and extend upward, forming the lower part of the montane zone, covering plateaus and mesas as well as mountain slopes, often bordered below by woodlands (*Pinus-Juniperus*) or semiarid sagebrush (*Artemisia* spp.) shrublands and grasslands.



Figure 1. A typical older ponderosa pine forest. Photo by W.L. Baker.

The fire problem in dry forests is a wicked problem [5], as much is at stake for people and nature, and the issues are difficult. Dry forests have become favored but dangerous locations for development, which is continuing expansion in the wildland–urban interface (WUI) [6]. Increasing wildfires ignited in wildland vegetation, including dry forests, spill over into expanding WUI-built environments [2]. There are significant concerns about ecological effects (e.g., [7]) and cost and effectiveness [8] of fire management.

This difficult situation is also the setting for what has been a lengthy scientific debate about historical forest structure and fire in dry forests. Since the 1990s, there has been competing evidence that dry forests historically were either (1) open, low-density forests maintained by frequent low-severity fires (e.g., [9]), called here the “low-severity fire” model, although limited moderate-severity fire is included, or (2) had both low- and high-density forests with a mixture of fire severities, including high-severity fires (e.g., [10–12]), the “mixed-severity fire” model. Evidence in this debate has become extensive and was reviewed previously by Odion et al. [12] and Hanson et al. [13]. A large body of evidence for both models has developed from local [14,15] and landscape-scale studies [11,16]. Findings have been challenged and rebutted in a complex series of papers, comments, and rebuttals (e.g., [17–19]). Hagmann et al. [3] do not call it the low-severity-fire model, but they explain that “high-severity fire is overrepresented in forests historically characterized by frequent low- to moderate-severity fire regimes” [3] (p. 13).

In general, sound evidence about the historical structure of dry forests and the ecosystem processes shaping them remains important today in understanding and managing dry forests, especially as climates change. Evidence about the historical range of variability

(HRV) of ecosystems provides an essential frame of reference for restoring and adapting ecosystems to maintain biological diversity and ecosystem services as climates change [20]. By historical, we mean prior to the expansion of industrial development and displacement of Indians, usually in the late 1800s in the eleven western states where dry forests occur. Biological diversity has embedded genetic composition from long-term response to HRV where organisms live, which constrains their ability to rapidly adapt to a changing climate, forest structure, and fire [21]. This makes departures from HRV undesirable for biological diversity, including dominant dry-forest trees, and heightens the importance of HRV as a major basis for sound forest management.

The fire regime is particularly a focus of HRV research in western North America because this area retains substantial original, if modified, vegetation that developed under the influence of wildfires and other natural disturbances. Wildfires have been modified by fire management (including logging), human-set fires, livestock, and climate change.

1.2. Rationale and Methodology of This Paper

Here, we rebut evidence in a recent review (Hagmann et al. [3], “H” hereafter) of HRV for forest structure and fire, focused on dry forests in North America. H extensively critiqued the mixed-severity-fire model and was a major basis for associated policy proposals [22,23] that are now disputed [7]. Here, we critically examine and rebut evidence against the mixed-severity model. This makes H a logical focus as they support only the low-severity model. We are authors of many papers critiqued in H, including its focus on the Williams and Baker [24] method (“WB” hereafter) of reconstructing historical forest structure and fire from US General Land Office (GLO) land survey data from the late 1800s; thus, we think it is appropriate to take this critical approach to evidence in H. The WB method is a new method using design-based statistical theory and Voronoi polygons to reconstruct forest structures from GLO bearing-tree data [24]. Science can benefit from a detailed examination of evidence about methods and models. We think it will be easier for readers to compare evidence and arguments if we follow H’s structure, including the tables they used to present evidence. The result is not a systematic or full review but instead a focused rebuttal of evidence. The result is a review of the substantial evidence that supports the mixed-severity model and rejects the low-severity model of HRV for forest structure and fire in dry forests of western North America.

Our method, which is generally replicable, given the focus in H on WB methods, was to seek key published rebuttals and original evidence relevant to each item in Tables 3–6 in H, as well as other evidence and critiques of the WB method in H’s text. Since our authorship includes the two original authors of the WB method and authors of other rebuttals of published studies by authors of H, we know all our original and rebuttal evidence. Evidence from other sources was primarily cited in these papers.

We found that evidence in H omitted major bodies of published evidence that do not support the low-severity model. H said our publications misrepresented the state of the science. Here, we refute this and show that it is the evidence in H where falsification occurred. Unfortunately, omissions and false evidence in H became a theme because these are so numerous and significant. H claimed the WB method should be evaluated with independent evidence, not WB’s evidence, but the validity of the WB method is not scientifically evaluated if original validations and rebuttals of critiques of the WB method are omitted, as H did. As we show, H not only omitted key original and rebuttal evidence about the WB method but also large bodies of independent evidence by other authors, including authors of H, that do not support the low-severity model.

We present three sections (Sections 2–4 below) to make the case about false evidence and omitted evidence in H. In Section 2, we show that an expansion of evidence about HRV for forest structure and fire in dry forests since the 1990s was omitted in H. In Section 3, we show that the evidence that H presented to argue that high-severity fire has been burning recently at rates that exceed historical rates shows the opposite. In Section 4, we refute evidence that H presented to argue that the WB method of reconstructing historical

forest structure and fire from land survey data is flawed. We show that all the evidence that shows that the WB method is valid and accurate was omitted in H. These sections together show the substantial evidence that rejects the low-severity model and supports the mixed-severity model.

2. Expanding Sources of Evidence about Dry-Forest HRV since the 1990s Omitted in H

H was rooted in earlier and now-disputed evidence about historical fire and forest structure in dry forests. Up to about the middle 1990s, fires in dry forests were often thought to have historically burned frequently and mostly at low to moderate severity in low-density dry forests dominated by large trees (Figure 1), as found in parts of the southwestern United States (e.g., [9]) (Table 1, Group 1). Fire histories in these forests often assumed low-severity fire dominated and did not reconstruct fire severity, as they were targeted at old park-like forests with long fire-scar records, which inherently lacked much higher-severity fire for long periods [25]. Dominance by low- to moderate-severity fire and low tree density were also found in some parts of land survey reconstructions, but other parts of these landscapes were found to have higher-severity fire and denser forests (Figure 2), so low- to moderate-severity fires were 85–98% of fires (Table 1, Group 1). Tree-ring [26] and land survey reconstructions agree that tree density was typically low, with ~140 trees/ha overall in these areas (Table 1, Group 1). These forests were in the driest areas, mostly in New Mexico and Arizona, but scattered in other states [25]. Thus, it is not in dispute that substantial area in historical dry forests, particularly in the driest areas of the southwest, had low- to moderate-severity fire and open, low-density forests. The issue is that evidence now shows that even these seemingly stable areas with low-severity fires, and other parts of these landscapes, at times also had infrequent high-severity fires and denser forests [27], as shown in Figure 2.

Table 1. The 15 dry-forest landscapes with reconstructions from US General Land Office survey data based on WB methods.

Study Area	Source	Area (ha)	Historical Fire Severity				Historical Tree Density (Trees/ha)			
			Low (%)	Mixed (%)	Low + Mixed (%)	High (%)	High Rotat. (yrs)	Mean	Median	C.V. (%)
Group 1—Mostly low- to moderate-severity fires and low tree density										
Mogollon Plateau, AZ	Williams and Baker [24]	405,214	62.4	23.1	85.5	14.5	828	141	124	54
Coconino Plateau, AZ	Williams and Baker [27]	41,214	58.8	38.7	97.5	2.5	2000 ¹	142	121	46
Group 2—Mostly moderate- to high-severity fires and intermediate to high tree density										
W Sierra, CA-North	Baker [28]	133,482	12.6	48.2	60.8	39.2	281	318	229	106
W Sierra, CA-South	Baker [28]	196,461	26.4	42.5	68.9	31.1	354	275	191	203
W. Sierra, CA—Greenhorn Mts. ²	Baker and Hanson [17]	49,050	-	-	-	26.9 ³	-	280 ⁴	-	-
Blue Mts., OR	Williams and Baker [24]	304,709	40.3	43.2	83.5	16.5	849	167	146	54
E Cascades, OR-N	Baker [29]	146,555	32.5	44.2	76.7	23.3	515	246	211	-
E Cascades, OR-C	Baker [29]	147,502	10.4	48.2	58.6	41.4	278	262	215	-
E Cascades, OR-S	Baker [29]	104,160	29.4	61.7	91.1	8.9	1180	233	224	-
Siskiyou Mts., OR ²	Baker [30]	46,445	-	-	-	-	-	272	253	-
Group 3—More high-severity fires and intermediate tree density										
Black Mesa, AZ	Williams and Baker [24]	151,080	12.0	32.8	44.8	55.2	217	144	137	47
Front Range, CO	Williams and Baker [24]	65,525	2.5	32.9	35.4	64.6	271	217	162	100
Front Range, CO-Section-line area	Williams and Baker [31]	624,156	-	-	-	-	249 ⁵	-	-	-
Uncompahgre Plateau, CO ⁴	Baker [32]	227,036	0.0	28.7	28.7	71.3	175	182	183	86
San Juan Mts., CO ²	Baker [33]	235,787	29.2	32.5	61.7	38.3	240	191	118	99

¹ This estimate was not reported in Williams and Baker [27] because so few (about 2.5%) high-severity fires were found. In order to have complete data, a rough estimate of 2000 years was used here. ² This study area is new since Baker and Williams [34] classified the other 11 landscapes, but it appears to best fit this group. ³ This is not from GLO survey data but instead a count of early timber inventory transects recording evidence of high-severity fire. ⁴ All estimates are the mean between the ponderosa pine and dry mixed-conifer estimates.

⁵ This estimate, the only one based on section-line data, is for “higher” severity fires, meaning pooled moderate- to high-severity fires, so the number is shown under each category but applies to them jointly.

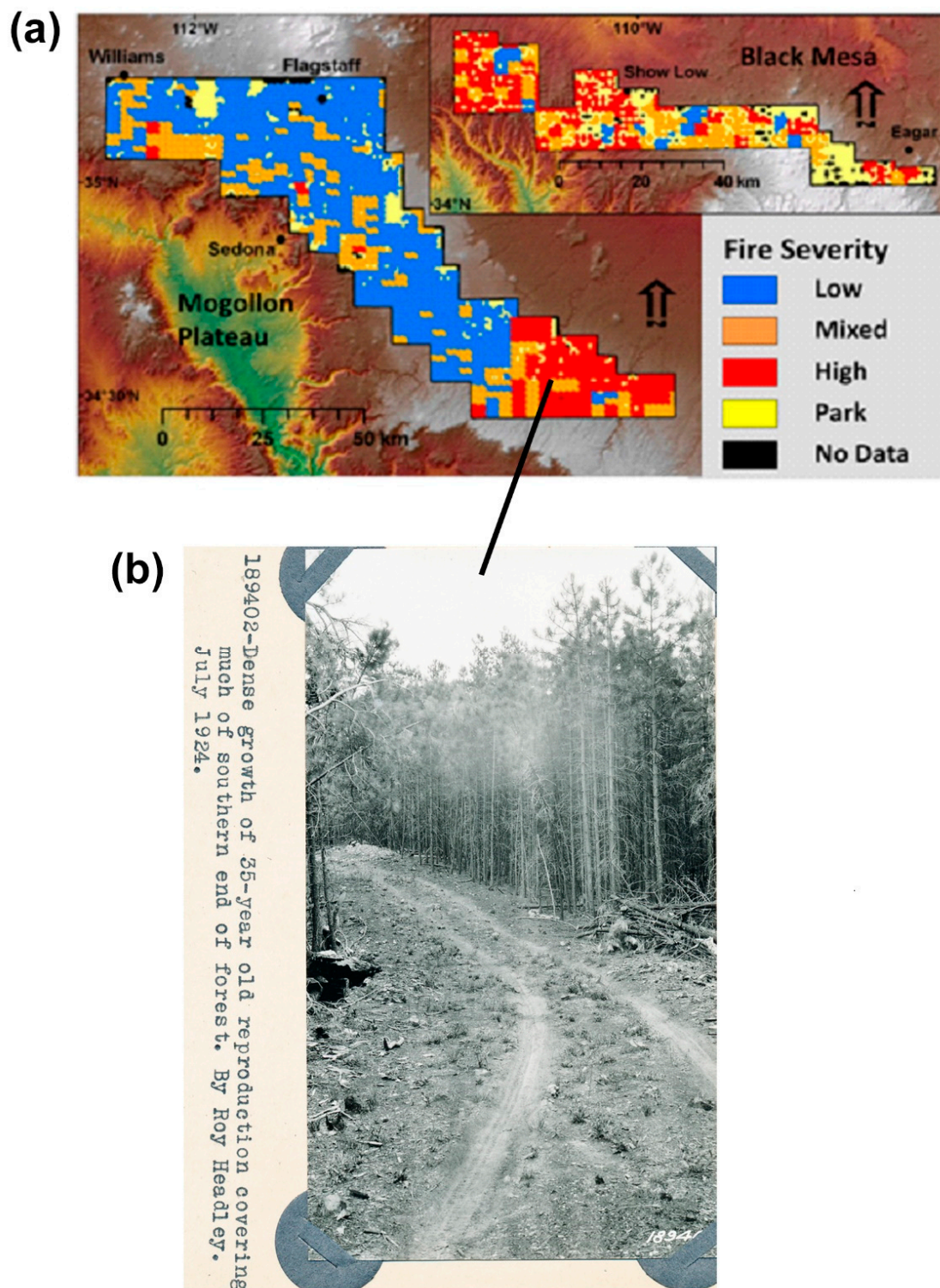


Figure 2. (a) “Fire severity evidence from forest structure, based on survey reconstructions on the Mogollon Plateau and nearby Black Mesa, Arizona”. Note the extensive mixed- to high-severity fire on Black Mesa compared to the more extensive low- to mixed-severity fire on the Mogollon Plateau. Reprinted from [24] with permission from John Wiley and Sons, and (b) a photograph taken in 1924 by Roy Headley, Historical Photo Collection, Region 3, U.S. Forest Service, Albuquerque, New Mexico.

This was discovered because fire histories expanded, between the late 1990s and the 2010s, from just fire scars in small plots, the primary basis for H's review and the low-severity model, to the landscape scale and due to the use of multiple sources of evidence (Table 2). This research found more severe historical fires, including moderate- (20–70% tree mortality) and high-severity (>70% tree mortality) fires and denser and more heterogeneous dry forests, even within Southwest landscapes but especially outside driest areas (Table 1, Group 2 and 3 landscapes). These findings were extensive enough to generally support the mixed-severity model. Sources included early photographs (Figure 3a), maps, and reports, including forest reserve reports [10,35], age structures showing trees likely killed by fire and regenerating abundantly after fire (Figures 2b and 3b), paleo-charcoal reconstructions [36,37], and early aerial photos (Figure 4). Where combinations of sources were available (Figures 2 and 5), validation is enhanced, as H mentioned. For example, Figure 2 shows that where a land survey reconstruction found that the southern end of the Mogollon Plateau had mixed- and high-severity fire in the late 1800s, an old oblique photograph shows dense tree regeneration, without large surviving trees, several decades later. In another example, Figure 5 shows that dry forests mapped as “severely burned” timber [38] ca 1900 were recorded by land surveyors in preceding decades as mature timber. This is independent multi-proxy evidence that the low-severity model is not supported, since mature forests historically burned with high severity at times.

Since historical fires in dry forests were previously considered nearly all low severity, it was surprising at this time to discover this was not the case over large areas. Hessburg et al. [11] used early aerial photography to reconstruct historical fires from forest structures across 178,902 ha of ponderosa pine and Douglas-fir cover types in eastern Washington and Oregon mixed-conifer forests (Figure 4). They explained their findings:

Table 2. Sources of evidence about historical forest structure and fires in dry forests, and selected methods papers and examples.

Historical Forest Structure	
Tree-ring reconstructions from remnant evidence	Fulé et al. [39]
Land survey reconstructions from bearing-tree data	Williams and Baker [24,40,41]
Early forest reserve reports	Baker et al. [35]
Early records from forest inventories	Baker and Williams [34]
Early aerial photographs	Hessburg et al. [11]
Early oblique photographs	Veblen and Lorenz [42]
Historical Fires	
Tree-ring reconstructions from fire scars—small plots	Baisan and Swetnam [43]
Tree-ring reconstructions from landscape-level data	Farris et al. [15]
Land survey reconstructions from bearing-tree data	Williams and Baker [24,40,41]
Land survey reconstructions from section-line data	Williams and Baker [31]
Forest Inventory and Analysis reconstructions	Odion et al. [12]
Paleo-charcoal reconstructions	Pierce et al. [36]
Early records from newspaper accounts	Baker [44]
Early scientific publications	Baker [44]
Early forest reserve reports	Baker et al. [35]
Early forest atlases	Baker [44]
Early oblique photographs	Baker [45]
Early aerial photographs	Hessburg et al. [11]

“The structure of mixed conifer patches, in particular, was formed by a mix of disturbance severities . . . evidence for low-severity fires as the primary influence, or of abundant old park-like patches, was lacking in both the dry and moist mixed conifer forests. The relatively low abundance of old, park-like or similar forest patches, high abundance of young and intermediate-aged patches, and widespread evidence of partial stand and stand-replacing fire suggested that variable fire severity and non-equilibrium patch dynamics were primarily at work” [11] (p. 5) and also: “ . . . before any extensive management had

occurred, the influence of fire in the dry forest was of a frequency and severity that intermittently regenerated rather than maintained large areas of old, fire tolerant forest” [11] (p. 19).

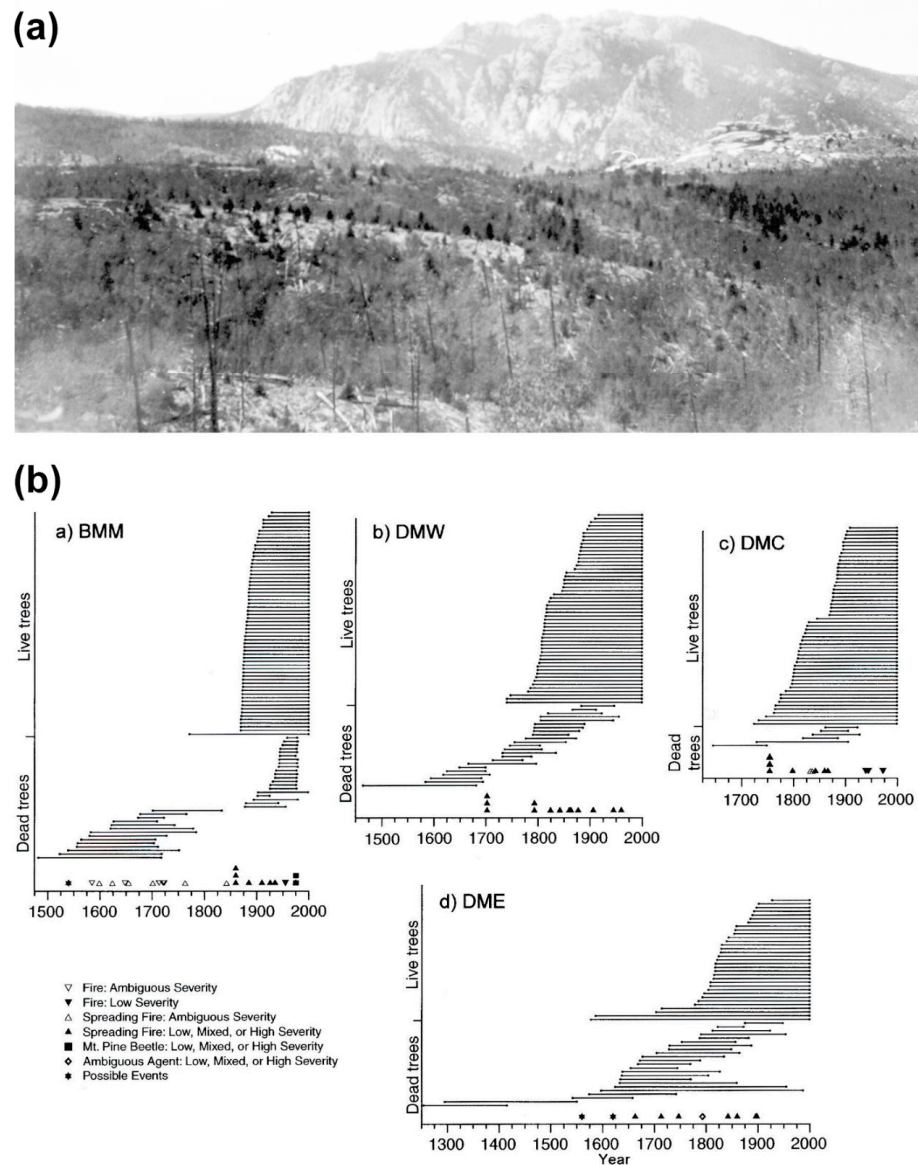


Figure 3. Two sources of evidence, early oblique photographs, and tree-ring reconstructions of age structure and fire scar dating of high-severity fires in dry forests in the Colorado Front Range: (a) high-severity fire identified by Jack [46] as in ponderosa pine and subsequent ponderosa pine (dark black) and quaking aspen regeneration in central Colorado in the Plum Creek Reserve; original photo, taken 18 August 1889, probably by John Jack, labeled “Looking north at Devils Head (Platte) Mt. from east side”. Original photo in the National Archives, FRA no. 008, (b) event diagram for four sample plots in which a tree-ring reconstruction of age structure of live and dead trees was completed in dry forests in Rocky Mountain National Park, Colorado. The severity of an event is indicated by the number of symbols stacked vertically: 1, low severity; 2, mixed severity; and 3, high severity. For example, in plot BMM, a high-severity fire was identified in the 1870s by a fire scar, extensive regeneration after the fire, and dead trees before the fire. Each line shows the lifespan of one tree. Reprinted from ([14] Figure 2) with permission from John Wiley and Sons.

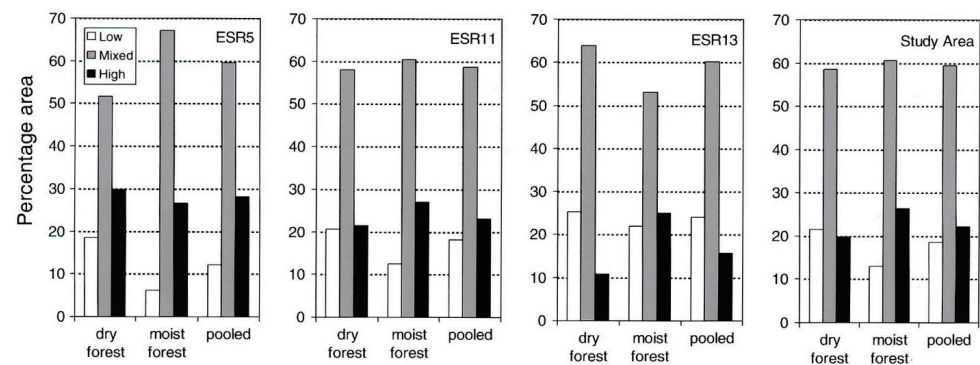


Figure 4. The proportions of the pre-management-era forest area (ha) by forest potential vegetation type in low-, mixed-, and high-severity fire (corresponding with percent canopy mortality values of $\leq 20\%$, 20.1–69.9%, and $\geq 70\%$, respectively) of Ecological Subregions 5, 11, and 13 and the study area. Comparisons are shown for the dry and moist forest potential vegetation types and pooled (sum of dry and moist). Note the dry-forest columns in each subregion and in the study area as a whole. Reprinted from Hessburg et al. ([11] Figure 5 part) with permission from Springer Nature.

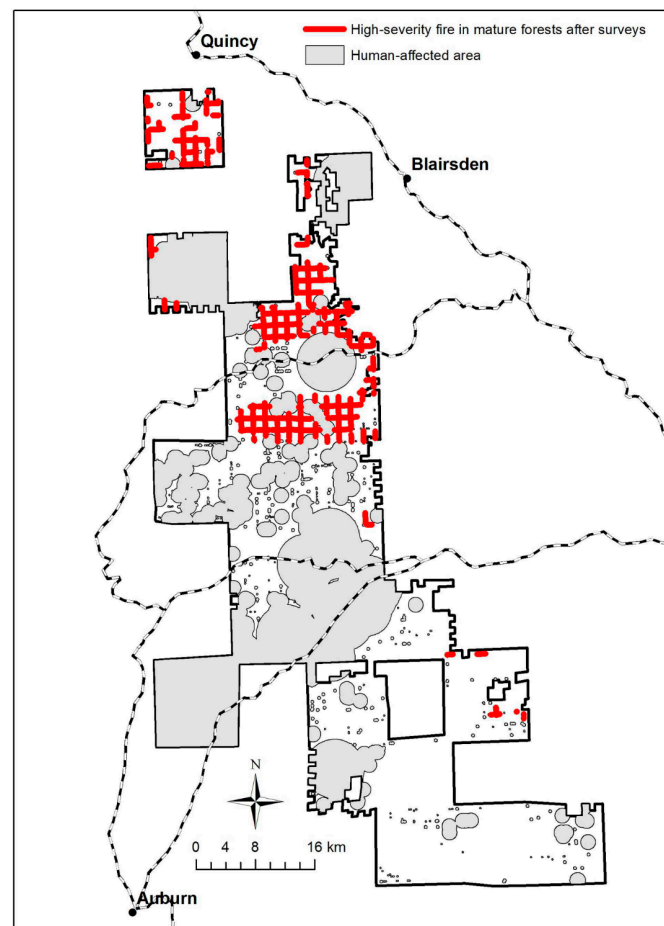


Figure 5. Areas of mature Sierran mixed-conifer forest burned at high severity after the surveys and before Leiberg's mapping ca 1900 [38]. The Leiberg 75–100% burned category from 1900 was overlain on the survey section-line data from 1865 to 1890. Section lines shown in red were described by surveyors as “heavily timbered”, “good timber”, or “excellent timber”, and thus as mature forest in 1865–1890 before Leiberg mapped these areas in 1900 as severely burned. Reprinted from Baker ([28] Figure 8) with permission from John Wiley and Sons.

In the 2010s, Williams and Baker [24,40,41] developed new methods to use General Land Office (GLO) surveys from the late 1800s to systematically reconstruct historical forest structures (e.g., tree density and basal area) and fire rates and severities from bearing-tree records across large landscapes (Figure 2a). They also validated that historical fire rotations and patch sizes could be reconstructed from GLO section-line data [31]. GLO survey reconstructions have been completed for 15 landscapes across > 2.5 million ha of dry forests of the western USA (Table 1, Figure 6). In eastern Oregon, GLO data were shown to be consistent with the findings of Hessburg et al. [11], corroborating both [29].

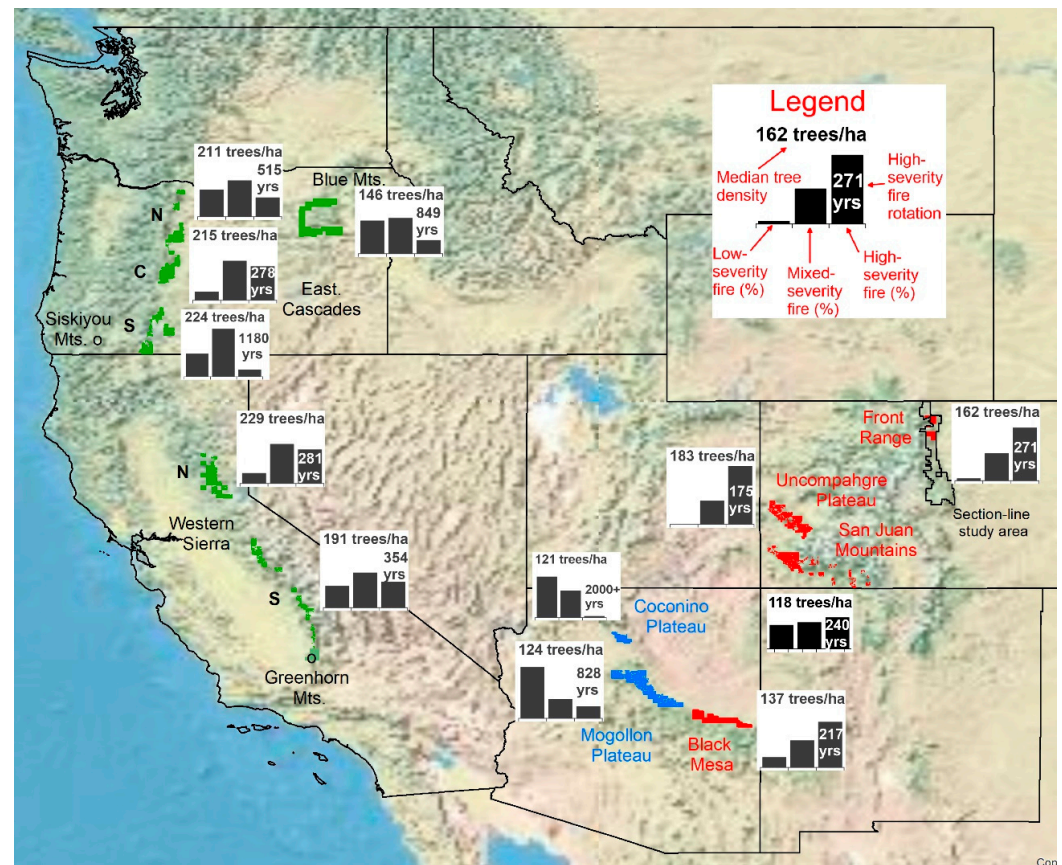


Figure 6. The 15 dry-forest landscapes with reconstructions from U.S. General Land Office survey data based on WB methods. The Greenhorn Mts., CA, and Siskiyou Mts., OR, study areas are small and do not have full reconstructions. The Front Range section-line study area, used in the [31] analysis, is shown by an outline.

During this same period, extensive research on methods of fire history reconstruction from fire scars and tree age structures led to improvements that enabled accurate estimation of fire rotation, the essential spatial measure of fire rates [47,48]. Earlier small-plot “composite fire interval” (CFI) methods, a primary basis for the low-severity model, created a pooled list of fire years from scars and then calculated mean, median, and other statistics from the list (e.g., [43]). However, these early CFI estimates, often called mean fire-return intervals (MFRIs), were further tested, found to overestimate rates of fires, and new, more accurate methods for small plots were derived [49–51].

Even more important, landscape-scale methods were also developed that do not depend on calculating intervals between fires in small plots. They instead use plots to simply detect and map fire-year occurrence, enabling the creation of landscape-scale fire-year maps (Figure 7) from which fire rotations can be directly estimated [15,52]. Inaccurate CFI methods were no longer needed. It also became possible to correct inaccurate CFI estimates to fire rotations via regression. This enabled historical fire rotations for low-

severity fires to be estimated and shown across dry forests (Figure 8). This evidence alone, omitted in H, showed that frequent low-severity fire (rotations < 25 years) was found across only ~14% of historical dry forests [25]. Forest atlases, mapped in the early 1900s, provided another source of landscape-scale maps [44], showing historical moderate- to high-severity fires in the 1800s (Figure 9). These also validated GLO methods of fire-history reconstruction [33]. Artificial intelligence methods [16,53] also have made it feasible to expand fire history evidence from plots across large land areas (Figure 10).

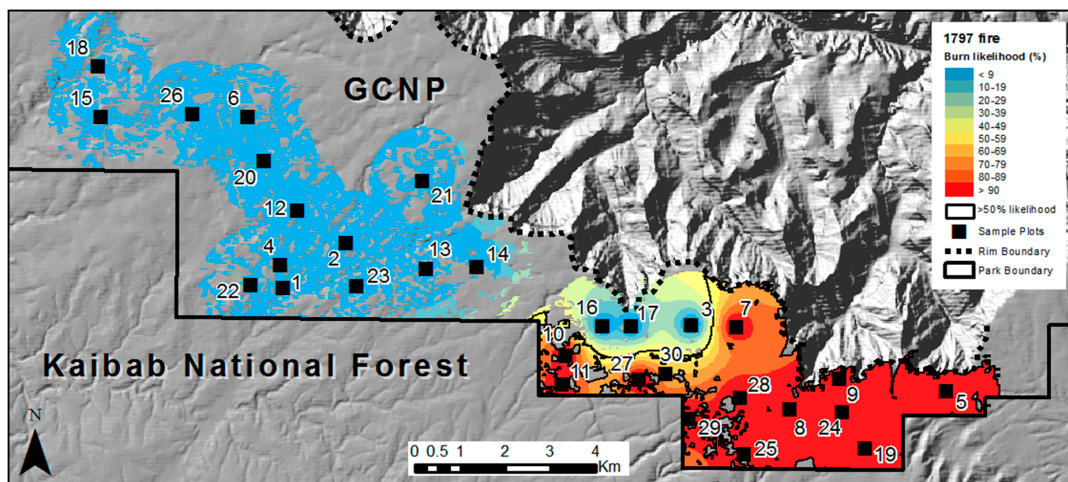


Figure 7. An example fire-year map for an 1829 fire on the south rim of Grand Canyon, Arizona. The reconstructed fire area was derived using inverse distance weighting based on plot records that showed where the fire did and did not burn. Reprinted from ([52] Figure S1) with permission from John Wiley and Sons.

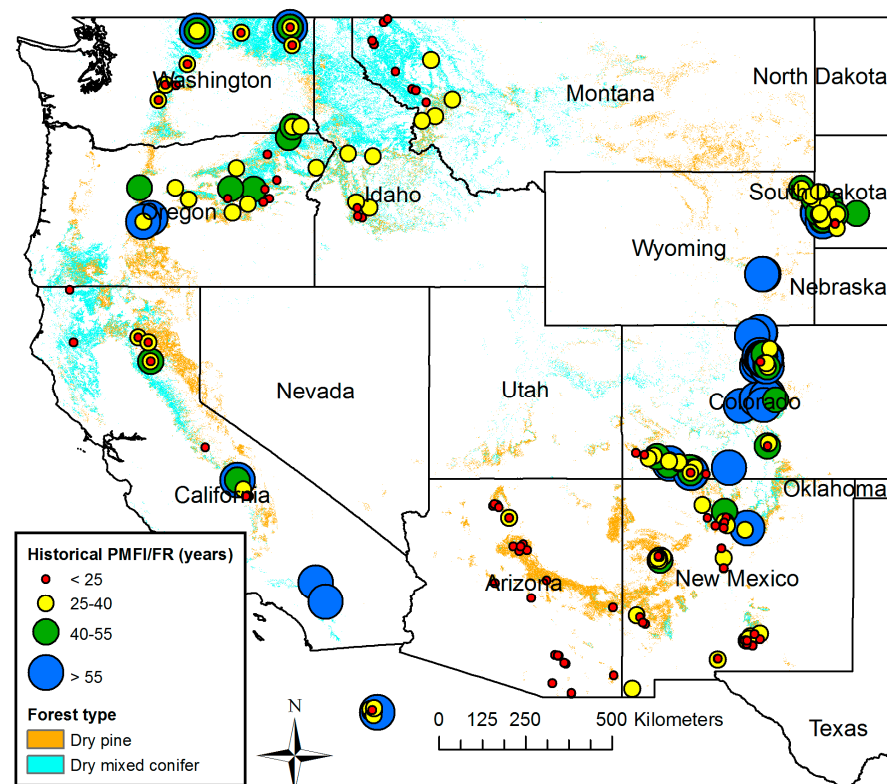


Figure 8. “Estimated historical low-severity population mean fire interval/fire rotation (PMFI/FR) for the combined set ($n = 342$) of calibration cases and prediction sites in dry forests of the western USA”. Reprinted from [25] with permission from PLoS ONE.

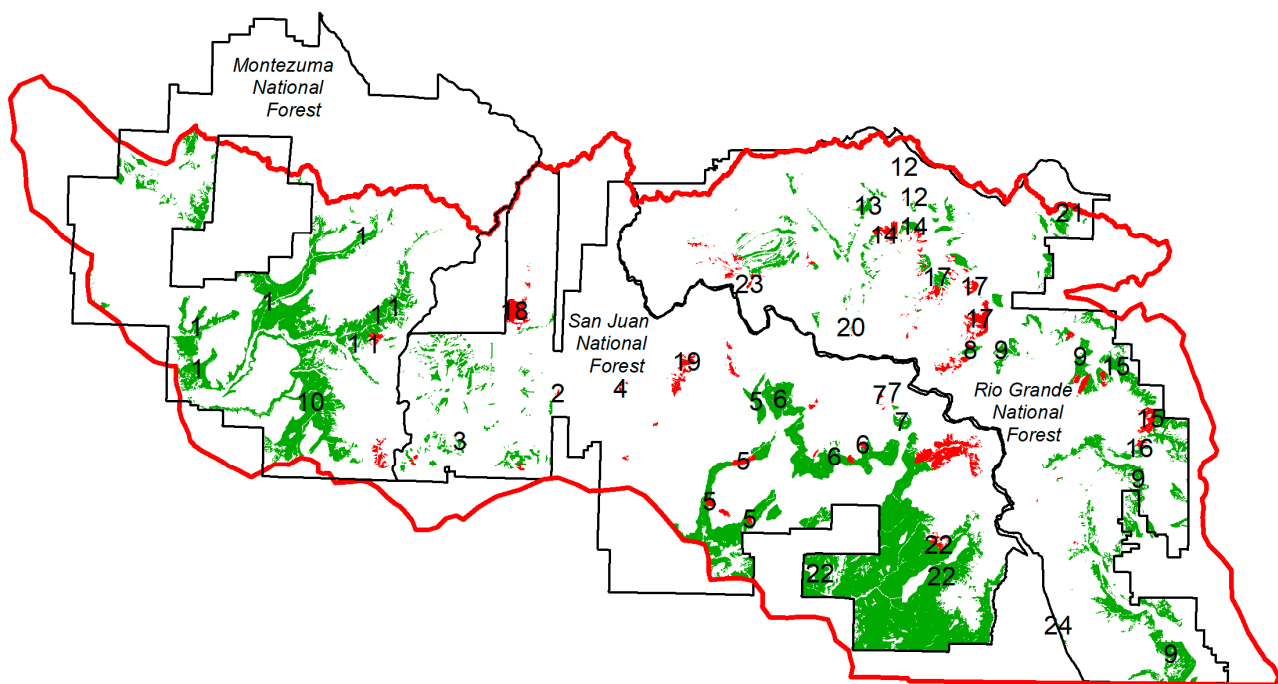


Figure 9. “The three atlas boundaries (black lines) and the fires (red) and woodlands (green) shown on the atlases with the area of ponderosa pine and mixed-conifer forests. Fires were likely mostly stand-replacing, and woodlands likely moderate- to high-severity fires, 1850–1909. Fire and woodland numbers are used in tables and text”. The red boundary is the study area in the southwestern San Juan Mountains, Colorado. Reprinted from ([44] Figure 1b) with permission of MDPI.

H omitted this history of development of new fire history and forest structure reconstruction methods, correction of old fire history methods, and addition of new sources (Table 2), including findings of limited areas (14%) of frequent low-severity fire and extensive evidence of widespread mixed-severity fire with infrequent moderate- to high-severity fire and denser dry forests. H instead used (1) only the early inaccurate CFI estimates of fire rates, which are documented to overestimate rates of fire unless corrected [25,47], and (2) just two sources, tree-ring reconstructions and modeling, when there are multiple sources of evidence (Table 2). We show here that H omitted most evidence since the mid-1990s that improved fire history methods and understanding of historical dry forest structure and fire, showing more evidence of mixed-severity fires. Evidence in H omitted even their own authors’ use of these new methods and their own findings of historically moderate- to high-severity fires and denser forests.

Given the large body of research and multiple methods (Tables 1 and 2) since the mid-1990s, evidence for the mixed-severity model of historically heterogeneous dry forests and mixed-severity fire has a sufficiently compelling evidence basis to qualify as an established model. Before the advances since the 1990s, there was also evidence in support of the low-severity model that historical dry forests were more uniform, low-density forests with predominantly low- to moderate-severity fire. H, for the first time, synthesized evidence against the mixed-severity and in support of the low-severity model.

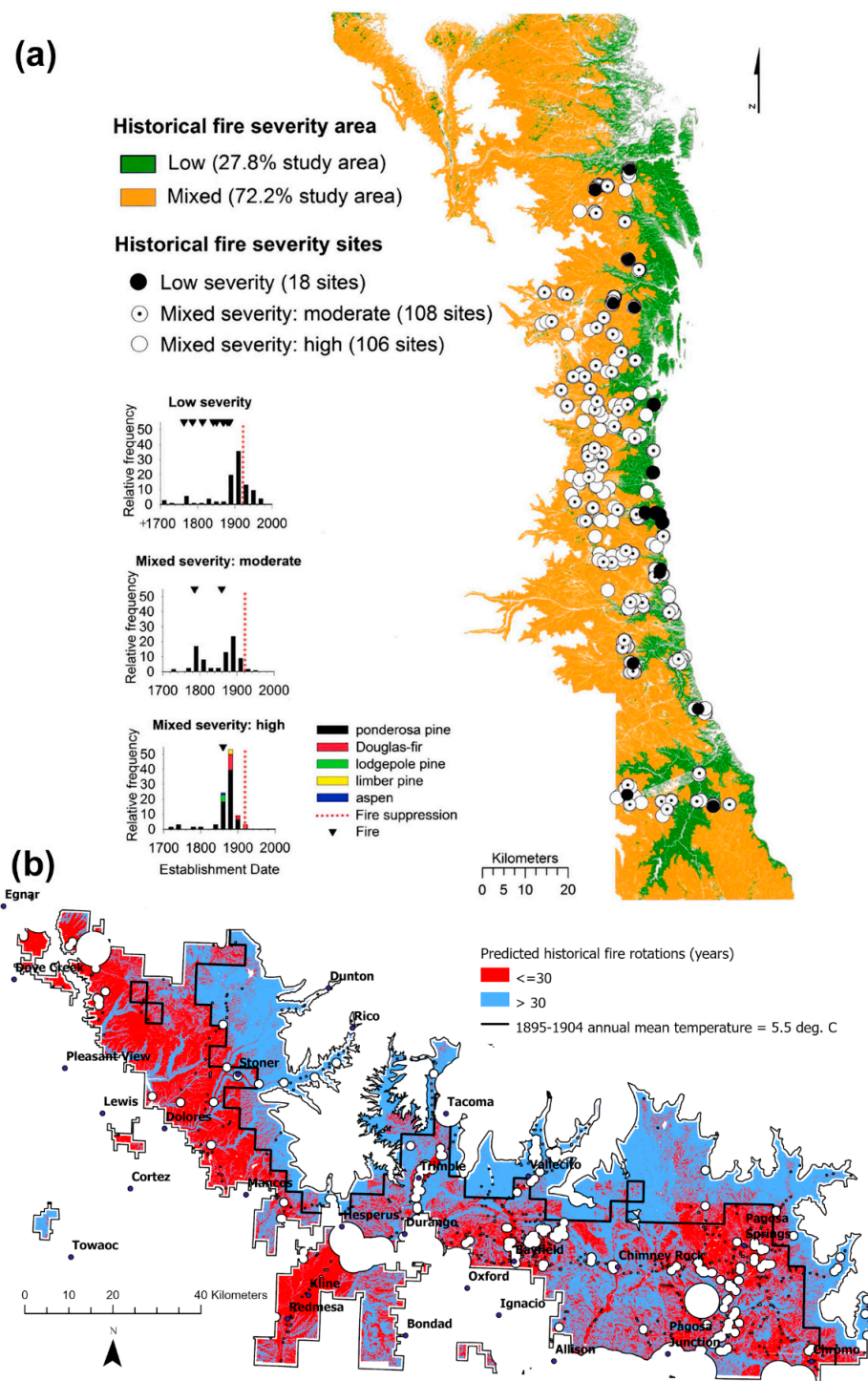


Figure 10. (a) “Distribution of 232 sites with historical (pre-1920) evidence of low-severity and mixed-severity fires.” This is a map of the montane zone in the Colorado Front Range, showing on the left the evidence used to classify the fire regime into low or mixed severity at three example sites. The map shows the result of using classification and regression trees (CARTs) to extrapolate from classified sites to map the two fire regimes across the whole landscape using physical predictors (e.g., elevation and slope). Reprinted from ([16] Figure 5) with permission of PLoS ONE, (b) “Relatively frequent fire (all-severity fire rotations ≤ 30 years) versus longer-rotation fire in historical montane forests overlain by the contour for 5.5 °C annual mean temperature between 1895 and 1904, which roughly corresponds with the upper limit of relatively frequent fire”. This map was derived by using random forest modeling of 28 tree-ring-based fire history sampling sites versus 14 topographic, soils, and climate predictors. Reprinted from ([53] Figure 5a) with permission of John Wiley and Sons.

3. Debunking H's: "Fire Regimes Are Significantly Departed"

H presented in their Table 2 [3] (pp. 14–15) a list of 14 publications they claimed show that "high-severity fire effects in recent fires exceed the pre-fire exclusion range of historical variation in landscapes historically dominated by frequent low- and moderate-severity fires." If this were true, it would imply that recent wildfires are too severe and warrant substantial active management via fuel reduction as claimed, but this is not the case, as all but 3 of the 14 publications were inadequate in area or data. The three adequate ones found the opposite: high-severity fire was occurring at or below historical rates.

To have an adequate sample area for this question, study areas need to be several times larger than the largest recent fires, which means generally on the order of 250,000 ha or more [4], because if only one or a few fires nearly fully burned an area, then $n = 1$ or a small number, and the sample is inadequate. Of the 14 cited studies, 1 was just a review, 8 had inadequate sampling areas <25,000 ha, and another [54] contained no evidence about historical fires. Nigro and Molinari [55] did not even report the area burned at high severity, so their finding was unusable. A total of 11 of 14 studies (79%) cited in H had an inadequate area or lacked necessary area-burned data. The remaining 3 studies [56–58] had study areas $\geq 250,000$ ha and reported area burned, meaning ample sample area and data.

These three studies found that high-severity fire was recently burning at similar or lower rates than historically, not at higher rates recently, as reported in H: (1) Reilly et al. [56] found that the 1985–2010 high-severity fire rotation in ponderosa pine forests was 1693 years vs. historical 705–849-year rotations, so the recent high-severity fire rate was deficient relative to historical rates; (2) Haugo et al. [57] did not calculate fire rotations, but area-burned data in their Table 4 (32 years/(337,000 ha Obs./6,400,000 ha Extent)) show that the 1984–2015 high-severity fire rotation was 608 years in Group 1 forests, which include ponderosa pine, relative to a historical 226-year mean return estimate for the ten forest types in Group 1. Haugo et al. [57] (p. 9) summarized these results as follows: "We also found smaller, and in some instances no, deficits of moderate and high-severity fire in FRG1 forests . . .". This indicates that high-severity fire rates were near or low relative to rates under the HRV; (3) Mallek et al. [58] also did not calculate fire rotations, but their data show that the 1984–2009 high-severity fire rotation was 413 years in yellow (ponderosa) pine (their Tables 1 and 3: 1/(3727 ha AAHS/1,540,923 ha Extent)) and 695 years in dry mixed conifer (1/(1061 ha AAHS/737,759 ha Extent)). They also reported these rates to be "within the range of the corresponding pre-settlement estimates" [58] (p. 9) and stated that "Modern regional rates of burning at high severity exhibited comparatively little or no departure from their pre-settlement levels in lower and middle-elevation forests" [58] (p. 11). In conclusion, only 3 of 14 publications cited in H had adequate sampling area and data; all 3 showed that high-severity fires in dry forests had burned recently at long rotations (413, 608, 695, and 1693 years) directly reported by 2 of 3 authors to have burned at rates longer than or within the range of high-severity fire rates under the HRV.

Evidence in H omitted findings from the largest study (25.5 million ha) of recent vs. historical rates of high-severity fires, covering nearly all dry forests of the western USA [4]. H included this study in their "Literature Cited", but there was no citation or use of this study in H's text, just a listing in their tables. This largest study showed that high-severity fires had burned between 1984 and 2012 in western USA dry pine forests at rotations from 470 to 15,043 years (mean = 1045 years) in 23 analysis regions averaging 547,982 ha in area and totaling 12.6 million ha. Additionally, high-severity fires in this period had burned in dry mixed-conifer forests at rotations from 212 to 7909 years (mean = 875 years) in 20 analysis regions averaging 645,960 ha and totaling 12.9 million ha. These means and ranges of recent fire rotations were similar to, or longer than, historical high-severity fire rotations, which ranged from 217 to 849 years, in 42 of 43 the analysis regions, except 1 region in southern California dry mixed conifer. The 217- to 849-year historical range is based on reconstructions from paleo-charcoal, land-survey, and aerial-photo reconstructions for 18 study areas covering millions of hectares. Conclusions from all four studies with

adequate samples [4,56–58] were the same: recent high-severity fire rates are not higher than under the HRV.

It is important to mention that one measure, high-severity fire area as a percentage of total burned area, does not provide a valid estimate to judge whether “high-severity effects” . . . “exceed the pre-fire exclusion range of variation” in H’s title for their Table 2. It is widely known, and shown by data in the studies H cited, that suppression of fires, since the end of the pre-industrial period, reduced low- to moderate-severity fires in dry forests. Mallek et al. [58] (p. 16) explained:

“ . . . nearly the only fires that reach any size are those that escape control under severe climatological conditions, in heavy fuels, and/or in inaccessible topography . . . Since almost all fires occurring under moderate conditions are put out, areas burned by wildfire in the contemporary study area suffer a statistical predisposition to burn at higher severity. This is especially evident in lower and middle elevation forests like yellow pine and mixed conifer . . . ”

The four studies show that more area is not burning annually at high severity in recent periods than burned annually at high severity in historical periods in dry forests. Suppressing low- to moderate-severity fires thus did not increase high-severity fires in dry forests. Adequate evidence in H’s Table 2 and the omitted study [4] shows the opposite of the table heading, as there are no “high-severity effects” that “exceed the pre-fire exclusion range of variation”.

In conclusion, the assertion in H that “high-severity fire effects in recent fires exceed the pre-fire exclusion range of variation” (H Table 2 title) is rejected. Evidence from the three studies with adequate sample areas and data [56–58] agrees with evidence from the larger study [4] omitted in H. Recent rates of high-severity fires are too low or within HRV. Evidence from the four studies shows that what is needed to restore historical fires in dry forests and adapt forests to climate change is just to increase low- to moderate-severity fires. Expanding fuel-reduction treatments, proposed in accompanying policy proposals [22,23] that are based on H, aim to reduce high-severity fires. This would generally be ecologically deleterious, if successful. Reducing fires that are burning at or below historical rates has many effects similar to the effects of intentional fire suppression, which is still occurring but is widely accepted to be ecologically damaging to dry forests.

4. Debunking H’s “Evaluating Evidence of Lack of Change”

H presented ~37 critiques in a section on “Evaluating evidence of lack of change” divided into two parts: “Misrepresented historical forest conditions” and “Misrepresented fire regimes.” We also divided our text here into these two sections. To facilitate the comparison of evidence in H and evidence reviewed here, we replicated Tables 3–6 in H and added the evidence omitted in H into a new column in each of the four corresponding tables here (Tables 3–6).

H summarized their critique in this section as:

“ . . . publications that suggest the preponderance of evidence misrepresents or overgeneralizes departures from active fire regimes. These publications then suggest that management actions aimed at recapturing the influence of abundant low- and moderate-severity fire lacks [sic] a sound ecological foundation. Over the past two decades, independent research groups have evaluated the methods and inferences proposed by these publications and documented multiple weaknesses. Despite demonstrated methodological biases and errors, new papers employing these methods, or results and conclusions derived from them, continue to pass peer review” H. (p. 16)

H did not cite any specific source that showed “evidence of lack of change”, but it would not be our publications. We documented that significant adverse cumulative impacts are widespread in dry forests from logging, livestock grazing, fire exclusion, and fuel-reduction treatments (e.g., [7,45]). In their last section, H presented evidence

of change. There (Section 3), we evaluated H's evidence and reported that change has occurred, particularly a reduction in low- to moderate-severity fires since the pre-industrial era, but not, through 2010–2015, increased rates of high-severity fires. It is unclear which publications H is referring to: “these publications then suggest that management actions aimed at recapturing the influence of abundant low- and moderate-severity fire lacks [sic] a sound ecological foundation.” The major body of evidence in the 15 land survey reconstructions, omitted in H, in fact showed an average of 76.0% low- to moderate-severity fire across dry forests (Table 1). Furthermore, “Restoring and managing low-severity fire in dry-forest landscapes of the western USA” [25] provides updated rates and guidance on restoring missing low-severity fires.

4.1. H's “Misrepresented Historical Forest Conditions” Section Omitted Evidence

The “Misrepresented historical forest conditions” section in H is largely a critique of an established method (WB method) to reconstruct historical tree density and other forest structure measures from original land survey records [41] and an assertion that early timber inventories provide valid estimates of historical tree density. The WB reconstruction method has been used across 15 large landscapes in >2.5 million ha of dry forests (Table 1). This method provided extensive evidence that dry-forest landscapes were heterogeneous in structure (e.g., tree density and basal area) and included both sparse and dense forests, supporting the mixed-severity model.

As we show below in the next two sections, (Section 4.1.1), evidence showing the validity and accuracy of the WB method, omitted in H, does not support their claim that the WB method overestimates historical tree density, and (Section 4.1.2), rebuttals of critiques of the WB method, omitted in H, includes original and rebuttal evidence that early timber inventories underestimate, also omitted in H. To show details and implications of evidence omitted in H, we replicated Table 3 in H in our Table 3. We then added a column that shows how conclusions in H were incorrect because H omitted all evidence that validates the WB method and omitted evidence that early timber inventories are flawed, which is shown in original papers and rebuttals that H omitted.

4.1.1. Evidence Showing Validity and Accuracy of the WB Method, Omitted in H

With historical reconstruction methods (e.g., WB method), there is a need to evaluate evidence about the development of the method and validations against independent modern and historical sources. Validations are inherently multi-proxy evidence, which H cited as most valuable, yet are omitted in H. Here, we present and discuss key evidence, omitted in H, showing the high accuracy and validity of the WB method and its findings in reconstructing historical tree density: (a) evidence of development of the WB method relative to earlier methods, (b) modern and historical validations showing high accuracy of the WB method, and (c) independent multi-proxy evidence of variable density and dense forests. This entire body of multi-proxy validation evidence, including evidence in original papers, rebuttals, and other publications, was omitted in H.

- Evidence of development of the WB method relative to earlier methods, omitted in H

H suggested that the analysis presented by Cogbill et al. [59] is the correct analysis to use in evaluating the WB method, implying the WB method was not derived and tested properly: “... valid methods exist for deriving estimates from spatial point patterns, such as GLO bearing trees” (p. 15). However, Cogbill et al. only tested older existing point-pattern measures, with no test of the WB method at all, and they did no testing in western dry forests, only moister forests in the Midwest. The study by Cogbill et al. is thus not relevant to the WB method. Cogbill et al. showed that old point-pattern measures typically have low accuracy, are biased, and require large sample sizes. These are known limitations that spurred the development of improved design-based estimators, including Voronoi-based estimators, that are more robust to a wide range of spatial patterns [60]. Williams and Baker [41] explicitly improved on older methods by developing Voronoi-based estimators for use in western dry forests. For comparison, Williams and Baker also

tested older point-pattern measures in modern validations; they did generally perform poorly, were biased, and required larger sample sizes, just as Cogbill et al. [59] found. Williams and Baker [40,41] had already shown by the time of Cogbill et al. that their WB method was well derived, statistically sound [60], and overcame the limitations of methods reviewed by Cogbill et al. These motivations, advances, and published comparisons of the WB method were omitted in H.

Table 3. Hagmann et al. ([3] Table 3) about historical tree density is replicated in the left two columns about counter-evidence, except that citations are changed to match our reference list. Omitted rebuttals and other omitted published evidence omitted by H are added in the right column to show Hagmann et al. omitted published evidence and made incorrect conclusions as a result.

Counter-Evidence		Evaluation of Counter-Evidence		H Omitted Rebuttals and Other Published Evidence also Essential to Evaluation of Counter-Evidence	
Citations	Counter-premise	Citations	Implications of evaluation	Citations	Implication of omitted evidence
Williams and Baker [41] Baker and Williams [34]	Novel methods provide estimates of tree density from point data, i.e., General Land Office (GLO) records of bearing trees	Levine et al. [61,62]	Multiple existing plotless density estimators (PDEs) provided less biased estimates than the PDE developed by Williams and Baker [41], which overestimated known tree densities by 24–667% in contemporary stands	Omitted Rebuttal by Baker and Williams [63]	Levine et al. [61] incorrectly coded and applied the WB method, producing spurious results that had no bearing on the WB method.
				Omitted Rebuttal by Baker and Williams [63], Omitted evidence in Williams and Baker [41]	Levine et al. [62] corrected their flawed 2017 code in [61], but then in [62] used incorrect equations. Baker and Williams [63] used corrected equations with their code at their sites and showed the WB method worked well. Williams and Baker [41] had shown that Voronoi-based estimators work better than existing PDEs and do not overestimate in western dry forests.
		Knight et al. [64]	Methods supported by PDE sampling theory and multiple accuracy assessments further demonstrate the potential for misrepresentation of historical tree density by biased estimators used at resolutions substantially smaller than the minimum recommended for –50% accuracy	Omitted evidence in Williams and Baker [41]	Knight et al. [64] did not use or test the WB method at all. They used old point-pattern measures that Cogbill et al. [59] had already shown were inaccurate, require large samples, and underestimate. The WB method was designed to overcome these known limitations and had already been validated [41] to be able to accurately estimate tree density at the –518 ha scale in western dry forests.
Williams and Baker [24]	Historical forests were denser than previously documented	Johnston et al. [65]	Existing method for estimating tree density from point data [66,67] yielded densities less than half as large as estimates using Williams and Baker [41]	Omitted Rebuttal by Baker and Williams ([63] Appendix S1)	This study roughly estimated Voronoi-based tree density of 89.6 trees/ha for Johnston et al.'s sites, a modest error of 20% if a Johnston et al. estimate of 112 trees/ha is considered truth. This is within expected accuracy for the WB method [41]. Their estimate is not from a random sample and is too small to compare, as they did, with the mean for the whole WB study area, but is within one s.d. of the reconstructed historical mean [24] and so is congruent with historical variability, as found in the reconstruction.

Table 3. Cont.

Counter-Evidence		Evaluation of Counter-Evidence		H Omitted Rebuttals and Other Published Evidence also Essential to Evaluation of Counter-Evidence	
Citations	Counter-premise	Citations	Implications of evaluation	Citations	Implication of omitted evidence
Williams and Baker [24] Baker [25,28,29,68]	Historical forests were denser than previously documented	Hagmann et al. [54,69–71], Collins et al. [72], Stephens et al. [73,74], Battaglia et al. [75], Johnston et al. [65]	Consistent with the finding that Williams and Baker [41] methods overestimate tree density (Levine et al. [61,62], Johnston et al. [65], and Knight et al. [64]), early timber inventory records and tree-ring reconstructions for the same study areas documented substantially lower tree densities than those estimated using Williams and Baker [41] methods	New evidence here	See above for why Levine et al. [61,62], Johnston et al. [65], and Knight et al. [64] do not show that the WB method overestimates tree density. Regarding early timber inventories, see the last line below. Battaglia et al.'s [75] study area was ~30 times WB's, a major scale-mismatch, and was based on sampling 97% in logged forests. There has been no validation that the method they used can accurately reconstruct historical tree density in their region, and there especially has been no validation of their method in heavily logged forests where evidence likely has been destroyed by harvesting and associated activities.
Hanson and Odion [76]	Managing for dense, old forest and high-severity fire is consistent with historical conditions	Collins et al. [77]	Fundamental errors compromise assertions about historical conditions including (1) inappropriate use of coarse-scale habitat maps and (2) inaccurate assumption that areas lacking timber volume in early inventories indicate past high-severity fire	Omitted rebuttal in Hanson and Odion [78]	Collins et al. [77] is not about tree density or forest density and did not belong in this table. However, Hanson and Odion [78] showed that (1) Collins et al. thought maps were wrong but missed that areas that were forested by 1992, having recovered from early high-severity fires, had burned again, and (2) Collins et al. had omitted including essential 1911 field survey notes that directly described high-severity fires.
Odion et al. [12], Baker [4,68], Baker and Hanson [17]	Spatially extensive early timber inventories and bias in their use and interpretation misrepresent historical conditions	Stephens et al. [73], Collins et al. [77], Hagmann et al. [18,54,71]	Fundamental errors compromise conclusions, including (1) the use of previously discredited methods (Williams and Baker [24]) to estimate tree density from GLO data as a baseline comparison; (2) incorrect assumptions about the methodological accuracy of early timber inventories; (3) inappropriate comparisons of studies of vastly different spatial scales, forest types, and diameter limits; (4) unsubstantiated assessment of bias in the locations of early timber inventories; and (5) unwarranted assumptions about vegetation patterns as indicators of fire severity	Omitted Rebuttal in Baker et al. [19]	Evidence in H omitted our rebuttal [19], where we showed that Hagmann et al. [18] did not contest Baker and Hanson's [17] key findings: (1) early two-chain-wide timber inventories, documented to underestimate, are unreliable and were abandoned by the 1930s, (2) comparisons between timber-inventory estimates and other sources showed it is timber inventory estimates that underestimate and need correction, (3) one-chain-wide inventories, if available data are used, could be fairly accurate, and (4) omission of immature conifers and non-conifers may lead to additional underestimation. In response, we revised our estimates of needed correction multipliers to 1.6–2.3. Hagmann et al. [18] still contended that inventories do not have biased placement, but we presented more evidence.

H also incorrectly implied that the WB method can only provide an accurate estimate over a very large land area, but this is just a known limitation of earlier methods, not a limitation of the WB method or an inherent property of land survey data. H incorrectly said “... the extremely low sampling density of this national land survey limits reliable estimates to the average forest density for a large area” [3] (p. 16). H listed some accuracies for large land areas (3000+ ha), but these are only from using the old, inaccurate, biased, point-pattern methods that require pooling data across large land areas [59]. Using the WB method, modern and historical validations (detailed below) showed that sample areas of

~518 ha in dry forests provide tree-density estimates with weighted mean errors of 19.3%. The WB method had already been well validated [40,41] as an advance over older methods that previously provided just one estimate for very large land areas [59]. Our conclusion from this updated evidence is that Cogbill et al. did not test the WB method, their study was about midwestern forests, and their study had no relevance to the WB method or its findings. All published evidence of higher validity and accuracy, from the development and testing of the WB method [40,41], was omitted in H.

- Modern and historical validations showing high accuracy of WB method, omitted in H

H did not mention or review substantial published evidence on the accuracy and lack of bias of the WB method from both modern and historical validations [24,28,29,31,34,40,41,63,68,79]. These validations documented considerable agreement with independent multi-proxy sources, something H had highlighted as strong evidence, but evidence in H did not include any of these highly relevant published validations.

In modern forests, the WB method's Voronoi estimators and 9 other existing estimators of tree density from land survey data were tested and compared in field validations at 499 section corners in dry forests in 3 states ([34] Appendix Table S1, [41]). The latest summary showed a weighted mean error of 19.3%, relative to plot estimates, using the WB method ([34] Appendix Table S1). As mentioned in the last section, nearly all other estimators, except the two new Voronoi estimators, including some tested by Cogbill et al. [59], were found to be significantly biased and underestimated modern tree density [41]. The WB method's Voronoi estimators are still validated as the most accurate, unbiased estimators of tree density for use with land surveys in modern dry forests in the western USA, but all this evidence was omitted in H.

In historical forests, evidence in H also did not cite or review published evidence ([34] Appendix Table S4) that the WB method is quite accurate in reconstructing historical tree density, based on specific and general cross-validations with multiple sources that also show high independent multi-proxy agreement. Specific cross-validations compare tree density from the six-corner reconstruction polygon, which intersects an alternative source location, with tree density at this source. Specific cross-validations at 18 source locations in Arizona, California, and Oregon had relative mean errors of 10.4–11.2% ([34] Appendix Table S4), much better than the 19.3% from modern validations. Relative mean errors were 9.6–10.7% in comparison with 12 tree-ring reconstructions, 10.0% in comparison with 2 early 1-chain-wide timber inventories, and 13.1% in comparison with 4 early permanent plots or other non-timber inventories. The WB method cross-validated well against multi-proxy historical sources, evidence that H said they especially valued, but all this evidence was omitted in H.

Published general cross-validations compared sets of mean tree densities from independent historical studies (imprecisely located so they cannot be overlaid) in or near reconstruction areas with tree-density reconstructions using the WB method for that area. For example, 19 tree-ring reconstructions across Arizona's Mogollon Plateau had a mean of 122 trees/ha, whereas the land survey reconstruction using the WB method had a mean of 141.5 trees/ha and a relative error of 16.0% ([34] Appendix Table S9). A recent compilation of 15 tree-ring reconstructions, early inventories, and land survey reconstructions for dry mixed conifer in the Southwest found a mean of 144.5 trees/ha [26], nearly identical to the WB method estimate for mixed conifer on the Mogollon Plateau of 144.3 trees/ha [24]. An author of H co-authored this publication showing the WB method works well [26], but this evidence was still omitted in H. Other general cross-validations [34] included Oregon's Blue Mountains (4 early inventories) with a relative error of 27.8%, Oregon's Eastern Cascades (2 early inventories and 2 tree-ring reconstructions) with a relative error of 14.2%, and California's western Sierra (18 early inventories and 1 tree-ring reconstruction) with a relative error of 6.0%. All this evidence was omitted in H.

These comparisons show that the implication in H, that the WB method overestimates historical tree density, is refuted. The WB method showed relative errors of only 6–28%

in validations across large land areas. The WB method's accuracy is also validated by cross-validations with multi-proxy evidence and independent compilations that H said they value. Yet, all this evidence of the accuracy of the WB method was omitted in H.

- Independent multi-proxy evidence of variable density and dense forests, omitted in H

H said they especially valued independent multi-proxy evidence, but again, all of it, supporting the mixed-severity model, was omitted in H. Baker et al. [35] reviewed evidence from 20 tree-ring reconstructions, forest reserve reports, and early scientific reports that dry forests in Rocky Mountain states had highly variable tree densities from 17 to 19,760 trees/ha. Baker ([29] Appendix Table A1) published nine quotes from early forest reserve reports and early scientific reports that historical dry forests in the eastern Cascades of Oregon varied in historical tree density, including some dense forests. Similarly, Baker [28] (Appendix A) published 47 quotes from early forest reserve reports and other scientific reports documenting that Sierran mixed-conifer forests in California were highly variable in density but typically dense. Additionally, Baker and Williams [63] published evidence from 30 independent early estimates of historical tree density in Sierran mixed-conifer forests in California that had a mean of 257 trees/ha and a standard deviation of 100 trees/ha, showing that these historical forests were highly variable in tree density and generally dense. H reported [3] (p. 16) "... as Fulé et al. (2014) observed, existing research documented even greater heterogeneity in historical forest conditions, including higher densities, than were reconstructed from GLO data." A total of 8 authors of Fulé et al. [80] are also authors of evidence in H, which omitted this highly relevant independent evidence, including their own, from more than half of the 11 western states. Omitted evidence shows that historical dry forests varied in density and included substantial areas that were dense. This omitted independent multi-proxy evidence, alone, without using evidence from land surveys and the WB method, rejects the low-severity model in favor of the mixed-severity model.

4.1.2. Rebuttals of Critiques of the WB-Method, Omitted in H

H said that papers that used the WB method "have suggested that densities and fire severities of dry forests were higher and more variable than previously thought (Table 3) ... " [3] (p. 16), implying that WB-method estimates of tree density are erroneous and too high. H summarized evidence the WB method overestimates tree density, stating that "Density estimates based on Williams and Baker (2011) methods are also inconsistent with tree-ring reconstructions and early 20th-century timber inventory records for areas where the data overlap ... " [3] (p. 16) and also that "Dendrochronological reconstructions and early timber inventories demonstrate consistency with each other and with other independent sources" [3] (p. 16). We reviewed in the last section that published validations show that historical tree density reconstructions from the WB method are very consistent with tree-ring reconstructions where they overlap, also shown in a publication [26] that included an author of H, but all this evidence was omitted in H.

H's evidence (their Table 3) that WB method estimates of historical tree density are too high rests largely on their comments, on publications using the WB method, which were rebutted in publications omitted in H. Here, we review that this omitted evidence shows that (a) Levine et al. modeling and plot tests are incorrectly coded, (b) older point-pattern methods have lower accuracy than the WB method, (c) the WB method has evidence of scale-matched high accuracy, (d) original and rebuttal evidence shows that early timber inventories underestimate, and (e) the fourth entry in H Table 3 is misplaced and belongs in H Table 5.

- Levine et al. modeling and plot tests shown to be incorrectly coded, omitted in H

Evidence from Levine et al. [61,62] simulation modeling that the WB method overestimated tree density actually showed the WB method works well, but the modeling was incorrect. First, Levine et al. [61] incorrectly coded the WB method, an error that invalidated their study, as shown in detail in a section of Baker and Williams [34], omitted

in H. Levine et al. [62] next used revised code in permanent plots and again reported over-estimation by the WB method. However, our rebuttal, Baker and Williams [63], omitted in H, showed that Levine et al. [62] this time used incorrect equations. For their three sample sites, using their coding of the WB method, Baker and Williams [63] showed that if correct equations were used, relative mean errors were only 6.2%, 7.0%, and 25.9%, well within expected accuracy for the WB method [41]. Levine et al. [61,62] are listed incorrectly in H Table 3 and their text as evidence the WB method is wrong, but both Levine et al. studies are fatally flawed because of the use of incorrect code and equations. Baker and Williams [63] showed that, after correcting Levine et al. errors, the WB method worked correctly and accurately even in highly altered modern forests in tiny plots, well outside their geographical setting and historical landscape-scale design, strong evidence of robust validity. All this published evidence that the WB method works well was omitted in H.

- Older point-pattern methods lower accuracy than WB method, rebuttal omitted in H

H said two studies [64,65] that used land survey data showed that tree densities from the WB method are too low. However, the findings of low tree density in these two studies were from the application of older point-pattern methods [64,65], not the WB method. As noted in Appendix S1 in [63], if Johnston et al. [65] had used the WB method at their sites, we estimated a relative error of ~20%, within the expected error for the WB method [41]. The methods they instead used have known lower accuracy than the WB method [41]. Neither Johnston et al. nor Knight et al. expressed any awareness of the significant limitation of the methods they used. However, these two studies also have no basis for claiming anything about the WB method, since they did not use or test the WB method at their sites. Evidence of the accuracy of the WB method, reviewed in Section 4.1.1, and the rebuttal [63] explaining this matter for the Johnston et al. study area, were both omitted in H. We explain, next, more about misinterpretation by Johnston et al.

- WB method has published evidence of scale-matched high accuracy, omitted in H

Critiques in the past, including several by these same authors, used a double standard on scale mismatches, as they do here again. H was concerned about mismatches in spatial scale in comparisons of a ~518 ha reconstruction polygon with a tree-ring reconstruction. This is just an inherent limitation of tree-ring reconstructions that their typically small plots produce scale mismatches with most other historical sources.

H was concerned about scale mismatches but did not cite, mention, or review evidence from the most closely scale-matched validations of the WB method, which are the modern validations performed in three states [41] that we reviewed in Section 4.1.1. These validations compared tree-density estimates from land-survey section-corner data with tree tallies in small plots placed over these same section corners [41]. These closely scale-matched comparisons showed the WB method has high accuracy. These closely scale-matched validations were omitted in H.

Although H critiqued smaller scale mismatches, they employed much larger scale mismatches as evidence against the WB method. Battaglia et al. [75] and Johnston et al. [65] were presented in H's Table 3 as showing the WB method overestimates tree density. Battaglia et al.'s study area is ~30 times that of the WB study area in the Front Range [24], and Battaglia et al. did not report estimates for just the WB study area portion, so this is a very large scale-mismatch. At most, only 6 of their 28 sampling points (21%) might occur in the WB study area. Johnston et al. [65] compared their tree-ring reconstructions in five small plots with the Williams and Baker [24] overall estimates for their 305,000 ha Blue Mountains study area. As explained in the Baker and Williams ([63] Appendix S1) rebuttal, "... Johnston et al. sampled and summarized Blue Mountains forests from only five clustered points covering the equivalent of perhaps 4 six-corner GLO pools, while our study sampled and summarized over a much larger area including over 500 six-corner GLO pools. Johnston et al. cannot validly infer from a small, nonrandom sample to the entire Blue Mountains landscape ... " These are examples of a double standard in H; neither comparison H used is valid because of very large scale-mismatches.

One small source, such as Johnston et al. [65], does not provide a valid comparison, particularly since its estimate is within reconstructed HRV. Land-survey reconstructions using the WB method show variability was large, based on the coefficient of variation (CV), across historical dry-forest landscapes (Table 1). H cited Johnston et al. as evidence the WB method overestimates tree density. However, the Blue Mountains reconstruction [24] had a mean of 167.3 trees/ha (median 146 trees/ha) and an s.d. of 89.8 trees/ha, so Johnston et al.'s weighted mean of 112 trees/ha [63] is well within the HRV for Blue Mountain forests. Additionally, Johnston et al. used a method biased toward underestimation [41,59] and is not a statistically valid sample of Blue Mountain dry forests.

When comparing other sources, at much finer spatial scales, to the overall study-area estimates from land-survey reconstructions, it is only valid to do a “general cross-validation” [34] with findings from multiple sites in the reconstruction area. The first largest general cross-validation is (1) in California's western Sierra, where Baker and Williams [63] compared means, quartiles, and confidence intervals from 30 independent historical estimates of tree density with similar data from the Baker [28] land survey reconstruction. They found overlapping 95% confidence intervals for historical mean tree density (independent = 257 trees/ha and land surveys = 293 trees/ha), similarity in distributions, and 14% relative error if independent estimates are considered the truth. The second largest general cross-validation is (2) on Arizona's Mogollon Plateau, where the mean study area estimate was 141.5 trees/ha versus the mean from 8 tree-ring reconstructions of 122.0 trees/ha, with a relative error of 16.0%, assuming tree-ring reconstructions represent truth ([34] Appendix Table S9). This is multi-proxy evidence that shows the WB method accurately reconstructs historical tree density across large landscapes, omitted in H.

In summary, the fuller set of evidence reviewed here shows scale mismatches to be inherent limitations of comparisons with some methods of reconstruction (e.g., tree-ring reconstructions). H criticized some WB method validations for scale mismatches, but evidence in H used much larger scale mismatches, which shows that a double standard was used in H. Published evidence of high accuracy from our closely scale-matched modern cross-validations was omitted in H. All our published general cross-validations with multiple independent reconstructions in land-survey study areas, which show compelling independent, multi-proxy evidence that the WB method accurately reconstructs historical tree density across large landscapes, were omitted in H.

- Original and rebuttal evidence early timber inventories underestimate, omitted in H

H implied that tree-density estimates from the WB method are too high. H in their Table 3 said “... early timber inventory records and tree-ring reconstructions for the same study areas documented substantially lower tree densities than those estimated using Williams and Baker (2011) methods”, implying that estimates from the WB method are in error. This conclusion is incorrect based on evidence in the original paper [17], evidence omitted in H, and evidence in the rebuttal [19], also omitted in H.

Early timber inventories using two-chain-wide strips failed early in modern evaluations and tests and later also in historical validations that found similar errors [17]. These inventories required visual estimation over too large a distance (2 chains or 40 m) to be accurate. Baker et al.'s [19] rebuttal of the Hagmann et al. [18] comment, which was omitted in H, confirmed that Hagmann et al. actually did not contest Baker and Hanson's [17] central findings about these early timber inventories, which we quote again here: (1) “early timber inventory data, particularly from two-chain-wide transects, were documented between 1911 and 1916 to underestimate and be unreliable and were abandoned and replaced by more accurate methods by the 1930s ...” [17] (p. 2), (2) “... comparisons between timber inventory estimates and other sources ... showed that it is timber inventory estimates, not other sources, that underestimate and need correction” [17] (p. 3), (3) “... one-chain-wide inventories, if all available data are used, could be fairly accurate, but further validation is needed ...” [17] (p. 3), (4) quantitative estimates of immature conifer density and non-conifer trees “were not included in Stephens et al. (2015)” [17] (p. 3), and, if included, historical tree density “... was ~17 times higher than the 25 trees/ha reported in ponderosa

pine, and ~7 times higher than the 75 trees/ha reported in mixed-conifer forests . . . by Stephens et al. (2015)” [17] (p. 3). This evidence, which Hagmann et al. [18] did not contest, was omitted in H.

Although Hagmann et al. [18] did not dispute the central findings of Baker and Hanson [17] that early timber inventory data substantially underestimate tree density and that 1.6–2.3 correction multipliers need to be used before reporting tree-density estimates, evidence in H omitted any mention of the rebuttal [19] and did not perform the necessary corrections. H still claimed that the study by Baker and Hanson [17] is among several papers where “Fundamental errors compromise conclusions, including . . . (2) incorrect assumptions about the methodological accuracy of early timber inventories” ([3] Table 3).

Regarding other points made by Hagmann et al. [18], (1) the rebuttal [19] showed that early inventory quality control records were not accuracy tests and did not correct erroneous estimates, (2) the rebuttal agreed that [17] had overestimated time available (more likely 15–30 min) for tallying trees in a transect, (3) the rebuttal updated ([17] Table 1) to address concern about matching tree species, sizes, and time periods, and (4) the rebuttal found that the needed correction multipliers for early timber inventory estimates were then 1.6–2.3, not 1.6–3.2, still large errors showing the need for correction multiplication of early timber inventory tree-density estimates, shown here in Table 7. Moreover, Baker and Hanson [17] and Baker et al. [19] did not discuss “assumptions” about the accuracy of early timber inventories as H put it in the above quote; they instead presented evidence, including documents, agency reports, and field tests, that showed that early timber inventories have low accuracy and need correction multipliers of 1.6–2.3 to estimate tree density. These are failures, not “assumptions” as H falsely characterized them, that led to government abandonment and replacement of early timber inventories by the 1930s [17,19].

Large underestimation bias by early two-chain-wide timber inventory estimates can be seen in other validations: (1) in comparing mean tree density from 3 early timber-inventory estimates (48 trees/ha) versus 19 estimates from independent sources (254 trees/ha) in the California western Sierra and (2) in comparing 2 early timber inventory estimates (67 trees/ha) versus estimates from 4 other independent sources (218 trees/ha) in the Oregon E. Cascades ([34] Appendix Table S9). These comparisons also document that it is only early timber inventories that substantially underestimate tree density, and it is only timber inventory estimates that need correction multipliers.

The papers that used two-chain-wide early timber inventories to estimate tree density and did not use correction multipliers, making their conclusions invalid, are in Table 7. Shown are the missing corrected estimates using 1.6–2.3 correction multipliers and corrections for missing non-coniferous trees and small trees in one case. What emerges from this evidence, after these corrections, is that the forests that received timber inventories often had historical tree-density estimates that were near the first quartile to the median tree density reconstructed from land survey data for these areas and thus are within the estimated HRV for tree density but have lower density (Table 1). We made the case [17,19] that areas that received timber inventories likely had concentrations of large trees that typically are less dense than in younger forests with smaller trees. Thus, the full set of available evidence, reviewed again here, shows that early records and reports had documented that timber inventories underestimate tree density, that correction multipliers of 1.6–2.3 must be applied, and that, if applied, these estimates may be congruent with independent estimates from other historical older forests with large trees, although these corrected estimates will be quite imprecise (Table 7). Evidence in H omitted all evidence, which Hagmann et al. [18] did not dispute, in both the original paper [17] and all evidence in the published rebuttal [19].

- The fourth entry in H Table 3 is misplaced and belongs in H Table 5

Collins et al. [77] is not about tree density or forest density and did not belong in this Table but instead in H Table 5. However, this is another case where H cited their own comment [77] on Hanson and Odion [76] but omitted the rebuttal of this comment by Hanson and Odion [78]. Hanson and Odion [78] showed that (1) Collins et al. [77] said maps

were wrong, and therefore, the interpretation that forests had burned at high severity was wrong, but Collins et al. [77] just missed areas that were forested by 1992, having recovered from early high-severity fires and burned again after the early high-severity fires, and (2) Collins et al. [77] had omitted including essential 1911 field survey notes that directly described these high-severity fires. Both errors show that Collins et al.'s [77] critiques were incorrect, and Hanson and Odion's [76] remains valid, which is more evidence that does not support the low-severity model.

4.1.3. Conclusion—Historically Heterogeneous Tree Density and Dense Forests

We showed here that what H [3] (p. 16) called “multiple weaknesses” and “... demonstrated methodological biases and errors” regarding land survey reconstructions of historical tree density using the WB method had already been shown, in original papers and in rebuttals omitted in H, to be false. H could have presented the evidence in these omitted original papers and rebuttals and then offered new counter-evidence, but they did not. H simply summarized their previous comments critical of the WB method and then omitted all evidence in published rebuttals of these comments and nearly all evidence in the original papers.

The WB method of reconstructing historical tree density from land survey data had been well validated, before the H review, to accurately estimate historical tree density, based on many closely scale-matched modern validations at section corners and through many specific and general historical cross-validations with independent multi-proxy evidence (Section 4.1.1). Abundant independent sources (not land-survey reconstructions) in more than half of the 11 western states agreed that historical dry forests were highly variable in tree density and included a substantial area of dense forests (Section 4.1.1). In contrast, early timber inventories, which H said closely match tree-ring reconstructions, were documented very early to produce estimates that were unreliable and far too low (Section 4.1.2), so these inventories were abandoned and replaced by the 1930s. H omitted this evidence that rejects the low-severity model, but supports the mixed-severity model.

4.2. “Misrepresented Fire Regimes” Section in H Omitted More Evidence

H began this section with a somewhat incorrect summary of publications cited in their Tables 4–6: “Counter-evidence publications have also posited that the high-severity component of contemporary wildfires is consistent with historical fire regimes.” What was found and reported in publications using the WB method was that the proportion of high-severity effects on historical landscapes was higher than previously thought. Thus, some modern wildfires considered uncharacteristic are more likely within HRV. Subsequent research that used land survey data, paleo-charcoal data, and aerial photo evidence together did find [4], as reviewed earlier, that recent high-severity fire rates were within or below rates under the HRV.

4.2.1. Evidence about Low-Severity Fires Omitted/Misinterpreted in H Table 4

Evidence in the H Table 4 “Counter-premise” list (Table 4) mentions concern about past methods of estimating rates of historical low-severity fires. H said, “Counter-evidence” publications showed that historical rates of low-severity fires were not as frequent (short) as reported using “composite fire interval” (CFI) methods. Yes, as reviewed in the Introduction, this began with Baker and Ehle [47,48], who critiqued the theoretical basis of CFIs and ITFIs (individual tree fire intervals) for estimating the essential fire rate parameters of fire rotation (FR) and population mean fire interval (PMFI), which they showed to be equivalent estimators of historical fire rates across landscapes. They hypothesized that the true fire rate, PMFI/FR, may lie between a CFI estimate, which is too short, and an ITFI estimate, which is too long. Baker and Ehle hypothesized and presented evidence that the omission of origin-to-scar intervals, the inclusion of small fires, targeted sampling, and known decline in mean CFI as samples increase could together explain CFI estimates that are too short. H Table 4 cited studies that presented evidence defending against these concerns with CFI

estimates (e.g., Van Horne and Fulé [81], Collins and Stephens [82], Brown et al. [83], and Stephens et al. [84]), but these studies did not analyze why CFI estimates are too short relative to PMFI/FR, as shown in Baker [25].

H Table 4 omitted citing and reviewing the much larger body of evidence in Baker ([25] S1 Text), where there is a detailed analysis, using 342 fire-history sampling sites, of all known hypotheses that could explain why CFI and ITFI estimates of PMFI/FR are inaccurate and biased toward intervals that, this study discovered, are both too short. These explanations included (1) overcompensation from the compositing process, (2) destruction of long fire intervals by compositing, (3) insufficient CFI restriction rules, (4) censoring causing loss of long fire intervals, (5) targeted sampling also causing loss of long fire intervals, and (6) unstudied fire severity inflating low-severity fire rates because some of the fires likely were not low severity.

As mentioned in the Introduction, even more important is that the H Table 4 column entitled “Implications of evaluation” omitted extensive new evidence about how much CFI and ITFI both underestimate PMFI/FR and how both now can be corrected to accurately estimate PMFI/FR [25]. Baker [25] used a 96-case calibration and analysis dataset from 44 fire history studies where both CFI and/or ITFI were calculated, or could be calculated, and could be compared with estimated PMFI/FR. CFI measures all produced estimates that were too short (biases of 38–72%) and were quite inaccurate (errors of 43–70%) in estimating PMFI/FR. ITFI measures also produced estimates that were too short but less so (biases of 3–28%) and also less inaccurate (errors of 16–33%). Most important, linear regression showed that historical PMFI/FR could be very accurately estimated from Weibull mean ITFI ($RMSE = 7.52$, $R^2_{adj} = 0.972$) and quite accurately ($R^2_{adj} > 0.900$) from eight other CFI/ITFI measures. These linear regressions (1) showed that all the CFI and ITFI measures and methods produced historical estimates that were too short and (2) enabled correction of all CFI/ITFI estimates to provide valid estimates of historical PMFI/FR at 342 sites across the western USA (Figure 8).

Fortunately, a new landscape-scale method has been developed and validated for directly estimating PMFI/FR using random or systematic plots in which all scarred trees are sampled, fire years are cross-dated, and individual fire years are reconstructed spatially from these plot data and then used to directly estimate PMFI/FR [15,51]. Baker [25] was able to find and use many of these spatial reconstructions, showing that the fire-year reconstruction method (e.g., Figure 7) is being used. This new landscape-scale method does not require further use of inaccurate CFI or ITFI estimates from small plots; thus, earlier debates over compositing, targeted sampling, etc., the focus of H’s comments, are now of little to no interest, as science has moved on.

Improved plot methods could still be used but have lower accuracy than these newer landscape methods, and, at a minimum, require pooling over several plots, limiting their value. CFI and a new all-tree-fire-interval (ATFI) plot method [49,50] were tested in a modern and historical validation at the Grand Canyon [51], omitted in H. In these tests, ATFI outperformed all CFI measures. ATFI was always correct in modern tests at the plot scale and CFI mostly failed. In historical tests, ATFI had a mean relative error of 14.3%, and the best traditional CFI measure, scar-to-scar 25% filtered CFI, had a mean relative error of 35.3%. ATFI was thus superior to all other plot-scale methods. ATFI at the plot scale can possibly achieve errors <26.6%, but errors <20% require at least 4 plots over 600–1000 ha [51]. H’s discussion of their Table 4 claimed that, “Additionally, as acknowledged by Kou and Baker (2006: Accessory Publication), ATFI will always be much longer than any MFI ... ” [3] (p. 20). H thought this was a failing of the ATFI method, but this is because CFI estimates are always erroneously too short [25], and ATFI is longer and more correct [49–51].

Estimates from the new landscape-scale methods (e.g., Figure 8) and corrected estimates from regression [25] show (Figure 8) that frequent low-severity fire was less common historically than was thought in the 1990s when the early, inaccurate CFI estimates were all that was available. For 342 sites, the mean low-severity fire rotation was 39 years and the

median was 30 years. Only ~14% of the dry-forest area, mostly at lower elevations in drier areas, had frequent fires (PMFI/FR < 25 years), and ~86% of the dry-forest area instead had multidecadal mean PMFI/FRs. In mountainous areas (Figure 8), landscapes often contained a wide range of low-severity fire rates. This is significant because PMFI/FR's ≥ 25 years will not generally keep fuels at low levels, as fuels can fully recover. This means that the idea that frequent fires kept fuel loads at low levels, and fires low in severity, was actually a condition found primarily only in limited areas, such as the driest areas in the Southwest and dry areas elsewhere at low elevations (Figures 8 and 10a).

Table 4. Hagmann et al.'s table ([3] Table 4) about rates of fire is replicated in the left two columns, with omitted published evidence added in the right column, to show Hagmann et al. omitted published evidence and made incorrect conclusions as a result.

Counter-Evidence		Evaluation of Counter-Evidence		Published Evidence, Essential to Evaluation of Counter-Evidence, That Was Omitted	
Citations	Counter-premise	Citations	Implications of evaluation	Citations	Implication of omitted evidence
Baker and Ehle [47,48], Ehle and Baker [14], Kou and Baker [49,50], Baker [25,85], Dugan and Baker [51]	Tree-ring reconstructions misrepresent historical fire regimes by overestimating fire frequency and extent because (1) unrecorded fires (e.g., fires that did not scar trees) increase uncertainty of mean fire interval (MFI); (2) interval between pith (origin) and first fire scar should be considered a fire-free interval and included in calculations of MFI; (3) targeted sampling of high scar densities biases MFI; and (4) mean point fire interval (mean of intervals between fire scars weighted by the number of fire scars) may more accurately represent historical fire rotation than MFI (mean between all fire scars)	Collins and Stephens [82]	Unrecorded fires (fire did not scar the tree) may contribute to underestimation, not overestimation, of fire frequency and extent in frequent fire systems. Probability of scarring decreased when intervals between successive fires were short in areas burned by up to four late-20th-century fires. Absence of scar does not indicate absence of fire.	Omitted evidence in Baker ([25] S1 Text)	It is generally agreed that each fire only scars some of the trees. However, with typical scarring fractions, only ~50 trees or ~1 ha need sampling to detect all the fires in a plot. In a sample of 262 reconstruction sites in dry forests of the western USA, 88% sampled ≥ 1.0 ha. Thus, underestimation from unrecorded fires is likely rare, and absence of a scar in a particular year likely does show lack of fire. A key problem with “composite fire intervals”, the primary source of evidence about historical low-severity fire rates, is the compositing process itself. Compositing makes a single list of all fire years in the plot. This assumes all fires burned the whole plot, which is not true, based on 11 studies [47]. Putting small fires in a single list with large fires reduces the “mean composite fire interval” to a small value, leading to large overestimation of rates of fire. Small fires can be filtered, but filtering is arbitrary, and compositing still destroys the long intervals that were found.
		Brown and Wu [86], Van Horne and Fulé [81], Brown et al. [83], Stephens et al. [84], Yocum Kent and Fulé [87], Meunier et al. [88]	Including origin-to-first-scar interval erroneously inflates MFI. Not all trees that survive fire are scarred. As an ambiguous indicator of fire-free interval, it should not be included in calculations of MFI. Additionally, tree establishment may not indicate a stand-replacing disturbance in dry forests where regeneration is strongly associated with climate	Omitted evidence in Kou and Baker [49], Polakow and Dunne [89], Moritz et al. [90] Omitted evidence in Dugan and Baker [52]	Fire history data typically have incomplete intervals at the start and end of a period of record. Real but long fire intervals have more chance of appearing at the beginning or end, and getting left out, than do real but short intervals. Thus, censoring the starting or ending incomplete intervals biases the record toward estimates that are too short and have reduced variability [49], as found in two other independent studies [89,90]. There is no citation in the counter-evidence list that assumed tree establishment indicates stand-replacing disturbance in dry forests. Brown and Wu [86] incorrectly assumed that a fire scar before a pulse of tree establishment <u>does not</u> indicate moderate- to high-severity fire [52].

Table 4. Cont.

Counter-Evidence		Evaluation of Counter-Evidence		Published Evidence, Essential to Evaluation of Counter-Evidence, That Was Omitted	
Citations	Counter-premise	Citations	Implications of evaluation	Citations	Implication of omitted evidence
		Fulé et al. [91] Van Horne and Fulé [81], Farris et al. [15,92], O'Connor et al. [93]	Complete, systematic (gridded), and random sampling at stand, watershed, and mountain range scale have repeatedly demonstrated fire frequencies similar to those derived from targeted sampling within forest types and scales. In direct comparison studies, no evidence was found that targeted sampling of fire-scarred trees biased MFI estimates. Targeted sampling reconstructed fire parameters comparable to those derived from systematic sampling of both a subset of the trees and all trees in a study area and from independent 20th-century fire atlases	Omitted evidence in Baker ([25] S1 Text)	Evidence cited in H in Farris et al. [92] and in Van Horne and Fulé [81] is not correct. Farris et al. instead found that using a targeted sample led to CFI estimates that were shorter (80–96%, comparing targeted and probabilistic sample size corrected in their Table 3) than that from a statistical sample. Van Horne and Fulé found that a targeted ITFI estimate was only 83% (inverse of 1.2 from p. 865) of ITFI from a random sample. These studies thus show that targeted samples produce CFI/ITFI estimates that are shorter than estimates from random samples.
		Farris et al. [15] Huffman et al. [94]	Rather than overestimating fire frequency as suggested in counter-premise papers, MFI may underestimate fire frequency, especially where small fires were abundant	Omitted evidence in Baker [25]	MFI as used in H is just composite fire interval (CFI), which has the well-established property of producing estimates that are too short relative to fire rotation, the gold standard, as shown in this monograph on this topic, which was omitted in H
		Van Horne and Fulé [81] Farris et al. [92]	Composite mean fire intervals (CMFI, e.g., fires recorded on 25% of samples) are relatively stable across changes in sample area or size. See the section on “Underestimated historical fire frequency” for a more detailed summary of CMFI and the highly problematic and inherently biased alternatives proposed in counter-evidence publications	Omitted evidence in Baker [25]	CFI estimates do vary with sample size, but they also definitely produce estimates that are too short relative to fire rotation, the gold standard, as shown in this monograph on this topic, which was omitted in H

In summary, H used out-of-date, inaccurate, biased methods of reconstructing historical rates of fire, and mentioned old arguments about these CFI methods that are no longer of interest, but omitted that these dated CFI measures and small-plot methods have been replaced by newer, more accurate PMFI/FR measures and spatial reconstruction methods. Also omitted in H is that 342 dated, inaccurate estimates have been corrected to new PMFI/FR estimates using regression, and corrected estimates show that historically frequent low-severity fires only dominated in ~14% of the dry-forest area [25]. This large, significant body of evidence, which does not support their model, that low-severity fires dominated and were frequent across dry forests, was omitted in H.

4.2.2. Evidence about Historical Fire Severity Omitted/Misinterpreted in H Table 5

In this section, which is about H Table 5, we show that H's low-severity fire model is based on more false and omitted evidence, in this case about historical fire severity: (a) published validations of WB-method fire-severity reconstructions vs. independent multi-proxy sources in both modern and historical forests, (b) abundant independent, multi-proxy evidence of historically severe fires, including tree-ring reconstructions and evidence in H's own publications, (c) incorrect use of fire scars and age structures omits the role of historically severe fires in H, (d) evidence that historical dry forests, in general, lacked high-

severity fires falsely based on tree-ring reconstructions of fire targeted in old growth, where high severity is inherently rare, (e) Baker et al.'s [19] rebuttal of Hagmann et al. [18] on high-severity fire, omitted in H, (f) Odion et al.'s [95] rebuttal of Stevens et al. [96], showing FIA data still can reconstruct fire severity, omitted in H, and (g) rebuttal, new evidence of historically large high-severity fire patches, omitted in H.

- Published validations of the WB-method fire-severity reconstructions vs. independent multi-proxy sources in both modern and historical forests, omitted in H

Williams and Baker [24] calibrated and then validated their fire-severity reconstruction method using information directly from tree-ring reconstructions or direct measurements from historical forest plots where fire severity had been assessed. Methods were directly calibrated using 55 estimates from areas where low-severity fire was dominant and from 9 areas where mixed- or high-severity fire was dominant. The calibrated definitions and methods correctly predicted fire severity at all the low-severity sites and all but 1 of the higher-severity sites, which was incorrectly assigned low severity as the high-severity event occurred 300 years ago.

For historical cross-validations with independent sources, Baker and Williams [34] reported that “For historical fire severity, 10 specific cross-validations in six study areas in four states had high mean accuracy of 89.1–90.1%, based on PSC . . .” [34] (p. 288), with the individual cross-validations listed in their Appendix S1 Table S7. PSC is percent similarity in composition. Additionally, Baker and Williams [34] reported that “There is substantial corroborating evidence that moderate/mixed-to-high-severity fires occurred and were extensive in some areas, based on evidence for five study areas in four states . . . These include 99 quotes from early forest reserve and other reports, four tree-ring reconstructions, two paleo studies, and two using early photographs.” This evidence was presented in detail in their Appendix S1 Tables S1 and S11 in [34], but was omitted in H.

Furthermore, Williams and Baker [31] validated the use of survey section-line data to characterize the modern moderate- to high-severity fire regime in the Colorado Front Range (Figure 6) and then analyzed 6904 km of historical section-line records and found a historical higher-severity fire rotation of 249 years. This estimate is similar to, and independent of, the WB method estimate (271 years) from Williams and Baker [24] for part of this area, further validating the WB method. Also important is that this is independent, direct surveyor-recorded evidence of historical moderate- to high-severity fires in historical dry forests, which does not use the WB method that H critiqued. All this evidence was omitted in H.

- Abundant independent, multi-proxy evidence of historically severe fires, including tree-ring reconstructions and evidence in H's own publications, omitted in H

The authors of H previously omitted evidence of historically severe fires in dry forests, the start of a pattern of omission of evidence. Fulé et al. [80], among whom were eight authors of H, said:

“W&B also fail to acknowledge the lack of contemporary evidence for large, patch-size crown fires in low- and mid-elevation dry forest landscapes, such as primary observation or photographic documentation in the 19th and early 20th centuries. The lack of direct documentary evidence of extensive crown fire in ponderosa pine forests in particular has been noted and reported repeatedly by ecologists and land-use historians for nearly 90 years . . .”. [80] (p. 826)

Williams and Baker's study [24], which they were critiquing, had, in fact, summarized direct documentary evidence of high-severity fires in their study areas in AZ, CO, and OR in this very publication ([24] Appendix S1). This evidence included early journal articles from the turn of the century, forest reserve reports by government scientists, analysis of early aerial photographs, tree-ring and fire scar studies, and paleo-charcoal reconstructions. All this evidence was omitted in Fulé et al. [80] and also omitted in H.

Early aerial photographic evidence of historically severe fires across 178,902 ha of mixed-conifer forests in eastern Washington and Oregon was published by Hessburg et al. [11], another author of H. However, as noted in Section 2, H remarkably omitted any review of the extensive evidence in this publication (e.g., Figure 4). A quote from this paper, also summarizing this omitted evidence, is reproduced in Section 2. Yet another author of H in Merschel et al. [97] thought “the wave of tree establishment that began in ~1900 ... was likely caused by a variety of factors, including changes in fire regimes, selective tree harvesting, and domestic livestock grazing” [97] (p. 1684), but rejected Baker’s [29] finding that moderate- to high-severity fires occurred in the late 1800s, explaining that “it would require moderate- to high-severity fires occurring over an immense area ... before 1900. Such fires are not recorded in written archives or tree-ring records from the region.” [97] (p. 1684). However, Baker ([29] Supplemental Materials Appendix A) presented evidence from written archives in early forest-reserve reports and other scientific reports of extensive high-severity fires in the late 1800s in and near Merschel et al.’s study area, evidence omitted in Merschel et al. [97] and in H.

A large body of independent evidence of historical mixed-severity fires, mentioned in other sections, was also omitted in H. Baker et al. [35] published 43 quotes from ca 1900 forest reserve reports from throughout the Rocky Mountains that showed a diversity of historical fire severities, including abundant evidence of moderate- and high-severity fires. Baker [45] published six early photographs of the aftermath of severe fires in dry forests in the Rocky Mountains, one reproduced here (Figure 3a). Baker ([28] Appendix A) published 208 quotes from early forest-reserve reports and other early scientific reports that documented historical moderate- to high-severity fires in Sierran mixed-conifer forests, including evidence that these fires burned extensive mature forests (Figure 5). Baker [32,33,44] documented that large moderate- to high-severity fires in the late 1800s occurred in dry forests on the Uncompahgre Plateau and in the San Juan Mountains, Colorado. This was based on forest atlases, early photographs, early scientific publications, and other early historical records, including newspaper reports. Figure 2b shows extensive dense tree regeneration after severe fires on the Mogollon Plateau in the late 1800s. All this independent evidence was omitted in H.

H said that “... fire scar records remain a primary means of exploring historical ecology” [3] (p. 8) and only used these records and modeling as their basis for evaluating changes in fire. However, there have been ≥ 18 tree-ring reconstructions that found evidence of moderate- to high-severity fires in historical dry forests, several of which were by authors of H. Many of these fire-scar studies, which found severe historical fires, were already reported in Odion et al. [12], including 6 published studies from the southern Cascades and Sierra in California, 1 from British Columbia, 10 from the Rocky Mountains (e.g., Figures 3b and 10a), and 2 from the Southwest. Others include Wu [98] and Tepley and Veblen [99] in the San Juan Mountains. Remarkably, again, H did not cite or review Brown et al. [100] and Huckaby et al. [101] from the Colorado Front Range, which included an author of H and together documented severe historical fires in dry forests. H also did not cite Taylor and Skinner [102], which included another author of H and showed tree-ring evidence of moderate- to high-severity fires in dry forests in the Klamath Mountains, California. The idea that there are no independent tree-ring reconstructions of historically severe fires in dry forests has been incorrect for over two decades. All this evidence, including studies by authors of H, was omitted in H.

H also did not cite or review that there have been seven paleo-charcoal studies that found evidence of severe fires in the last 500–600 years in dry forests, cited in Table 1 in Baker [4] with his estimates of fire rotation. These include studies by Long et al. [103] from the eastern Cascades, Oregon (estimated fire rotation = 333 years), Fitch [104] from northern New Mexico (~500 years), Pierce et al. [36] and Pierce and Meyer [37] from central Idaho (154–286 years, mean = 220 years), Jenkins et al. [105] from northern Arizona (250 years), Bigio [106] from southwestern Colorado (>471 years), and Colombaroli and Gavin [107] from southern Oregon (500 years). The overall estimated high-severity fire rotation from

these studies, cited in Baker [4], had a mean of ~379 years and a range of 154–500 years. The mean is 515 years, and the range 217–849 years from eight land-survey reconstructions [4]. Both sources, which are independent, document and validate each other in showing that infrequent high-severity fires occurred historically in dry forests. All this independent evidence was omitted in H.

The repeated idea that there are no independent records of historically severe fires in dry forests is clearly incorrect. These records have been known since the early 1900s and reported by us and other authors since the 1990s, including in reviews by Odion et al. [12] and Hanson et al. [13], several land-survey reconstructions (Table 1), and other papers. Evidence from eight authors of H earlier in Fulé et al. [80] omitted a large body of evidence of historical moderate- to high-severity fires. Evidence in H again here omitted all evidence from early forest-reserve reports and other early reports, analysis of early aerial photographs and early oblique photographs, landscape-scale tree-ring reconstructions, forest atlases, newspaper reports, and paleo-charcoal reconstructions, including studies by authors of H. This evidence alone is more than sufficient, without land survey reconstructions using WB methods, to reject the low-severity model and support the mixed-severity model.

- Incorrect use of fire scars and age structures omits role of historically severe fires in H

In H Table 5, the citation of Brown [108] as a counter to Shinneman and Baker [10] repeats an incorrect interpretation of evidence by Brown [108]. Prior to Brown [108], Brown et al. [100] and Huckaby et al. [101] studied fire history in the Colorado Front Range. Trees that died about the time of a dated fire and abundant trees that regenerated after a dated fire provided evidence of high-severity fire, as also shown in Figure 3b. However, a little later, Brown and Wu [86] found the same evidence yet interpreted tree regeneration pulses as having an unknowable disturbance cause, but instead regional climate forcing: “... cohort structure is uncoupled from any single mortality event and instead appears to be the result of broader scale climate forcing of fire timing that resulted in successful recruitment episodes” [86] (p. 3036). This is not a valid inference, as right before the tree-regeneration pulses was strong evidence of fire, which was not excluded as a cause of post-fire tree regeneration. The flaw in the Brown and Wu [86] interpretation is that disturbance history and climate history can be confounded; to determine the effect of one, the other must be controlled, which Brown and Wu did not do. It is not possible to validly conclude climate forcing alone was the cause without showing fire was not the cause of tree regeneration pulses; more likely, both contributed, with fire being the primary cause.

The next year, Brown ([108] Figure 3) showed the same evidence that should have led to the recognition of confounding and the possible role of high-severity fire, as interpreted earlier [100,101], but Brown said “Abundant synchronous tree recruitment affected by optimal climate forcing is probably the reason for extensive stands of even-aged forests in the Black Hills, rather than widespread crown fires ...” [108] (p. 2507). However, Brown [108] provided no explanation for how trees present before this period were all killed so that regenerating stands became even-aged. If trees had not been mostly killed prior to a pulse of tree regeneration, resulting stands would not have been even-aged but instead multi-aged. Evidence of a fire and this logic effectively excludes climate forcing by wet periods as the sole cause of regeneration pulses. Again, the more likely explanation was not analyzed, which is that moderate- to high-severity fires, followed by a climate favorable for regeneration, produced the evidence presented in Brown ([108] Figure 3). Failure to exclude a confounded variable, fire, before assuming climate forcing as the sole cause, has been a repeated error, as also made by O’Connor et al. [109].

The climate forcing model of tree recruitment pulses was not supported in a key test. Dugan and Baker [52] directly tested whether fires, fire-quiescent periods, droughts, or pluvials, in some combination or permutation, had separate or combined influences on the occurrence of historical tree recruitment pulses in ponderosa pine forests in Grand Canyon National Park, Arizona. The conclusion was:

“Permutation analysis showed that mortality-inducing influences of fire and drought played the primary role in initiating pulses as they occurred first for 90% of pulses, significantly more than expected . . . drought was the most important single initiator . . . as the first influence for 65% of pulses. Mixed-severity fire was the initial influence for 30% of fires . . . none of the 20 pulses had a pluvial influence alone”. [52] (p. 704)

Table 5. Hagsmann et al.’s ([3] Table 5) about severity of historical fires is replicated in the left two columns, with omitted published evidence added in the right column, to show Hagsmann et al. omitted published evidence and made incorrect conclusions as a result.

Counter-Evidence		Evaluation of Counter-Evidence		Omitted Rebuttals and Other Published Evidence also Essential to Evaluation of Counter-Evidence	
Citations	Counter-premise	Citations	Implications of evaluation	Citations	Implication of omitted evidence
Shinneman and Baker [10]	Based on early forest inventory age datasets, “nonequilibrium” areas of extensive, high-severity fires in the Black Hills led to landscapes dominated by dense, closed-canopy forests	Brown [108]	Tree-ring reconstructions of ponderosa pine forest age structures and fire regimes across the Black Hills found synchronous regional tree recruitment largely in response to pluvials and longer intervals between surface fires, especially during the late 1700s/early 1800s, which is when early inventory data report similar patterns of recruitment. No evidence of crown fires was found in relation to past fire dates	Omitted evidence in Brown [108], Dugan and Baker [52]	Brown [108] said “ . . . dense stands were still present at settlement . . . , and likely contributed to extensive patches of crown fire noted by early explorers and scientists during the late 1800s (Dodge 1965, Graves 1899)” (p. 2509). This is entirely consistent with early reports of severe fires across the Black Hills by Graves and Dodge [10]. However, Brown [108] (p. 2507) also interpreted “abundant synchronous tree recruitment” as caused by climate forcing (e.g., wet periods), without excluding these high-severity fires as also a cause. The lack of control of this potentially confounded causative variable leaves Brown’s conclusion invalid. A test of Brown’s conclusion at the Grand Canyon did not support his findings [52].
Baker et al. [35]	Most ponderosa pine forests in the Rocky Mountains were capable of supporting high-severity crown fires as well as low-severity surface fires	Brown et al. [83]	Tree-ring reconstruction of ponderosa pine forests in the Black Hills of South Dakota (included in Baker et al., 2007) demonstrated that roughly 3.3% of the study area burned as crown fire between 1529 and 1893; however, tree density in most stands in 1870 could not have supported crown fire	Omitted evidence in Baker [25], Baker et al. [35]	This review [25] emphasized that the historical fire regime in the Rocky Mountains included variable fire severities with some areas having mostly low severity in old-growth forests. The 517 ha Mount Rushmore area of [83] is consistent with this review. Additionally, old growth inherently exists because it lacked high-severity fire for long periods, so it is impossible to draw valid conclusions about larger land areas from these old forests, since they are not random samples of larger land areas [25].
Williams and Baker [24], Baker [28,29]	Fire severity inferred from tree density by size class estimated from GLO bearing trees [41] and surveyors’ descriptions suggests low-severity fire dominated only a minority of ponderosa and mixed-conifer forests	Levine et al. [61,62]	Plotless density estimator used by Williams and Baker [41] overestimated known tree densities due to a scaling factor that does not correct for the number of trees sampled and therefore systematically underestimates the area per tree relationship	Omitted Rebuttals by Baker and Williams [34,63]	Levine et al. [61] incorrectly coded and applied the WB method, producing erroneous results that had no bearing on the WB method. Levine et al. [62] corrected their flawed 2017 code [61] but then used incorrect equations. Baker and Williams [63] used corrected equations with their code at their sites and showed that the WB method worked well and that both Levine et al. studies are fatally flawed.

Table 5. Cont.

Counter-Evidence		Evaluation of Counter-Evidence		Omitted Rebuttals and Other Published Evidence Also Essential to Evaluation of Counter-Evidence	
Citations	Counter-premise	Citations	Implications of evaluation	Citations	Implication of omitted evidence
Williams and Baker [24], Baker [28,29]	Fire severity inferred from tree density by size class estimated from GLO bearing trees [41] and surveyors' descriptions suggests low-severity fire dominated only a minority of ponderosa and mixed-conifer forests	Fulé et al. [80], Merschel et al. [97], O'Connor et al. [109]	Substantial errors of method and interpretation invalidate inferences about historical fire severity. These include the fact that (1) tree size is an ambiguous indicator of tree age; (2) tree regeneration is an ambiguous indicator of disturbance severity, particularly in dry forests where climate conditions strongly influence regeneration; and (3) lack of direct documentary evidence (e.g., primary observation) of extensive crown fire in historical ponderosa pine forests has been widely noted for nearly 90 yr.	Omitted Rebuttal in Williams and Baker [79], Omitted evidence in Baker [4,25,28,29]	Williams and Baker [79] showed that Fulé et al. [80] mistook the WB method, misquoted WB, misused evidence, and created three new false narratives. Merschel et al. [97] did not contest the WB method but said there were no reports of high-severity fires in the late 1800s, even though direct reports of them were extensively quoted by Baker [28,29]. O'Connor et al.'s study [109] has no bearing on the WB method. Extensive evidence of crown fires in historical ponderosa pine forests is widely published and reviewed in [4] and the text here. Baker [25] also showed, using tree-ring reconstructions, that low-severity fire was the primary severity across only ~34% of historical dry forests, mostly in the Southwest.
		Stephens et al. [73], Huffman et al. [94], Miller and Safford [110], Hagmann et al. [54]	Multi-proxy records documented substantially lower levels of high-severity fire in ponderosa and Jeffrey pine and mixed-conifer forests in overlapping study areas	Omitted evidence in Baker and Hanson [17] and Baker [29]	Baker and Hanson [17] showed that Stephens et al.'s lower estimate is because they omitted timber inventory documents that recorded high-severity fires. Huffman et al.'s study does not overlap our study area. Miller and Safford repeated critiques we already refuted (see above). However, GLO reconstructions identify fires before the mining era, and their finding of few trees at low elevations today likely is due to loss of low-elevation forests. Hagmann et al.'s [54] estimate of 6% high-severity fire is similar to that of Baker [29] of 8.9% historically.
Baker [29], Baker and Hanson [17]	Estimates of area burned at high severity by Hessburg et al. [11] validate estimates derived using Williams and Baker [41] methods <i>Note: [17] Baker and Hanson 2017 did not belong here, as it has nothing to do with the Hessburg et al. matter, so it was not defended</i>	Hagmann et al. [18], Spies et al. [111]	Inappropriate comparisons are not validation. Baker [29] limited assessment of high-severity fire to tree mortality in dry forests whereas Hessburg et al. [11] estimated high-severity fire in the dominant cover type whether that be grass or tree for "moist and cold forest" type, with lesser amounts of dry forests	Omitted and incorrect evidence in Hessburg et al. [11]	This argument is incorrect. Hessburg et al.'s Table 2 shows that specifically in forest cover types (not grass, shrub), their pooled forest percentages in ESR5 were 20.7% low, 55.0% moderate, and 24.3% high, which are even more similar to the Baker [29] estimates of 18.1% low, 59.9% moderate, and 23.0% high. Hessburg et al. Figure 4 also shows that ponderosa and Douglas-fir cover types had a mean of about 18% low, 59% moderate and 23% high, almost identical to the Baker [29] estimates.

Table 5. Cont.

Counter-Evidence		Evaluation of Counter-Evidence		Omitted Rebuttals and Other Published Evidence also Essential to Evaluation of Counter-Evidence	
Citations	Counter-premise	Citations	Implications of evaluation	Citations	Implication of omitted evidence
Odion et al. [12]	Modern, high-severity crown fires are within historical range of variation. Inferred fire severity from current tree-age data for unmanaged forests in the U.S. Forest Service Inventory and Analysis (FIA) program. Compared inferences about modern fire severity to estimates of historical forest conditions and fire severity inferred using Williams and Baker [41] methods	Fulé et al. [80], Levine et al. [61,62], Knight et al. [64]	Overestimation of historical tree density and unsupported inferences of fire severity from GLO records weaken conclusions based on Williams and Baker [41] methods	Omitted Rebuttals in Williams and Baker [79], Baker and Williams [34,63]	Fulé et al. [80] mistook the WB method, misquoted publications, misused evidence, and created three new false narratives [79]. Levine et al. incorrectly coded the WB method [61], then used incorrect equations [62], and both are fatally flawed. Knight et al. did not use or test the WB method and has no relevance.
		Stevens et al. [96]	Substantial errors of method and interpretation invalidate inferences about historical fire severity. These include (1) FIA stand age variable does not reflect the large range of individual tree ages in the FIA plots and (2) recruitment events are not necessarily related to high-severity fire occurrence	Omitted rebuttal in Odion et al. [95]	With same definition of high-severity fire, there was 68% agreement between these two studies; 3/4 of evidence of historical high-severity fire not from FIA data and not disputed; Stevens et al. agreed “High-severity fire was undoubtedly a component of fire regimes in ponderosa pine and drier mixed-conifer forests”
		Spies et al. [111]	In contradiction of the counter-premise, Odion et al. [12] documented only three patches of high-severity fire larger than >1000 ha in OR and WA in the early 1900s, which account for 1% of the area of historical low-severity fire regime managed under the Northwest Forest Plan	Omitted rebuttal in Odion et al. [95]; Omitted evidence in DellaSala and Hanson [112,113]	Two sources in omitted 2015 paper, reviewed in the omitted Odion et al. [95] paper, found high-severity patches $\geq 14,000$ ha in OR and WA, two others [112,113] found many large patches in OR and WA; numerous other large patches > 1000 ha reported in OR and WA in omitted [113] paper.
Baker and Hanson [17]	Stephens et al. [73] underrepresented the historical extent of high-severity fire in their interpretation of surveyor notes in early timber inventory. <i>Note: because they omitted key records of high-severity fire that were readily available in the inventory records.</i>	Hagmann et al. [18]	Substantial errors of method and interpretation invalidate inferences about the historical extent of high-severity fire. Inferences were based on (1) inappropriate assumptions about the size and abundance of small trees given the ambiguity of data describing small trees in the 1911 inventory, (2) averaging of values derived from different areas and vegetation classifications, and (3) inappropriate assumptions that the presence of chaparral (common on sites with thin soils and high solar radiation) indicates high-severity fire	Omitted rebuttal in Baker et al. [19]	Hagmann et al. [18] did not dispute that Stephens et al. [73] had omitted most trees, and when omitted trees were included, forests were 7–17 times as dense as they reported, and they also did not dispute the abundant data, from numerous historical sources, showing occurrence of substantial high-severity fire patches, small and large, including chaparral, presented in Baker and Hanson [17]

It remains essential to test for effects of canopy-opening disturbances before assuming that moist periods alone trigger regeneration pulses; this test showed moist periods do not trigger pulses without a canopy-opening event, such as a moderate- to high-severity fire, drought, or possibly a beetle outbreak (not reconstructed). The climate forcing conclusions of the Brown studies are invalid because (1) no evidence was presented to exclude that severe fires were instead, or in addition, the cause of pulses, and (2) regeneration in moist periods cannot alone produce even-aged stands, as existing trees must be killed first. The analyses [52] that must be completed to be able to exclude high-severity fire or other severe disturbances were not performed, invalidating the conclusion of just climate forcing in

studies by Brown and others [86,108,109]. Moreover, Dugan and Baker's study [52], which performed valid testing and did not find forcing by wet periods, was omitted in H.

- Evidence historical forests, in general, lacked high-severity fires, falsely based on tree-ring reconstructions of fire targeted in old growth, where high severity is inherently rare

Tree-ring reconstructions of fire history have commonly been biased against the detection of historical moderate- to high-severity fires because tree-ring reconstructions require old trees, which could not have persisted until today without a lack of moderate- to high-severity fires for long periods. Older methods of collecting fire history evidence targeted these old forests, which strongly biased these methods against finding evidence of historical moderate- to high-severity fires. For example, Grissino-Mayer [114] showed this strong and intentional bias against detecting historical high-severity fires when he said of dry forests in volcanic landscapes of northern New Mexico:

"We found no fire-scarred samples on the kipukas in the northern and eastern portions of the malpais, and found few samples in the southern portions. These areas contained ponderosa forests that appeared younger than elsewhere, perhaps due to more recent, intense stand-replacing fires . . . ". [114] (p. 136)

This study performed no analysis of fire severity and instead just intentionally excluded areas with possible evidence of high-severity fires, as documented in this quote, and targeted older forests with abundant scars and lower-severity fires. Conclusions about historical fire severity from this biased sampling cannot be extrapolated to other areas but often are. This bias is not unusual for fire history studies in dry forests. Baker [25] found that 32% of 342 fire-history sites targeted plots in old forests with concentrations of fire scars, where moderate- to high-severity fires likely had not occurred. Moreover, 74% of fire history sites did not include any analysis of fire severity and just assumed historical fires were low severity. In contrast, where fire severity was studied, some higher-severity fire was usually found, showing the low-severity bias in most studies [25].

Two examples illustrate this bias. Brown's study [108], which is cited in H Table 5 as countering Shinneman and Baker's [10] finding of historically severe fires in the Black Hills, based on historical records, was similarly conducted in mostly old growth, where the probability of finding high-severity fires is inherently low. Therefore, it is expected that Brown [108] would find little evidence of historical high-severity fires because this is a highly biased sample. This evidence from a small area of old growth of course does not counter evidence of moderate- to high-severity historical fires in the much larger Shinneman and Baker [10] study area in the Black Hills. Merschel et al. [97], similarly, intentionally sampled in "areas of older forest" [97] (p. 1673) but nonetheless claimed that "The ubiquitous presence of large, multi-aged ponderosa pine at all sites, regardless of environmental setting, suggests historical fires were frequent and predominantly low severity . . . " These two cases illustrate the inherent bias, not admitted by either author, against finding evidence of historical mixed- and high-severity fires from studying fires in areas of older forest.

Thus, most previous fire-history studies, including those cited in H Table 5, Brown [108], and Merschel et al. [97], do not provide valid inferences about historical fire severity across larger landscapes, as they are not random samples and are mostly from rarer old-growth forests that inherently lacked moderate- to high-severity fires for long periods. This is a bias recognized by Hessburg et al. [11] and expanded by Baker [25]. H also knew this, as they said " . . . landscape-scale assessments and simulation models encompassing multiple forest types can address concerns that sampling bias of fire-scar studies favors detection of low-severity fire regimes (e.g., see arguments in Hessburg et al., 2007)." Yet, all landscape-scale fire histories, including that presented by Hessburg et al. [11], that avoid this known bias from sampling only in older forests, were omitted in H.

- Baker et al.'s [19] rebuttal of Hagmann et al. [18] on high-severity fire, omitted in H

H, in their Table 5, cited Hagmann et al. [18] as evidence ostensibly rebutting Baker and Hanson [17] regarding their findings of historical high-severity fires in ponderosa pine and mixed-conifer forests of the Sierra Nevada. However, Baker et al.'s study [19], which rebutted Hagmann et al. [18], was omitted in H. Baker et al. [19] explained that the Hagmann et al. [18] critique actually did not challenge or dispute the abundant evidence of historical high-severity fire presented by Baker and Hanson [17]. This evidence included (a) extensive U.S. Forest Service field notes and maps documenting the occurrence of high-severity fire, and young, naturally regenerating conifer forests following severe fire from forest surveys circa 1911 in two different areas of the Sierra Nevada and (b) explicit notes and observations from three U.S. Forest Service reports, ca 1904 1912, regarding small and large high-severity fire patches and naturally regenerating conifer forest following a severe fire. The rebuttal [19] and all this evidence was omitted in H.

- Odion et al.'s [95] rebuttal of Stevens et al. [96], showing FIA data can still reconstruct fire severity, omitted in H

H Table 5 argued that Stevens et al. [96] had shown that “errors of method and interpretation invalidate inferences about fire severity” from FIA stand-age data. However, the rebuttal of Stevens et al. [96] by Odion et al. [95] was omitted in H. The Odion et al. [95] rebuttal found/noted that (a) with the same definition of high-severity fire, there was 68% agreement between Stevens et al. [96] and Odion et al. [12] in terms of classifying historical high-severity fire using FIA stand-age plot-data; (b) 75% of the evidence for historical high-severity fire, which did not pertain to FIA, was not disputed or challenged by Stevens et al. [96]; and (c) while Stevens et al. questioned whether the current occurrence of high-severity fire patches >1000 ha is within the natural range of variation, they acknowledged that “High-severity fire was undoubtedly a component of fire regimes in ponderosa pine and drier mixed conifer forests” [96] (p. 3), including patches >50 ha in area. The Odion et al. rebuttal [95] and this evidence were omitted in H.

- Rebuttal/new evidence of historically large high-severity fire patches, omitted in H

H Table 5 argued that Spies et al. [111] had shown that Odion et al. [12] documented “only three patches of high-severity fire larger than >1000 ha in OR and WA in the early 1900s”. However, the rebuttal of Stevens et al. [96] by Odion et al. [95] was omitted in H. The Odion et al. [95] rebuttal summarized data presented by DellaSala and Hanson [112] (p. 31), wherein four different sources were discussed regarding the historical occurrence of high-severity fire patches >1000 ha in mixed-conifer and ponderosa pine forests of OR and WA. Two of these sources documented individual high-severity fire patches of 14,000 ha and 24,000 ha, while the other two sources documented dozens of occurrences of such patches. Additional data regarding numerous historical high-severity patches of this size in OR and WA, as well as the Sierra Nevada and elsewhere across the western USA, were presented by DellaSala and Hanson [113], new evidence that was also omitted in H. Baker [28] (p. 26) had reported for the Sierra that “... the reconstructions show that contiguous areas of historical high-severity fire commonly exceeded 250 ha and reached as high as 9400 ha”, which was also omitted in H. Furthermore, in the Colorado Front Range, Williams and Baker [31] found that the maximum historical high-severity patch size was 8331 ha based on direct surveyor reports along section lines, evidence also omitted in H. Thus, all this evidence that rejects the low-severity model was omitted in H.

4.2.3. Evidence about Recent Fire Severity Omitted/Misinterpreted in H Table 6

H (Table 6) claimed that Odion and Hanson [115] stood for the proposition that “High severity fire was rare in recent fires”, whereas Odion and Hanson [115] actually stood for the proposition that long unburned forests are not experiencing higher fire severity in modern fires. H also cited Safford et al. [116] as rebutting Odion and Hanson [115] but failed to mention that Safford et al.'s study [116] was refuted by Odion and Hanson [117].

Odion and Hanson [117] found that Safford et al. [116] had arbitrarily combined two time-since-fire categories, which created a false impression of slightly higher fire severity in long unburned forests. Odion and Hanson [117], using the same vegetation severity data, analyzed all time since fire categories and found that forests, that had not burned in the longest period of time, had similar or lower fire severity, not higher severity.

H also cited Spies et al. [118] as rebutting Hanson et al. [119] regarding current fire severity trends but failed to mention that Spies et al. [118] was subsequently refuted by Hanson et al. [120]. Hanson et al. [120] found that a mathematical error, and reliance on an inaccurate anecdotal assertion, had led to an erroneous conclusion that the rate of high-severity fires in old forests of the Pacific Northwest was outpacing the old-forest recruitment rate from growth. Widespread rollbacks of forest protections, and increased logging, were being proposed based on the false data. Spies et al. [118] did not dispute that the errors had been made but hypothesized that the initial conclusion might still hold if a much broader high-severity fire definition was used. Hanson et al. [120] analyzed the Forest Service's own fire-severity field-plot validation data and rates of high-severity fire in old forests from satellite imagery, finding that, even with the broader high-severity fire definition, old-forest recruitment still outpaced the rate of high-severity fire in old forests by 7 to 29 times, depending on the subregion, and most mature trees survived fire under this broader definition.

H listed a few studies as rebutting Williams and Baker's [24] evidence that severity distributions in some modern wildfires were not different from severity distributions in historical fire patterns they reconstructed. However, H did not mention or cite the many published studies, discussed above, that have refuted these critiques, or the rebuttals and other counter-evidence regarding these few studies. Steel et al. [121] reported no relationship between time since fire and high-severity fire for some forest types. They reported such a relationship for mixed conifer, but the model was based on data for only one narrow time-since-fire category, and the authors excluded from their analysis the longest unburned forests, those with no recorded history of fire ([121] Table 4, Figure 4). Evidence presented by Odion et al. [122], Miller et al. [123], and van Wagendonk et al. [124], which included the longest unburned forests and all time-since-fire categories, and found similar or lower proportions of high-severity fire in the longest unburned forests, was omitted in H. Steel et al. [121] also reported historical high-severity fire proportions of 4–8% for mixed-conifer forests, based on only a theoretical model, but both Steel et al. [121] and evidence in H omitted mention of numerous studies finding much higher historical proportions of high-severity fire in these forests based on historical field data, maps, and reports, including those by Baker [28], Hanson and Odion [76,78], and Baker and Hanson [17]. Steel et al. [125] reported an increase in high-severity fire proportion since 1984 in some regions but used a fire-history database that is known to disproportionately omit large, severe fires in the earlier years of the dataset, causing a bias and potential to report false trends [126]. Evidence in H omitted Hanson and Odion [126] and Baker [4], who used more comprehensive data and found no trends in high-severity fire proportion in the same regions. Guiterman et al. [127] analyzed a single 38 ha high-severity fire patch, with very limited inferential potential for landscapes. Reilly et al. [56] reported no increase in high-severity fire proportion in the Pacific Northwest since 1985 but an increase in large high-severity fire patches. Evidence in H omitted DellaSala and Hanson [113], who found that the increase in large high-severity fire patches occurred from the 1980s through the 1990s but found no statistically detectable increase over about the past two decades. H cited Safford et al. [128] as rebutting Hanson and Odion [129] but neglected to cite or mention that Safford et al.'s study [128] was refuted by Hanson and Odion [126]. Safford et al. [128] questioned fire severity trend analyses reported by Hanson and Odion [129] for the Sierra Nevada and hypothesized several potential methodological flaws. Hanson and Odion [126] re-analyzed their initial data, using the new methods proposed by Safford et al. [128], and found their initial conclusions were robust to re-analysis under Safford et al.'s new methods. All this substantial evidence in published papers and rebuttals was omitted in H.

Table 6. Hagmann et al.’s study ([3] Table 6) about severity of modern fires is replicated in the left two columns, with omitted published evidence added in the right column, to show Hagmann et al. omitted published evidence and made incorrect conclusions as a result.

Counter-Evidence		Evaluation of Counter-Evidence		Omitted Rebuttals and Other Published Evidence also Essential to Evaluation of Counter-Evidence	
Citations	Counter-premise	Citations	Implications of evaluation	Citations	Implication of omitted evidence
Odion and Hanson [115]	High-severity fire was rare in recent fires in the Sierra Nevada based on analysis of Burned Area Emergency Response (BAER) soil burn severity maps	Safford et al. [116]	BAER maps greatly underestimate stand-replacing fire area and heterogeneity in burn severity for vegetation. BAER maps are soil burn-severity maps, not vegetation burn-severity maps.	Omitted rebuttal in Odion and Hanson [117]	Safford et al. [116] arbitrarily combined two time-since-fire categories, creating slightly higher fire severity in long unburned forests. Odion and Hanson [117] used all categories and found similar or lower fire severity in long unburned forests.
Hanson et al. [119]	Change in conservation strategies for northern spotted owl (NSO) were unwarranted due to overestimation of high-severity fire in the NSO recovery plan	Spies et al. [118]	Use of a higher relative delta normalized burn ratio (RdNBR) threshold substantially increased misclassification errors and reduced estimates of high-severity fire extent. Hanson et al. [120] used an RdNBR threshold of 798 rather than 574 as recommended in the literature in Miller et al. [130] they cited as the source of the threshold used	Omitted rebuttal in Hanson et al. [120]	Spies et al. [118] had cited evidence with a math error and incorrect anecdotal evidence to conclude high-severity fire was outpacing old-forest recruitment but did not dispute these and then tried a broader high-severity fire definition. Hanson et al. [120], however, showed this new definition still led to old-forest recruitment outpacing high-severity fire by 7–29 times
Williams and Baker [24]	Severity distributions in recent fires do not depart from historical	Steel et al. [121], Guiterman et al. [127], Reilly et al. [56], Steel et al. [125]	Extent and spatial patterns of fire severity in some recent fires have departed from pre-fire exclusion range of variation for some forest types	Omitted evidence in Odion et al. [122], Hanson and Odion [126,129], Della-Sala and Hanson [113], and many others (see text)	Steel et al. [121] based historical high-severity proportions on only a theoretical model. Guiterman et al. [127] was from only one 38 ha patch, with little inferential power. Reilly et al. [56] found no trend in high-severity proportion and more large, high-severity patches, but H omitted the study by DellaSala and Hanson [113] who found no such increase over the last two decades. Steel et al. [125] used a database that Hanson and Odion [126] showed can produce false trends.
Hanson and Odion [129]	Previous assessments overestimate extent of high-severity fire in modern fires	Safford et al. [128]	Use of coarse-scale, highly inaccurate, and geographically misregistered vegetation map and averaging across unrelated vegetation types and diverse ownerships undermine confidence in Hanson and Odion [129]	Omitted rebuttal in Hanson and Odion [126]	Hanson and Odion [126] re-analyzed Safford et al.’s [128] initial data, using new methods that Safford et al. proposed, and found Hanson and Odion’s [129] initial conclusions were robust to re-analysis using Safford et al.’s [128] proposed new methods.

4.2.4. Conclusions—Abundant Multi-Proxy Evidence of Historical Moderate- to High-Severity Fires, including in Their Own Publications, Omitted in H

Fire-history research has moved beyond old composite fire interval (CFI) rate measures, as reviewed in the Introduction, but H cited old debates about CFIs. Publications on new methods that use the much sounder fire-rotation measure and have corrected old CFI measures to fire rotations [25] were omitted in H. These new estimates show that frequent low-severity fire was much less prevalent than previously thought (Figure 8). Low-to moderate-severity fire remains supported as having been at least temporarily dominant historically across a substantial area in the Southwest and scattered elsewhere (Figure 8, Table 1), usually at lower elevations and on drier sites (e.g., Figure 10a). However, frequent low-severity fires with rotations <25 years only covered 14% of dry-forest areas [25]. Even in these areas, it was historically possible for fires to infrequently burn extensive areas at moderate to high severity (e.g., compare the Mogollon Plateau and nearby Black Mesa in

Figure 2). Hessburg et al. [11] (p. 20) articulated this contingent equilibrium in eastern Washington mixed-conifer forests:

“Low-severity fires and equilibrium dynamics likely occurred in eastern Washington dry forests, where they fostered fire tolerant, park-like pine stands, however, these dynamics were perhaps ephemeral in nature, lasting one or more centuries at a location, and then switching concordant with regional climate forcing to non-equilibrium states”.

Table 7. Reported early timber inventory tree-density estimates and corrected estimates with 1.6–2.3 correction multipliers applied, along with estimated total tree density (conifer and hardwood). Data are from studies that used early timber inventories to estimate historical tree density in dry forests.

Study Area	Source	Tree Diam. Recorded (cm)	Trees Recorded	Reported Tree Density (Trees/ha)	Corrected Tree Density (Trees/ha)	Estimated Conifer Plus Hardwood Tree Density (Trees/ha)
<i>Two-chain-wide timber inventories that underestimate tree density by 16–2.3 times (Baker et al., 2018 [19])</i>						
E. Ore. Cascades-N	Hagmann et al. [70]	15.0+	Main conifers	66	106–152 ^a	106–152 ^a
E. Ore. Cascades-S	Hagmann et al. [69]	15.0+	Main conifers	65	104–150 ^a	106–152 ^a
E. Ore. Cascades-S	Hagmann et al. [71]	15.0+	Main conifers	68	109–156 ^a	109–156 ^a
S. California Sierra	Collins et al. [131]	15.2+	Only conifers	44–52	70–120 ^a	90–155 ^b
S. California Sierra	Collins et al. [72]	15.2+	Only conifers	48	77–110 ^a	99–142 ^b
S. California Sierra	Scholl & Taylor [132]	15.2+	All trees	99	158–228 ^a	158–228 ^a
<i>One-chain-wide timber inventory not known to underestimate tree density at this time (Baker et al., 2018 [19])</i>						
S. California Sierra	Stephens et al. [73]	30.5	Only conifers	55	244 ^c	498 ^d

^a Estimate is calculated, as in the text here, as 1.6–2.3 times “Reported tree density”. ^b Estimate is calculated from direct tallies of trees by species in the land-survey records for the southern Sierra, which found that a mean of 22.4% of total trees were oaks; thus, conifer plus hardwood tree density is estimated as corrected tree density/0.776.

^c Stephens et al. [73] were unique in omitting data for conifers < 30.5 cm dbh. Baker and Hanson [17] reconducted the Stephens et al. inventory count of trees for their study area and found that for all conifers, tree density had a mean of 196–292 trees/ha for pine/ponderosa and mixed conifer, which are averaged here to be 244 trees/ha.

^d Estimate is calculated by the recorded percentages of total trees in the land surveys that were conifers and non-conifers in ponderosa pine (59.5%) and mixed-conifer forests (38.5%) in the area of the Stephens et al. [73] inventory, which averaged together equals a fraction of 0.49. The corrected tree density is thus divided by 0.49 to estimate conifer plus hardwood tree density. Note, 49% of non-conifer trees is high but not historically outside the HRV in the southern Sierra overall, where the third quartile of oaks as a percentage of all trees begins at 34.9% [28].

In other areas away from the lower elevations and driest areas, where low severity had fire rotations > 25 years (Figure 8), landscapes had greater proportions of moderate- to high-severity fire (Table 1) and were likely more frequently vulnerable to rapid change, as on the Uncompahgre Plateau, Colorado, after extensive severe fire in the late 1800s [32].

Regarding historical fire severity, (1) research that suggested climate forcing, not high-severity fires, led to pulses of tree regeneration did not separate these confounded variables, and their conclusions are invalid; (2) research from rarer old-growth forests, showing a lack of high-severity fires, is invalid evidence that other large parts of landscapes without old trees lacked severe fires; (3) early timber inventories, reported in H to show low-severity fires dominated, had omitted key documents showing evidence of high-severity fires; and (4) H and some of its authors claimed there was no evidence of historically severe fires in dry forests but omitted abundant published evidence of these fires, including by authors of H.

The very large body of evidence omitted in H included hundreds of quotes from early historical documents, many direct observations by land surveyors and observations by scientists in early forest-reserve reports, detailed mapping in early forest-atlases performed by the Forest Service, direct newspaper accounts, early oblique photographs, extensive analysis of early aerial photographs, ≥18 tree-ring reconstructions, 7 paleo-charcoal reconstructions, 15 land survey reconstructions (Table 1), and extensive reconstructions using modern forest inventory and analysis (FIA) age data. Of course, each source has limitations and warrants some critiques, but not omission. Nearly all available evidence that

found historically severe fires in dry forests was omitted in H. Omitted evidence clearly shows dry forests historically had infrequent moderate- to high-severity fires, rejecting the low-severity model and supporting the mixed-severity model.

Moreover, entire bodies of scientific evidence and rebuttal studies regarding time since fire and fire-severity trends were omitted in H. This created the false impression that long unburned forests experience higher fire severity and that high-severity fire is increasing, when, in fact, the strong weight of scientific evidence indicates that long unburned forests experience similar or lower fire severity and that high-severity fire rates are not outside HRV. This was also found in the three large studies H cited [56–58], as reviewed in Section 3, but these were misreported in H.

5. Conclusions and Implications of False and Omitted Evidence in H

5.1. All Evidence Not Supporting the Low-Severity Model Omitted in H, Leaving Falsified Science

H framed their review as an independent and objective critique of “dissent in the scientific literature” and an “incomplete assessment of the best available science” by providing “a framework for objectively assessing change” [3] (p. 3). This critique-of-dissent approach, however, quickly turned from objectivity and the best available science to false evidence and omission of evidence. As we have shown throughout, it is H that is an incomplete and biased assessment of the best available science due to extensive false and omitted evidence.

H’s model is that historical fires burned at low to moderate severity, with little to no high severity, but the primary evidence basis in H shows this to be false. The evidence for their model, in H Table 2, that they claimed shows recent high-severity fire is outside the HRV, instead was shown to be false based on evidence and quotes in the three key papers and largest study, omitted in H. These four studies show that high-severity fire was burning recently at rates near or below rates under the HRV.

In the rest of H, there is a pattern of consistent omission of evidence. Most evidence was omitted in H when it does not support their low-severity model, including 10 published rebuttals of their papers (Table 8) and 25 other very germane published papers (Table 9), as well as papers by the authors of H that do not support H’s model. To elucidate the extent of omission and misrepresentation in H, our review included (1) corrected replacement tables (Tables 3–6) that add the evidence omitted in H in their published tables, (2) summary tables that list all omitted rebuttals (Table 8) and omitted published studies with evidence that does not support their model (Table 9), and (3) extensive text detailing these omissions, including the omission of papers, by authors of H, that do not support H’s low-severity model. Together, this large body of evidence shows H’s evidence about their model, in both text and tables, is incorrect and rebutted in publications omitted in H.

We suggested this also may be a pattern, as it has occurred before. Earlier we showed in Baker et al.’s study [19], in a rebuttal omitted in H, that Hagmann et al. [18] cited 11 papers that they said pointed out “errors in methodology or misrepresentation of the work of others” [18] (p. 8). However, evidence for these alleged misrepresentations and errors was not presented at all in the paper. There also was no presentation of the evidence in 9 published studies that had specifically rebutted these 11 papers [19]. Additionally, earlier (Section 4.2.2), we mentioned that Fulé et al. [80] said there was no evidence of historical high-severity fires in dry forests, when this evidence was actually detailed in Williams and Baker’s study [24] itself, the paper Fulé et al. were critiquing. Finally, in the case of the early timber inventories, which we showed were known before the 1930s to underestimate tree density [17], Hagmann et al. [18] replied and we rebutted their reply [19]. However, as we explained earlier, in Section 4.1.2, Hagmann et al. [18] did not reject a central conclusion of [17] that early timber inventory data substantially underestimate tree density and require the use of 1.6–2.3 correction multipliers before reporting tree-density estimates. However, since the 2018 exchange, three new applications of early timber inventory data have appeared in studies by Stephens et al. [74], Bernal et al. [133], and North et al. [134] that include authors of Hagmann et al. [18], who knew about the original paper [17] and rebuttal [19]. Yet, none of these three new papers cited or discussed either

of these papers or made any of the necessary corrections to tree-density estimates (Table 7). Omission and false evidence are now a repeating pattern.

We want to be very clear that we focus here on evidence “omitted in H”, which simply describes the state of the evidence in H and the resulting state of the broader scientific record. We have no evidence about the authors’ intentions or motivations or how or why all this evidence was omitted in H. Nonetheless, the major false evidence and omission of evidence in H show that H is invalid and not replicable science. H leaves a falsified state of the scientific record regarding historical dry-forest structure and fires. The mixed-severity model, which is that dry forests had a heterogeneous structure and a mixture of fire severities, was not refuted in H and remains supported by the large body of scientific evidence, as shown in Tables 1–9 and the text, that was omitted in H. The low-severity fire model is rejected by this large body of false and omitted evidence in H.

The U.S. Office of Research Integrity (<https://ori.hhs.gov/investigations>, accessed on 10 January 2023) indicates that potential scientific misconduct is particularly important where there are significant public policy implications or where there may be more than one instance. We showed that there is a repeated pattern of omission in H, and that pattern of omission continues [74,133,134]. Omission of evidence and false evidence in H have significant land management implications, as millions of hectares of dry forests could be inappropriately managed in a futile and ecologically damaging attempt to prevent high-severity fires that are well documented in the omitted evidence to have occurred historically. The two accompanying public policy papers [22,23], which are substantially based on the false and omitted evidence in H, are thus shown here to not have a sound scientific basis.

Table 8. Published rebuttals, omitted in H, of ten publications used as evidence in H. Listed are the tables containing a summary of the omitted evidence, which refuted the rebutted articles and H’s conclusions. Details of these rebuttals are in the text.

Omitted Rebuttal	Article Rebutted	Rebuttal Summarized in Table(s)
Baker and Williams [34]	Levine et al. [61]	Tables 3 and 5
Baker and Williams [63]	Levine et al. [62]	Tables 3 and 5
Baker and Williams [63]	Johnston et al. [65]	Table 3
Baker et al. [19]	Hagmann et al. [18]	Tables 3 and 5
Hanson and Odion [78]	Collins et al. [77]	Table 3
Williams and Baker [79]	Fulé et al. [80]	Table 5
Odion et al. [95]	Stevens et al. [96]	Table 5
Odion and Hanson [117]	Safford et al. [116]	Table 6
Hanson et al. [120]	Spies et al. [118]	Table 6
Hanson and Odion [126]	Safford et al. [128]	Table 6

Table 9. Twenty-five original publications, with evidence of historically heterogeneous forest structure and mixed- to high-severity fires, omitted in H.

Omitted Evidence in These Sources	Evidence Omitted in H
Williams and Baker [40]	Omitted all evidence showing low bias and error in land-survey records.
Williams and Baker [41]	Omitted all evidence of validations of the WB method.
Williams and Baker [24]	Omitted all evidence of validations of the WB method and evidence of historically variable tree density and fire severity in dry forests in Arizona, Colorado, and Oregon.
Williams and Baker [31]	Omitted all direct evidence of extensive moderate- to high-severity fire in historical dry forests in the Colorado Front Range, evidence validating the WB method of reconstructing historical moderate- to high-severity fires, and evidence of very large high-severity fire patches (up to 8331 ha).
Baker and Williams [34]	Omitted all evidence of validations of the WB method and all evidence of historically variable tree density and fire severity documented in multiple historical sources cited in this paper.
Baker et al. [35]	Omitted all evidence from tree-ring reconstructions, forest-reserve reports, and other early scientific reports showing that historical dry forests in the Rocky Mountains had tree densities varying from 17 to 19,760 trees/ha.

Table 9. Cont.

Omitted Evidence in These Sources	Evidence Omitted in H
Baker [29]	Omitted quotes from early forest-reserve reports and other early scientific reports showing that historical dry forests in the eastern Cascades of Oregon had variable tree density as well as many direct reports of moderate- to high-severity fire.
Baker [28]	Omitted 47 quotes from early forest-reserve reports and other early scientific reports documenting that Sierran mixed-conifer forests were highly variable in tree density, but typically dense, and omitted numerous early reports of extensive moderate- to high-severity fire in historical Sierran mixed-conifer forests. Omitted 208 quotes from early forest reserve reports and other early scientific reports that documented historical moderate- to high-severity fires in Sierran mixed-conifer forests. Omitted evidence of high-severity fire patches commonly > 250 ha and up to 9400 ha in area.
Baker [25]	Omitted all evidence in this monograph analyzing why old CFI-based estimates of historical rates of fire are too short, why moderate- to high-severity fires were seldom found using these old methods, and how these old estimates can be corrected to accurately estimate fire history.
Farris et al. [15], Dugan and Baker [52]	Omitted any mention of the development of new landscape-scale methods of conducting fire history studies that overcome the limitations of earlier CFI-based fire history studies that H cite.
Hessburg et al. [11]	Omitted evidence of severe fires in northwestern dry forests; even though Hessburg is an author of H, and also authored this publication, H did not review its evidence. Hessburg et al. studied 303,156 ha in E. OR and E. WA and found “widespread evidence of partial stand and stand-replacing fire” [11] (p. 5) in mixed-conifer forests.
Baker [45]	Omitted evidence in six early photographs of the aftermath of severe fires in dry forests in the Rocky Mountains.
Baker [32,33,44]	Omitted evidence that documented that large moderate- to high-severity fires occurred in the late 1800s in dry forests on the Uncompahgre Plateau and in the San Juan Mountains, Colorado, based on forest atlases, land survey records, early photographs, early scientific publications, and other early records, including newspaper reports.
Pierce et al. [36], Pierce and Meyer [37], Long et al. [103], Fitch [104], Jenkins et al. [105], Bigio [106], Colombaroli and Gavin [107]	Omitted evidence in these paleo-charcoal studies from Arizona, Colorado, Idaho, New Mexico, and Oregon that infrequent moderate- to high-severity fires occurred historically in western USA dry forests.
DellaSala and Hanson [112,113]	Omitted evidence of numerous large historical high-severity fire patches in OR, WA, CA, and other parts of the western USA

5.2. After Correcting False and Omitted Evidence in H, What Are Key Findings?

First, high-severity fire rates in dry forests are within or lower than under the HRV, based on our correction of the evidence in H Table 2. The four studies with adequate samples [4,56–58] showed that recent fire rotations were within or longer (413, 608, 695, 875, 1045, and 1693 years) than the historical range of 217–849 years documented by Baker [4], based on land-survey reconstructions, paleo-charcoal studies, and reconstructions from early aerial photographs. The central premise of H that high-severity fires are burning at rates that exceed the HRV is refuted by H’s own evidence in their Table 2, as shown here in Section 3, and in the larger study [4] that H omitted. This evidence shows that there is no ecological need to reduce high-severity fire through fuel reduction; doing so successfully would likely have effects similar to fire suppression, which is widely understood to have deleterious ecological effects [7]. The primary fire restoration need is to just focus on restoring low- to moderate-severity fires, which is likely to also continue to include high-severity fires not expected to lead to rates exceeding historical rates. As Mallek et al. [58] explained, virtually all fires burning under moderate conditions are put out, so simply reducing fire suppression is likely to allow fires to burn closer to the HRV. Of course, it is sensible to periodically monitor rates of all severities of fires.

Second, the mixed-severity model of historical fires in dry forests is not rejected, and thus still supported, and the low-severity model is rejected by the evidence reviewed here, particularly based on newer landscape-scale evidence and newer methods, where more variability in historical fire has been found. H recognized that “... landscape-level assessments ... can address concerns that sampling bias of fire-scar studies favors detection of low-severity fire regimes” [3] (p. 8), but H did not review and use the available landscape-scale assessments. There are three major sources, at this point, that provide landscape-level assessments for historical wildfire: (1) tree-ring reconstructions at the landscape scale (e.g., Farris et al. [15], Sherriff et al. [16], and Dugan and Baker [52]), (2) land survey reconstructions (Table 1), and (3) early aerial photography [11]. This landscape-scale evidence of historical mixed-severity fires is also supported by multiple

other sources, including paleo-charcoal reconstructions, early newspaper accounts, early scientific publications, early forest-reserve reports, early forest atlases, and early oblique photographs (Table 2). Evidence in H omitted all these sources, as detailed in this paper. This evidence is sufficiently extensive and compelling, as reviewed in this paper and in earlier syntheses [12,13,34] to conclude that the low-severity model is rejected. The mixed-severity model is the best available science for the HRV of wildfire in all western USA dry forests.

Third, for forest structure, landscape-scale tree-ring and land-survey reconstructions have been shown through cross-validations to produce congruent findings, reviewed in this paper, that also strongly support the mixed-severity model and reject the low-severity model. Land survey reconstructions can cover larger land areas (Table 1) than can tree-ring reconstructions, but tree-ring reconstructions can provide more detailed evidence, if over a smaller area, as the base unit for land survey reconstructions is ~518 ha. However, tree-ring reconstructions are often biased toward older forests, which typically lack severe fires, by the need to have older trees to provide evidence from the historical period. Early aerial photographs can reconstruct categories of forest structure (e.g., old single-story forests vs. closed-canopy stem exclusion) but not the details of tree density and basal area [11]. The best source of landscape-scale historical forest structure may be from both landscape-scale tree-ring and land-survey reconstructions, possibly aided by early aerial photography, in conjunction with other sources (Table 2), which can provide certain evidence (e.g., maps and histories) that cannot otherwise be obtained.

The mixed-severity model is supported, and the low-severity model is rejected also for forest structure, which means dry forests had both low and high tree density and substantial heterogeneity in density from interaction with mixed-severity fires.

5.3. Implications and Outlook

It appears it is primarily the high-severity component of mixed-severity fires that concerns proponents of the rejected low-severity fire model. Are they forever hoping to retain the false image of low- to moderate-severity fires, in perpetual balance with old growth, which has now even been labeled “good fire” by some scientists [1]? Will they continue the fight to eliminate high-severity “bad fire”, using expensive mechanical fuel reductions and fire suppression even though, as shown here, high-severity fire was part of historical fire regimes in dry forests and is occurring at rates that recently are below or within the HRV? Will false evidence, omission of evidence, and misleading labeling of historical high-severity fires be ongoing? This study shows this is an unnecessary, futile, and false effort—Section 3 demonstrates that H’s own evidence shows no increase, at this point, relative to historical rates of high-severity fires. Moreover, the high-severity component of mixed-severity fires was and still is quite infrequent, with fire rotations from 175 to 2000 years (Table 1), allowing for considerable, if temporary, persistence of old, park-like dry forests. This finding is very consistent with Hessburg et al.’s [11] (p. 20) finding of contingent equilibrium in historical eastern Washington mixed-conifer forests: “Low-severity fires and equilibrium dynamics likely occurred in eastern Washington dry forests, where they fostered fire tolerant, park-like pine stands, however, these dynamics were perhaps ephemeral in nature, lasting one or more centuries at a location, and then switching concordant with regional climate forcing to non-equilibrium states.” We end by noting again that H did not even mention the findings of this large, key study, by the second author of H, that supports the mixed-severity fire model and rejects H’s low-severity fire model. H is documented here to have left behind a falsification of the scientific record, yet was used as a key basis for public land-management policy proposals [22,23] likely to have ecologically detrimental effects on dry forests [7].

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Having Your Voice Heard



JANUARY 2021

The Council on Environmental Quality (CEQ) revised this guide in January 2021 to reflect the updated National Environmental Policy Act (NEPA) Implementing Regulations that became effective on September 14, 2020. CEQ modernized and clarified the regulations to facilitate more efficient, effective, and timely NEPA reviews by Federal agencies. This guide provides an explanation of NEPA, how it is implemented, and how people outside the Federal Government—individual citizens, private sector applicants, members of organized groups, and representatives of Tribal, State, or local government agencies—can participate in the assessment of environmental impacts conducted by Federal agencies. This guide is informational, does not establish new requirements, and is not formal CEQ guidance. The contents of this document do not have the force and effect of law and are not meant to bind the public in any way. This document is intended only to provide clarity to the public regarding existing requirements under the law or agency policies.

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List of Acronyms

CE	Categorical Exclusion
CEQ	Council on Environmental Quality
CFR	Code of Federal Regulations
EA	Environmental Assessment
EIS	Environmental Impact Statement
EPA	U.S. Environmental Protection Agency
FONSI	Finding of No Significant Impact
NCECR	McCain Center for Environmental Conflict Resolution
NEPA	National Environmental Policy Act
NOI	Notice of Intent
ROD	Record of Decision
U.S.C.	United States Code

Purpose of the Guide

CEQ developed this guide to help citizens and organizations effectively participate in Federal agencies' environmental reviews under the National Environmental Policy Act (NEPA), which requires the consideration of environmental effects in Federal decision making.¹ With some limited exceptions, all Federal agencies in the executive branch have to comply with NEPA before they make final decisions about major Federal actions that could have environmental effects. The Federal Government takes hundreds of actions every day that may be subject to NEPA, including Federal construction projects, plans to manage and develop federally owned lands, and Federal approvals of non-Federal activities such as grants, licenses, and permits.

The environmental review process under NEPA provides an opportunity for citizens to get involved in a Federal agency's decision-making process. This guide will help you understand proposals for Federal actions, when to offer your thoughts on alternative ways for the agency to accomplish what it proposes, and how to offer your comments on the agency's analysis of the environmental effects of the proposed action and possible mitigation of potential harmful effects of such actions. NEPA requires Federal agencies to consider the effects of their actions on the environment, including interrelated social, cultural, and economic effects. Citizens often possess helpful information about the potential environmental, social, and economic effects that proposed Federal actions may have on people, places, and resources. NEPA's requirements provide you the opportunity to provide information to a Federal agency so it can take your input and unique perspective into account during the decision-making process.

History and Purpose of NEPA

Congress enacted NEPA in December 1969, and President Nixon signed it into law on January 1, 1970. NEPA established this country's national environmental policy and a process to implement it. Section 101 of NEPA declares that the national policy is "to use all practicable means and measures, including financial and technical assistance, in a manner calculated to foster and promote the general welfare, to create and maintain conditions under which man and nature can exist in productive harmony, and [to] fulfill the social, economic, and other requirements of present and future generations of Americans." 42 U.S.C. 4331(a).

What are the Procedural Requirements of NEPA?

Section 102 of NEPA contains procedures to ensure Federal agencies carry out the national policy of Section 101. These procedures require Federal agencies to engage in an environmental review process that integrates the consideration of the environment in Federal agency decision-making. NEPA also directs Federal agencies, to the fullest extent possible, to interpret and administer the policies, regulations, and public laws of the United States consistent with the policies set forth in NEPA.²

In NEPA, Congress recognized that the Federal Government's actions may cause significant environmental effects. Using the NEPA process, agencies must determine if their proposed actions will have significant environmental effects and consider the reasonably foreseeable environmental and related social and economic effects of their proposed actions that have a reasonably close causal relationship to the proposed actions. NEPA does not require particular

results or outcomes. Rather, NEPA encourages better decisions by requiring agencies to consider the environmental effects of their proposed actions in making their decisions. This environmental review process has two major purposes: ensuring that agencies consider the significant environmental consequences of their proposed actions and informing the public about their decision making.

NEPA also created the Council on Environmental Quality (CEQ). One of the responsibilities of CEQ is to consult with Federal agencies on procedures to implement NEPA's procedural requirements. In 1978, CEQ issued binding regulations directing agencies on the fundamental requirements necessary to fulfill their NEPA procedural obligations. CEQ updated these regulations in 2020 to facilitate more efficient, effective, and timely NEPA reviews by Federal agencies and to improve interagency coordination.³

Who is Responsible for Implementing NEPA?

NEPA's procedural requirements apply to all Federal agencies in the executive branch and some Federal boards, commissions, independent agencies, and committees. NEPA does not apply to the President, to Congress, or to the Federal courts.⁴

Because NEPA implementation is an important responsibility of the Federal Government, many Federal agencies have established offices dedicated to NEPA policy and program oversight. Employees in these offices prepare NEPA guidance, policy, and procedures for the agency, and often make this information available to the public through the Internet. A "senior agency official" oversees the agency's overall compliance with NEPA and resolves any implementation issues that may arise, including those related to agency timelines and schedules for environmental reviews.⁵ Federal agencies must develop their own capacity within a NEPA program in order to develop analyses and documents (or review those prepared by others) to ensure informed decision making.⁶ Most agency NEPA procedures are available online at [NEPA.gov](https://www.nepa.gov) or on individual agency websites, which agencies are required to maintain to allow agencies and the public to efficiently and effectively access information about NEPA reviews.⁷ Agency NEPA procedures also are published in the *Federal Register* for public review and comment when first proposed and some are later codified and published in the Code of Federal Regulations.⁸ If you experience difficulty locating an agency's NEPA procedures, you can contact the agency NEPA point of contact and ask for a copy of their procedures.⁹

To What Do the Procedural Requirements of NEPA Apply?

NEPA's procedural requirements apply to a Federal agency's decisions on proposed actions, including providing permits for private actions; financing, assisting, conducting, or approving projects or programs; issuing agency rules, regulations, plans, policies, or procedures; making Federal land management decisions; and an agency's legislative proposals.¹⁰ NEPA applies when a Federal agency has discretion to choose among one or more alternative means of accomplishing a particular goal.¹¹

Frequently, private individuals or companies will become involved in the NEPA process when they need a permit issued by a Federal agency. When a company applies for a permit (for example, for crossing Federal lands or impacting waters of the United States), the agency that is

being asked to issue the permit must evaluate the reasonably foreseeable environmental effects of the permit decision that have a reasonably close causal relationship to the agency decision. Federal agencies might require the private company or developer to pay for the preparation of analyses, but the agency remains responsible for the scope and accuracy of the analysis.

When Does NEPA Apply?

NEPA requires agency decision makers to make informed decisions. Therefore, the NEPA process must be completed before an agency makes a final decision on a proposed action. As a threshold matter, agencies start the NEPA process early by evaluating in their agency NEPA procedures the extent to which a proposed action requires environmental analysis.¹² NEPA does not require the decision maker to select the environmentally preferable alternative or prohibit adverse environmental effects. Indeed, decision makers in Federal agencies often must take into account other concerns and policy considerations in the decision-making process, such as social, economic, technical or national security interests. But NEPA does require that decision makers be informed of the environmental consequences of their decisions.

Federal agencies also can use the NEPA process to comply with other environmental requirements like the Endangered Species Act, the National Historic Preservation Act, the Environmental Justice Executive Order, and other Federal, State, Tribal, and local laws and regulations.¹³ Agencies often coordinate to conduct these other environmental reviews concurrently to increase efficiency and avoid duplication.¹⁴

Who Oversees the NEPA Process?

There are two Federal agencies that have particular responsibilities relating to NEPA. CEQ has primary responsibility for overseeing implementation of NEPA by Federal agencies. Congress placed CEQ in the Executive Office of the President and gave it many responsibilities, including the responsibility to ensure that Federal agencies meet their obligations under the Act. CEQ oversees implementation of NEPA, principally through issuance and interpretation of NEPA regulations that implement the procedural requirements of NEPA. CEQ also reviews and approves Federal agency NEPA procedures, approves alternative arrangements for compliance with NEPA in the case of emergencies, and helps to resolve disputes between Federal agencies and with other governmental entities and members of the public.

The CEQ regulations set forth requirements for agencies and call for agencies to update their own implementing procedures that implement these requirements based on each agency's specific mandates, obligations, and missions.¹⁵ These agency-specific NEPA procedures account for the slight differences in agencies' NEPA processes.

The Environmental Protection Agency's (EPA's) Office of Federal Activities also conducts NEPA oversight as it reviews environmental impact statements (EISs) and some environmental assessments (EAs) issued by Federal agencies.¹⁶ It provides its comments to the public by publishing summaries of them in the *Federal Register*, a daily publication that provides notice of Federal agency actions. Appendix B has information on the *Federal Register*. EPA's reviews are intended to assist Federal agencies in improving their NEPA analyses and decisions.¹⁷

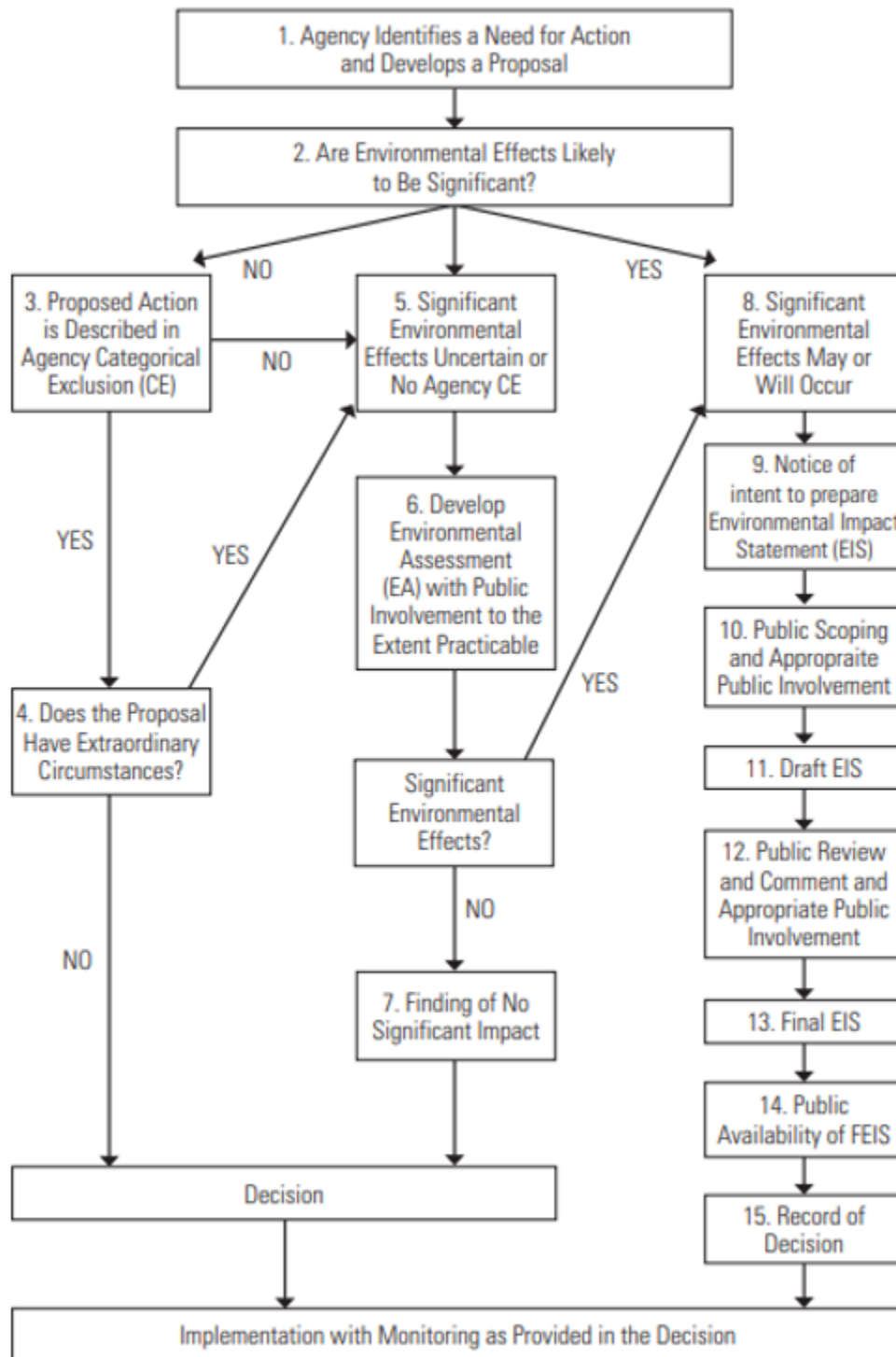
In addition to CEQ's and EPA's oversight, other agencies also may assist in the NEPA process, particularly in issue resolution (for example, the McCain Center for Environmental Conflict Resolution (NCECR) and Federal Permitting Improvement Steering Council).

Navigating the NEPA Process

Each year, Federal agencies prepare thousands of EAs and hundreds of EISs. These documents provide citizens and communities with an opportunity to learn about and be involved in the agencies' environmental reviews that are part of the Federal agency decision-making process. It is important to understand that commenting on a proposal is not a "vote" on whether the proposed action should take place. Nonetheless, the information you provide during the EA and EIS process can influence the decision makers and their final decisions because NEPA requires that Federal decision makers be informed of the environmental consequences of their decisions.

This guide will help you better navigate the NEPA process and better understand the roles of the various other actors. While reading the guide, please refer to the flowchart, "The NEPA Process," in Figure 1, which details the steps of the NEPA process. For ease of reference, each step of the process is designated with a number that is highlighted in the text discussing that particular step. While agencies may differ slightly in how they comply with NEPA, understanding the basics will give you the information you need to work effectively with any agency's process.

The NEPA Process (Figure 1)



* Significant new circumstances or information relevant to environmental concerns or substantial changes in the proposed action that are relevant to environmental concerns may necessitate preparation of a supplemental EIS following either the draft or final EIS, or the Record of Decision. 40 CFR 1502.9(d).

The NEPA process begins when an agency develops a proposal to address a need to take an action. The need to take an action may be something the agency identifies itself, or it may be a need to make a decision on a proposal brought to it by someone outside of the agency, for example, an applicant for a permit. Based on the need, the agency develops a proposal for action (Number 1 in Figure 1). If it is the only Federal agency involved, that agency will automatically be the “lead agency,” which means it has the primary responsibility for compliance with NEPA.

Some large or complex proposals involve multiple Federal agencies along with State, Tribal, and local agencies. If another Federal, State, Tribal, or local agency has a major role in the proposed action and also has NEPA responsibilities or responsibilities under a similar NEPA-like law,¹⁸ that agency may be a “joint lead agency.” A “joint lead agency” shares the lead agency’s responsibility for management of the NEPA process, including public involvement and the preparation of documents.

Other Federal, State, Tribal, or local government agencies may have a decision or special expertise regarding a proposed action, but less of a role than the lead agency. In that case, such a Federal, State, Tribal, or local government agency may be a “cooperating agency.” A “cooperating agency” is an agency that has jurisdiction by law or special expertise with respect to any environmental impact involved in a proposal (or a reasonable alternative). Thus, a “cooperating agency” typically will have some responsibilities for the analysis related to its jurisdiction or special expertise.

Special Situations

- Congress may exempt an action from NEPA.
- If the agency needs to take an action that would typically require preparation of an EIS in response to an emergency, and there is insufficient time to follow the regular NEPA process, then the agency can proceed immediately to mitigate harm to life, property, or important resources, and work with CEQ to develop alternative arrangements for compliance with NEPA (40 CFR 1506.12). The NEPA analyses and document may involve classified information. If the entire action is classified, the agency will still comply with the analytical requirements of NEPA, but the information will not be released for public review. If only a portion of the information is classified, the agency will organize the classified material so that the unclassified portions can be made available for review (40 CFR 1507.3(f)).

Implementing the NEPA Process

The CEQ NEPA regulations establish three levels of review for Federal agencies to assess proposals for agency action: a categorical exclusion (CE), an EA, or an EIS. Once it has developed a proposed action, the agency will determine which level of NEPA review the agency will pursue. Agencies may review expeditiously those actions that normally do not have

significant effects by using CEs or, for actions that are not likely to have significant effects, by preparing EAs. By using CEs and EAs whenever appropriate, agencies then can focus their limited resources on those actions that are likely to have significant effects and require an EIS.

Categorical Exclusions (CEs) (Number 3 in Figure 1)

A CE is a category of actions that the agency has determined does not normally have a significant effect on the human environment.¹⁹ Examples include issuing administrative personnel procedures, making minor facility renovations (such as installing energy-efficient lighting), and reconstruction of hiking trails on public lands. Agencies develop a list of CEs specific to their operations when they develop or revise their NEPA implementing procedures in accordance with CEQ's NEPA regulations.²⁰

A CE is based on an agency's experience with a particular kind of action and its environmental effects. The agency may have studied the action in previous EAs, found no significant impact on the environment based on the analyses, and validated the lack of significant impacts after the implementation. If this is a type of action that will be repeated over time, the agency may decide to amend their implementing regulations to include the action as a CE. In these cases, the draft agency procedures are published in the *Federal Register*, and a public comment period is required. Members of the public may comment on draft agency procedures that are proposing new CEs or amending existing CEs to ensure the agency takes into consideration relevant information and views.

An agency may comply with NEPA by determining that a CE applies to a proposed action and verifying that no extraordinary circumstances exist that may cause the proposed action to have a significant effect. Extraordinary circumstances typically include such matters as effects to endangered species, protected cultural sites, and wetlands (Number 4 in Figure 1). If there are no extraordinary circumstances indicating that the effects of the action may be significant, or there are circumstances that lessen the impacts or other conditions sufficient to avoid significant effects, then the agency can proceed with the action.

If the proposed action is not included in the description provided in the CE established by the agency, or there are extraordinary circumstances, the agency must prepare an EA or an EIS, or develop a new proposal that may qualify for application of a CE. When the agency does not know or is uncertain whether significant impacts are expected, the agency should prepare an EA to determine if there are significant environmental effects.

Environmental Assessments (EA) (Number 6 in Figure 1)

The purpose of an EA is to determine the significance of the potential environmental effects of a proposed Federal action and to look at alternative means to achieve the agency's objectives. The EA is a concise public document to aid an agency's compliance with NEPA and support its determination whether to prepare an EIS (Number 6 in Figure 1) or a finding of no significant impact (FONSI) (Number 7 in Figure 1).²¹

Agencies must complete EAs within one year of the agency decision to prepare an EA unless a senior agency official of the lead agency approves a longer period in writing and establishes a new time limit.²²

An EA should include brief discussions of:

- The purpose and need for the proposal;
- Alternative courses of action for any proposal that involves unresolved conflicts concerning alternative uses of available resources;
- The environmental impacts of the proposed action and alternatives; and
- A listing of agencies and persons consulted.²³

Because the EA serves to evaluate the significance of a proposal for agency action, it should focus on the potentially affected environment and degree of the effects of the action.²⁴ Often the EA will identify ways in which the agency can revise the action to minimize environmental effects.

When preparing an EA, the agency has discretion as to the level of public involvement (Number 6 in Figure 1). The CEQ regulations state that the agency must involve, to the extent practicable, the public, State, Tribal, and local governments, other relevant agencies, and applicants in preparing EAs.²⁵ Sometimes agencies will choose to use the scoping and public comment periods that are found in the EIS process. In other situations, agencies make the EA and a draft FONSI available to interested members of the public.

Some agencies require that interested parties be notified of the decision to prepare an EA. Some agencies keep a notification list of parties interested in a particular kind of action or in all agency actions. Other agencies simply prepare the EA. It is important that you read the specific NEPA procedures of the proposing agency or ask the local NEPA point of contact working on the project about the process and let the appropriate agency representative know if you are interested in being notified of all NEPA documents or NEPA processes related to a particular type of action.

The EA process concludes with either a FONSI (Number 7 in Figure 1) or a determination to proceed to preparation of an EIS. A FONSI is a document that presents the reasons why the agency has concluded that there are no significant environmental impacts projected to occur upon implementation of the action.²⁶ The FONSI either includes the EA or incorporates the EA by reference.

In two circumstances, the CEQ NEPA regulations require agencies to make the proposed FONSI available for public review for 30 days.²⁷ Those situations are:

- If the type of proposed action has not been done before by the particular agency, or
- If the action is something that typically would require an EIS under the agency NEPA procedures.

If this is the case, the agency usually publishes a notice of availability of the FONSI with information on how and where to provide your comments. The agency may post it on its website, publish it in local newspapers, publish it in the *Federal Register*, or make available in some other manner. If you are interested in a particular action that is the subject of an EA, you should find out from the agency how it will make the FONSI available.

Environmental Impact Statements (EIS) (Number 8 in Figure 1)

A Federal agency must prepare an EIS if it is proposing a major Federal action significantly affecting the quality of the human environment.²⁸ The regulatory requirements for an EIS are more detailed than the requirements for an EA or a CE. The EIS process consists of four main stages, which are explained below: scoping with a public notice of intent (NOI) to prepare an EIS, the draft EIS and public comment period, the final EIS, and the record of decision (ROD).

To the extent practicable, if a proposal will require action by more than one Federal agency, the lead and cooperating agencies will evaluate the proposal in a single EIS and issue a joint ROD. Agencies must complete EISs within two years from the date of the NOI unless a senior agency official of the lead agency approves a longer period in writing and establishes a new time limit.²⁹

Scoping and Public Notice of Intent (Number 10 in Figure 1)

When a proposed action is sufficiently developed for agency consideration, the agency may begin the process of determining the scope of issues for analysis in an EIS. Scoping generally involves identifying significant issues, eliminating non-significant issues from further study, and determining the range of actions, alternatives, and impacts to be considered by the EIS.³⁰

A cornerstone of the scoping process is the publication of a NOI to prepare an EIS in the *Federal Register*, which provides information on the proposed action (Number 10 in Figure 1).³¹ The lead agency publishes the NOI as soon as practicable after the agency determines that the proposal is sufficiently developed to allow for meaningful public comment on alternatives, information, and issues for analysis in the EIS. The NOI briefly summarizes the proposal, including the purpose and need, expected impacts, and possible alternatives. Under the updated CEQ regulations, agencies must request in the NOI public comment specifically on potential alternatives, information, and analyses relevant to the proposed action. The NOI also provides a schedule for the decision-making process including anticipated permits and other authorizations, and describes the agency's proposed scoping process, including any meetings and how the public can get involved. The NOI also contains an agency point of contact who can answer questions about the proposed action and the NEPA process. Scoping also may include pre-application communication with potential cooperating agencies, an applicant, and survey work conducted before or after the publication of the NOI. The scoping process is the best time to identify issues, determine points of contact, establish project schedules, and provide recommendations to the agency. The overall goal is to define the scope of issues to be addressed in depth in the analyses that will be included in the EIS. Specifically, the scoping process will:

- Identify the significant issues to be analyzed in the EIS and eliminate from detailed study non-significant issues;
- Identify people or organizations who are interested in the proposed action and invite them to participate;
- Determine the roles and responsibilities of lead and cooperating agencies;
- Identify any related EAs or EISs;
- Identify gaps in data and informational needs;
- Identify other environmental review and consultation requirements so they can be integrated with the EIS; and

- Indicate the relationship between the development of the environmental analysis and the agency's tentative decision-making schedule.³²

As part of the process, agencies must identify and invite the participation of interested persons. The agency should choose whatever communications methods are best for effective involvement of communities, whether local, regional, or national, that are interested in the proposed action, and the agency must consider the ability of affected persons to access electronic media. Video conferencing, public meetings, conference calls, formal hearings, or informal workshops are among the legitimate ways to conduct scoping. It is in your interest to become involved as soon as the EIS process begins and to use the scoping opportunity to make thoughtful, rational presentations on impacts and alternatives. Some of the most constructive and beneficial interaction between the public and an agency occurs when citizens identify or develop reasonable alternatives that the agency can evaluate in the EIS.

NEPA is About People and Places

Draft EIS (Number 11 in Figure 1)

The next major step in the EIS process that provides an opportunity for your input is when the agency publishes a draft EIS for public comment. The agency publishes its EIS on an agency website and the EPA publishes a Notice of Availability in the *Federal Register* informing you and other members of the public that the draft is available for comment (Number 11 in Figure 1). Based on the communication plan established by the agency, websites, local papers, or other means of public notice may also be used. The comment period is at least 45 days long. During this time, the agency may conduct webinars, public meetings, or hearings as a way to solicit comments.³³ The agency will also request comments from other Federal, State, Tribal, and local agencies that may have jurisdiction or interest in the matter.

One key aspect of a draft EIS is the statement of the underlying purpose and need.³⁴ Agencies draft a "Purpose and Need" statement to describe what they are trying to achieve by proposing an action. The purpose and need statement explains to the reader why an agency action is necessary, and serves as the basis for identifying the reasonable alternatives that meet the purpose and need.

Another fundamental part of the draft EIS is the identification and evaluation of alternative ways of meeting the purpose and need of the proposed action. The lead agency or agencies must, "evaluate reasonable alternatives, and for alternatives that were eliminated from detailed study, briefly discuss the reasons for their elimination."³⁵ Reasonable alternatives are those that are technically and economically feasible, meet the proposal's purpose and need, and, where applicable, meet the goals of the applicant.³⁶ If the agency is considering an application for a permit or other Federal approval, the agency must still consider all reasonable alternatives. Reasonable alternatives include those that are practical or feasible from the technical and economic standpoint, rather than simply desirable from the standpoint of the applicant. Agencies are obligated to evaluate a reasonable range of feasible alternatives in enough detail so that a reader can compare and contrast the environmental effects of the various alternatives.

If an agency has a preferred alternative when it publishes a draft EIS, the draft must identify which alternative the agency prefers. All agencies must identify a preferred alternative in the final EIS, unless another law prohibits it from doing so.³⁷

Agencies must always describe and analyze a “no action” alternative. The “no action” alternative is simply what would happen if the agency did not act upon the proposal for agency action. For example, in the case of an application to the U.S. Army Corps of Engineers for a permit to place fill material from a dredging project in a particular area, the “no action” alternative is no permit. But in the case of a proposed new management plan for the National Park Service’s management of a national park, the “no action” alternative is the continuation of the current management plan. The “no action” alternative describes reasonably foreseeable environmental trends or planned actions in the area that would be affected by the proposed action.³⁸

Definition of Effects

CEQ NEPA Regulation, 40 CFR 1508.1(g)

Effects or impacts means changes to the human environment from the proposed action or alternatives that are reasonably foreseeable and have a reasonably close causal relationship to the proposed action or alternatives, including those effects that occur at the same time and place as the proposed action or alternatives and may include effects that are later in time or farther removed in distance from the proposed action or alternatives.

- (1) Effects include ecological (such as the effects on natural resources and on the components, structures, and functioning of affected ecosystems), aesthetic, historic, cultural, economic (such as the effects on employment), social, or health effects. Effects may also include those resulting from actions that may have both beneficial and detrimental effects, even if on balance the agency believes that the effect will be beneficial.
- (2) A “but for” causal relationship is insufficient to make an agency responsible for a particular effect under NEPA. Effects should generally not be considered if they are remote in time, geographically remote, or the product of a lengthy causal chain. Effects do not include those effects that the agency has no ability to prevent due to its limited statutory authority or would occur regardless of the proposed action.
- (3) An agency’s analysis of effects shall be consistent with this paragraph (g). Cumulative impact, defined in 40 CFR 1508.7 (1978), is repealed.

The environmental consequences section discusses the effects of the proposed action, no action, and reasonable alternatives. It also forms the scientific and analytic basis for the comparisons of the proposed action and reasonable alternatives made under the alternatives section. For purposes of NEPA, “effects” and “impacts” mean the same thing—changes to the human

environment from the proposed action or alternatives that are reasonably foreseeable and have a reasonably close causal relationship to the proposed action or alternatives. This includes those effects that occur at the same time and place as the proposed action or alternatives and may include effects that occur later or are farther removed in distance from the proposed action or alternatives.³⁹ Impacts include ecological, aesthetic, historic, cultural, economic, social, or health impacts, whether adverse or beneficial.⁴⁰ It is important to note that human beings are part of the environment (indeed, that is why Congress used the phrase “human environment” in NEPA), so when an EIS is prepared and economic or social and natural or physical environmental effects are interrelated, the EIS should discuss all of these effects.⁴¹

In addition to the environmental impacts of the proposed action and alternatives, the environmental consequences section will discuss:

- Any potential unavoidable adverse environmental effects;
- The relationship between short-term uses of man’s environment and the maintenance and enhancement of long-term productivity;
- Any potential irreversible or irretrievable commitments of resources;
- Possible conflicts with land use plans, policies, and controls for the area;
- Energy and natural or depletable resource requirements and conservation potential of alternatives and mitigation measures;
- Urban quality, historic and cultural resources, and the design of the built environment, including the reuse and conservation potential of various alternatives and mitigation measures;
- Mitigation of adverse environmental impacts; and
- Applicable economic and technical considerations, including the economic benefits of the proposed action.

The draft EIS will also contain a summary of alternatives, information, and analysis submitted by commenters during the scoping process.⁴² The agency will specifically invite comment on this summary.

The EIS also will have a list of the individuals who prepared the document and their qualifications⁴³ and a table of contents.⁴⁴ The agency may choose to append the EIS with additional material relevant to the decision, including material prepared in connection with the EIS or that substantiates its analysis.⁴⁵

Final EIS (Number 13 in Figure 1)

When the public comment period is finished, the agency analyzes comments, conducts further analysis as necessary, and prepares the final EIS. The agency may respond to individual comments or groups of comments by making changes to the proposed action or alternatives, developing new alternatives, modifying its analyses, making factual corrections, or explaining why a comment does not require the agency’s response.⁴⁶ Often the agency will meet with other agencies that may be affected by the proposed action in an effort to resolve an issue or mitigate project effects. The final EIS also will include a summary that identifies all relevant alternatives, information, and analyses submitted by commenters for consideration by the lead and cooperating agencies.⁴⁷

When it is ready, the agency will publish the final EIS and EPA will publish a Notice of Availability in the *Federal Register*. The Notice of Availability can mark the start of a waiting period (Number 14 in Figure 1), during which a minimum of 30 days must pass before the agency can make a decision on its proposed action, unless the agency couples the 30 days with a formal internal appeals process or is authorized to issue a combined final EIS and ROD.⁴⁸ A waiting period provides time for the agency decision maker to consider public comments, the purpose and need for agency action, weigh the alternatives, balance the objectives and policy considerations, and make a decision.

There is an additional (but rarely used) procedure worth noting: pre-decisional referrals to CEQ.⁴⁹ This referral process takes place when EPA or another Federal agency determines that proceeding with the proposed action is environmentally unacceptable. If an agency reaches that conclusion, the agency can refer the issue to CEQ within 25 days after the Notice of Availability for the final EIS is issued. CEQ then works to resolve the issue with the agencies concerned. CEQ might also refer the agencies to the NCECR to try to address the matter before formal elevation.⁵⁰ There is no provision for citizens to formally refer an action to CEQ; however, CEQ typically provides an opportunity for public involvement in a referral.

Record of Decision (ROD) (Number 15 in Figure 1)

The ROD is the final step for agencies in the EIS process. The ROD is a document that states what the decision is; identifies the alternatives considered, including the environmentally preferred alternative; and discusses mitigation plans, including any enforcement and monitoring commitments.⁵¹ In the ROD, the agency discusses all the factors, including any considerations of national policy that were contemplated when it reached its decision on whether to, and if so how to, proceed with the proposed action. The ROD will also discuss if all practical means to avoid or minimize environmental harm have been adopted, and if not, why they were not. The ROD will summarize any monitoring and enforcement program that it has adopted for any enforceable mitigation requirements or commitments.⁵² The ROD also will contain a certification by the decision maker that, in developing the EIS, the agency has considered all of the alternatives, information, analysis, and objections submitted by State, Tribal, and local governments and public commenters.⁵³ The ROD is a publicly available document. Sometimes RODs are published in the *Federal Register* or on the agency's website, but if you are interested in receiving the ROD, you should ask the agency's point of contact for the EIS how to obtain a copy of the ROD.

Supplemental EIS

Sometimes a Federal agency is obligated to prepare a supplement to an existing EIS. An agency must prepare a supplement to either a draft or final EIS if the proposed action has not been completed and the agency makes substantial changes in the proposed action that are relevant to environmental concerns or there are significant new circumstances or information relevant to environmental concerns and bearing on the proposed action or its impacts. An agency may also prepare a supplemental EIS if it determines that doing so will further the purposes of NEPA.⁵⁴ An agency prepares a supplemental EIS in the same way as a draft or final EIS, except that scoping is not required. If a supplement is prepared following a draft EIS, the final EIS will address both the draft EIS and supplemental EIS. An agency may find that substantial changes

in a proposed action or new circumstances or information do not result in significant environmental concerns. In such cases, the agency will document the finding consistent with its procedures, or, if necessary, in a FONSI supported by an EA.

EPA's Review

EPA plays a critical role in other agencies' NEPA processes. EPA must review and provide comments on the adequacy of the analysis and the impact to the environment.⁵⁵ EPA must refer a matter to CEQ if it determines that the action is environmentally unsatisfactory.

The Office of Federal Activities in EPA is the official recipient of all EISs prepared by Federal agencies, and publishes the notices of availability in the *Federal Register* for all draft, final, and supplemental EISs. The publication of these notices start the official clock for public review and comment periods and wait periods.⁵⁶ In addition to the *Federal Register*, EISs are available in the EIS database at <https://www.epa.gov/nepa>.

When and How to Get Involved

It Depends on the Agency

To determine the specific steps in the process where public involvement will be the most effective, it is very important to review the agency's NEPA procedures and the agency's NEPA website.⁵⁷ As previously mentioned, NEPA procedures may differ among agencies.

In addition, new legislation and presidential directives can change the way NEPA is implemented in agencies. Congress has enacted a number of statutes to improve coordination among agencies, integrate NEPA with other environmental reviews, and bring more transparency to the NEPA process. Presidents also have directed agencies, through Executive orders and Presidential memoranda, to undertake various initiatives that improve the timeliness and efficiency of the NEPA process.

Infrastructure Projects under FAST-41

In 2015, Congress enacted Title 41 of the Fast Act (FAST-41) to provide for a more efficient environmental review and permitting process for "covered projects."⁵⁸ These are projects that require Federal environmental review under NEPA, are expected to exceed \$200 million, and involve the construction of infrastructure for renewable or conventional energy production, electricity transmission, water resource projects, broadband, pipelines, manufacturing, and other sectors.

FAST-41 created the Federal Permitting Improvement Steering Council (FPISC or Permitting Council), composed of agency Deputy Secretary-level members and chaired by an Executive Director appointed by the President. FAST-41 establishes new procedures that standardize interagency consultation and coordination practices. FAST-41 also codifies into law the use of a Permitting Dashboard to track project timelines (www.permits.performance.gov). The Permitting Dashboard is an online tool for Federal agencies, project developers, and interested members of the public to track the Federal Government's environmental review and

authorization processes for large or complex infrastructure projects. Project sponsor participation in FAST-41 is voluntarily.

FAST-41 codified certain roles and responsibilities required by the NEPA regulations, such as the concepts of lead and cooperating agencies, and the different levels of NEPA analysis—EISs, EAs, and CEs—and the requirement for CEQ to resolve any dispute over designation of a facilitating or lead agency for a covered project.⁵⁹ Additionally, Congress addressed interagency coordination on key aspects of the NEPA process, including scoping, identification of the range of reasonable alternatives for study in an EIS, and the public comment process. Finally, Congress established a two-year statute of limitations for covered projects.⁶⁰ The Permitting Council has more resources on FAST-41 posted on the Permitting Dashboard.

Be Informed of Actions

Sometimes citizens generally are interested in actions taking place in a particular area (for example, in your community or in an ecosystem or a facility that affects you). If this is the case, you can inform the appropriate agency or agencies that you would like to be notified of any proposed action or any environmental impact analysis that might be prepared in that area. In addition, CEQ now requires agencies to have websites where they post environmental documents, relevant notices, and other relevant information for use by interested persons.⁶¹

Active Involvement

Being active in the NEPA process requires you to dedicate some of your resources to the effort. Environmental impact analyses can be technical and lengthy. Agencies can be expected to provide general responses to general comments on a NEPA document, so active involvement in the NEPA process requires a commitment of time and a willingness to share information with the decision-making agency and other citizens. For example, during the scoping process for an EIS, you are encouraged to identify alternatives, information, and analyses relevant to the proposed action for consideration by the agency.⁶² The agency will summarize that information in the draft EIS and invite further comment on that information.⁶³ However, you must submit your comments during the comment periods in order for the agency to consider the information and to ensure informed decision making.⁶⁴

You may participate as an individual, get involved by working with other interested individuals or organizations, or by working through your State, Tribal, or local government. For example, if an agency is taking an action for which your State, Tribal, or local government has special expertise or approval authority, the appropriate State, Tribal, or local agency can become a “cooperating agency” with the Federal agency.⁶⁵ This formal status does not increase their role in decision making, but it does allow the governments to use their knowledge and authorities to help shape the Federal decision-making process.

Another way to participate is to check with local experts such as biologists or economists at a university to assist with your review of the NEPA analyses and documents. You can also form study groups to review environmental impact analyses and enlist experts to review your comments on the documents.

Your involvement in the NEPA process does not have to be confined to commenting on the analysis. If the agency adopts monitoring and mitigation in the ROD, upon request, it must make available to the public the results of relevant monitoring.⁶⁶ Upon request, it also must inform cooperating or participating agencies on progress in carrying out mitigation measures that they have proposed and that were adopted by the agency making the decision.⁶⁷ Community groups also can be involved in monitoring.

In summary, there are several opportunities to get involved in the NEPA process:

- When the agency prepares its NEPA procedure;
- Prior to and during preparation of a NEPA analysis;
- When a NEPA document is published for public review and comment;
- When a final decision is pending before the agency decision-maker; and
- When monitoring the implementation of the proposed action and the effectiveness of any associated mitigation.

Other Processes that Require Public Involvement

When a proposed action is part of a permitting process, the statute or regulations for that permitting process also may provide opportunities to comment in addition to the NEPA public involvement opportunities discussed above. For example, most Federal agency land use planning regulations require public involvement. While this guide does not explore all of those additional possibilities for comment, the NEPA team working on a particular proposal will be familiar with the various comment periods and will be able to inform you of those opportunities. Note that the permitting and NEPA processes should be integrated or run concurrently in order to have an effective and efficient decision-making process.

Public Comment Periods

Agencies must make diligent efforts to involve the public in development and implementation of their NEPA procedures.¹ In requesting comments on a draft EIS, Federal agencies must affirmatively solicit comments in a manner designed to inform those persons or organizations who may be interested in or affected by the proposed action.¹ Citizens involved in the process should ensure that they know how agencies will inform the public that an action is proposed and the NEPA process is beginning (via the *Federal Register*, websites, newspapers, direct mailing, etc.); that certain documents are available; and that preliminary determinations have been made on the possible environmental effects of the proposal (e.g., what level of analysis the agency will initially undertake).

Agencies solicit different levels of involvement when they prepare an EA versus an EIS. In preparing an EIS, agencies must invite the identification of alternatives, information, and analyses relevant to the proposed action during the scoping process. Agencies must summarize that information in the draft EIS and have a 45-day comment period after the draft EIS is made available. In the case of an agency preparing an EA, the CEQ regulations require the agency to involve the public to the extent practicable, but each agency has its own guidelines about how to involve the public for EAs. In any case, citizens are entitled to receive “environmental documents,” such as EAs, involved in the NEPA process.¹

In terms of a specific agency, required public comment periods associated with an EA or an EIS can be found in its NEPA procedures. An agency may grant requests to extend the comment period to ensure enough time for the public and other agencies to review and comment.

Citizens who want to raise issues with the agency should do so as specifically as possible and at the earliest possible stage in the process. Agencies are much more likely to evaluate a new alternative or address a concern if it is raised in a clear and timely manner.

How to Comment

Comments may be the most important contribution from citizens because they promote informed decision making. Comments should provide sufficient detail for the agency to understand the commenter’s position and why the issues raised are important to the decision. Accordingly, comments should be clear, concise, relevant to the analysis of the proposed action, and submitted during the public comment periods. Take the time to organize thoughts and edit the document submitted.⁶⁸ As a general rule, the tone of the comments should be polite and respectful. Those reviewing comments are public servants tasked with a job, and they deserve the same respect and professional treatment that you and other citizens expect in return. Comments that are solution-oriented and provide specific examples will be more effective than those that simply oppose the proposed project. Comments that contribute to developing alternatives that address the purpose and need for the action also are effective. Agencies must invite the submission of alternatives

during the scoping process to facilitate timely submission of comments that contribute to developing alternatives.

Commenting is not a form of “voting” on an alternative. The number of negative comments an agency receives does not prevent an action from moving forward. Agencies typically respond collectively to numerous comments that repeat the same basic message of support or opposition. In addition, general comments that state an action will have “significant environmental effects” will not help an agency make a better decision unless the comment explains the relevant causes and environmental effects. If you think the proposed action will have a significant environmental effect, explain why the issues you raise are significant to the consideration of potential environmental impacts and alternatives to the proposed action. In drafting comments, try to focus on the purpose and need of the proposed action, the proposed alternatives, the assessment of the environmental impacts of those alternatives, and the proposed mitigation.

Finally, remember that decision makers also receive other information and data, such as operational and technical information related to implementing an action, which they will have to consider when making a final decision.

What If Involvement Is Not Going Well?

For the purposes of this discussion, “not going well” means that you or your organization believes that the lead agency is not giving the public sufficient opportunity to get involved or is not using that involvement effectively. Perhaps you think that the agency should hold a public meeting. Or you or your community or group has developed an alternative that you think meets the purpose and need of the proposed action and reflects the policies set forth in NEPA. Maybe you want an extension of the comment period because the document’s appendix is very lengthy, and you simply need more time to review it. Or maybe you feel that communications between your organization and the lead agency have, for some reason, not been constructive.

The most appropriate steps to take if you find yourself in these kinds of situations always depend, of course, on the particular people, timing, and proposal at hand. Nonetheless, here are some possible factors and courses of action to consider.

Do Not Wait Too Long

First, do not wait too long to raise your concerns; raise them as soon as practicable, and be mindful of the comment period and when it ends. If you just sit back and hope that things will get “better” or that your comments will have greater effect later, you may hear that “you should have raised this sooner.” At times, waiting can be detrimental to your interests as well as to the rest of the public and the agency involved. For example, if you feel strongly that a particular alternative should be addressed and do not raise it during the scoping process, then it will not get the benefit of comparative analysis with the other alternatives. In addition, it could result in a more expensive and lengthy process (costing taxpayers, including yourself, more) if your delayed suggestion results in the agency deciding to issue a supplemental EIS analyzing that alternative. Or, if you or your organization later go to court to argue that a certain alternative should have been analyzed in the NEPA document, the judge may find that the court will not

consider that information because you should have raised your concern earlier during the NEPA process.⁶⁹

Contact the Agency

Your first line of recourse should be with the individual that the agency has identified as being in charge of this particular process. See if you can sit down with him or her to discuss your concern(s). You may be pleasantly surprised at the response.

Collaboration and Conflict Resolution Support

Some decisions necessarily involve conflicting views, so Federal agencies may choose to engage an impartial third-party to support stakeholder engagement and conflict resolution in a NEPA process. Impartial third-party support may include facilitation, mediation, stakeholder engagement process design, and other services to enhance collaboration between the lead agency and its partners, stakeholders, and citizens. These approaches, referred to as environmental collaboration and conflict resolution (ECCR), are often beneficial if the process ahead may be particularly contentious or challenging and include a past history of deeply divided interests. If you believe the process that you are involved with has a high-level of conflict or contention, consider raising with the lead agency the possibility of enhancing collaborative opportunities within the NEPA process using outside assistance.

In recent years, the Federal Government has used ECCR due to its numerous benefits. The Office of Management and Budget (OMB) and CEQ underscored ECCR's utility by jointly issuing memoranda that directed Federal agencies to increase the effective use of environmental conflict resolution and build institutional capacity for collaborative problem solving.⁷⁰ These memoranda highlighted basic principles for agency engagement in ECCR processes, including informed commitment, balanced and voluntary representation, group autonomy, informed process, accountability, openness, timeliness, and implementation.

ECCR offers many advantages over adversarial approaches to resolve environmental challenges, such as litigation. A 2018 report examining the use of ECCR in Federal processes over the previous decade found that these approaches lead to a savings in time and financial resources, an improvement in relationships between government and stakeholders, and improved outcomes.⁷¹ For example, between 2011 and 2014, the EPA reported that ECCR took 45 percent less time to reach a decision, 30 percent fewer staff, and 79 percent fewer lead attorney hours.⁷² And in a 2009 study, those involved in ECCR reported improved relationships, ability to work together, and level of trust.⁷³ Other benefits to ECCR include:

- Better information, diverse expertise, better-informed decisions;
- Fairer process, especially for traditionally disadvantaged/under-represented parties;
- Better integration, enhanced coordination, and streamlining;
- Conflict prevention and resolution of differences;
- Improved fact-finding and common understanding of the facts;
- Increased social capital through the promotion of trust and partnership;
- Easier implementation “vesting” stakeholders in decision implementation;
- Enhanced stewardship promoted through cooperation; and

- Reduced litigation by solving problems at lowest possible level and narrowing issues.

The NCECR is a Federal agency⁷⁴ that provides collaboration, consensus-building, and conflict resolution services on a range of environmental, natural and cultural resources, Tribal, and public lands issues involving the Federal Government. Citizens can work with lead agencies to express their interest in a collaborative approach and may recommend the involvement of the NCECR.⁷⁵ There may also be an environmental conflict resolution office in your state that can provide assistance, and there are also many other individuals and organizations in the private sector that provide various types of conflict resolution services.

NEPA's Requirements

Perhaps your concern involves understanding a legal requirement. There are, of course, many ways to obtain the advice of lawyers knowledgeable about the NEPA process: the lead agency, private attorneys, and public interest attorneys. Build your own understanding by reading information on the [NEPA.gov](http://www.nepa.gov) website. You may also call the General Counsel's office or the Associate Director for NEPA at the Council on Environmental Quality for assistance in understanding NEPA's legal requirements or for advice and assistance if you have tried to work with the lead agency but feel those efforts have been unsuccessful (see Appendix A for contact information).

Remedies Available

Finally, of course, there are both administrative and judicial remedies available. A few Federal agencies, such as the Bureau of Land Management and the Forest Service, have an administrative appeals process. Each process is specific to that agency. If an appeal is available, you may find it beneficial to invoke it to try to resolve your concerns with the agency's decisions without the need for a legal challenge. Moreover, a statute or agency regulation may require you to exhaust such an appeal procedure before seeking judicial review. Citizens who believe that a Federal agency's actions violate NEPA may seek judicial review (after any required administrative appeals) in Federal court under the Administrative Procedure Act. If you are represented by a lawyer, you should consult with him or her about appropriate options and about communicating with the Federal agencies.

Final Thoughts

This guide was developed to explain NEPA, how it is implemented, and how people outside the Federal Government—individual citizens, private sector applicants, members of organized groups, or representatives of Tribal, State, or local government agencies—can better participate in the assessment of environmental impacts conducted by Federal agencies. To learn more about CEQ and NEPA, visit our web sites at <http://www.whitehouse.gov/ceq> and [NEPA.gov](http://www.nepa.gov) or contact the CEQ Associate Director for NEPA at (202) 395-5750. Your thoughts and comments on improving this Guide for future editions are always welcome.

Appendix A: About the Council on Environmental Quality

The National Environmental Policy Act (NEPA) established the Council on Environmental Quality (CEQ) in 1970 within the Executive Office of the President. CEQ oversees Federal agency NEPA implementation and develops and recommends national policies to the President that promote the improvement of environmental quality and meet the Nation's goals. In addition, CEQ is assigned various duties and responsibilities under other statutes, Executive Orders, and Presidential Memoranda, including with regard to Federal ocean policy, Federal sustainability, and timely environmental review and permitting processes for infrastructure development, and other matters.

The Council on Environmental Quality is housed within the Executive Office of the President. CEQ has offices within the Eisenhower Executive Office Building (EEOB) and within the Jackson Place townhouses on Lafayette Square.

Mailing Address

Council on Environmental Quality
730 Jackson Place, NW
Washington, DC 20503

Main Line: (202) 395-5750

Fax: (202) 456-6546

Appendix B: Useful Websites

NEPA.gov

[NEPA.gov](https://www.nepa.gov) is the Council on Environmental Quality's NEPA website that is supported by the U.S. Department of Energy. It contains a wealth of information related to NEPA. The site contains CEQ guidance as well as studies and reports and information on NEPA training.

Under the "Laws & Regulations" section, there are several useful links including:

- The NEPA Statute
- Executive Orders
- CEQ Regulations for Implementing NEPA
- State NEPA Information
- The Legislative History of NEPA
- Individual Federal Agency Procedures for Implementing NEPA⁷⁶

The other sections provide information about:

- Guidance
- How to get involved
- Resources on NEPA Practice
- CEQ Publications
- CEQ Reports

The *Federal Register* and How to Use It

<https://www.federalregister.gov/>

The *Federal Register* is the official daily publication for rules, proposed rules, and notices of Federal agencies and organizations, as well as executive orders and other presidential documents. It is updated daily by 6 a.m. and is published Monday through Friday, except Federal holidays.

This is where you will find notices from Federal agencies regarding their NEPA actions. Information on the availability of documents, schedule of meetings, and notices of intent to prepare EISs are also published in the *Federal Register*. In addition, EPA publishes a list of EISs that they have received from agencies each week, and a summary of ratings on EISs that they have reviewed.

The easiest way to pull up notices is to have as much information as possible. Key words such as the name of the agency, location of the action, date or date ranges of the publication are all helpful in the search.

The Electronic Code of Federal Regulations (e-CFR)

www.ecfr.gov

The Electronic Code of Federal Regulations (e-CFR) is a currently updated version of the Code of Federal Regulations (CFR). It is not an official legal edition of the CFR. The e-CFR is an editorial compilation of CFR material and *Federal Register* amendments produced by the National Archives and Records Administration's Office of the Federal Register (OFR) and the Government Publishing Office. The OFR updates the material in the e-CFR on a daily basis. The current update status appears at the top of all e-CFR web pages.

The United States Code

The United States Code is a compilation of most public laws currently in force, organized by subject matter. When a law has been amended by another law, the U.S. Code reflects this change. The U.S. Code collates the original law with subsequent amendments, and it deletes language that has later been repealed or superseded.

The full text of the official version of the U.S. Code is provided on www.govinfo.gov at www.govinfo.gov/app/collection/uscode. You can do fielded searches to look for Code material by popular name of the law, the public law number, U.S. Code citation, Statutes at Large citation, or word or phrase. You can also browse the U.S. Code by individual Code titles, down to the section level, for the latest available update.

The U.S. House Office of the Law Revision Counsel also provides the full text of the official version of the U.S. Code at uscode.house.gov/. You can do fielded searches or download entire titles or chapters. This site also provides classification tables that show where recently enacted laws will appear in the United States Code and which sections of the Code have been amended by those laws.

The Federal Infrastructure Permitting Dashboard

www.permits.performance.gov

The Permitting Dashboard is an online tool for Federal agencies, project developers, and interested members of the public to track the Federal Government's environmental review and authorization processes for large or complex infrastructure projects, part of a government-wide effort to improve coordination, transparency, and accountability.

A major function of this Dashboard is to track infrastructure projects designated as "Covered Projects" under Title 41 of the Fixing America's Surface Transportation Act (FAST-41). The Dashboard also provides information on most DOT projects, as well as other infrastructure projects. Follow the "Projects" link for project-specific information.

Appendix C: Agency NEPA Contacts

The list of Federal NEPA Contacts is maintained on [NEPA.gov](https://nepa.gov) under the heading “NEPA Practice” and is periodically updated.

The complete list is available via the link entitled “Federal NEPA Contacts” or available directly at <https://ceq.doe.gov/nepa-practice/agency-nepa-contacts.html>. If you do not have computer access, call CEQ at (202) 395-5750 for assistance.

Appendix D: Statutory References

The National Environmental Policy Act of 1969

42 U.S.C. 4321. Congressional declaration of purpose [Sec. 2]

The purposes of this chapter are: To declare a national policy which will encourage productive and enjoyable harmony between man and his environment; to promote efforts which will prevent or eliminate damage to the environment and biosphere and stimulate the health and welfare of man; to enrich the understanding of the ecological systems and natural resources important to the Nation; and to establish a Council on Environmental Quality.

(Pub. L. 91–190, § 2, Jan. 1, 1970, 83 Stat. 852)

SUBCHAPTER I—POLICIES AND GOALS [TITLE I]

42 U.S.C. 4331. Congressional declaration of national environmental policy [Sec. 101]

(a) The Congress, recognizing the profound impact of man’s activity on the interrelations of all components of the natural environment, particularly the profound influences of population growth, high-density urbanization, industrial expansion, resource exploitation, and new and expanding technological advances and recognizing further the critical importance of restoring and maintaining environmental quality to the overall welfare and development of man, declares that it is the continuing policy of the Federal Government, in cooperation with state and local governments, and other concerned public and private organizations, to use all practicable means and measures, including financial and technical assistance, in a manner calculated to foster and promote the general welfare, to create and maintain conditions under which man and nature can exist in productive harmony, and fulfill the social, economic, and other requirements of present and future generations of Americans.

(b) In order to carry out the policy set forth in this chapter, it is the continuing responsibility of the Federal Government to use all practicable means, consistent with other essential considerations of national policy, to improve and coordinate Federal plans, functions, programs, and resources to the end that the Nation may—

- (1) fulfill the responsibilities of each generation as trustee of the environment for succeeding generations;
- (2) assure for all Americans safe, healthful, productive, and esthetically and culturally pleasing surroundings;
- (3) attain the widest range of beneficial uses of the environment without degradation, risk to health or safety, or other undesirable and unintended consequences;
- (4) preserve important historic, cultural, and natural aspects of our national heritage, and maintain, wherever possible, an environment which supports diversity, and variety of individual choice;
- (5) achieve a balance between population and resource use which will permit high standards of living and a wide sharing of life’s amenities; and
- (6) enhance the quality of renewable resources and approach the maximum attainable recycling of depletable resources.

(c) The Congress recognizes that each person should enjoy a healthful environment and that each person has a responsibility to contribute to the preservation and enhancement of the environment.

(Pub. L. 91–190, title I, § 101, Jan. 1, 1970, 83 Stat. 852)

42 U.S.C. 4332. Cooperation of agencies; reports; availability of information; recommendations; international and national coordination of efforts [Sec. 102]

The Congress authorizes and directs that, to the fullest extent possible: (1) the policies, regulations, and public laws of the United States shall be interpreted and administered in accordance with the policies set forth in this chapter and (2) all agencies of the Federal Government shall—

(A) utilize a systematic, interdisciplinary approach which will insure the integrated use of the natural and social sciences and the environmental design arts in planning and in decisionmaking which may have an impact on man's environment;

(B) identify and develop methods and procedures, in consultation with the Council on Environmental Quality established by subchapter II of this chapter, which will insure that presently unquantified environmental amenities and values may be given appropriate consideration in decision- making along with economic and technical considerations;

(C) include in every recommendation or report on proposals for legislation and other major Federal actions significantly affecting the quality of the human environment, a detailed statement by the responsible official on—

(i) the environmental impact of the proposed action,

(ii) any adverse environmental effects which cannot be avoided should the proposal be implemented,

(iii) alternatives to the proposed action,

(iv) the relationship between local short-term uses of man's environment and the maintenance and enhancement of long-term productivity, and

(v) any irreversible and irretrievable commitments of resources which would be involved in the proposed action should it be implemented.

Prior to making any detailed statement, the responsible Federal official shall consult with and obtain the comments of any Federal agency which has jurisdiction by law or special expertise with respect to any environmental impact involved. Copies of such statement and the comments and views of the appropriate Federal, State, and local agencies, which are authorized to develop and enforce environmental standards, shall be made available to the President, the Council on Environmental Quality and to the public as provided by section 552 of title 5, and shall accompany the proposal through the existing agency review processes;

(D) Any detailed statement required under subparagraph (C) after January 1, 1970, for any major Federal action funded under a program of grants to States shall not be deemed to be legally insufficient solely by reason of having been prepared by a state agency or official, if:

(i) the State agency or official has statewide jurisdiction and has the responsibility for such action,

- (ii) the responsible Federal official furnishes guidance and participates in such preparation,
- (iii) the responsible Federal official independently evaluates such statement prior to its approval and adoption, and
- (iv) after January 1, 1976, the responsible Federal official provides early notification to, and solicits the views of, any other state or any Federal land management entity of any action or any alternative thereto which may have significant impacts upon such state or affected Federal land management entity and, if there is any disagreement on such impacts, prepares a written assessment of such impacts and views for incorporation into such detailed statement.

The procedures in this subparagraph shall not relieve the Federal official of his responsibilities for the scope, objectivity, and content of the entire statement or of any other responsibility under this Act; and further, this subparagraph does not affect the legal sufficiency of statements prepared by State agencies with less than statewide jurisdiction.

(E) study, develop, and describe appropriate alternatives to recommended courses of action in any proposal which involves unresolved conflicts concerning alternative uses of available resources;

(F) recognize the worldwide and long-range character of environmental problems and, where consistent with the foreign policy of the United States, lend appropriate support to initiatives, resolutions, and programs designed to maximize international cooperation in anticipating and preventing a decline in the quality of mankind's world environment;

(G) make available to States, counties, municipalities, institutions, and individuals, advice and information useful in restoring, maintaining, and enhancing the quality of the environment;

(H) initiate and utilize ecological information in the planning and development of resource-oriented projects; and

(I) assist the Council on Environmental Quality established by subchapter II of this chapter.

(Pub. L. 91–190, title I, § 102, Jan. 1, 1970, 83 Stat. 853; Pub. L. 94–83, Aug. 9, 1975, 89 Stat. 424)

42 U.S.C. 4333. Conformity of administrative procedures to national environmental policy [Sec. 103]

All agencies of the Federal Government shall review their present statutory authority, administrative regulations, and current policies and procedures for the purpose of determining whether there are any deficiencies or inconsistencies therein which prohibit full compliance with the purposes and provisions of this chapter and shall propose to the President not later than July 1, 1971, such measures as may be necessary to bring their authority and policies into conformity with the intent, purposes, and procedures set forth in this chapter.

(Pub. L. 91–190, title I, § 103, Jan. 1, 1970, 83 Stat. 854)

42 U.S.C. 4334. Other statutory obligations of agencies [Sec. 104]

Nothing in section 4332 [Sec. 102] or 4333 [Sec. 103] shall in any way affect the specific statutory obligations of any Federal agency (1) to comply with criteria or standards of environmental quality, (2) to coordinate or consult with any other Federal or State agency, or (3)

to act, or refrain from acting contingent upon the recommendations or certification of any other Federal or State agency.

(Pub. L. 91–190, title I, § 104, Jan. 1, 1970, 83 Stat. 854)

42 U.S.C. 4335. Efforts supplemental to existing authorizations [Sec. 105]

The policies and goals set forth in this chapter are supplementary to those set forth in existing authorizations of Federal agencies.

(Pub. L. 91–190, title I, § 105, Jan. 1, 1970, 83 Stat. 854)

SUBCHAPTER II – COUNCIL ON ENVIRONMENTAL QUALITY [TITLE II]

42 U.S.C. 4341. [Sec. 201] Omitted

Section 201 which required the President to transmit to Congress annually an Environmental Quality Report, was terminated by Congress, effective May 15, 2000, pursuant to section 3003 of Pub. L. 104–66, as amended, set out as a note under section 1113 of Title 31, Money and Finance.

(Pub. L. 91–190, title II, § 201, Jan. 1, 1970, 83 Stat. 854; Pub. L. 104–66, title III, § 3003, Dec. 21, 1995 of as amended, 31 U.S.C. 1113)

42 U.S.C. 4342. Establishment; membership; Chairman; appointments [Sec. 202]

There is created in the Executive Office of the President a Council on Environmental Quality (hereinafter referred to as the “Council”). The Council shall be composed of three members who shall be appointed by the President to serve at his pleasure, by and with the advice and consent of the Senate. The President shall designate one of the members of the Council to serve as Chairman. Each member shall be a person who, as a result of his training, experience, and attainments, is exceptionally well qualified to analyze and interpret environmental trends and information of all kinds; to appraise programs and activities of the Federal Government in the light of the policy set forth in subchapter I of this chapter; to be conscious of and responsive to the scientific, economic, social, esthetic, and cultural needs and interests of the Nation; and to formulate and recommend national policies to promote the improvement of the quality of the environment.

(Pub. L. 91–190, title II, § 202, Jan. 1, 1970, 83 Stat. 854)

Provisions stating that notwithstanding this section, the Council was to consist of one member, appointed by the President, by and with the advice and consent of the Senate, serving as chairman and exercising all powers, functions, and duties of the Council, were contained in the Department of the Interior, Environment, and Related Agencies Appropriations Act, 2006, Pub. L. 109–54, title III, Aug. 2, 2005, 119 Stat. 543, and were repeated in provisions of subsequent appropriations acts which are not set out in the Code.

42 U.S.C. 4343. Employment of personnel, experts and consultants [Sec. 203]

(a) The Council may employ such officers and employees as may be necessary to carry out its functions under this chapter. In addition, the Council may employ and fix the compensation of such experts and consultants as may be necessary for the carrying out of its functions under this Act, in accordance with section 3109 of title 5, (but without regard to the last sentence thereof).

(b) Notwithstanding section 1342 of Title 31, the Council may accept and employ voluntary and uncompensated services in furtherance of the purposes of the Council.

(Pub. L. 91–190, title II, § 203, Jan. 1, 1970, 83 Stat. 855; Pub. L. 94–52, § 2, July 3, 1975, 89 Stat. 258)

42 U.S.C. 4344. Duties and functions [Sec. 204]

It shall be the duty and function of the Council—

(1) to assist and advise the President in the preparation of the Environmental Quality Report required by section 4341[Sec. 201] of this title;¹

(2) to gather timely and authoritative information concerning the conditions and trends in the quality of the environment both current and prospective, to analyze and interpret such information for the purpose of determining whether such conditions and trends are interfering, or are likely to interfere, with the achievement of the policy set forth in subchapter I of this chapter, and to compile and submit to the President studies relating to such conditions and trends;

(3) to review and appraise the various programs and activities of the Federal Government in the light of the policy set forth in subchapter I of this chapter for the purpose of determining the extent to which such programs and activities are contributing to the achievement of such policy, and to make recommendations to the President with respect thereto;

(4) to develop and recommend to the President national policies to foster and promote the improvement of environmental quality to meet the conservation, social, economic, health, and other requirements and goals of the Nation;

(5) to conduct investigations, studies, surveys, research, and analyses relating to ecological systems and environmental quality;

(6) to document and define changes in the natural environment, including the plant and animal systems, and to accumulate necessary data and other information for a continuing analysis of these changes or trends and an interpretation of their underlying causes;

(7) to report at least once each year to the President on the state and condition of the environment; and

(8) to make and furnish such studies, reports thereon, and recommendations with respect to matters of policy and legislation as the President may request.

(Pub. L. 91–190, title II, § 204, Jan. 1, 1970, 83 Stat. 855)

42 U.S.C. 4345. Consultation with Citizens' Advisory Committee on Environmental Quality and other representatives [Sec. 205]

In exercising its powers, functions, and duties under this Act, the Council shall—

(1) consult with the Citizens' Advisory Committee on Environmental Quality established by Executive Order numbered 11472, dated May 29, 1969, and with such representatives of science, industry, agriculture, labor, conservation organizations, State and local governments and other groups, as it deems advisable; and

¹ CEQ notes that Congress amended 42 U.S.C. 4341 to remove the Environmental Quality Report requirement.

(2) utilize, to the fullest extent possible, the services, facilities and information (including statistical information) of public and private agencies and organizations, and individuals, in order that duplication of effort and expense may be avoided, thus assuring that the Council's activities will not unnecessarily overlap or conflict with similar activities authorized by law and performed by established agencies.

(Pub. L. 91–190, title II, § 205, Jan. 1, 1970, 83 Stat. 855)

42 U.S.C. 4346. Tenure and compensation of members [Sec. 206]

Members of the Council shall serve full time and the Chairman of the Council shall be compensated at the rate provided for Level II of the Executive Schedule Pay Rates (5 U.S.C. 5313). The other members of the Council shall be compensated at the rate provided for Level IV of [f] the Executive Schedule Pay Rates (5 U.S.C. 5315).

(Pub. L. 91–190, title II, § 206, Jan. 1, 1970, 83 Stat. 856)

42 U.S.C. 4346a. Travel reimbursement by private organizations and Federal, State, and local governments [Sec. 207]

The Council may accept reimbursements from any private nonprofit organization or from any department, agency, or instrumentality of the Federal Government, any State, or local government, for the reasonable travel expenses incurred by an officer or employee of the Council in connection with his attendance at any conference, seminar, or similar meeting conducted for the benefit of the Council.

(Pub. L. 91–190, title II, § 207, as added Pub. L. 94–52, § 3, July 3, 1975, 89 Stat. 258)

42 U.S.C. 4346b. Expenditures in support of international activities [Sec. 208]

The Council may make expenditures in support of its international activities, including expenditures for: (1) international travel; (2) activities in implementation of international agreements; and (3) the support of international exchange programs in the United States and in foreign countries.

(Pub. L. 91–190, title II, § 208, as added Pub. L. 94–52, § 3, July 3, 1975, 89 Stat. 258)

42 U.S.C. 4347. Authorization of appropriations [Sec. 209]

There are authorized to be appropriated to carry out the provisions of this chapter not to exceed \$300,000 for fiscal year 1970, \$700,000 for fiscal year 1971, and \$1,000,000 for each fiscal year thereafter.

(Pub. L. 91–190, title II, § 209, formerly § 207, Jan. 1, 1970, 83 Stat. 856, renumbered § 209, Pub. L. 94–52, § 3, July 3, 1975, 89 Stat. 258)

The Clean Air Act—Section 309

42 U.S.C. 7609. Policy review [Sec. 309]

(a) Environmental impact

The Administrator shall review and comment in writing on the environmental impact of any matter relating to duties and responsibilities granted pursuant to this chapter or other provisions of the authority of the Administration, contained in any (1) legislation proposed by any Federal

department or agency, (2) newly authorized Federal projects for construction and any major Federal agency action (other than a project for construction) to which section 4332(2)(C) of the title applies, and (3) proposed regulations published by any department or agency of the Federal Government. Such written comment shall be made public at the conclusion of any such review.

(b) Unsatisfactory legislation, action, or regulation

In the event the Administrator determines that any such legislation, action, or regulation is unsatisfactory from the standpoint of public health or welfare or environmental quality, he shall publish his determination and the matter shall be referred to the Council on Environmental Quality.

(July 14, 1955, ch. 360, title III, § 309, as added Pub. L. 91–604, § 12(a), Dec. 31, 1970, 84 Stat. 1709)

¹ National Environmental Policy Act of 1969, *as amended*, 42 U.S.C. 4321–4347 provided in Appendix D.

² Section 102 of the National Environmental Policy Act of 1969, 42 U.S.C. 4332.

³ CEQ NEPA Regulations, 40 CFR parts 1500–1508, available at NEPA.gov.

⁴ Council on Environmental Quality, “Regulations for Implementing the Procedural Provisions of the National Environmental Policy Act,” 40 CFR 1508.1(k) available at [NEPA.gov](https://www.nepa.gov).

⁵ 40 CFR 1507.2(a) and 1508.1(dd).

⁶ 40 CFR 1507.2.

⁷ 40 CFR 1507.4.

⁸ Agencies publish their draft NEPA procedures in the *Federal Register*, and the CEQ NEPA regulations require a public comment period prior to CEQ approval. 40 CFR 1507.3. Members of the public may participate in the development of agency NEPA procedures by providing comments. Most agencies already have NEPA procedures; however, when they are changed, the agency will again provide for public comment on the proposed changes.

⁹ See Appendix C for information on how to access agency points of contact and agency websites.

¹⁰ 40 CFR 1508.1(q)(2). Note that this section applies only to legislation drafted and submitted to Congress by Federal agencies. NEPA does not apply to legislation initiated by members of Congress or by the President of the United States.

¹¹ 40 CFR 1508.1(x).

¹² 40 CFR 1501.1.

¹³ 40 CFR 1502.24.

¹⁴ 40 CFR 1506.2.

¹⁵ 40 CFR 1507.3.

¹⁶ Clean Air Act, 42 U.S.C. 7609.

¹⁷ For additional information see www.epa.gov/nepa.

¹⁸ About a quarter of the States have such laws; for example, New York, Montana, Washington, and California all have such laws. New York City also has such a law. A list with references is available at NEPA.gov by clicking on “Laws & Regulations,” the “State NEPA Information” or directly at <https://ceq.doe.gov/laws-regulations/states.html>.

¹⁹ 40 CFR 1508.1(d).

²⁰ CEQ has developed a comprehensive list of the Federal agencies’ CEs, which is available at <https://ceq.doe.gov/nepa-practice/categorical-exclusions.html>. Citizens may consult this resource but also should review the relevant agency’s NEPA procedures to ensure that a CE is currently available for use.

²¹ 40 CFR 1508.1(h).

²² 40 CFR 1501.10(b)(1).

²³ 40 CFR 1501.5(c)(2).

²⁴ 40 CFR 1501.3(b).

²⁵ 40 CFR 1501.5(e).

²⁶ 40 CFR 1508.1(l).

²⁷ 40 CFR 1501.6(a)(2).

²⁸ 40 CFR 1502.3.

²⁹ 40 CFR 1501.10(b)(2).
³⁰ 40 CFR 1508.1(cc).
³¹ 40 CFR 1501.9(d).
³² 40 CFR 1501.9.
³³ Public hearings are run in a formal manner, with a recording or minutes taken of speakers' comments. Public meetings may be held in a variety of formats, and may be much more informal than hearings.
³⁴ 40 CFR 1502.13.
³⁵ 40 CFR 1502.14.
³⁶ 40 CFR 1508.1(z).
³⁷ 40 CFR 1502.14(d).
³⁸ 40 CFR 1502.14(c).
³⁹ 40 CFR 1508.1(g).
⁴⁰ 40 CFR 1508.1(g)(1).
⁴¹ 40 CFR 1502.16(b).
⁴² 40 CFR 1502.17(a).
⁴³ 40 CFR 1502.18.
⁴⁴ 40 CFR 1502.10.
⁴⁵ 40 CFR 1502.19.
⁴⁶ 40 CFR 1503.4.
⁴⁷ 40 CFR 1502.17(b).
⁴⁸ 40 CFR 1506.11(b) references statutory provisions for combining a final EIS and ROD. If the end of the 30 day wait period is less than 90 days after the notice of availability of the Draft EIS, was published in the *Federal Register*, then the decision must await the expiration of the 90 days.
⁴⁹ 40 CFR part 1504.
⁵⁰ The NCECR reports disputes it is involved with to CEQ and requests concurrence from CEQ to engage in those disputes involving two or more Federal agencies.
⁵¹ 40 CFR 1505.2.
⁵² 40 CFR 1505.2(a)(3).
⁵³ 40 CFR 1505.2(b).
⁵⁴ 40 CFR 1502.9(d).
⁵⁵ Clean Air Act, 42 U.S.C. 7609.
⁵⁶ 40 CFR 1506.11.
⁵⁷ 40 CFR 1507.4(a).
⁵⁸ Public Law 114–94, sec. 41001–41014, 129 Stat. 1312, 1741 (42 U.S.C. 4370m—4370m–12).
⁵⁹ 42 U.S.C. 4370m–2(a)(6)(B).
⁶⁰ 42 U.S.C. 4370m–6.
⁶¹ CEQ NEPA Regulations, 40 CFR 1507.4.
⁶² CEQ NEPA Regulations, 40 CFR 1501.9(d).
⁶³ CEQ NEPA Regulations, 40 CFR 1502.17, 1503.1(a)(3).
⁶⁴ 40 CFR 1500.3(b).
⁶⁵ 40 CFR 1501.8, 1508.1(e).
⁶⁶ 40 CFR 1505.3(d).
⁶⁷ 40 CFR 1505.3(c).
⁶⁸ There are many reference books for how to research issues, review documents, and write comments. One in particular is “The Art of Commenting” by Elizabeth Mullin from the Environmental Law Institute (Mullin, Elizabeth D. 2000. The Art of Commenting: How to Influence Environmental Decisionmaking with Effective Comments, Environmental Law Institute, Washington, DC). Another useful reference for those involved in commenting on transportation projects is the American Association of State Highway and Transportation Officials (AASHTO) Practitioner’s Handbook 05-Utilizing Community Advisory Committees for NEPA Studies, December, 2006, <http://environment.transportation.org> or available through AASHTO’s Center for Environmental Excellence by calling (202) 624-3635.
⁶⁹ 40 CFR 1500.3(b), 1503.3(b).

⁷⁰ Memorandum on Environmental Conflict Resolution (Nov. 28, 2005), as expanded by Memorandum on Environmental Collaboration and Conflict Resolution (Sept. 7, 2012), <https://ceq.doe.gov/nepa-practice/environmental-collaboration-and-conflict-resolution.html>.

⁷¹ Federal Forum on Environmental Collaboration and Conflict Resolution, Environmental Collaboration and Conflict Resolution (ECCR): Enhancing Agency Efficiency and Making Government Accountable to the People (May 2, 2018), https://ceq.doe.gov/docs/nepa-practice/ECCR_Benefits_Recommendations_Report_%205-02-018.pdf.

⁷² Hall, W.E. (2016, June). “Assessing the value of environmental collaboration and conflict resolution: A census of litigation related cases to estimate comparative process costs at the U.S. Environmental Protection Agency.” Concurrent session presentation, the 29th Annual Conference of the International Association for Conflict Management, Columbia University, New York, NY.

⁷³ Emerson, K., Orr, P.J., Keyes, D.L., & McKnight, K.M. (2009). Environmental conflict resolution: Evaluating performance outcomes and contributing factors. *Conflict Resolution Quarterly*, 27(1), 27–64.

⁷⁴ The McCain Center is a program of Udall Foundation, is an independent, nonpartisan Federal agency. Environmental Policy and Conflict Resolution Act of 1998, 20 U.S.C. 5601–5609, as amended.

⁷⁵ The McCain Center can be contacted via www.ecr.gov; mailing address: John S. McCain III National Center for Environmental Conflict Resolution, 130 S. Scott Ave. Tucson, AZ 85701; phone: (520) 901-8501; or electronic mail: usiecr@ecr.gov.

⁷⁶ The agency implementing procedures can be accessed at https://ceq.doe.gov/laws-regulations/agency_implementing_procedures.html and are mentioned throughout the Citizen’s Guide as an important part of the process.



Large Trees Dominate Carbon Storage in Forests East of the Cascade Crest in the United States Pacific Northwest

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Large-diameter trees store disproportionately massive amounts of carbon and are a major driver of carbon cycle dynamics in forests worldwide. In the temperate forests of the western United States, proposed changes to Forest Plans would significantly weaken protections for a large portion of trees greater than 53 cm (21 inches) in diameter (herein referred to as “large-diameter trees”) across 11.5 million acres (~4.7 million ha) of National Forest lands. This study is among the first to report how carbon storage in large trees and forest ecosystems would be affected by a proposed policy. We examined the proportion of large-diameter trees on National Forest lands east of the Cascade Mountains crest in Oregon and Washington, their contribution to overall aboveground carbon (AGC) storage, and the potential reduction in carbon stocks resulting from widespread harvest. We analyzed forest inventory data collected on 3,335 plots and found that large trees play a major role in the accumulated carbon stock of these forests. Tree AGC (kg) increases sharply with tree diameter at breast height (DBH; cm) among five dominant tree species. Large trees accounted for 2.0 to 3.7% of all stems (DBH ≥ 1 ” or 2.54 cm) among five tree species; but held 33 to 46% of the total AGC stored by each species. Pooled across the five dominant species, large trees accounted for 3% of the 636,520 trees occurring on the inventory plots but stored 42% of the total AGC. A recently proposed large-scale vegetation management project that involved widespread harvest of large trees, mostly grand fir, would have removed ~44% of the AGC stored in these large-diameter trees, and released a large amount of carbon dioxide to the atmosphere. Given the urgency of keeping additional carbon out of the atmosphere and continuing carbon accumulation from the atmosphere to protect the climate system, it would be prudent to continue protecting ecosystems with large trees for their carbon stores, and also for their co-benefits of habitat for biodiversity, resilience to drought and fire, and microclimate buffering under future climate extremes.

Keywords: carbon, climate mitigation, eastside screens, forests, global change, large-diameter trees, 21-inch rule

INTRODUCTION

Forest carbon accumulation is crucial for mitigating ongoing climatic change, with individual large trees storing a substantial portion of the overall carbon in living trees. Globally, forests store about 862 Gt carbon in live and dead vegetation and soil, with 42% of it stored in live biomass (above- and belowground; Pan et al., 2011). Globally, forests removed the equivalent of about 30% of fossil fuel emissions annually from 2009 to 2018 (Friedlingstein et al., 2019), and 44% of that was by temperate forests. Temperate forests of the United States are the largest category of land sinks in the country, consistently offsetting about 14% of the Nation's CO₂ emissions (EPA, 2020). Projections indicate that ecological systems have significant additional climate mitigation potential, with forest carbon accumulation serving as a central component of a natural climate solutions framework (Griscom et al., 2017; Fargione et al., 2018; Moomaw et al., 2019; Cook-Patton et al., 2020).

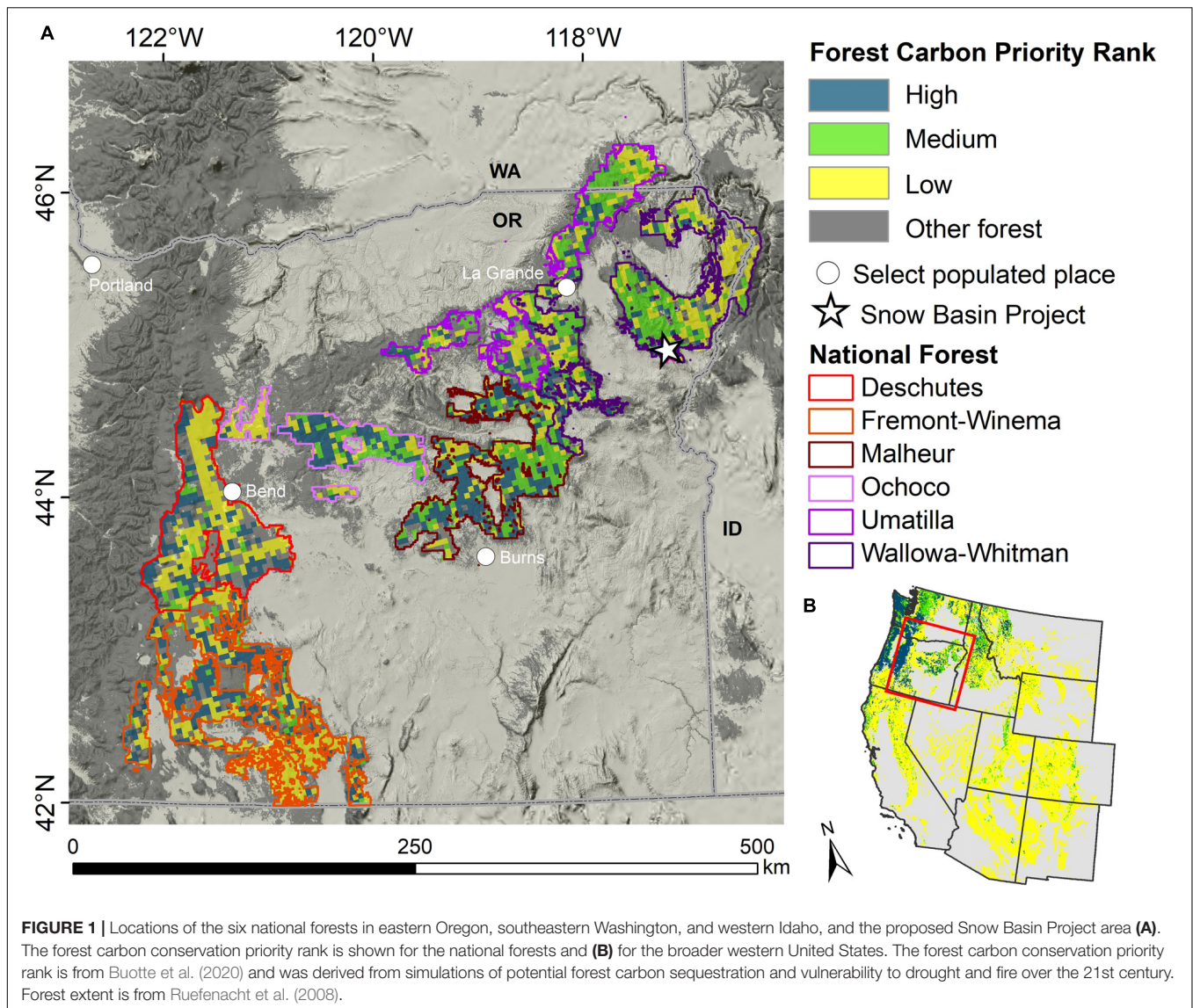
Large-diameter trees constitute about half of the mature forest biomass worldwide and are key to the ability of forests to accumulate substantial amounts of carbon needed to mitigate climate change (Luyssaert et al., 2008; Lutz et al., 2018). Trees exceeding 60 cm (23.6 in) diameter at breast height (DBH) comprise ~41% of the world's aboveground live tree biomass (Lutz et al., 2018). Furthermore, on average, 50% of the live tree biomass carbon in all types of forests globally is stored in the largest 1% of trees, but the value for the United States is lower, ~30% in the largest 1% of trees due to widespread historical logging of large trees (Lutz et al., 2018). A single large tree can add the same amount of carbon to the forest within a year as is contained in a single mid-sized tree of the same species (Stephenson et al., 2014). The relationship between large-diameter trees and overall forest biomass suggests that forests cannot accumulate aboveground carbon (AGC) to their ecological potential without large trees (Lutz et al., 2018). Recognition of the importance of large-diameter trees in determining global atmospheric carbon stocks has led to management recommendations to conserve existing large-diameter trees and those that will soon reach large diameters (Lindenmayer et al., 2014; Lutz et al., 2018; Moomaw et al., 2019).

In addition to comprising a substantial portion of forest carbon storage and accumulation, large-diameter trees fulfill a variety of unique ecological roles such as increasing drought-tolerance, reducing flooding from intense precipitation events, altering fire behavior, redistributing soil water, and acting as focal centers of mycorrhizal communication and resource sharing networks (Bull et al., 1997; Brooks et al., 2002; Brown et al., 2004; Luyssaert et al., 2008; Beiler et al., 2015; Lindenmayer and Laurance, 2017). In the United States Pacific Northwest (PNW), carbon dense old growth forests buffer against increasing temperatures by creating microclimates that shelter understory species from rising temperatures (Frey et al., 2016; Davis et al., 2019a). Forests with large-diameter trees often have high tree species richness, and a high proportion of critical habitat for endangered vertebrate species, indicating a strong potential to support biodiversity into the future and promote ecosystem

resilience to climate change (Lindenmayer et al., 2014; Buotte et al., 2020).

In the PNW region of the United States a 21-inch (~53 cm) diameter criteria (USDA Forest Service, 1995) was enacted in 1994 to slow the loss of large, older trees and old forest patches in national forests east of the Cascade Mountains crest in Oregon and Washington. The forests under consideration shall be referred to as “eastside forests” to be consistent with United States Forest Service (USFS) terminology. In our study, we refer to trees that equal or exceed this value to be “large-trees” or “large-diameter trees.” Extensive studies determined that the large tree component of old forest structure had decreased due to human uses, and that sensitive wildlife species associated with old growth forest such as the American Marten and the Northern Goshawk were also in decline (Interior Columbia Basin Ecosystem Management Project, 2000; Wisdom et al., 2000; Bull et al., 2005; Greenwald et al., 2005). Site-specific exceptions have allowed removal of some trees ≥ 21 in DBH. However, the 21-inch rule has prevented large-scale harvest of trees ≥ 21 in DBH. For example, the 2010 Snow Basin Vegetation Management Project proposed harvesting over 43,000 trees ≥ 21 in DBH but was stopped by litigation [League of Wilderness Defenders, et al., v. Connaughton, et al., No. 3:12-cv-02271-HZ (D. Or. Dec. 9, 2014)]. Recently restoration of these forests was claimed to be inhibited by the 21-inch rule that protects large trees from logging (Hessburg et al., 2020). The rationale for harvesting large trees is premised upon the use of historical baselines of stand structure and species composition as management targets, and assuming that by removing large shade-tolerant species like grand fir and Douglas-fir it will promote resilience to future drought and disturbance (Johnston et al., 2018; Merschel et al., 2019; Hessburg et al., 2020). However, ongoing climate change and many other anthropogenic stressors such as habitat fragmentation, invasive species, and declines in biodiversity, heighten concerns over use of historical conditions as management targets (Millar et al., 2007; IPCC, 2018; Ripple et al., 2020). Proposed amendments to the 21-inch diameter limit would allow widespread harvesting of the larger trees up to 30 in DBH (76.2 cm) across six National Forests of eastern Oregon and southwestern Washington with major implications for forest carbon dynamics and important environmental co-benefits (Figure 1).

Carbon storage is an increasingly important management objective for National Forest Lands in the United States (Depro et al., 2008; Dilling et al., 2013; Dugan et al., 2017). USFS Forest Inventory and Analysis (FIA) data have been used to develop baseline stocks and trends of forest carbon for every region and individual National Forest, including assessment of the main disturbance, management, and environmental factors that drive changes (Birdsey et al., 2019). Western United States forests show considerable potential to accumulate additional carbon over the coming century, especially forests within the PNW that are projected to have relatively low to moderate vulnerability to future drought and fire (Figure 1; Buotte et al., 2020). Strategies to mitigate climate change effects on forests require careful examination of the tradeoffs of proposed forestry practices on forest carbon stock accumulation, water cycling, and additional environmental co-benefits of forests, such as biodiversity and



microclimatic buffering (McKinley et al., 2011; Law et al., 2018; Sheil, 2018; Buotte et al., 2020).

The potential effects of changing the 21-inch rule on carbon storage of eastside forests have not been adequately considered. Proposed changes to the management of large trees should be carefully assessed prior to adoption of new rules given that forest carbon storage is critically important in the context of a warming climate, and that large trees disproportionately store more accumulated carbon, keeping it out of the atmosphere. The objective of this study is to evaluate the potential impact of removal of the 21-inch rule on forest carbon stocks and carbon stock accumulation in eastside forests (Figure 1). We examine the relationship between tree DBH and AGC storage at tree to population scales for selected dominant tree species in eastside forests, focusing on the following questions:

1. How does AGC storage change with tree diameter among the dominant eastside tree species?

2. How common are large trees across the eastside forests by species and what proportion of each species total AGC do they account for?
3. What are the potential consequences on carbon stores of widespread removal of large trees in eastside forest restoration projects?

MATERIALS AND METHODS

Study Area

Our study area included the six national forests in eastern Oregon, two of which extend slightly into southwestern Washington and western Idaho (Figure 1). Located within the East Cascades and Blue Mountains ecoregions, these national forests together cover approximately 11.5 million acres (4.7 million ha) of complex mountainous terrain, characterized by a broad range of environmental gradients in climate regimes and

TABLE 1 | Summary of forest inventory data and climate conditions for the six National Forests included in the analysis.

National forest	Forest inventory			Climate conditions			
	N. Plots	DBH (cm)	AGC (kg)	Annual precipitation (mm)	Temperature (°C)		
					Annual	Jan.	July
Deschutes	474	16 (187)	131 (11359)	817 (170)	6.1 (0.6)	−1.5 (1.9)	16.1 (1.8)
Fremont-Winema	740	17 (204)	145 (14269)	681 (156)	6.3 (0.6)	−1.4 (1.9)	16.6 (1.8)
Malheur	687	15 (145)	108 (8392)	571 (104)	5.9 (0.7)	−2.9 (2.0)	17.2 (2.1)
Ochoco	253	16 (188)	119 (13462)	484 (101)	7.1 (0.6)	−1.0 (1.9)	17.6 (1.9)
Umatilla	487	14 (139)	99 (8759)	802 (121)	6.8 (0.7)	−1.5 (1.8)	17.5 (2.0)
Wallowa-Whitman	694	13 (164)	91 (8301)	792 (119)	6.0 (0.7)	−3.0 (1.8)	17.3 (2.1)

Forest inventory data were from the USFS Forest Inventory and Analysis program for the period 2000 to 2018. The variables include number of inventory plots and the average (max) diameter at breast height (DBH) and aboveground carbon (AGC) of trees on inventory plots. Average (± 1 SD) climate conditions were based on the 40-year period from 1981 to 2019 and derived from 4-km resolution gridded PRISM data (Daly et al., 2008).

associated forest types (Johnson and Clausnitzer, 1992; Peterson and Waring, 1994; Berner and Law, 2015). The Blue Mountains ecoregion functions ecologically and floristically as a transverse bridge between the Cascade Mountains to the west and the Rocky Mountains to the east (Kerns et al., 2017). This “mega-corridor” links together some of the most intact habitat remaining in the PNW region and is of great importance to regional-scale connectivity and carbon storage in response to a warming climate (McGuire et al., 2016; Buotte et al., 2020).

Climatic conditions from 1981 to 2019 were similar across National Forests with regard to January and July average temperatures but show substantial variation in annual precipitation (Table 1). The Deschutes and Fremont-Winema National Forests are in the eastern Cascade Mountains and thus receive relatively high annual precipitation (817 mm and 681 mm) due to orographic uplift. National Forests in the southwestern Blue Mountains receive less precipitation (Malheur, 571 mm; Ochoco, 484 mm) due to their location within the rainshadow of the Cascade Mountains. In the northeastern portion of our study area annual precipitation increases to 802 mm and 792 mm, respectively, for the Umatilla and Wallowa Whitman National Forests due to the intrusion of moisture-laden air masses through the Columbia River Gorge. The region’s complex precipitation patterns are similarly reflected in the distribution of forest types and biophysical characteristics.

Forest Inventory and Analysis Dataset

We relied on forest inventory measurements from the USFS FIA program and our own selection of biomass equations to evaluate relationships between tree diameter and AGC storage, as well as assess potential effects of tree removal proposed as part of the 2010 Snow Basin project. The FIA conducts systematic forest inventories across the United States, with one field sampling plot for every ~2,400 hectares (6,000 acres) of forest (O’Connell et al., 2017). In the western United States, the FIA surveys 10% of sampling plots each year, meaning that each sampling plot is revisited every 10 years. At each sampling plot, field crews collect data on tree species, tree size, and other forest attributes. These inventory data are available online through the FIA DataMart¹.

¹<https://apps.fs.usda.gov/fia/datamart/>

We downloaded the latest FIA data (FIADB_1.8.0.02) for Oregon, Washington, and Idaho, and then identified inventory plots that occurred within the six national forests based on a spatial overlay with national forest boundaries ($n = 3,973$ plots). To maintain landowner privacy the FIA plot coordinates available to the public are slightly altered (“fuzzed”), thus our spatial overlay may include some plots just outside the national forests or miss some plots that occur around the edges of these forests. We selected the latest inventory measurements from each surveyed plot for the years 2010 to 2018 (the most recent year available). We filtered these data to include only live trees of the five tree species of interest. A total of 54,651 individual trees were measured on the microplots, subplots, and macroplots, which represented 636,520 trees after applying the expansion factors for each type of plot. Overall, we used data from 3,335 plots in this analysis. A summary of FIA plot conditions within our study area shows that relatively small variations in average DBH across National Forests, between 13 cm and 17 cm, translate into substantial differences in AGC (Table 1). The Fremont Winema National Forest had the largest average DBH (~17 cm) and the largest average AGC stores (~145 kg), whereas the Wallowa-Whitman had the smallest average DBH (~13 cm) and the smallest average AGC stores (~91 kg).

We performed the analysis using the statistical software R (v. 3.6.1; R Core Team, 2020) with add-on packages including data.table (Dawle and Srinivasan, 2019), ggplot2 (Wickham, 2016), maptools (Bivand and Lewin-Koh, 2019), raster (Hijmans, 2019), and rgdal (Bivand et al., 2019).

Estimating Tree Aboveground Carbon Storage

We estimated AGC storage (AGC; kg C) for each tree on the FIA inventory plots using tree biometric measurements from FIA along with species- and component-specific biomass allometric models and traits (Supplementary Tables 1, 2). Using species-specific allometric models that include height measurements are essential to capture climate and site effects on taper and height (volume), and reduce uncertainty compared to estimates that are diameter-based and generalized models. For example, Van Tuyl et al. (2005) found that generalized models underestimated

biomass by 50% on the eastside. Specifically, for each tree we estimated AGC as the sum of carbon stored in stem, branch, bark, and foliage biomass. To estimate stem biomass, we first derived stem volume using species-specific allometric models that incorporated tree diameter and height (Cochran, 1985; **Supplementary Table 1**). We then converted stem volume to biomass using published information on the wood density of each species, which we found was important to reducing uncertainty in estimates (Van Tuyl et al., 2005; Ross, 2010; Berner and Law, 2015). We also estimated branch, bark, and foliage biomass using species-specific allometric models, substituting equations for similar species where necessary (Gholz et al., 1979; Means et al., 1994; Jenkins et al., 2004). We assumed dry stem, branch, and bark biomass was 47.6 to 52.5% carbon depending on species (Lamloom and Savidge, 2003) and that dry foliage biomass was 46.1 to 51.4% carbon depending on species (Berner and Law, 2016). For trees smaller than 10 cm DBH, total aboveground biomass was estimated from tree height and then converted to AGC assuming wood carbon content.

Tree-Level Analysis of Relationship Between Tree Diameter and Aboveground Carbon

To better understand how carbon storage varies with tree size in eastside forests, we examined relationships between tree diameter (DBH; cm) and aboveground carbon (AGC; kg) among trees sampled on the FIA plots. Specifically, we rounded the DBH of each tree to the nearest centimeter and then computed the average AGC of trees within each 1-cm size class. To account for sampling uncertainty, we repeatedly computed these metrics using a resampling approach where each draw ($n = 10^4$) utilized data from a random 25% of inventory plots sampled with replacement. We computed the median across these 10^4 draws as our best-estimate of these metrics and also derived 95% confidence intervals.

Population-Level Analysis of Tree Diameter and Aboveground Carbon

We evaluated how the cumulative percentage of tree stems and AGC varied with tree diameter across the six national forests of our study area using forest inventories. This involved quantifying the cumulative percentage of tree stems and total AGC by DBH for each tree species, as well as pooled across tree species. To account for sampling uncertainty, we again implement a resampling approach ($n = 10^4$ draws).

Snow Basin Case Study: Carbon Consequences of Proposed Large Tree Removal

To estimate the carbon consequences of large tree removal we utilized USFS National Environmental Policy Act (NEPA) documentation from the Snow Basin Vegetation Management Project located on the Wallowa-Whitman National Forest, Whitman Ranger District, Baker County, Oregon (**Figure 1**). The project area includes 10,721 ha (26,493 acres) of National Forest land and primarily encompasses two main subwatersheds (Paddy

Creek-Eagle Creek and Little Eagle Creek). Elevations within the project area range from approximately 1,340 m (4,400 feet) on the southern boundary to approximately 1,980 m (6,500 feet) at the northern boundary near the border with the Eagle Cap Wilderness. The USFS's preferred alternative plan was to remove over 43,000 large trees (DBH ≥ 21 in or 53.3 cm) from the project area. The large tree removal was prevented by litigation, but the project nonetheless provides a realistic framework to evaluate the carbon cost of large tree removal associated with dry forest landscape-scale restoration projects.

The Snow Basin NEPA document estimated the number of large trees to be removed or retained per acre by biophysical environment (Snow Basin DEIS). The USFS's preferred alternative proposed removal of grand fir from cool/moist and warm/dry grand fir biophysical environments, as well as removal of ponderosa pine, Douglas-fir, and western larch from mixed ponderosa pine and Douglas-fir biophysical environments (**Table 2**). Based on this information we computed the total number of large trees proposed for removal vs. retention across the project area by biophysical environment. To evaluate the potential carbon consequences of large tree removal, it was necessary to know tree size class distribution. The FIA only surveyed 13 plots in the Snow Basin project area between 2010 and 2018, of which 12 plots included large trees. After an initial analysis, we deemed this inadequate for rigorous assessment of tree size class distribution and therefore used tree measurements from all FIA plots in the Wallowa-Whitman National Forest that included large trees for the species of interest ($n = 217$ plots). To estimate the size class distribution of large trees, we computed the fraction of large trees that occurred at 0.1 cm DBH intervals between 53.3 cm and the largest observed DBH for grand fir, and after combining ponderosa pine, Douglas-fir, and western larch into a mixed-species group. For each biophysical environment, we then distributed the large trees proposed for removal or retention among these size classes in proportion to the occurrence of trees in each size class. Lastly, we estimated the total large tree AGC that would be removed or retained by multiplying the number of trees in each size class by the mean AGC of trees in that size class and then summing AGC across size classes. To account for sampling uncertainty, we again computed these metrics using a resampling approach where each of 10^4 draws utilized data from a random 25% of inventory plots sampled with replacement. As before, we computed the median across these 10^4 draws as our best-estimate and derived 95% confidence intervals. For verification, we also performed this assessment using only FIA plots in the Snow Basin Project area rather than across the Wallowa-Whitman National Forest and present these findings in the **Supplementary Material**.

RESULTS

Relationships Between Tree Diameter and Aboveground Carbon

Average tree aboveground carbon (AGC; kg) rapidly increased with tree diameter at breast height (DBH; cm) among dominant tree species measured on USFS inventory plots located in the

TABLE 2 | The total number of trees ≥ 21 in DBH proposed for removal and retention across the Snow Basin project area as reported in the NEPA plan prepared by the USFS (Table 53 of Snow Basin Vegetation Management Project DEIS, 2011).

Biophysical environment	Acres treated	Species removed	Mean number of large trees per acre		Total number of large trees	
			Removed	Retained	Removed	Retained
Cool/moist grand fir	973	Grand fir	5	9	4,865	8,757
Warm/dry grand fir	5,262	Grand fir	5	5	26,310	26,310
Warm/dry Douglas-fir and ponderosa pine	6,136	Douglas-fir, ponderosa pine, western larch	2	3	12,272	18,408
Warm/moist Douglas-fir						
Grand total	12,371	–	–	–	43,447	53,475

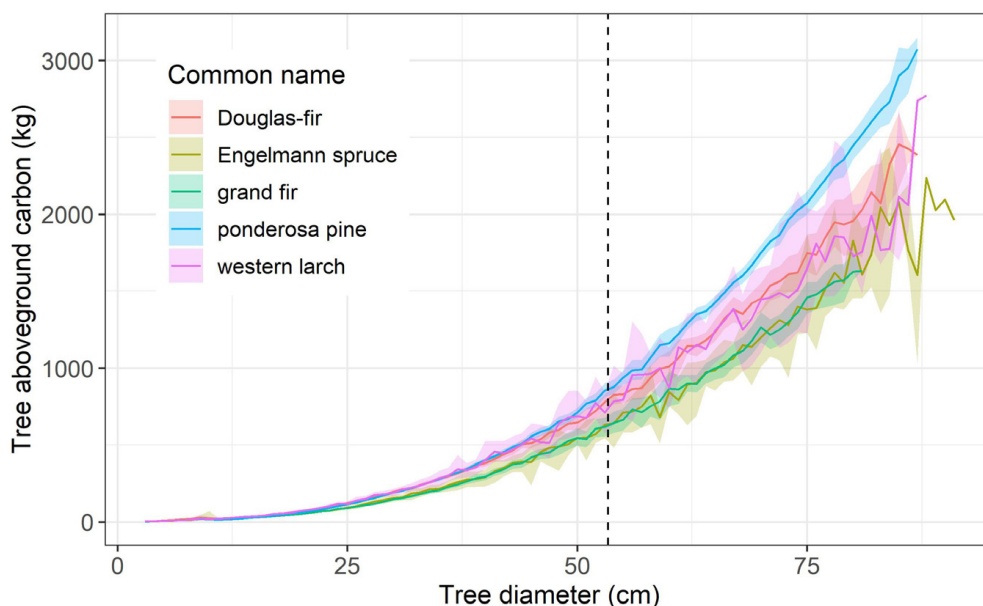


FIGURE 2 | Average tree aboveground carbon (AGC; kg) increases sharply with tree diameter at breast height (DBH; cm) for five dominant tree species measured on USFS inventory plots located in the six eastside national forests. The 21-inch (53.3 cm) DBH limit of the Eastside Screens (vertical dash line) is shown for reference. Tree AGC (≥ 10 cm DBH) was estimated from tree diameter and height using regional species-specific allometric equations, wood density, and carbon content. Similarly, sapling AGC (DBH = 3–9.9 cm) was estimated from sapling height using regional species-specific allometry and wood carbon content. These estimates and 95% confidence intervals were derived by resampling ($n = 10^4$) measurements from inventory plots ($n = 3,335$). For visual clarity, the tree diameter limit for each species excludes the largest 0.5% of trees, generally those larger than 80 to 88 cm DBH.

six eastside national forests (Figure 2). For instance, an average 25 cm (~ 10 ") diameter tree stored 90–121 kg of AGC depending on tree species, while a 50 cm (~ 20 ") diameter tree stored 541–683 kg of AGC. Thus, doubling tree diameter over this range led to a 5.3–6.2-fold increase in AGC. Similarly, tripling tree diameter from 25 cm to 75 cm led to a 13.8–18.2-fold increase in AGC, with the largest increase observed for ponderosa pine. For any given diameter, ponderosa pine, Douglas-fir and western larch tended to store more AGC than an Engelmann spruce or grand fir.

These results clearly showed that for large trees, a small increase in diameter corresponds to a massive increase in additional carbon storage relative to a small tree increasing by the same diameter increment. Overall, as trees grow larger,

each additional centimeter of stem diameter corresponds with a progressively larger increase in tree carbon storage.

Tree Diameter and Aboveground Carbon Storage Within Tree Populations

Large trees (DBH ≥ 21 in or 53.3 cm) accounted for a small percentage of each species' tree stems, but a substantial percentage of the total AGC stored by each species on FIA inventory plots located in the six eastside national forests (Figure 3 and Table 3). Specifically, large trees accounted for ~ 2.0 to $\sim 3.7\%$ of all stems (DBH ≥ 1 " or 2.54 cm) among the five dominant tree species; however, these trees held ~ 33.3 to $\sim 45.8\%$ of the total AGC stored by each species (Table 3). Grand

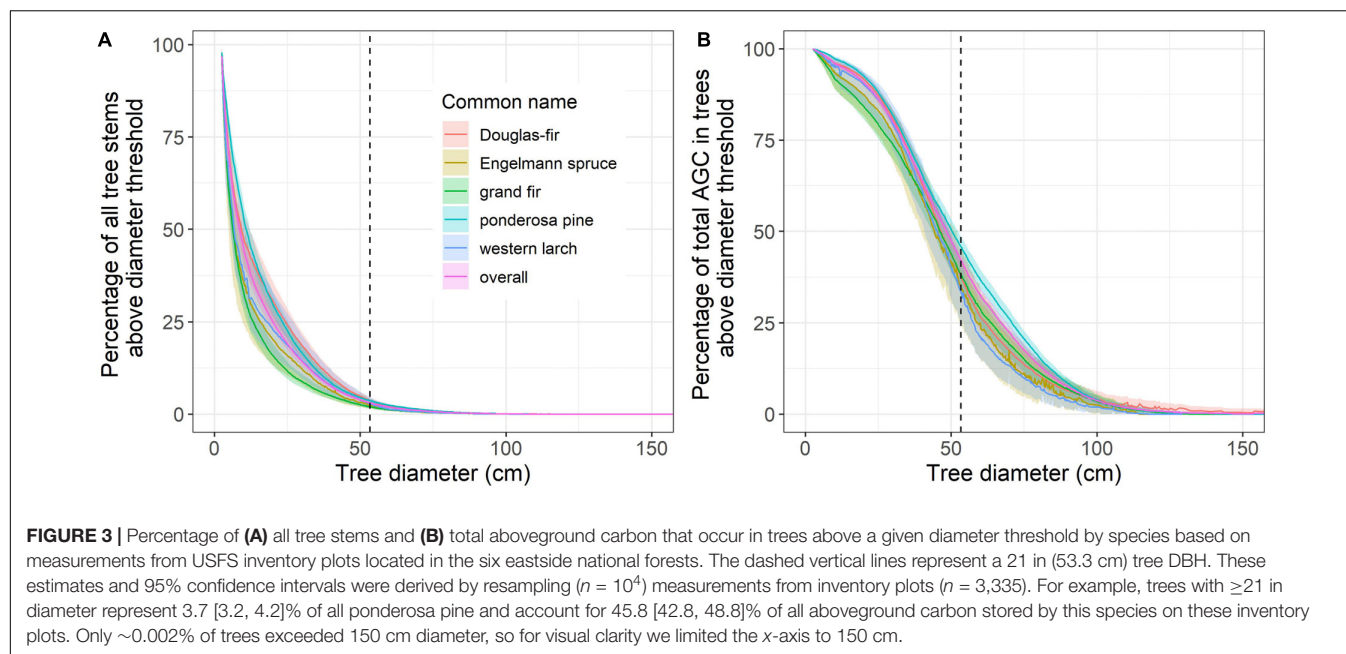


TABLE 3 | Percentage of all trees stems and total aboveground carbon (AGC) occurring in trees above and below the 21 in DBH threshold based on measurements from USFS inventory plots located in the six eastside national forests.

Common name	Number of trees in the population	% of total species stems in trees...		% of total species AGC in trees...	
		<21 in	≥ 21 in	<21 in	≥ 21 in
Douglas-fir	100021	96.3 [95.4, 97.1]	3.7 [2.9, 4.6]	62.5 [58.2, 66.7]	37.5 [33.3, 41.8]
Engelmann spruce	31375	97.6 [96.0, 98.6]	2.4 [1.4, 4.0]	65.3 [54.6, 75.5]	34.7 [24.5, 45.4]
Grand fir	187445	98.0 [97.5, 98.5]	2.0 [1.5, 2.5]	61.6 [57.0, 66.2]	38.4 [33.8, 43.0]
Ponderosa pine	286970	96.3 [95.8, 96.8]	3.7 [3.2, 4.2]	54.2 [51.2, 57.2]	45.8 [42.8, 48.8]
Western larch	30708	97.2 [95.5, 98.4]	2.8 [1.6, 4.5]	66.7 [58.4, 74.8]	33.3 [25.2, 41.6]
Overall	636520	96.9 [96.6, 97.3]	3.1 [2.7, 3.4]	57.8 [55.7, 60.0]	42.2 [40.0, 44.3]

Live trees were measured on 3,335 plots surveyed between 2010 and 2018. The estimates and 95% confidence intervals below were derived by resampling bootstrap sampling. Each bootstrap sample ($n = 10^4$) utilized measurements from 25% of inventory plots selected at random.

fir had the lowest percentage of stems ≥ 21 in DBH ($\sim 2.0\%$), while ponderosa pine and Douglas-fir had the highest percentage of stems exceeding this threshold (both $\sim 3.7\%$). However, large grand fir trees accounted for 38.4% of total species AGC, whereas large Douglas-fir accounted for 37.5% of total species AGC. Western larch had the lowest percentage of total species AGC in trees ≥ 21 in ($\sim 33.3\%$), whereas ponderosa pine had the highest percentage ($\sim 45.8\%$).

Pooling across the five dominant species, large trees accounted for $\sim 3.1\%$ of the 636,520 trees occurring on the inventory plots, but stored $\sim 42.2\%$ of the total AGC. Similarly, based on the FIA's CRM method, the five tree species together accounted for 44.9% of total AGC, with species-specific AGC stores ranging from ~ 36.5 to $\sim 49.3\%$ of the total stored by each species (Supplementary Table 3 and Supplementary Figure 1).

A similar examination for trees > 30 in DBH (76.2 cm) showed that among the five species, trees larger than 30 in DBH accounted for 0.4–0.9% of stems, but held ~ 9.0 to $\sim 19.0\%$ of each species AGC (Table 4). Ponderosa pine had the highest percentage of stems > 30 in DBH ($\sim 0.9\%$) and accounted for

the highest percentage of AGC ($\sim 19.4\%$) among the five species. Douglas-fir had $\sim 0.6\%$ of stems > 30 in DBH, and these stems held $\sim 12.9\%$ of total species AGC. Engelmann Spruce, grand fir, and western larch each had $\sim 0.4\%$ stems > 30 in DBH, and accounted for ~ 10.3 , ~ 14.0 , and $\sim 9.2\%$ of each species total AGC, respectively. Overall, trees > 30 in DBH represent 0.6% of stems on inventory plots, but stored 16.6% of the total AGC.

Snow Basin Case Study: Carbon Consequences of Proposed Large Tree Removal

As per the USFS NEPA documentation, the Snow Basin project preferred alternative would have removed $\sim 43,447$ large trees (DBH ≥ 21 in or 53.3 cm) across 12,371 acres, while retaining $\sim 53,475$ large trees over this area (Table 2). We estimate this would translate into removing ~ 131.3 Gg AGC stored in large trees, while retaining ~ 164.1 Gg AGC in large trees (1 Gg = 10^9 g; Table 5). Grand fir from cool-moist and warm-dry environments would together comprise $\sim 67\%$ (~ 87.3 Gg C) of large tree AGC

TABLE 4 | Percentage of all tree stems and total aboveground carbon (AGC) occurring in trees above and below the 30 in diameter threshold based on measurements from USFS inventory plots located in the six eastside national forests.

Common name	Number of trees in the population	% of total species stems in trees...		% of total species AGC in trees...	
		<30 in	>30 in	<30 in	>30 in
Douglas-fir	100021	99.4 [99.2, 99.5]	0.6 [0.5, 0.8]	87.1 [83.7, 90.2]	12.9 [9.8, 16.3]
Engelmann spruce	31375	99.6 [99.2, 99.9]	0.4 [0.1, 0.8]	89.7 [82.7, 95.7]	10.3 [4.3, 17.3]
Grand fir	187445	99.6 [99.5, 99.7]	0.4 [0.3, 0.5]	86.0 [82.3, 89.3]	14.0 [10.7, 17.7]
Ponderosa pine	286970	99.1 [99.0, 99.3]	0.9 [0.7, 1.0]	80.6 [78.3, 82.7]	19.4 [17.3, 21.7]
Western larch	30708	99.6 [99.3, 99.8]	0.4 [0.2, 0.7]	90.8 [86.2, 94.8]	9.2 [5.2, 13.8]
Overall	636520	99.4 [99.3, 99.4]	0.6 [0.6, 0.7]	83.4 [81.8, 85.0]	16.6 [15.0, 18.2]

Tree AGC was derived using the FIA's Component Ratio Method. Live trees were measured on 3,335 plots surveyed between 2010 and 2018. The estimates and 95% confidence intervals below were derived by resampling bootstrap sampling. Each bootstrap sample ($n = 10^4$) utilized measurements from 25% of inventory plots selected at random.

TABLE 5 | Removal and retention of aboveground carbon (AGC) in large trees (DBH ≥ 21 in) within the proposed Snow Basin project area.

Common name	Env.	Total AGC in large trees (Gg C)...		% of large tree AGC...	
		Removed	Retained	Removed	Retained
Grand fir	Cool moist	13.6 [12.0, 15.3]	24.5 [21.6, 27.5]	35.7 [35.7, 35.7]	64.3 [64.3, 64.3]
Grand fir	Warm dry	73.7 [65.0, 82.5]	73.7 [65.0, 82.5]	50.0 [50.0, 50.0]	50.0 [50.0, 50.0]
Douglas-fir + ponderosa pine + western larch	Warm dry or moist	43.8 [40.5, 47.3]	65.7 [60.8, 71.0]	40.0 [40.0, 40.0]	60.0 [60.0, 60.0]
Overall	–	131.3 [120.8, 141.5]	164.1 [151.8, 176.2]	44.4 [44.2, 44.7]	55.6 [55.3, 55.8]

Estimates of total AGC are given in gigagrams of carbon (1 Gg = 10^9 g). Values in brackets denote 95% confidence intervals derived by resampling ($n = 10^4$) inventory plots used to determine tree size class distributions.

removed by the project, while ponderosa pine, Douglas-fir, and western larch from warm-dry and warm-moist environments would comprise the remaining $\sim 33\%$ (~ 43.8 Gg C). The project would have the largest relative and absolute impact on grand fir occurring in warm-dry environments, where 50% (~ 73.7 Gg C) of large tree AGC would be removed. We obtained similar results albeit with substantially wider confidence intervals using only FIA inventory plots occurring in the Snow Basin project area (Supplementary Table 4).

DISCUSSION

Our study reveals the large carbon stocks associated with large-diameter trees in the region, and the potential for significant losses in AGC with large tree logging. The 21-inch rule was initially conceived to protect remaining late successional and old-growth forest and the native species that depend on these unique ecosystems for survival in forests east of the Cascades Crest in Oregon and Washington (Henjum et al., 1994). The carbon storage associated with the 21-inch rule on the six national forests is a significant co-benefit of this protective measure. Large trees (DBH ≥ 21 in or 53.3 cm) constitute $\sim 3\%$ of the total stems, but store $\sim 42\%$ ($\sim 45\%$ with CRM; Supplementary Table 3) of the AGC across the six eastside forests (Table 3). This finding highlights the important role of large trees in storing carbon in eastside forest ecosystems, and is consistent with previous findings on the disproportionately important role of large trees in the forest carbon cycle (Hudiburg

et al., 2009; Lutz et al., 2012, 2018; Stephenson et al., 2014). The sharp increase in carbon storage with increasing tree diameter (Figure 2) speaks to the importance of preserving mature and old large trees to keep this carbon stored in the forest ecosystem where it remains for centuries (Law et al., 2018; Lutz et al., 2018). Once trees attain large stature, each additional DBH increment results in a significant addition to the tree's total carbon stores, whereas small-diameter trees must effectively ramp up to size before the relationship between DBH and AGC results in significant carbon gains. Harvest of large-diameter trees, even focused on a specific species (e.g., grand fir) can remove upward of 50% of the large tree AGC from these ecosystems (Table 5).

There are substantial differences in wood density among the species examined that contributes to the observed differences in AGC at a given DBH. Wood density tends to be higher for ponderosa pine (~ 0.45 g cm $^{-3}$), Douglas-fir (~ 0.52 g cm $^{-3}$), and western larch (~ 0.56 g cm $^{-3}$) than for Engelmann spruce (~ 0.37 g cm $^{-3}$) or grand fir (~ 0.40 g cm $^{-3}$; Supplementary Table 2). An evaluation of the relationship between tree height and diameter suggests the observed differences in AGC are not driven by ponderosa pine, Douglas-fir and western larch tending to be taller at a given DBH than Engelmann spruce or grand fir. Although stem decay is common in mature and old grand fir, a synthesis shows no evidence of carbon consequences of heart rot in grand fir (Harmon et al., 2008). While we could not estimate heart rot loss in grand fir due to lack of sufficient data, heart rot respiration has been estimated for another species and it had a scant contribution to ecosystem respiration (Harmon et al., 2004).

Forestry practices exert significant controls on stand structure and forest carbon dynamics, and alterations of harvest practices can substantially alter carbon storage and accumulation (Masek et al., 2011; Turner et al., 2011; Krankina et al., 2012; Kauppi et al., 2015; Law et al., 2018). Generally, there is a negative relationship between harvest intensity and forest carbon stocks whereby as harvest intensity increases, forest carbon stocks decrease while emissions increase (Hudiburg et al., 2009; Mitchell et al., 2009; Simard et al., 2020). It can take centuries to reaccumulate forest carbon stocks reduced by harvest (Birdsey et al., 2006; McKinley et al., 2011). It would take 180 to 310 years to reach maximum biomass in the Blue Mountains and East Cascades, respectively, after harvest of the large trees (Hudiburg et al., 2009), too long to help reach climate mitigation targets in the next few decades. The amount of harvested carbon that remains stored in wood products is insufficient to offset the loss of carbon stored in the forest. If harvested, life cycle assessment shows that 65% of the wood harvested in Oregon over the past 115 years has been emitted to the atmosphere, 16% is in landfills and only 19% remains in wood products (Hudiburg et al., 2019). Thus, harvesting the large trees will increase, not decrease emissions and end centuries of long-term carbon storage in the forests.

The 21-inch rule has preserved the trees that store and accumulate a disproportionately large amount of carbon in these forests (**Figure 3** and **Table 3**). Of the ~3% of tree stems over 21 in DBH in the study area, about 81% of these are between 21 in and 30 in DBH and account for ~61% of the AGC in all large trees (**Supplementary Figure 2**). Trees over 30 inches DBH account for ~19% of the large tree stems and hold ~39% of AGC in large trees. These findings are similar to those reported by Stephenson et al. (2014) and emphasize the relative importance of the sub-30 in DBH large trees in the study area, and the value in allowing these trees to continue growing and replenish the stock of trees over 30 inches DBH that are rare on the landscape. This proforestation strategy is among the most rapid means for accumulating additional quantities of carbon in forests and out of the atmosphere (Moomaw et al., 2019).

The importance of forest carbon storage is now greatly amplified by a warming climate that must urgently be addressed with reductions in greenhouse gasses and natural climate solutions (IPCC, 2018; Ripple et al., 2020). The preponderance of forests in our study area have medium to high carbon accumulation potential and low future climatic vulnerability (**Figure 1**), which reinforces the value of protecting large trees to help abate our current trajectory toward massive global change (Fargione et al., 2018; Buotte et al., 2020). Rather than holding ecosystems to an idealized conception of the past using historical conditions as management targets, a good understanding of the environmental co-benefits associated with large tree protection is needed to inform management strategies that contribute toward solving humanity's most pressing Earth system challenges (Millar et al., 2007; Rockström et al., 2009; Barnosky et al., 2017; Ripple et al., 2020). Replacing large diameter trees with seedlings will create a major carbon loss to the atmosphere during harvest (Harris et al., 2016) and not achieve storage of comparable atmospheric carbon for the indefinite future.

Large grand fir trees (DBH ≥ 21 in or 53.3 cm) accounted for the lowest proportion of tree stems for the five species we evaluated (2%), possibly reflecting the ingrowth of young, small grand fir within portions of our study area (Merschel et al., 2014; Johnston et al., 2016). One of the main premises for the proposed removals of large-diameter grand fir trees is that they have increased over historic levels, especially in drier biophysical environments. However, our data does not indicate an uncharacteristic abundance of large grand fir trees across these forests. Given the recent history of high-grade logging that focused on large and old trees (Henjum et al., 1994; Rainville et al., 2008), historical abundances of large trees were much greater than today (Wales et al., 2007; Hagmann et al., 2013; Kauppi et al., 2015), and thus would have represented a larger fraction of aboveground biomass than currently found on these forests.

Interestingly, with respect to the overall representation of species abundance based on inventory plots (**Table 3**), the population of grand fir with 187,445 stems is a distant second to ponderosa pine with 286,970 stems, followed by Douglas-fir (100,021 stems), Engelmann spruce (31,375 stems), and western larch (30,708 stems). To the extent that current forest stand structure is skewed toward smaller diameter classes, ponderosa pine, grand fir, and Douglas-fir trees are notable for the number of trees on inventory plots.

It is important to note that a diameter limit that emphasized protection of carbon stores would ideally protect trees starting at a lower DBH limit (~12–15 inches). The 21-inch rule has provided significant leeway for fuels reduction and ecological restoration toward historical baselines to proceed in eastside forests. Ecological restoration treatments generally recommend giving protection to large and old trees, while reducing surface and ladder fuels, and accompanied by understory thinning treatments where appropriate and reintroduction of low-intensity fire at intervals (Allen et al., 2002; Brown et al., 2004; Agee and Skinner, 2005; Noss et al., 2006). From a forest carbon perspective, it would be prudent to restore an appropriate number of the approximately 97% of tree stems <21 inches DBH to replenish the stock of large trees over time (Lutz et al., 2018; Moomaw et al., 2019), while continuing to protect large trees for their carbon stores and the myriad other benefits that they provide. This approach would achieve the benefits of proforestation in the larger, most fire-resistant trees and reduction of fuel loads and stem density in the smaller diameter tree stems.

The Snow Basin project provides a case study for how amending the 21-inch rule to allow logging large trees could change management across the six eastside National Forests. Overall the Snow Basin project would have removed 44% or 131,000 metric tons of carbon stored in large trees. This is equivalent to ~0.75% of annual Oregon statewide carbon emissions, and this does not include carbon in tree roots or the AGC losses due to the removal of trees <21 in DBH. Removing nearly half of the carbon content from the large trees over 12,000 acres would create a carbon deficit in the live, dead, and soil carbon pools that will persist for many decades to the end of the century. However, left standing, these 43,445 large

trees continue to grow, sequestering more carbon into long-term stores (Stephenson et al., 2014; Law et al., 2018; Domke et al., 2020). Older trees (~100 years) are the next generation of old growth and already possess qualities associated with large, old trees, such as large canopies, deep root systems, and thick, fire-resistant bark.

Co-Benefits of Carbon, Habitat, Biodiversity, Water Availability, Resilience to Climate Extremes

High carbon conservation-priority forests support important components of biodiversity and are associated with increased water availability (McKinley et al., 2011; Perry and Jones, 2016; Berner et al., 2017; Law et al., 2018; Buotte et al., 2020). Large-diameter snags persist as standing snags for many years, providing valuable wildlife habitat, and account for a relatively high proportion of total snag biomass in temperate forests (Lutz et al., 2012). In PNW forests, large hollow trees, both alive and dead, are the most valuable for denning, shelter, roosting, and hunting by a wide range of animals (Bull et al., 2000; Rose et al., 2001). In the Interior Columbia River Basin, grand fir and western larch form the best hollow trees for wildlife uses (Rose et al., 2001). Downed hollow logs continue to serve as important hiding, denning, and foraging habitat on the forest floor (Bull et al., 1997; Bull et al., 2000). Large decaying wood influences basic ecosystem processes such as soil development and productivity, nutrient immobilization and mineralization, and nitrogen fixation (Harmon et al., 1986). Continuing to protect large trees in eastside forests provides the greatest benefit for carbon, habitat, and biodiversity.

Water availability and microclimatic buffering are also disproportionately affected by large trees and intact forests (Frey et al., 2016; Buotte et al., 2020). Forest canopies of the PNW buffer extremes of maximum temperature and vapor pressure deficit, with biologically beneficial consequences (Davis et al., 2019a). Removal of large trees quickly leads to a large increase in soil and canopy heating, which increases enough to impact photosynthesis (Kim et al., 2016), seedling survival, and regeneration (Kolb and Robberecht, 1996; Davis et al., 2019b). The climatic changes toward warmer and drier conditions expected in the next decades will likely increase forest stress and mortality (Allen et al., 2015). Eastside forests experienced hotter and drier conditions from 2003 to 2012 concentrated in the months of August and September, especially in drier forest type groups (i.e., ponderosa pine, juniper), whereas spring months (April–June) showed trends toward cooler and wetter conditions (Mildrexler et al., 2016). Projections suggest that proportionally, the largest changes in microclimatic buffering capacity will occur in lower elevation or dry forests, which currently have more limited buffering capacity (Davis et al., 2019a). In these drier regions, microclimatic buffering by forest canopies may create important microsites and refugia in a moisture-limited system (Meigs and Krawchuk, 2018). In an old growth ponderosa pine stand in eastern Oregon, ~35% of the total daily water used from the upper 2 m was replaced by hydraulic redistribution from deep soil by

deep-rooted larger trees in summer (Brooks et al., 2002). The bigger trees rarely reach 80% loss of hydraulic conductivity, and both mature pine and mesic Douglas-fir were better buffered from the effects of drought on photosynthesis compared with young pine (~20-year old) due to full root development and larger stem capacitance in older trees (Kwon et al., 2018). Redistribution of deep soil water can increase seedling survival during summer drought when young trees lack the root development to reach deep soil water (Brooks et al., 2002). While large tree composition may have shifted today relative to European settlement times, these large trees nonetheless continue to perform important functional attributes related to water and climate such as carbon storage, hydraulic redistribution, shielding the understory from direct solar radiation, and providing wildlife habitat. These functional attributes of large trees, irrespective of species, characterize ecosystems through thousands to millions of years (Barnosky et al., 2017), and are not quickly replaced.

In mesic forest environments, microclimatic buffering and transpirational cooling are amplified because sites with higher moisture availability are better able to shift energy to latent as opposed to sensible heat fluxes (Dai et al., 1999; Mildrexler et al., 2011). During midday in full sun the surface temperature of a closed canopy moist forest is warmer than the temperature beneath the canopy which is protected from direct solar radiation (Thomas, 2011). Microclimates in moist forests are strongly linked to their closed-canopy structure (Chen et al., 1999; Aussenac, 2000). Removal of the overstory creates canopy openings that increase solar radiation penetration resulting in increased drying of the understory vegetation and the forest floor, and a thermal response of rising land surface temperatures (Chen et al., 1993, 1999). This alteration in the subcanopy thermal regime changes atmospheric mixing between the ground, subcanopy, and canopy, which in turn modifies the microclimate condition of the affected stand. Microclimate modifications associated with forest harvesting are expected to be greatest in moist forests and may affect resilience to climate change and increase the risk of occurrence and severity of wildfires (Lindenmayer et al., 2009). Maintaining mesic microclimates may give undisturbed moist forests and the species they support some inherent resilience to climate change. Moreover, an evaluation of the effects of water limitations on forest carbon cycling in the eastern Cascade Mountains found that grand fir radial growth was not strongly associated with variability in temperature or water variability (Berner and Law, 2015). A lengthening of the growing season may increase productivity in high-elevation grand fir stands. The microclimatic buffering, current and future potential carbon stores, and intact nature of previously unlogged grand fir and Douglas-fir forest types are co-benefits of protecting large trees that need to be considered in future management decisions.

Potential Solutions

The consequences of reducing protection of large trees are significant reduction in forest carbon stores and their climate mitigation, impacts on habitat for animals including birds, and

resilience to a changing climate for decades to centuries to come. Given the rarity of large trees across the landscape, and their outsized role in *storing carbon removed from the atmosphere*, our findings call into question the value of removing large trees for forest modification in eastside forests.

If the 21-inch rule were retained on these lands, continued protection of the existing carbon stock would prevent large quantities of harvest related biogenic carbon from being released to the atmosphere. It is also essential to let a sufficient number of sub-21 inch trees remain to become additional large, effective carbon stores, and assure that carbon accumulation continues in these forests. Rather than weakening the 21-inch rule, we suggest strengthening this important measure and expanding large tree protections to other western United States public lands that have been adversely affected by a similar history of large-tree logging. Protecting and growing more large trees is the most effective near-term option for accumulating more carbon out of the atmosphere, and will benefit other ecosystem services as well.

Some public lands (local, state, and federal) could become part of a *designated reserve system* that includes intact forest landscapes, and carbon rich forests, that hold most of these large, older trees. This is an appropriate use of public lands because the services they provide including biodiversity, water retention, carbon accumulation and storage, and regional cooling by evapotranspiration serve the public interest, and promote sustainable economies that benefit from land protection and restoration.

The critical need to adapt to more wildfire in the west is congruent with protecting large-trees in fire-prone forests. Older forests experience lower fire-severity compared with younger, intensively managed forests, even during extreme weather conditions (Zald and Dunn, 2018). A shift in policy and management from restoring ecosystems based on historical baselines to adapting to changing fire regimes and from unsustainable defense of the wildland–urban interface to developing fire-adapted communities is needed (Schoennagel et al., 2017). Improved fire and forest management is part of the solution, but the most effective changes in terms of protection of people and property, will be near homes and on private property. Prioritizing federal fuel treatments around communities and creating better mechanisms for reducing fuels on private land can help reduce home loss and better protect communities (Moritz et al., 2014; Schoennagel et al., 2017). Given the natural role of fire in the West, managing more wild and prescribed fires with a range of severities will help reduce future wildfire threats and increase ecological benefits in many systems (Schoennagel et al., 2017).

Eastside forests are surrounded predominantly by rural communities. For production forests, lengthening the forest cycle will keep a larger amount of carbon in trees and soils rather than in the atmosphere (Law et al., 2018). Rather than coupling funding of forest restoration or community payments to logging large trees and disturbing older forests, new policies could be enacted that compensate rural communities for protecting large trees in older forests and some of the younger trees that will become large with their associated carbon stores.

To implement such policies, the amount to be paid to a community needs to be marginally greater than the revenue earned from cutting these large trees and the older forests in which they are located. Policies that provide compensation for setting aside reserves and individual trees with microhabitats are already in place in Europe (European Commission, 2015). Because this service benefits society as a whole and is an irreplaceable part of the natural climate solutions framework urgently needed for climate stabilization, the payment funds should come from the treasury since all citizens benefit from carbon accumulation by these trees and forests. With over half of Oregon's forest east of the Cascade Mountains crest, these forests are key to the State's climate mitigation and biodiversity conservation goals.

CONCLUSION

Conducting a quantitative assessment using empirical data has determined the large carbon stock that would be lost and the resulting climate consequences if these large trees are harvested. Research indicates that 2021 begins a pivotal decade for humanity to transition off of fossil-fuels (IPCC, 2018) and move “to achieve ... stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system.” (UNFCCC, 1992, Article 2). To meet net-zero carbon goals by 2050, it is estimated that reductions in net carbon emissions must be 7.6% per year over the decade of the 2020s (UNEP, 2019). This is most readily accomplished by reducing fossil fuel, bioenergy and industrial carbon dioxide emissions while simultaneously accumulating more carbon dioxide by protecting existing older forests that contain the largest share of carbon, and by allowing more forests to continue to accumulate carbon through proforestation (IUCN, 2020). Proforestation allows existing forests to continue growing without harvest or other management practices so that more trees can reach the large tree size that accumulates more carbon in the near and long term than do reforestation and afforestation (Moomaw et al., 2019). No additional land is required as is the case with afforestation, and proforestation is the lowest cost opportunity for reaching the zero net carbon goal by 2050. In fire-prone forests such as in our study area, a diameter limit strikes the balance between protecting the most fire-resistant trees that store the most carbon and allowing fuels reduction with reintroduction of fire in dry biophysical environments. Intact mesic forests are ideal locations for proforestation. Harvesting large trees will add very large amounts of biogenic carbon to the atmosphere (Harris et al., 2016), and make the net zero carbon goal difficult or impossible for Oregon to achieve. The young trees will never be able to recover and accumulate the amount of carbon that is in the growing and older forests during these next critical decades, and will only equal current levels a century or more from now.

Protecting large trees to help stabilize climate is critically important for managing forest ecosystems as social-ecological

systems. Knowledge of the disproportionately large amount of carbon stored in a small fraction of trees creates an opportunity to engage the public, decision makers, and forest managers in their importance as an integral part of the climate solution.

DATA AVAILABILITY STATEMENT

Publicly available datasets were analyzed in this study and are available through the FIA DataMart: https://apps.fs.usda.gov/fia/datamart/CSV/datamart_csv.html.

AUTHOR CONTRIBUTIONS

DM and LB conceived the ideas and designed the study with critical suggestions from BL and RB. DM and LB collected the data. LB processed the data and carried out the modeling. DM and LB analyzed the data and led the writing of the manuscript. BL, RB, and WM assisted with writing, and contributed critically to the drafts. All authors gave final approval for publication.

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SUPPLEMENTARY MATERIAL

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PERSPECTIVE

Protect large trees for climate mitigation, biodiversity, and forest resilience

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Abstract

Protecting the climate system requires urgently reducing carbon emissions to the atmosphere and increasing cumulative carbon stocks in natural systems. Recent studies confirm that large trees accumulate and store a disproportionate share of aboveground forest carbon. In the temperate forests of the western United States, a century of intensive logging drastically reduced large-trees and older forest, but some large trees remain. However, recent changes to large tree management policy on National Forest lands east of the Cascade Mountains crest in Oregon and southeastern Washington allows increased harvesting of large-diameter trees (≥ 53 cm or 21 inches) that account for just 3% of all stems, but hold 42% of total aboveground carbon. In this article, we describe synergies with protecting large trees for climate mitigation, biodiversity, and forest resilience goals to shift species composition, reduce fuel loads and stem density, and adapt to climatically driven increases in fire activity in eastern Oregon.

KEYWORDS

aboveground forest carbon, biodiversity, climate change, eastern Oregon, large trees

1 | INTRODUCTION

Society has a narrow window of opportunity left to avert catastrophic consequences from the intertwined climate and biodiversity crises (IPCC, 2022), and forests offer major solutions at the intersection of these urgent imperatives. Forests account for 92% of all terrestrial biomass globally (Pan et al., 2013), store about 45% of the total organic carbon on land in their biomass and soils (Bonan, 2008), and removed the equivalent of about 30% of fossil fuel emissions annually from 2009 to 2018, of which 44% was by temperate forests (Friedlingstein et al., 2019). Moreover, forests provide critical habitats to

more than half of all known plant and animal species on Earth (Gibson et al., 2011; Vié et al., 2009). As climate change increases and accelerates amplifying feedbacks, preserving species- and carbon-rich forests becomes ever more important, alongside a rapid transition to net-zero fossil fuel CO₂ emissions (Matthews et al., 2022).

Forests of the western US contain large stocks of carbon and remove significant quantities of CO₂ from the atmosphere to help protect climate, biodiversity, and water security (Buotte et al., 2020; Law et al., 2021). But how we manage these forests will play a large role in determining future outcomes (Fargione et al., 2018; Hudiburg et al., 2009; Law et al., 2018). Oregon stands

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out with the most forested area in the western USA, yet the lowest proportion of its forests protected (Law et al., 2021), and significant opportunities to create strategic forest reserves (Law, Berner, et al., 2022). About 80% of tree mortality in Oregon and Washington is attributed to harvest (Berner et al., 2017). In this article we provide insights from a recent study that quantified large tree carbon stocks in diverse forests of eastern Oregon (Mildrexler et al., 2020), and describe synergies with protecting disproportionately valuable large trees for biodiversity and climate mitigation, and forest resilience goals.

2 | LARGE TREES DOMINATE ABOVEGROUND CARBON STORAGE

Trees capture and store massive amounts of carbon, thus forests are an essential component of limiting global warming to 1.5–2°C (IPCC, 2018). However, trees are not all equal in their capacity to slow climate change in the coming critical decades. Large trees play an inordinately large role in removing carbon from the atmosphere and storing it in long-lived tissues (Figure 1; Lutz et al., 2012; Leverett et al., 2021). Globally, studies have found that about half the aboveground carbon is concentrated in a small proportion of large trees (1%–5% of total stems) (Lutz et al., 2018; McNicol et al., 2018). Because most global forests are well below their potential carbon stocks due to past and current land management practices, they could store twice the carbon than now (Erb et al., 2018). As large trees grow larger, small increases in diameter add a relatively large amount of volume and biomass (Mildrexler et al., 2020; Stephenson et al., 2014). Protecting existing forests with large trees and letting more forests mature and develop additional large trees is crucial for preventing carbon emissions and for continued

accumulation of carbon from the atmosphere in the coming decades (Birdsey et al., 2023; Law, Moomaw, et al., 2022; Moomaw et al., 2019).

2.1 | The 21-inch rule and carbon stocks

Forests in eastern Oregon and southeastern Washington are recovering from a century of intensive logging that eliminated much of the region's large trees by selective harvest of the largest, most robust trees including clear-cutting older forests. Nevertheless, the United States Forest Service (USFS) recently weakened protection for trees 21 inches diameter at breast height (DBH) and larger ("21-inch rule") across six national forests in this region. The 21-inch rule specifically applied to large-diameter trees on millions of acres of federal public lands. To assess the consequences of the loss of these trees it is essential to quantify large tree carbon stocks prior to changes in management actions. Mildrexler et al. (2020) evaluated carbon storage in large-diameter trees across the six national forests located east of the Cascade Crest in Oregon and Washington ("eastside forests") (Figure 2). Specifically, we quantified the relative contribution of large trees (≥ 21 inches DBH) to aboveground carbon (AGC) storage based on analysis of 636,520 trees on 3335 USFS Forest Inventory & Analysis (FIA) plots, and also assessed the carbon implications of relaxing the 21-inch rule. In these forests, large trees compose a small fraction of total stems (2.0% to 3.7% of all stems among five dominant tree species) yet hold 33% to 46% of total AGC stored by each species (Figure 3). The very largest trees, >30 inches DBH, held an even greater proportion of carbon (16.6%) relative to their small numbers (0.6%) demonstrating the importance of letting large trees grow larger and accumulate more carbon. Our research contributes to growing recognition that forests with large trees play a



FIGURE 1 Large-diameter grand fir (*Abies grandis*) in a mesic, mixed-conifer forest of northeast Oregon. These carbon-rich forests have a large cooling effect on maximum temperatures, provide thermal refugia for biodiversity including sensitive species, and are a high priority for protection. Large grand fir form the best hollow trees for wildlife (Rose et al., 2001).

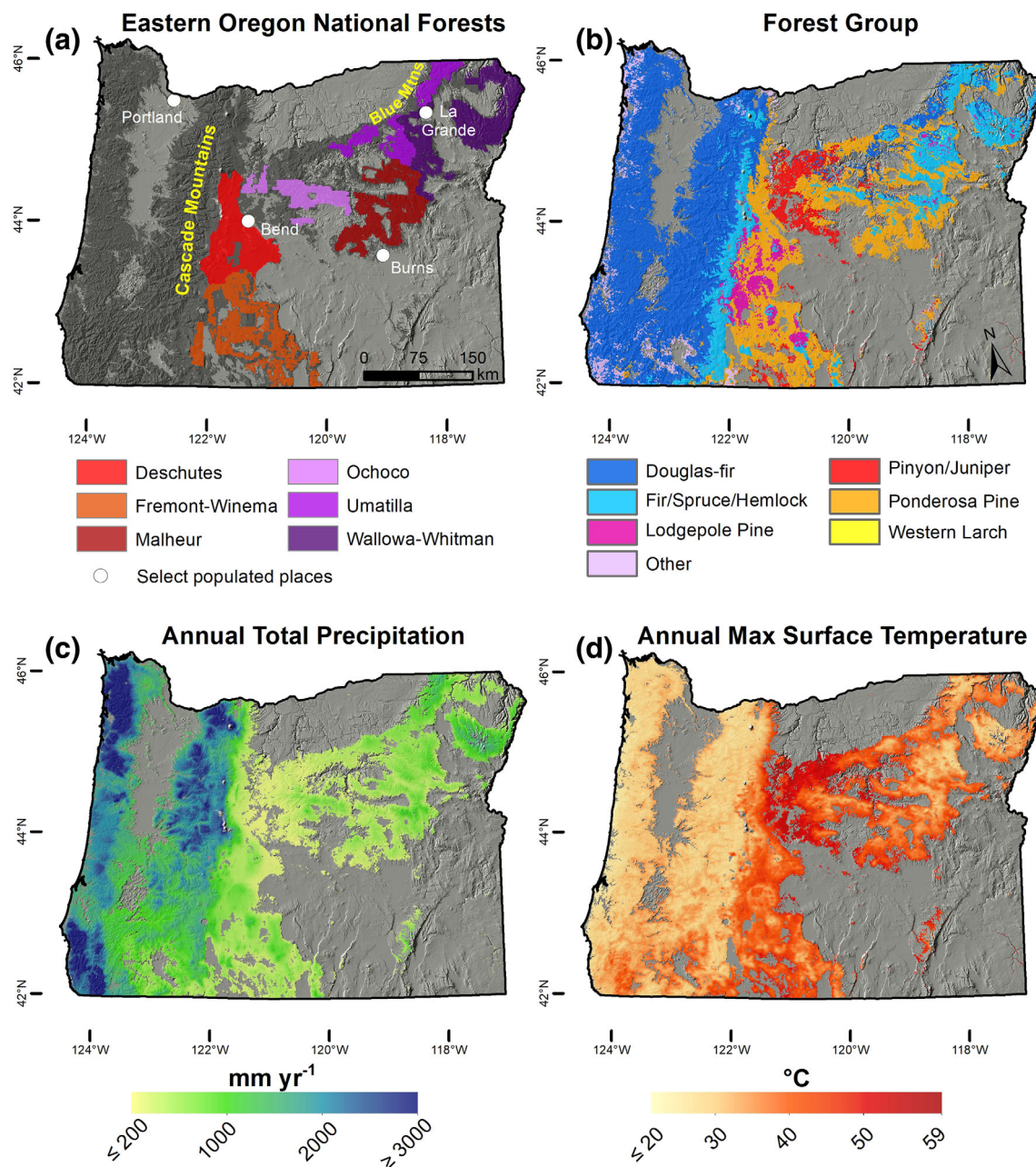


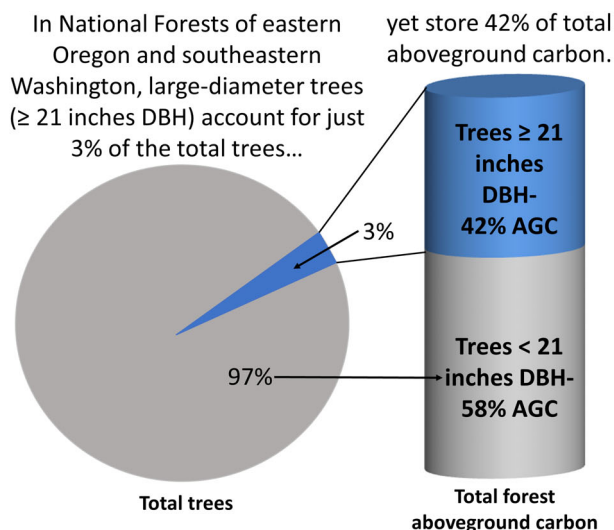
FIGURE 2 Forest and climatic diversity across the state of Oregon. (a) Extent of National Forests in Eastern Oregon. (Note a small portion of southeastern Washington included in our original study is not shown in this figure). (b) Distribution of forest groups. (c) Mean annual total precipitation from 1981 to 2010. (d) Mean annual maximum land surface temperature [LSTmax] from 2003 to 2020. Data sources include forest groups from Ruefenacht et al. (2008), precipitation climatology from Daly et al. (2008), and annual LSTmax derived using MODIS Aqua satellite data from Wan (2014).

very important role in climate mitigation now and in the near future (Lutz et al., 2018; Stephenson et al., 2014).

2.2 | A wildlife protection measure with a crucial carbon co-benefit

The 21-inch rule was implemented in the early 1990s as a habitat and species protection measure to recover large

tree structure and to protect remaining late successional and old-growth forest and associated species (e.g., American Marten, Northern Goshawk) (Bull et al., 2005; Bull & Hohmann, 1994; Henjum et al., 1994), similar to the Northwest Forest Plan (NWFP) that was implemented to ensure persistence of old-growth forest species and their habitat in the western portion of the region (FEMAT, 1993). The NWFP resulted in a strong carbon benefit for climate mitigation, in addition to protecting sensitive species and riparian



Species common name	% of total species trees ≥ 21 inches	% of total species AGC in trees ≥ 21 inches
Douglas-fir	3.7	37.5
Engelmann spruce	2.4	34.7
Grand fir	2.0	38.4
Ponderosa pine	3.7	45.8
Western larch	2.8	33.3
Overall	3.1	42.2

FIGURE 3 Percentage of all tree stems above and below the 21-inch DBH threshold and their total aboveground carbon (AGC) stores overall, and for five dominant tree species, evaluated based on measurements from USFS inventory plots located in the six eastside national forests.

systems (Turner et al., 2011). Mildrexler et al. (2020) showed that carbon storage associated with the 21-inch rule on the six eastside national forests is a significant co-benefit of this protective measure (Pörtner et al., 2021).

Detailed analysis of stand structure and carbon impacts is essential for science-based decision-making about large-tree forest management policies because such policies affect many different values and services provided by forests (Davis et al., 2019; Teich et al., 2022), including consequences on greenhouse gas emissions and for increasing atmospheric carbon removal and accumulation in forests (Fargione et al., 2018; Griscom et al., 2017). Moreover, large live trees eventually create large-diameter snags and downed wood that continue to store carbon for decades and contribute directly to biodiversity by providing unique specialized habitats such as hollow trees and logs, and micro-environments (Lutz et al., 2021; Rose et al., 2001). However, the USFS General Technical Report on the 21-inch rule did not assess large tree carbon stocks (Hessburg et al., 2020), even though storage and accumulation of carbon in forests is an increasing priority in National Forests (Depro et al., 2008; Dilling et al., 2013; Dugan et al., 2017). Consequently, quantitative assessments of management effects on both forest carbon and biodiversity are important, including assessment of the effects of long-standing rules before they are eliminated or weakened (Mildrexler et al., 2020).

The 21-inch rule has since been amended. Grand fir (DBH ≥ 53 cm and < 150 years) has lost protections in stands not designated as Late and Old Structure, and protections for all tree species have been significantly weakened from a standard to a guideline (USDA, 2021). This represents a major shift in management of large trees across the region, highlighting escalating tradeoffs between goals for carbon sequestration to mitigate climate change, and efforts to increase the pace, scale, and intensity of cutting across national forest lands. The potential impacts of removal of large grand fir on wildfire

are unclear, although a trait-based approach to assess fire resistance found that the grand fir forest type had the second highest fire resistance score, and one of the lowest fire severity values among forest types of the Inland Northwest USA (Moris et al., 2022).

3 | ARE LARGE GRAND FIR OUTCOMPETING LARGE PONDEROSA PINE AND LARCH?

The key rationale for amending the 21-inch rule is that increased cutting of large-diameter fir trees (≥ 53 cm DBH and < 150 years) is needed to facilitate the conservation and recruitment of early-seral, shade-intolerant old ponderosa pine (*Pinus ponderosa*) and western larch (*Larix occidentalis*) by reducing competition from shade-tolerant large grand fir (*Abies grandis*) (USDA, 2021). Previous studies have looked at tree age-size relationships (Merschel et al., 2019; Perry et al., 2004), large tree numbers, and changes in basal area (Hessburg et al., 2022), but there has been no spatial analysis of close-range comingling of large-diameter tree species across the six national forests covered by the 21-inch rule. This is an important consideration because the competitive interaction among large-diameter trees, and their protection under the 21-inch rule, should not be conflated with small tree dynamics and common dry forest restoration strategies to reduce small tree density and favor retention of early-seral species.

We therefore examined how often large trees (≥ 53 cm DBH) of these species co-mingle on USFS FIA plots (~ 1 acre) across the same six eastside national forests where we previously examined carbon storage by large trees (Mildrexler et al., 2020). Drawing on the same USFS FIA measurements as our prior study, we found that large ponderosa pine, grand fir, and western larch were present

TABLE 1 Coverage, mean annual precipitation from 1981 to 2010, and mean annual maximum land surface temperature from 2003 to 2020 for the major FTG's within the six national forests, standard deviations in parenthesis (excludes lands in Washington and Idaho).

Forest type group	Area (km ²)	Mean annual precipitation		Mean annual maximum LST	
		mm	in	°C	°F
Douglas-fir	2372	679.9 (187.3)	26.8 (7.4)	37.8 (3.5)	100.1 (6.3)
Fir/Spruce/Hemlock	12,224	974.3 (369.5)	38.4 (14.5)	33.2 (2.8)	91.8 (5.0)
Lodgepole pine	3573	822.0 (274.6)	32.4 (10.8)	37.2 (3.8)	99.0 (6.8)
Pinyon/Juniper	634	329.5 (96.0)	13.0 (3.8)	50.4 (3.7)	122.7 (6.6)
Ponderosa pine	18,514	548.3 (139.2)	21.6 (5.5)	39.8 (3.5)	103.6 (6.3)
Western larch	47	746.9 (78.9)	29.4 (3.1)	31.5 (2.4)	88.7 (4.3)

on 56%, 18%, and 7% of all plots ($n = 3335$). Large ponderosa pine co-mingle with large grand fir about 14% of the time (259 plots), leaving 86% of plots with large ponderosa pine without large grand fir (1616 plots). Similarly, large western larch co-mingle with large grand fir about 56% of the time. Large ponderosa pine and grand fir are found together on only 8% of all plots in the region, while large larch and grand fir are found together on only 4% of all plots in the region. In other words, large ponderosa pine are by far the most common tree species found in these six National Forests and infrequently co-mingle with large grand fir at the FIA plot scale, whereas large western larch are far less common and co-mingle with large grand fir about half the time, which is expected since these species occupy similar environmental settings that receive more moisture (Table 1, Johnson & Clausnitzer, 1992).

The relative prevalence of large ponderosa pine in eastside forests is good for climate resilience given that large-diameter pines are exceptionally drought and fire-resistant trees (Irvine et al., 2004; Irvine et al., 2007). In the drought-prone region of central Oregon, mature and old ponderosa pine forests had 60% to 85% higher seasonal gross photosynthesis than a young forest (Irvine et al., 2004). Large ponderosa pine trees experienced only 34% mortality in moderate severity fire, and accounted for 91% of post-fire stemwood production, while small trees experienced 82% mortality (Irvine et al., 2007).

Across the entirety of all six national forests large grand fir represent 2% of the total species population, a proportion slightly lower, but roughly on par with other dominant species (Figure 3, Mildrexler et al., 2020). It is not uncommon for grand fir to reach 250 to 300 years of age (Howard & Aleksoff, 2000). Thus, large grand fir ≥ 53 cm DBH and <150 years of age can continue growing and play an important role in storing and accumulating carbon from the atmosphere to help abate the climate crisis.

Synergy: Enhancing forest resilience does not necessitate widespread cutting of any large-diameter tree species. Favoring early-seral species can be achieved with a

focus on smaller trees and restoring surface fire, while retaining the existing large tree population.

4 | LARGE TREES, VULNERABILITY, STAND DYNAMICS, AND THE CARBON COST OF THINNING

As eastside forests recover from a century of intensive logging, it is important to distinguish between the shift of AGC stocks into small-diameter, fire-sensitive trees and the retention of a small fraction of the largest more fire-resistant trees that store disproportionately massive amounts of carbon. Small tree carbon stores are relatively unstable and at risk of loss to fire and drought, whereas large tree carbon stores are relatively stable and resistant (Hurteau et al., 2019). Physiological-based studies in ponderosa pine forests of Oregon have found that small trees are most vulnerable during drought relative to mature trees that have reached full root, bark and canopy development and respond to climate variability better than smaller trees (Domec et al., 2004; Irvine et al., 2004; Vickers et al., 2012). Buotte et al. (2019, 2020) identified forests in the western U.S. with high potential carbon accumulation and low vulnerability to future drought and fire using the Community Land Model and two climate models with high CO₂ emissions (RCP8.5), and species-specific traits capturing sensitivity of different species to water limitations and to drought and fire. The Eastern Cascades and Blue Mountains contain substantial area with opportunity to enhance forest carbon in large trees (Buotte et al., 2020; Law et al., 2018).

In dry forests historically maintained by a frequent, low-severity fire regime, the priority ought to be restoring the process of periodic surface fire. Prescribed fires create landscape heterogeneity, reduce surface and ladder fuels, lower stand density, and confer drought resistance to surviving trees (Knapp & Keeley, 2006; van Mantgem et al., 2016). In

these forests prescribed fire can modulate future fire activity (Schoennagel et al., 2017), and favor early-seral species such as ponderosa pine, western larch and Douglas-fir. Large trees of these species and grand fir are resilient to prescribed fire because they have attained the thick bark that provides resistance to low- and moderate-severity fire (Howard & Aleksoff, 2000; Pellegrini et al., 2017).

Thinning also has an inherent carbon cost that increases as larger trees are harvested, thereby putting thinning of larger trees in conflict with carbon goals because it takes so long to replace the harvested biomass (James et al., 2018; Law & Harmon, 2011). The underlying principle for these losses is the negative relationship between harvest intensity and forest carbon stocks whereby as harvest intensity increases, forest carbon stocks decrease and emissions increase (Hudiburg et al., 2009; Mitchell et al., 2009; Simard et al., 2020). Claims that carbon stores will be “stabilized” by increasing harvest of large-diameter trees that store and accumulate the most carbon (Johnston et al., 2021) are inconsistent with basic science on thinning (Zhou et al., 2013) and the carbon cycle (Campbell et al., 2012; Law et al., 2018). These claims ignore the large amounts of CO₂ rapidly released to the atmosphere following harvest (Hudiburg et al., 2019), and that large trees cannot be replaced in short timeframes. It can take centuries to reaccumulate forest carbon stocks reduced by harvest of large trees (Birdsey et al., 2006).

Even thinning smaller trees involves substantial carbon tradeoffs in the short term, a 30%–40% reduction in live tree carbon stores in some forests (Krofcheck et al., 2017; North et al., 2009). To minimize reductions in carbon stocks and emissions, focus on removing smaller-sized trees, restoring surface fire, and managed wildfire in favorable weather conditions (Mitchell et al., 2009; Stenzel et al., 2021).

Synergy: Small trees are more relevant to drought and fire vulnerability and store less carbon, whereas large trees are more resilient to fire and drought and are the highest priority for keeping carbon in the forest.

5 | DIVERSE CLIMATE REGIMES AND FOREST TYPES REDUCE CLIMATIC EXTREMES

It is critical to accurately represent the diversity of climatic regimes and forest types in decisions affecting large tree management because large trees play unique roles in ecosystem water and energy cycles, and these biophysical effects can promote local climate stability by reducing extreme temperatures in all seasons and times of day (Lawrence et al., 2022). Forest modulation of summer maximum temperature is especially powerful (Mildrexler et al., 2018) and can partly offset the projected increases in temperature due

to anthropogenic climate change (de Frenne et al., 2019). With heatwave frequency and severity projected to increase, the capacity of forests to buffer against temperature extremes and provide refugia is increasingly recognized as important to sustaining biodiversity in a warming world (Davis et al., 2019; de Frenne et al., 2019).

The six eastside national forests affected by the 21-inch rule cover a region of pronounced geographic and climatic variation and associated forest types (Figure 2; Johnson & Clausnitzer, 1992; Wyatt, 2017). Mean annual precipitation varied from 484 to 571 mm per year on the Ochoco and Malheur National Forests, to ~800 mm per year on the Deschutes, Umatilla and Wallowa Whitman National Forests (Mildrexler et al., 2020). We further examined the climatic regimes of the major forest types across the six national forests using satellite-based annual maximum land surface temperature (LST_{max}) and mean annual precipitation datasets (Figure 2D, Table 1). Our analysis shows that ponderosa pine and fir/spruce/hemlock types cover the largest area on the six national forests. The fir/spruce/hemlock type received the most total precipitation (~974 mm yr⁻¹) and had the second lowest annual LST_{max} (33.2°C). Average LST_{max} for the fir/spruce/hemlock type was 6.6°C (~12 °F) cooler than ponderosa pine (39.8°C), and 4.6°C (~8 °F) cooler than Douglas-fir (37.8°C). The pinyon juniper type had the lowest total precipitation (329 mm yr⁻¹) and highest annual LST_{max} (50.4°C) due to low canopy cover and heating of the dry surface during summer. These results show the region's pronounced variability in hydrologic and forest thermal regimes and highlight the thermal offsetting capacity of closed-canopied mesic forest systems. These valuable ecosystem services can be severely degraded by industrial logging (Lindenmayer et al., 2009).

Synergy: Mature and old mesic forests are a high priority for protection, provide crucial biophysical benefits on climate, including a large cooling effect on maximum temperatures regulating climate extremes and protecting biodiversity. Large grand fir is essential to this ecology.

6 | CONCLUSIONS

The 21-inch rule is an excellent example of a policy initiated for wildlife and habitat protection that has also provided significant climate mitigation values across extensive forests of the PNW Region. The rule resulted in a valuable resource of large-diameter trees in a landscape that remains below historical levels for large live trees and large snags due to historical logging (Bell et al., 2021). We have described synergies between protecting these disproportionately valuable large trees and forest resilience goals, providing common potential solutions for these urgent challenges.

Inland PNW forests can make a significant contribution to climate mitigation goals by protecting and enhancing carbon stores in large trees that accumulate and store the most carbon and are much more resistant to fire and drought than small trees, even when the current status of ecosystems has changed from historical baselines. Climate science makes clear that we do not have time to wait for regrowth after logging to accomplish these important ecosystem services (IPCC, 2022).

AUTHOR CONTRIBUTIONS

David J. Mildrexler led the writing. David J. Mildrexler and Logan T. Berner performed data analysis, investigation, and visualization. All authors commented on drafts, and assisted with writing, review and editing. All authors gave final approval for publication.

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interest.

DATA AVAILABILITY STATEMENT

This study relied on data from prior studies that are publicly available: Forest Inventory Data from the United States Forest Service (<https://apps.fs.usda.gov/fia/datamart/datamart.html>), MODIS Land Surface Temperature data from the Land Processes Distributed Active Archive Center accessed through Google Earth Engine, and gridded precipitation climatologies from Oregon State University PRISM Climate Group (<https://prism.oregonstate.edu/normals/>).

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**UNITED STATES DISTRICT COURT
DISTRICT OF OREGON
PENDLETON DIVISION**

**GREATER HELLS CANYON COUNCIL,
OREGON WILD, CENTRAL OREGON
LANDWATCH, SIERRA CLUB, GREAT
OLD BROADS FOR WILDERNESS, and
WILDEARTH GUARDIANS,**

Plaintiffs,

v.

HOMER WILKES, in his official capacity as
Undersecretary for Natural Resources and
Environment; **GLENN CASAMASSA**, in his
capacity as Regional Forester for Region 6; and
the **UNITED STATES FOREST SERVICE**,

Defendants,

**AMERICAN FOREST
RESOURCE COUNCIL**, and
**EASTERN OREGON COUNTIES
ASSOCIATION**,

Defendants-Intervenors.

Case No. 2:22-cv-00859-HL

**DECLARATION OF
DR. BEVERLY LAW**

I, Beverly E. Law, declare as follows:

INTRODUCTION

1. My name is Beverly E. Law. I reside in Corvallis, Oregon. I have personal knowledge of the matters stated herein and if called as a witness I would and could truthfully testify thereto.
2. I am a Professor Emeritus of Global Change Biology and Terrestrial System Science in the Department of Forest Ecosystems and Society at Oregon State University. I have been a tenured professor at Oregon State University since 2003. I make this declaration in my personal capacity.
3. I am a fellow at the American Geophysical Union and an Aldo Leopold Fellow.
4. My research focuses on the effects of climate change, fire, and management on ecosystem carbon and water cycling, addressing issues such as vulnerability of forests to drought-related mortality, and ecological implications of human actions intended to minimize drought and fire effects.
5. I am an author or co-author of over 240 peer-reviewed publications. I have led teams that have explored our understanding of the interactions between ecosystems and the atmosphere, and regional analysis of carbon and water cycling and potential vulnerability of forests under future climate.
6. I served as the Science Chair of the AmeriFlux network of 100+ land-atmosphere research sites (2000-2011), Chair of the Global Terrestrial Observing System – Terrestrial Carbon Observations (GTOS-TCO), and I served on the National Research Council committees “Verifying Greenhouse Gas Emissions: Methods to Support International Climate Agreements,” and “Air Quality Management in the U.S.”

7. I have testified before U.S. House and Senate Committees on a variety of issues, such as "Managing forests in response to climate change including for natural resource adaptation and carbon sequestration," and "Monitoring, measurement, and verification of greenhouse gas emissions."
8. My curriculum vitae is attached as **Exhibit 1**.
9. I have reviewed the Proposed Amicus Curiae Brief and other submissions filed by James Johnston in this case. James Johnston is technically an Assistant Professor, Senior Research at Oregon State University. This means he is fixed-term non-tenure-track and brings in his own salary from grants and contracts, which I understand comes in part from grants and monies for work that he does for the United States Forest Service.
10. I make this declaration to rebut the claim made by James Johnston that "no meaningful controversy among scientists exists," regarding the historical and present conditions of the forests at issue in the Eastside Screens Amendment and the environmental effects of the Screens Amendment. **ECF 58**.
11. Although James Johnston and a handful of other scientists have signed a letter to the Court providing their opinions based on a selection of studies, I write to inform the court that Dr. Johnston's claim that there is a "scientific consensus" does not accurately reflect the state of the science generally on these topics, nor specifically the best available science and field verification of the condition of the forests at issue in this case.
12. There has been extensive scientific debate in publications regarding the historical and current state of the eastside forests. See for example, the debate between Hagmann et al. (2013) and Baker and Hanson (2017) discussing the use of different methodologies to

arrive at different conclusions regarding historical forest composition and fire severity published in peer-reviewed publications.

13. Indeed, James Johnston fails to point out in his letter that, during the public process for the Screens Amendment, I and over 150 other scientists submitted a letter to the Forest Service requesting that the 21” rule protecting large trees remain in effect. **See AR 49237.** In the letter conveyed our opposition and the controversy:

The Forest Service is amending its forest plans on >9 million acres (3.6 million ha) in eastern Oregon to lift protections for large trees that have been in place for over two decades. The planned amendment threatens to set back forest recovery that has only recently followed a century of clear-felling and high-grade logging. Protections were put into place in 1994 targeting forests that needed time to recover depleted large-tree populations particularly for wildlife species that are highly dependent on that now scarce element in these Eastside forest. Although ***removing protections for large trees is highly controversial from a scientific perspective***, the Forest Service is rushing forward without adequately analyzing the impacts of the proposal on wildlife habitat, aquatic ecosystems, hydrological cycles and carbon values. (emphasis added)

14. I have first-hand knowledge of the administrative process for the Screens Amendment because I was an invited panelist at the Forest Service’s Science Forum on May 11, 2020 (Available at: <https://vimeo.com/727176378/bf53flea42>, starting at 1:43:00). The purpose of that panel was to present scientific information. I delivered information on our peer-reviewed publications during my presentation that directly contradicts James Johnston’s assertion that there is a “scientific consensus” on the issues presented in this case. That testimony includes but is not limited to the rarity of large trees in eastside forests and their significant contribution to forest carbon and biodiversity, which is a priority for protection nationally and internationally (Law et al. 2021, 2018, 2001, Hudiburg et al., 2013, Buotte et al. 2019, 2020). See AR 32289-32294, which is the powerpoint I delivered.

15. I also submitted testimony to the United States Forest Service on the proposed elimination of protections for trees 21 inches and greater, and I have attached that as **Exhibit 2** to this declaration. In that letter I relayed the findings of over 30 years of study on Eastside forests, and presented findings which directly contradict James Johnston's claims, namely:
 - Forests with large trees and closed canopies create a microclimate that buffers climate extremes. ***Removal of the overstory creates canopy openings that increase solar radiation penetration to the forest floor, resulting in increased drying and heating of the understory and litter), which increases risk of fire ignitions and higher wildfire severity*** (Law et al. 2001, Burnett et al. 2019, Ma et al. 2010).
 - Near-future vulnerability to fire is generally low on the eastside, with the exception of a few areas (Buotte et al. 2019). ***The large trees are less vulnerable to fire, and removing them will not reduce fire occurrence.*** Contrary to common belief, 40-60% of fire emissions are from litter and duff in all fire severities, and 0% of large conifer bole biomass combusts in all severities, and up to 5% of large tree branch biomass combusts, whereas smaller trees have much higher combustion in all burn severities (Campbell et al. 2007). For context, harvest-related emissions in Oregon are 10 times wildfire emissions (Hudiburg et al. 2019).
16. In his amicus letter and his recent article, Johnston suggested without evidence that protecting old-growth ponderosa pine and western larch from large grand fir *requires* cutting large diameter trees across the landscape (Johnston et al. 2021). According to my field research, he appears to have overestimated the co-occurrence of these species.
17. Our research team intensively analyzed measurements of 636,520 trees on 3,335 USFS inventory plots (~1 acre plots) to determine the percentage of large trees on the same eastern Oregon and Washington landscape that is governed by the screens and is the subject of this case. We quantified aboveground biomass carbon of those trees, and the co-occurrence of grand fir species with those trees. We found that large-diameter trees ($\geq$

53 cm or 21 inches) account for just 3% of all stems, but hold 42% of total aboveground carbon (Mildrexler et al. 2020).

18. Contrary to James Johnston's assertions, we found that across all FIA plots in the region, large ponderosa pine and large grand fir are found together on only 8% of all plots; large larch and large grand fir are found on only 4% of all plots. Large ponderosa pine are the most common tree species (56% of plots) on all six national forests in the region and infrequently co-mingle with large grand fir (14% of the time) at the FIA plot scale. Large larch trees are far less common (7% of total plots) and co-mingle with large grand fir about half the time (56% of the time) (Mildrexler et al. in press).
19. Therefore, across the entire study area, only 8% of all plots have the large-tree co-occurrence of ponderosa pine and grand fir that the Forest Service says is the primary reason for weakening the 21-inch rule. Whether or not the Forest Service's determination that grand fir is encroaching on ponderosa pine to their detriment holds true, the fact that 92% of the forested landscape does not fit their rationale is a major finding, and is at the center of the scientific controversy.
20. Stated plainly, the Forest Service, and their consultant James Johnston, are making a mountain out of a molehill. We cannot know where wildfire will burn, yet they have eliminated protections for large trees based on a gross overstatement of the risk so that the Forest Service will be able to authorize the preemptive killing and removal of large grand fir trees across the entire landscape. James Johnston's theory is based on questionable assumptions and selective presentation of data that preemptive actions will protect large ponderosa pine trees.
21. To the further underscore the controversy, decades of research indicate that fire behavior

is heavily driven by weather and climate, not stand density (e.g., Abatzoglou et al. 2016, Bradley et al. 2016, Povak et al. 2020, among many others), and it has long been known that, in mild and moderate fire weather, one can conduct prescribed fire or managed wildfire to achieve lower-severity fire effects, if that is the goal.

22. All of that is not to say there are not some areas of agreement. In particular, Johnston states that there is “little or no meaningful debate or controversy in the scientific literature about the importance of large and old trees,” and recognizes “significant variability in historical forest structure and composition.” ECF 58 at 11. True, but such statements distract from the core issue.
23. The controversy lies in the Forest Service’s decision to remove protections for large trees across eight million acres of national forestland, replacing it with an uncertain guideline—regardless of historical or current forest structure and composition, contrary to the findings and recommendations of many scientists, and at the expense of many other important values including carbon storage. When applied across this vast landscape, the justifications for and effects of blanket removal of large-tree protections are scientifically controversial.

Pursuant to 28 U.S.C. § 1746, I declare under penalty of perjury that the foregoing is true and correct to the best of my knowledge.

Executed this 24th day of March, 2023 in Corvallis, Oregon

BE Law

Dr. Beverly Law

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AN OPEN LETTER FROM NATIONAL & INTERNATIONAL SCIENTISTS ON THE IRREPLACEABLE IMPORTANCE OF LARGE TREES AND MATURE/OLD-GROWTH FORESTS OF ALL TYPES TO HELP STEM THE BIODIVERSITY AND CLIMATE CRISES

“As a researcher of old-growth forests and old trees I am strongly convinced that every possible old tree should be preserved if possible. We have lost too many of those living witnesses of the past. Old trees and old-growth forests are the "island" containing the genes of resilience, which is increasingly important in these times of climate extremes and forecasts of warming and more extreme climate.”

Dr. M Panayotov, University of Forestry, Bulgaria

As scientists with expertise in forest ecosystems, climate change, and natural resources, we fully support global efforts to protect large trees and old forests (mature and old growth) from logging along with restoring logged forests as natural climate solutions¹. In response to President Joe Biden’s [Executive Order 14072](#) on forests and climate change, federal agencies are in the process of formulating conservation policies for large trees and old forests in the United States via national rulemaking. We urge conservation groups, decision makers, and land managers to maximize protection of **large trees of all species as well as mature and old-growth forests** as the nation’s most effective “[climate-smart forestry](#)” approach. Simply put, there is no substitute for their outsized role, and no ecological reason to log large trees and old forests. Many of the current and proposed forest management actions in the US are inconsistent with climate goals and global efforts to protect 30% to 50% of lands for myriad ecosystem and biodiversity benefits².

Large, old trees are among the biggest terrestrial organisms on the planet but are on the decline globally mainly due to logging and development³. These trees and old forests that harbor them are the most critical terrestrial sinks and easily scalable climate mitigation opportunities at our disposal. They are also essential to soil integrity and complexity, water circulation, stable microclimates, nutrient exchange, and wildlife habitat, regardless of whether the trees are alive or dead. Protecting them enables irreplaceable ecosystem services and biodiversity benefits, including habitat for hundreds of imperiled species⁴. Importantly, while large trees make up just 1% (globally)⁵ to 3% (eastern Oregon)⁶ of the overall tree density in older forests, they contain 50% to 42%, respectively, of the above ground carbon in forests. As they age, large trees and old forests continue to sequester and accumulate massive amounts of atmospheric carbon in biomass and soils⁷. From a climate and biodiversity perspective, it is essential not to log the large trees and old forests. Broad-scale thinning (e.g., ecoregions, regions) to reduce fire risk or severity results in more carbon emissions than fire, creating a long-term carbon deficit that undermines climate goals⁸.

¹Lindenmayer et al. 2012. Global decline in large old trees. *Science* Vol 338:1305-1306. 10.1126/science.1231070. Mackey et al. 2015. Policy options for the world’s primary forests in multilateral environmental agreements *Conservation Letters* 8:139-147 doi: 10.1111/conl.12120. Griscom et al. 2017. Natural climate solutions. *PNAS* 114 pnas.org/cgi/doi/10.1073/pnas.1710465114. Moomaw et al. 2019. Intact forests in the United States: proforestation mitigates climate change and serves the greatest good. *Frontiers in Forests and Global Change* <https://doi.org/10.3389/ffgc.2019.00027>. Law et al. 2022. Creating strategic reserves to protect forest carbon and reduce biodiversity losses in the United States. *Land* 2022, 11, 721. <https://doi.org/10.3390/land11050721>. DellaSala et al. 2022. Mature and old-growth forests contribute to large-scale conservation targets in the conterminous United States. *Frontiers in Forests and Global Change* <https://doi.org/10.3389/ffgc.2022.979528>. Faison et al. 2023. The importance of natural forest stewardship in adaptation planning in the United States. *Conservation Science and Practice* 2023;e12935. <https://doi.org/10.1111/csp2.12935>.

²Law, B.E. et al. 2022. *Ibid*

³Lindenmayer et al. 2012. *Ibid*. Lutz et al 2018. Global importance of large diameter trees. *Global Ecol Biogeogr*:1-16.

⁴Brandt et al. 2014. Multi-functionality and biodiversity: ecosystem services in temperate rainforests of the Pacific Northwest, USA. *Biol. Cons.*169:362-71. <http://dx.doi.org/10.1016/j.biocon.2013.12.003>. Buottie et al. 2020. Carbon sequestration and biodiversity co-benefits of preserving forests in the western United States. *Ecol. Applic.* 30(2), 2020, e02039. Morgan et al. 2021. Capturing multiple forest ecosystem services for just benefit sharing. The basket of benefits approach. *Ecos. Services* <https://doi.org/10.1016/j.ecoser.2022.10142>. Law et al. 2022. *Ibid*. DellaSala et al. 2022. *Ibid*.

⁵Lutz et al. 2018. *Ibid*

⁶Mildrexler et al. 2020. *Ibid*

⁷Stephenson et al. 2014. *Ibid*. Birdsey et al. 2023. Assessing carbon stocks and accumulation of mature forests and large trees in U.S. federal forests. *Frontiers in Forests and Global Change* <https://doi.org/10.3389/ffgc.2022.1074508>.

⁸Law et al. 2022. *Ibid*

In the US, most large trees and old forests were eliminated decades ago following European colonization and expansion. What remains is mostly on federal lands⁹ that are vital to restoring forest ecosystems from the coastal rainforests of Alaska and the endemic *Sequoia giganteum* groves of California to mixed hardwoods, pines, oaks, cypress and many other forest types across the country. We note this includes protecting forests affected by insect outbreaks and characterized by **frequent and infrequent fire return intervals and mixed fire severities**. Notably, large trees and old forests tend to be the most fire and drought resistant ecosystem elements, serving as critically important fire¹⁰ and climate refugia¹¹. They also contain unique genetic adaptations for conferring resilience to natural disturbances and climate change¹². Increased forest protections would not result in greater fire risks¹³ but would instead allow forest managers to focus limited resources on previously logged and degraded forests that are more likely to burn and be located near residences. However, large trees have often been inappropriately targeted in “fuel reduction” by land managers,¹⁴ despite myriad benefits and relative fire resistance and resilience.

In some regions (e.g., eastern Cascades of Oregon/Washington, Appalachia, northeast, Great Lakes), large tree densities and older forests have increased due to restrictions on logging and fire suppression policies but still remain far below historical levels due to logging writ-large¹⁵. Concerns have been raised by federal managers in eastern hardwood forests that despite the rarity of old forests in the landscape, they are replacing young forests and now need intensive logging to reset succession even though such practices do not mimic complex early seral forests generated by natural disturbances¹⁶. Moreover, trees in eastern Oregon’s Cascades have increased in density and size due largely to fire suppression, potentially competing for limited resources. However, large trees in these forests now represent the remaining old-forest cohort that have developed fire resistant properties overtime by dropping lower branches and increasing bark thickness. A recent study showed that despite a claim that there is substantial encroachment and competition from large grand fir (*Abies grandis*) on ponderosa pine (*Pinus ponderosa*) and western larch (*Larix occidentalis*) in six National Forests in eastern Oregon, large pine and large fir are found together on only 8% of all plots (n=1616) sampled, while large grand fir and large larch co-mingled on only 4%. In other words, large pines are by far the most common tree and infrequently co-mingle with large grand firs at the plot scale. **There is no ecological reason to remove large trees of any species**¹⁷.

As federal agencies develop older forest conservation policies, we note that the recent [IPCC AR6 report](#) states, in relation to natural climate solutions, “...protection of forest ecosystems is the highest priority for reducing GHG emissions (Moomaw et al. 2019) and restoration may not always be practical.” Protecting irreplaceable natural systems from logging is therefore the first step. The IPCC report also noted that maintaining the resilience of biodiversity and ecosystem services at a global scale is “fundamental” to climate mitigation and adaptation, requiring “effective and equitable conservation of approximately 30 to 50% of Earth’s land, freshwater and ocean areas, including current near-natural ecosystems”¹⁸.

⁹ DellaSala et al. 2022. Ibid.

¹⁰Leismaster et al. 2019. Mixed-severity wildfire and habitat of an old-forest obligate. Ecosphere 10(4) Article e02696. Leismaster et al. Northern spotted owl nesting forests as fire refugia: a 30-year synthesis of large wildfires. Fire Ecology 17 <https://doi.org/10.1186/s42408-021-00118-z>. Law et al. 2022. Ibid. Mildrexler et al. 2023. Protect large trees for climate mitigation, biodiversity, and forest resilience. Conservation Science and Practice DOI: 10.1111/csp2.12944.

¹¹Frey et al 2016. Spatial models reveal the microclimatic buffering capacity of old-growth forests. Sci Adv 2016:e1501392. 9 pp. doi: 10.1126/sciadv.1501392. Law et al. 2022. Ibid.

¹²Faison et al. 2023. Ibid.

¹³Bradley et al. 2016. Does increased forest protection correspond to higher fire severity in frequent-fire forests of the western United States? Ecosphere 7:Article e01492.

¹⁴DellaSala et al. 2022. Have western USA fire suppression and megafire active management approaches become a contemporary Sisyphus? Biol. Cons. <https://doi.org/10.1016/j.biocon.2022.109499>

¹⁵Mildrexler et al. 2020. Ibid. Kellett et al. 2023. Forest-clearing to create early-successional habitats: questionable benefits, significant costs. Frontiers in Forests and Global Change <https://doi.org/10.3389/ffgc.2022.1073677>. Mildrexler et al. 2023. Ibid.

¹⁶Swanson et al. 2010. The forgotten stage of forest succession: early-successional ecosystems on forest sites. Front. Ecol. Environ doi:10.1890/090157. Kellett et al. 2023. Ibid.

¹⁷Mildrexler et al. 2023. Ibid.

¹⁸Law et al. 2022. Ibid.

The White House recently released its “[roadmap to nature-based solutions](#),” noting the importance of green infrastructure and other natural climate solutions. We submit that this policy must embrace the protection of large trees and old forests **of all types and species** as one of the nation’s most important “nature-based solutions.” Such an effort in rulemaking would send a clear message to the global community that the United States is making a serious effort to enlist natural climate solutions, while reducing its emissions across all sectors, including forestry. Protecting large trees and old forests in national rulemaking would also comply with the Paris Climate Agreement (Article 5.1), the IPCC AR6 Report, demonstrate steps toward the President’s [Executive Order 14008](#) to protect 30% of all lands and waters by 2030, and showcase the President’s national commitment to the [Glasgow Forest Pledge](#) to end global forest degradation and deforestation.

Sincerely (affiliations for identification purposes only),
35 Lead Signatures

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Forest riparian buffers reduce timber harvesting effects on stream temperature, but additional climate adaptation strategies are likely needed under future conditions

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ABSTRACT

Stream water temperature imposes metabolic constraints on the health of cold-water fish like salmonids. Timber harvesting can reduce stream shading leading to higher water temperatures, while also altering stream hydrology. In the Pacific Northwest, riparian buffer requirements are designed to mitigate these impacts; however, anticipated future changes in air temperature and precipitation could reduce the efficacy of these practices in protecting aquatic ecosystems. Using a combined modeling approach (Soil and Water Assessment Tool (SWAT), Shade, and QUAL2K), this study examines the effectiveness of riparian buffers in reducing impacts of timber harvest on stream water temperature in Lookout Creek, Oregon across a range of potential future climates. Simulations assess changes in riparian management alone, climate alone, and combined effects. Results suggest that maximum stream water temperatures during thermal stress events are projected to increase by 3.3–7.4 °C due to hydroclimatic change alone by the end of this century. Riparian management is effective in reducing stream temperature increases from timber harvesting alone but cannot fully counteract the additional effects of a warming climate. Overall, our findings suggest that the protection of sensitive aquatic species will likely require additional adaptation strategies, such as the protection or provisioning of cool water refugia, to enhance survival during maximum thermal stress events.

Key words | climate sensitivity, forestry, QUAL2K, riparian buffer, SWAT, water temperature

HIGHLIGHTS

- A range of future climate conditions can negatively impact cold-water fish species during extreme thermal events.
- Riparian forest management can help play a role to mitigate impacts of future climate conditions.
- Cold-water species may be unable to thrive in pristine traditional ecosystems due to climate impacts which cannot be mitigated during extreme thermal events.
- Mature forests are unlikely to shelter pristine freshwater streams from impacts of climate change.
- The importance of cold-water refugia and alternative mitigation strategies are likely to become increasingly important by the end of the century to combat climate change.

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INTRODUCTION

The effects of timber harvest on the thermal regime of streams have been observed globally from Oregon to Amazonia. Beginning in the 1950s, watershed studies of clearcut harvesting showed that removal of riparian vegetation resulted in dramatic increases in water temperature in the Oregon Coast Range (e.g., Brown & Krygier 1970). Reduction in baseflow, which can occur due to soil compaction during harvest, may also reduce the capacity to absorb increased thermal inputs. Preventing these adverse effects was a motivating factor in the development of riparian buffer management practices for forestry. Riparian buffers influence water temperature through shading that reduces direct insolation to streams, and secondary effects on microclimate (air temperature and wind) within the buffer are also important. Additional benefits of buffers include reducing peak storm runoff, maintaining stable baseflow, and potentially filtering out sediment, nutrients, and other pollutant loads from the surrounding area. Whether performance of riparian buffers under future conditions is sufficient to offset the effects of warming remains an open question, however.

There is growing concern about the potential effects of climate change on stream temperature. Stream temperatures are projected to increase in most North American rivers during the next century, resulting in increased stress on cold-water fish species such as salmonids. In the Pacific Northwest (PNW), air temperature increased from 1900 to 2000 by about 0.8 °C (Mote & Salathé 2010), mostly due to warming since 1976 (Isaak *et al.* 2018). Increases were larger for daily minimum temperature than for daily maximum temperature (Hamlet & Lettenmaier 2007). Future increases in air temperature in the PNW are projected to be 1.8–2.9 °C by the 2080s compared with climate 1970–1999 (Mote *et al.* 2014). Precipitation changes are less certain, but most models suggest wetter conditions in fall and winter, earlier snowmelt, and drier summers in the PNW.

Several studies have examined the potential impact of these changes on water temperatures in the PNW (e.g., Wu *et al.* 2012; Beechie *et al.* 2013; Dugdale *et al.* 2018). Results show that the thermal responses of individual

streams depend on the complex interaction of hydrology, topography, and climate. Maximum water temperature in some PNW streams could be impacted more by reductions in summer streamflows, associated with drier and warmer conditions, than the direct impact of projected increases in air temperature (Cristea & Burges 2010; Wu *et al.* 2012). Streams that will shift from snow-dominant regimes to mixed or rain-dominant hydrology are also theorized to be more sensitive to increases in air temperature (Mote & Salathé 2010). Luce *et al.* (2014) showed that rain-dominated, low elevation watersheds were more sensitive to climate warming than streams draining steeper topography where flows are dominated by snowmelt in the PNW.

Reducing the risk of future water temperature increases requires understanding the interactions between hydroclimatic conditions and forest management practices. In this study, we examine the effectiveness of riparian buffers for reducing the impacts of timber harvest on stream temperature across a range of potential future changes in air temperature and precipitation. As our study system, we use Lookout Creek (LC), a mixed rain-and-snow catchment, representative of forested catchments on the western, windward side of the Oregon Cascades in a steep headwater catchment. Specifically, we address the following questions:

1. How will projected future changes in air temperature and precipitation affect water temperature in a small, forested PNW watershed?
2. How effective are forest riparian buffers in mitigating anticipated increases in water temperature?

METHODS

In this study, we use an upland watershed model (Soil and Water Assessment Tool (SWAT)), a riparian vegetation shading model (Shade), and an instream hydraulic model (QUAL2K) to simulate streamflow and water temperature responses to a range of air temperature, precipitation, and forest management scenarios.

The SWAT is a lumped-parameter watershed model that simulates the watershed hydrologic response, including

changes to summer baseflow, and provides the flow boundary conditions for the stream temperature model. The SWAT model was selected to represent boundary conditions to QUAL2K for the following reasons: (1) it is a widely accepted, tested, and often-used tool for predicting the impacts of climate and land management practices on hydrology at the watershed scale, (2) it incorporates explicit representation of plant growth needed to represent the effects of changes in temperature, precipitation, humidity, and ambient CO₂ concentration on vegetation and hydrology, and (3) it can be successfully implemented with readily available GIS data. The impact of changes in CO₂ concentrations in the SWAT model are manifested in the simulation of stomata control, which reduces water vapor losses by reducing leaf stomata conductance as CO₂ concentrations rise (Easterling *et al.* 1992). The SWAT simulates water temperature, but in a simple manner that does not fully account for riparian shading and other aspects of the heat balance. The Shade model (developed by Washington Ecology; shade_ver40b04a06.xlsm; www.ecy.wa.gov/programs/eap/models.html) evaluates the effects of canopy and topographic shading on energy exchanges between the atmosphere and stream and is often paired with QUAL2K. QUAL2K is a steady-state stream water-quality model that provides a robust instream simulation of the heat balance during thermal stress events. QUAL2K is a one-dimensional model capable of representing a well-mixed channel with steady-state hydraulics, non-uniform steady flow, and diel heat budget and water-quality kinetics (Chapra *et al.* 2012; Version 2.12b1; <http://www.qual2k.com/>). The QUAL2K model is run for steady-state hydrologic conditions with diel variability in meteorology and shade.

Study site

Model simulations were applied to LC, a fourth-order mountain headwater stream in the McKenzie 8-digit hydrologic unit code #17090004, tributary to the Willamette River in Oregon. The LC watershed lies completely within the United States Forest Service H. J. Andrews Experimental Forest (Andrews Forest) and is part of the Long-Term Ecological Research (LTER) network. The watershed occupies approximately 63.5 km², with an elevation range from 420 to 1,630 m, and is almost entirely covered with

old-growth conifer forest of Douglas fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), and western red cedar (*Thuja plicata*) (Figure 1). About 25% of the watershed was historically logged around 60–80 years ago; however, the riparian areas are almost exclusively densely vegetated old-growth conifers which provide ample shade. LC is generally steep (858-m change over 17.5 km; slope of 0.05) and is well-aerated, shaded, and cool with groundwater-fed baseflow. The climate of LC is wet and mild in winter and warm and dry in summer, typical of the Western Cascades, with winter snowpack historically present above 1,050 m. The stream supports salmonid populations of native cutthroat trout (*Oncorhynchus clarkii*) and non-native rainbow trout (*Oncorhynchus mykiss*).

Recently, there has been an observed increase in the number and duration of warm events per decade in the Cascades that may negatively affect aquatic biota. Historical data (1976–2015) throughout the PNW suggest that August air temperature in this region (zone H, Oregon Coastal) has increased by about 0.29 °C per decade, while August streamflow has decreased by nearly 4% (Isaak *et al.* 2017).

Model development

The SWAT represents a watershed as subbasins subdivided into smaller hydrologic response units (HRUs). HRUs are unique combinations of land-use, soil, and slope classes. Land-use and soil property inputs for the LC watershed represented in the SWAT are based on the National Land Cover Database 2011 version (NLCD; 30-m resolution resampled to 10-m resolution) and the State Soil Geographic (STATSGO; polygon-based, converted to 10-m resolution raster). Slope is based on a 10-m resolution digital elevation model. Water yield is generated at the HRU level (10-m resolution) and aggregated by subbasin and transferred to the stream reach routing component of the SWAT. The stream reach routines simulate the transport and transformation of nutrients in the stream network. The SWAT application for the LC watershed consists of 111 HRUs (using two slope classes and zero thresholds for HRU land-use, soil, and slope definition) and 11 subbasins and reaches. The SWAT was configured to approximate observed streamflow and water quality in the LC watershed 2000–2015.

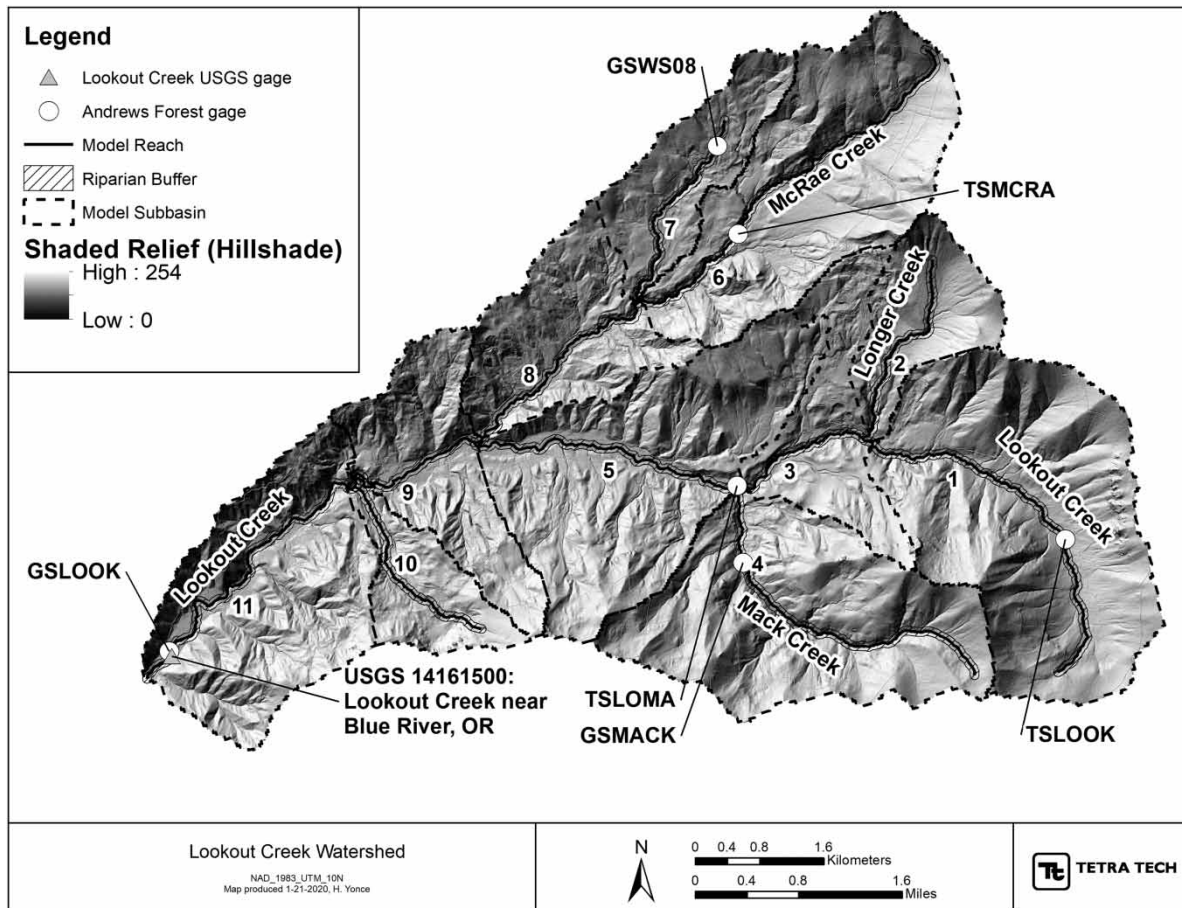


Figure 1 | Subbasin boundaries, model reaches, riparian buffers, and monitoring sites in the LC watershed.

Meteorological forcing in the SWAT is based on gridded, daily weather data. Daily precipitation time series were developed for each subbasin using the PRISM (Parameter-elevation Relationships on Independent Slopes Model), a 4-km resolution daily gridded precipitation product (<http://prism.oregonstate.edu>). Meteorological data at a coarser scale of approximately 14-km resolution were retrieved from NLDAS2 (North American Land Data Assimilation System version 2) and processed to develop air temperature, relative humidity, solar radiation, and wind speed forcing series. Potential evapotranspiration (PET) was automatically computed by the SWAT during the simulation from these series using the Penman Monteith method. Both gridded datasets were area-weighted to the SWAT subbasins.

A QUAL2K model was developed for the LC mainstem and major tributaries to examine potential changes to water

temperature during critical conditions for thermal stress and risk to biota. Each QUAL2K stream segment or ‘reach’ was subdivided into 0.1-km computational elements. Inputs for each QUAL2K reach include physical characteristics (e.g., bed slope and reach length), boundary conditions (diffuse flows to the stream network provided by the SWAT), meteorological data (hourly air temperature, dew point temperature, wind speed, and cloud cover), and shading (provided by the Shade model). Runoff to LC as simulated by the SWAT over a longer period of time is dominated by shallow subsurface pathways, and baseflow is supported solely by subsurface flow. Therefore, the watershed flow entering QUAL2K reaches is represented as nominal ($0.01 \text{ m}^3/\text{s}$) diffuse inflow at upstream headwaters. The key linkage between the SWAT and QUAL2K models for both baseline model development and scenario application is characterized by hydrological changes in baseflow.

QUAL2K reach length, width, slope, and elevation were developed from Light Detection And Ranging (LiDAR) spatial datasets flown in August 2008 (data available from <https://andrewsforest.oregonstate.edu/>). Reach hydraulics were simulated using the rating curve method. Velocity (U , m/s) and depth (H , m) were generated as power functions of streamflow (Q , m³/s) produced by the SWAT and scaled to observed streamflow at the watershed outlet: $U = a \cdot Q^b$; and $H = A \cdot Q^B$. Exponents b and B were set to typical values of 0.43 and 0.45, respectively, as suggested in the QUAL2K documentation (Chapra *et al.* 2012). Coefficients a and A were approximated using observed channel geometry from Andrews Forest research (Anderson *et al.* 2005). The relationships between observed depth and drainage area were used along with field observations of channel width and velocity at the U.S. Geological Survey (USGS) gage #14161500 (LC near Blue River, OR) to estimate depth and velocity coefficients.

The term ‘critical conditions’ for the QUAL2K model application refers to forcing that causes the highest reasonably expected 7-day average daily maximum water temperature (MWAT). Daily water temperature data are available for LC at USGS gage #14161500 (2012–2017). The highest MWAT was observed for the week prior to 8 July 2015 (20.9 °C), so that was the date selected for QUAL2K calibration. The second highest annual 7-day average daily MWAT was observed for the week prior to 4 August 2014 (19.4 °C), so that date was selected for model corroboration. A second corroboration application was constructed for 28 July 2013, the day which exhibited the highest 7-day average of the maximum air temperature (32.7 °C) at the Andrews Forest primary weather station (PRIMET).

Air temperature datasets for QUAL2K were constructed by reach using observed daily minimum and maximum temperatures from Andrews Forest meteorological stations averaged over the preceding week and interpolated to hourly values. Dew point temperatures were calculated using the hourly air temperature inputs and the average daily relative humidity observed across the watershed for the week prior to the simulation date. Hourly wind speeds were based on average observed summer conditions across the watershed, disaggregated to hourly inputs. Hourly cloud cover during daylight hours was based on observed

solar radiation at the primary meteorological gage in the Andrews Forest and estimated cloudless sky radiation. Night cloud cover estimates were based on weather observations at Mahlon Sweet Airport in Eugene, OR (GHCND: USW00024221), which is the closest first-order station in the region.

Heat content from groundwater inflows and small, groundwater-driven tributaries has an important effect on mainstem water temperature in LC (Tague *et al.* 2008). To estimate these inputs for QUAL2K, we used data from five continuous water temperature monitoring sites on small streams in the watershed that were not included in the explicitly modeled stream network. These were used to develop a regression against elevation for the temperature of discharging groundwater. The combined impact of groundwater to the system was input to QUAL2K as steady diffuse inflow along the length of each model reach.

Stream shading associated with different riparian management scenarios was estimated using the Shade model. We used TTools, a GIS extension for sampling geospatial data on channel morphology, land cover, and topography, to assemble high-resolution inputs for the Shade model (available through Washington Ecology; www.ecy.wa.gov/programs/eap/models.html). TTools was used to sample August 2008 LiDAR-derived topography and vegetation height rasters, measure topographic angles, and stream gradients. Longitudinal GIS sampling output from TTools was used to tabulate hourly effective stream shading to the water due to topography and vegetation in the Shade model. The Shade model was run for the calibration model date (8 July 2015), and hourly shade estimates were averaged by model reach. The average daily shade from topography and vegetation for the 8 July 2015 calibration model was 69.2%. Average shade for the mainstem is lower than the tributaries (58.9% compared with 80.9%), which are narrower channels with a greater frequency of north–south orientation.

We conducted sensitivity analyses to test the relative impact of major model inputs on temperature predictions from the QUAL2K models. Results are summarized per QUAL2K output as either diel output at a single point instream or longitudinally from headwaters to the outlet. Of the parameters tested, simulated water temperature was less sensitive to changes in cloud cover and wind speed,

and more sensitive to changes in air temperature, ground-water temperature, and shade.

Model calibration approach

The calibration of the QUAL2K model focused on minimizing mean error and mean absolute error between simulated and observed water temperatures during warm summer conditions. Water temperature data were available for comparison at USGS site 14161500, and Andrews Forest monitoring gages on simulated reaches: GSLOOK, GSMACK, GSW08, TSLOMA, TSLOOK, and TSMCRA (Figure 1). We used the sensitivity analysis to identify key input parameters for calibration. The performance of the model was corroborated through application to two additional dates with high air and water temperatures.

The SWAT application was calibrated against observed continuous streamflow at the USGS gage. Moriasi *et al.* (2007) provide guidelines for the evaluation of SWAT applications based on Nash–Sutcliffe Efficiency (NSE) and percent bias (PBIAS) at a monthly time step. We also examined fit of daily time series and checked for seasonal biases as part of the calibration exercise.

Scenario development

Climate scenarios

Scenarios representing a range of potential future climatic conditions for LC were developed based on the statistically downscaled global climate model (GCM) output from the Multivariate Constructed Analogs (MACA) version 2 archive (<http://maca.northwestknowledge.net/>). The constructed analog approach uses a library of historical observations to scale from monthly to daily time step, ensuring a reasonable representation of the temporal structure of local rainfall (Abatzoglou & Brown 2012). The MACA dataset also provides simultaneous downscaling of precipitation, temperature maxima and minima, humidity, wind, and radiation, helping to ensure physical consistency in the outputs and allowing energy–balance estimation of PET. We selected a set of five downscaled GCM projections (Table 1) to approximate a plausible range of potential future changes. Specifically, we selected GCMs that approximate the median, 10th, and

Table 1 | GCMs selected for LC future climate simulations (CMIP5, RCP 8.5)

Future climate condition (relative increases)	GCM
10th percentile precipitation, 10th percentile air temperature	MRI-GCM3
90th percentile precipitation, 10th percentile air temperature	inmcm4
90th percentile precipitation, 90th percentile air temperature	IPSL-CM5A-LR
10th percentile precipitation, 90th percentile air temperature	MIROC-ESM
50th percentile precipitation, 50th percentile air temperature	CNRM-CM5

90th percentiles of changes in annual average precipitation and air temperature assuming relatively high greenhouse gas projections (Coupled Model Intercomparison Project Phase 5 [CMIP5] Representative Concentration Pathway [RCP] 8.5). We focus on GCM output for late-century conditions (2071–2100) to provide a wider range of possible conditions and because achieving maximum riparian vegetation is a process that takes decades. Projected results for intermediate time horizons will fall along a trajectory between the historic and end-of-century scenarios, although the form of that trajectory is not known.

Daily gridded MACA precipitation, maximum and minimum air temperature, solar radiation, wind speed, and relative humidity data were area-weighted and aggregated by SWAT subbasins for the LC watershed for the hindcast (1950–2005) and forecast (2069–2099) periods. While we simulate the hindcast period (1950–2005), we report changes in water yield for the forecast period relative to the last 30 years (1976–2005).

Atmospheric CO₂ concentration also has direct effects on plant growth and physiology, which has a potentially significant impact on evapotranspiration (ET). Plants decrease stomatal conductance to mitigate transpiration water loss as atmospheric CO₂ concentrations rise. Ambient CO₂ concentrations at the midpoint of each simulation period were specified from the IPCC CMIP5 estimates (historic [1978] – 335 ppm; mid-century [2045] – 513 ppm; late century [2085] – 801 ppm). The SWAT then reduces water vapor losses by reducing stomata conductance as CO₂ concentrations rise (Easterling *et al.* 1992). Plant radiation use efficiency also changes in response to increased CO₂.

The QUAL2K model inputs are steady-state representations of critical periods for which high water temperatures are expected, and meteorological inputs were developed from the same MACA datasets used with the SWAT. Air temperature inputs for QUAL2K scenarios represent the absolute increase in 7-day average daily maximum air temperatures from hindcast (1976–2005) to forecast (2070–2099) periods as applied to the calibration model. Dew point temperature inputs were calculated as a function of air temperatures and average relative humidity for the simulation date in the forecast period. Scenario boundary conditions for flows in QUAL2K were developed using model output from the SWAT, with the percent change in lowest annual 7-day average flow between hindcast and forecast periods applied to the existing condition flow rates. The reach-averaged change in baseflow for each climate scenario as simulated by the SWAT ranged from –0.7 to 22.4% across all GCMs. QUAL2K inputs for boundary condition (primarily diffuse inflow) water temperatures were estimated as the difference between average daily air temperatures over the hindcast and forecast periods, as shallow groundwater temperatures tend to reflect average annual air temperatures.

Forest management/shade scenarios

The buffer width needed to maintain shading varies with aspect, but almost all the shade effect takes place within about one tree height of the stream edge (e.g., within about 50–60 m for old-growth conifers in the PNW) (Moore *et al.* 2005). Groom *et al.* (2011) reported that 68–75% of post-harvest shade in western Oregon streams were accounted for by basal area within 30 m of the stream and that 30 m riparian buffers minimized post-harvest temperature increases.

For modeling purposes, the riparian buffer zone width was defined as 54 m on either side of all stream channels to incorporate the approximate distance at which effects on stream shade approach zero (Groom *et al.* 2011). Management options in the riparian buffer are explicitly represented in Shade/QUAL2K, ranging from clearcut to restoration of system potential vegetation (SPV), and including two partial harvest options within the buffer, as summarized in Table 2. Shade reported in the table is the reduction of daily

Table 2 | Riparian forestry buffer management and shade simulation scenarios

Forest riparian buffer shade scenario	Shade results and QUAL2K components
Existing shade	LiDAR/TTools/Shade analysis of existing vegetation conditions. Average daily shade 69% (59% for mainstem and 68–87% for tributaries).
SPV	100-year SPV in riparian buffer with 85% density, tree height 50 m, overhang 3 m. Average daily shade 95% (93% for mainstem and 95–97% for tributaries).
Full clearcut	Fully clearcut, stream shading provided only by topography. Average daily shade 13% (9% for mainstem and 12–21% for tributaries).
Outer clearcut	Inner 24 m maintained at the existing shade, outer riparian areas clearcut. Average daily shade 68% (57% for mainstem and 67–87% for tributaries).
Outer thinning	Riparian buffer simulated with inner 6 m maintained at the existing shade, then 18 m of forest thinning (existing vegetation with density decreased by 25% and clearcut the rest of the outer buffer). Average daily shade 65% (54% for mainstem and 64–82% for tributaries).

insolation caused by topography and vegetation, before accounting for reflection from the water surface.

The SWAT does not provide a spatially explicit way to represent changes within the riparian buffer; therefore, scenarios for forest clearcutting and thinning were applied to all forest HRUs. The results are then extrapolated to the fraction of forest area lying within the buffer zone, which captures the basic upstream impact on hydrology for boundary condition inputs, while Shade and QUAL2K capture the key local effects on the stream. We evaluated four clearcutting scenarios; simulated on 5, 10, 20, and 50% of all forested HRUs. To simulate the immediate after-effects of clearcutting, we set the initial biomass and leaf area index (LAI) for affected HRUs to zero. We also increased the initial curve number (CN2) to be representative of barren or no vegetation conditions consistent with previous SWAT studies. For thinning, we also evaluated application to 5, 10, 20, and 50% of all forested HRUs. Forest thinning was represented by reducing the initial

biomass and LAI limit of forested HRUs to half of their initial values.

RESULTS AND DISCUSSION

Results: model performance

SWAT-simulated daily flow time series show good agreement with observations at the LC outlet (USGS #14161500; [Figure 2](#)). Daily NSEs for the calibration and validation periods (water years 2011–2015 and 2006–2010) are 0.749 and 0.674, respectively. Monthly NSE and PBIAS for the calibration and validation periods are 0.918 and 11.3%, and 0.826 and 18.5%, respectively. Based on the NSE alone, the model performance is rated as ‘very good’ based on the criteria of [Moriasi *et al.* \(2007\)](#), although the relatively high PBIAS puts the model performance in the

‘good’ to ‘satisfactory’ range. Comparison of flow duration curves shows that the model tends to overestimate mid-range stream flows in the winter but provides an accurate representation of late summer baseflow conditions when water temperature maxima occur.

QUAL2K-simulated water temperatures for both the calibration and corroboration models provide good matches to observed data at the watershed outlet, especially for the daily maximum ([Table 3](#)). The simulated daily temperature cycles were also reasonably accurate ([Figure 3](#)).

Results: stream response

Simulated total water yield ([Figure 4](#)) increases in four out of five future scenarios by late century (2071–2100), with additional increases in yield associated with forest thinning, which reduces ET. SWAT simulations for clearcut scenarios

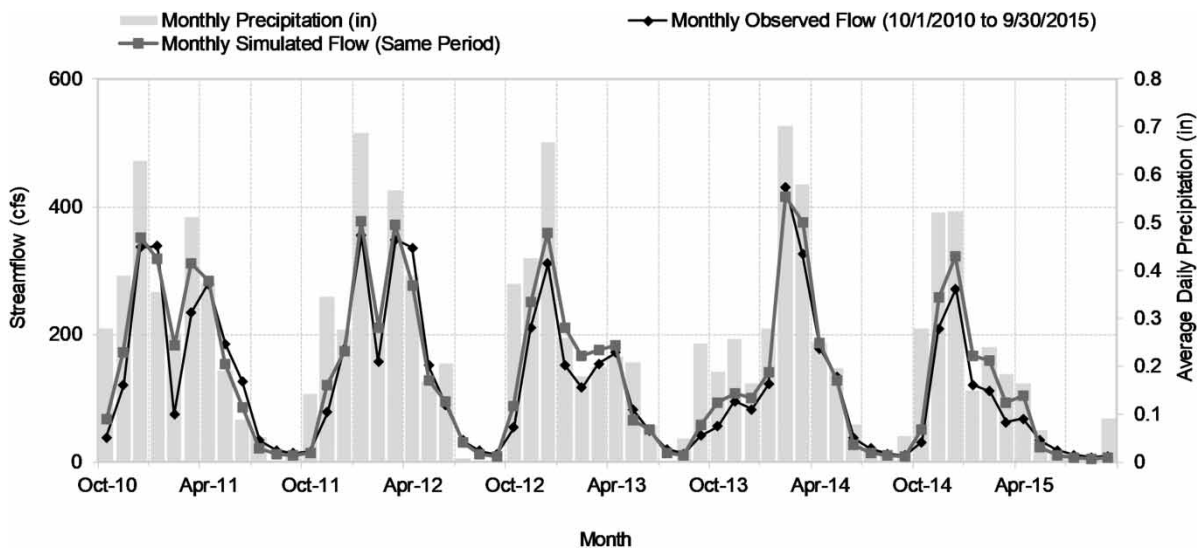


Figure 2 | SWAT-simulated and -observed streamflow at USGS gage #14161500.

Table 3 | QUAL2K-simulated and -observed hourly water temperature statistics at Andrews Forest gage GSLOOK

Simulation date	Model type	Simulated mean (°C)	Observed mean (°C)	Simulated maximum (°C)	Observed maximum (°C)	Mean error	Mean absolute error
8 July 2015	Calibration	18.79	19.18	21.56	21.50	−0.39	0.86
4 August 2014	Corroboration	17.09	17.71	19.17	19.10	−0.62	0.74
28 July 2013	Corroboration	16.37	16.13	18.81	18.00	0.24	0.68

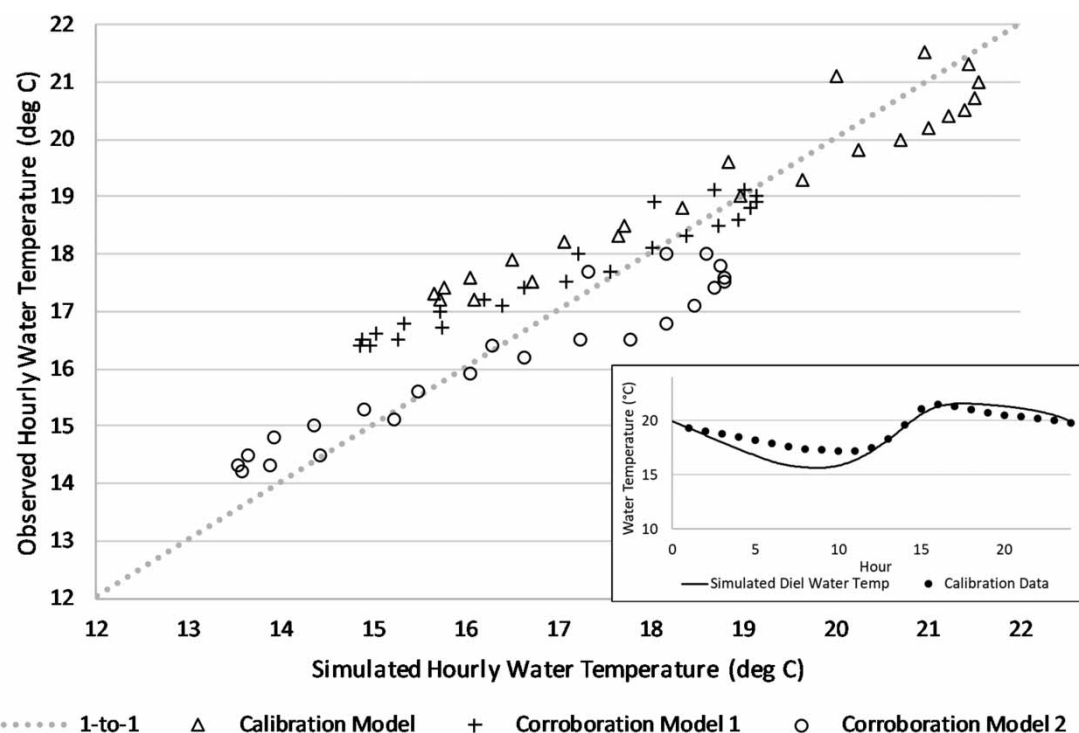


Figure 3 | QUAL2K-simulated and -observed hourly water temperature time series at Andrews Forest gage GSLOOK.

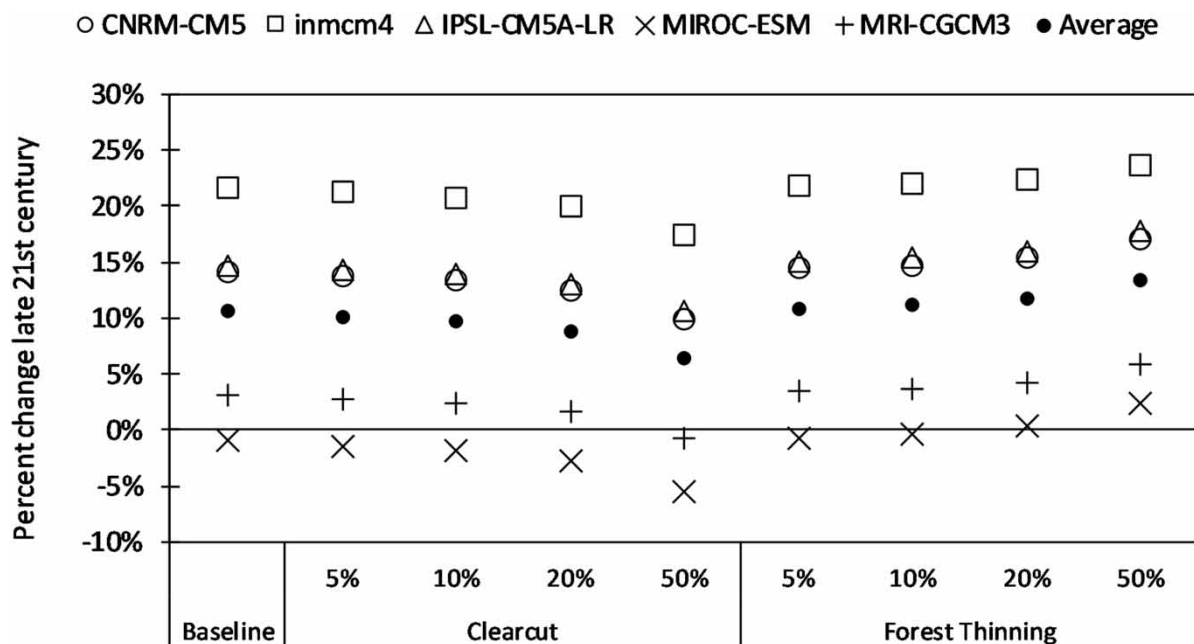


Figure 4 | Change in the projected future annual water yield relative to historic conditions.

represent changes in the first year after harvest. Under the clearcut scenario, the SWAT simulates a decrease in ground-water discharge and an increase in ET associated with rapid

regrowth. The net result is a slightly smaller increase in water yield compared with simulations of current forest cover under future climate.

The calibrated model run for the event of 8 July 2015 is referred to hereafter as ‘historic climate’ with the ‘existing shade’ for which scenario responses will be compared to as relative change. SWAT hydrologic simulations under future climate show precipitation increasing more than ET in four of five late-century scenarios (all but scenario MIROC-ESM), leading to an overall increase in water yield and critical period baseflow volume under existing shade conditions (Figure 5). This result is due to change in the net balance between input precipitation and outputs such as ET.

Forest management and shade effects on water temperature were evaluated longitudinally along LC per QUAL2K model output from headwaters to outlet. Under historic climate, longitudinal simulation results for the MWAT show that clearcutting can increase temperatures throughout the length of LC, while SPV can reduce temperatures. A similar pattern is seen under late-century conditions, but the baseline is shifted upward due to increased thermal inputs (Figure 6).

Longitudinal results were spatially averaged across all model reaches and are summarized in Table 4. Under historic climate, clearcutting leads to large increases in water temperatures throughout LC. The most aggressive management response, the SPV scenario, can reduce temperatures, but not to baseline conditions under most

late-century climate scenarios. A similar pattern is seen under late-century conditions; however, overall temperatures are increased due to warmer air temperatures. QUAL2K simulations suggest that in the absence of modified management practices (i.e., maintaining existing riparian conditions), MWATs are anticipated to rise from 17 to 38% during extreme thermal events. The results for each forest management practice show that clearcutting increases MWATs by nearly 30%, while SPV decreases MWATs by nearly 20% under historic climate. Riparian buffer outer clearcut and outer thinning scenarios result in only small increases in MWATs for LC relative to the existing shade because the intact portion of the buffer adjacent to the narrow stream channel is sufficient to fully shade the water surface.

All future climate scenarios suggest future increases in MWAT, amplified by clearcutting, or mitigated by SPV. Among the GCMs considered in this study, MIROC-ESM has the largest increase in future air temperature ($\sim 7.4^{\circ}\text{C}$ warmer than historic) and the lowest projected future baseflow volume. MRI-CGCM3 has the smallest increase in future air temperatures ($\sim 3.3^{\circ}\text{C}$ warmer than historic). The simulated water temperatures under future conditions closely follow the trends in air temperature. Figure 7 summarizes the corresponding changes in the MWAT during critical thermal stress conditions.

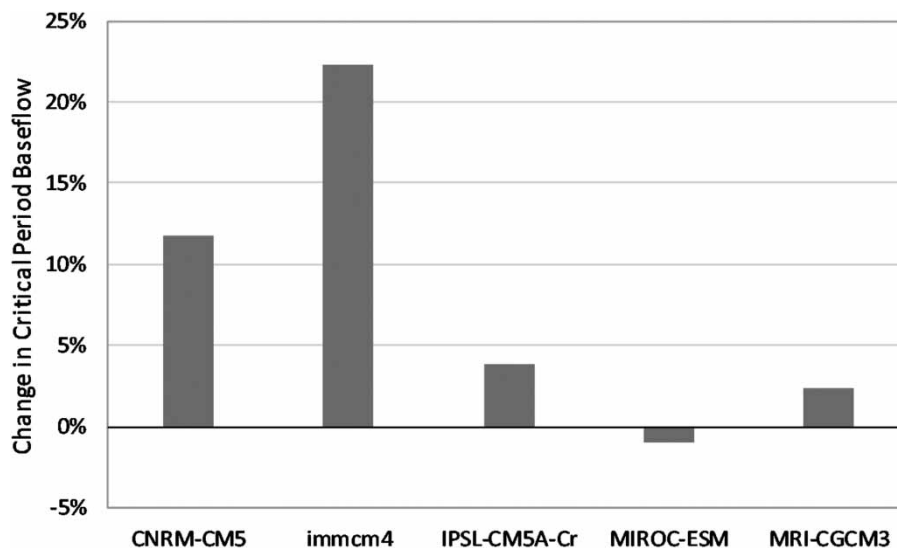


Figure 5 | Late-century changes in baseflow for critical thermal stress events by GCM with the existing shade conditions.

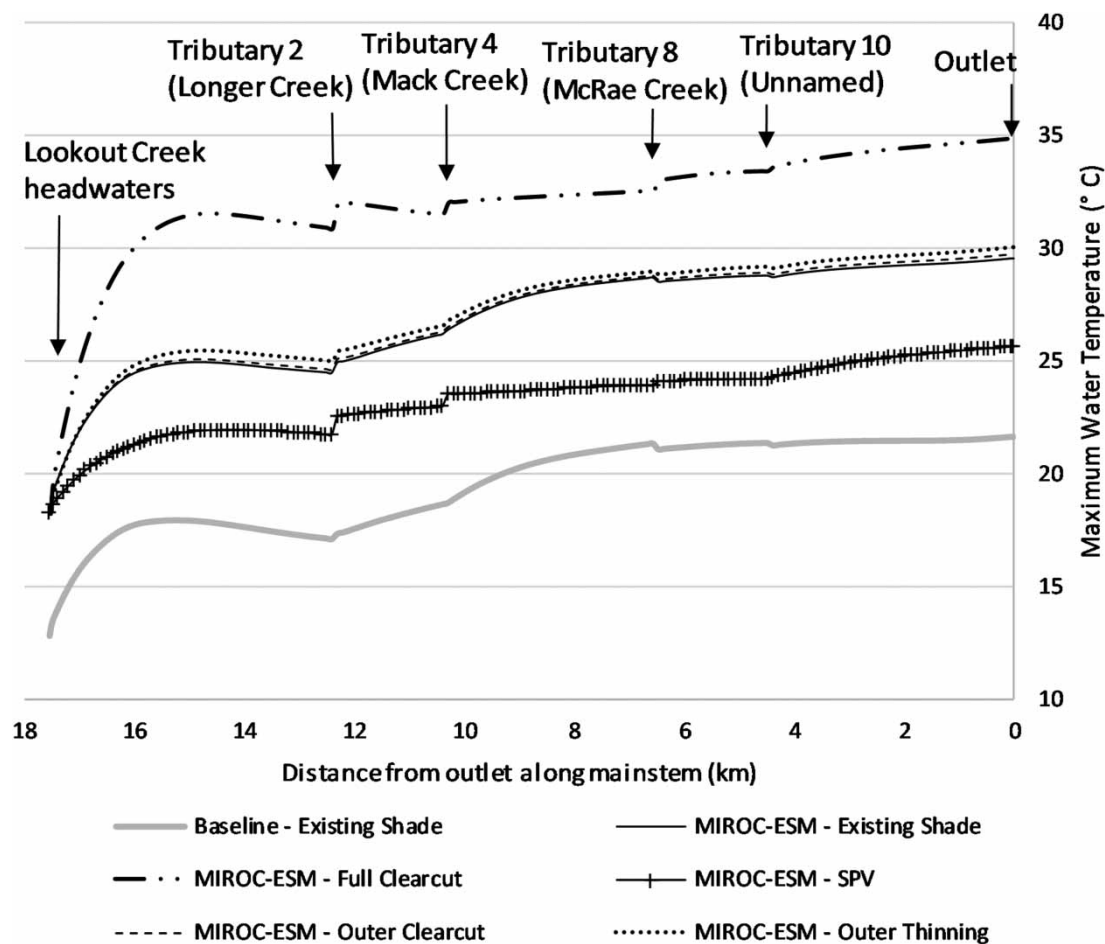


Figure 6 | Longitudinal temperature changes along the LC mainstem for late-century (2071–2100) climate, MIROC-ESM GCM compared with historic conditions with the existing shade. Note that cold headwater temperatures are depicted at the far left of the graphic, and so increases in water temperature shown as LC creek flows downstream are due to the impact of solar radiation, shade, and tributary inflows.

Table 4 | Future maximum (spatially averaged) water temperature (°C and % change relative to historic climate and existing forest cover) for critical thermal stress conditions

Climate scenario	Forestry management/shade scenario				
	Existing shade	Full clearcut	SPV	Outer clearcut	Outer thinning
Historic	19.5	25.0 (28.1%)	15.7 (– 19.7%)	19.6 (0.6%)	19.9 (2.2%)
MIROC-ESM	26.9 (37.8%)	32.0 (64.1%)	23.3 (19.3%)	27.0 (38.3%)	27.3 (39.8%)
inmcm4	23.5 (20.6%)	28.4 (45.5%)	20.2 (3.3%)	23.6 (21.0%)	23.9 (22.5%)
CNRM-CM5	24.2 (23.9%)	29.2 (49.7%)	20.7 (6.0%)	24.3 (24.4%)	24.6 (25.9%)
MRI-CGCM3	22.8 (17.0%)	28.1 (44.1%)	19.1 (– 1.9%)	22.9 (17.5%)	23.2 (19.1%)
IPSL-CM5A-LR	25.7 (31.5%)	30.8 (57.7%)	22.1 (13.2%)	24.3 (24.4%)	26.1 (33.5%)

Note that while simulations indicate that SPV provides a consistent benefit relative to other scenarios, it is not sufficient to mitigate projected increases in the MWAT by the

end of century, as MWAT is still projected to increase in four out of five climate scenarios, while average water temperatures increase in all five.

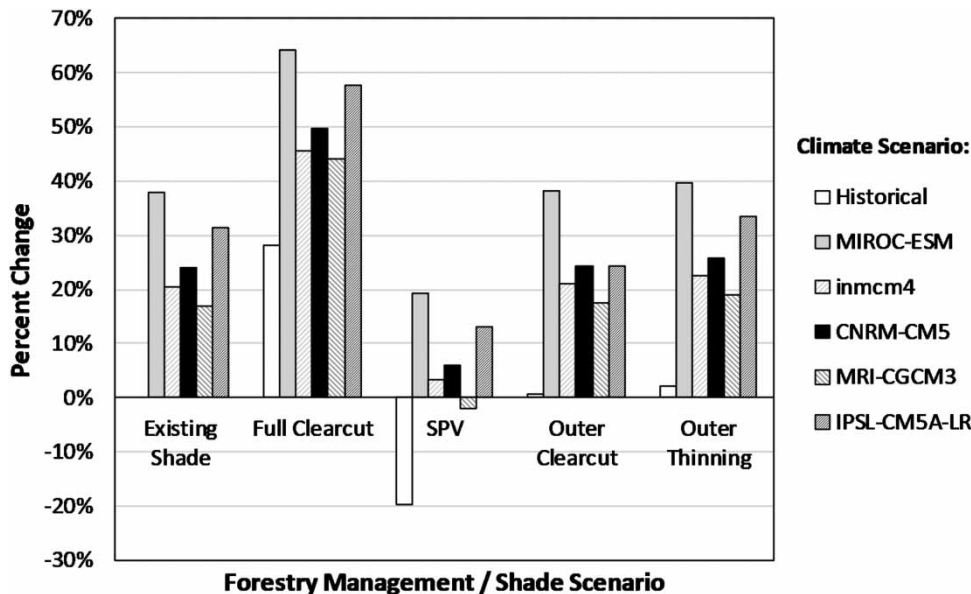


Figure 7 | Late-century percent change in daily MWAT during critical thermal stress conditions relative to historic climate and the existing shade.

Linked to the stream temperature response, QUAL2K also simulated decreases in dissolved oxygen (DO) concentrations which are integral drivers of stream ecological health. The decline in DO is mostly due to increases in water temperature, which reduces the attainable DO saturation concentration. DO concentrations were projected to decrease by nearly 10% or 1 mg/l during summer critical events.

DISCUSSION

This study evaluates the effectiveness of riparian buffers for reducing the impacts of timber harvest on stream temperature in LC, a forested catchment in the Oregon Western Cascades, across a range of potential future changes in air temperature and precipitation. We first consider how potential future changes in air temperature and precipitation could affect water temperature. Simulations in this study suggest precipitation increasing more than ET, leading to increases in annual average water yield for the LC watershed. Shifts in the snow regime do not appear to be a major factor for future conditions in LC, which is already a mixed snow and rain system and is expected to remain so by late century. Despite increasing air temperatures and reduced winter snowpack, no significant decline in summer baseflow is projected.

In contrast, a study of the entire Columbia River basin by Hamlet *et al.* (2013) suggested that summer low flows will decline in most streams west of the Cascades due to increased ET. The difference could be due, in part, to reduced ET, offsetting the effects of increasing temperatures, in our SWAT simulations resulting from representation of vegetative response to increased atmospheric CO₂ concentrations.

In snow-dominant regimes, much of the winter precipitation is stored in snowpack, which melts in spring and early summer, resulting in peak flows in early spring, maintenance of baseflow, and a continued supply of cold melt water through summer (Hamlet & Lettenmaier 2007). Where air temperature increases, snow-dominant systems shift to mixed or rainfall-dominant regimes, significant reductions in summer flows and cool groundwater discharge are likely, potentially exacerbating the effects of increased air temperatures (Mote & Salathé 2010). Streams with marginal snow-dominant hydrology (Butcher *et al.* 2016) may be particularly sensitive due to a shift from winter snow to rain. Luce *et al.* (2014) and Isaak *et al.* (2016a) both noted that observed rates of water temperature change relative to air temperature change are typically lower in headwater streams because of short residence times and strong local temperature gradients associated with topography.

Despite minimal change in summer baseflow, our results suggest that stream temperatures will increase

under future conditions. The QUAL2K simulation results suggest that MWATs during critical thermal stress events in LC will increase 17–38% (3.3–7.4 °C) under an existing riparian buffer condition by late century. The projected temperature increases are greater than the 2–3 °C increases in average summer temperatures suggested by Wu *et al.* (2012) for the end of century in this region, and the summer temperature increase of about 2 °C in LC suggested by Isaak *et al.* (2016b). Simulations reported by Wu *et al.* and Isaak *et al.* are based on the CMIP3 A1B scenario, which projects substantially lower air temperature increases by the end of the 21st century than the CMIP5 RCP 8.5 scenario used in this study. We also expect that extreme thermal events (MWATs) may increase by more than the summer average, and it is these extreme events that will impose metabolic constraints on fish species survival.

A second question we address is how effective forest management strategies using riparian buffers are to mitigate anticipated increases in water temperature in forest streams. We consider five forest management options simulated in QUAL2K representing a range of disturbances in the riparian buffer, from clearcutting (full loss of vegetative shade) to SPV, the expected condition of maximal shade absent anthropogenic disturbance. Resulting MWAT varies in accordance with this gradient in shade provided by riparian vegetation. The lowest MWAT is associated with SPV. It is important to note that riparian areas in most forests are not currently at SPV because of past activities.

QUAL2K simulation results suggest that MWATs under summer critical conditions are likely to increase under most late-century scenarios, regardless of riparian management practices. Restoration of full SPV in the riparian buffer can reduce temperatures relative to current vegetation under future climate but does not provide sufficient mitigation to prevent substantial temperature increases during critical thermal stress events relative to the historic temperature regime. Similar results have been suggested elsewhere in the PNW (e.g., Cristea & Burges 2010; Mantua *et al.* 2010; Butcher *et al.* 2016),

Simulations suggest that all riparian buffer strategies are effective in offsetting water temperature increases associated with harvest. Notably, the riparian management scenarios designed to allow harvest while protecting streamside shade resulted in an increase in MWATs relative to existing vegetation conditions, but only by a small amount

(a maximum predicted increase of 0.4 °C). However, in most cases, riparian buffers cannot fully compensate/offset the possible increase in thermal stress events due to projected future increases in air temperature. This finding is consistent with findings from a modeling study of the South Fork Nooksack River (WA), which predicted that the risk of higher water temperatures will accelerate over time and is likely to overwhelm the mitigation provided by increased shade from riparian buffers (Butcher *et al.* 2016).

Our findings suggest that additional strategies beyond SPV will be necessary to meet temperature criteria designed to protect cold-water species. Protecting and restoring local cold-water refugia – such as log-jam pools, small tributaries, side channels, and backwaters fed by groundwater seeps or hyporheic upwelling – may help to mitigate the effects of warming temperatures on aquatic life during high-temperature events, as fish can thermoregulate by moving to such microhabitats (Beechie *et al.* 2013). The effectiveness of these strategies under future conditions, in combination with riparian buffer restoration, should be further examined.

As in all studies, there are several limitations which should be noted. First, effective riparian buffer widths are likely to vary depending upon the physical setting. LC has a mountain valley hydrologic landscape, with relatively steep hillslopes and very little flat riparian area. These hillslopes provide significant shading to the stream regardless of forest management. Thus, buffer widths likely be relatively smaller in mountain valley hydrologic landscapes than other less dissected landscapes. To achieve the same temperature-moderating effects, wider buffers may be needed in hydrologic landscapes with wider, flatter riparian areas and less significant topographical shading.

Second, impacts in the QUAL2K model from various forestry management techniques were largely limited to impacts on localized hydrology and baseflow. The realized impact of changing climate patterns on vegetation and the potential interactions between climate and management scenarios would require further analysis and different modeling applications. Mature SPV within the riparian corridor will be more resilient to changes in climate than less well-established communities; however, changing hydrologic and thermal regimes may also impact which species thrive relative to changes in PET in response to changes in CO₂. Higher CO₂ concentrations may increase mortality rates in

case of drought due to an enhanced dependence on soil moisture as watershed hydrology changes. Watershed elevation also plays a role in the variable forest response to changing temperature and CO₂ concentrations. The ecosystem response to changing CO₂ is also dependent on tree species, age, stand density, stand health in response to pests, management or natural disturbance, seasonality, and nutrient availability which are not only difficult to simulate and predict, but have combined impacts as well on both forest response and management options and opportunities (Kallarackal & Roby 2012). For example, the increased likelihood of wildfire due to changing climate may be reduced by thinning operations. These interdependencies stress the need for whole forest ecosystem modeling to further analyze the effects and interactions between climate, vegetation, and management.

Finally, it should be noted that the results of this study are conditional on the methods, models, and scenarios used. The QUAL2K model implemented here may omit some important physical factors that could ameliorate water temperatures such as increased hyporheic flow or discharges of deeper, cooler groundwater. Conversely, increased tree mortality or changes in species composition under a future climate might yield incorrect assumptions about the amount of shading that can be achieved with SPV, although strategies are available to increase the climate resilience of riparian buffers.

CONCLUSIONS

This study evaluates the effectiveness of riparian buffers for reducing impacts of timber harvest on stream temperature in LC, a forested catchment in the Oregon Western Cascades, across a range of potential future changes in air temperature and precipitation. Results suggest that riparian shade can play an important role in mitigating extreme water temperature events and can remain an effective practice for counteracting the effects of timber harvesting; however, anticipated temperature increases by late century will likely be greater than can be addressed through maintaining – or even increasing – riparian shade alone. While results presented are specific to LC in Oregon, they illustrate the risk of future temperature increases in other mid-altitude PNW streams. Although high-temperature events will likely increase even with attainment

of SPV in riparian areas, achieving SPV will produce significantly better outcomes than scenarios under which riparian buffers are only partially protected. In addition to robust riparian buffers, other climate adaptation strategies, such as providing and/or protecting cold-water refugia, are likely needed to protect cold-water species from higher stream temperature events in the future.

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DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

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Effects of fine sediment inputs from a logging road on stream insect communities: a large-scale experimental approach in a Canadian headwater stream

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Abstract

A forest headwater stream was manipulated (logging road-crossing amended) to induce fine sediment inputs. Benthic inorganic sediment concentrations (particles 1.5–250 μm) increased from a 2-year pre-disturbance average of about 800 g m^{-2} to over 5000 g m^{-2} that persisted for 3 years. Aquatic insect communities were examined over the 5-year study period in the manipulated and nearby reference streams. Overall, the effects of the fine sediment increases on aquatic insect communities were minimal. There were no significant effects of sedimentation on total aquatic insect abundance or biomass. An index of multivariate dispersion gave no evidence of community stress at the manipulated site. Multivariate ordination plots and time trends among univariate community metrics indicated only subtle changes in community structure. Among the univariate metrics (16 time series analyses in total), six gave evidence of a sediment impact on aquatic insect communities. Of those, the clearest indications of an effect were small reductions in diversity and richness of spring communities. These resulted from a significant decline in the proportion of spring shredders, accompanied by a significant increase in the percent Chironomidae. This large-scale experimental approach integrated the realism of a whole-stream study with the control of a manipulative study by including pre-manipulation measurements and excluding other confounding catchment disturbances. In this regard, it may provide a more realistic measure of benthic community-level responses to sedimentation in streams at a magnitude associated with logging activity than many previous studies.

Introduction

Catchment disturbances that result in fine sediment inputs to streams have been identified as major contributors to the degradation of freshwater habitats and to impacts on aquatic biota (Waters 1995). In particular, agricultural tillage, urbanization, mining, road construction and logging activity can result in fine sediment inputs and deposition in stream beds. The biological consequences can include declines in benthic invertebrate abundance and/or changes in community structure (Tebo 1957; Somer and Hassler

1992; Wood and Armitage 1997; Fossati et al. 2001). However, most studies that have examined the effects of fine sediments in streams on benthos have relied on small-scale experiments in artificial stream channels or flumes (Rosenberg and Wiens 1978; McClelland and Brusven 1980), or on comparative studies between reference and sediment-impacted sites after the disturbance or along a gradient of sedimentation (Lenat et al. 1981; Ermin and Ligon 1988; Hogg and Norris 1991; Barton and Farmer 1997; Zweig and Rabeni 2001). These provide valuable information on the potential effects of sedimentation on stream

insects, but may not be representative of real-world responses by benthic insects to fine sediment inputs because of limitations in scale and the lack of baseline or pre-disturbance community measures.

Logging activities, particularly in recent years under best management practices, often produce only moderate amounts of fine sediment deposition on stream beds in comparison to more destructive land use activities (Wood and Armitage 1997). Reported effects of fine sediments on stream habitat and benthos from logging (Campbell and Doeg 1989; Webster et al. 1992; Binkley and Brown 1993; Keenan and Kimmins 1993) may have been confounded by other catchment disturbances such as canopy modification or changes in nutrient flux (Murphy et al. 1981; Grown and Davis 1994). While some studies have demonstrated adverse effects on benthos under conditions of heavy sediment loading, others have detected little or no response to more moderate fine sediment increases (Culp et al. 1985; Fairchild et al. 1987). Thus it is not clear to what extent stream benthos is adversely affected by sedimentation at magnitudes associated with logging activity, especially logging road construction or maintenance. Models have been developed to estimate sediment yield from logging activities (Burns and Hewlett 1983) from which sediment deposition in streams could be predicted, but few studies have linked actual benthic sediment concentrations to responses by benthic invertebrates under realistic conditions.

To examine benthic insect responses to stream sedimentation under realistic conditions, we measured fine sediment concentrations and aquatic insect communities in a headwater stream below a logging road crossing. No actual logging or any other disturbance occurred in the stream catchment. Sampling was initiated two years before the road crossing was disturbed, and continued for three years after. In this sense, the study was a stream reach-scale manipulation providing pre-disturbance baseline measurements and comparisons to a nearby reference stream. This approach integrated the realism of a natural stream reach study with the experimental control of a manipulative study by including pre-sedimentation measurements and excluding other catchment disturbances. The study differed from most previous sediment impact studies in that it 1) focused on time course analyses over five years, examining trends before and after the sedimentation at the impacted and reference sites, 2) was not confounded by other catchment disturbances such as canopy modification

or altered nutrient fluxes, and 3) quantified and characterized stream bed sedimentation.

Materials and methods

Study site

The study was conducted in the Turkey Lakes Watershed, a 10.5-km² area located approximately 60 km north of Sault Ste. Marie, Ontario, Canada, and 13 km inland from Lake Superior. The watershed is dominated by old-growth sugar maple (*Acer saccharum* Marsh.) with scattered yellow birch (*Betula alleghaniensis* Britton). The two streams sampled for this study were first-order streams draining hardwood forest catchments with channel slopes of about 10%. Average bankfull width of the reference stream was 1.9 m, while the sediment impacted stream had an average bankfull width of 1.6 m. Study reaches were 120 m in both streams. Bottom substrates were characterized by visual estimates of percent composition of sand, gravel, rubble and boulder in 10-m sections and averaged over the study reaches. The reference stream was 13% boulder, 34% rubble, 48% gravel and 5% sand, and the sediment impacted stream was 8% boulder, 25% rubble, 58% gravel and 9% sand.

Sampling sites in the manipulated catchment were downstream from an old road crossing. The road crossing (gravel road over culvert) was considered stable in that there were no indications of erosion or sediment inputs from the road at the beginning of the sampling period, and pre-disturbance benthic sediment concentrations were similar to sites without road crossings. For example, in the fall of 1996 (one year before the disturbance) average fine sediment concentration at the manipulated site was 680 g m⁻², and at the reference site was 280 g m⁻², while in three other nearby streams without road crossings average concentrations were 214, 536, and 1380 g m⁻². In preparation for logging in the watershed, the road crossing at the manipulated site was upgraded to facilitate logging truck access. The road was widened, boulders removed, and ditches cleared on either side to increase drainage. No sediment control measures were taken during the road amendment.

Benthic sediment concentrations

Procedures for collection, processing, and analyses of fine sediment samples are given in Kreutzweiser and

Capell (2001). Briefly, fine sediments ($<250\ \mu\text{m}$) were collected from stream beds with a 17.5-cm diameter core sampler in which the substrates were stirred vigorously with a steel rod to suspend the fine particles. A manual bilge pump with a 2-mm screened intake was used to pump 3.8 l of water with suspended sediments through a 250- μm sieve into a collection bucket. The water and sediment that passed through the sieve were stirred vigorously and a 25-mL subsample was drawn with a syringe and placed in a glass vial. The sediments were suction filtered on to 1.5- μm glass fibre filters, dried at 60 °C for 48 h, weighed, then placed in a muffle furnace at 500 °C for 2 h, and weighed again to determine the organic (ash-free dry weight) and inorganic fractions. This provided a sediment size fraction of 1.5–250 μm and the sediment concentration on stream beds was expressed as g m^{-2} . Ten sediment bed load samples were collected from each stream at each sample time. The samples were collected in a stratified random manner from each section; immovable substrates, plunge pools, and distinct depositional areas were avoided to ensure that sediment bed load measures were taken from similar habitats in which stream benthos was measured. Benthic sediment samples were collected in late spring (end of May) and in early fall (early October) of each year from 1995 to 1999. The road amendment occurred in the late summer of 1997 so the sampling regime provided 5 pre-disturbance and 5 post-disturbance measurements. Differences in trends over time of sediment loading on stream beds between reference and disturbed sites were examined following the BACI approach of Stewart-Oaten et al. (1986), but with randomization tests to detect nonrandom patterns in differences between reference and the sediment impacted site before and after the disturbance.

The average surface cover of deposited sediment was visually estimated in both streams under base-flow conditions at the end of the study period. Fifteen 1-m wide transects were placed randomly within each study reach. The percent of stream bottom covered by fine sediment, in 5% increments, was visually estimated at each transect by a single observer. This approach was used by Zweig and Rabeni (2001) and found to be as effective as calculated embeddedness measures in representing the extent of deposited sediments as a proportion of stream bottom area. Differences between the two sites in sediment surface cover were compared by ANOVA following an arcsine/square root transformation for percentages.

Aquatic insect communities

Aquatic insect communities were quantitatively sampled with modified Surber-type samplers. The collection net was 0.32 m wide with a 363 μm mesh, and the side walls were 1 m long. The side walls were held in place by rods driven into the stream bed at the upstream ends of the samplers, and the total area sampled was 0.32 m^2 . Ten replicate samples were collected by random allocation at each study reach each time the sediment core samples were taken. Plunge pool or boulder locations were excluded. The contents of the collection net were preserved in 10% formaldehyde and dyed with phloxine B to facilitate sorting under a magnifier. Insects were identified to genus or species, except for Chironomidae. Seven of the 10 sorted and identified samples from each site and time were randomly selected, dried at 60 °C for 72 h, weighed and recorded as total insect biomass per sample. On every sampling occasion, insects comprised more than 99% of the total macroinvertebrates collected, and other invertebrates occasionally encountered (e.g., gastropods, oligochaetes, hirudiniids) were excluded from the analyses.

Insect community data analyses

This was an un-replicated, whole-stream study design with emphasis on comparisons over time among response variables between the manipulated and reference streams. Because the insect communities were sampled twice each year (in late May and early October), the data were separated into spring communities and fall communities and analyzed separately to reduce seasonal variability among annual comparisons. Aquatic insect community structure was examined by a multivariate ordination procedure, nonmetric multidimensional scaling (NMDS) (Clarke 1993). For these analyses, data were restricted to those taxa that contributed at least 1% of the total abundance on at least two sample dates. This adjustment reduced the confounding effects of highly uncommon species in the data. Data were log transformed ($\log x + 1$) to down-weight the influence of abundant taxa and take into account rarer taxa as well. Average abundance for each taxon at both sites at each sample time was used in the NMDS ordination. Two-dimensional scatter plots were constructed from the first two ordination axes in NMDS with each point representing the average community structure of a given site at a specified sample time. The NMDS points for both sites were tracked over time through ordination space to determine if a divergence of trends occurred after the

disturbance. Differences in community structure between sites were determined by a multivariate analysis of similarity (ANOSIM), which is based on a non-parametric permutation procedure applied to the ranked Bray-Curtis similarity matrix underlying the NMDS ordination (Clarke 1993). The similarity matrices were further analyzed to provide multivariate measures of community stress. An index of multivariate dispersion (IMD) was computed which contrasted the ranked similarities between sites averaged over time. The IMD values range from -1 to $+1$. Values near zero imply no differences between groups, whereas values near the extremes infer that variability in multivariate structure differs between groups. Relative dispersion values were also computed for both sites, with larger values corresponding to greater within-group variability (Clarke and Warwick 1994).

Insect communities were further examined by a number of univariate metrics taken from Davis et al. (2001). These included mean abundance, biomass, taxon richness, percent shredders (functional feeding group classifications following Merritt and Cummins 1996), percent scrapers, percent filterers, percent EPT (Ephemeroptera, Plecoptera, Trichoptera), and percent Chironomidae. We also included Shannon-Wiener diversity and Margalef's richness indices. Trends over time in these univariate metrics were compared between sites by a repeated measures orthogonal polynomial coefficient analysis (Meredith and Stehman 1991). This approach accounts for the correlation among samples drawn from the same experimental unit over time. It also focuses on the response curve, i.e., a comparison of changes over time among groups, and does not require the stringent assumptions of a split-plot analysis. Trends were examined by linear response models, and the treatment X time interaction in the analysis was the primary term of interest for detecting significant treatment effects. Significant treatment X time interaction ($p < 0.05$) indicated that trends over time were not parallel between the two sites. A significant divergence over time was taken as a treatment (sedimentation) effect. Abundance data were log transformed, and percent data were arcsine/square root transformed to improve normality and increase homogeneity of variances. The Shannon-Wiener diversity and Margalef's richness indices were based on average abundance data at each time, precluding statistical comparisons between sites, and trends over time for these indices were examined visually.

Results

The road amendment resulted in a significant increase (BACI $p=0.0032$) in fine sediment bed load (from an average 2-yr pre-disturbance level of 800 g m^{-2} to about 5000 g m^{-2} after the disturbance) that was sustained for the duration of the study period. Most of the increased sediment (about 80%) consisted of particles in the coarse silt size range of $40\text{--}63 \text{ }\mu\text{m}$ (Kreutzweiser and Capell 2001). By the end of the study period, the average visible surface cover of deposited sediment at the manipulated site was 46.7% and at the reference site was 14.5% (ANOVA $p=0.0006$). The inorganic fraction of fine sediments increased from a pre-disturbance average of 85.9% to 97.1% (ANOVA $p=0.0299$) at the manipulated site, while the average inorganic fraction of fine sediments at the reference site was 82.1% before the disturbance and 81.2% after.

There were no measurable effects of sedimentation on total insect abundance in spring samples. Mean abundance of spring samples was consistently higher in the reference than in the manipulated site, but this occurred before as well as after the sediment inputs (Figure 1A), and trends over time were essentially parallel (time X treatment interaction $p=0.6737$). There was also little indication of a sedimentation effect on abundance of aquatic insects in fall samples. By the fall of 1999, total abundance at the reference site showed a sharp increase (Figure 1B), and this resulted in a marginally significant divergence in trends (time X treatment interaction $p=0.0750$). The large error associated with the mean estimate in fall 1999 indicates a highly patchy distribution within the study reach, and most of this increased abundance (90%) was comprised of Chironomidae. When the fall 1999 samples were removed from the repeated measures analysis, trends over time between sites were similar (time X treatment interaction $p=0.5269$).

There was also no indication that the increased sediment affected total insect biomass. Biomass in spring samples from the manipulated site tended to be lower than in the reference, but this trend was apparent before the sediment inputs (Figure 1C) and there were no detectable differences in time trends overall (time X treatment interaction $p=0.8581$). Total biomass in fall samples was more variable, and while there tended to be a post-disturbance decline at the manipulated site, followed by an increase the following year (Figure 1D), no significant difference in time

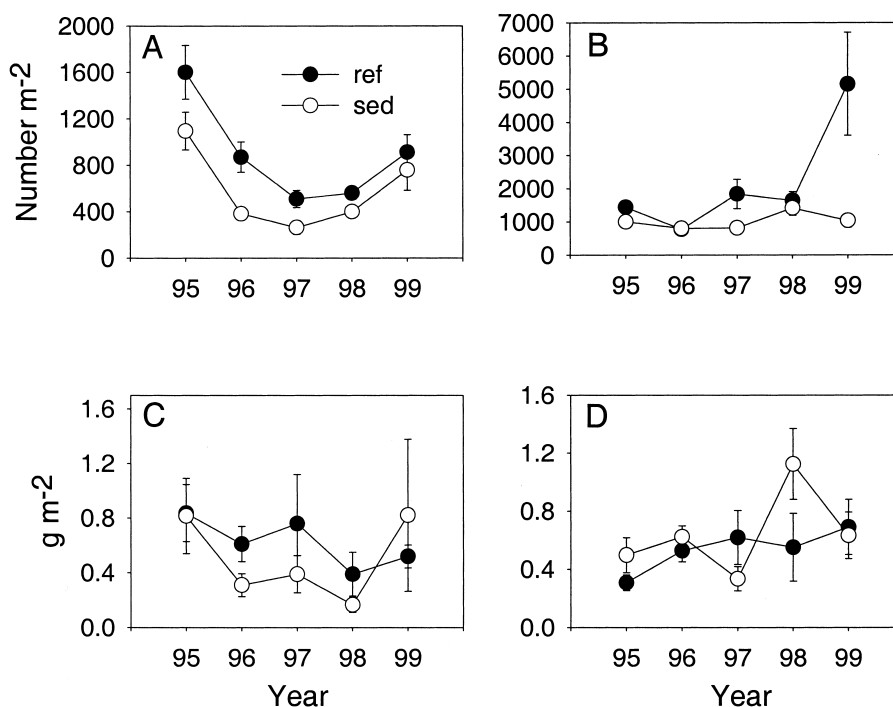


Figure 1. Mean (± 1 SE, $n=10$) abundance of aquatic insects in spring (A) and fall (B), and mean total biomass of aquatic insects in spring (C) and fall (D) samples from the reference (ref) and sediment (sed) sites. Disturbance occurred in late summer 1997.

trends could be detected (time $\times$ treatment interaction $p=0.9800$).

The NMDS ordination showed that among both spring and fall samples, there were distinct differences in community structure between the two sites both before and after the sediment inputs (Figure 2). The ANOSIM procedure detected significant differences between sites ($p < 0.01$) at all paired site comparisons at each sample time both before and after the disturbance. The distinct grouping of samples by site indicated that there were greater differences between sites than between years, although there were also considerable shifts in community structure over time.

In spring samples, community composition changed in roughly the same direction at both the reference and disturbed site before the manipulation, but to a greater extent at the manipulated site (Figure 2). There was little change in community structure at the reference site between the third and fourth sample times (last pre- and first post-disturbance) while a slightly greater shift occurred at the manipulated site. Between the fourth and fifth sample time, the community structure at both sites shifted in ordination space but the distance was slightly greater and in a different direction at the manipulated site than at the

reference. Changes in community structure of fall samples between the first two (pre-disturbance) sample times were of similar magnitude and direction (Figure 2). Post-disturbance trends among fall communities were more divergent. Between sample times 2 and 3, there was a comparatively large shift in community structure at the reference site, while the shift in community structure at the manipulated site was much smaller and in the opposite direction. Over the next two post-disturbance sample times, the shifts in community structure at the manipulated site were slightly greater and in different directions than at the reference site. Overall, the ordination indicated that there were distinct differences in community structure between the manipulated and reference sites, before and after the sediment inputs. However, the slightly divergent trends through ordination space over the post-disturbance period suggested some changes in community structure in response to the sedimentation.

The index of multivariate dispersion (IMD) value for a comparison of variability in multivariate structure between sites averaged over time was -0.16 . This low value indicated that there was little difference in multivariate structure between sites overall, and the negative sign indicated that there was greater

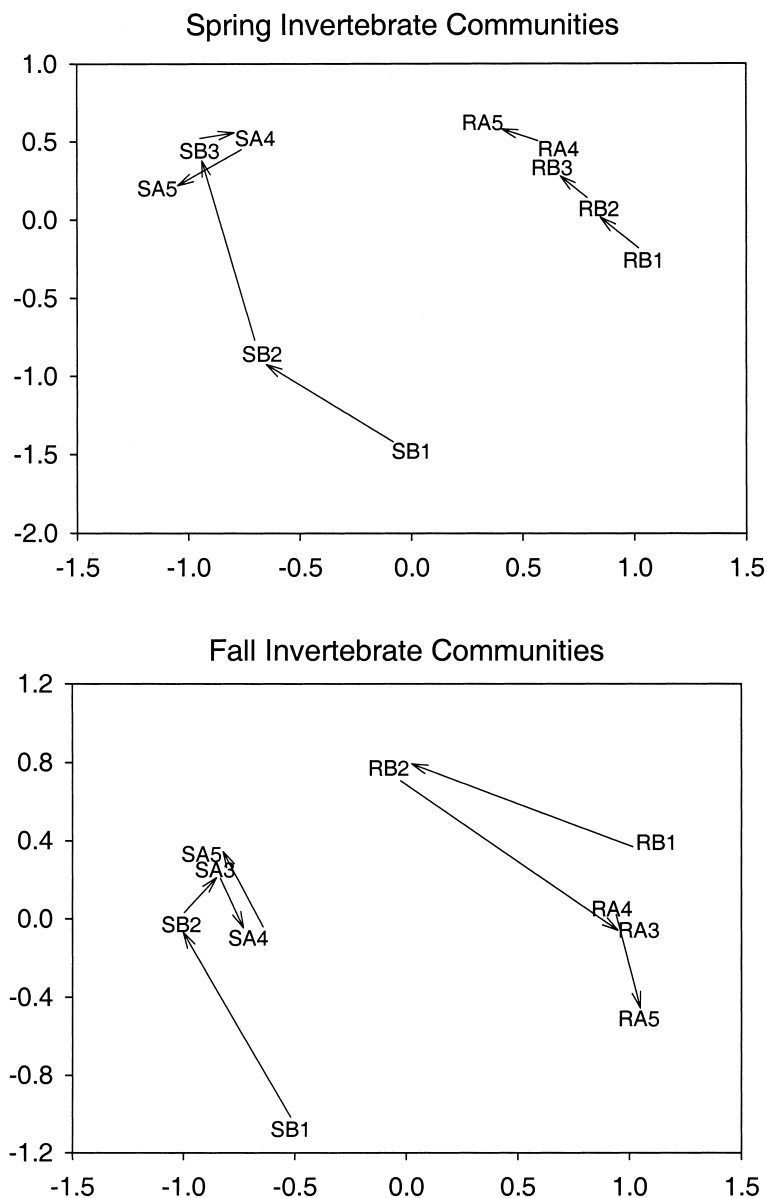


Figure 2. Two-dimensional plots from NMDS ordination of spring and fall invertebrate communities. At each point, the first letter indicates site (S = sediment site, R = reference), second letter indicates before (B) or after (A) the sediment inputs, and the numbers indicate sequential sample times.

variability among reference samples than among manipulated site samples (see Clarke and Warwick (1994) for interpretation of IMD). This was further indicated by the relative dispersion values based on the ranked Bray-Curtis similarity matrix. The value for the reference site was 1.076 and for the sediment site was 0.924, indicating that there was slightly higher within-group variability in multivariate structure at the reference site than at the manipulated site.

The subtle changes in community structure at the sediment site in comparison to the trends at the reference site were examined further by the community-level, univariate metrics. The Shannon-Wiener diversity and the Margalefs' richness indices both showed a declining post-disturbance trend among spring communities at the manipulated site in comparison to the reference site (Figure 3A, Figure 3C). Trends in diversity and richness were less apparent among fall

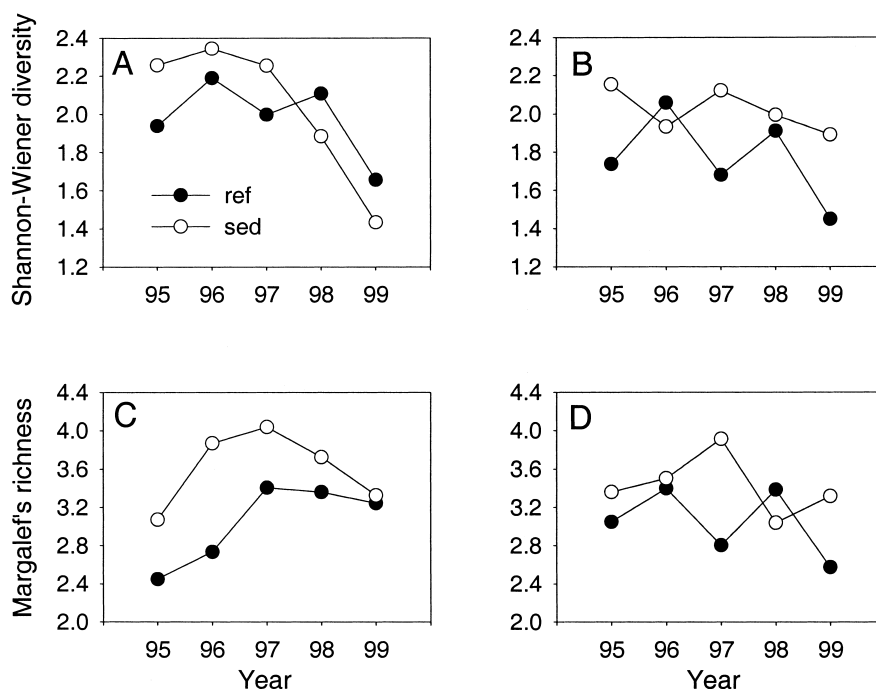


Figure 3. Shannon-Wiener diversity (A and B) and Margalef's richness indices (C and D) based on mean relative abundances from reference (ref) and sediment site (sed) samples in spring (A and C) and fall (B and D) of each year. Disturbance occurred in late summer 1997.

communities, and year to year differences were as great or greater than site differences with no clear indication of a sediment effect on fall communities (Figure 3B, Figure 3D).

Among the six community metrics that were examined by the repeated measures analysis, differences between sites in time trends overall were small but tended to be more pronounced in spring than in fall. Three metrics showed significant divergence over time in spring (Figure 4). The percent shredders in spring communities clearly declined at the manipulated site after the disturbance (time X treatment interaction $p < 0.0001$). The spring EPT component also declined at the manipulated site, but reflected a similar decline at the reference site over the same time. The greatest difference in percent EPT between sites occurred before the disturbance, and this contributed to the significant difference (time X treatment interaction $p < 0.0001$) in EPT trends overall. The percent Chironomidae increased at the manipulated site after the disturbance, and while there tended to be an increase at the reference site as well by the end of the sampling period, a significant difference in time trends between sites was detected (time X treatment interaction $p = 0.0006$). Taxa richness declined at the impacted site (Figure 4), but most of this de-

cline occurred before the disturbance and the time trends overall were only marginally different between sites (time X treatment interaction $p = 0.0897$). There were no clear treatment-related patterns or significant differences in time trends between sites among spring scraper and filterer community metrics.

Only one of six community metrics among fall samples showed a significant divergence in trends (Figure 5). The filterer component of fall communities at the sediment site declined slightly after the disturbance, and as a result of increases in the percent filterers over the same period at the reference site, the time trends were significantly different between sites (time X treatment interaction $p = 0.0246$). The significant decline in percent shredders that occurred in spring samples was less apparent in fall samples. There was a tendency for lower percent shredders in fall samples on two occasions after the disturbance, but no significant difference between sites could be detected (time X treatment interaction $p = 0.6887$). There was no evidence of sediment impact among the four remaining metrics for fall communities (Figure 5).

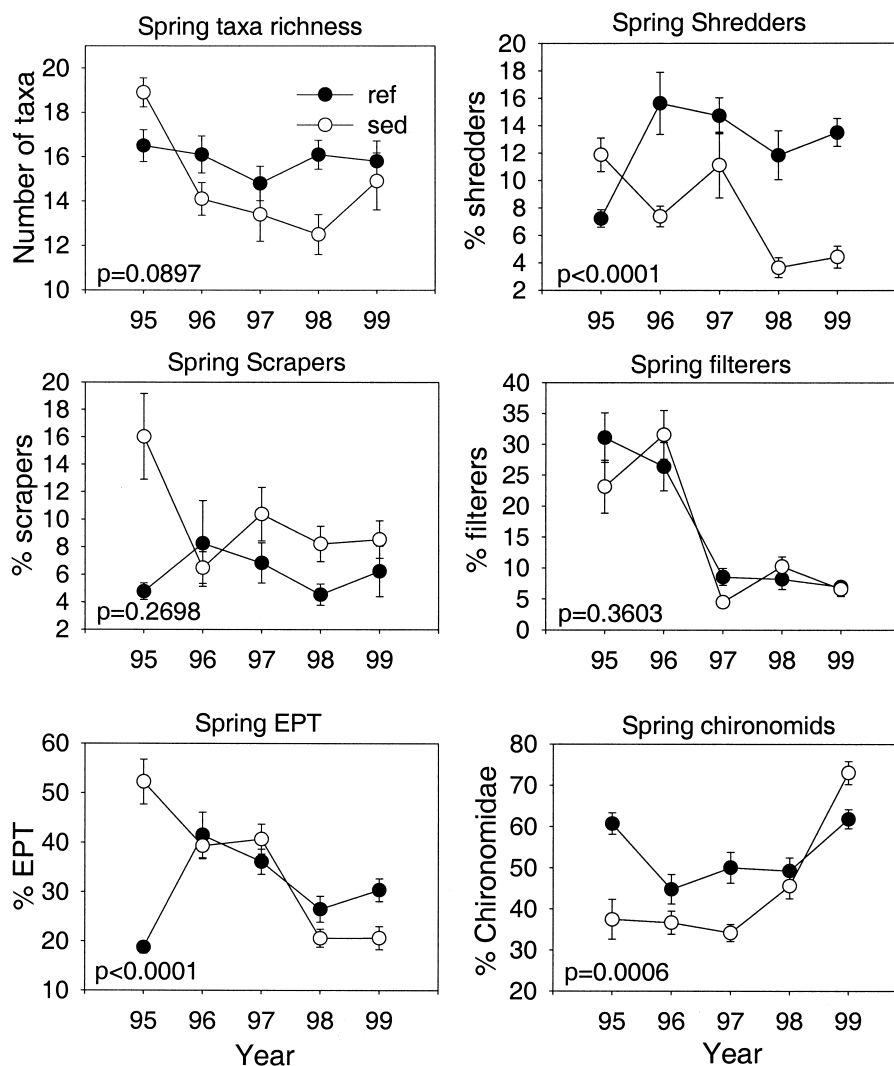


Figure 4. Community-level metrics (mean $\pm$ 1 SE, $n=10$) for reference (ref) and sediment site (sed) samples in spring of each year. Probability values are from time X treatment interaction terms in repeated measures orthogonal polynomial coefficient analyses. Disturbance occurred in late summer 1997.

Discussion

Despite large and sustained increases in fine sediment deposition in a forest stream (about a 7-fold increase for 3 yr), only small changes in aquatic insect community structure could be detected. The NMDS ordination demonstrated that over the 5-yr study period there were generally greater differences in community structure between sites than among years, but that these differences were as large or larger before the sediment inputs. There was only a subtle divergence in ordination time trends between sites after the disturbance. The multivariate measures of community

stress (indices of multivariate dispersion) indicated little difference between sites overall, with a tendency for higher variability in multivariate structure among reference samples. Of the univariate metrics examining spring and fall communities at the manipulated and reference sites, the clearest indications of an effect on community structure were small reductions in diversity and richness of spring communities. These were primarily the result of a decline in the proportion of spring shredders after the disturbance, accompanied by a significant increase in the percent Chironomidae. None of these metrics gave evidence of a sediment impact on fall communities.

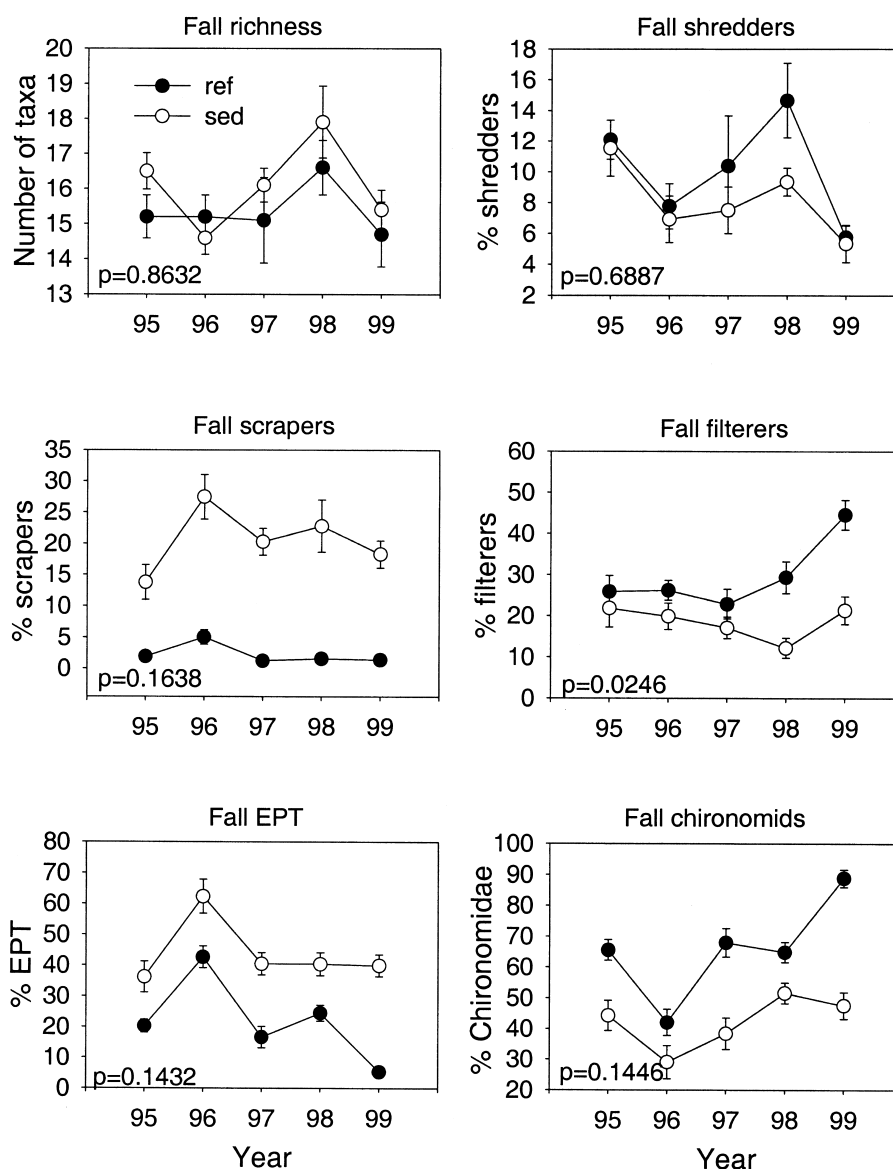


Figure 5. Community-level metrics (mean $\pm$ 1SE, $n=10$) for reference (ref) and sediment site (sed) samples in fall of each year. Probability values are from time X treatment interaction terms in repeated measures orthogonal polynomial coefficient analyses. Disturbance occurred in late summer 1997.

The decline in Shannon-Wiener diversity and Margalef's richness indices of spring communities at the manipulated site reflected shifts in relative abundance among taxa, not loss of species. Richness tended to be lower at the manipulated site, but the greatest reduction in mean number of taxa occurred before the sediment inputs. The increase in the proportion of chironomids in spring communities at the manipulated site is consistent with their general tolerance of sediment-degraded habitats (Davis et al. 2001),

although Angradi (1999) showed differential responses among Chironomidae subfamilies to sediment additions. However, changes in the proportion of chironomids also occurred in fall communities at the reference site suggesting that natural environmental factors are as likely to cause shifts of this magnitude as the sediment inputs from the road crossing. The significant decline in the proportion of spring shredders suggests that the sedimentation affected the habitat or resource quality for this functional group.

Since there were no catchment disturbances to alter organic matter inputs at the sediment site, the effects on shredders presumably occurred from siltation of the coarse particulate matter on which this functional group feeds. This could result from physical interference by sediment particles on feeding behaviour, or from reduced palatability to shredders through a reduction in microbial colonization and conditioning of particulate organic matter surfaces. There was no indication of these effects on fall shredders, and this may have resulted from shredder utilization of fresh litter inputs from leaf fall before impingement of organic matter surfaces by fine sediments from high-water events.

It is noteworthy that the multivariate ordination, the analysis of similarity, and many of the univariate community metrics showed a clear distinction in community structure between the two sites, before the sediment inputs. The reference site was selected because of its apparent similarity to the manipulated site. Catchment characteristics, bankfull widths, study reach lengths and orientation, forest composition, canopy structure and density, environmental influences (given the proximity of sites), and substrate qualities were similar at the two streams. The differences in aquatic insect community structure were consistent over the study period, with greater differences between sites than among years within a site. The distinct natural differences in community structure of these paired streams underscores the need for establishing baseline or pre-treatment conditions to which after-impact conditions can be compared in unreplicated, whole-stream experiments such as this one (Stewart-Oaten et al. 1986). A paired-stream comparison after the disturbance would have indicated distinct differences in aquatic insect community structure suggesting a marked effect of sedimentation on benthic insects, but it is clear that distinct differences in community structure in these streams were present before the disturbance.

Many other studies have shown significant effects of fine sediment deposition on stream benthic invertebrates (Tebo 1957; Barton 1977; Lenat et al. 1981; Erman and Ligon 1988; Barton and Farmer 1997; Fossati et al. 2001). These, and others, have indicated that benthic insect communities are generally intolerant of fine sediment deposition, and that catchment disturbances resulting in sediment inputs to streams are likely to cause reductions in abundance and/or changes in community structure (Campbell and Doeg 1989; Waters 1995; Wood and Armitage 1997). How-

ever, across a range of sedimentation studies, results have been variable. In extreme cases where sediment inputs have been extensive, aquatic insects have clearly been adversely affected (Somer and Hassler 1992; Fossati et al. 2001). In other studies, conclusions are drawn from small-scale, controlled experiments and short-term responses (Rosenberg and Wiens 1978; McClelland and Brusven 1980). These are useful for examining response mechanisms or processes but may not adequately represent community-level responses to fine sediment deposition because they do not incorporate all the biophysical properties and processes or temporal dynamics inherent in natural systems. Many studies have made comparisons of benthic invertebrate communities among disturbed and reference sites, or across a gradient of sediment deposition or manipulation, after the sediment input disturbance. Among these, some have detected significant adverse effects (Lenat et al. 1981; Hogg and Norris 1991; Barton and Farmer 1997; Zweig and Rabeni 2001), while others have reported subtle or no effects on stream benthos (Murphey et al. 1981; Culp et al. 1985; Fairchild et al. 1987; Richards and Bacon 1994; Angradi 1999).

Some of this discrepancy among studies may arise from differences in deposited sediment characteristics and amounts. Many of the previous studies have described sediments ranging from silt- to sand-size particles up to 2mm, have not reported organic or inorganic fractions, and have quantified in terms of a measure of surface cover or concentrations of suspended sediment additions rather than actual benthic sediment concentrations. Differential invertebrate responses may reflect differences in fine sediment composition in terms of particle size and structure. While an association between substrate particle size and benthic invertebrate community structure has been demonstrated across a broad range of sizes (Minshall 1984; Bourassa and Morin 1995), it is less clear how differences in particle sizes or composition across fine sediment size groups (< 2mm) may affect benthic communities. In the present study, we showed that sediment inputs from the road crossing were mostly comprised of small particles in the 40-63 μm size range, consisted almost entirely (97%) of inorganic materials, and resulted in an average benthic concentration of over 5000 g m^{-2} , roughly a 7-fold increase. Given this magnitude of sediment increase, and that the small particle sizes of this sediment were in the range of particle sizes that have been shown to interfere with hydrological

exchange processes and stream bed quality (Schälchli 1992; Brunke and Gonser 1997), we expected to find marked effects on the benthic insect community.

The fact that the subtle changes in insect community structure of this study were less than in many previous studies of sediment inputs at similar magnitudes may also reflect differences in sampling frequency and study duration. Most previous studies focused on short-term (several weeks to months) measurements, or one-time comparisons, which may have detected short-term impacts, while sampling in the present study was integrated over a longer time period. More frequent sampling immediately after the sediment disturbance might have detected significant, short-term changes in insect abundance and diversity. However, our sampling frequency (5 times over 2 ½ yr after the disturbance) integrated responses over the longer term taking into account the natural stochastic variability over time and may have provided a more realistic indication of ecologically-significant effects of the disturbance than measures of short-term changes.

The minimal effects of sedimentation on aquatic insects communities of this study may have occurred because these stream substrates were largely comprised of finer materials (about 55% gravel, 7% sand) before the sediment inputs and the insect communities may have already been naturally adapted to finer substrates. A fine sediment load of this magnitude and duration in a stream with more coarse materials may have had a greater effect on benthic insects. On the other hand, stream beds of more coarse materials are less prone to embeddedness (the proportion of substrate particles covered by fine materials), and therefore may not be more susceptible to sedimentation effects on benthos.

Conclusions

Fine sediment inputs from logging activities, particularly road crossings, have been implicated in stream habitat degradation (Campbell and Doeg 1989; Webster et al. 1992; Binkley and Brown 1993; Keenan and Kimmins 1993). However, except under extreme cases, effects of sedimentation on benthic communities in large-scale field studies have been equivocal. In some cases, responses by benthic invertebrate communities to sedimentation from logging operations may have been confounded by concomitant influences of activities such as canopy modifications or

other catchment disturbances. The large-scale experimental approach of the present study integrated the realism of a whole-stream study with the control of a manipulative study by including pre-sedimentation measurements and excluding other confounding catchment disturbances. The road crossing at the manipulated site was upgraded in an actual logging operation to accommodate haul trucks and other equipment, so the resulting sediment inputs to the stream should be typical of road-crossing disturbances in similar hardwood forest terrain. In this regard, the present study may provide a more realistic measure of community-level effects on benthic insects of sedimentation at a magnitude associated with logging activity, than many previous studies. That only subtle effects on aquatic insects were detected indicates that benthic insect communities of northern hardwood forest streams can exhibit some degree of resistance to fine sediment inputs at this magnitude. Additional experimental studies at this or larger scales in other hardwood forest watersheds would be useful to further evaluate the degree to which stream insect communities can tolerate fine sediment inputs from forestry operations. The implementation of sediment control strategies under best management guidelines for stream crossings would further reduce the likelihood of adverse effects on stream habitat quality and benthic communities.

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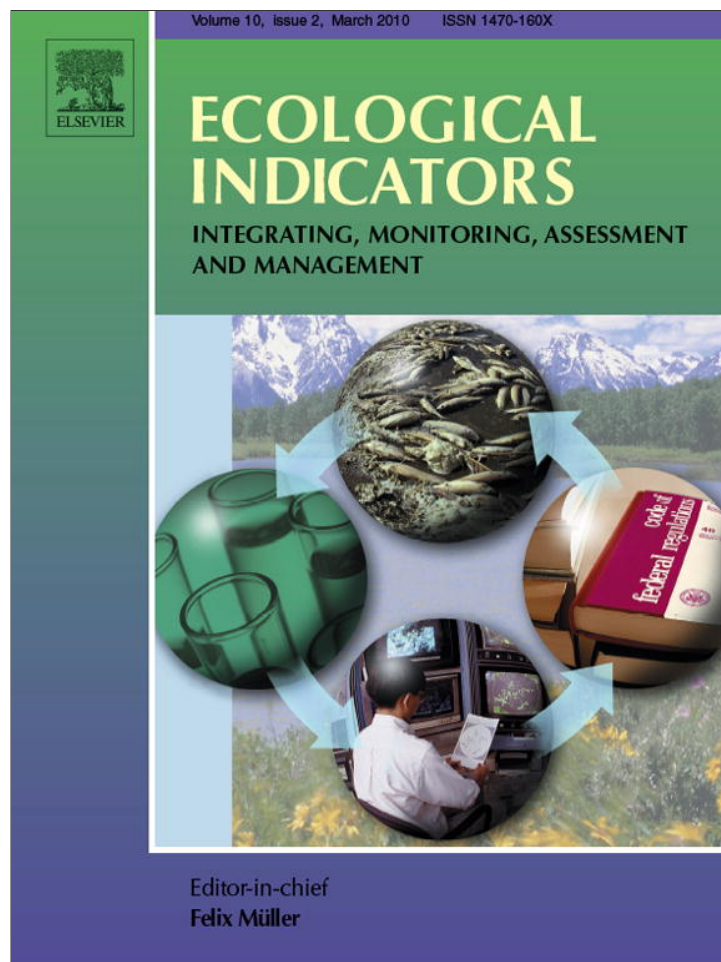
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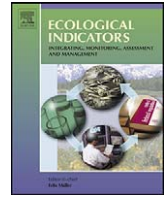
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The effects of land use on environmental features and functional organization of macroinvertebrate communities in Patagonian low order streams

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ABSTRACT

Benthic invertebrates, water quality variables, chlorophyll *a* and particulate organic matter (POM) were studied in 18 sites of mountain streams in Patagonia (Argentina) subjected to six different land uses: native forest, pine plantation, pasture, harvest forest, urban and reference urban. Three streams of each land use were studied in March 2006. Macroinvertebrates were sampled in three riffles and three pools ($n = 108$) and biomass of detrital fractions of POM were quantified. Overall benthic POM biomass was higher at native and harvest forest than pastures, whereas fine fraction (FPOM) was higher in harvest forest than in pastures. Regarding to autotrophic subsidies bryophytes were the only that changed consistently among uses. These differences in energy resources were correlated with changes in community attributes. Shredder richness was clearly higher at native and harvest forest than exotic pine plantations, collector gatherers density was consistently high at harvest sites, and total density was significantly higher at urban and harvest forest. Multidimensional scaling ordination based on macroinvertebrate density data showed a clear separation of forested (either native or exotic species) from riparian modified areas (pasture, urban and harvest sites). Environmental variables having explanation power on macroinvertebrate assemblages were mostly related with: detritus availability (wood and leaves biomass) and impairment (total phosphorous and % sand). These results confirm that macroinvertebrate assemblage structure in Patagonian low order streams can be altered by land use practices. Among guild structure measures, those indicators based on benthic community functional attributes, shredders richness and collectors density, resulted good candidates to assess land use impacts. On account of riparian corridor management may be critical to the distribution of benthic taxa, the maintenance of good conditions of vegetation adjacent to rivers will enhance water quality and the environment for highly endemic headwater communities of Patagonian streams.

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1. Introduction

Land use changes have resulted in profound modifications of running waters in mountain streams of temperate areas (Allan et al., 1997). In developed basins the conversion of riparian margins from forest to grassland has been one of the most widespread phenomena. Moreover, shifts in land use practices (agriculture to silviculture) have also determined the replacement of native by exotic vegetation as for example pine plantations, sometimes affecting riparian corridors (Baillie and Davies, 2002; Albariño and Balseiro, 2002). Changes in the adjacent vegetation can alter the complex biotic and abiotic processes that govern the function and attributes of aquatic communities including benthic macroinvertebrates (Roy et al., 2003; Thompson and Townsend,

2004; Utz et al., 2009). Early works revealed the consequences that the loss of riparian vegetation can have on physical properties of streams (Newbold et al., 1980). The extirpation of riparian corridors can alter the light and temperature regimes and lead to increases in nutrients and augment primary production (Scarsbrook and Halliday, 1999; Kratzer et al., 2006). The establishment of pastures for agricultural grazing can significantly reduce the amount of allochthonous leaf material entering the river, which consecutively produces a significant effect on trophic structure (Abelho and Graça, 1998; Miserendino, 2005). Scrimgeour and Kendall (2003) found that shredder biomass was significantly lower in Canadian streams under intense pasture development than in those without pastoral use. Loss of insect species (Collier et al., 2001) and higher invertebrate densities have been documented in New Zealand streams in which forest was converted to pasture (Quinn et al., 1997). Thompson and Townsend (2004) observed significant differences between shredders and collector filterer biomass in basins under different land uses such as pasture conversion, native, or exotic vegetation. In a recent

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study of 60 sites in streams subjected to different land uses Kratzer et al. (2006) found that macroinvertebrate communities responded considerably to land use and most species variation was accounted by variables related to watershed and riparian conditions. Similarly, Compin and Céréghino (2007) detected significant responses of functional feeding groups to landscape alterations. They recognized macroinvertebrate trophic measurements useful for assessing river health with development along riparian zones with collector gatherers showing higher percentages in urban landscapes and scrapers and predators increasing in agricultural landscapes.

The functional structure along streams in Patagonian mountains seems to correspond well with the predictions of the RCC (River Continuum Concept, Vannote et al., 1980) with biomass of shredders decreasing and collector's biomass increasing downstream from the headwaters (Miserendino and Pizzolón, 2003; Miserendino, 2007). Some differences in the functional attributes of benthic communities between rivers bordered by the native *Nothofagus* forest and those dominated by *Salix* spp. have been

observed, the former showed higher biomass of shredders while *Salix* bordered rivers had dominance of collector and filterers (Miserendino and Pizzolón, 2003, 2004). In recent experimental studies, Albariño and Balseiro (2002) suggested that the litter processing mechanisms would be affected as a consequence of the substitution of native *Nothofagus* forest by exotic pine plantations.

Streams in Patagonian mountains are subjected to different land use practices; the most common are pasture conversion, exotic plantations, and wood collection/deforestation. The consequences that these activities are having on functional structure of the local, highly endemic biota are not explicitly known. A better understanding of these relationships may contribute to the development of improved land use management practices and to purpose mitigation plans on freshwater ecosystems (Morley and James, 2002; Danger and Robson, 2004).

The aim of this study was to assess the impacts of different land use practices (native forest, exotic, pasture, wood collection, reference urban and urbanizations) on physicochemical characteristics and macroinvertebrate community, and also to evaluate

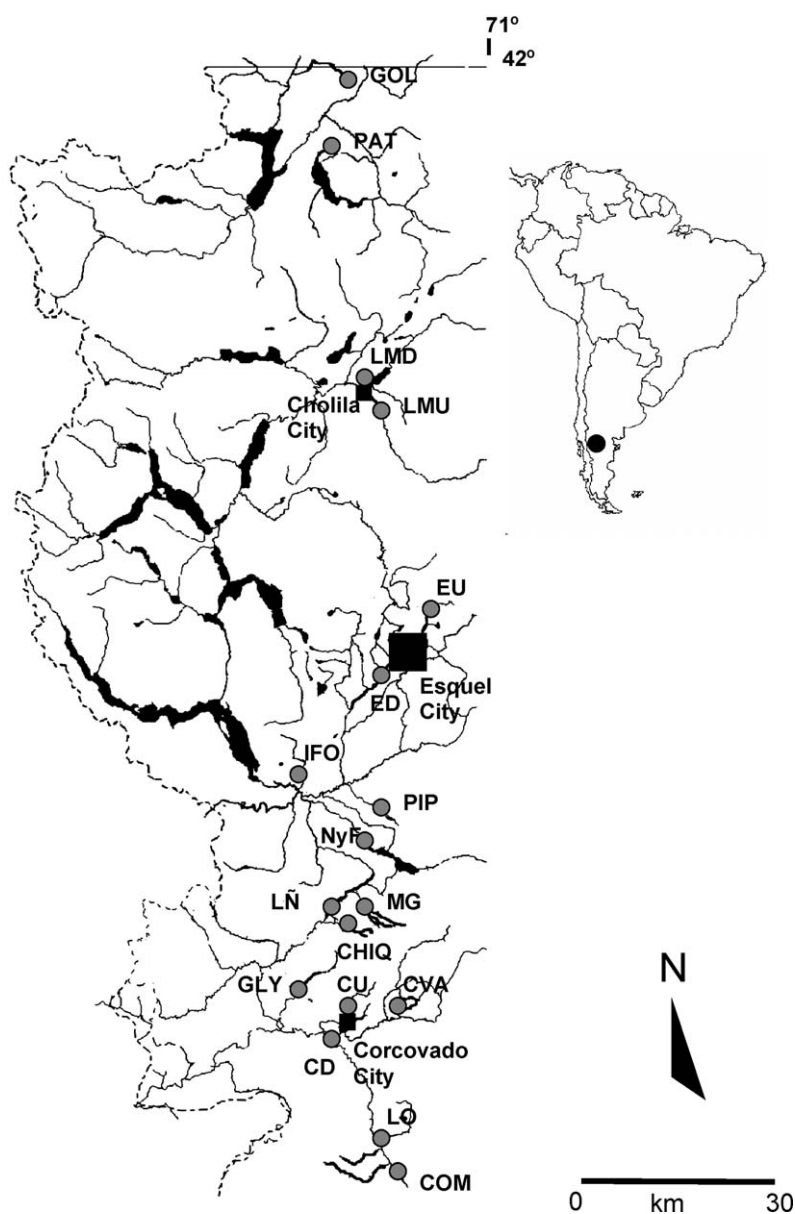


Fig. 1. Map of study area with locations of sampling points, Chubut, Argentina. Site codes are shown in Table 1.

the usefulness of guild structure indicators in mountain streams of northwest Patagonia (Argentina).

2. Materials and methods

2.1. Study area and site selection

The study area is located in the Northwest of Chubut Province (Fig. 1) in the ecotone between the Subantarctic forest and the Patagonian steppe, and exhibits a marked altitudinal gradient. Perennial (*Austrocedrus chilensis*, *Nothofagus dombeyi* and *Maiteinus boaria*) and deciduous tree species (*Nothofagus pumilio*, *Nothofagus antarctica*) constitute the Subantarctic forests. The shrub and herbaceous strata are characterized mainly by *Chusquea culeou*, *Berberis buxifolia*, and *Fuchsia magellanica*. The valleys are dominated by the herbaceous, shrub-like steppe vegetation (Tell et al., 1997). Exotic vegetation is composed mainly by the implanted forest of *Pseudotsuga menziesii*, *Pinus ponderosa*, *Pinus radiata* and *Pinus lambertiana*, in many cases species are included in the riparian corridor and replacing most of the native species.

The survey was conducted on March 2006 on 18 sites in 15 rivers. We choose this month because most streams have stable flow conditions (Coronato and del Valle, 1988). Three sampling sites were placed on non-impacted native *Nothofagus* forest (Chiquito, Comisario, and Loro), these sites were considered as reference sites to test for impacts: pasture, pine plantation and wood collection.

Three sites on streams with native *N. antarctica* and *N. pumilio* forest with wood collection or extraction (Glyn, Cabeza de vaca, Pipo) were selected. At these sites the harvest was achieved during previous spring (3 months before the survey). The logging operations imply a selective cut of trees, which should exclude specimens from the river corridor. Operators also have to keep free the streams of debris and branches once the tree is logged. However during the survey we observed some river corridors having logged trees, and some watercourses obstructed with debris.

For pasture sites we selected three sites that had been cleared from the native *Nothofagus* forest and had been grazed intensively (Nant y Fall, Manguera and Los Nires). At the present pasture sites sustain cattle livestock mainly cow and sheep. The pine plantations sites corresponded to streams running (950–1200 m) through plantations of *Pseudotsuga menziesii*, *Pinus ponderosa*, *P. radiata* and *P. lambertiana* (Golondrinas, Patriada and Ifona). Three urban rivers (Las Minas, Carbón and Esquel) plus their respective reference sites (Las Minas ref, Carbón ref and Esquel ref) were also included in the survey (Fig. 1).

In order to know basin features, vegetation types and coverage, a LandSat image was analyzed and processed in the Dirección General de Bosques y Parques. Per each surveyed site we obtained sub-basin area, % of native vegetation coverage, % of exotic vegetation coverage, % pasture and % of urban areas. We included percentage of the basin corresponding to lakes and areas above the timberline.

2.2. Environmental characterization and macroinvertebrate analysis

Percentages of boulder, cobble, gravel, pebble, and sand in the reach were estimated at each site using a 1-m² grid. Current speed was measured in mid-channel on three occasions by timing a float (average of three trials) as it moved over a distance of 10 m (Gordon et al., 1994). Average depth was estimated from five measurements along one transversal profile across the channel with a calibrated stick. Discharge was obtained by combining depth, wet width and current velocity as in Gordon et al. (1994). At

each site, air and water temperature were measured with a mercury thermometer.

Specific conductance ($\mu\text{S cm}^{-1}$), pH, dissolved oxygen (DO, $\text{mg O}_2 \text{ l}^{-1}$) were measured with a Horiba U2-probe. For nutrient analyses water samples were collected below the water surface and kept at 4 °C prior to analysis and transported to the laboratory. Total suspended solids (TSS), total nitrogen (TN), total phosphorus (TP), nitrate plus nitrite nitrogen ($\text{NO}_3 + \text{NO}_2$), ammonia (NH_4), and soluble reactive phosphate (SRP) were analyzed following standard methods (APHA, 1994).

Quantitative macroinvertebrate samples were taken with a Surber sampler (0.09 m²; 250 μm mesh size). On each reach, three samples were taken from riffles and three from pools ($n = 6$). Samples were fixed *in situ* with 4% formaldehyde, and sorted in the laboratory under at least 5 $\times$ magnification. Macroinvertebrate species were identified using available keys (Fernández and Domínguez, 2001). Functional feeding groups (FFGs) were assigned using available references, knowledge of feeding modes (mouthpart morphology and behaviour), and analysis of gut contents (Merritt and Cummins, 1996; Albariño and Balseiro, 2002).

Macroinvertebrate total density (TD), total richness of taxa (TR), diversity (H: Shannon Weaver diversity index) and dominance (BPD; Berger–Parker dominance index) were calculated per site. Species richness, percentage and total density of shredders (S), grazer/scrapers (SG), collector filterers (CF), collector gatherers (CG) and predators (P) were also obtained (Ludwig and Reynolds, 1988).

2.3. Particulate organic matter and chlorophyll analysis

Detritus from benthic samples was divided into fine (250–1000 μm) and coarse (>1000 μm) particulate fractions (FPOM and CPOM, respectively). CPOM was separated into wood, leaves (mainly entire leaves), macrophytes, bryophytes and others (fragments of leaves, grass, seeds, roots, buds, etc.) (Miserendino, 2007). All fractions were dried (105 °C for 4 h) and weighed on an electronic balance (Scout Pro) to ± 0.003 g. Total benthic particulate organic material (BPOM) was obtained by adding FPOM and CPOM.

Algal biomass (as chlorophyll *a*) was determined by scraping algae from five randomly selected rocks (length range: 5–16 cm) collected within a 20 m run/riffle reach at each site. Scrapings were brought back to the laboratory and drawn on to GF/FF filters. Chlorophyll *a* was extracted from pulverized filters in 90% acetone and the extract was measured, spectrophotometrically as described by Wetzel and Likens (1991).

2.4. Statistical analyses

Land use differences in physicochemical parameters and functional groups attributes were tested using one-factor analysis of variance, or where assumption of normality was violated, Kruskal–Wallis tests. Location of significant differences was tested using Tukey's post hoc test for ANOVA and Duncan test for Kruskal–Wallis (Sokal and Rohlf, 1995). Variability within and among groups of selected streams was assessed by means of a similitude matrix based on taxa percentage composition (Bray Curtis matrix) (Ludwig and Reynolds, 1988), this was performed using the program Bio Diversity Pro. A Pearson correlation matrix based on quantitative macroinvertebrate data was performed, this matrix was employed to produce a non-linear ordination using the Multidimensional Scaling method. This is an alternative to the factorial method and the objective is to display significant distances among investigated objects (Ludwig and Reynolds, 1988). The associations between the MDS ordination scores and environmental variables were tested by mean Spearman rank correlations.

3. Results

3.1. Environmental features

Stream shade was lower at Pas and Ur than at forested sites, as a consequence lower water temperatures were recorded in canopied sites (Table 1) thus NF sites had significantly lower water temperature than Pas and Ur sites (ANOVA, Tukey test $p < 0.01$). Both wet (range 0.9–21 m) and channel width (range 1.8–31.5 m) varied considerably between sites, having Pas and Ur the most expanded channels, however differences were only significant when comparing pasture dry width with all types of forested channels (ANOVA, Tukey test $p < 0.01$). At most sites the substrate particle size was mainly boulder, cobble and pebble. However, a higher proportion of sand was observed in some particular sites (LN, PAT, PIP) having sedimentation symptoms (Table 1). Current velocity was comprised between 0.1 and 3 m seg^{-1} but did not vary in a systematic manner among different uses.

There were also evident differences in physico-chemistry, with dissolved oxygen contents higher ($F_{(5,12)} = 4.0$, $p < 0.02$) and conductivity lower (Kruskal–Wallis, $p < 0.02$) at NF sites than the rest. Most chemical variables including SRP, TP, NO_3 and NH_4 showed higher mean values at Ur use than at the rest, including R-ur sites. However, values were highly variable, possibly explained by extreme values at ED urban site. A comparison of the same variables, excluding Ur sites from the analyses, highlighted that SRP was slightly lower at Pn sites, TP increased at Pas sites, NO_3 – NO_2 , TN and NH_4 values were higher at HF, but none of these results was statistically significant ($p > 0.05$). Nonetheless, the trend was that lowest values of NH_4 , TN, and TP were recorded at reference native forest sites (Table 1).

The land coverage with exotic (pine) plantation at selected sites ranged between 3.1 and 7.3% and land coverage with pastures primarily dedicated to sustain livestock varied between <1% and 25%. Urbanization affected between 0.4% and 9% of the basins however, rural/agricultural areas affected near of 18.7% of some basins as was the case of ED.

3.2. Particulate organic matter and chlorophyll a

Differences in energy availability with land uses were also evident particularly in the POM fractions: wood, leaves, CPOM, BPOM, and FPOM (Fig. 2). HF streams contain significantly higher leaves than Ur sites (Mann–Whitney U test, $p < 0.04$) and NF showed more wood than Pas and Ur sites (Kruskal–Wallis, $p < 0.04$). As expected, BPOM biomass was higher at NF and HN than Pas (Mann–Whitney U test, $p < 0.05$). Although FPOM did not varied in a consistent manner among land uses (ANOVA, $p = 0.30$), HF sites had higher FPOM biomass than Pas sites (Mann–Whitney, $p < 0.04$, $n = 6$, Fig. 2).

Regarding to autotrophic subsidies differences observed in chlorophyll a (range 2.3 and 17.9 mg m^{-2}) and macrophytes biomass (0 and 15.56 g m^{-2}) were not significant among land uses ($p = 0.84$, 0.77, respectively). However, bryophytes biomass was significantly higher at native forest sites than at Pn, Ur, Pas and R-ur sites (Kruskal–Wallis, $p < 0.01$, Fig. 2).

3.3. Functional metrics and land use

A total of 112 taxa were identified in the survey, being 22 shredders, 27 collector gatherers, 13 collector filterers, 23 scrapers/grazers and 27 predators. Taxa richness ranged between 20 and 48 taxa, and as expected the lowest value (20 taxa) corresponded to ED an urban site (Table 2).

Significant differences in shredder richness were observed among land uses with more shredders species found in HF and NF

Table 1
Environmental features of 18 sampled sites of streams in Patagonia (Argentina) during March 2006. Land use codes—Pas: pasture, R-ur: reference for urban site, Ur: urban site, Pn: pine plantation, HF: harvest forest, NF: native forest reference site.

Site	Site code	Land use	Elevation (m.a.s.l.)	% Native forest	% Pasture wetlands	% Exotic forest	% Urban areas	% Timberline and lakes	Water temperature (°C)	Wet and dry width (m)	Depth (cm)	W. velocity (m seg^{-1})	Sand %	Conductivity ($\mu\text{S cm}^{-1}$)	DO (mg l^{-1})	TSS (mg l^{-1})	SRP ($\mu\text{g l}^{-1}$)	TP ($\mu\text{g l}^{-1}$)	NO_3 ($\mu\text{g l}^{-1}$)	TN ($\mu\text{g l}^{-1}$)	NH_4 ($\mu\text{g l}^{-1}$)	Chlorophyll a (mg m^{-2})
Nant y Fall	NyF	Pas	690	56.3	25.9	0.2	0	14.6	13.6	21–31.5	19	1.5	15	113	7.7	1.7	0.4	8	0.0	410	0.7	2.3
E. Los ñires	LN	Pas	667	81.0	7.9	0	0	11.0	9.4	4.3–22.5	20.6	0.9	40	98	9.8	1.1	0.5	18	0.1	<10	0.5	5.4
Manguera	MG	Pas	699	76	<1	0	0	23	8.8	5.9–14.2	13.2	1.5	5	87	10.2	1.3	0.7	13	0.0	<10	0.4	10.4
Esquel	EU	R-ur	721	11.3	0	0.7	0	88.0	13.1	2.8–10.8	13.1	3	10	105	7.8	0.1	0.4	4	0.0	467	0.5	4.5
Las Minas	LMU	R-ur	573	93.1	1.3	0	0	5.6	10.2	3.25–5	17.9	0.9	10	164	9.2	0.8	0.4	2	nd	228	0.8	3.5
Carbón	CU	R-ur	384	97.0	1.4	1.4	0.2	0	9.3	7.57–10.5	18.9	1.7	10	93	9.8	1.7	0.7	14	0.1	319	0.5	9.3
Esquel	ED	Ur	491	11.0	18.7	5.6	9.0	55.7	11.9	9.8–10.5	23.1	1.9	20	366	8.3	4.1	21.0	816	69.0	5464	271.5	3.7
Las Minas	LMD	Ur	555	92.3	1.3	0	0.7	5.7	12.4	4.3–9	10.5	3	5	72	8.3	1.1	0.3	6	0.1	55	0.6	4.2
Carbón	CD	Ur	403	96.6	1.5	1.1	0.4	0.4	10.1	15.8–23.5	27.6	1.3	10	84	9.3	0.7	0.4	13	0.3	45	0.6	5.6
Golondrinas	GOL	Pn	361	57.1	1.3	4.7	0	38.2	8.7	1.6–4.2	20.5	2.6	30	126	10.1	1.5	0.2	12	0.1	<10	0.4	5.7
Patriada	PAT	Ph	495	79.1	0	7.3	0	13.6	8.4	1.2–1.8	7.8	0.3	35	76	10.3	0.5	0.6	8	0.5	354	0.6	4.1
Ifona	IFO	Ph	368	69.5	0	3.1	0	27.4	10.5	8.8–10.5	33.2	1.3	5	38	9.1	4.1	0.2	7	0.0	<10	1.6	15.5
Cab. De vaca	CVA	HF	699	90.4	2.8	0	0	6.8	9.4	4.6–6.9	21.6	1.1	15	63	9.7	0.7	0.7	13	0.2	10	0.6	3.7
Pipo	PIP	HF	728	99.0	0	0	0	1.0	10.6	0.9–2.3	7.7	0.1	35	84	8.9	0.6	0.9	13	0.2	280	4.0	15.5
Glyn	GLY	HF	615	98.8	0	0	0	1.2	8	3.5–5.7	15.6	1.5	15	69	10.6	0.7	0.2	2	0.4	524	0.5	3.7
Comisario	COM	NF	770	60.8	0	0	0	37.2	7.4	8.7–9	24.1	0.8	5	29	11.0	0.7	0.2	4	0.0	<10	0.5	5.0
Loro	LO	NF	820	43.7	0	0	0	56.3	6.1	3.4–4	22.1	1.1	5	26	11.9	4	0.2	3	nd	153	0.5	4.7
Chiquito	CHI	NF	670	97.1	0	0	0	2.9	6.8	2.4–3.5	10.4	3	10	9	11.3	5.2	0.9	26	0.2	<10	0.5	17.9

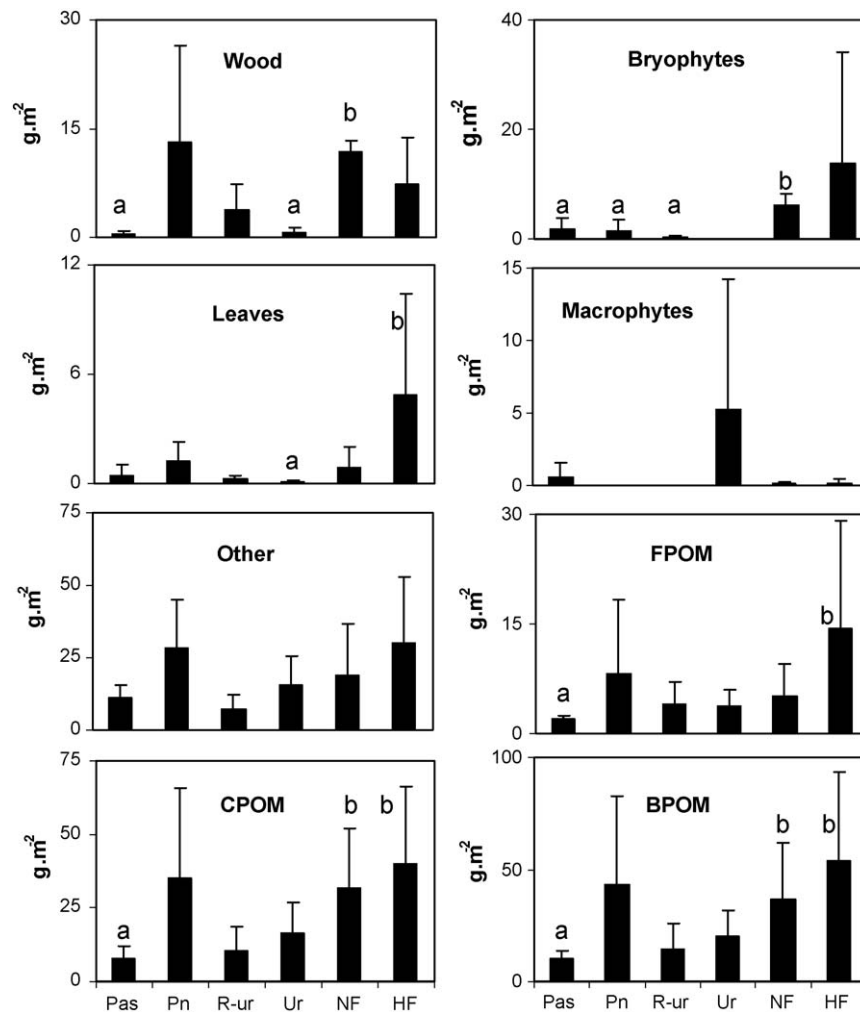


Fig. 2. Mean dry biomass of different energy supplies across different land uses at 18 sampling sites, Chubut, Argentina. Items: wood, entire leaves, other (mainly fragments), macrophytes, bryophytes, fine particulate organic matter (FPOM), total coarse benthic organic matter (CPOM) and total benthic organic matter (BPOM) ($\pm$ SD, g m⁻²). Superscript letters show significant differences among sites ($p < 0.05$) as indicated by Kruskal–Wallis test.

Table 2

TR: total richness, TD: total density (ind. 0.09 m⁻²), D: density (S: shredders, CG: collector gatherers, CF: collector filterers SG: scrapers/grazers and P: predators), H: Shannon Weaver diversity index, BPD: Berger–Parker dominance (%) ($n = 6$). Data from 18 sampled streams in Patagonia Argentina (March 2006).

	Site (land use)																		
	NyF (Pas)	LÑ (Pas)	MG (Pas)	EU (R-ur)	LMU (R-ur)	CU (R-ur)	ED (Ur)	LMD (Ur)	CD (Ur)	GOL (Pn)	PAT (Pn)	IFO (Pn)	CVA (HF)	PIP (HF)	GLY (HF)	COM (NF)	LO (NF)	CHI (NF)	
TR	32	37	28	31	31	36	20	31	42	38	27	31.5	39	32	48	33	34	45	
S	3	6	5	5	4	4	1	7	6	4	5	5	7	6	7	6	9	8	
CG	9	11	4	7	8	9	9	6	12	8	6	6	11	7	13	10	9	13	
CF	5	2	3	5	4	2	1	3	5	2	1	3	4	1	2	2	2	1	
SG	9	9	10	9	8	11	7	7	11	10	6	8	9	7	13	7	4	10	
P	6	9	6	5	7	10	2	8	8	14	9	9.5	8	11	13	8	10	13	
% S	48.3	12.8	42.1	15.2	23.2	12.2	0.1	20.7	17.2	9	35.1	13.5	11.1	17.5	62.6	26.4	42.8	21.1	
%CG	18.5	16.2	21	18.1	36.8	34.8	93.1	42.4	26.8	8.3	9.9	24.4	39.8	43.6	16.1	28.4	14	20.3	
%CF	16.5	42.1	5.9	7.7	8	5.8	0.1	16.6	14.3	36.8	14.9	5	25.4	0.2	3.5	2.8	2.7	35.7	
% SG	15.5	25.6	27.9	54.7	26.1	40.3	0.2	14.5	37.6	37.6	33.5	52.9	17.9	16.3	14.6	37.4	26.1	19.6	
% P	1.2	3.2	3.1	4.3	5.9	6.9	6.5	5.8	3.9	8.2	6.7	4.2	5.8	22.4	3.2	5	14.4	3.2	
TD	1119	548.5	629.6	273	229.8	494.5	3497	2126	1051	457.6	141.8	83.9	1192.2	1488.5	1866.8	206.8	108.9	614.3	
DS	540.8	70.2	265.5	41.5	53.3	60.3	1	440.8	181.1	41	49.6	11.3	132.2	260	1168.5	55.2	46.6	129.8	
D CG	207.6	89	132	49.3	84.5	172.1	3254.6	901	282.2	38	14	20.5	474	649	299	58.6	15.2	124.6	
D CF	184.6	231	37.2	21.2	18.3	28.5	5.3	353.2	150.8	168.6	21.2	4.2	302.3	3.7	66.3	5.8	3	219.5	
D SG	172.8	140.5	175.5	149.3	60	199.5	8.7	308.2	395.5	172.3	47.5	44.4	213.8	242.6	272.8	77.3	28.3	120.6	
D P	13	17.8	19.5	11.6	13.6	34	227.3	122.8	41.3	37.6	9.5	3.5	69.8	333.2	60.2	9.8	15.7	19.6	
H	2.33	2.04	2.12	2.51	2.58	2.81	1.89	2.07	2.93	2.34	2.11	2.28	2.75	2.26	1.86	2.7	2.89	2.42	
BPD	27.4	41.9	36.7	22.5	20.8	17.4	39.3	20.9	13.2	33.1	31.1	30.2	14.9	32.3	60.5	20.5	13.6	35.7	

Table 3Results of Kruskal–Wallis (H , $n = 6$) or Mann–Whitney U test (pair site comparison) for effects of land use on macroinvertebrates functional attributes.

	$H_{(5,18)}$	p	Post hoc test	Mann–Whitney U test ($p < 0.05$)
Total richness	4.58	0.46		
Richness				
Shredders	9.7	0.026	HF, NF > Pn, R-ur	
Collector gatherers	5.89	0.32		
Collector filterers	4.48	0.48		
Scraper/grazers	2.56	0.76		
Predators	8.02	0.15		
Relative abundance				
% Shredders	4.32	0.50		
% Collector gatherers	7.90	0.16		
% Collector filterers	2.77	0.73		
% Scraper/grazer	9.98	0.07		Pn > HF
% Predator	6.49	0.26		
Density				
Total density	13.8	0.016	Ur, HF > Pn, NF, R-ur	
Shredders	9.6	0.08		HF, Pas > Pn
Collector gatherers	14.5	0.012	HF > Pas, NF, R-ur > Pn	
Collector filterers	3.30	0.65		
Scraper/grazer	7.61	0.18		HF > Pn, NF, Pas
Predator	11.7	0.04	Ur > Pas, NF	

Significant differences are in bold. Significant post hoc comparisons (Duncan, Kruskal–Wallis). Land use codes—Pas: pasture; R-ur: reference for urban site; Ur: urban site; Pn: pine plantation; NH: native harvest; NF: native forest reference site.

than Pn and R-ur sites (Kruskal–Wallis, $p < 0.026$, Table 3, Fig. 3). Collector gatherers richness was highly variable (range: 4–13 taxa) and collector filterers (range: 1–5 taxa) were poorly represented mostly at Pas, Pin and NF sites.

As expected total density of macroinvertebrates was significantly higher at Ur and HF than at Pn, NF and R-ur sites (Kruskal–Wallis, $p < 0.016$). Collector's gatherers were significantly more abundant at HF sites than at the rest of uses, moreover, Pas, NF and R-ur had higher density of collector's gatherers than Pn (Kruskal–Wallis, $p < 0.012$). Abundance of shredders was significantly lower at Pn sites than at Pas and HF. Density of scrapers/grazer was higher at HF compared to Pn, Pas and NF (Mann–Whitney U test, $p < 0.05$) (Table 3, Fig. 3). More predators were recorded at Ur sites than at Pas and NF sites.

Diversity values ranges between 1.86 and 2.93 and lowest values were found at Pas and Pn sites. Dominance values were high at HF, Pn and Pas uses. However, some particular sites showed extreme values as GLY by recruitment of the stonefly *Antarctoperla michaelsoni* (Gripopterygidae) and ED by high density of *Hyaella araucana*, *Limnodrilus* spp. and Orthoclaadiinae species.

Regarding to species composition, similarity percentages showed that sites in Pas overlapped more with Ur, R-ur, and HF (Table 4), than with Pn and NF. Similarities between sites within land uses were relatively high, although Ur sites were more variable probably reflecting differences in degrees of impairment. NF selected sites showed a reasonable similitude among each other (55.8%), the same as HF sites (57%). However, the within similarity of land cover types ranged 36.9–64.3%. This might show a fairly

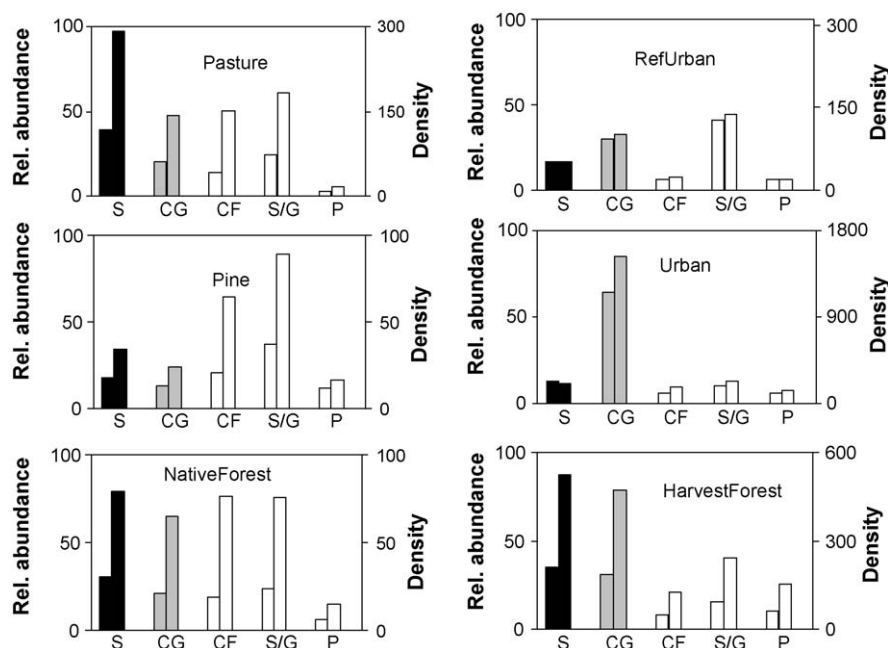


Fig. 3. Relative abundance (left bar) and total density (right bar, ind. m^{-2}) of functional feeding groups at 18 sampling sites subjected to different land uses, Chubut, Argentina. Functional feeding groups are—S: shredders, CG: collector gatherers, CF: collector filterers, S/G: scrapers/grazers and P: predators.

Table 4

Similarity percentages (Bray Curtis matrix) showing similarity in species composition within and between land uses types.

	Pasture	Ref urban	Urban	Pine	Native forest	Harvest forest
Pasture	53.4					
Ref urban	58.9	64.3				
Urban	50.3	49.6	36.9			
Pine	39.8	50.6	37.4	40.7		
Native forest	41.5	51.3	38.1	47.3	55.8	
Harvest forest	53.1	55.6	46.7	47.7	51.6	57

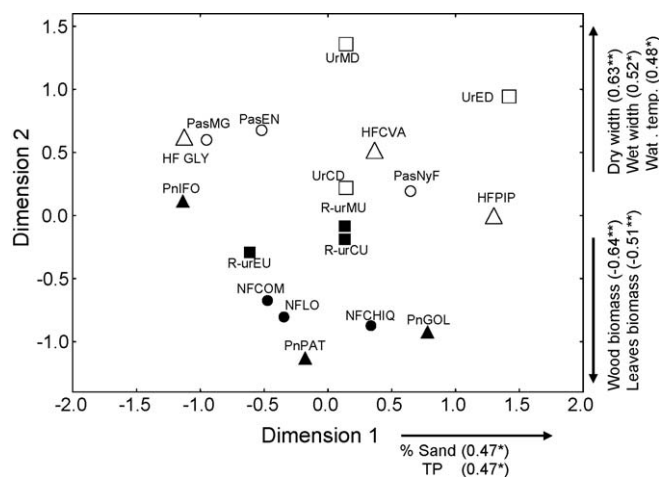


Fig. 4. Results of the multidimensional ordination of macroinvertebrate community data collected from 18 sampling sites subjected to different land uses (Chubut, Argentina). Symbols are: (□) urban sites; (△) harvest forest; (○) pasture; (■) reference urban sites; (●) reference native forest and (▲) pine plantations. Significant correlation coefficients (Spearman rank) with dimension 1 and 2 are given (* $p < 0.05$, ** $p < 0.005$).

high within-type variability, which was largest in Ur and Pn forest sites.

3.4. Environmental relationships

The separation of forested either with native or exotic species, plus reference sites (dark symbols) from pasture, urban and harvest sites (open symbols) was clearly evident in the site ordination using density data of macroinvertebrate species (dimension 2, Fig. 4). Thus, NF, R-ur and Pn sites were grouped to the negative end of dimension 2, whereas Pas, HF and Ur were positioned to the positive end of dimension 2. When identifying variables associated with dimension 2, wood and leaf biomass which were negatively correlated, resulted as significant variables explaining sites ordination (Fig. 4). Other significant variables were water temperature and dry and wet width, which were positively correlated with dimension 2. On the other hand, dimension 1 highlighted a water quality impairment gradient or sedimentation; variables that were positively correlated with dimension 1 were TP and sand percentage (Fig. 4).

4. Discussion

Our results suggest that distribution of detritus and trophic structure were different among land uses, this is in agreement with what has been reported for streams draining in developed catchments (Delong and Brusven, 1993; Riley et al., 2003; Thompson and Townsend, 2004; Danger and Robson, 2004). Changes in physical attributes of streams produced by some land use practices were also documented in this study. The buffering

effect of the riparian corridors on water temperature (Collier et al., 2000; Scrimgeour and Kendall, 2003; Leunda et al., 2009) was evident at native forested sites. Similarly the percentage of fine sediment on the streambed varied among uses and explained some differences in benthic assemblages. Agricultural land use and pasture development often results in an increased supply of fine sediments (Scarsbrook and Halliday, 1999; Brisbois et al., 2008). Niyogi et al. (2007) found that fine sediment covered 33% of the stream bottom at agricultural streams in New Zealand. We observed that LN showed a high percentage of sand as consequence of the constant supply from the severely eroded banks while post-harvest sedimentation effects were evident at Pipo (HF). At this site several trees from the riparian zone were cut and the channel was obstructed by debris (branches and leaves) after the harvest. Logging operations can produce increases of sand and silt in the streambed and this is more significant in small streams (Death et al., 2003). Some groups favored by the presence of fine sediments such as the collector gatherers *Lumbriculus variegatus* and *Pseudochironomus* sp. were abundant at these sites.

We expected higher values of chlorophyll *a* at pasture sites as a consequence of higher temperatures or nutrients levels. On the contrary, chlorophyll *a* pattern was highly variable as were nutrients, which is consistent with observations of Scrimgeour and Kendall (2003) and Scarsbrook and Halliday (1999). They did not report any significant difference in chlorophyll patterns and land uses, although values increased with distance to forested areas. In our study, even though nutrients did not show any significant differences among land use practices, lower values of NH_4 , TN, and TP were recorded at reference native forest sites. Likely we were unable to separate cattle effects from other effects in our sampling design, thus cattle is an all-pervading land use in the area. Then some of the patterns observed at harvest sites could result of the combined effect of wood collection and pasturing.

A noteworthy feature in our study was that bryophyte biomass was significantly higher at native forest than at other land uses. Similarly, Riley et al. (2003) found lower moss coverage of streambeds at developed pasture streams than at undeveloped or lightly grazed native grassland catchments. They concluded that the presence of moss can be influenced by substrate stability (bedrock, boulder vs. gravel, sand), trampling by stock and nutrient concentrations.

As reported in other studies (Delong and Brusven, 1993; Benfield, 1997), forested sites showed consistently more allochthonous material than pastures, thus BPOM was significantly higher at NF and HF than at Pas. However, BPOM varied in a non-systematic manner among Pn sites. Scarsbrook et al. (2001) found more detritus material in pine plantations than in pasture sites and Thompson and Townsend (2004) reported a decreasing gradient of detritus biomass from pine, native forest to pasture grasslands. Likely differences observed in the attributes of the riparian corridors (composition, coverage and age) at our Pn sites, were the cause of high variation in detritus biomass.

The contribution of shredders to benthic community at pasture sites was higher than expected (12.8–48%). Most selected pasture sites have remnant forest upstream and this probably had some positive effects on detritus supply. From all of our pasture sites, the most extreme situation was Los Nires with benthos community basically dominated by filterers (Hydropsychidae: *Smicridea annulicornis*). Predominance of filtering and gathering collectors in rural and agricultural landscapes has been well documented (Shieh et al., 2003; Ometto et al., 2004; Utz et al., 2009) and as suggested by Strand and Merritt (1999) a high proportion of filterers reflect the relative amount of fine particulate organic matter in transport in the water column. In addition, in our study collectors dominated or co-dominated at all urban sites (Compín and Céréghino, 2007) but also at most harvest forest sites. Thus, the

relative proportion of collectors was higher at HF sites than forest reference sites suggesting a compositional shift from CPOM feeders to FPOM feeders. McCord et al. (2007) observed that increases of collector's relative abundance was a significant response indicating detrimental effects of silviculture operations when comparing harvest from reference sites. As we mentioned the accumulation of logging slash due to the incorrect treatment of debris was evident at least in one site (Pipo).

Whiles and Wallace (1997) stated that conversion of forest to pine monoculture can influence benthic taxonomic composition. In our study, when assessing the possible effects of exotic plantation on functional feeding groups, the most marked finding was that shredder richness was consistently higher at HF and NF than Pn sites. Besides, total density of shredders was significantly lower at Pn than at Pas and HF. Valdovinos (2001) found more density and biomass of shredders in leaf packs of *N. pumilio* (native) than *Pinus radiata* in experimental trials in Chilean streams. In other experiments, Albariño and Balseiro (2002) noticed that large shredders (*Klapopteryx kuscheli* and *Parasericostoma cristatum*) fed only on *N. pumilio* and not on *Pinus ponderosa* leaf packs in Patagonian streams. The most dramatic situation in our study was the site PAT, in which the riparian corridor is completely replaced by *Pseudotsuga menziesii* and the stream runs through 1200 m of the exotic forest. Paradoxically, the shredder *Myotricha murina* was well represented at this site and constitutes almost 35% of the benthic community. However, other common shredders such as *Senzilloides panguipulli*, *Antarctoperla michaelsoni*, and *Udamocercia arumifera* that were well represented at HF and NF sites were practically absent at Pn sites.

As was showed in the ordination analysis, pasture sites were clearly separated from forested sites (either native or exotic) in agreement with reports from similar studies (Hall et al., 2001; Death et al., 2003; Thompson and Townsend, 2004; Compin and Céréghino, 2007). Moreover, in our study harvest forest sites were also clearly separated from undisturbed native forest suggesting changes in composition and structure of benthic community. Detritus biomass, sedimentation processes and habitat impoverishment, factors associated with the removal of the forest canopy and harvesting procedure, appear as important variables determining those assemblages. As we tested here there is also a potential usefulness of functional metrics in particular shredders richness and collectors density to indicate the impact of land use change on macroinvertebrates.

This study reinforces the idea that riparian corridor management may be critical to the distribution of benthic taxa as well functional organization in headwater Patagonian streams. For this reason maintenance of good conditions of vegetation adjacent to rivers will enhance water quality and the environment for stream communities.

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Research Article

Stream and bed temperature variability in a coastal headwater catchment: influences of surface-subsurface interactions and partial-retention forest harvesting

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Abstract

Stream temperature was recorded between 2002 and 2005 at four sites in a coastal headwater catchment in British Columbia, Canada. Shallow groundwater temperatures, along with bed temperature profiles at depths of 1 to 30 cm, were recorded at 10-min intervals in two hydrologically distinct reaches beginning in 2003 or 2004, depending on the site. The lower reach had smaller discharge contributions via lateral inflow from the hillslopes and fewer areas with upwelling (UW) and/or neutral flow across the stream bed compared to the middle reach. Bed temperatures were greater than those of shallow groundwater during summer, with higher temperatures in areas of downwelling (DW) flow compared to areas of neutral and UW flow. A paired-catchment analysis revealed that partial-retention forest harvesting in autumn 2004 resulted in higher daily maximum stream and bed temperatures but smaller changes in daily minima. Changes in daily maximum stream temperature, averaged over July and August of the post-harvest year, ranged from 1.6 to 3 °C at different locations within the cut block. Post-harvest changes in bed temperature in the lower reach were smaller than the changes in stream temperature, greater at sites with DW flow, and decreased with depth at both UW and DW sites, dropping to about 1 °C at a depth of 30 cm. In the middle reach, changes in daily maximum bed temperature, averaged over July and August, were generally about 1 °C and did not vary significantly with depth. The pre-harvest regression models for shallow groundwater were

not suitable for applying the paired-catchment analysis to estimate the effects of harvesting. However, shallow groundwater was warmer at the lower reach following harvesting, despite generally cooler weather compared to the pre-harvest year. Copyright © 2012 John Wiley & Sons, Ltd.

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Land use strategies to mitigate climate change in carbon dense temperate forests

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Strategies to mitigate carbon dioxide emissions through forestry activities have been proposed, but ecosystem process-based integration of climate change, enhanced CO₂, disturbance from fire, and management actions at regional scales are extremely limited. Here, we examine the relative merits of afforestation, reforestation, management changes, and harvest residue bioenergy use in the Pacific Northwest. This region represents some of the highest carbon density forests in the world, which can store carbon in trees for 800 y or more. Oregon's net ecosystem carbon balance (NECB) was equivalent to 72% of total emissions in 2011–2015. By 2100, simulations show increased net carbon uptake with little change in wildfires. Reforestation, afforestation, lengthened harvest cycles on private lands, and restricting harvest on public lands increase NECB 56% by 2100, with the latter two actions contributing the most. Resultant cobenefits included water availability and biodiversity, primarily from increased forest area, age, and species diversity. Converting 127,000 ha of irrigated grass crops to native forests could decrease irrigation demand by 233 billion m³·y⁻¹. Utilizing harvest residues for bioenergy production instead of leaving them in forests to decompose increased emissions in the short-term (50 y), reducing mitigation effectiveness. Increasing forest carbon on public lands reduced emissions compared with storage in wood products because the residence time is more than twice that of wood products. Hence, temperate forests with high carbon densities and lower vulnerability to mortality have substantial potential for reducing forest sector emissions. Our analysis framework provides a template for assessments in other temperate regions.

forests | carbon balance | greenhouse gas emissions | climate mitigation

Strategies to mitigate carbon dioxide emissions through forestry activities have been proposed, but regional assessments to determine feasibility, timeliness, and effectiveness are limited and rarely account for the interactive effects of future climate, atmospheric CO₂ enrichment, nitrogen deposition, disturbance from wildfires, and management actions on forest processes. We examine the net effect of all of these factors and a suite of mitigation strategies at fine resolution (4-km grid). Proven strategies immediately available to mitigate carbon emissions from forest activities include the following: (i) reforestation (growing forests where they recently existed), (ii) afforestation (growing forests where they did not recently exist), (iii) increasing carbon density of existing forests, and (iv) reducing emissions from deforestation and degradation (1). Other proposed strategies include wood bioenergy production (2–4), bioenergy combined with carbon capture and storage (BECCS), and increasing wood product use in buildings. However, examples of commercial-scale BECCS are still scarce, and sustainability of wood sources remains controversial because of forgone ecosystem carbon storage and low environmental cobenefits (5, 6). Carbon stored in buildings generally outlives its usefulness or is replaced within decades (7) rather than the centuries possible in forests, and the factors influencing product substitution have yet to be fully explored (8). Our analysis of mitigation strategies focuses on the first four strategies, as well as bioenergy production, utilizing harvest residues only and without carbon capture and storage.

The appropriateness and effectiveness of mitigation strategies within regions vary depending on the current forest sink, competition with land-use and watershed protection, and environmental conditions affecting forest sustainability and resilience. Few process-based regional studies have quantified strategies that could actually be implemented, are low-risk, and do not depend on developing technologies. Our previous studies focused on regional modeling of the effects of forest thinning on net ecosystem carbon balance (NECB) and net emissions, as well as improving modeled drought sensitivity (9, 10), while this study focuses mainly on strategies to enhance forest carbon.

Our study region is Oregon in the Pacific Northwest, where coastal and montane forests have high biomass and carbon sequestration potential. They represent coastal forests from northern California to southeast Alaska, where trees live 800 y or more and biomass can exceed that of tropical forests (11) (Fig. S1). The semiarid ecoregions consist of woodlands that experience frequent fires (12). Land-use history is a major determinant of forest carbon balance. Harvest was the dominant cause of tree mortality (2003–2012) and accounted for fivefold as much mortality as that from fire and beetles combined (13). Forest land ownership is predominantly public (64%), and 76% of the biomass harvested is on private lands.

Significance

Regional quantification of feasibility and effectiveness of forest strategies to mitigate climate change should integrate observations and mechanistic ecosystem process models with future climate, CO₂, disturbances from fire, and management. Here, we demonstrate this approach in a high biomass region, and found that reforestation, afforestation, lengthened harvest cycles on private lands, and restricting harvest on public lands increased net ecosystem carbon balance by 56% by 2100, with the latter two actions contributing the most. Forest sector emissions tracked with our life cycle assessment model decreased by 17%, partially meeting emissions reduction goals. Harvest residue bioenergy use did not reduce short-term emissions. Cobenefits include increased water availability and biodiversity of forest species. Our improved analysis framework can be used in other temperate regions.

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The authors declare no conflict of interest.

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Data deposition: The CLM4.5 model data are available at Oregon State University (terraweb.forestry.oregonstate.edu/FMEC). Data from the >200 intensive plots on forest carbon are available at Oak Ridge National Laboratory (https://daac.ornl.gov/NACP/guides/NACP_TERRA-PNW.html), and FIA data are available at the USDA Forest Service (<https://www.fia.fs.fed.us/tools-data/>).

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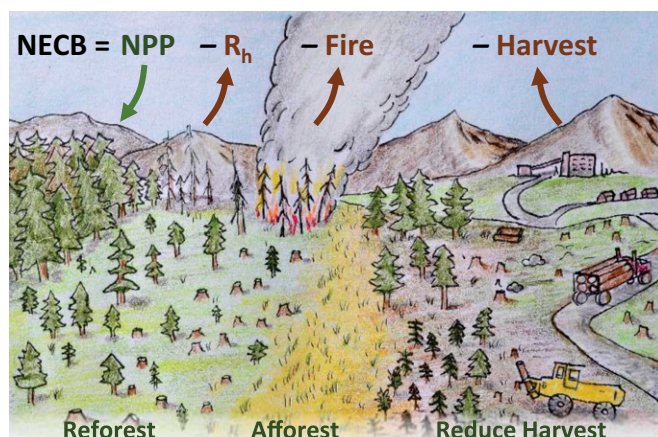


Fig. 1. Approach to assessing effects of mitigation strategies on forest carbon and forest sector emissions. NECB is productivity (NPP) minus R_h and losses from fire and harvest (red arrows). Harvest emissions include those associated with wood products and bioenergy.

Many US states, including Oregon (14), plan to reduce their greenhouse gas (GHG) emissions in accordance with the Paris Agreement. We evaluated strategies to address this question: How much carbon can the region's forests realistically remove from the atmosphere in the future, and which forest carbon strategies can reduce regional emissions by 2025, 2050, and 2100? We propose an integrated approach that combines observations with models and a life cycle assessment (LCA) to evaluate current and future effects of mitigation actions on forest carbon and forest sector emissions in temperate regions (Fig. 1). We estimated the recent carbon budget of Oregon's forests, and simulated the potential to increase the forest sink and decrease forest sector emissions under current and future climate conditions. We provide recommendations for regional assessments of mitigation strategies.

Results

Carbon stocks and fluxes are summarized for the observation cycles of 2001–2005, 2006–2010, and 2011–2015 (Table 1 and Tables S1 and S2). In 2011–2015, state-level forest carbon stocks totaled 3,036 Tg C (3 billion metric tons), with the coastal and montane ecoregions accounting for 57% of the live tree carbon (Tables S1 and S2). Net ecosystem production [NEP; net primary production (NPP) minus heterotrophic respiration (R_h)] averaged 28 teragrams carbon per year (Tg C y^{-1}) over all three periods. Fire emissions were unusually high at 8.69 million metric tons carbon dioxide equivalent ($tCO_2e\ y^{-1}$, i.e., $2.37\ Tg\ C\ y^{-1}$) in 2001–2005 due to the historic Biscuit Fire, but decreased to 3.56 million $tCO_2e\ y^{-1}$ ($0.97\ Tg\ C\ y^{-1}$) in 2011–2015 (Table S4). Note that 1 million tCO_2e equals 3.667 Tg C.

Our LCA showed that in 2001–2005, Oregon's net wood product emissions were 32.61 million tCO_2e (Table S3), and 3.7-fold wildfire emissions in the period that included the record fire year (15) (Fig. 2). In 2011–2015, net wood product emissions were 34.45 million tCO_2e and almost 10-fold fire emissions, mostly due to lower fire emissions. The net wood product emissions are higher than fire emissions despite carbon benefits of storage in wood products and substitution for more fossil fuel-intensive products. Hence, combining fire and net wood product emissions, the forest sector emissions averaged 40 million $tCO_2e\ y^{-1}$ and accounted for about 39% of total emissions across all sectors (Fig. 2 and Table S4). NECB was calculated from NEP minus losses from fire emissions and harvest (Fig. 1). State NECB was equivalent to 60% and 70% of total emissions for 2001–2005 and 2011–2015, respectively (Fig. 2, Table 1, and Table S4). Fire emissions were only between 4% and 8% of total emissions from

all sources (2011–2015 and 2001–2004, respectively). Oregon's forests play a larger role in meeting its GHG targets than US forests have in meeting the nation's targets (16, 17).

Historical disturbance regimes were simulated using stand age and disturbance history from remote sensing products. Comparisons of Community Land Model (CLM4.5) output with Forest Inventory and Analysis (FIA) aboveground tree biomass (>6,000 plots) were within 1 SD of the ecoregion means (Fig. S2). CLM4.5 estimates of cumulative burn area and emissions from 1990 to 2014 were 14% and 25% less than observed, respectively. The discrepancy was mostly due to the model missing an anomalously large fire in 2002 (Fig. S3A). When excluded, modeled versus observed fire emissions were in good agreement ($r^2 = 0.62$; Fig. S3B). A sensitivity test of a 14% underestimate of burn area did not affect our final results because predicted emissions would increase almost equally for business as usual (BAU) management and our scenarios, resulting in no proportional change in NECB. However, the ratio of harvest to fire emissions would be lower.

Projections show that under future climate, atmospheric carbon dioxide, and BAU management, an increase in net carbon uptake due to CO_2 fertilization and climate in the mesic ecoregions far outweighs losses from fire and drought in the semiarid ecoregions. There was not an increasing trend in fire. Carbon stocks increased by 2% and 7% and NEP increased by 12% and 40% by 2050 and 2100, respectively.

We evaluated emission reduction strategies in the forest sector: protecting existing forest carbon, lengthening harvest cycles, reforestation, afforestation, and bioenergy production with product substitution. The largest potential increase in forest carbon is in the mesic Coast Range and West Cascade ecoregions. These forests are buffered by the ocean, have high soil water-holding capacity, low risk of wildfire [fire intervals average 260–400 y (18)], long carbon residence time, and potential for high carbon density. They can attain biomass up to 520 Mg C ha^{-1} (12). Although Oregon has several protected areas, they account for only 9–15% of the total forest area, so we expect it may be feasible to add carbon-protected lands with cobenefits of water protection and biodiversity.

Reforestation of recently forested areas include those areas impacted by fire and beetles. Our simulations to 2100 assume regrowth of the same species and incorporate future fire responses to climate and cyclical beetle outbreaks [70–80 y (13)]. Reforestation has the potential to increase stocks by 315 Tg C by 2100, reducing forest sector net emissions by 5% by 2100 relative to BAU management (Fig. 3). The East and West Cascades ecoregions had the highest reforestation potential, accounting for 90% of the increase (Table S5).

Afforestation of old fields within forest boundaries and non-food/nonforage grass crops, hereafter referred to as “grass crops,” had to meet minimum conditions for tree growth, and crop grid cells had to be partially forested (SI Methods and Table S6). These crops are not grazed or used for animal feed. Competing land uses may decrease the actual amount of area that can be afforested. We calculated the amount of irrigated grass crops (127,000 ha) that could be converted to forest, assuming success of carbon offset programs (19). By 2100, afforestation increased stocks by

Table 1. Forest carbon budget components used to compute NECB

Flux, Tg C·y ⁻¹	2001–2005		2006–2010		2011–2015		2001–2015
NPP	73.64	7.59	73.57	7.58	73.57	7.58	73.60
Rh	45.67	5.11	45.38	5.07	45.19	5.05	45.41
NEP	27.97	9.15	28.19	9.12	28.39	9.11	28.18
Harvest removals	8.58	0.60	7.77	0.54	8.61	0.6	8.32
Fire emissions	2.37	0.27	1.79	0.2	0.97	0.11	1.71
NECB	17.02	9.17	18.63	9.14	18.81	9.13	18.15

Average annual values for each period, including uncertainty (95% confidence interval) in Tg C y^{-1} (multiply by 3.667 to get million tCO_2e).

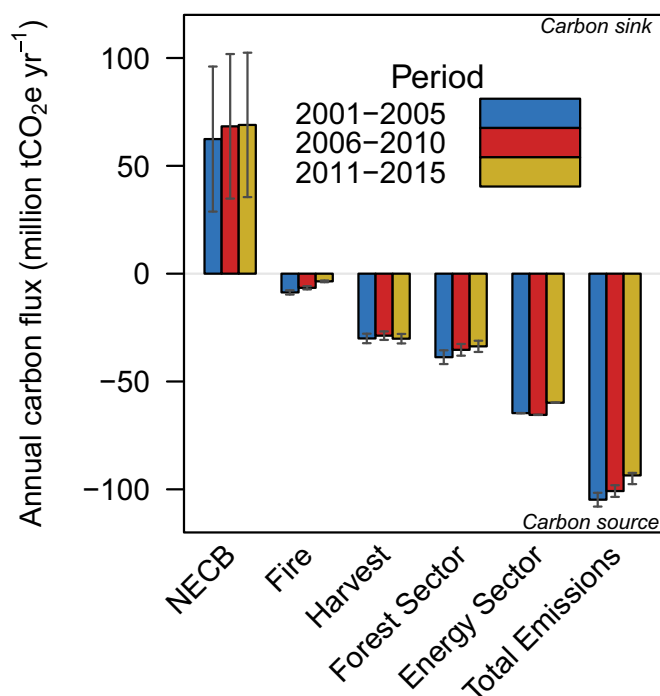


Fig. 2. Oregon's forest carbon sink and emissions from forest and energy sectors. Harvest emissions are computed by LCA. Fire and harvest emissions sum to forest sector emissions. Energy sector emissions are from the Oregon Global Warming Commission (14), minus forest-related emissions. Error bars are 95% confidence intervals (Monte Carlo analysis).

94 Tg C and cumulative NECB by 14 Tg C, and afforestation reduced forest sector GHG emissions by 1.3–1.4% in 2025, 2050, and 2100 (Fig. 3).

We quantified cobenefits of afforestation of irrigated grass crops on water availability based on data from hydrology and agricultural simulations of future grass crop area and related irrigation demand (20). Afforestation of 127,000 ha of grass cropland with Douglas fir could decrease irrigation demand by 222 and 233 billion m³·y⁻¹ by 2050 and 2100, respectively. An independent estimate from measured precipitation and evapotranspiration (ET) at our mature Douglas fir and grass crop flux sites in the Willamette Valley shows the ET/precipitation fraction averaged 33% and 52%, respectively, and water balance (precipitation minus ET) averaged 910 mm·y⁻¹ and 516 mm·y⁻¹. Under current climate conditions, the observations suggest an increase in annual water availability of 260 billion m³·y⁻¹ if 127,000 ha of the irrigated grass crops were converted to forest.

Harvest cycles in the mesic and montane forests have declined from over 120 y to 45 y despite the fact that these trees can live 500–1,000 y and net primary productivity peaks at 80–125 y (21). If harvest cycles were lengthened to 80 y on private lands and harvested area was reduced 50% on public lands, state-level stocks would increase by 17% to a total of ~3,600 Tg C and NECB would increase 2–3 Tg C y⁻¹ by 2100. The lengthened harvest cycles reduced harvest by 2 Tg C y⁻¹, which contributed to higher NECB. Leakage (more harvest elsewhere) is difficult to quantify and could counter these carbon gains. However, because harvest on federal lands was reduced significantly since 1992 (NW Forest Plan), leakage has probably already occurred.

The four strategies together increased NECB by 64%, 82%, and 56% by 2025, 2050, and 2100, respectively. This reduced forest sector net emissions by 11%, 10%, and 17% over the same periods (Fig. 3). By 2050, potential increases in NECB were largest in the Coast Range (Table S5), East Cascades, and Klamath

Mountains, accounting for 19%, 25%, and 42% of the total increase, whereas by 2100, they were most evident in the West Cascades, East Cascades, and Klamath Mountains.

We examined the potential for using existing harvest residue for electricity generation, where burning the harvest residue for energy emits carbon immediately (3) versus the BAU practice of leaving residues in forests to slowly decompose. Assuming half of forest residues from harvest practices could be used to replace natural gas or coal in distributed facilities across the state, they would provide an average supply of 0.75–1 Tg C y⁻¹ to the year 2100 in the reduced harvest and BAU scenarios, respectively. Compared with BAU harvest practices, where residues are left to decompose, proposed bioenergy production would increase cumulative net emissions by up to 45 Tg C by 2100. Even at 50% use, residue collection and transport are not likely to be economically viable, given the distances (>200 km) to Oregon's facilities.

Discussion

Earth system models have the potential to bring terrestrial observations related to climate, vulnerability, impacts, adaptation,

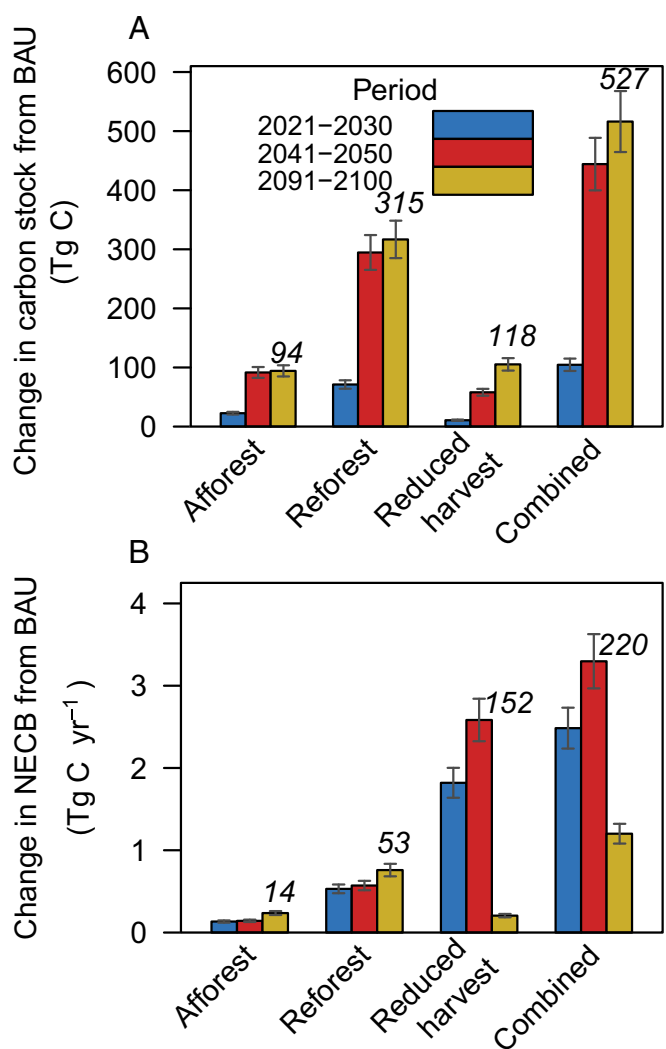


Fig. 3. Future change in carbon stocks and NECB with mitigation strategies relative to BAU management. The decadal average change in forest carbon stocks (A) and NECB relative to BAU (B) are shown. Italicized numbers over bars indicate mean forest carbon stocks in 2091–2100 (A) and cumulative change in NECB for 2015–2100 (B). Error bars are ±10%.

and mitigation into a common framework, melding biophysical with social components (22). We developed a framework to examine a suite of mitigation actions to increase forest carbon sequestration and reduce forest sector emissions under current and future environmental conditions.

Harvest-related emissions had a large impact on recent forest NECB, reducing it by an average of 34% from 2001 to 2015. By comparison, fire emissions were relatively small and reduced NECB by 12% in the Biscuit Fire year, but only reduced NECB 5–9% from 2006 to 2015. Thus, altered forest management has the potential to enhance the forest carbon balance and reduce emissions.

Future NEP increased because enhancement from atmospheric carbon dioxide outweighed the losses from fire. Lengthened harvest cycles on private lands to 80 y and restricting harvest to 50% of current rates on public lands increased NECB the most by 2100, accounting for 90% of total emissions reduction (Fig. 3 and Tables S5 and S6). Reduced harvest led to NECB increasing earlier than the other strategies (by 2050), suggesting this could be a priority for implementation.

Our afforestation estimates may be too conservative by limiting them to nonforest areas within current forest boundaries and 127,000 ha of irrigated grass cropland. There was a net loss of 367,000 ha of forest area in Oregon and Washington combined from 2001 to 2006 (23), and less than 1% of native habitat remains in the Willamette Valley due to urbanization and agriculture (24). Perhaps more of this area could be afforested.

The spatial variation in the potential for each mitigation option to improve carbon stocks and fluxes shows that the reforestation potential is highest in the Cascade Mountains, where fire and insects occur (Fig. 4). The potential to reduce harvest on public land is highest in the Cascade Mountains, and that to lengthen harvest cycles on private lands is highest in the Coast Range.

Although western Oregon is mesic with little expected change in precipitation, the afforestation cobenefits of increased water availability will be important. Urban demand for water is projected to increase, but agricultural irrigation will continue to consume much more water than urban use (25). Converting 127,000 ha of irrigated grass crops to native forests appears to be a win-win strategy, returning some of the area to forest land, providing habitat and connectivity for forest species, and easing irrigation demand. Because the afforested grass crop represents only 11% of the available grass cropland (1.18 million ha), it is not likely to result in leakage or indirect land use change. The two forest strategies combined are likely to be important contributors to water security.

Cobenefits with biodiversity were not assessed in our study. However, a recent study showed that in the mesic forests, cobenefits with biodiversity of forest species are largest on lands with harvest cycles longer than 80 y, and thus would be most pronounced on private lands (26). We selected 80 y for the harvest cycle mitigation strategy because productivity peaks at 80–125 y in this region, which coincides with the point at which cobenefits with wildlife habitat are substantial.

Habitat loss and climate change are the two greatest threats to biodiversity. Afforestation of areas that are currently grass crops would likely improve the habitat of forest species (27), as about 90% of the forests in these areas were replaced by agriculture. About 45 mammal species are at risk because of range contraction (28). Forests are more efficient at dissipating heat than grass and crop lands, and forest cover gains lead to net surface cooling in all regions south of about 45° latitude in North American and Europe (29). The cooler conditions can buffer climate-sensitive bird populations from approaching their thermal limits and provide more food and nest sites (30). Thus, the mitigation strategies of afforestation, protecting forests on public lands and lengthening harvest cycles to 80–125 y, would likely benefit forest-dependent species.

Oregon has a legislated mandate to reduce emissions, and is considering an offsets program that limits use of offsets to 8% of

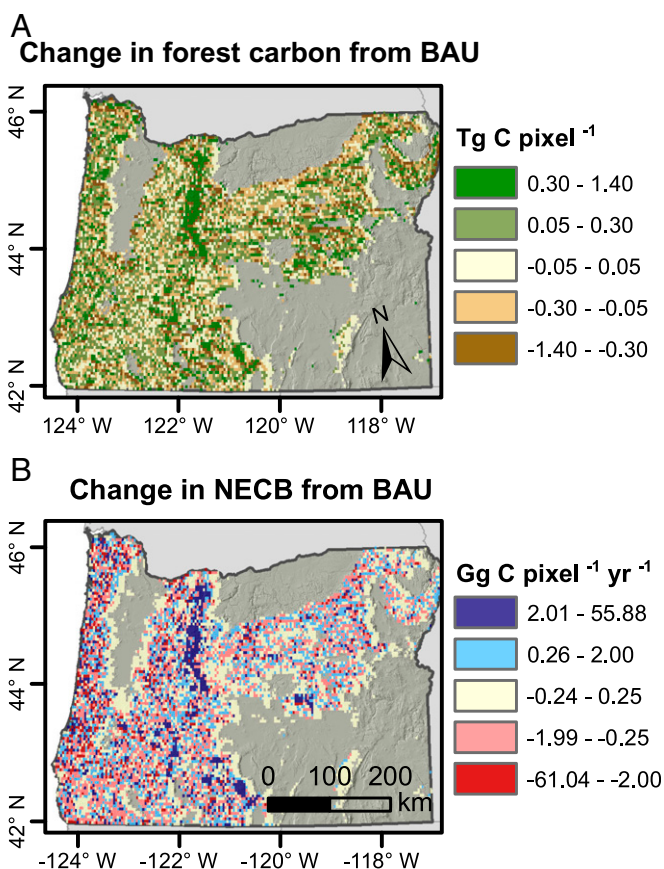


Fig. 4. Spatial patterns of forest carbon stocks and NECB by 2091–2100. The decadal average changes in forest carbon stocks (A) and NECB (B) due to afforestation, reforestation, protected areas, and lengthened harvest cycles relative to continued BAU forest management (red is increase in NECB) are shown.

the total emissions reduction to ensure that regulated entities substantially reduce their own emissions, similar to California's program (19). An offset becomes a net emissions reduction by increasing the forest carbon sink (NECB). If only 8% of the GHG reduction is allowed for forest offsets, the limits for forest offsets would be 2.1 and 8.4 million metric tCO₂e of total emissions by 2025 and 2050, respectively (Table S6). The combination of afforestation, reforestation, and reduced harvest would provide 13 million metric tCO₂e emissions reductions, and any one of the strategies or a portion of each could be applied. Thus, additionality beyond what would happen without the program is possible.

State-level reporting of GHG emissions includes the agriculture sector, but does not appear to include forest sector emissions, except for industrial fuel (i.e., utility fuel in Table S3) and, potentially, fire emissions. Harvest-related emissions should be quantified, as they are much larger than fire emissions in the western United States. Full accounting of forest sector emissions is necessary to meet climate mitigation goals.

Increased long-term storage in buildings and via product substitution has been suggested as a potential climate mitigation option. Pacific temperate forests can store carbon for many hundreds of years, which is much longer than is expected for buildings that are generally assumed to outlive their usefulness or be replaced within several decades (7). By 2035, about 75% of buildings in the United States will be replaced or renovated, based on new construction, demolition, and renovation trends (31, 32). Recent analysis suggests substitution benefits of using wood versus more fossil fuel-intensive materials have been overestimated by at

least an order of magnitude (33). Our LCA accounts for losses in product substitution stores (PSSs) associated with building life span, and thus are considerably lower than when no losses are assumed (4, 34). While product substitution reduces the overall forest sector emissions, it cannot offset the losses incurred by frequent harvest and losses associated with product transportation, manufacturing, use, disposal, and decay. Methods for calculating substitution benefits should be improved in other regional assessments.

Wood bioenergy production is interpreted as being carbon-neutral by assuming that trees regrow to replace those that burned. However, this does not account for reduced forest carbon stocks that took decades to centuries to sequester, degraded productive capacity, emissions from transportation and the production process, and biogenic/direct emissions at the facility (35). Increased harvest through proposed thinning practices in the region has been shown to elevate emissions for decades to centuries regardless of product end use (36). It is therefore unlikely that increased wood bioenergy production in this region would decrease overall forest sector emissions.

Conclusions

GHG reduction must happen quickly to avoid surpassing a 2 °C increase in temperature since preindustrial times. Alterations in forest management can contribute to increasing the land sink and decreasing emissions by keeping carbon in high biomass forests, extending harvest cycles, reforestation, and afforestation. Forests are carbon-ready and do not require new technologies or infrastructure for immediate mitigation of climate change. Growing forests for bioenergy production competes with forest carbon sequestration and does not reduce emissions in the next decades (10). BECCS requires new technology, and few locations have sufficient geological storage for CO₂ at power facilities with high-productivity forests nearby. Accurate accounting of forest carbon in trees and soils, NECB, and historic harvest rates, combined with transparent quantification of emissions from the wood product process, can ensure realistic reductions in forest sector emissions.

As states and regions take a larger role in implementing climate mitigation steps, robust forest sector assessments are urgently needed. Our integrated approach of combining observations, an LCA, and high-resolution process modeling (4-km grid vs. typical 200-km grid) of a suite of potential mitigation actions and their effects on forest carbon sequestration and emissions under changing climate and CO₂ provides an analysis framework that can be applied in other temperate regions.

Materials and Methods

Current Stocks and Fluxes. We quantified recent forest carbon stocks and fluxes using a combination of observations from FIA; Landsat products on forest type, land cover, and fire risk; 200 intensive plots in Oregon (37); and a wood decomposition database. Tree biomass was calculated from species-specific allometric equations and ecoregion-specific wood density. We estimated ecosystem carbon stocks, NEP (photosynthesis minus respiration), and NECB (NEP minus losses due to fire or harvest) using a mass-balance approach (36, 38) (Table 1 and *SI Materials and Methods*). Fire emissions were computed from the Monitoring Trends in Burn Severity database, biomass data, and region-specific combustion factors (15, 39) (*SI Materials and Methods*).

Future Projections and Model Description. Carbon stocks and NEP were quantified to the years 2025, 2050, and 2100 using CLM4.5 with physiological parameters for 10 major forest species, initial forest biomass (36), and future climate and atmospheric carbon dioxide as input (Institut Pierre Simon Laplace climate system model downscaled to 4 km × 4 km, representative concentration pathway 8.5). CLM4.5 uses 3-h climate data, ecophysiological characteristics, site physical characteristics, and site history to estimate the daily fluxes of carbon, nitrogen, and water between the atmosphere, plant state variables, and litter and soil state variables. Model components are biogeophysics, hydrological cycle, and biogeochemistry. This model version does not include a dynamic vegetation model to simulate resilience and

establishment following disturbance. However, the effect of regeneration lags on forest carbon is not particularly strong for the long disturbance intervals in this study (40). Our plant functional type (PFT) parameterization for 10 major forest species rather than one significantly improves carbon modeling in the region (41).

Forest Management and Land Use Change Scenarios. Harvest cycles, reforestation, and afforestation were simulated to the year 2100. Carbon stocks and NEP were predicted for the current harvest cycle of 45 y compared with simulations extending it to 80 y. Reforestation potential was simulated over areas that recently suffered mortality from harvest, fire, and 12 species of beetles (13). We assumed the same vegetation regrow to the maximum potential, which is expected with the combination of natural regeneration and planting that commonly occurs after these events. Future BAU harvest files were constructed using current harvest rates, where county-specific average harvest and the actual amounts per ownership were used to guide grid cell selection. This resulted in the majority of harvest occurring on private land (70%) and in the mesic ecoregions. Beetle outbreaks were implemented using a modified mortality rate of the lodgepole pine PFT with 0.1% y⁻¹ biomass mortality by 2100.

For afforestation potential, we identified areas that are within forest boundaries that are not currently forest and areas that are currently grass crops. We assumed no competition with conversion of irrigated grass crops to urban growth, given Oregon's land use laws for developing within urban growth boundaries. A separate study suggested that, on average, about 17% of all irrigated agricultural crops in the Willamette Valley could be converted to urban area under future climate; however, because 20% of total cropland is grass seed, it suggests little competition with urban growth (25).

Landsat observations (12,500 scenes) were processed to map changes in land cover from 1984 to 2012. Land cover types were separated with an unsupervised K-means clustering approach. Land cover classes were assigned to an existing forest type map (42). The CropScape Cropland Data Layer (CDL 2015, <https://nassgeodata.gmu.edu/CropScape/>) was used to distinguish nonforage grass crops from other grasses. For afforestation, we selected grass cropland with a minimum soil water-holding capacity of 150 mm and minimum precipitation of 500 mm that can support trees (43).

Afforestation Cobenefits. Modeled irrigation demand of grass seed crops under future climate conditions was previously conducted with hydrology and agricultural models, where ET is a function of climate, crop type, crop growth state, and soil-holding capacity (20) (Table S7). The simulations produced total land area, ET, and irrigation demand for each cover type. Current grass seed crop irrigation in the Willamette Valley is 413 billion m³·y⁻¹ for 238,679 ha and is projected to be 412 and 405 billion m³ in 2050 and 2100 (20) (Table S7). We used annual output from the simulations to estimate irrigation demand per unit area of grass seed crops (1.73, 1.75, and 1.84 million m³·ha⁻¹ in 2015, 2050, and 2100, respectively), and applied it to the mapped irrigated crop area that met conditions necessary to support forests (Table S7).

LCA. Decomposition of wood through the product cycle was computed using an LCA (8, 10). Carbon emissions to the atmosphere from harvest were calculated annually over the time frame of the analysis (2001–2015). The net carbon emissions equal NECB plus total harvest minus wood lost during manufacturing and wood decomposed over time from product use. Wood industry fossil fuel emissions were computed for harvest, transportation, and manufacturing processes. Carbon credit was calculated for wood product storage, substitution, and internal mill recycling of wood losses for bioenergy.

Products were divided into sawtimber, pulpwood, and wood and paper products using published coefficients (44). Long-term and short-term products were assumed to decay at 2% and 10% per year, respectively (45). For product substitution, we focused on manufacturing for long-term structures (building life span >30 y). Because it is not clear when product substitution started in the Pacific Northwest, we evaluated it starting in 1970 since use of concrete and steel for housing was uncommon before 1965. The displacement value for product substitution was assumed to be 2.1 Mg fossil C/Mg C wood use in long-term structures (46), and although it likely fluctuates over time, we assumed it was constant. We accounted for losses in product substitution associated with building replacement (33) using a loss rate of 2% per year (33), but ignored leakage related to fossil C use by other sectors, which may result in more substitution benefit than will actually occur.

The general assumption for modern buildings, including cross-laminate timber, is they will outlive their usefulness and be replaced in about 30 y (7). By 2035, ~75% of buildings in the United States will be replaced or renovated, based on new construction, demolition, and renovation trends, resulting in threefold as many buildings as there are now [2005 baseline (31, 32)]. The loss of

the PSS is therefore PSS multiplied by the proportion of buildings lost per year (2% per year).

To compare the NECB equivalence to emissions, we calculated forest sector and energy sector emissions separately. Energy sector emissions ["in-boundary" state-quantified emissions by the Oregon Global Warming Commission (14)] include those from transportation, residential and commercial buildings, industry, and agriculture. The forest sector emissions are cradle-to-grave annual carbon emissions from harvest and product emissions, transportation, and utility fuels (Table S3). Forest sector utility fuels were subtracted from energy sector emissions to avoid double counting.

Uncertainty Estimates. For the observation-based analysis, Monte Carlo simulations were used to conduct an uncertainty analysis with the mean and SDs for NPP and Rh calculated using several approaches (36) (*SI Materials and Methods*). Uncertainty in NECB was calculated as the combined uncertainty of NEP, fire emissions (10%), harvest emissions (7%), and land cover estimates

(10%) using the propagation of error approach. Uncertainty in CLM4.5 model simulations and LCA were quantified by combining the uncertainty in the observations used to evaluate the model, the uncertainty in input datasets (e.g., remote sensing), and the uncertainty in the LCA coefficients (41).

Model input data for physiological parameters and model evaluation data on stocks and fluxes are available online (37).

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Atmospheric greenhouse gases (GHGs) must be reduced to avoid an unsustainable climate. Because carbon dioxide is removed from the atmosphere and sequestered in forests and wood products, mitigation strategies to sustain and increase forest carbon sequestration are being developed. These strategies require full accounting of forest sector GHG budgets. Here, we describe a rigorous approach using over one million observations from forest inventory data and a regionally calibrated life-cycle assessment for calculating cradle-to-grave forest sector emissions and sequestration. We find that Western US forests are net sinks because there is a positive net balance of forest carbon uptake exceeding losses due to harvesting, wood product use, and combustion by wildfire. However, over 100 years of wood product usage is reducing the potential annual sink by an average of 21%, suggesting forest carbon storage can become more effective in climate mitigation through reduction in harvest, longer rotations, or more efficient wood product usage. Of the ~10 700 million metric tonnes of carbon dioxide equivalents removed from west coast forests since 1900, 81% of it has been returned to the atmosphere or deposited in landfills. Moreover, state and federal reporting have erroneously excluded some product-related emissions, resulting in 25%–55% underestimation of state total CO₂ emissions. For states seeking to reach GHG reduction mandates by 2030, it is important that state CO₂ budgets are effectively determined or claimed reductions will be insufficient to mitigate climate change.

Introduction

Heat trapping greenhouse gases (GHGs) are being added to the atmosphere at an accelerating rate by fossil fuel combustion and land use change. Climate change consequences were recently described by the Intergovernmental Panel on Climate Change (IPCC) and the United States National Climate Assessment (USGCRP 2018). The IPCC Special Report (IPCC 2018), Global Warming of 1.5 °C, concludes that to keep global average temperature below 1.5 °C by 2100, it is essential to reduce fossil fuel emissions by

45% by 2030, while substantially increasing the removal of atmospheric CO₂. Both reports emphasize the need to increase atmospheric CO₂ removal strategies by forests in addition to sustaining current forest carbon uptake (Houghton and Nassikas 2018). Some states in the US have set targets for reducing GHGs that include forest climate mitigation options (Anderson *et al* 2017, Law *et al* 2018), yet consistent, rigorous accounting methods are required for evaluating options. Challenges include determining the extent that forests, harvest operations, and wood products affect GHG budgets and emissions accountability.

The most recent global carbon budget estimate indicates that land-based sinks remove 29% of anthropogenic emissions (including land use change) with a significant contribution from forests (Le Quéré *et al* 2018). However, none of the agreements or policies (IPCC 2006, NRCS 2010, Brown *et al* 2014, Doe 2017, EPA 2017, Duncan 2017) provides clear and consistent procedures for quantitatively assessing the extent forests and forest products are increasing or reducing carbon dioxide concentrations in the atmosphere. Assessments are challenging because they involve components that require multiple types of expertise and accounting methods (i.e. forest ecosystem processes, wood products, and inherently uncertain substitution credits). Methods are often in disagreement over the wood product Life Cycle Assessment (LCA) assumption of *a priori* carbon neutrality, where biogenic emissions from the combustion and decomposition of wood is ignored because the carbon released from wood is assumed to be replaced by subsequent tree growth in the following decades (EPA 2016). Despite a multitude of analyses that recognize that the assumption is fundamentally flawed (Harmon *et al* 1996, Gunn *et al* 2011, Haberl *et al* 2012, Schulze *et al* 2012, Buchholz *et al* 2016, Booth 2018), it continues to be used in mitigation analyses, particularly for wood bioenergy.

Forests are sustainable *net* sinks as long as forest carbon uptake from the atmosphere exceeds emissions from harvesting, wood product use and decomposition, and wildfire. Wood products ultimately release CO₂ to the atmosphere as they are manufactured, disposed of, and decompose or are burned. However, because of concerns about double-counting, significant emissions associated with harvest and wood product use have not been counted for any sector (EPA 2018). These emissions are often not included in state CO₂ budget estimates (Brown *et al* 2014, Oregon Global Warming Commission 2017), even when they are included in national budgets (EPA 2017) (table S1 is available online at stacks.iop.org/ERL/14/095005/mmedia). If US states intend to use forests for mitigation strategies, they must account for all contributing sources and sinks of forests and forest-derived products (Stockmann *et al* 2012, IPCC 2014).

By focusing on a region with sufficient information to conduct a meaningful LCA, we demonstrate how a quantitative assessment of forests, management practices and wood products can assess the actual role played by forests and forestry practices in managing atmospheric CO₂. We calculate the regional forest carbon balance (from 2001 to 2016) using observations from over 24 000 forest inventory plots in Washington, Oregon, and California (states with GHG reduction mandates). Net forest sector carbon balance is quantified using an improved LCA including harvest, transportation, manufacturing, wood product pool storage and decay, emissions associated with fire, and

substitution for both building construction and energy production. We specifically consider global warming potential associated with carbon dioxide and do not include additional GHGs such as nitrous oxide and methane. Our aim is to provide an accurate cradle-to-grave, transparent and transferable accounting method of all forest-derived carbon for other states and countries with GHG reduction mandates (figure 1; box 1; figure S1; tables S2–S6).

Results

Western US forest ecosystem CO₂ balance (2001–2016)

Forest carbon uptake and release (net ecosystem production (NEP); figure 1(a)) controlled by ecosystem biological processes is calculated as the balance between forest carbon uptake (net primary production (NPP)) and forest carbon release through the decomposition of dead organic matter (heterotrophic respiration; R_h). In this study, a negative number indicates a net carbon sink (removal from the atmosphere) and a positive number indicates a net carbon source (addition to the atmosphere). The coastal Western US states together are a strong forest carbon sink with NEP of -292 ± 36 million metric tonnes (MMT) CO₂e per year ($-857 \text{ g CO}_2\text{e m}^{-2} \text{ yr}^{-1}$) (table 1; table S1), and account for approximately 60% of total Western US forest NEP (coastal, southwestern, and intermountain regions).

In addition to NEP, disturbances from harvest and wildfire influence estimates of net ecosystem carbon balance (NECB = NEP minus losses Chapin *et al* 2006; figure 1(a)). In the Western US states, the significant carbon losses from the forest are primarily from removals of wood through harvest, decomposition or burning of aboveground and belowground harvest residues, and wildfire (Law and Waring 2015). Significant harvest has been occurring in the western US since the early 20th century (figure S2). Up to 40% of the harvested wood does not become a product and the products themselves decay over time, resulting in product accumulation much smaller than the total amount harvested (figure 2(a); solid line) (Harmon *et al* 1996, Dymond 2012, Williams *et al* 2016, EPA 2017). Emissions include combustion of wood that does not become a product, combustion for energy, decomposition and/or combustion at end-of-life (table 1; rows 5, 6, 9, and 10). When these carbon losses are accounted for, these forests remain significant carbon sinks at -187 ± 33 MMT CO₂e per year ($-551 \text{ g CO}_2\text{e m}^{-2} \text{ yr}^{-1}$), with the largest sink in California (40%) followed by Oregon (33%) and Washington (27%). Despite California having twice the fire emissions of the other states (~ 10 versus ~ 5 MMT CO₂e yr⁻¹ per state) the ranking is due to much lower harvest removals in California (~ 12 MMT CO₂e yr⁻¹) compared to almost double in

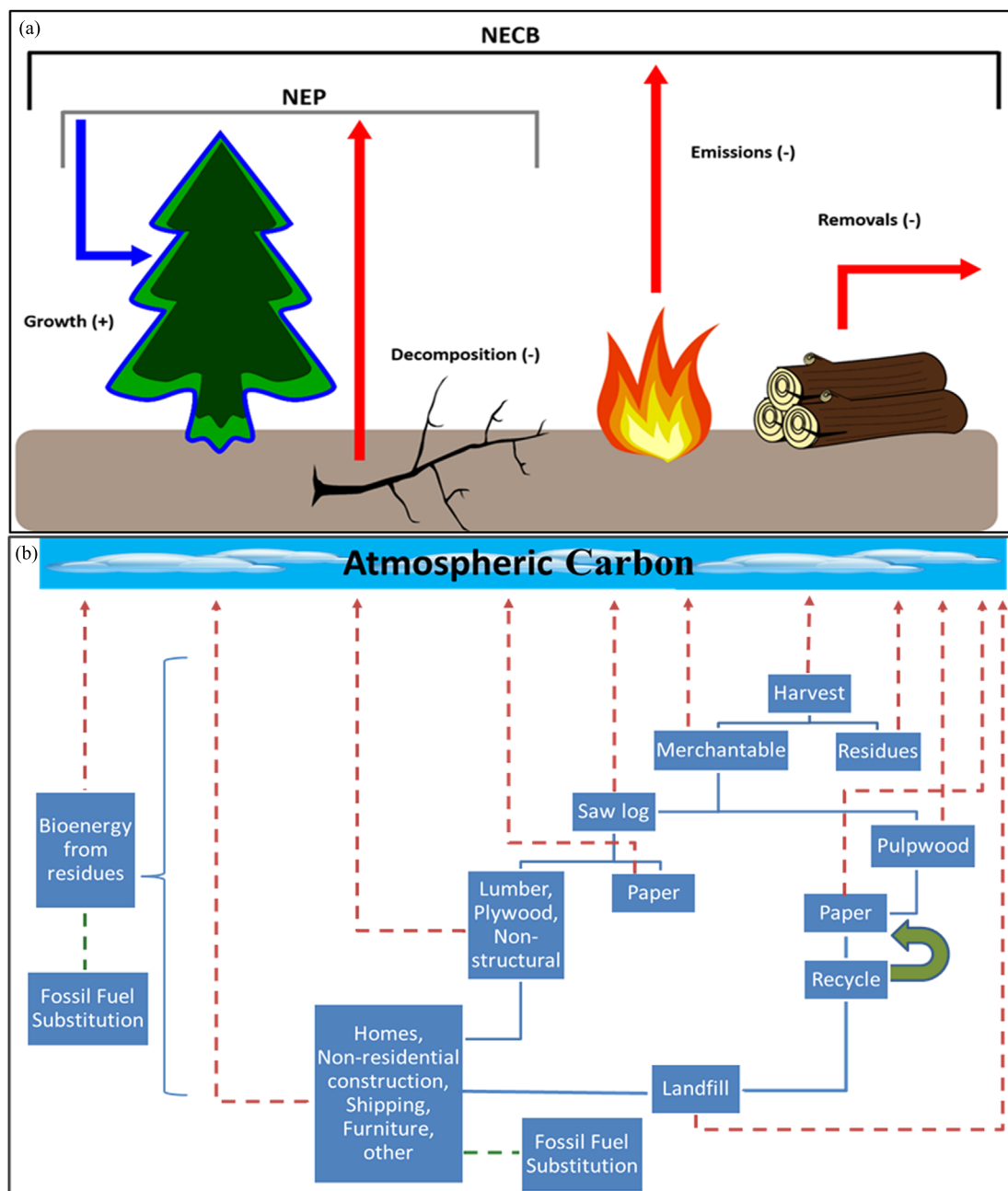


Figure 1. Conceptual diagram of Forest-GHG (a) describes the natural, land-based forest carbon sink where the net of growth and decomposition is net ecosystem production (NEP), and after accounting for removals from fire and harvest, the balance is net ecosystem carbon balance (NECB), (b) describes the cascade of wood products until eventual deposition in landfills or the atmosphere and shows the pathway of emissions.

Washington ($\sim 20 \text{ MMT CO}_2\text{e yr}^{-1}$) and triple in Oregon ($\sim 31 \text{ MMT CO}_2\text{e yr}^{-1}$). Fire emissions are a third of harvest removals region-wide.

Building on our earlier work (Harmon *et al* 1996, Hudiburg *et al* 2011, Law *et al* 2018), we developed a modified cradle-to-grave model (Forest-GHG) for combining the balance of carbon captured in forest ecosystems, wood product use, lifetime emissions, and eventual return to the atmosphere or long-term storage in landfills. Forest-GHG tracks emissions associated with harvest of wood and manufacturing, transport and use of wood products. Harvest removals result in immediate (combustion of residues on-site or

as mill residues with and without energy recapture), fast (short-lived products such as paper), decadal (long-lived products such as wood) and centuries-long (older buildings and land-filled) timeframes before emissions are released back to the atmosphere (figures 1(b) and S1). Our model includes seven product pools and temporally dynamic recycling and landfill rates. Most importantly, we now include a more mechanistic representation of longer-term structural wood in buildings, by moving beyond a simple half-life with exponential decay (figure 3 and SI methods and SI tables 2–6). Our new building cohort-component method tracks decay of short- and

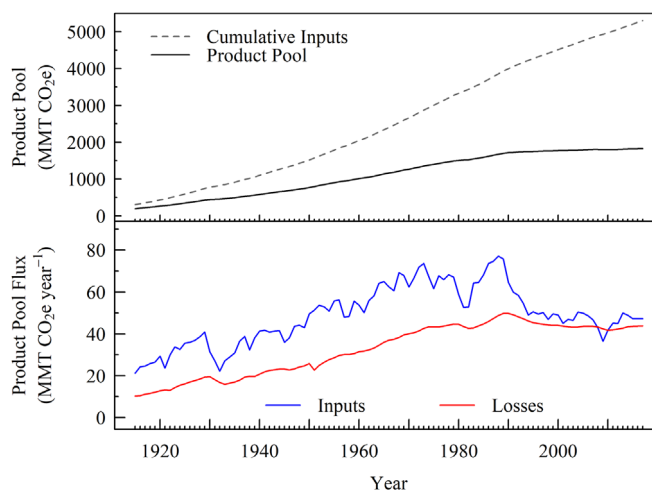


Figure 2. Wood product inputs and outputs from 1900 to 2016 for Washington, Oregon, and California. (A) Cumulative production in MMT CO₂e per year assuming no losses over time (dotted grey line) versus the realized in-use wood product pool over time after accounting for decay (losses). (B) Yearly product inputs over time (blue line) that represents the fraction of harvest (removed wood) that becomes a product versus the decay emissions from the pool over time (red line).

Table 1. Average annual total fluxes by state and region from 2001 to 2016. All units are in million MT CO₂e. Negative numbers indicate a carbon sink (CO₂ is being removed from the atmosphere). The more negative the number, the stronger the sink. Grey shading is used to indicate net values that represent carbon sink strength both before and after removals are accounted for.

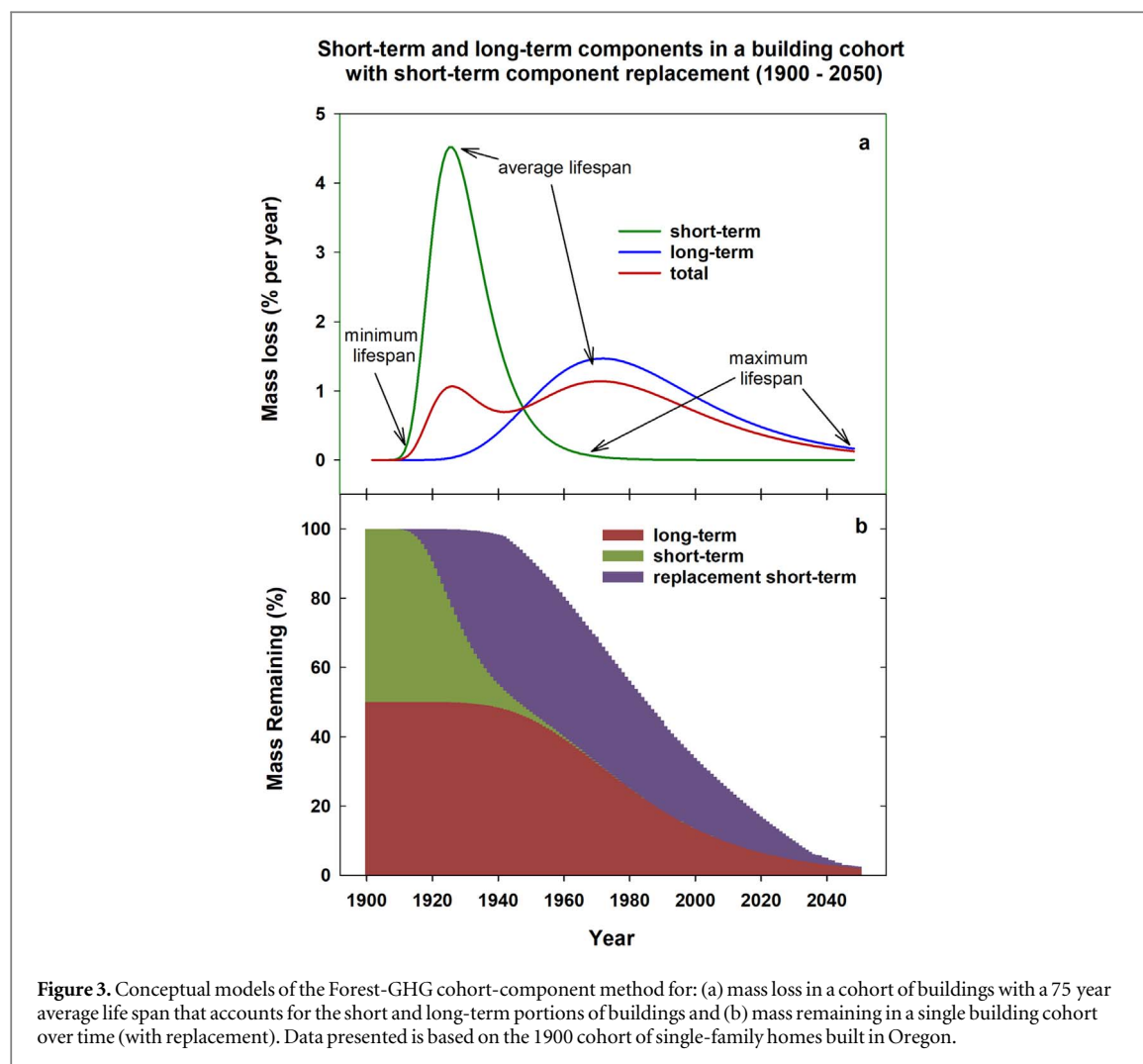
Ecosystem	Washington	Oregon	California	Total
1. Forested area (million hectares)	9.7	12.4	11.9	34.0
2. Net ecosystem production (NEP)	−89.9	−102.0	−99.8	−291.6
3. Fire emissions	5.1	5.3	10.3	20.7
4. Harvest removals	18.5	30.5	11.5	60.5
Net ecosystem carbon balance (NECB) (sum of rows 1 through 4)	−66.4	−66.2	−78.0	−210.5
Forest industry	Washington	Oregon	California	Total
5. Harvest residue combustion (onsite)	3.9	6.5	2.5	12.9
6. Harvest, transportation, manufacturing (FFE emissions)	2.8	4.6	1.6	9.0
7. Wood product pool annual inputs	−18.5	−30.5	−11.5	−60.5
8. Landfill annual inputs (from products)	−6.8	−11.9	−4.2	−22.9
9. Wood manufacturing losses	3.9	6.5	3.9	14.3
10. Wood product and landfill decomposition	21.4	36.2	13.3	71.0
Net forest sector carbon balance (NECB + sum of rows 5 through 10)	−59.5	−54.7	−72.4	−186.6
11. Wood product substitution (wood)	−3.0	−4.9	−1.6	−9.4
12. Wood product substitution (energy)	−1.8	−3.0	−1.8	−6.6
Net forest sector carbon balance (with credits; NECB + sum of rows 5 through 12)	−64.3	−62.6	−75.8	−202.7

long-lived building components annually, and the lag time associated with these losses (figure S3). Our wood bioenergy substitution credits (Sathre and O'Connor 2010) include wood waste from harvest, mill residues, and wood products displacement of more fossil fuel intensive materials.

Using our component tracking LCA, we found that of the ~10 700 MMT CO₂e of wood harvested in all three states since 1900 (figure 2), only 2028 MMT CO₂e are currently stored in wood products with half stored in Oregon (1043 MMT CO₂e). In just over 100 years, Oregon has removed the equivalent of all live trees in the state's Coast Range forests (Law *et al* 2018), and returned 65% to the atmosphere and transferred 16% to landfills. Even though these are some of the most productive and carbon dense forests in the world

(Hudiburg *et al* 2009), the carbon accumulated in much of the removed biomass took up to 800 years to accumulate—and cannot be recovered if current management practices continue.

Forest harvest-related emissions have averaged 107 MMT CO₂e annually from 2001 to 2016 (table 1; row 5, 6, 9, and 10). Emissions are highest from decay of the wood product pool that has been accumulating for over 100 years (table 1 row 10; figures 3 and S3). This is after accounting for recycling and semi-permanent storage in landfills. Structural wood product decay for long- and short-term components (wood in buildings; figure 3) account for about 30%–35% of wood product and landfill decomposition while paper and non-building wood products account for about 65%–70%. Under this complete accounting, the



lowest contribution to overall emissions is from fossil fuel usage during harvest, transportation, and manufacturing, i.e. less than 10% of total wood product-related emissions in the three states.

We found that wood-related substitution for construction materials (0.54 fossil fuel carbon emissions avoided per unit carbon of wood; table 1 row 11) and energy (0.68 fossil fuel carbon emissions avoided; table 1 row 12) may offset 18% of forest industry emissions. This assumes 50% of wood-derived construction products are substituted for a non-wood product and that 75% of mill residues are substituted for fossil fuel energy (Berg *et al* 2016).

We varied the maximum average life spans of the wood products used in construction (e.g. buildings) to examine its effect on emissions estimates. Emissions are minimally reduced by 2%–4% in each state when a longer average maximum lifespan is used (100 years) for the long-term building components and minimally increased by 2%–3% when a shorter average maximum lifespan is used (50 years, which is the mean lifetime of buildings in the US EPA 2013).

Combined, the US west coast state forest sector (cradle-to-grave) is a net carbon sink, removing ~187 MMT CO₂e annually from the atmosphere and

potentially reducing fossil fuel emissions by up to another 20 MMT CO₂e through product and energy substitution. Harvest-related emissions reduce the natural sink (NEP—Fire) by 34, 46, and 27% for Washington, Oregon, and California, respectively. When substitution credits are included, this changes to reductions of 27%, 37%, and 23%. Harvest rates have been highest in Oregon (table 1), contributing to increasing wood product emissions and the largest reductions to forest sink capacity.

Discussion

NECB is a good estimate of ecosystem carbon uptake, e.g. for carbon offsets programs (Anderson *et al* 2017), and can be compared spatially with changing environmental conditions or disturbances, but is an incomplete calculation of the entire forest sector emissions. It does not include emissions from wood products caused by machinery, transport, manufacturing and losses—emissions that can equal up to 85% of the total versus 15% from fire, insects, and land use change (Williams *et al* 2016). Nor does it account for the storage and subsequent release of carbon in varying

end uses with varied product lifetimes. Given that not all harvested wood is an immediate source to the atmosphere and very little harvested wood is stored in perpetuity, it is essential to track associated emissions over time. For state- or region-level carbon budgets, a cradle-to-grave carbon LCA should be combined with the ecosystem carbon balance (NEP and NECB) to account for how much the forestry sector is contributing to or offsetting total carbon emissions.

If wood buildings are replaced by wood buildings, substitution is not occurring, and because wood is preferred for construction of single-family housing in North America, some of our substitution values are overestimated (Sathre and O'Connor 2010). Wood products store carbon temporarily, and a larger wood product pool increases decomposition emissions over time (figure 3). This emphasizes that increasing the wood product carbon sink will require shifts in product allocation from short-term to long-term pools such as reclaimed (re-used) wood products from demolition of buildings, and reduction of product manufacturing losses (EPA 2016). Clearly, there is potential for climate mitigation by using forests to sequester carbon in biomass and reduce losses associated with the wood product chain (Law *et al* 2018).

It is argued that there may be reductions in fossil carbon emissions when wood is substituted for more fossil fuel intensive building materials (e.g. steel or concrete) or used as an alternative energy source (Butarbutar *et al* 2016). Substitution is a one-time credit in the year of the input. Studies have reported a range of substitution displacement factors (from negative to positive displacement; Sathre and O'Connor 2010, Smyth *et al* 2017), but we found no study that has tracked the actual amount of construction product substitution that is occurring or has occurred in the past in the United States. This makes substitution one of the most uncertain parts of this carbon budget. It may be more easily tracked in the fossil fuel sector through a decrease in emissions because of reduction in product supply, in which case it would be double counting to then include it as a credit for the forest sector. We show results with and without the substitution credit (a decrease in forest sector emissions) because it cannot be verified. We show the potential impact it has on the overall forest sector carbon sink, even though the displacement factor may be unrealistically high (Smyth *et al* 2017, Dugan *et al* 2018). For forest sector emissions assessments, the uncertainty suggests exclusion of the credit.

Currently, state's GHG accounting budgets are incorrect because they are not full cradle-to-grave estimates of all CO₂ emissions associated with forest natural processes and human influences. For accurate GHG accounting, these emissions should be included in the forestry sector as they are not accounted for by state's energy and transportation sectors (IPCC 2006) (table S1). The US EPA reported average fossil fuel CO₂ emissions of 491 MMT CO₂e yr⁻¹ for the three

states combined (2013–2016). Forest industry harvest, transportation, and manufacturing fossil fuel emissions are included in this total. However, it is unclear to what extent wood product decay and combustion emissions are also counted in state budgets. In Oregon, they are not included at all, resulting in state CO₂ emissions that have been underestimated by up to 55% (Oregon Global Warming Commission 2017, Law *et al* 2018). Washington includes combustion emissions from the current year's harvest (table 1; Manufacturing losses; row 9), but not from wood product decay, resulting in up to a 25% underestimation of state CO₂ emissions. Because California's emissions from other sectors are so high (76% of regional total), and harvest rates have been historically lower than in Oregon and Washington, the impact of not including these emissions is very small as a proportion of the total. Although fire in California has received much attention, it only accounts for 3% of the state's total fossil fuel CO₂ emissions.

These underestimates are especially alarming for Oregon where GHG reduction targets are to be 10% below 1990 levels by 2020 and at least 75% below 1990 levels by 2050 (Pietz and Gregor 2014). California and Washington emissions are to be reduced to 1990 levels by 2020 (Nunez 2006), and 80% and 50% below 1990 levels by 2050 (Washington State 2008), respectively.

In contrast, the US EPA reports emissions from wood product decay and landfills (EPA 2017) per the IPCC guidelines (IPCC 2006) (table S1). However, combustion emissions from logging and mill residues are not reported (EPA 2017). Moreover, ecosystem carbon losses are indirectly estimated through changes in biomass pools with measurement uncertainty that can be greater than the change (Ferster *et al* 2015). So even at the national level, emissions (as a fraction of fossil fuel emissions) would be underestimated by 10% and 24% in Washington and Oregon, respectively. Undoubtedly, there are implications for reduction mandates when the magnitude of emissions themselves are incorrect.

Conclusions

The goal for all societies and governments as stated in Article 2 of the *United Nations Framework Convention on Climate Change* (Oppenheimer and Petsonk 2005) should be '...stabilization of GHG concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system.' The Paris Climate Agreement (UNFCCC 2015) aims to keep global average temperature from rising by no more than 2 °C above preindustrial levels, and if possible no more than 1.5 °C. Forests are identified as part of the strategy (UNFCCC 2015).

Although some US states have attempted to quantify a portion of forest-related emissions, improved estimates are essential to track emissions to meet

reduction goals. We identified the main components that should be part of the forest sector state estimates. We found that emissions have been underestimated by up to 55% in Oregon and 25% in Washington, and that at present, these emissions are not reported in state GHG reporting guidelines. The accuracy of forest sector emissions estimates can be improved with sub-regional data on residential and commercial building lifespans, recycling, verifiable substitution benefits and accurate monitoring of growth rates of forests. However, verifiable substitution of one material for another may be more readily quantified in the fossil fuel sector.

The 2006 IPCC GHG guidelines provide three different approaches for calculating emissions from harvested wood products (IPCC 2006) (including reporting 'zero') and reporting of this component is not required by UNFCCC. To complicate accounting further, several studies have shown that using the different recommended approaches results in emissions that differ by over 100% (Green *et al* 2006, Dias *et al* 2007). Moreover, according to IPCC and UNFCCC, emissions of CO₂ from forest bioenergy are to be counted under land use change and not counted in the energy sector to avoid double counting. However, this provides a 'loophole' leading to their not being counted at all.

The United States government currently requires all federal agencies to count forest bioenergy as carbon neutral because the EPA assumes replacement by future regrowth of forests somewhere that may take several decades or longer (EPA 2018). While it is theoretically possible that a replacement forest will grow and absorb a like amount of CO₂ to that emitted decades or a century before, there is no guarantee that this will happen, and the enforcement is transferred to future generations. In any rational economic analysis, a benefit in the distant future must be discounted against the immediate damage associated with emissions during combustion. Furthermore, the goal for climate protection is not climate neutrality, but rather reduction of net GHGs emissions to the atmosphere to avoid dangerous interference with the climate system. Allowing forests to reach their biological potential for growth and sequestration, maintaining large trees (Lutz *et al* 2018), reforesting recently cut lands, and afforestation of suitable areas will remove additional CO₂ from the atmosphere. Global vegetation stores of carbon are 50% of their potential including western forests because of harvest activities (Erb *et al* 2017). Clearly, western forests could do more to address climate change through carbon sequestration if allowed to grow longer.

Since it is now clear that both CO₂ emissions and removal rates are essential to meet temperature limitation goals and prevent irreversible climate change, each should be counted and reported. We recommend that international agreements and states utilize a consistent and transparent carbon LCA that explicitly

accounts for all forest and wood product storage and emissions to determine compliance with goals to lower atmospheric GHGs. Only by using a full accounting of GHGs can the world manage its emissions of heat trapping gases to achieve concentrations in the atmosphere that will support a stable climate.

Materials and methods

We calculated the 2001 to 2016 average net forestry sector emissions from cradle-to-grave, accounting for all carbon captured in biomass and released through decomposition by forest ecosystems and wood products industry in Washington, Oregon, and California. Building on our previous work (Harmon *et al* 1996, Hudiburg *et al* 2011, Law *et al* 2013, Law *et al* 2018), we developed a modified and expanded LCA method to combine with our ecosystem carbon balance, now called Forest-GHG (version 1.0; figure 1 and box 1). We accounted for all carbon removed from forests through fire and harvest. All harvested carbon was tracked until it either was returned to the atmosphere through wood product decomposition/combustion or decomposition in landfills, minus the amount semi-permanently stored in landfills (buried). This required calculating the carbon removed by harvest operations starting in 1900 to present day because a portion of the wood removed in the past century is still in-use or decomposing. In addition to carbon in biomass, we also accounted for all carbon emissions associated with harvest (equipment fuel, transportation, manufacturing inputs). Moreover, our wood product life-cycle assessment includes pathways for recycling and deposition in landfills. Finally, we give substitution credits for not using more fossil fuel intensive materials than wood used in construction of buildings and energy production.

Observed carbon stocks and fluxes (ecosystem carbon balance)

Carbon stock and flux estimates were calculated from over 30 000 forest inventory plots (FIA) containing over 1 million tree records in the region following methods developed in previous studies (Law *et al* 2018) (SI Methods). Flux calculations include NPP (Clark *et al* 2001) NEP, and NECB. The NECB represents the net rate of carbon accumulation in or loss from ecosystems.

Off-site emissions associated with harvest (LCA)

Decomposition of wood through the product cycle was computed using a LCA (Harmon and Marks 2002, Law *et al* 2018). A 117 year wood products pool (1900–2016) was simulated using reported harvest rates from 1900 to 2016 for Oregon and Washington (Harmon *et al* 1996, DNR 2017, Oregon Department of Forestry 2017) and from the California State Board of Equalization (CA 2018). Harvest was converted to

Box 1. Terminology and Flux Definitions for table 1

1. Forest Area = sum of all forest area in each state derived from US Forest Service forest area map (30 m resolution). Includes all ownerships.
1. NEP = Net Primary Production—heterotrophic respiration; microbial respiration as they decompose dead organic matter in an ecosystem.
1. Fire emissions = the emissions associated with *combustion* of organic matter at the time of the fire. Most of what burns is fine surface fuels, averaging 5% of aboveground biomass in mixed severity fires of Oregon and Northern California.
1. Harvest removals = Wood actually removed from the forest (not the total aboveground biomass killed). Removals are not equal to emissions but are the removed carbon from the forests at the time of harvest. This is subtracted from NEP along with fire emissions to calculate the net forest carbon balance from the viewpoint of the forest ecosystem.

NECB = NEP + Fire Emissions + Harvest Removals. The term is the simplest expression of forest carbon balance without tracking wood through the product life cycle. Although not all of the harvest removals will result in instant or near-term emissions, NECB still captures the impact of the removed carbon on the forest ecosystem carbon balance, and is consistent with international agreements (REDD+, conservation).

1. Harvest Residue Combustion = the emissions associated with combustion of slash piles; the branches, foliage, and non-merchantable wood left after harvest operations (remains in the forest) and burned onsite (assumed to be 50% of slash).
1. Harvest, Transportation, Manufacturing (FFE emissions) = the fossil fuel emissions associated with harvest (skidding, sawing, etc), transportation of logs to mills, manufacturing of wood and paper products, and transportation of products to stores (see table S5 for coefficients).
1. Wood Product Pool Annual Inputs = Harvest removals
1. Landfill Annual Inputs (from products) = The amount of wood and paper that is sent to landfills at end of life. In Forest-GHG, this occurs incrementally from 1950 to 1960 and then in 1961 is assumed to be constant at the current rate.
1. Wood Manufacturing Losses = fraction of wood that is lost at the mill (sawdust, etc) and is assumed to be returned to the atmosphere within one year through combustion (with 75% energy recapture) or decomposition.
1. Wood Product and Landfill Decomposition = fraction of the total wood product and non-permanent landfill carbon pools that is returned to the atmosphere annually.

Net Forest Sector Carbon Balance = sum of NECB and rows 5 through 10. Emission sources are rows 5, 6, 9, and 10. Sinks are rows 7 and 8.

1. Wood product substitution (Wood) = carbon credits that account for the displaced fossil fuel emissions when wood is substituted for a fossil fuel derived product in buildings (e.g. concrete or steel). We assume 0.54 g C fossil fuel emissions avoided per g of C of wood biomass used.

Box 1. (Continued.)

1. Wood product substitution (Energy) = carbon credits that account for the displaced fossil fuel emissions when wood is substituted for energy. In the Oregon, Washington, and California this primarily a mix of natural gas and coal. We include the biogenic emissions from combustion of forest-derived woody biomass and include an energy substitution credit if it is combusted with energy recapture.

Net Forest Sector Carbon Balance (with substitution credit) = sum of NECB and rows 5 through 12.

total aboveground biomass using methods from (Law *et al* 2018). The carbon emissions to the atmosphere from harvest were calculated annually over the time-frame of the analysis (1900–2016).

The coefficients and sources for the Forest-GHG LCA (figures 1(b) and S1) are included in table S1 through S6 with all units expressed as a function of the wood biomass being cut, transported, manufactured, burned, etc. We accounted for the fossil fuel emissions that occur during harvest (fuel for equipment) and the fossil fuel emissions associated with transport of wood to mills. Then, we accounted for the fossil fuel emissions associated with manufacturing of products followed by a second transportation emission for delivery of products to stores and warehouses. Wood that is not made into a wood or paper product (e.g. waste) is assumed to be combusted onsite at the mill (with 50% energy recapture as combined heat and power) or used in a product that will return the carbon to the atmosphere within one year (table 1 and box 1; Wood Manufacturing Losses).

Wood products are divided into varying product pools and are then tracked through the wood product cascade until end of life (figure 1(b)). Wood products are split into seven product pools: single-family homes, multi-family homes, mobile homes, non-residential construction, furniture and manufacturing, shipping, and other wood. We simulated wood product storage and emissions to 2050 for display purposes in the figures assuming a constant harvest rate after 2016.

We estimate the carbon pools and fluxes associated with buildings by separating buildings into components with different life spans (figures 3 and S3). This allows components and buildings to have a lag time before significant losses occur, and recognizes the difference between building life span and the residence time of carbon in a building. This also allows capacity for Forest-GHG to have component and building life spans evolve over time as construction practices and the environment (including biophysical, economic, and social drivers) change.

In Forest-GHG, a fraction of each year's new harvest is allocated to residential (single-family, multi-family, and mobile homes) and non-residential construction (Smith *et al* 2006). This fraction is further divided into the short-term (23%) and long-term (77%) components. The

resulting pools are tracked independently, quantifying losses through decay and demolition from the year they start until then end of the simulation.

All the components created in a given year are considered a building cohort that is also tracked separately each year. All components are summed to give the total amount of building carbon remaining in a cohort at a given time (figure S3). For each year, the amount lost to the atmosphere or to the landfills through demolition, is simply the current year's total wood product carbon pool plus the current years inputs and minus last year's total wood product carbon pool.

Substitution

We calculated wood product substitution for fossil fuel derived products (concrete, steel and energy). The displacement value for product substitution was assumed to be 0.54 Mg fossil C/Mg C (Smyth *et al* 2017, Dugan *et al* 2018) wood use in long-term structures (Sathre and O'Connor 2010). Although the displacement value likely fluctuates over time, we assumed it was constant for the simulation period. We accounted for losses in product substitution associated with building replacement (Harmon *et al* 2009), but ignored the leakage effect related to fossil C use by other sectors. We assumed 75% of 'waste wood' was used for fuelwood in homes or at mills (wood manufacturing losses in table 1). We accounted for displacement of fossil fuel energy sources using a displacement factor of 0.68 assuming a mix of coal and natural gas replacement (Smyth *et al* 2017, Dugan *et al* 2018).

Uncertainty estimates and sensitivity analysis

We calculate a combined uncertainty estimate for NEP and NECB using the uncertainty in the observations and input datasets (climate, land cover, harvest amounts). For the biomass and NPP observations, we performed Monte Carlo simulations of the mean and standard deviations for NPP (Hudiburg *et al* 2011) derived for each plot using three alternative sets of allometric equations. Uncertainty in NECB was calculated as the combined uncertainty of NEP, fire emissions (10%), harvest removals (7%), and land cover estimates (10%) using the propagation of error approach. Sensitivity analysis was only used for the long-term wood product pool by varying the average life spans of buildings by ± 25 years in our new cohort component method. Our estimates varied by 7%. This was combined with the uncertainty in NECB to calculate total uncertainty on the net forest sector carbon balance.

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Forest Carbon Emission Sources Are Not Equal: Putting Fire, Harvest, and Fossil Fuel Emissions in Context

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Climate change has intensified the scale of global wildfire impacts in recent decades. In order to reduce fire impacts, management policies are being proposed in the western United States to lower fire risk that focus on harvesting trees, including large-diameter trees. Many policies already do not include diameter limits and some recent policies have proposed diameter increases in fuel reduction strategies. While the primary goal is fire risk reduction, these policies have been interpreted as strategies that can be used to save trees from being killed by fire, thus preventing carbon emissions and feedbacks to climate warming. This interpretation has already resulted in cutting down trees that likely would have survived fire, resulting in forest carbon losses that are greater than if a wildfire had occurred. To help policymakers and managers avoid these unintended carbon consequences and to present carbon emission sources in the same context, we calculate western United States forest fire carbon emissions and compare them with harvest and fossil fuel emissions (FFE) over the same timeframe. We find that forest fire carbon emissions are on average only 6% of anthropogenic FFE over the past decade. While wildfire occurrence and area burned have increased over the last three decades, per area fire emissions for extreme fire events are relatively constant. In contrast, harvest of mature trees releases a higher density of carbon emissions (e.g., per unit area) relative to wildfire (150–800%) because harvest causes a higher rate of tree mortality than wildfire. Our results show that increasing harvest of mature trees to save them from fire increases emissions rather than preventing them. Shown in context, our results demonstrate that reducing FFEs will do more for climate mitigation potential (and subsequent reduction of fire) than increasing extractive harvest to prevent fire emissions. On public lands, management aimed at less-intensive fuels reduction (such as removal of “ladder” fuels, i.e., shrubs and small-diameter trees) will help to balance reducing catastrophic fire and leave live mature trees on the landscape to continue carbon uptake.

Keywords: carbon, forests, fire, climate change mitigation, GHG emissions

INTRODUCTION

Climate change has intensified and increased the scale of global wildfire impacts in recent decades (Bowman et al., 2020). The western United States 2020 fire season exemplified intensifying fire impacts (Higuera and Abatzoglou, 2020), including high loss of life and property, and the record area burned in the last century in California, Oregon, and Colorado

(Higuera and Abatzoglou, 2020). Historically, similar catastrophic wildfires events (i.e., the 1910 Big Burn) instigated development of management policies to prevent and contain wildfire, including a century of fire suppression. In the western United States, climate change is now amplifying the negative effects of these management practices, resulting in unprecedented catastrophic wildfire outcomes (Parks and Abatzoglou, 2020).

Forests provide many ecosystem services such as wildlife habitat, hydrologic benefits, recreation opportunities, and wood harvest (Lawler et al., 2014), and also serve as a critical “natural climate solution”; they act as extensive and persistent carbon sinks that sequester large amounts of carbon from the atmosphere (Turner et al., 2011; Fargione et al., 2018). Increases in climate change-driven wildfire events (Westerling et al., 2006) have led to proposals to increase extractive forest harvest (i.e., the removal of large, mature trees, including altering policy to increase diameter limits to remove even larger trees; **Table 1**) in areas at high-risk of wildfire to decrease fire risk (**Figure 1**; Executive Order, 2018; Infrastructure Investment and Jobs Act, 2021). Public opinion and policies have been shaped by the misconception that harvest can reduce fire risk (or save other trees), or that harvest of a singular tree can save that tree from “burning down” (**Table 2**). These beliefs are widespread (**Table 2**), but their impact on policy and subsequent impact on on-the-ground harvest has not been quantified. While prescribed fire has been shown to decrease fire risk (Kolden, 2019) and increase carbon storage (Wiedinmyer and Hurteau, 2010), removal of biomass through large-diameter tree thinning or logging produces mixed outcomes for fire risk mitigation and forest resilience (Sohn et al., 2016) and reduces forest carbon storage and sequestration for decades to centuries (Campbell et al., 2012; Bartowitz et al., 2019; Stenzel et al., 2021). The misconception that trees need to be saved from wildfire through harvest (Zinke, 2018; Infrastructure Investment and Jobs Act, 2021; **Table 2**) may lead to unintended consequences through increased logging. These consequences include increased fire risk, a decreased forest carbon sink, decreased forest resiliency, and loss of the forest as a natural climate solution (Hudiburg et al., 2013; Law et al., 2018; Zald and Dunn, 2018; Stephens et al., 2020).

Although high intensity fire combusts less than 5% of mature, live tree biomass (Knorr et al., 2016), discussions of fire policy and forest management have framed tree biomass combustion as an undesirable outcome requiring mitigation through extractive forest management (i.e., harvest of mature trees for timber sales; Mater, 2017; Zinke, 2018; Newhouse, 2021; Senate Bill 762, 2021). Increasing, i.e., extractive forest management (**Table 1**), to “lock” carbon into man-made structures, to increase forest productivity (CORRIM, 2020), or reduce fire risk ignores the volume of forest fire emissions relative to the direct emissions of such strategies (Hudiburg et al., 2019; Stenzel et al., 2019). Previous studies have shown that timber harvest directly kills more trees than forest fire in the western United States (Berner et al., 2017), but it remains unclear how much fire and harvest are contributing to regional total carbon emissions in the western United States, especially in the context of how these emissions compare with anthropogenic FFE (Hudiburg et al., 2019; Stenzel et al., 2019).

Here, we calculate forest fire emissions (average for the last decade; large historical events, and the record 2020 fire season) and compare those to (1) average and hypothetical timber harvest emissions, and (2) average decadal FFE. Our comparisons clarify the relative contributions of extractive forest management, fire, and fossil fuels to atmospheric carbon dioxide concentrations and help provide clarity for future management scenarios intended to reduce carbon emissions and/or increase carbon uptake and the scientific observations that show the opposite occurs. We further show how misrepresentation of fire and management impacts on forest carbon cycling leads to discussions and policies that overestimates benefits to carbon stocks and sequestration, or downplays carbon consequences (**Figure 1**). Finally, we discuss how policy and management based on carbon cycle science and observations could be used to both reduce fire risk and to increase and maintain carbon storage.

MATERIALS AND METHODS

Study Area: Forest Fires Across the Western United States

We calculated carbon emissions from the forest fires in the western United States (**Figure 1**) between 1984 and 2020 and the largest fire in the continental United States, the 1910 Big Burn. Here, we group the “western United States” as the 11 states in the contiguous United States West (i.e., Arizona, California, Colorado, Idaho, Montana, Nevada, New Mexico, Oregon, Utah, Washington, and Wyoming). While there were other extreme forest fires in the 20th century (e.g., 1902 Yacoult Burn in Washington, 1933 Tillamook Burn), historic records of forest attributes were not available for analysis.

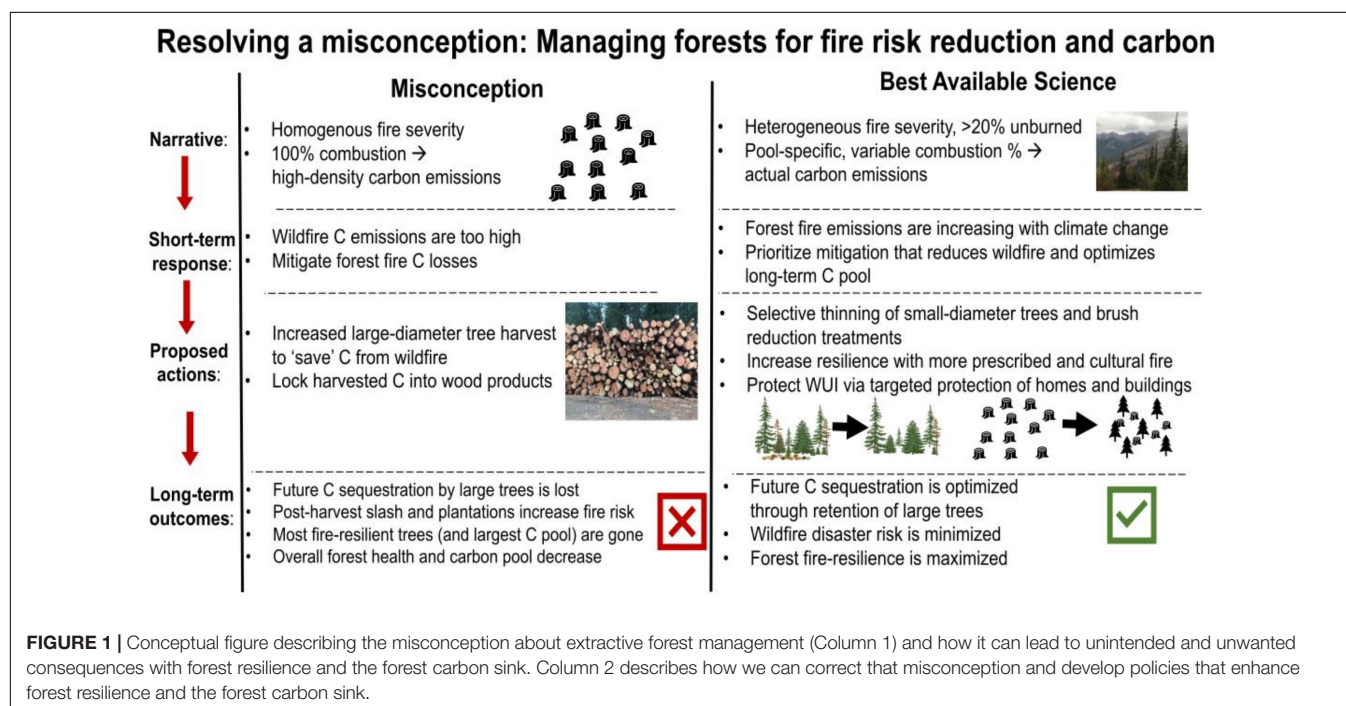
Extreme fires have continued to occur in recent decades (**Figure 2**). Availability of high-resolution fire perimeter and burn severity data allows for analysis of fires since 1984 (Eidenshink et al., 2007) through 2020. All wildfires >526 ha (1,000 acres) with >50% forest area within the burn perimeter were included in this analysis. In addition, we selected large, notable forest fires (or complexes of individual fires; referred to as “extreme” fires throughout the manuscript) that occurred between 1984 and 2020. Fires were selected based on how notable they were at the time for size, duration, volatile fire behavior and legacy of impact in the subsequent years. Wildfires included in the “extreme fires” list (**Table 1**) were chosen from the created emissions database based on high area burned (i.e., >40,000 ha), and overall significance of fire event (i.e., most were record events in some way such as: highest area burned in that state or human impacts).

Emissions have not been previously calculated for the Big Burn. We have calculated an estimate from the Big Burn not only because it is the largest known fire to have occurred in the continental United States, but also to serve as a baseline or reference for the range of emissions possible in the absence of fire suppression. While the Big Burn emissions estimate is calculated differently from modern fires due to lack of forest data from that time, the comparison between modern fire emissions and the Big Burn is still useful and has been completed with the best possible methodology given data availability. The 1910 Big Burn

TABLE 1 | Common management scenario types in western United States forests.

Management scenario	Description	Calculation scenario (Figure 4)	Extractive management?
Thin-from-below	Removal of understory brush and small-diameter trees. No tree sales.	30% harvest	No
Commercial thin	Removal of understory brush, small-diameter trees, and some larger, mature trees for sale.	50% harvest	Yes
Clear-cut removal	Removal of all trees (small and mature) for sale.	100% harvest	Yes
Prescribed burn	Intentionally set fire to remove ground fuels. Often coupled with a restoration or commercial thin.	Not included in calculations	No

Tree removal scenarios (thins and clear-cut) were used in the hypothetical harvest carbon loss calculations (Figure 4). Extractive management (i.e., if there is also a financial sale from the management rather than just for restoration or fuels reduction) is noted for each scenario.

**TABLE 2** | Examples of recent public opinions surrounding the logging-forest-carbon misconceptions and how they lead to policy that increases harvest.

Source	Year	Author	Quote or summary	Description
Public	2020	Logging and grazing organizations (Radke, 2020)	"Log it, graze it, or watch it burn it"	Logging and grazing orgs believe logging and grazing will solve the fire problem
Public	2020	Consortium for Research on Renewable Industrial Materials (CORRIM, 2020)	"Sustainably harvesting forest carbon not only provides significant opportunities for carbon storage"	Logging industry promotion trying to show "reducing carbon emissions by using wood products"
Public	2018	Former United States Cabinet Member Ryan Zinke (Zinke, 2018)	"When an entire forest burns to the ground"	Advocates for logging to prevent wildfires, boost the economy, and to save lives.
Public	2017	Catherine mater (Mater, 2017)	"Half of those emissions are due to tree mortality"	Misconception that tree mortality equals direct emissions
Policy	2021	United States Government – Infrastructure Investment and Jobs Act (Infrastructure Investment and Jobs Act, 2021)	Significant increase in project funds to increase logging and commercial thinning on public lands for fire risk reduction	\$3.3 billion allocated to hazard fuels reduction with no diameter limits set. 12 million ha opened to logging on public lands.
Policy	2020	United States Government – Twisp River Restoration Project (USFS, 2020)	Increase diameter limits on trees harvest to cut down larger trees for fire risk reduction and restoration	>30,000 ha forest management project in fire-prone forest in Washington state

encompassed an area throughout Washington, Montana, and Idaho (Figure 2). We calculated fire-induced carbon emissions of the Big Burn using historical accounts and records (Koch, 1942).

The fire perimeters used in this study were a cross-reference between Koch's account and the 1910 Fire perimeters (USFS, 1978; Gibson, 2005).

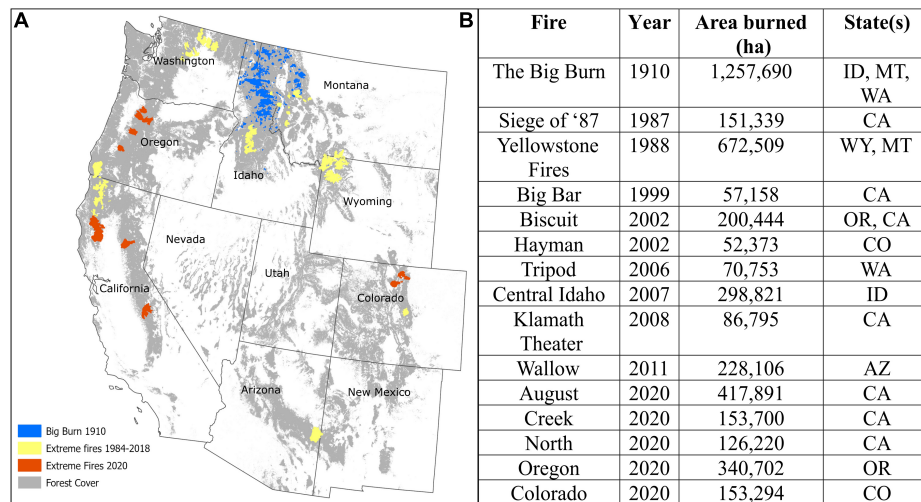


FIGURE 2 | Extreme forest wildfires in the western United States. **(A)** Perimeters of forest fires in 2020, a selection of extreme forest fire events 1984–2018, and of the 1910 Big Burn fires. **(B)** Fire statistics of the 1910 Big Burn and contemporary fires (1984–2020) within the Western United States.

Forest Fire Emissions Calculations

Direct carbon emissions for contemporary forest fires (1984–2020) were calculated using, fire severity and area burned from the Monitoring Trends in Burn Severity database (MTBS; Eidenshink et al., 2007) for forest fires between 1984 and 2019, and Burned Area Emergency Response (BAER; Parsons, 2003) and National Interagency Fire Coordination (NIFC, 2020) data products for 2020 fires. Carbon emissions for the 1910 Big Burn were calculated using area burned from the Northern Rockies Fire atlas for the Big Burn (Gibson, 2005). All carbon stock calculations were from forest type and ecoregion-specific carbon data (**Supplementary Figure 1**; Buotte et al., 2019; Stenzel et al., 2019), and severity-specific combustion factors (**Supplementary Table 1**; Stenzel et al., 2019). Only fires that burned > 526 ha and in 50% or greater forested area within the burn perimeter (Ruefenacht et al., 2008) were used in this analysis. Aboveground carbon stocks were calculated for each forest fire area based on average carbon stocks for the forest type and ecoregion and area of the specific forest type within the burn perimeter (**Supplementary Table 1**). Aboveground carbon stocks were multiplied by the appropriate combustion factor for the fire severity value of that forested area to obtain carbon losses (**Supplementary Table 1**). Fires between 1984 and 2019 were calculated using MTBS severity classes. Smaller forest fires in 2020 were calculated using an average combustion factor. Big Burn carbon emissions were calculated using a range for the moderate-severe combustion factor which gives us a range (uncertainty) of emissions for this fire. Extreme 2020 forest fire emissions were calculated using BAER severity classes (which are precursors to MTBS severity classes). While we used contemporary forest structure data to calculate emissions from the Big Burn, our range of carbon emissions (**Table 3**) from this fire are robust because we used stem tree biomass and demographic data from pre-1910 timber cruise inventories (Koch, 1942) to identify FIA plots

of similar structure (diameter and heights) and age classes to the 1910 inventory.

We calculated mean annual forest fire emissions for the western United States and each western United States state based on the 2009–2018 decade, to best represent the observed trends toward increased area burned under climate change (Abatzoglou and Williams, 2016). These calculations were completed using MTBS (Eidenshink et al., 2007) perimeter and severity data, carbon stock data, and combustion factors. All fire carbon losses were converted to Tg CO₂e (i.e., tera-grams CO₂ equivalent) for comparison with fossil fuel emissions (FFE). To normalize emissions on a per area basis, we calculated megagrams of carbon lost per hectare burned.

To calculate uncertainties for contemporary forest fire emissions, we used a propagation of error approach. We combined uncertainty estimates of each emissions calculation component to calculate total uncertainty for each individual extreme fire event (**Table 4**). We used the Combining Uncertainties Propagation of Error estimates from the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (Eq. 1; IPCC, 2006).

Equation 1: Combining Uncertainties (individual associated uncertainties):

$$U_{\text{total}} = \frac{\sqrt{(U_1 * x_1)^2 + (U_2 * x_2)^2}}{|x_1 + x_2|}$$

Where U_{total} = the percentage uncertainty in the sum of the quantities (half the 95 percent confidence interval divided by the total, i.e., mean and expressed as a percentage).

x_i and U_i = the uncertain quantities and the percentage uncertainties associated with them. x_i refers to the specific fire emission calculation. U_1 refers to the uncertainty in biomass calculations (0.05) and U_2 refers to remote sensing uncertainty in area burned calculations (0.10).

TABLE 3 | Carbon emissions from 1910 Big Burn and extreme contemporary (1984–2020) forest fires in the western United States.

Complex	Year	Area burned (ha)	Tg C	Mg C ha ⁻¹	Tg CO _{2e}
Big Burn	1910	966,564–1,257,690*	29.79–49.87**	23.69–39.65**	94.60–158.51**
Siege of '87	1987	151,339	3.58 ± 0.20	23.70 ± 1.32	13.10 ± 0.73
Yellowstone	1988	672,509	16.3 ± 0.91	24.30 ± 1.36	59.90 ± 3.35
Big Bar	1999	57,158	1.40 ± 0.08	24.60 ± 1.38	5.10 ± 0.29
Biscuit	2002	200,444	4.40 ± 0.25	22.0 ± 1.23	16.10 ± 0.90
Hayman	2002	52,373	1.53 ± 0.09	29.20 ± 1.63	5.60 ± 0.31
Tripod	2006	70,753	1.31 ± 0.07	18.60 ± 1.04	4.80 ± 0.27
Central Idaho	2007	298,821	5.96 ± 0.33	20.0 ± 1.12	21.90 ± 1.22
Klamath Theater	2008	86,795	2.16 ± 0.12	25.0 ± 1.40	7.90 ± 0.44
Wallow	2011	228,106	3.39 ± 0.19	14.90 ± 0.83	12.40 ± 0.69
August	2020	417,891	9.02 ± 0.50	24.60 ± 1.38	33.10 ± 1.85
Creek***	2020	153,700	4.56 ± 0.25	29.70 ± 1.66	16.70 ± 0.93
North	2020	126,220	2.35 ± 0.13	18.60 ± 1.04	8.60 ± 0.48
Oregon****	2020	340,702	8.18 ± 0.46	24.0 ± 1.34	30.0 ± 1.68
Colorado****	2020	153,294	4.23 ± 0.24	27.60 ± 1.54	16.90 ± 0.94

*The area estimates are from Koch (1942) and Gibson (2005).

**The emissions from Big Burn were calculated using present day USFS Forest Inventory and Analysis data. Here we provide a range of carbon emissions values, rather than uncertainty ranges, because we use a range of combustion factors as we do not have detailed severity data.

***Prior to the 2020 Creek Fire, this forest had a proportion killed by the 2012–2017 drought and was subsequently salvaged logged. Our emissions calculation is likely an overestimate due to the large amount of biomass already removed from salvage logging, and slash left behind from salvage logging driving higher fire severity.

****Oregon and Colorado forest fire area and emissions are aggregated for the 2020 extreme forest fires.

TABLE 4 | Average fossil fuel emissions, forest fire emissions, and harvest emissions, 2020 fire emissions, and record year fire emissions (2008–2020) for all western United States and for the entire western United States region.

State	10-year average fossil fuel emissions (TgCO _{2e})	10-year average forest fire emissions (TgCO _{2e})	10-year average harvest emissions (TgCO _{2e})	2020 forest fire emissions (TgCO _{2e})	Record year forest fire emissions (TgCO _{2e})	Record year
Arizona	92.76 ± 1.57	7.02 ± 0.79	0.41 ± 0.07	8.28 ± 0.93	21.34 ± 2.39	2011
California	370.71 ± 4.98	14.09 ± 1.58	7.38 ± 1.27	121.92 ± 13.63	121.92 ± 13.63	2020
Colorado	91.84 ± 1.04	4.92 ± 0.55	0.36 ± 0.06	33.35 ± 3.73	33.35 ± 1.91	2020
Idaho	17.66 ± 0.32	9.81 ± 1.09	5.03 ± 0.86	5.06 ± 0.56	31.37 ± 3.51	2012
Montana	32.86 ± 0.71	8.95 ± 1.00	2.03 ± 0.35	2.20 ± 0.25	35.30 ± 3.95	2017
Nevada	37.24 ± 1.46	0.64 ± 0.07	0.03 ± 0.01	0.59 ± 0.07	2.46 ± 0.28	2018
New Mexico	52.68 ± 1.06	0.16 ± 0.02	0.17 ± 0.03	1.33 ± 0.15	1.33 ± 0.15	2020
Oregon	39.58 ± 0.60	7.36 ± 0.82	19.38 ± 3.33	35.78 ± 4.00	35.78 ± 4.00	2020
Utah	63.01 ± 0.89	1.85 ± 0.21	0.14 ± 0.03	3.21 ± 0.36	7.01 ± 0.78	2018
Washington	77.70 ± 1.21	4.10 ± 0.45	14.72 ± 2.53	2.11 ± 0.24	18.68 ± 2.09	2015
Wyoming	64.91 ± 0.60	0.94 ± 0.01	0.18 ± 0.03	10.25 ± 1.15	10.25 ± 1.15	2020
Total WUS	941.00 ± 10.74	59.95 ± 6.70	49.88 ± 8.56	224.09 ± 25.05	224.09 ± 25.05	2020

Calculations and spatial analyses were conducted using R (R Core Team, 2017) and ESRI software (ESRI, 2020).

Timber Harvest and Wood Product Emissions

Hypothetical Harvest Carbon Losses

Hypothetical harvest carbon losses were calculated for all states for burned areas between 2009 and 2019; these are the exact burned areas and pre-fire carbon stocks used to calculate forest fire emissions for this study. This hypothetical calculation allows us to directly compare fire carbon losses to harvest carbon losses on a per area basis. Here, we calculated three scenarios for

standing tree carbon (both live trees and snags): 30% harvest, 50% harvest, and 100% harvest. Both 30% and 50% harvests are meant to represent different levels of thinning (thinning-from-below and commercial, respectively), while 100% harvest is akin to a clear-cut harvest (Table 1). For these scenarios, the fraction (30, 50, or 100%) of aboveground carbon for standing live and dead trees was calculated and counted as carbon loss, and then converted to a per area basis. For the hypothetical thinning scenarios we did not include carbon stored in wood products because very little to no long-term wood products would be created from the smaller-diameter trees removed from these types of thinning. These smaller-diameter trees will most likely be used in short-term wood products such as

paper (Hudiburg et al., 2019). The hypothetical clear-cut used a static 60% emission from aboveground tree carbon stocks, with 40% remaining in long-term wood products pools. To normalize harvest carbon losses on a per area basis, we calculated megagrams of carbon lost per hectare harvested.

Actual Timber Harvest Carbon Losses

Actual timber harvest calculations were aggregated from publicly-available state and federal historical harvest sources (Hudiburg et al., 2019), including privately-owned lands. Detailed methodology can be found in Hudiburg et al. (2019). We calculated a mean annual harvest loss for the most recent available harvest data for each state (2007–2016) as well as an annual average for the entire western United States. Reported harvest volumes (merchantable) were converted to grams carbon using board feet to cubic volume estimates from Keegan (Keegan et al., 2010). Our calculations include the carbon stored (and released from at end of life) in wood products for the years of this analysis. Wood was assumed to enter short-term (1 to 10 years before emissions return to the atmosphere; includes wood waste at the mills) and long-term (50-year half-life) product pools at rates of 60 and 40%, respectively, (Heath et al., 2010; Hudiburg et al., 2019). All timber harvest carbon losses were converted to Tg CO₂e for comparison with FFE.

Fossil Fuel Emissions

Fossil fuel emission numbers were sourced from the Environmental Protection Agency (EPA, 2020). The EPA provides yearly, state-by-state FFE from nearly all emissions sources (i.e., commercial, industrial, residential, transportation, and electric power; Table 4). We calculated an annual average for the most recent available data for each state (2009–2018), as well as an annual average for the entire western United States.

Forest Fire and Fossil Fuel Emission Comparisons

Carbon emissions from forest fires and FFE were normalized on a state-by-state basis by normalizing both average FFE (2009–2018) and record year fire emissions (1984–2020) with average fire emissions (2009–2018). A factor of both average FFE and record year fire emissions over the average fire emissions was calculated for each state.

RESULTS

Forest Fire Carbon Emissions

Carbon emissions from 1984 to 2020 wildfire events varied considerably by fire severity (Supplementary Figure 1), forest type (e.g., varied carbon density), and size (Figure 2). As forest fire carbon emissions are a product of forest type (pre-fire aboveground carbon density per area, Figure 3) and fire severity (Table 4), it is notable that Colorado fires generally exhibit higher emissions per unit area (27.60 Mg C ha⁻¹) compared to other 2020 fire events in Oregon and California, although the 2020 Creek Fire in California had the highest emissions per unit area for a single contemporary fire (Table 4). This highlights how

severely Colorado wildfires have burned in recent decades, given their lower pre-fire carbon density. By contrast, Oregon forests have much higher pre-fire carbon density and slightly lower area-normalized emissions compared to other western fires because they burned, on average, at a lower severity (Figure 3). When normalized by area burned to control for size, there is notable variation amongst the contemporary extreme fires in emissions per hectare (Table 4). For example, the 2020 Creek Fire in California had the highest emissions per hectare (29.7 Mg C ha⁻¹), and fires in Colorado all exceed the Idaho, Montana, and Wyoming fires, and many in the carbon-dense forests of Oregon when normalized by area. In addition, >90% of the burn area of extreme forest fires in 2020 were in low-to-moderate severity classes (Supplementary Figure 1).

Harvest Carbon Losses

Total average annual western United States total harvest emissions were lower than total average forest fire emissions (Table 4), however, actual harvest area is much lower than area burned (Berner et al., 2017). Actual harvest carbon losses vary greatly by state, with carbon-dense Oregon and Washington having the highest biomass harvest removals (Table 4). However, on a per unit area basis, hypothetical 100% harvest is 2–8 times greater than fire for the same perimeters across the entire region (Figure 4). We calculated hypothetical harvest carbon losses for the exact burn areas in each state to compare per area harvest losses. We found that for all states a 50–100% harvest would have led to greater carbon losses than fire for those burned areas, and even a 30% harvest led to greater carbon losses than fire for all but four of the western United States. Hypothetical harvest carbon losses continue to outpace fire carbon losses on a per unit area basis for most scenarios (Figure 4).

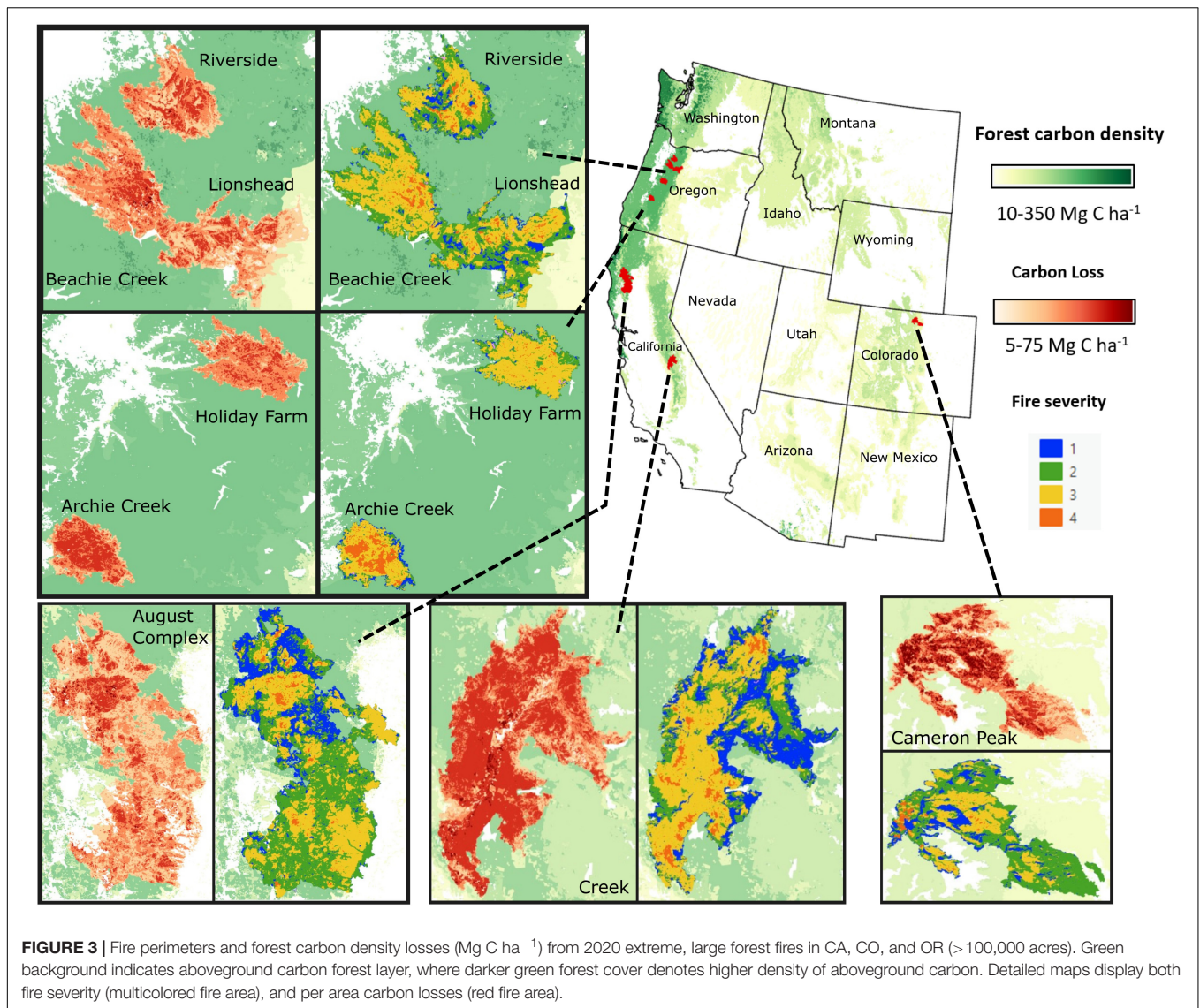
Anthropogenic Fossil Fuel Emissions

Anthropogenic fossil fuel emissions (AFFE) for each western United States and for the total western United States substantially exceed forest fire carbon emissions (mean annual and 2020), and average actual timber harvest (Figure 5 and Table 4). Mean annual AFFE in the western United States were over 15× higher than mean annual forest fire emissions and mean annual AFFE were 420% higher than forest fire emissions from the 2020 record fires across the west (Figure 5). Emissions vary widely by state, primarily due to population size (i.e., population and FFE are positively correlated) and large-scale, high-emissions industries within the state. Total western United States 2020 fire emissions were higher than the mean annual fire emissions (2009–2018), driven by large fire events in California, Oregon, and Colorado (Table 4). California, Oregon, Colorado, and New Mexico all had record-high forest fire emissions in 2020.

DISCUSSION

Forest Fire, Harvest, and Fossils Fuels: Putting Emissions in Context

Public perception and existing overestimates of forest mortality and carbon emissions from wildfire feeds into the misconception



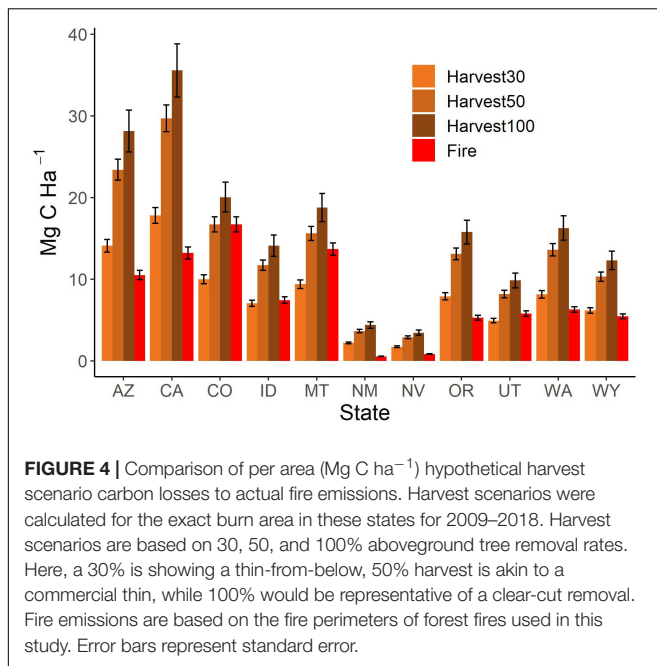
that wildfire kills all live forest cover and combusts all forest carbon (Wiedinmyer and Neff, 2007; Mater, 2017; Zinke, 2018). The reality of actual fire emissions calculated from mixed-severity combustion rather than overestimates calculated from the false high-severity narrative highlights the need to disentangle ecological impacts of wildfire from societal impacts (i.e., loss of lives and houses). This will help to ensure that risk-reduction solutions can decrease wildfire disasters while still maintaining ecosystem services, such as live tree carbon uptake and wildlife habitat (Kolden, 2020).

As wildfire policy discussions increasingly include extractive forest harvest to mitigate forest fires (Executive Order, 2018; Newhouse, 2021; Senate Bill 762, 2021), a comparison of emissions from forest fire, timber harvest, and fossil fuels provides a more complete understanding of the relative contributions of emissions sources to anthropogenic climate change. Despite increasing area burned trends across the western United States (Parks and Abatzoglou, 2020), FFE still greatly

outpace forest fire emissions in the last decade, including 2020. FFE are also significantly higher than intensive and large-scale land management operations like timber harvest in many United States (i.e., California).

Policy Implications and Ways Forward

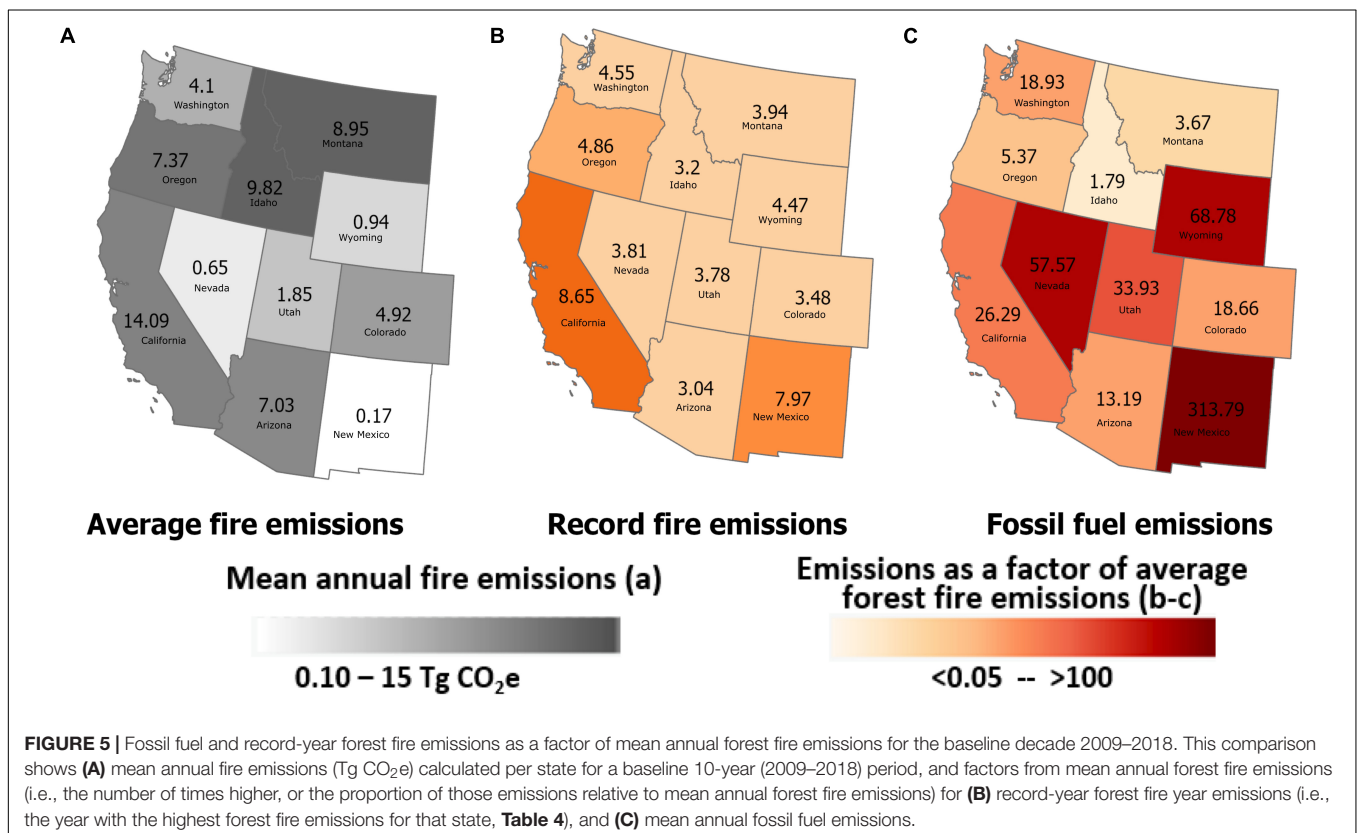
Much of United States fire management and policy has been shaped by specific, previously unprecedented wildfire events. The Big Burn of 1910 was the first massive fire event for the fledgling United States Forest Service (Koch, 1942), and is still the largest wildfire complex that has occurred in the contiguous United States. Fire suppression as the main form of fire control persisted until the late 20th century, when ecological restoration efforts began seeking to reduce hazardous fuels and increase ecologically beneficial fire effects (Parsons et al., 1986). However, these efforts have not yet altered the fire suppression culture instilled by 1910 (Stephens and Ruth, 2005; Kolden, 2019; McWethy et al., 2019).



Like past extreme fire events, the 2020 and 2021 fire seasons have accelerated fire policy and forest management discussions at all levels of government – federal, state, and local – including recent bills introduced in the United States

Senate (S.4625, S.4331). Many new policy discussions on fire and forest management are being based upon the misconception that harvest will protect forests from mortality and carbon loss (Executive Order, 2018; Zinke, 2018; Infrastructure Investment and Jobs Act, 2021; Newhouse, 2021), and decrease fire risk (Forest Climate Action Team, 2018; **Figure 1**) despite substantial uncertainty over long-term impacts to forest climate resilience (i.e., forest treatments may decrease forest resilience in the era of climate change). Our results and the majority of full-carbon accounting studies conclude that any type of harvest (logging or commercial thinning) decreases forest carbon storage (Law et al., 2013), and this research shows harvest emits more carbon per unit area than fire at all scales (**Figure 5**).

To mitigate climate change, it is key we understand exactly where emissions are originating. While increased intensity and size of fires are increasing overall fire emissions, these emissions are still substantially less than FFE. This is true even in record forest fire years (**Figure 5**). While the 2020 fire season was unprecedented in many ways (record forest area burned in California, Oregon, and Colorado; societal devastation including fatalities, thousands of homes consumed, dramatic evacuations, and regional hazardous air quality events), ecologically, most of the forested area burned in extreme fires was in a low-to-moderate severity class (>90%, **Supplementary Figure 1**). Moreover, while there is assumed high tree mortality in these forest fire perimeters, many of these burns were mixed-severity fires (**Supplementary Figure 1** and **Figure 3**) meaning many live trees will persist across most of the low-to-moderate severity



burned areas. Locations with high-harvest rates and carbon-dense forests, such as the Pacific Northwest United States, see higher carbon losses from harvest than fire compared to areas in the Southwest United States with low harvest rates and carbon-sparse forests (e.g., Oregon versus Arizona). Forest management needs to be specific to forest type and region; old-growth and wet forests in the Northwest are best left preserved while dry, fire-prone forests or areas in the Wildland Urban Interface benefit from fire risk-reduction strategies like small-diameter thinning and prescribed fires (Law et al., 2018; Case et al., 2021). Inclusion of specific diameter limits in policy for public lands could help prevent large-diameter tree removal and subsequent unintended consequences.

Forest management strategies that are site-specific and balance the immediate protection of life and property with long-term preservation of existing and potential carbon stocks in forests are critical to mitigating the negative impacts of climate change. The most effective forest management strategy to protect forest carbon stocks on public lands is to preserve forests through decreased harvest and thinning, lengthened harvest rotations, increased proportion of long-term wood products, reduced harvest and mill waste, and working toward afforestation and reforestation (Hudiburg et al., 2013; Law et al., 2018; Buotte et al., 2020; **Figure 1**). Prescribed burns reduce fire risk while minimizing carbon losses and amplify tree growth and carbon sequestration in large-diameter trees in fire-adapted forests (Hurteau and North, 2009). In western United States forests, 33 to 46% of aboveground live biomass is stored in the large diameter trees (> 60 cm; Lutz et al., 2018; Mildrexler et al., 2020). Carbon-smart treatments on public lands need to be specific about diameter limits to avoid large-diameter tree removal.

Here, we have shown that FFE for the western United States are 7 times greater than emissions associated with timber harvest and fire (**Figure 5** and **Table 4**). As more forest-fire policy and management plans are expanded, written, and discussed following extreme fires of the recent decades, and especially the extreme forest fires of 2020 (DNR, 2020), it is crucial that these policy changes focus on the largest driving factor of these fires – anthropogenic climate change. In practice, large-scale extractive forest management efforts will hamper climate mitigation and may be futile for decreasing fire risk. To be most effective, policy will need to focus on fire-wise adaptations for homes and property and disentangle ecologically-good fire from destructive fires (Kolden, 2020). Protecting forests with ecologically sound principles, rather than increasing extractive management, may be the best scenario for the mitigation of climate change (Law et al., 2018), and protecting humans, biodiversity, and forests (Walsh et al., 2019; Buotte et al., 2020; Law et al., 2021). The continued

escalation of fires throughout the 21st century is evidence of climate change-mediated intensification of fire regimes in the United States (Williams et al., 2019). Fire catastrophes will continue to occur and worsen if we do not focus on decreasing FFE, the primary driver of climate change (IPCC, 2018).

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are openly available from the University of Idaho at <http://doi.org/10.7923/VVZE-7642> (Bartowitz et al., 2022).

AUTHOR CONTRIBUTIONS

KB and TH: conceptualization, formal analysis, investigation, and writing—original draft preparation. KB, JS, EW, CK, and TH: methodology and writing—review and editing. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: <https://www.frontiersin.org/articles/10.3389/ffgc.2022.867112/full#supplementary-material>

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Tree mortality from fires, bark beetles, and timber harvest during a hot and dry decade in the western United States (2003–2012)

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Supplementary material for this article is available [online](#)

Abstract

High temperatures and severe drought contributed to extensive tree mortality from fires and bark beetles during the 2000s in parts of the western continental United States. Several states in this region have greenhouse gas (GHG) emission targets and would benefit from information on the amount of carbon stored in tree biomass killed by disturbance. We quantified mean annual tree mortality from fires, bark beetles, and timber harvest from 2003–2012 for each state in this region. We estimated tree mortality from fires and beetles using tree aboveground carbon (AGC) stock and disturbance data sets derived largely from remote sensing. We quantified tree mortality from harvest using data from US Forest Service reports. In both cases, we used Monte Carlo analyses to track uncertainty associated with parameter error and temporal variability. Regional tree mortality from harvest, beetles, and fires ($MORT_{H+B+F}$) together averaged 45.8 ± 16.0 Tg AGC yr^{-1} ($\pm 95\%$ confidence interval), indicating a mortality rate of $1.10 \pm 0.38\%$ yr^{-1} . Harvest accounted for the largest percentage of $MORT_{H+B+F}$ ($\sim 50\%$), followed by beetles ($\sim 32\%$), and fires ($\sim 18\%$). Tree mortality from harvest was concentrated in Washington and Oregon, where harvest accounted for $\sim 80\%$ of $MORT_{H+B+F}$ in each state. Tree mortality from beetles occurred widely at low levels across the region, yet beetles had pronounced impacts in Colorado and Montana, where they accounted for $\sim 80\%$ of $MORT_{H+B+F}$. Tree mortality from fires was highest in California, though fires accounted for the largest percentage of $MORT_{H+B+F}$ in Arizona and New Mexico ($\sim 50\%$). Drought and human activities shaped regional variation in tree mortality, highlighting opportunities and challenges to managing GHG emissions from forests. Rising temperatures and greater risk of drought will likely increase tree mortality from fires and bark beetles during coming decades in this region. Thus, sustained monitoring and mapping of tree mortality is necessary to inform forest and GHG management.

1. Introduction

Forests help regulate Earth's climate in part by sequestering carbon from the atmosphere (Bonan 2008, Pan *et al* 2011a, Anderson-Teixeira *et al* 2012), yet tree mortality from disturbance accelerates carbon transfer from these ecosystems back to the atmosphere (Kurz *et al* 2008, Baccini *et al* 2012, Brinck *et al* 2017).

Forests globally store a similar amount of carbon as the atmosphere (Pan *et al* 2011a, Houghton 2013), with much of the carbon ($\sim 42\%$) held in the biomass of living trees (Pan *et al* 2011a). Disturbances such as forest fires, insect outbreaks, and timber harvest can kill trees over large areas each year (Goetz *et al* 2012, Meddens *et al* 2012, Kautz *et al* 2016, Williams *et al* 2016) and thus contribute to increased regional

carbon emissions as tree biomass subsequently decomposes or is rendered into wood products that have finite longevity (Harmon *et al* 1990, Harmon *et al* 2011, Ghimire *et al* 2015). Carbon emissions from forest disturbance can challenge efforts to meet greenhouse gas (GHG) emission targets (Gonzalez *et al* 2015), but are also highly uncertain in many parts of the world (Pacala *et al* 2010). Both the growing global demand for wood products (FAO 2017) and the increase in forest disturbance due to ongoing climatic change (Allen *et al* 2010, Williams *et al* 2012, Kautz *et al* 2016) underscore the need to better understand carbon implications of tree mortality from disturbance.

Forest disturbance by fires and insects increased during recent decades in the western contiguous United States as the regional climate became warmer and more arid (Westerling *et al* 2006, Williams *et al* 2012, Dennison *et al* 2014, Hicke *et al* 2015, Abatzoglou and Williams 2016). Regional mean annual air temperature increased 0.8 °C–1.1 °C from 1895 to 2011, with most of the warming having occurred in recent decades (Mote *et al* 2014, Walsh *et al* 2014). Higher temperatures contributed to higher atmospheric vapor pressure deficits (Abatzoglou and Williams 2016), reduced mountain snowpack (Mote *et al* 2005), and more frequent and severe drought (McCabe *et al* 2004, Diffenbaugh *et al* 2015). For instance, the western US recently experienced its most severe drought (2000–2004) in the past 800 years (Schwalm *et al* 2012), with hot and dry conditions then prevailing through the 2000s (Diffenbaugh *et al* 2015, Abatzoglou and Williams 2016). These conditions contributed to extensive forest disturbance by fires and bark beetles relative to recent decades (Williams *et al* 2012, Creeden *et al* 2014, Abatzoglou and Williams 2016). High temperatures and drought increase regional forest fire occurrence (Littell *et al* 2016) and the likelihood of post-fire tree mortality (i.e. increased fire severity; van Mantgem *et al* 2013), while also increasing beetle populations and the vulnerability of drought-stricken trees to beetle attack (Raffa *et al* 2008, Creeden *et al* 2014, Hart *et al* 2014). Projections indicate that regional temperatures could rise another ~3.8 °C–5.5 °C by the end of the 21st century and that much of the region, particularly the Southwest, could become increasingly arid and prone to drought under a high GHG scenario (RCP 8.5; Kunkel *et al* 2013, Walsh *et al* 2014, Cook *et al* 2015). These changes in regional climate could further accelerate tree mortality (Adams *et al* 2009, Allen *et al* 2015) and increase carbon emissions from forest ecosystems (Spracklen *et al* 2009, Jiang *et al* 2013, McDowell *et al* 2015).

Several states in the western US have GHG reduction targets (e.g. Oregon, California) and would thus benefit from information on the magnitude and primary causes of recent tree mortality. Prior studies have shown that fires, bark beetles, and timber harvest

are important causes of tree mortality in this region (Masek *et al* 2011, Meddens *et al* 2012, Hicke *et al* 2015). In this study, we asked ‘What was the magnitude and relative contribution of mean annual tree mortality from fires, bark beetles, and timber harvest from 2003–2012 both regionally and among the 11 western states?’ Tree mortality can be quantified over large areas in terms of carbon using remote sensing estimates of tree aboveground biomass (AGB) together with information on the carbon content of AGB, as well as disturbance extent and severity (Baccini *et al* 2012, Hicke *et al* 2013). Here we quantified tree mortality as the amount of *carbon stored in tree AGB* (AGC) killed by disturbance (e.g. Mg AGC ha yr⁻¹ or Tg AGC state yr⁻¹). Specifically, we developed spatially explicit estimates of annual tree mortality from fires and bark beetles across regional forestland building off of a remote sensing framework from an earlier study (Hicke *et al* 2013). In addition to the remote sensing analysis, we estimated mean annual tree mortality from timber harvest for each state using harvest statistics from the US Forest Service (USFS; Smith *et al* 2009, Oswalt *et al* 2014). The USFS recently recommended that metrics related to fire and insect effects be used to track national climate change impacts (Heath *et al* 2015), further underscoring the importance of quantifying the magnitude and regional variation in tree mortality from these types of disturbance.

2. Methods

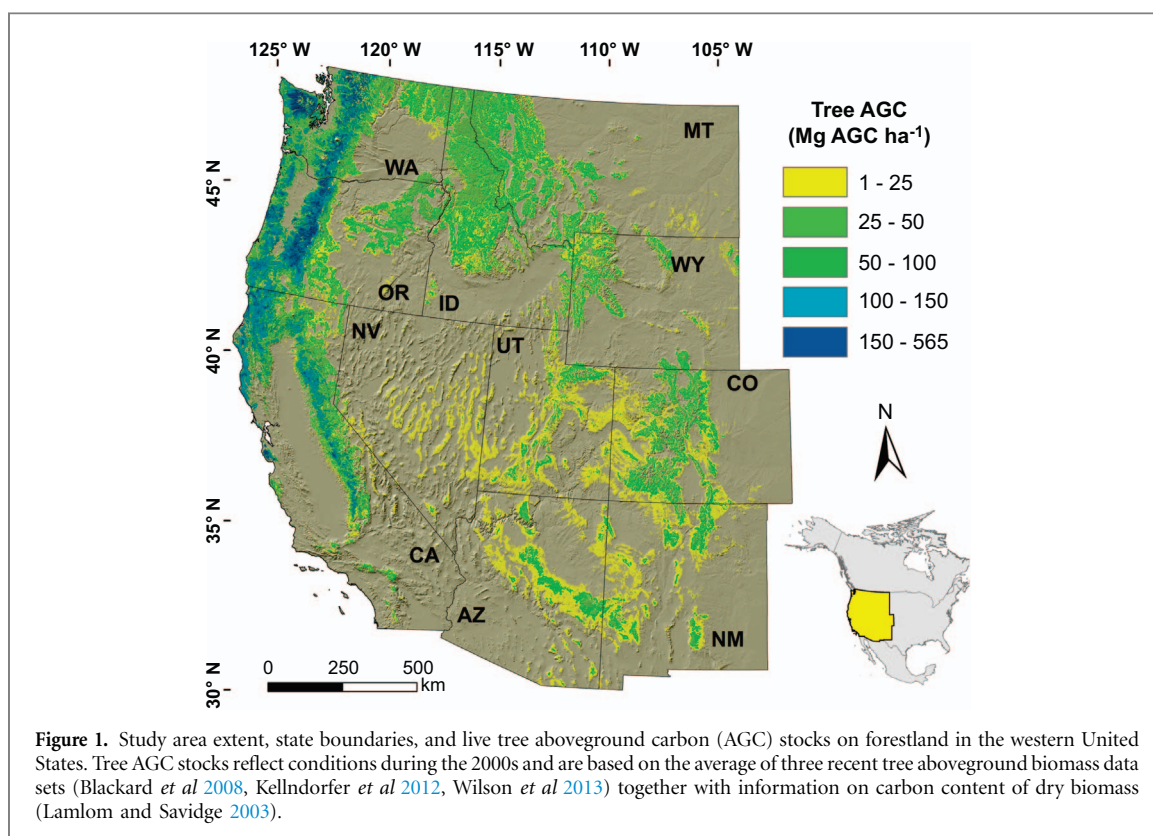
2.1. Geospatial data sets and preprocessing

2.1.1. General information

We quantified tree mortality from fires (MORT_{fire}, Mg AGC ha yr⁻¹) and bark beetles (MORT_{beetle}, Mg AGC ha yr⁻¹) from 2003–2012 across 802 575 km² of forestland located in the western US (figure 1, 31.3°N–49.0°N, 102.0°W–124.5°W). Forestlands were included if consistently mapped as forest by three separate tree AGB data sets (described below). We conducted the analysis on a 1 km resolution grid in an Albers Equal Area projection, with the resolution chosen to match the bark beetle data set (Meddens *et al* 2012). We analyzed and visualized the data using ArcGIS 10 (ESRI, Redlands, CA) and R statistical software (version 3.2; R Core Team 2015), along with the R packages *raster* (Hijmans and van Etten 2013) and *dplyr* (Wickham and Francois 2015). The data preprocessing work flow described in the following section is illustrated in figure 2.

2.1.2. Tree carbon stocks

We estimated tree AGC using three tree AGB geospatial data sets together with information on the fraction of dry biomass that is carbon (f_{carbon}). The tree AGB data sets were produced by spatially interpolating USFS inventory measurements (FIA)



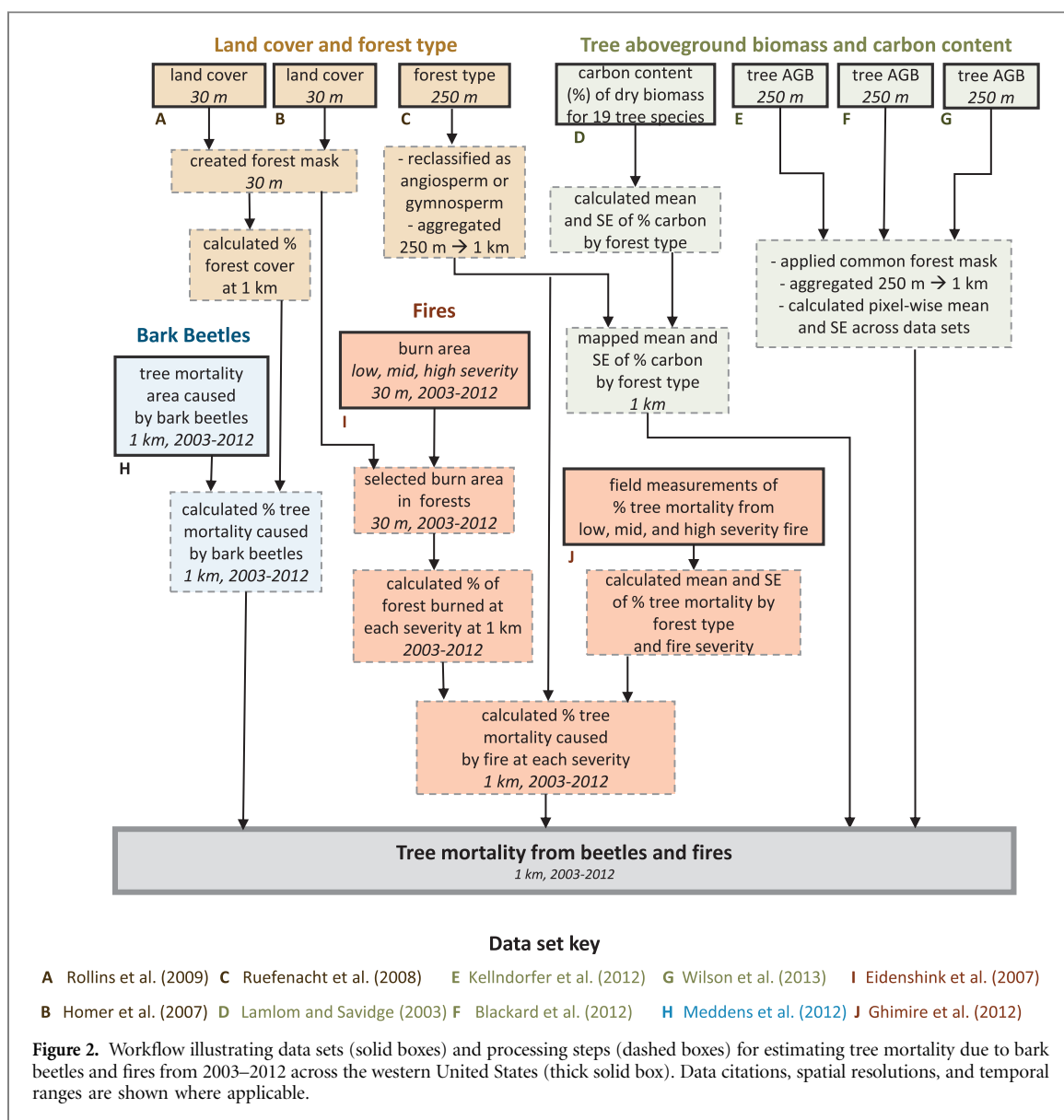
using satellite and geophysical data sets in conjunction with machine learning algorithms (Blackard *et al* 2008, Kellndorfer *et al* 2012, Wilson *et al* 2013). Wilson *et al* (2013) provided estimates of tree AGC assuming that f_{carbon} was 50%; we converted these estimates back to tree AGB by multiplying by 2. We acquired each data set at 250 m spatial resolution, reprojected it to an Albers Equal Area projection, and then applied a common mask that identified pixels consistently classified as forest among all data sets. Next, we aggregated from 250 m to 1 km resolution by computing total tree AGB in each 1 km pixel (i.e. Mg AGB pixel⁻¹).

We quantify tree mortality from 2003–2012 and required estimates of tree AGB prior to disturbance. Two of these data sets represented tree AGB circa 2000 (Blackard *et al* 2008, Kellndorfer *et al* 2012), whereas the third depicted tree AGB circa 2003–2008 (Wilson *et al* 2013). Therefore we needed to estimate pre-disturbance tree AGB in pixels that were disturbed during these six years in the Wilson *et al* (2013) data set. We followed an existing approach (Hicke *et al* 2013) that involved comparing tree AGB in fire and beetles disturbed pixels against the average tree AGB of undisturbed pixels from the same forest type (Ruefenacht *et al* 2008) and ecoregion (Omernik 1987). If tree AGB in the disturbed pixel was less than the average tree AGB of undisturbed pixels, then we set tree AGB in the disturbed pixel to this average, but otherwise left the disturbed pixel unaltered. After implementing this correction, we then computed the average and standard error (SE) of tree AGB for each

pixel over the three data sets ($\overline{\text{AGB}}$, SE_{AGB}). Lastly, we estimated the average and SE of f_{carbon} ($\overline{f_{\text{carbon}}}$, $\text{SE}_{\text{carbon}}$) for hardwood (angiosperm, $48.49 \pm 0.42\%$ C, $n = 8$) and softwood (gymnosperm, $50.87 \pm 0.63\%$ C, $n = 11$) tree species found in this region (Lamloom and Savidge 2003).

2.1.3. Forest fires

We used annual maps of fire severity from 2003–2012 produced by the Monitoring Trends in Burn Severity (MTBS) project (Eidenshink *et al* 2007). These maps were derived from 30 m resolution Landsat images acquired before and after fires larger than ~ 405 ha in the western US. Fire severity was mapped using the differenced Normalized Burn Ratio (dNBR) and then each pixel was classified as one of five thematic severity classes. We focused on pixels that burned at low-, moderate-, or high-severity and excluded those that were unburned or showed post-fire greening. We identified fire in forests by generating a 30 m resolution forest mask based on forestland in either of two national land cover maps (Homer *et al* 2007, Rollins 2009). We then computed the annual fraction of forest area that burned at each severity (A_s) in each 1 km pixel over the decade. Lastly, we incorporated field measurements of the fraction of tree biomass killed at each severity (f_{fire}). We computed the mean and SE of f_{fire} ($\overline{f_{\text{fire}}}$, SE_{fire}) for forests dominated by hardwood and softwood tree species based on 116 estimates of f_{fire} synthesized from 29 field studies conducted in the western United States (table S1 available at stacks.iop.org/ERL/12/065005/mmedia, Ghimire *et al* 2012).



2.1.4. Bark beetles

We used annual maps depicting the areal extent of tree canopy mortality due to 12 bark beetle species that were produced from aerial surveys, forest inventory measurements, and high-resolution satellite imagery (Meddens *et al* 2012). The USFS National Forest Health Monitoring program conducts aerial detection surveys (ADS) that provide a coarse snapshot of insect and other forest disturbance at a landscape scale (Johnson and Wittwer 2008). Trained observers conduct surveys from fixed-wing aircraft that involve sketch-mapping the extent of areas affected by insect outbreaks and then visually estimating the number of dead trees within affected areas. Meddens *et al* (2012) used the ADS observations to estimate annual canopy mortality area due to bark beetles across the western US from 1997–2010, with estimates then extended through 2012 (Hicke *et al* 2015).

Canopy mortality area was estimated at 1 km resolution based on the number of recently killed trees and the average canopy area of each tree species. The

ADS observations underestimated the number of trees killed by bark beetles in comparison with field observations and QuickBird (2.4 m resolution) satellite imagery from Colorado, Idaho, and New Mexico. This led to the development of adjustment factors for several forest types based on the ratio of dead trees mapped from satellite to ADS tree kill counts. These adjustment factors were then used to modify ADS tree kill counts, resulting in high and middle (most realistic) estimates as well as low estimates based on unaltered ADS numbers (more details in: Meddens *et al* 2012, Hicke *et al* 2015). The data set includes bark beetle species that cumulatively killed at least 100 000 trees across the domain from 1997–2010. Specifically, it includes (from greatest to least mortality area) mountain pine beetle (*Dendroctonus ponderosae* Hopkins), piñon ips (*Ips confusus* (LeConte)), Douglas-fir beetle (*Dendroctonus pseudotsugae* Hopkins), western balsam bark beetle (*Dryocoetes confusus* (Swaine)), and fir engraver (*Scolytus ventralis* LeConte), as well as seven additional beetle species

that killed fewer trees. We masked these data to our domain and then summed canopy mortality area across bark beetle species for each 1 km pixel in a given year (i.e. a voxel) from 2003–2012. We then calculated annual tree canopy mortality fraction (f_{beetle}) for each voxel by dividing canopy mortality area by forest cover fraction. We incorporated the lower, middle, and upper estimates of f_{beetle} into our analysis.

2.2. Tree mortality from bark beetles and fires

We developed spatially explicit estimates of annual $\text{MORT}_{\text{beetle}}$ and $\text{MORT}_{\text{fire}}$ from 2003–2012 and quantified uncertainty in these estimates using a Monte Carlo approach (e.g. Harmon *et al* 2007, Gonzalez *et al* 2015). The Monte Carlo involved generating 100 realizations of annual tree mortality in each voxel, where each realization iteratively varied tree $\overline{\text{AGC}}$, $\bar{f}_{\text{carbon}}$, f_{beetle} , and $\bar{f}_{\text{fire}}$ based on uncertainty in each term. We assumed that tree AGC within a pixel only changed due to disturbance (i.e. no tree growth or recruitment), which potentially caused us to slightly underestimate tree mortality. For instance, stand age averaged 97 ± 73 years ($\pm 1\text{SD}$) in our study area (Pan *et al* 2011b). Field measurements from the Western Cascades (Hudiburg *et al* 2009) and the Rocky Mountains showed that tree AGC could increase 6%–7% between stands that are 97 and 106 years old. This simplifying assumption made it so that cumulative tree mortality within a pixel could not exceed the initial tree AGC. For each realization r we first computed tree AGC for pixel p of year $t = 2003$ as

$$\overline{\text{AGC}}_{r,p,t} = (\overline{\text{AGB}}_{p,t} + a_{\text{AGB},r} \times \text{SE}_{\text{AGB},p}) \times (\bar{f}_{\text{carbon}} + a_{\text{carbon},r} \times \text{SE}_{\text{carbon}}) \quad (1)$$

where $\bar{f}_{\text{carbon}}$ and $\text{SE}_{\text{carbon}}$ varied by forest type and a_x was a random number from a normal distribution (mean = 0, SD = 1) that differed for each variable with each realization. The other variables are defined above. We then calculated $\text{MORT}_{\text{beetle}}$ as

$$\text{MORT}_{\text{beetle},r,p,t} = \overline{\text{AGC}}_{r,p,t} \times \hat{f}_{\text{beetle},p,t} \quad (2)$$

where $\hat{f}_{\text{beetle},p,t}$ was an estimate of f_{beetle} drawn randomly from a triangular distribution defined uniquely for each voxel using the low, middle, and high estimates of f_{beetle} . Each distribution was fit using the *triangle* package in R (Carnell 2016). Next, we computed $\text{MORT}_{\text{fire}}$ as

$$\text{MORT}_{\text{fire},r,p,t} = (\overline{\text{AGC}}_{r,p,t} - \text{MORT}_{\text{beetle},r,p,t}) \times \sum_s^{\text{severity classes}} [A_{p,t,s} \times (\bar{f}_{\text{fire},s} + a_{\text{fire},s,r} \times \text{SE}_{\text{fire},s})] \quad (3)$$

We then reduced tree AGC stocks at the start of year $t + 1$ to account for mortality in year t , such that

$$\overline{\text{AGC}}_{r,p,t+1} = \overline{\text{AGC}}_{r,p,t} - (\text{MORT}_{\text{beetle},r,p,t} + \text{MORT}_{\text{fire},r,p,t}) \quad (4)$$

We repeated equations 2–4 annually through 2012 and then repeated the entire process 100 times.

We then computed the mean and standard deviation (SD) of these multiple estimates of tree mortality for each voxel, where the SD represented uncertainty due to parameter error (e.g. SE_{AGB} , SE_{fire}). Altogether, this process yielded spatially explicit estimates of annual tree mortality (and uncertainty) caused by fires and bark beetles from 2003–2012 across forestland in the western United States.

Following the geospatial analysis, we estimated mean annual tree mortality from fires ($\overline{\text{MORT}}_{\text{fire}}$) and bark beetles ($\overline{\text{MORT}}_{\text{beetle}}$) for each state from 2003–2012 and used a Monte Carlo analysis to propagate uncertainty in these estimates associated with parameter error and temporal variability. As illustrated for fires (identical for beetles), we first derived 1000 realizations r of annual tree mortality in each state from $t = 2003$ to 2012, as per

$$\text{MORT}_{\text{fire},r,\text{state},t} = \sum_{p=1}^{\text{N pixels}} (\text{MORT}_{\text{fire},p,t} + a_r \times \text{SD}_{\text{MORT}_{\text{fire},p,t}}) \quad (5)$$

and then summarized (mean and SD) each set of realizations. We used these annual statewide summaries to then repeatedly ($N_r = 1000$) estimate both the mean and SE of annual mortality from 2003–2012 for each state, where each realization randomly varied annual statewide mortality by its parameter error (i.e. SD computed above). We again summarized (mean and SD) each set of realizations for both statistics. This yielded an estimate of $\overline{\text{MORT}}_{\text{fire}}$ for each state and produced estimates of uncertainty in $\overline{\text{MORT}}_{\text{fire}}$ due to parameter error (i.e. SD of estimates of $\overline{\text{MORT}}_{\text{fire}}$) and temporal variability (i.e. mean of estimates of SE of annual mortality).

In the final step, we derived a 95% confidence interval (CI) around each estimate of $\overline{\text{MORT}}_{\text{fire}}$ that accounted for both parameter error and temporal variability. This involved repeatedly estimating $\overline{\text{MORT}}_{\text{fire}}$ for each state, where each realization randomly varied $\overline{\text{MORT}}_{\text{fire}}$ by uncertainty associated with both parameter error and temporal variability. We then computed the 95% CI (Gonzalez *et al* 2015) for each state, as per

$$95\% \text{ CI} = \frac{\overline{\text{MORT}}_{\text{fire}}^{97.5} - \overline{\text{MORT}}_{\text{fire}}^{2.5}}{2} \quad (6)$$

where $\overline{\text{MORT}}_{\text{fire}}^{97.5}$ and $\overline{\text{MORT}}_{\text{fire}}^{2.5}$ were the 97.5th and 2.5th percentiles of 1000 realizations of $\overline{\text{MORT}}_{\text{fire}}$. Overall, this approach yielded estimates of mean annual tree mortality ($\pm 95\%$ CI) due to fires and bark

beetles from 2003–2012 for each state in the western US. We present these estimates both in terms of absolute tree mortality (Tg AGC yr^{-1}) and mortality rate (i.e. % of statewide $\overline{\text{AGC}}$ in tree biomass killed each year).

2.3. Tree mortality from timber harvest

We estimated mean annual tree mortality caused by timber harvest from 2003–2012 for each state ($\overline{\text{MORT}}_{\text{harvest}}$, Tg AGC yr^{-1}) using timber product output surveys from several USFS reports along with information on tree characteristics. We again used a Monte Carlo analysis to propagate uncertainty in $\overline{\text{MORT}}_{\text{harvest}}$ that was associated with parameter error and temporal variability. The USFS reported harvest for each state in terms of mean annual growing stock volume (GSV, $\text{m}^3 \text{yr}^{-1}$) removed from c. 2003 to 2007 (Smith *et al* 2009) and c. 2008 to 2012 (Oswalt *et al* 2014). Harvest was reported separately for hardwood and softwood tree species. We calculated mean annual GSV removed across these two periods for both species groups in each state ($\overline{\text{GSV}}$). In each case, we then estimated $\overline{\text{MORT}}_{\text{harvest}}$ from $\overline{\text{GSV}}$ using information on the ratio of total aboveground tree volume to GSV(R), wood density (WD , Mg m^{-3}), and the fraction of dry woody biomass that is carbon (f_{carbon}), as per

$$\overline{\text{MORT}}_{\text{harvest}} = \overline{\text{GSV}} \times \overline{R} \times \overline{WD} \times \overline{f} \quad (7)$$

We repeatedly ($N_r = 1000$) estimated $\overline{\text{MORT}}_{\text{harvest}}$ for both species groups in each state as part of the Monte Carlo analysis, which again involved randomly varying each parameter by its uncertainty (e.g. as shown in equation (1)). It was not possible to directly compute the uncertainty (SE) associated with temporal variability in the amount of GSV annually removed since annual harvest data were not available for each state. We therefore assumed that the SE was 4.6% of $\overline{\text{GSV}}$ based concurrent annual harvest data from Oregon, which accounted for 35% regional harvest (Oregon Department of Forestry 2017). We used estimates of average R ($\overline{R}$) that varied for hardwoods and softwoods, as well as among states in different USFS regions (Birdsey 1992), assuming in each case that the SE was 2% of $\overline{R}$ (Levy *et al* 2004). We computed average WD ($\overline{WD}$) and its uncertainty (SE_{WD}) for both species groups in each state by (1) identifying the four tree species in each group that accounted for the largest percentages of total tree AGB on USFS inventory plots surveyed 2003–2012 (Smith 2002) and then (2) summarizing WD measurements among these species (Miles and Smith 2009). After repeatedly estimating $\overline{\text{MORT}}_{\text{harvest}}$, we then derived the average and associated 95% CI from each set of 1000 realizations (e.g. equation (6)). We report $\overline{\text{MORT}}_{\text{harvest}}$ ($\pm 95\%$ CI) for each state both in terms of absolute mortality (Tg AGC yr^{-1}) and mortality rate ($\% \text{yr}^{-1}$).

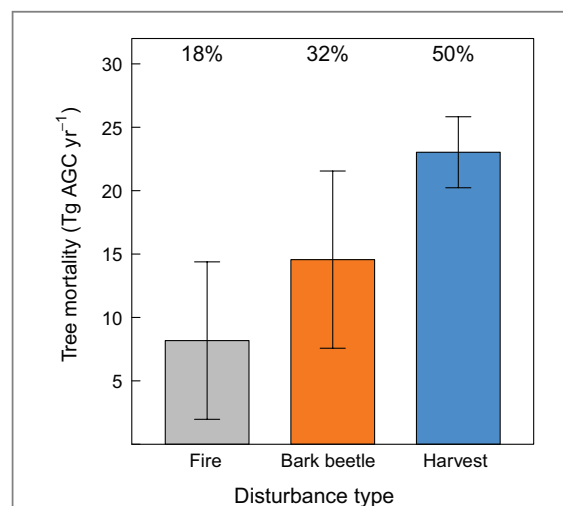


Figure 3. Regional mean annual tree mortality from fires, bark beetles, and timber harvest from 2003–2012 on forestland in the western United States. Tree mortality was quantified as the amount of aboveground carbon (AGC) stored in tree biomass killed by disturbance (Tg AGC yr^{-1}). Tree mortality from bark beetles and fires was derived using remote sensing, whereas harvest was derived from USFS forest resource assessments (Smith *et al* 2009, Oswalt *et al* 2014). Error bars depict 95% confidence intervals computed with a Monte Carlo analysis and incorporate temporal variability and parameter error. We estimated that regional mean annual tree mortality from fires, beetles, and harvest was $45.8 \pm 16.0 \text{ Tg AGC yr}^{-1}$ from 2003–2012, with fires, beetles, and harvest accounting for 18%, 32%, and 50% of annual mortality, respectively.

3. Results

3.1. Regional tree mortality from disturbance

Taken together, mean annual tree mortality from timber harvest, bark beetles, and fires ($\overline{\text{MORT}}_{\text{H+B+F}}$) was $45.8 \pm 16.0 \text{ Tg AGC yr}^{-1}$ from 2003–2012 across the western US. Regional tree $\overline{\text{AGC}}$ stocks totaled $4.16 \pm 0.12 \text{ Pg}$, suggesting that the tree mortality rate was $1.10 \pm 0.38\% \text{yr}^{-1}$. Timber harvest accounted for the largest percentage of $\overline{\text{MORT}}_{\text{H+B+F}}$ (50%), followed by bark beetles (32%) and then fires (18%, figure 3, table S2).

3.2. Tree mortality from fire

Mean annual tree mortality from fires ($\overline{\text{MORT}}_{\text{fire}}$) was $8.2 \pm 6.2 \text{ Tg AGC yr}^{-1}$ and the mortality rate was $0.20 \pm 0.15\% \text{yr}^{-1}$ from 2003–2012 in the western US (figures 3, 4(a), 5, table S2). Absolute $\overline{\text{MORT}}_{\text{fire}}$ was highest in northern California, central Idaho, and western Montana, with these states accounting for 64% of regional $\overline{\text{MORT}}_{\text{fire}}$. Forests in Arizona and New Mexico experienced the highest annual rates of $\overline{\text{MORT}}_{\text{fire}}$ ($0.36\%–0.57\% \text{yr}^{-1}$), as well as the highest percentage of $\overline{\text{MORT}}_{\text{H+B+F}}$ caused by fire (51%–55%). Conversely, forests in Colorado, Oregon, and Washington had the lowest rates of $\overline{\text{MORT}}_{\text{fire}}$ ($0.03\%–0.08\% \text{yr}^{-1}$) and the lowest percentage of $\overline{\text{MORT}}_{\text{H+B+F}}$ caused by fire (2%–8%). Fires that occurred in Oregon and Washington were largely concentrated along the eastern slopes of the Cascade Range.

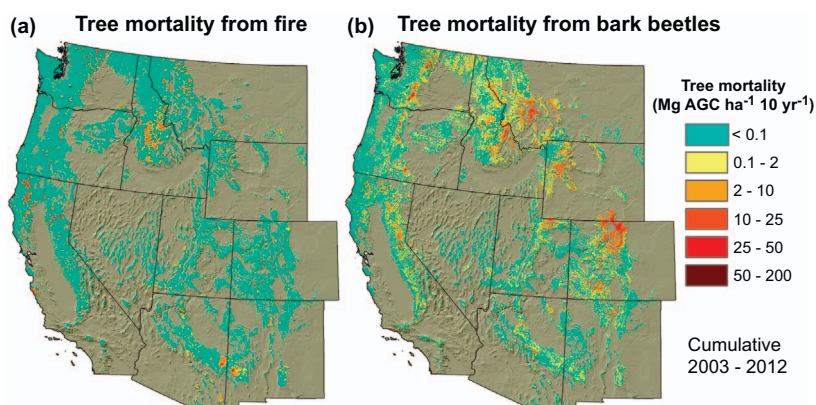


Figure 4. Cumulative tree mortality due to (a) fires and (b) bark beetles from 2003–2012 on forestland in the western United States. Tree mortality was quantified as the amount of aboveground carbon (AGC) stored in tree biomass killed by disturbance (e.g. Mg AGC ha⁻¹). Note that (a) tree mortality from fire was concentrated in northern California, central Idaho, and southern Arizona/New Mexico, whereas (b) tree mortality from bark beetles occurred widely at low levels, but was highest in northern Colorado and western Montana.

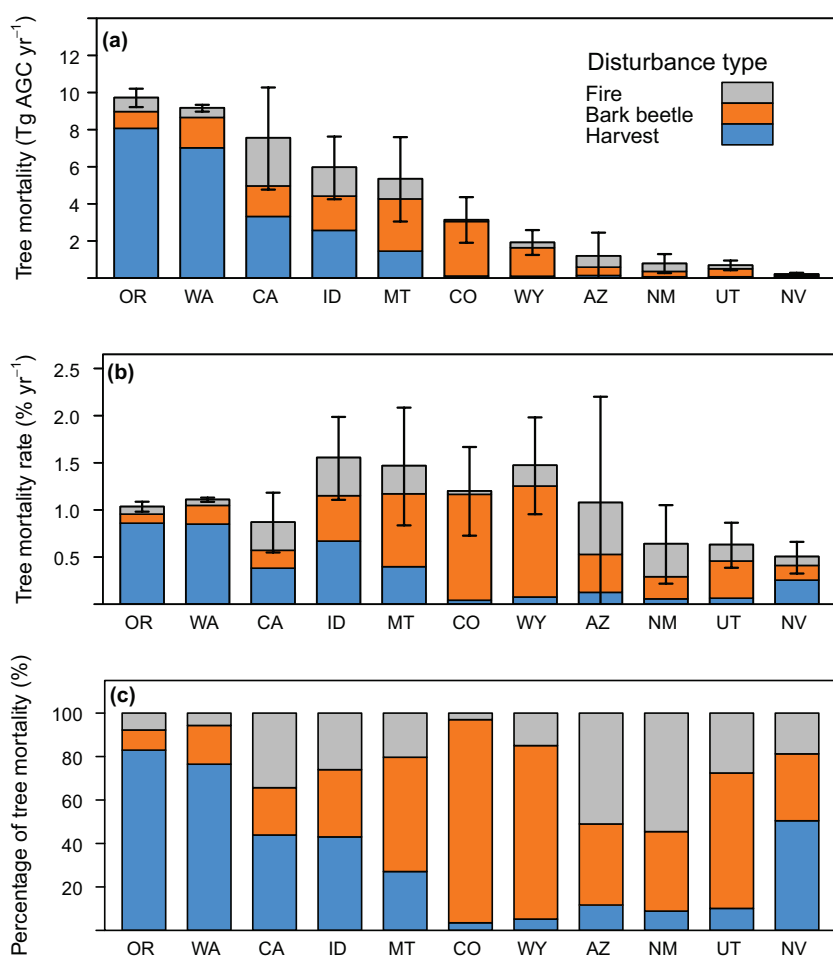


Figure 5. Mean annual tree mortality from fires, bark beetles, and timber harvest on forestland from 2003–2012 for each state in the western US. Tree mortality was quantified as the amount of aboveground carbon (AGC) stored in tree biomass killed by disturbance. Mean annual tree mortality is summarized in terms of (a) absolute mortality (Tg AGC yr⁻¹), (b) mortality rate (percentage of statewide AGC in tree biomass killed per year, % yr⁻¹), and (c) percentage of mortality caused by each type of disturbance (%). Tree mortality from bark beetles and fires was derived using remote sensing, whereas harvest was derived from USFS forest resource assessments. Error bars depict 95% confidence intervals that were computed using a Monte Carlo analysis and incorporate temporal variability and parameter error from all three types of disturbances.

3.3. Tree mortality from bark beetles

Mean annual tree mortality from bark beetles ($\overline{\text{MORT}}_{\text{beetle}}$) was 14.6 ± 7.0 Tg AGC yr^{-1} and the mortality rate was $0.35 \pm 0.17\%$ yr^{-1} from 2003–2012 in the western US (figures 3, 4(b), 5, table S2). Absolute $\overline{\text{MORT}}_{\text{beetle}}$ was highest in northern Colorado, western Montana, and central Idaho, with these states accounting for 52% of regional $\overline{\text{MORT}}_{\text{beetle}}$ (table S2). Forests in Colorado and Wyoming had the highest annual rates of $\overline{\text{MORT}}_{\text{beetle}}$ (1.12% – 1.22% yr^{-1}) and the highest percentage of $\overline{\text{MORT}}_{\text{H+B+F}}$ caused by bark beetles (80%–93%). Conversely, Oregon and Washington had not only two of the lowest rates of $\overline{\text{MORT}}_{\text{fire}}$, but also two of the lowest rates of $\overline{\text{MORT}}_{\text{beetle}}$ (0.10% – 0.20% yr^{-1}).

3.4. Tree mortality from timber harvest

Mean annual tree mortality from timber harvest ($\overline{\text{MORT}}_{\text{harvest}}$) was 23.0 ± 2.8 Tg AGC yr^{-1} and the mortality rate was $0.55 \pm 0.07\%$ yr^{-1} from 2003–2012 in the western US (figures 3, 5, table S2). Timber harvest in Oregon and Washington accounted for 67% of regional $\overline{\text{MORT}}_{\text{harvest}}$. These two states had the highest rates of $\overline{\text{MORT}}_{\text{harvest}}$ (0.85% – 0.86% yr^{-1}) and the highest percentage of $\overline{\text{MORT}}_{\text{H+B+F}}$ caused by harvest (76%–83%). Conversely, forest in Colorado, Utah, and New Mexico had the lowest rates of $\overline{\text{MORT}}_{\text{harvest}}$ (0.04% – 0.06% yr^{-1}) and among the lowest percentage of $\overline{\text{MORT}}_{\text{H+B+F}}$ caused by harvest (3%–10%).

4. Discussion

4.1. Regional tree mortality from disturbance

In this study, we quantified tree mortality caused by timber harvest, bark beetles, and fire from 2003–2012 across the western US. We found that regional mean annual tree mortality from timber harvest (~ 23 Tg AGC yr^{-1}) was quite similar in magnitude to the mortality from fires and bark beetles combined (~ 22 Tg AGC yr^{-1}). Regional timber harvest declined about 40% since the 1980s (Oswalt *et al* 2014), yet harvest still caused significantly more tree mortality than bark beetles or fires during the 2000s, a period during which hot and dry conditions contributed to extensive beetle and fire activity relative to the last several decades (Williams *et al* 2012, Hicke *et al* 2015, Meddens *et al* 2015, Abatzoglou and Williams 2016, Cohen *et al* 2016).

Our regional tree mortality estimates were comparable with several other studies from the western US. For instance, forest inventories indicated that the regional tree mortality rate was $\sim 0.72\%$ – 0.92% yr^{-1} during the 2000s in areas unaffected by harvest or land clearing (Oswalt *et al* 2014). By comparison, we estimated that the tree mortality rate from fires and bark beetles combined was $\sim 0.56\%$ yr^{-1} , suggesting that these disturbances together might have accounted

for $\sim 60\%$ – 80% of regional tree mortality not associated timber harvest or land clearing. Our estimates of tree mortality from fires and bark beetles were also of similar magnitude to those predicted using the CASA (Carnegie–Ames–Stanford Approach) carbon cycle model with fire (Ghimire *et al* 2012) and beetle (Ghimire *et al* 2015) disturbances prescribed using MTBS and ADS data sets, respectively. Lastly, our estimates of regional $\text{MORT}_{\text{H+B+F}}$ in 2003 and 2004 differed by $\pm 5\%$ from estimates of tree mortality due to all forms (undifferentiated) of disturbance that were derived using a regional sample of Landsat scenes from 1986–2004 (Powell *et al* 2014). Similarity among these field, modeling, and remote sensing estimates of regional tree mortality is encouraging. Together these comparisons suggest that timber harvest, bark beetles, and fires were the primary causes of tree mortality from disturbance during the 2000s in the western US.

Our current estimates of cumulative (2003–2012) regional tree mortality from fires and bark beetles were both $\sim 40\%$ lower than earlier best-estimates reported by Hicke *et al* (2013). This downward revision stemmed from refinements in representation of tree AGB, more realistic parameterization of tree mortality from fire, and a routine that tracked prior tree mortality within each grid cell. Estimating tree mortality in terms of carbon necessitated accurate representation of tree AGB; however, estimates of tree AGB differ among available data sets, particularly at the pixel scale (figure S1; Huang *et al* 2015, Neeti and Kennedy 2016). We therefore chose to incorporate an ensemble of available data sets (Blackard *et al* 2008, Kellndorfer *et al* 2012, Wilson *et al* 2013) rather than rely on a single data set as was done in the prior study (Hicke *et al* 2013). This improvement accounted for $\sim 10\%$ of the reduction in tree mortality between studies and allowed us to propagate uncertainty in tree AGB into our estimates of mortality. Our current analysis also used a regional synthesis of field measurements (Ghimire *et al* 2012) to vary tree mortality fraction by forest type and fire severity rather than assume complete tree mortality in areas that burned at moderate or high severity (Hicke *et al* 2013). This further reduced our estimates of tree mortality from fire. Lastly, we found that 39% of the forested grid cells (1 km^2) registered multiple disturbances during this decade. We therefore implemented a routine that reduced tree AGB each time a disturbance occurred, which reduced the amount of tree AGB killed by subsequent disturbance. These improvements enabled more robust estimation of how much carbon was stored in trees recently killed by fires and beetles across the western US.

The carbon in trees killed by disturbance will be emitted to the atmosphere as dead trees decompose in the forest and as harvested trees are processed into wood products with varying efficiency and longevity (Harmon *et al* 1990, Harmon *et al* 2011, Hudiburg *et al* 2011). Ignoring pyrogenic emissions, mean

annual tree mortality from fire and beetles could together lead to emission of $\sim 0.8 \text{ Tg C yr}^{-1}$ during the following decade assuming exponential decay and an intermediate rate constant ($k = 0.04$; Harmon *et al* 1986). Actual decomposition rates will depend on the interplay among snag fall rate, microclimate, wood chemistry, tree anatomy, and other factors (Weedon *et al* 2009, Harmon *et al* 2011). Similarly, mean annual tree mortality from harvest could lead to emission of around 1.3 Tg C yr^{-1} during the following decade assuming that (1) 60% of harvested material went to fuels, paper, and residues that lasted < 5 years (Harmon *et al* 1990) and that (2) the remaining 40% of material went to longer term storage that remained unchanged during the first decade. In other words, tree mortality from harvest, bark beetles, and fires averaged $45.8 \pm 16.0 \text{ Tg AGC yr}^{-1}$, yet annual emissions are potentially closer to 2 Tg C yr^{-1} during the first decade following disturbance and will persist for decades to centuries. By comparison, regional fossil fuel emissions averaged $\sim 260 \text{ Tg C yr}^{-1}$ during this period (US Energy Information Administration 2015). More accurate assessment of the timing and magnitude of carbon emissions requires using robust ecosystem models and life cycle assessments (e.g. Hudiburg *et al* 2011, Ghimire *et al* 2015).

4.2. Tree mortality from fire

Tree mortality from fire was highest in the Southwest, with regional variation in mortality associated with differences in drought severity and human activities. States with more consistently severe summer drought tended to have the highest average tree mortality rate from fires (e.g. Arizona), whereas states that experienced less consistently severe drought tended to have lower mortality from fires (e.g. Oregon, figures S2, S3). Hot and dry conditions increase fuel flammability and fire occurrence among forests in the western US (Littell *et al* 2009, Morton *et al* 2013, Abatzoglou and Williams 2016). Furthermore, drought-stricken trees are more likely to die if a fire does occur (van Mantgem *et al* 2013), potentially because of more extensive xylem cavitation and hydraulic impairment during a fire (Michaletz *et al* 2012, van Mantgem *et al* 2013).

Human activities, including fire ignition and suppression, also affected regional variation in fire activity. Human fire ignitions were responsible for $\sim 32\%$ of the total area burned in the western US from 1992–2012 and were concentrated among the more densely populated coastal states (Balch *et al* 2017). Humans were the primary cause of fire ignitions in California (Balch *et al* 2017), where drought-affected forests experienced the highest tree mortality from fire of any western state ($\sim 2.6 \text{ Tg AGC yr}^{-1}$). The federal government recently spent $\sim \$400$ million per year on fire suppression in California, which accounted for about half of the federal expenditure on fire suppression in the western US (Kenward and Raja 2013). Nevertheless, plant mortality from fire averaged

$\sim 4.3 \text{ Tg AGC yr}^{-1}$ across all of California's ecosystems from 2001–2010, which led to an estimated 8% net reduction in statewide plant AGC and complicated efforts to reduce GHG emissions (Gonzalez *et al* 2015).

It is widely perceived in California and other parts of the region that beetle-killed trees increase subsequent risk of fire (Heller 2017), yet recent studies indicate that beetle outbreaks have not increased burn area (Hart *et al* 2015, Meigs *et al* 2015a) or fire severity in the region (Bond *et al* 2009, Meigs *et al* 2016, Reilly and Spies 2016). Similarly, we found no association between statewide tree mortality rates from fires and beetles ($P = 0.88$), though lagged effects and the spatial scale of this comparison could mask potential interactions.

The MTBS data set provides a valuable record of fire extent and severity across the nation (Eidsenink *et al* 2007), though additional efforts are needed to better quantify fire severity (Kolden *et al* 2015). It is also important to better understand how mortality differs among co-occurring trees species with different functional traits (e.g. bark thickness; Pausas 2015). Projected shifts towards a hotter, drier climate could potentially increase fire activity in the Southwest and other dry parts of the region over the coming century (Spracklen *et al* 2009, Williams *et al* 2014), contributing to increases in tree mortality, cost of wildfire management, and GHG emissions.

4.3. Tree mortality from bark beetles

Tree mortality from bark beetles was pervasive at low levels across the region, though was pronounced in the Rocky Mountains (e.g. Colorado) where beetles accounted for $\sim 80\%$ of $\overline{\text{MORT}}_{\text{H+B+F}}$. Mountain pine beetles were responsible for $\sim 62\%$ of tree mortality area (5.37 Mha) caused by bark beetles from 1997–2010 in the western US and largely affected lodgepole pine (Meddens *et al* 2012, Hicke *et al* 2015). In this region, lodgepole pine occur most extensively in the Rocky Mountains, where stands were generally of an age (80–120 years) and stem density ($> 500 \text{ stems ha}^{-1}$) that rendered them more susceptible to beetle attack than lodgepole pine in the coastal states, which tended to be younger and less widespread (Hicke and Jenkins 2008). Forests in the Rocky Mountains also experienced long-term warming that increased beetle populations by reducing beetle mortality during winter and accelerating beetle development (Logan and Powell 2001, Creeden *et al* 2014). Furthermore, states with the most severe single-year summer drought tended to have the highest average tree mortality rate from beetles (e.g. Colorado, Wyoming), whereas states that experienced less acute drought tended to have lower mortality from beetles (e.g. Oregon, figure S4). High tree mortality from beetles in the Rocky Mountains has been linked to acute drought that weakened tree defense against beetle attack (Creeden *et al* 2014, Hart *et al* 2014) in stands already subject to

high structural susceptibility and larger beetle populations (Logan and Powell 2001, Hicke and Jenkins 2008).

Lower tree mortality from beetles in other parts of the region could reflect less severe drought (Abatzoglou and Redmond 2017), lower stand susceptibility (Hicke and Jenkins 2008), and different host and beetle species. Low insect-caused mortality among wet coastal forests (Meddens *et al* 2012, Meigs *et al* 2015b, Reilly and Spies 2016) might be linked to higher tree species diversity (Waring *et al* 2006) limiting host availability and population size of host-specific beetle species (Anderegg *et al* 2015). The coastal forests also tend to be younger than forests in the Rocky Mountains (Pan *et al* 2011b) due to more extensive timber harvest (Masek *et al* 2011), which might reduce the likelihood of beetle attack since beetles tend to prefer large old trees (Raffa *et al* 2008). On the other hand, land-use activities that homogenize forest structure and composition can increase the likelihood of beetle outbreaks (Raffa *et al* 2008). Future efforts are needed to reduced uncertainty in the causes, extent, and severity of bark beetle outbreaks (Meddens *et al* 2012, Gartner *et al* 2015) given that rising temperatures and local land use activities could amplify tree mortality from bark beetles over the coming century in parts of this region (Hicke *et al* 2006, Raffa *et al* 2008).

4.4. Tree mortality from harvest

Tree mortality from timber harvest was highest in Oregon and Washington and accounted for $\sim 80\%$ of $\overline{\text{MORT}}_{\text{H+B+F}}$ in these states. Much of the timber harvest in these states occurs in highly productive coastal forests, where rates of timber extraction per unit of forest area are the highest in the country (Masek *et al* 2011, Law and Waring 2015). These coastal forests were a net source of carbon to the atmosphere during the 1980s due to high rates of timber harvest, though declines in harvest following implementation of the Northwest Forest Plan in 1994 led forests on public lands to become a net carbon sink during the 2000s (Turner *et al* 2011). Forests in Oregon and northern California currently store ~ 3.2 Pg C, yet could theoretically store ~ 5.9 Pg C if stand-replacing disturbances did not occur on the landscape (Hudiburg *et al* 2009). Recent tree mortality from timber harvest far exceeded tree mortality caused by both bark beetles and fires in the Pacific Northwest, highlighting that reductions in timber harvest could help these states meet GHG emission reduction targets. Continued warming and drying could reduce the amount of tree biomass available to be harvested in the Interior West over the coming century (Williams *et al* 2012, Jiang *et al* 2013, Berner *et al* 2017), yet it is currently unclear how the net effects of ongoing climate change will affect tree biomass and resource availability in the Pacific Northwest (Hudiburg *et al* 2013, Jiang *et al* 2013, Kang *et al* 2014, McDowell *et al* 2015).

5. Conclusions

Timber harvest, bark beetles, and fires were important causes of tree mortality from 2003–2012 across forests in the western US. This was a period during which high temperatures contributed to severe drought that increased both fire and bark beetle activity relative to recent decades. Regional tree mortality from these disturbances together averaged 45.8 ± 16.0 Tg AGC yr^{-1} , with harvest accounting for a significantly larger percentage (50%) than bark beetles (32%) or fires (18%). The amount of carbon in tree aboveground biomass killed each year by these disturbances was equivalent to $\sim 18\%$ of concurrent regional carbon emissions from fossil fuel consumption (US Energy Information Administration 2015). Tree mortality from timber harvest was concentrated in the high-biomass forests of the Northwest, where it accounted for $\sim 80\%$ of the mortality caused by these three types of disturbance. Shifts in management priorities in the Northwest could reduce tree mortality and subsequent GHG emissions as a means of mitigating climate change. Bark beetles caused tree mortality widely across the region, though mortality was concentrated in the Rocky Mountains. Tree age and high stem density in these mountain forests made them susceptible to beetle attack, while rising air temperatures increased beetle populations and caused drought-induced reductions in tree defense against beetle attack. Lastly, tree mortality from fires peaked in California, where high temperatures increased fuel aridity and human activities increased fire ignitions. Tree mortality from bark beetles and fires will likely increase in parts of the regions over the coming decades as anthropogenic GHG emissions drive higher temperatures and increased risk of drought. Efforts to manage natural resources and meet GHG emission targets will all benefit from better understanding of the magnitude, location, and causes of tree mortality.

Data Availability

Our geospatial estimates of tree mortality from fires and bark beetles will be publicly archived with Oak Ridge National Laboratory at the Distributed Active Archive Center for Biogeochemical Dynamics (<https://daac.ornl.gov/>). This data set includes annual estimates of tree mortality and uncertainty from 2003 to 2012 at 1 km spatial resolution for the western US.

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Appendix B

Appendix B includes the expert reports referenced in our comment.

Surveys for Closed and Decommissioned Roads in the Ochoco National Forest Black Mountain
Vegetation Management Project Area: What Did We Learn?

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Great Old Broads for Wilderness; Central Oregon Bitterbrush Broadband

Abstract

The Great Old Broads for Wilderness (Broads) are concerned about the high density of roads on the Ochoco National Forest (Forest) particularly roads that are designated as “closed” roads (Maintenance Level 1 (ML-1)) which are roads held in storage for future use, and “decommissioned” roads, that have been removed from the Forest Service system. Both types of roads sometimes receive unauthorized vehicular use by the public. The Broads conducted road surveys in the Black Mountain Vegetation Management Project (Project) area, a 34,013-acre area that will receive extensive vegetation treatments along with new construction and re-construction of temporary and Forest system roads. Roads that are depicted and evaluated as closed or decommissioned on a map but are functionally open, cause the same impacts to fish and wildlife species and their habitats as open roads. Many of the closed and some of the decommissioned roads on the Forest are not physically blocked and the public has no way of knowing the roads are closed unless they use a Motor Vehicle Use Map (MVUM) to determine whether the road is open for public use or not. We surveyed 115 ML-1 closed, decommissioned and user-created roads in the Project area in July and September 2020 to determine their status of use. Of the 115 roads we found, 63 roads were closed, decommissioned, or user-created and were receiving unauthorized vehicular use, 45 roads were closed and/or decommissioned primarily due to vegetation growth of which 7 had physical barriers or were “ripped” to deter use, and 7 decommissioned roads were not surveyed. We documented that 63% of the ML-1 closed roads (45 of 71 ML-1 roads surveyed), 33% of the decommissioned roads (13 of 39), and at least 5 user created-roads were documented in the Project area that had unauthorized vehicular use. The latter appeared to be user-created roads that were not on any map and were driven by the public, and in some cases, were part of a much larger network.

Introduction

The Forest has a high density of roads across the landscape except in designated Wilderness Areas and some special Management Areas. Most of the extensive road system has been in existence for more than 40-50 years and were generally constructed for access to timber harvest areas (Black Mountain Final Environmental Impact Statement (FEIS) p. 275).

The Forest Service designates areas of the forest by a management area (MA). Most of the Black Mountain Project area is in MA-F22 (General Forest) while there are smaller MAs identified as the North Fork Crooked River Recreation Corridor, Visual Corridor (along the 22 and 3010 roads), Summit Trail Retention (along the historic Summit Trail), Summit Trail Preservation, Old Growth, and Developed Recreation (Allen Creek Horse Camp). Management Area designation determines the type and amount of vegetation management treatments and legally open road densities for each MA.

The Forest Service under the 2005 Travel Management Rule implemented a new road closure system in 2011 on the Ochoco Forest. Prior to that all roads were open unless designated closed and Forest users had motorized access to anywhere on the forest that they could drive as long as they did not cause resource damage such as wildlife disturbance, damage to meadows and scablands, and the spread of weeds. Under the 2011 Travel Management Plan (TMP), Forest users are only allowed to use open roads that are designated on the MVUM maps. In addition, under the Forest Land and Resource Management Plan (LRMP) (USDA 1989), the Forest has a standard of a maximum of 3 miles of open road density per square mile in General Forest MA (LRMP 4-259). This standard is used by forest personnel to determine whether a proposed project area complies with or violates the forest road density standard.

Forest Service roads are assigned a Maintenance Level (ML) of 1-5 depending on the service level provided and maintenance required for the road (Black Mountain FEIS p. 276). Open roads are defined as all roads that the public legally can drive and are designated ML 2-5 roads. ML 2-5 roads are open to motorized traffic and receive varying levels of maintenance depending on the service provided (FSH 7709.59). The Forest Service states that "Operational Maintenance Levels define the degree of maintenance required for a specific road and the level of service which that road provides, consistent with road management objectives and maintenance criteria (Forest Service Handbook (FSH) 7709.58, Transportation System Maintenance Handbook)." ML 1 roads are considered "closed" which means they are suitable for nonmotorized uses such as pedestrians and equestrians but closed to motorized traffic and receive minimal custodial maintenance (Black Mountain FEIS p. 512). Appropriate traffic management strategies on ML 1 roads are "prohibit" and "eliminate" all traffic (FSH 7709.59_63.32). It is prohibited to possess or operate a motor vehicle on ML 1 roads (36 CFR 261.15). Only open roads, and not closed and decommissioned roads are shown on MVUM maps.

The Black Mountain FEIS reported a pre-project open road density of 2.66 miles per square mile (mi/mi²) (Black Mountain FEIS p. 209). Forest specialists calculated the open road density only using ML 2-5 designated roads and assume all ML 1 "closed" roads in the project area are not driven, and that there are no additional (i.e., undesignated, temporary hauling roads, user-created routes) roads in the Project area.

The following three paragraphs enumerate Forest and project specific open road densities as compared to the Project calculated open road densities.

The 2003 Ochoco Forest Roads (Roads) Analysis stated in Chapter 2 that "In most areas [on the forest] existing open road densities are higher than the identified objectives. The Trail System and Off-Highway Vehicle Management and Development EIS completed by the Forest in 1996 found open road densities ranged from 0 to > 12 mi/mi². Average road densities in General Forest and Winter Range designations were 4.01 mi/mi² and 3.14 mi/mi² respectively" (USDA 2003). The 2003 Roads analysis also reported that many roads were constructed in close proximity to or across streams, contributing to sediment instream, and that the current road system has increased the hydrologic network causing water to flow more quickly from most 5th field watersheds.

The Summit Trail Supplemental Final Environmental Impact Statement (Summit Trail SFEIS) which encompassed the Black Mountain Project area, analyzed the miles of existing roads and proposed trail miles within 300 feet of streams (Summit Trail SFEIS Table 116, p. 221). The Summit Trail SFEIS reported that the Peterson Creek subwatershed (that includes Peterson Creek, Allen Creek and tributaries) had an open (legally) road density of 3.7 mi/mi² within 300 feet of streams while the total road density including closed roads was 4.7 mi/mi². For the Porter Creek subwatershed road densities within 300 feet of streams were even higher at 4.4 mi/mi² for legally open roads and 6.5 mi/mi² for total road mile density. Within the entire watershed but particularly within the designated 300 feet of Riparian Habitat Conservation Areas, the areas in the Summit Trail Project area failed to meet the LRMP road standards for open road densities of no greater than 3 mi/mi² in General Forest.

The Black Mountain FEIS disclosed that the open road density in the project area is 2.66 mi/mi² (Black Mountain FEIS Table 63, p. 209), which is substantially less than reported in the Summit Trail SFEIS. Unlike the Summit Trail SFEIS which informed the public of combined open and closed road densities, the Black Mountain FEIS did not disclose this information. The FEIS stated that the entire project area is 34,013 acres (FEIS p. 161) and "There are approximately 193 miles of roads within the project area (141.2 miles open, 51.5 miles closed) (FEIS p. 78). Therefore, the most conservative estimate of road densities in the project area is 3.63 mi/mi² including open and closed roads (ML 1-5). This estimate does not include decommissioned roads that are driven by the public, temporary roads or user-created roads, underestimating the real impact to natural resources.

The high density of open, closed, decommissioned and user-created roads that are driven by motor vehicles on the Forest impact aquatic and wildlife species and their habitats in a variety of ways. Essentially closed, decommissioned and user-created roads that are driven have the same impact as an open road. For wildlife species, impacts from open roads and closed roads that are driven include loss of habitat, habitat fragmentation and reduced connectivity, direct and indirect mortality, increased stress and impaired reproduction, and alteration of habitats by increased invasive plant and animal species. For streams and aquatic species, forest roads and associated activities (such as logging roads or off highway vehicle trails) increase the hydrologic network connecting uplands with streams, alter peak flows and contribute to increases in sediment and turbidity. Riparian areas, streams and wetlands generally occupy a small percentage of the landscape but are sensitive and more vulnerable to destruction, and their impairment is critical because of their scarcity and importance to riparian dependent aquatic and terrestrial species. Similarly, wetlands serve essential functions, such as improving water quality by filtering sediments, nutrients, and contaminants from the water column (US Environmental Protection Agency 2002). Wetlands also provide essential habitat for a wide variety of plants and animals including many federally threatened, endangered species, as well as sensitive species. For example, the Columbia spotted frog and redband trout are aquatic federal species of concern in the Black Mountain project area. Riparian areas and wetlands are a small percentage of forest lands but are generally more productive in plant and animal biomass and higher in species biodiversity than surrounding areas. When the Forest underestimates the road density in a project area by only including legally open roads, and discounts vehicle use of unauthorized roads, the Forest

underestimates the impact to other forest resources including fish and wildlife species and their habitats, riparian areas and wetlands, and water quantity and quality in streams.

Methods

The Black Mountain Project is generally located on the east side of Big Summit Prairie of the Ochoco National Forest (Figure 1). The purpose of the 34,013-acre Project is to “restore characteristic (i.e., dry forest vegetative) conditions in the Project area, thereby increasing resilience to insects, disease, fire, and drought, reducing the risk of uncharacteristic high severity fires” (Black Mountain FEIS p. 10). The Project will implement commercial harvest, noncommercial thinning, prescribed burning, hardwood enhancement, new road construction, temporary road construction, reopen “closed” roads, and close and decommission roads on 15,810 acres of the Project area.

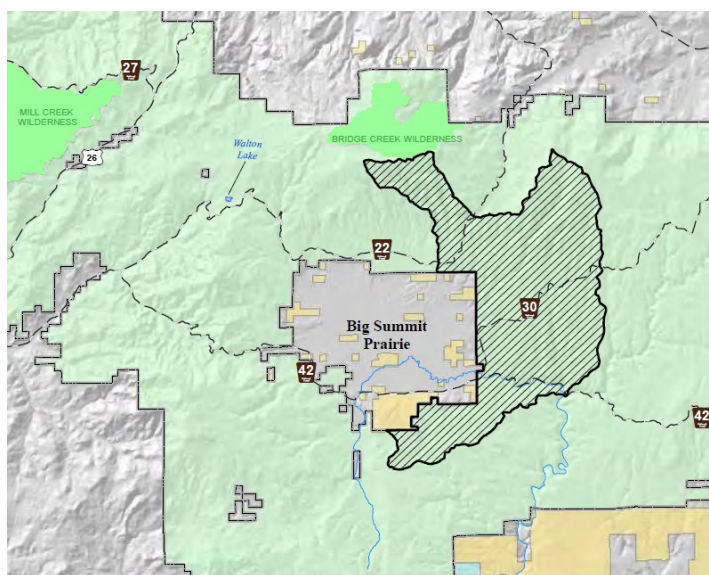


Figure 1. Black Mountain Vegetation Management Project vicinity. The project is located approximately 40 miles east of the City of Prineville, Oregon (Source: Black Mountain FEIS Map 1).

The Black Mountain FEIS provided maps of open and closed roads (Figure 2), with Maps 11 and 12 delineating “closed” roads in the thin red lines while “open” roads are black lines. However, the maps in the FEIS did not show some of the decommissioned roads on the administrative maps for the Lookout Mountain and Paulina Ranger districts, newly discovered roads not represented on any maps, nor any temporary roads and unauthorized routes and user-created trails.

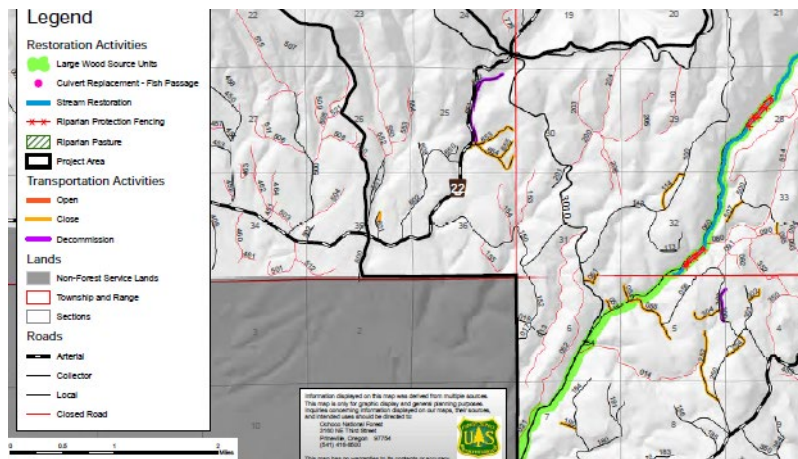


Figure 2. Portion of the Black Mountain Project area with open (black lines) and closed (thin red lines) roads (Source: Black Mountain FEIS, Maps 11 and 12). These maps did not include roads on the administrative maps or other roads found on the ground.

Members of the Broads surveyed ML 1 closed and decommissioned roads in the Project area (Figure 1) during the summer and early fall of 2020 to assess how many of the closed or decommissioned roads in the area were physically closed or had a barrier, and if they had been driven that year. The surveys occurred for four days on July 14-15, September 1, and September 7. Open roads were determined using the Forest MVUM maps #2 and #3 for the Project area, while closed and decommissioned roads were determined using maps from the Black Mountain FEIS and the Ranger District Administrative Maps available for purchase on Avenza Maps, one of several mapping applications. The two administrative maps used for the Project area were the Prineville Ranger District Map (Lookout Mountain) (1983) and Paulina Ranger District Map (1998). Roads on the Ranger District maps and not on the MVUM maps are most of the closed or decommissioned roads in the Project area. However, because these maps are decades old, the Broads found several roads on Forest Service lands that were unmarked with any road system number, that were well driven, and not on either MVUM or Ranger District administrative maps. Surveyors documented numerous user-created roads not found on any maps.

The Project area was broken into 4 quadrants and each area assigned to teams of 1 to 2 people per team. Each quadrant encompassed one of 4 stream areas on the Forest including Allen Creek, Peterson Creek, Porter Creek and Hickey Meadow and their respective tributaries. Each team was given a map of their quadrant delineating open and closed/decommissioned roads, and a list of roads to survey. Each team documented their survey of closed and decommissioned roads determining if roads were open and being driven, open and driven with a breached barrier such as a log or berm, closed due to a barrier or closed due to vegetation growing in and obscuring the road, or not surveyed. The surveyors recorded their observations on a table with their list of roads to surveys and took photos of closed and decommissioned roads and a few of the user-created roads that were driven by the public.

On the list of roads to survey, many but not all closed and decommissioned roads have a number based on the Forest Service road system with arterial, collector and “spur” roads having either a 4-digit or 3-digit number. For example, the 22 (2200 road), 30 (3000 road) and 42 (4200 road) are main arterial

roads while several roads such as the 4230 and 3010 roads are connector roads. In addition, many roads have a 3-digit road number which can be either a connector road, or a spur road that ends in the forest or on a scabland. All legally open, closed, and decommissioned roads were identified on maps so that teams could travel to the roads and determine their status. Roads were determined to have been driven by evidence of recent vehicle wheel tracks and current year's vegetation that had been flattened by vehicles. Surveyors took photos of each of the closed or decommissioned roads that had been driven in 2020 to document their use and any barriers. In many cases, the surveyors hiked closed roads that had been driven to determine how far vehicles had traveled and if there were any barriers further on the roadway not visible from the open road. In a few cases, the lower ends of spur roads from a main road are designated open for a certain distance and then legally closed for the remainder, usually in the upper reaches in a watershed (e.g. the 30-500 road in the Peterson Creek watershed).

Results:

We surveyed 115 closed, decommissioned and user-created roads in the Black Mountain Vegetation Project Area in July and September 2020 (Table 1). Of the 115 roads, we did not survey 7 of the decommissioned roads. However, we found 63 closed, decommissioned or user-created roads that were open, most with no barrier, and extensively used by the public and 45 roads that were closed and not driven. Of the 45 closed or decommissioned roads that were unused by the public, only 7 had effective physical barriers (logs that had fallen or rocks that had been placed) or were ripped to block access while the remaining 38 roads were grown in with vegetation. Of the 63 closed, decommissioned and user-created roads that were open and driven, 11 (17%) had ineffective barriers such as small berms that had been breached and 52 (83%) had no barriers at all to discourage public use. We did not document all of the user-created roads that we found but noted locations of 5 well used user-created roads driven by the public.

Table 1. Survey results of closed and decommissioned roads in the Black Mountain project area to determine if they were open and driven by the public in 2020. 65% of closed roads and 33% of decommissioned roads were open and driven by the public.

	Closed Roads		Decommissioned Roads			User-Created	
Survey Results	Closed	Open but driven	Decommissioned	Decommissioned by driven	Not Surveyed	Open	Total
Number of Roads	26	45	19	13	7	5	115
% driven by the public	63%		33%			100%	

Only one closed road had a sign to discourage use and had been well driven indicating a disregard for signage. The sign which was labeled “No Motorized Vehicles – to provide habitat security for elk” had

been broken and dropped next to a livestock barb wire gate. Many of the closed roads with barriers that had been breached had all-terrain vehicles (ATV) and ultra-terrain vehicles (UTV) tracks. The advent of these types of vehicles has allowed forest users to successfully travel around or go over what few barriers had been installed that were intended to eliminate motorized use.

These results are similar to a survey done by the Oregon Department of Fish and Wildlife (ODFW) in 2016 in the 301,000-acre area of the Summit Trail System, a proposed 137 mile Off Highway Vehicle trail system that included the Black Mountain Project area. ODFW conducted a motorized vehicle survey on randomly sampled closed roads per the 2015 MVUM maps for 3 weeks from mid-May to early June 2016. By June 3, 2016, the last day of the survey, 60% of the closed roads had received motorized vehicle use (Jackle 2016).

During the course of the Black Mountain Project road survey, we found several user-created roads and ATV and UTV trails that were not on any map but were well used. Most were found in the Allen, Peterson, and Porter creek subwatersheds although a few were found in the Hickey Meadow area as well. The most user-created roads were found in the upper reaches of Peterson, Dry Porter and East Porter creeks, with a network of user-created roads and trails connecting to the 30-500 road and between ridges and creeks. Many user-created roads were not documented in this survey.

The Black Mountain FEIS states that “Many of the three digit roads receive very light use until the beginning of archery season, at which time road use increases. Off-highway vehicle (OHV) use within the project area has also increased in the last 10 years, which is primarily associated with the beginning of archery season and related in general to other hunting activities” (FEIS p. 208-209). Further in the document the FEIS asserts that “Public road use in the project area becomes more moderate in the spring after snow melt and continues to increase through the summer. The heaviest road use occurs in late summer and fall with the commencement of deer and elk hunting season” (Black Mountain FEIS p. 275). We disagree with the statement that many of the three digit roads receive very light use until the beginning of the archery season as we saw lots of public use of open, closed and decommissioned roads during our survey in July 2020.

The ODFW surveys occurred in May and June while the Broads surveys occurred in mid-July and early September. Big game hunting seasons commence with an archery season for both species in late August, a rifle buck season in late September and early October, and 2 rifle elk seasons in late October and early November. Both the ODFW and Broads surveys occurred prior to most of the big game hunting seasons. If recreation use of closed roads increases during hunting seasons, then both the number of closed roads and the amount of use of closed roads would increase beyond the approximate 60-63% noncompliance for closed roads found in the two studies. This amount of public use of closed roads and decommissioned roads from April after snow recedes to late November each year fails to protect fish and wildlife species and their habitats from the intended designation of road closures.

The following figures are examples of photos taken of closed, decommissioned and user-created roads that were surveyed in July and September 2020 and demonstrate the ineffectiveness of using simply MVUM maps or small berms that are easily breached, to manage road closures (Figures 3-11).

1. Hickey Meadow and tributaries



Figure 3. 4240-051 road, a closed road that has been driven. The second photo is a user-created road not on any map from the 4230-820 road. Well driven roads with no barriers or signs to indicate closed.



Figure 4. 4230-706 road west and east ends, respectively. Note sign with “No motorized use – to provide habitat security for elk” on a road that has been well driven. The sign was found on the ground.

2. Allen Creek and tributaries



Figure 5. Well driven 22-453 road with berm and illegal road created to go around berm and cross Allen Creek.



Figure 6. Well driven user-created road across from 22-602 road. No road number found or signs to indicate closed road. Road not found on any administrative or FEIS maps.

3. Porter Creek and tributaries



Figure 7. 42-420 road. Well driven closed road. Second picture shows a breached berm.



Figure 8. 30-502 and 30-506 roads. Well driven closed roads in the Port Spring area. No signs or barriers found.

4. Peterson Creek and tributaries



Figure 9. 3010-080 road. Entrance road from the 3010 road and then crosses Peterson Creek. Very small and ineffective berm found across the creek at the west end. Well driven from the 3010 road to the 30-500 road (east end). No sign or barrier was found.



Figure 10. East end of 3010-080 closed road and across from this road the 30-514 closed road, both well driven. No barriers found.



Figure 11. 30-500 road at its legal end. Breached berm in the left picture that is well driven and a go around road in the right picture. This breached road allows unauthorized motor vehicle access to another 2 miles of closed road in the headwaters of Peterson Creek and a network of roads and trails that extends to Dry Porter Creek.

Discussion

Historic ranges of variability (HRV) for the Forest were developed from the Viable Ecosystem Management Guide and other historic information sources such as USGS land survey notes from the 1870's, fire history records, and the 1907 Forest Establishment Report for the Forest (Black Mountain FEIS p. 10). Historically old forests had several major components that affected the distribution and density of tree species on the landscape. They generally include 1) plant association groups, which is the potential natural vegetation of a site based on climate, elevation, soil type, soil moisture and geology, 2) seral conditions or structure, and 3) frequency of and resilience to disturbance. Resource specialists believe that the patchy nature of dry site old growth forests in central and eastern Oregon included ponderosa pine and western larch in dry mixed conifer, along with a mix of moist mixed conifers in sites with a higher moisture regime such as at higher elevations or north facing slopes (Franklin et al. 2013). These HRV descriptions of tree stands are considered the most resilient to disturbance by some Forest Service personnel (insects, disease, and wildfire), that are supposed to provide conditions necessary for dependent and other wildlife species. However, restoration for forest health by managing for tree species, size and density does not address other aspects of ecological restoration that is much needed on the forest such as reducing and improving road systems, addressing pressures from livestock grazing, and restoring riparian habitats and instream conditions in streams and rivers.

The umbrella of Forest Health guided by HRV is used to pursue vegetation management projects with the purpose of increasing resilience from wildfires. However, the Forest fails to include and adequately analyze road densities that also need to be addressed to improve ecological conditions on the forest. The Project is largely designed to restore the vegetative forest health component but not the fish and wildlife habitats and streams, riparian areas and wetlands that are affected by the large number of roads on the landscape. While the Project includes proposed road closures and decommissioning of 17.16 and 2.97 miles, respectively, these actions are dependent on future mitigation funds or grants and not assured.

For the Black Mountain Project, Forest personnel only used roads designated as ML 2-5 or legally "open" in their analyses of road density and did not include ML 1 "closed" roads, decommissioned roads and other roads including undesignated roads, temporary hauling routes and user-created motorized trails in the Project area that are driven by the public. By using only ML 2-5 roads, the Forest alleges that it is compliant with the LRMP standard of less than 3 mi/mi². The 2005 Travel Management Planning process, when road status was changed and road density standards were set, was preceded more than 16 years by the LRMP that was completed in 1989, when virtually all roads were considered open on the landscape. However, changing the legal status of a road on the landscape does not resolve its impact to other resources when it is driven. It is well documented in the scientific literature that all roads, especially those actively driven, no matter their designated status, have detrimental effects on fish and wildlife, and their habitats.

The Broads' survey was conducted to determine the number of closed, decommissioned and user-created roads that were physically open and used on the landscape and did not evaluate road densities. However, it is clear from reviewing the FEIS documents and maps, and administrative maps, that only

roads that are considered legally open are evaluated for Forest Service management projects in terms of meeting road density standards. None of the closed roads, decommissioned or user-created roads that are driven by the public were used in the FEIS analyses to determine the true road densities and consequently the real impacts to fish and wildlife populations and their habitats.

Impacts to Aquatics: Sediment is the greatest pollutant of forest streams, and in the absence of wildfire, forest road networks are the main annual sources of sediment in forest watersheds (Elliot et al. 2011). Numerous authors have reported the impacts of high road densities on streams and aquatic life. Overgrown and revegetated roads that are cleared and used for logging traffic increase erosion rates by 100-fold (Foltz et al. 2009). Years of compaction by vehicles effectively and permanently impacts runoff and sedimentation from lack of infiltration. Stream crossings along with roads are also a major source of sediment to streams (Furniss et al. 1991). There is no “safe” threshold road density for aquatic species because negative impacts begin with the first road segment in a watershed, and highly significant impacts begin at road densities as low as 1 mi/mi² (Carnefix and Frissell 2009).

Research has demonstrated increased sediment production by traffic. Roads with heavy traffic generate 4 to 5 times the sediment of roads compared to roads with light traffic (Elliot et al. 2010). One study documented that heavily used roads increase sediment production by 130 times greater than abandoned roads (Reid and Dunne 1984), while another study showed that roads with traffic deliver 2 to 30 times sediment production compared to unused roads (Luce and Black 2001).

Sediment in streams from roads causes filling of pools, enlarged channel widths and widening width-to-depth ratios (Jackson and Beschta 1984; Lisle 1982). As road density increases, pool frequency quantity and quality decline (Lee et al. 1997) which are in turn correlated with declines in native trout such as bull trout, Westslope cutthroat trout, Yellowstone cutthroat trout, and redband trout. All species exist at low population levels in highly roaded areas. Pools filled with sediment support fewer fish and fish suffer higher mortality, while elevated levels of fine sediment adversely affect salmonid embryo survival, decrease fry emergence and juvenile densities, cause loss of winter carrying capacity and increased predation (Bjornn and Reiser 1991; Chapman 1988; Everest et al. 1987; Jensen et al. 2009; Magee et al. 1996; Scrivener and Brownlee 1989; Weaver and Fraley 1993; Young et al. 1991). High sediment load reduces intragravel dissolved oxygen, decreases space for aquatic life, and limits movement of young fry (Bjornn and Reiser 1991; Chapman 1988; Everest et al. 1987; Baird et al. 2012).

Sediment also increases turbidity which reduces aquatic insects and plankton and the foraging efficiency of fish. Sediment fills interstitial spaces in the substrate, reduces habitable area in streams, and when it exceeds 20 percent of the total area on the substrate, smothers fish and frog eggs, increases mortality and reduces aquatic insect and plant production (Bjornn and Reiser 1991). A 20% embeddedness of substrates with sediment fines is a threshold well established in the literature for decline of aquatic habitat (Bjornn and Reiser 1991, Rhodes et al. 2000). While few streams have been measured for embeddedness on the Forest, both Peterson Creek (45%) and Porter Creek (22%) exceeded the 20% embeddedness indicating impairment (USDA Summit OHV FEIS 2016).

Increases in stream temperature are correlated with roads constructed along valley bottoms and next to stream channels that can remove riparian vegetation and canopy cover. Higher stream temperature impacts aquatic species by raising stream temperatures above the tolerable range, increases vulnerability to disease, reduces metabolic efficiency; shifts fish species assemblages, and inhibits upstream migration (Beschta et al. 1987; Hicks et al. 1991). Climate change with warming and drying trends will further impact streams with reduced flows further fragmenting populations and leading to local extinctions.

Impacts to Wildlife: Roads fragment forests by dissecting large patches into smaller pieces and by converting forest interior habitat into edge habitat (Reed et al. 1996). Habitat fragmentation alters the distribution of wildlife species across the landscape and affects functions such as feeding, courtship, breeding, and migration. Fragmentation from roads and other human infrastructure has been identified as one of the greatest threats to biological diversity worldwide (Noss 1987, Wilcove 1987, Noss and Cooperrider 1994). Climate change further compounds the threats of habitat fragmentation and biodiversity loss. As animals migrate due to changing climate, landscape connectivity will be increasingly important to ensure the survival of many species (Hansen et al. 2001; Kettunen et al. 2007).

Road effects are pervasive and include consequences such as population and habitat fragmentation, accelerated rates of soil erosion, and invasion of exotic plants along roadways (Rowland et al. 2005). Reduced habitat connectivity disrupts plant and animal movement and dispersal, resulting in altered population dynamics and reduced potential for recolonization if a species is extirpated from a given habitat (Haddad et al. 2015). Disturbance from roads can create a range of physiological impacts including stress and mortality such as breaking nest-supporting vegetation, collapsed burrows, inner ear bleeding, and vehicle-animal collisions as well as altered behaviors and population distribution/dispersal patterns, which leads to declines in population size, survival, and productivity.

Elk are a “management indicator species” (MIS) for the Forest. Recent studies of elk in Montana showed that motorized routes were the strongest variable that determined elk movement, even more so than canopy cover during hunting seasons (Proffitt et al. 2012, Ranglack et. al. 2016). Significantly greater habitat protection for elk was recommended in order to maintain elk on public lands including recommendations to manage for areas that are at least 5,000-acre blocks of elk security habitat, have greater than 13% canopy cover, and are greater than 1 mile from motorized routes.

The Ochoco Forest is a mixed conifer forest located generally in canyons and draws associated with riparian areas and shrub-scrublands occupying ridge tops between forested areas. However, the topography in the Ochoco Mountains is gentler and has higher road densities that are driven by forest users compared to the areas studied in Montana. Leaving large blocks of habitat unfragmented by roads on the Ochoco Forest is particularly important given the gentler topography, high road densities and with ongoing and future impacts of climate change on forest habitats.

Policy and Implementation: As stated above, ML 1 roads are considered “closed” which means they are suitable for nonmotorized uses but closed to motorized traffic and receive minimal custodial maintenance (FEIS p. 512). Appropriate traffic management strategies for closed roads are “prohibit”

and "eliminate" all traffic (FSH 7709.59_63.32). The Forest Service road maintenance guidelines reports the following: ML 1 roads have the following attributes that include but are not limited to: motor vehicular traffic is prohibited, including administrative motor vehicle traffic, and the road entrance is physically blocked or disguised (USDA 2012).

The failure to implement and enforce road maintenance level guidelines by not physically blocking or disguising closed roads continues to impact and degrade fish and wildlife habitat and is a detriment to fish and wildlife populations. The excessively high road density from all ML roads, undesignated routes and unauthorized trails on the landscape causes significant ecological harm. While some roads are included for closure and decommissioning as part of the Black Mountain Project, the road density in the Project area as calculated by the Broads is at a minimum 3.63 mi/mi² and likely much greater since it does not include non-system roads, decommissioned and user-created roads that are driven in the Project area. Merely changing the road status during the course of a project implementation and designating roads as a closed road on a map and not physically blocking access allows forest users to continue driving "closed" roads. By continuing to allow high road densities, the Forest fails to address degraded riparian and instream conditions and continues to harm wildlife habitat such as elk security cover.

There are many roads on the Forest. Unlike forest health, road densities are not evaluated in the context of HRV even though in 1870-80 there were few roads on what was eventually designated as the Ochoco Forest. Roads and trails cause forest fragmentation, destroy habitats, and cause disturbance while also providing avenues for the spread of invasive plants, human-caused fire starts, trash, poaching, and increased stream peak flow events which can result in increased sediment loads and degraded stream channels. Habitat disturbance and destruction is further complicated by the impacts of livestock grazing, particularly in riparian areas. Failure to do a complete analysis of all routes and trails in the project area gives a false view of the impacts to the existing habitat condition within the Project area.

Many miles of unauthorized roads and trails on the landscape, whether closed, decommissioned, or user-created, are illegal to drive. When forest users see a road without any noticeable restrictions (i.e., gates, signs, or physical barriers) many people drive these roads in some cases knowing its illegal and in other cases ignorant of road designation. Many members of the public are unaware of Travel Management rules that limit driving to open roads, or they do not have MVUM maps with them when driving on the forest.

There are many reports and surveys done to support environmental documents for proposed vegetation management projects. These can include surveys and analyses of resources for transportation, forest, soils, wildlife, fish and aquatic resources, botany, recreation, and cultural resources. They are intended to evaluate the "affected environment" and "environmental consequences" of a proposed project. In order to adequately assess the transportation system and its impacts on other resources, it is essential that Forest personnel use field-collected data and physically survey all roads in the proposed Project area, including those designated as closed roads or decommissioned, to determine whether they are open and driven by forest users and whether road density standards are met. Project planning efforts should also evaluate unauthorized trails and user-created trails that are driven by forest users with

motorized vehicles. In our study and in a similar study conducted in 2016 by ODFW, we found that over 60% of the closed roads and 33% of the decommissioned roads are not physically closed and are driven by the public. In addition, there were many user-created roads that were seen or documented and a large network of connected unauthorized roads in the upper watersheds of Peterson, Dry Porter and East Porter creeks. Clearly, for the Project area, an assessment of all existing ML roads, undesignated routes and unauthorized trails was not conducted.

Conclusion and Recommendation

We conclude from our road surveys in the Project area that many of the roads identified as closed and decommissioned are in fact not physically closed and the majority are driven. As shown in Figures 3-11, many of the ML 1 roads and decommissioned roads are wide open and forest users drive them. Road densities are much higher than the Project environmental documents state including the FEIS and ROD. Failing to disclose the correct information to the public misrepresents what is happening on the landscape and leads to mistrust of the Forest Service and their actions on public lands. Promising to do work after treatments are implemented would be disingenuous, especially when the promise for future road closures includes a statement that the Forest “may include trees, rocks, berms or gates” (Black Mountain FEIS p. 108). The use of the word “may” is not the same as the word “shall”, the latter which is a commitment while the former is contingent on funding and willpower to do the work.

We appreciate that there are new efforts on the Forest to close some roads although it appears to be largely based on grant funding. We support this work. But we are also concerned that Forest managers may just re-designate ML 2 roads to ML 1 roads for roads proposed for future closures in the Black Mountain Project area as a mapping exercise, with minimal effort to physically block access to “closed roads”, both on existing roads on the landscape that are being used and roads proposed for closure and decommissioning.

We recommend three strategies for the Forest to meet their legal obligations identified in the LRMP and policy, law and regulation. First, the analyses and environmental documents for any proposed project must include field surveys of all roads in the project area including but not limited to ML 1-5 roads, undesignated roads (temporary), and unauthorized routes and trails to correctly determine motorized road density and disclose that information to the public. Second, project area mitigation measures must be funded and implemented to physically block or disguise current ML 1 roads, decommissioned roads, and future proposed “closed” roads and user-created roads and trails. These should include using signage, ripping and/or blocking access with locked gates, and physical barriers such as large berms and boulders that ATVs and UTVs cannot breach. Third, increase law enforcement to enforce road closures. When violators are found, assign penalties that are meaningful that could include monetary consequences, forfeiture of vehicles, and community service. To aid law enforcement, we suggest the use of video trail cameras to strategically monitor and document the forest users driving closed, decommissioned roads and user-created roads.

The 2011 Travel Management Plan and subsequent MUVN are now in the 11th year of implementation. To be honest and transparent, the Forest must initiate efforts to effectively manage and enforce the

motor vehicle policy outlined in the TMP and MUV. The Broads believe that the Forest would likely be in compliance with LRMP standards if Forest staff physically closes or disguises all ML 1 closed, decommissioned, and temporary roads and unauthorized routes and trails, to prevent use.

The Broads will continue to monitor ongoing and future Forest projects to determine compliance with LRMP standards for road densities. It is hoped our monitoring efforts will aid the Forest in project planning and implementation and result in many more effective road closures. We are sharing the results of this report and the table of our results in Appendix A to aid in the closure of many of the closed, decommissioned and user-created roads that are currently driven by the public. Additionally, the Central Oregon Broads have supported the Deschutes, Ochoco and Malheur forests for stewardship projects such as surveys, riparian plantings and weed pulls of invasive species. We extend our volunteerism with service projects to help the Forest monitor and implement road closures.

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Appendix A. Black Mountain Road Surveys by the Great Old Broads for Wilderness, Bitterbrush Broadband. Summer and Fall 2020.

* Roads in the table that are “open” are illegally driven; yellow highlighted roads are main arterial roads

Hickey Meadow / South and West of North Fork Crooked River

Road Number	Open/Closed	Picture (Y/N)	Barrier?	Driven this year?
4230	Legal open			
4230-050	Legal open			
508	Decom			No
570	Decom but open			Yes
704	Decom			No
705	Decom but open		No	Yes, well driven both east & west end
890	Decom			No
902	Decom but open		No	Yes
903	Decom			No
904	Legal open			
906	Legal open			
908	Legal open			
912	Decom but open			Yes
913	Legal open			
914	Decom	Yes	Pasture fence	No
915	Legal open			
916	Decom but open			Yes
917	Legal open			
918	Legal open			
919	Legal open			
920	Legal open			
921	Legal open			
922	Decom			No
923	Decom but open			Yes
924	Legal open			
925	Legal open			
926	Open		No	Lightly driven
927	Open		No	Yes
928	Legal open			Merges into 140
300/900	Legal open			
800	Legal open			
802	Legal open			
809	Decom			
810	Decom			
820	Legal open			
User created road off 820	Open	Yes	No	Well driven from 820

Road Number	Open/Closed	Picture (Y/N)	Barrier?	Driven this year?
821	Legal open			
822	Legal open			
850	Legal open			
851	Decom			
852	open			
855	Legal open			
856	Open			
860	Legal open?			New gravel road; likely newly open road
865	Legal open			
870	Legal open			
4240	Legal open			
4240-050	Legal open			
051	Open	Yes	No	Yes
052	Decom			
053	Decom			Behind fence
051	Open	Yes	No	Yes
054	Decom but open	Yes	No	Yes ATVs
056	Legal open			
061	Closed			
070	Closed			
073	Closed	Yes	No	Little to no use
077	Legal open			
080	Legal open	Yes	No	No use
081	Closed			
082	Closed			
091	Legal open			
092	Legal open			
100	Closed			
110	Legal open			
113/117	Legal open			
140	Legal open			
142	Decom but open	Yes	No	High ATV use
104	Decom			

Allen Creek & North Peterson

Road Number	Open/Closed	Picture (Y/N)	Barrier?	Driven this year?
22 Rd (2200)	Legal open			
550	Lower mile legal open; upper closed			
551	Legal open			
552	Open	No	No	Yes

Road Number	Open/Closed	Picture (Y/N)	Barrier?	Driven this year?
553	Open	Yes	No	Yes
554	Open	Yes	No	Yes
601	Legal open			
602	Legal open			
User created road across from 602	Open	Yes	No	Across from 602 road
603	Open	Yes	No	Yes
604	Decom but open	No	No	Yes
605	Decom but open	No	No	Yes
608	Legal open			
609	Decom			
650	Legal open			
651	Legal open			
653	Legal open			
654	Legal open			
655	Legal open			
656	Decom			
2230	Legal open			
110	?			Not surveyed
114	?			Not surveyed
2630	Legal open			
555	Legal open			
556	Legal open			
557	Legal open			
600	Legal open			
601	Legal open			
606	Legal open			
3010	Legal open			
3010-050	Legal open			
053	Closed			Grown in
054	Closed			Grown in
055	Legal open			
056	Legal open			
080	Open east & west ends	Yes, X from 30- 514 on east end	No	Yes
90	Open	No	No	Yes
91	Open	Yes	No	Yes
92	Closed	Yes	Ripped	Yes at least once
94	Closed	Yes	No	Grown in
352	Closed	No	No	Grown in
100	Legal open			
110	Open			Yes
112	Decom			
113	Legal open			
114	Legal open			

Road Number	Open/Closed	Picture (Y/N)	Barrier?	Driven this year?
200	Open	Yes, both ends	Yes small ineffective berm at Yellow Jacket Res	Yes
201	Legal open			
202	Decom but open	Yes	No	Yes
203	Open	Yes	No	Yes
204	Closed	No	Log	No
205	Open	No	No	Yes
206	Closed	No	No	Grown in
New road ~ ¼ mile downhill of Scott Camp	Open	Yes	No	Yes, no #
3010-150	Legal open			
153	Open	No	No	No
154	Closed			
155	Closed			
30 Rd (3000)	Legal open			
30-400	Legal open			
500	Legal open to NW ¼ sec 33 then open beyond tiny berm	Yes, 3 pix	Tiny berm ATVs and 4WD vehicles well driven	Yes, well driven both around berm and over ridge
501	Closed	Yes, 2 pix	No	Grown in
502	Open to 506, then ATV track	Yes	Berm jct with 506	Yes
505	Legal open			
User created road between 505 & 510	Open	Yes	No	Yes, well driven
506	Open	Yes	No	Yes
507	Legal open	Yes		
510	Legal open until jct 511 & 512			
511	Open	Yes	No	Yes
512	Open	Yes	No	Yes
514	Open	Yes	No	Yes
User created road west of 514	Open	Yes	No, but fallen log may deter use until reopened	Yes
550	Legal open			
552	Open	Yes	No	Observed people & vehicles
553	Open	No	No	Yes
554	Legal open			
555	Legal open			

Road Number	Open/Closed	Picture (Y/N)	Barrier?	Driven this year?
556	Legal open			
557	Legal open			
600	Open	Yes	No	Yes, observed people & vehicles
603	Open	Yes	No	Yes

Lower Peterson and West Porter

Road Number	Open/Closed	Picture (Y/N)	Barrier?	Driven this year?
30 (3000)	Open			
050	Legal open			
051	Open	Yes	No	Yes
053	Decom	No	No	No, grown in
100	Legal open			
103	Legal open			North end of 150 road on FS map
104	Legal open			
105	Legal open			
106	Legal open			
107	?			Not surveyed
150	Legal open			
151	Open	Yes	Old ineffective barrier	Yes
180	Legal open			
183	Legal open			
184	Legal open			
185	Legal open			
186	Legal open			
190	Legal open			
191	Legal open			
200	Legal open			
201	Legal open			
202	Legal open			
203	Closed			
204	Decom			
205	Closed	Yes 3	No	Yes
206	Decom			
207	Decom			
250	Legal open			
252	Legal open			
254	Legal open			
300	Legal open			
301	Legal open			
302	Legal open			
303	Legal open			

Road Number	Open/Closed	Picture (Y/N)	Barrier?	Driven this year?
304	Legal open			
350	Open	Yes	No	Well driven
400	Legal open			
450	Open	No	No	Yes
451	Open	Yes 2	No	Yes, camp site
453	Open	Yes	No, berm	Yes
3010	Legal open			
011	Decom but open	Yes 2	No	Yes
012	Decom but open	Yes 1	No	Yes
013	Legal open but blocked	No	Yes Rocks	No
014	Closed	No	Yes Log	No
015	Closed	No	Yes Log	No
016	Closed			
017	Legal open			
018	Legal open			
42 (4200)	Legal open			
109	Legal open			Not on admin map
320	Closed			No
350	Legal open			
353	Legal open			
355	Legal open			
357	Legal open			
358	Closed			
359	?			Not surveyed
360	Legal open			
361	Open		No	Yes, dispersed camp
362	Legal open			
363	Closed			
364	Legal open			
366	Legal open			

East Porter

Road Number	Open/Closed	Picture (Y/N)	Barrier?	Driven this year?
42 (4200)	Legal open			
400	Legal open			
410	Legal open			
419	?			Not surveyed
420	Open	Yes 6 pix	Berm & log; easily driven around	Well driven
421	Legal Open	Yes		Well driven
422	Open	Yes, 2 Track	No	Yes

Road Number	Open/Closed	Picture (Y/N)	Barrier?	Driven this year?
423	Legal Open			
424	?			Not surveyed
425	?			Not surveyed
426	Legal open			
427	Open	Yes	No	ATV tracks to scablands
428	Open		Breached berm	Yes, ATV tracks
430	Legal open			
431	Legal open			
436	Open	Yes	No	Yes
440	Open	Yes	Yes	Yes
441	Closed			
442	Open	No	No	Yes
450	Legal Open			
451	Open	Yes	No	Yes, camp site
453	Open	Yes	No	Yes
454	Open	Yes	No	Yes
455	Legal open			
456	Legal open			
457	Closed			
458	Legal open			

60-Day Notice of Intent to Sue

EXHIBIT B

Letter from Amy Stuart, Fisheries Biologist

April 19, 2022

Oliver Stiefel
Crag Law Center
3141 E Burnside St.
Portland, OR 97214

Re: Review and Comment on the “Forest Plans Amendment – Forest Management Direction for Large Diameter Trees in Eastern Oregon and Southeastern Washington Environmental Assessment” and the Consequences and Impacts to Fish Habitat and Native Fish Populations in Riparian Areas

Dear Mr. Stiefel,

I have reviewed the environmental documents issued by the US Forest Service, Region 6, in the limited National Environmental Policy Act (NEPA) Environmental Assessment (EA) process used to remove the standard from the “Eastside Screens” that formerly protected large mature trees over 21 inches for the past 25 years on six national forests (USDA 2021a) and other related research and literature (Appendix A, Literature Cited). The following are my comments regarding the future impacts and consequences of the removal of protection of trees over 21 inches in riparian areas, including those designated as “riparian habitat conservation areas” (RHCAs), on native fish species. In my letter, I will comment on the importance of large trees in riparian areas and the consequences to native fish habitats, and the lack of adequacy of the environmental documents to fully assess and account for impacts to native fish populations, particularly species listed under the Endangered Species Act (ESA).

I am a fisheries biologist with over 38 years of experience, including 30 years with the Oregon Department of Fish and Wildlife (ODFW) before retiring in February 2014. My work for ODFW included the lands within the Project area including the John Day and Deschutes basins. I was the lead author and co-lead author for research papers on bull trout and redband trout, and restoration of anadromous fish including spring chinook and steelhead in the Deschutes basin. Attached as Appendix B is my resume.

I. Introduction

From the 1960s to the early 1990s, there were few limits on federal land management agencies to control management actions on or near streams (Roper et al. 2019). There were no safeguards to balance the risks posed by increasing timber harvest and a lengthening road system (Hicks et al. 1991; Dose and Roper 1994). This led to simplified stream channels with less instream large woody debris, increased fine sediments in streambeds, and destabilized streambanks (Meehan 1991). This period followed decades (since the 1920s) when management actions such as the construction of valley bottom roads, channel straightening, and log drives had a great direct effect on stream channels (Burnett et al. 2007; Steel et al. 2016). Beschta et al. (1995) reported that “*Land management practices in the interior Columbia... have profoundly impacted forest, grassland, and aquatic ecosystems. Watersheds and forests have been degraded (e.g., ecosystems fragmented, habitats simplified or lost, disturbance regimes altered). At every level of biological organization -- within populations, within assemblages, within species, and across the landscape -- the integrity of biological systems has been severely degraded. This is best seen in the marked reduction in the biological diversity in the region.*” The authors further stated that “*By narrowly concentrating on forest health (often a euphemism for tree health, recently referring to carbon cycling), federal land*

managers have embarked on an ambitious attempt to address forest management in ways that risk neglecting watershed health and the ecology of aquatic ecosystems.”

One of the primary goals of amending planning documents in the 1990s was to implement new management practices that would support viable native salmonids (Ratner et al. 1997; McHugh et al. 2017). The primary method to improve stream conditions was to limit activities such as timber harvest, road construction, livestock grazing, and other activities near streams that had caused the degradation of streams and riparian areas (Boisjolie et al. 2017; Roper et al. 2018). In addition, there were new policy guidelines to minimize sediment runoff from roads, requirements to conduct watershed scale analysis, identify priority watersheds where aquatic species would have greater protection, and implement stream restoration projects. Concurrent with changes in management policies that were expected to improve streams and riparian areas was a 70% reduction in the quantity of timber harvested from public lands in the Pacific Northwest region (Adams et al. 2006).

The Eastside Screens were implemented in 1994 in order to preserve the relatively few remaining large and old trees, and to protect ecosystem values such as wildlife and stream habitats. Those protections were recently removed by the scientifically controversial decision that the Forest Service implemented when the Undersecretary of the Department of Agriculture signed the Environmental Assessment (EA) Decision Document to remove protections for trees over 21 inches on east side forests in Oregon. Those protections were not in place long enough to restore the large and old trees, and the crucial fish and wildlife habitat they provided for recovery from the past century of logging that had removed the largest and oldest trees in eastern Oregon. There remains a significant deficit of large live and dead trees on the landscape in eastern Oregon as well as a relatively low abundance of the fish and wildlife species that the Eastside Screens were designed to protect and maintain.

Large trees are the foundation of old growth and mature forests, and they have a crucial role in supporting biodiversity, native fish and wildlife including numerous sensitive and Endangered Species Act (ESA) listed species, and clean water (DellaSala and Baker 2020). It is important to retain ALL remaining large trees since they are scarce on a landscape that has experienced decades of high grading timber harvest for large old trees. Large, old trees are also important since they are much more resistant to surviving wildfires. There is a large body of science that shows that increased harvest of large trees results in long-term and irreparable harm to fish and wildlife, ecosystem processes, biodiversity, and water quality. Since the Eastside Screens were implemented in the mid-1990s, the ecological basis for the Screens has not changed.

Since the Eastside Screens were amended, all six eastside forests have proposed projects to cut trees over 21 inches, with some projects including large tree removal in riparian areas. These proposed projects range in size from 35,000 to over 110,000 acres. Thus far three projects, Morgan Nesbit on the Wallowa-Whitman, Ellis on the Umatilla, and South Warner on the Fremont-Winema forests have proposed cutting trees over 21 inches in Riparian Habitat Conservation Areas (RHCA's).

For example, the Ellis project proposes to thin trees up to 9 inches diameter breast height (dbh) in the first 100 feet of the RHCA along fish bearing streams (Category 1), but contains no restrictions on harvest for the next 200 feet to the edge of the RHCA. In Category 2 and 3 streams (perennial and non-fish bearing streams and ponds, lakes and wetlands), harvest is limited for the first 75 feet from the stream but not the remaining 75 feet to the outer edge of the RHCA. Category 4 intermittent streams have even less protection, with restricted harvest only in the first 50 feet from the edge of the stream and no limits to the edge of the RHCA. Another project, Cliff Knox,

proposes logging operations in RHCAs including roadwork and ground-based skidding, and the cutting of trees over 21 inches adjacent to RHCAs.

Even though riparian buffers were established in PACFISH and INFISH to protect riparian areas, streams, and wetlands, the Forest Service has increasingly pursued cutting in RHCAs. The Forest Service has been using silvicultural prescriptions and modifying buffers to eliminate protections for and allow the cutting of large trees that provide critical shade for the streams. The impact of cutting large, mature trees, particularly in mixed conifer forests, old growth, connectivity corridors, and especially in riparian areas, will significantly and directly impact many important functions that old trees provide. Large trees over 21" are important because they support a large variety of fish and wildlife species by providing a multi layered large structural habitat as well as habitat for snag dependent and old growth species, provide shading that preserves clear, cold water for native aquatic species and assists in the improvement of soil health.

II. Riparian areas provide countless benefits for aquatic species.

Riparian areas provide very important ecosystem functions for their associated aquatic environments. Riparian areas, wetlands, seeps, and springs are among the most critical fish and wildlife habitats. Riparian areas provide habitat for over 80% of the native fish and wildlife species in western landscapes. However, riparian areas typically comprise less than 5% and sometimes as little as 1-2% of National Forest lands.

Historically, river systems provided highly complex habitat conditions for a diversity of native salmonids including Chinook salmon, steelhead, bull trout, cutthroat trout, and redband trout populations. Mature forests covered almost 75% of the region's forest types (DellaSala and Baker 2020). Old growth and large mature forests buffered the river system and contributed large quantities of wood to stream channels, forming pools and creating hiding and feeding cover for adult and juvenile fish. Mature trees and streamside riparian vegetation shaded streams, provided connectivity between floodplains and stream channels, and kept water cool for fish production.

Large trees in riparian areas are particularly important, either live or dead, standing or fallen onto the forest floor or into the stream, and are critical to forest and watershed function. In streams, they are referred to as large woody debris (LWD). There is abundant literature on the importance of LWD to the structure, function, and biodiversity of forests and particularly in aquatic ecosystems.

Large trees in riparian areas provide stream-side shading and, when they fall into streams, hiding cover for aquatic species. Large mature trees accrue soil depth, cycle nutrients, mitigate pollution, purify water, release oxygen, and provide habitat for wildlife at levels far superior to logged forests (Mackey et al. 2013, Mackey 2014, Mackey et al. 2014, Brandt et al. 2014).

The bulk of the National Forest System roads were built in the last 50 years, and with a few exceptions, most were constructed for access to timber harvest, particularly along streams and in riparian areas. Given the depletion of large trees over the past century from high grading timber harvest practices, the remaining large trees in riparian areas are all the more important to retain. Large trees provide a far greater benefit than smaller trees since they are critical for forming and retaining high quality pools, they last longer than smaller trees, and are more effective at providing instream habitat and reducing bank erosion.

Forest Service research scientists have repeatedly acknowledged the importance of large old trees for aquatic ecosystems. Hessburg et al. (2020) reported that "*Larger stems with root wads are more effective at creating deep*

*plunge pools and side channels than smaller wood pieces because they can deflect greater hydraulic power and they last longer in streams. Smaller stems are often removed by spring flood and other channel reorganizing events...Large trees play a vital role in creating instream structure by adding complexity to stream channels and providing shade, and they are essential to creating high-quality, durable fish habitats...Across a continuum of habitats from the headwaters to the mouth, large wood plays a functional role in the creation of complex habitats for native fish (Fauschet al. 2002, Vannote et al. 1986).” The authors further stated that “Highly complex habitat mosaics reflect a combination of dead trees, log and debris jams, sediment, varying hydraulic power, and stream width (fig. 24). In headwater areas, habitat is often characterized by high-gradient streams. In such environments, large wood moderates flow, providing cold water and micro refugia for fish, salamanders, and other aquatic species (Vannote et al. 1986). Farther downstream, large wood contributes to the complexity of floodplain habitats by slowing water, depositing sediment, and facilitating seasonal flooding. These floodplain environments are highly productive for native fish and are enhanced by the presence of embedded large wood (Jeffres et al. 2008). Large water-adjacent trees often contribute to the complexity of aquatic habitats (Reeves et al. 2018). Large wood slows water, resulting in the deposition of sediment that is then sorted into microhabitats by the water column (Montgomery and Buffington 1997). Instream cover provided by large wood offers critical refuge from predation, particularly for rearing juvenile fishes (Dolloff and Warren 2003). Large wood is also important to the life histories of dozens of species of fish that associate with large wood for cover, spawning, and feeding. Other aquatic organisms such as crayfish, freshwater mussels, and turtles also use large wood during part of their life cycles (Dolloff and Warren 2003). Further, large wood with root wads tends to survive longer in the stream channel, providing cover and moderating sediment and flow dynamics more effectively than small wood, which tends to be washed downstream. Shading of stream channels by large riparian trees provides critical thermal regulation and cooling (Poole and Berman 2001), particularly for cold-water dependent fish such as bull trout (*Salvelinus confluentus*) (Dunham et al. 2011). As climate changes, anticipated thermal stress will further reduce habitat available to native coldwater fish (Falke et al. 2016, Isaak et al. 2017, Rieman et al. 2007). Large trees throughout the river network are important regulators of thermal condition.”*

Yet, despite these extensive observations on the importance of large old trees to aquatic ecosystems and their importance for fish and other aquatic species, the Eastside Screens amendment removes the protections for large trees in riparian areas.

III. Riparian logging impacts ESA-listed aquatic species—such as Chinook salmon, steelhead, bull trout—and other native fish and aquatic species in many ways, including by impacting nutrient loads, stream temperatures, sediment loads and LWD, all important components for habitat quality.

It is well established that timber harvest and thinning near streams and in riparian areas can harm water quality and fish habitat. An established and growing body of science has documented the negative effects associated with the removal of large, commercial-sized trees. While non-commercial thinning and prescribed burning of fine fuels may only cause a short-term loss of habitat (i.e. a few months or a few years), the impacts from removal of large trees and their canopies, snags and down wood can last decades or longer.

Effects on fish habitat from loss of riparian vegetation include increased stream temperature, loss of cover, increased erosion, a widening and shallowing of the stream channel, and reduction or loss of perennial flow. Degraded habitat is characterized by increased sediment, increased water temperatures, a decline in pool depth, quality and frequency, reduced LWD, increased cutbanks and bank instability, and high width/depth ratios. Water

quantity and quality problems, primarily flow reduction or loss, temperature, sedimentation, and turbidity, limit fish distribution and production (Bottom et al. 1985).

Studies have shown that logging, even selective logging, has increased instream fine sediments (Kreutzweiser et al. 2005, Miserendino and Masi 2010) and increased instream water temperatures (Guenther et al. 2012), and these changes in habitats are found decades after logging.

Most streams on National Forest lands in Eastern Oregon already have high stream temperatures and currently violate state water quality standards for temperature. These high temperatures are a serious and long-term threat to water quality and continue to be a problem due to both past and current land management practices of timber harvest, high road densities and livestock grazing. High stream temperatures, especially in violation of state and Forest Plan stream temperature standards, create chronic and acute lethal conditions for native fish and limit their productivity and survival in areas where they are already stressed by high loads of fine sediments. Even local increases of water temperatures and sediment at the subwatershed or reach scale can cause local extinctions which cannot be recovered (Frissell 2017).

Logging in riparian areas can increase nutrient loads, stream temperature, and sediment to the stream, compromising fish habitat and water quality. Any active management in stream corridors and riparian areas causes a risk of harm to stream ecosystems (Frissell 2017) via the release of nutrients and increases in sediment and stream temperatures.

A. Nutrients. One important function of riparian areas is nutrient filtering and retention from harvest activities up slope. Any vegetation disturbance from wildfires or timber harvest, but especially logging, frees phosphorus and nitrogen in soils and causes a spike of these elements in streams. Excess nitrogen causes increased growth of aquatic plants and algae and in excessive amounts it can use up dissolved oxygen and block light to deeper waters. Too much nitrogen and phosphorus in the water causes algae to grow quickly and beyond what riparian areas and streams can handle. Large increases in algae harms water quality and aquatic habitats and decreases oxygen that fish and other aquatic life need to survive. In large algal blooms, oxygen is depleted leading to toxicity and even death of large numbers of fish. Sweeney and Newbold (2014) reported that *“The most familiar aspect of streamside forest buffers is their role in creating sufficient space to intercept pollutants created by upland activity before they enter the stream. It is intuitive that wider buffers with more vegetation have greater potential for intercepting, sequestering, degrading, and processing pollutants.”* The authors conducted modeling of nitrogen removal and, for a site with average water flux, predicted *“removal efficiency of 48% for a 30-m buffer, increasing to 90% for a 100-m buffer.”*

Wetlands, like riparian areas, also serve important ecosystem functions, such as improving water quality by filtering sediments, and nutrients and contaminants from the water column (Environmental Protection Agency 2002). Wetlands are also critically essential habitats for a wide variety of plants and animals, including many federally threatened and endangered species.

B. Stream temperature. Timber harvest, particularly in streamside zones, increases water temperature that can become chronic or acutely lethal to native fish species especially salmonids that are adapted to cold water. Yonce et al. (2021) observed that *“The effects of timber harvest on the thermal regime of streams have been observed globally”*. The authors stated that *“Reduction in baseflow, which can occur due to soil compaction during harvest, may also reduce the capacity to absorb increased thermal inputs. Preventing these adverse effects was a motivating*

factor in the development of riparian buffer management practices for forestry. Riparian buffers influence water temperature through shading that reduces direct insolation to streams, and secondary effects on microclimate (air temperature and wind) within the buffer are also important. Additional benefits of buffers include reducing peak storm runoff, maintaining stable baseflow, and potentially filtering out sediment, nutrients, and other pollutant loads from the surrounding area."

Stream temperatures are also increased by the interruption of hillslope drainage patterns. By intercepting surface and subsurface flow, and increasing the channels on the landscape, it alters the magnitude of peak flows and lowers base stream discharge (Furniss et al. 1991). Loss of riparian cover and shading from timber harvest is well known while a third mechanism is the conversion of groundwater to surface water (Williams 1999). When water is changed from subsurface to surface water, as streams degrade, it is exposed to solar heating (Fissekis and Hersh-Burdick 2007; Wonzell et al. 2009). The hyporheic zone is the area under or beside a stream channel or floodplain that contributes water to the stream and has been found to have a significant role in supporting cold water refugia for salmonid fish populations.

Stream temperature is also impacted by the density of the road network in watershed. Increases in stream temperature have been strongly correlated with roads constructed along valley bottoms and next to stream channels and from the associated loss of riparian vegetation and canopy cover adjacent to those roads. Higher stream temperature impacts aquatic species by raising stream temperatures above the tolerable range for rearing, and increases vulnerability to disease, reduces metabolic efficiency, shifts fish species assemblages, and inhibits upstream migration (Beschta et al. 1987; Hicks et al. 1991).

C. Sediment. It is well documented in many studies that elevated sediment occurs as a result of timber harvest and the increase of "temporary" roads that continue to have impacts on the landscape well after the project is completed. PACFISH and INFISH both expounded on the contribution of timber harvest and roads to the degradation of streams and listing of anadromous and inland fish species. Many scientists and reports have affirmed that 200-to-300-foot riparian buffers along streams are minimums to protect aquatic habitats. A review of numerous authors in INFISH reported that *"non-channelized sediment flow rarely travels more than 300 feet and that 200-to-300-foot riparian 'filter strips' are generally effective at protecting streams from sediment from non-channelized flow."* Beschta et al. (1995) reported that *"Analysis of sediment impacts, for example, that accept existing conditions as the baseline are highly inappropriate because these have been increased over natural background levels for 50-100 years in many cases. There is considerable evidence that current conditions are insufficient to maintain viable populations of many native species, including sensitive and declining trout, salmon, and other fishes."* The large influxes and biological impacts of sediment beyond those normally in a stream are well documented, and come from timber harvest, roads construction and grazing. Bowerman et al. (2014) reported that fine sediments reduced percolation of water through salmonid redds (fish nests) and impacted both egg and early life survival. The authors also reported that management actions that restore hydraulic complexity in the form of pool-riffle sequences, point bars, and large wood enhance hyporheic exchange and support higher salmonid survival. May and Lee (2004) also stated that *"increases in coarse sediment supply can be associated with sharp reductions in salmonid habitat and productivity"* and reported that streams with discontinuous flow caused by sediment increases reduced survival by crowding fish into smaller areas with reduced food availability. The additional bedload of both fine sediments and coarser sediment delivered to streams fills in the channel, losing pool depth, channel complexity, and changing the channel profile to a shallow, wide profile (Chin et al 2004; May and Lee 2004).

Increased sediment in streams connected with roads causes increased aggradation of stream beds, filling of pools, enlarged channel widths and widening width-to-depth ratios (Jackson and Beschta 1984; Lisle 1982). Lee et al. (1997) (referenced in Carnefix and Frissell 2009) reported as road density increases, there is a clear decline in pool frequency and frequency of large pools, both of which are essential requirements for high-quality fish habitat. Lee et al. (1997) reported that increasing road densities are correlated with declines in the four non-anadromous salmonid species including bull trout, Westslope cutthroat trout, Yellowstone cutthroat trout, and redband trout. All species are less likely to use streams in highly roaded areas for spawning and rearing, and if found are less likely to be at strong population levels. As pools are filled in by sediment, they support fewer fish, and they suffer higher mortality. Elevated levels of fine sediment also adversely affect salmonid embryo survival (Bjornn and Reiser 1991; Jensen et al. 2009) and have been linked to decreased fry emergence and juvenile densities, loss of winter carrying capacity, and increased predation (Chapman 1988; Everest et al. 1987; Scrivener and Brownlee 1989; Magee et al. 1996; Weaver and Fraley 1993; Young et al. 1991). Furthermore, higher fine sediment load reduces intragravel dissolved oxygen, increases metabolic waste concentrations, decreases intergravel space for aquatic life, and restricts movements of alevins (young fry) (Bjornn and Reiser 1991; Chapman 1988; Everest et al. 1987; Baird et al. 2012).

Sedimentation affects fish habitat quality by increasing the amount of fines and causing embeddedness of gravel and substrate. This in turn reduces spawning habitat by making gravel difficult to lift, oxygenation to incubating eggs, and cover for rearing fish. Turbidity affects fish production by reducing production of aquatic insects and plankton, and foraging efficiency of fish. Sediment fills interstitial spaces between rocks in the substrate, reduces habitable area in streams, and when it exceeds 20 percent of the total area on the substrate, can smother fish and frog eggs and increase mortality (Bjornn and Reiser 1991). Increased turbidity reduces the ability of fish to feed on aquatic insects and reduces aquatic insect and plant production. Many streams throughout eastern Oregon forests have documented sediment embeddedness of greater than 20% and in some cases has high as 45% (USDA Summit Off Highway Vehicle Trail (USDA, Ochoco National Forest SFEIS 2016)).

As another example, the US Fish and Wildlife Service (2010) describes the effects of sediment on ESA-listed bull trout: *"The introduction of sediment in excess of natural amounts can have multiple adverse effects on bull trout and their habitat (Rhodes et al. 1994, pp. 16-21; Berry, Rubinstein, Melzian, and Hill 2003, p. 7). The effect of sediment beyond natural background conditions can be fatal at high levels. Embryo survival and subsequent fry emergence success have been highly correlated to percentage of fine material within the streambed (Shepard et al. 1984, pp. 146, 152). Low levels of sediment may result in sublethal and behavioral effects such as increased activity, stress, and emigration rates; loss or reduction of foraging capability; reduced growth and resistance to disease; physical abrasion; clogging of gills; and interference with orientation in homing and migration (McLeay et al. 1987a, p. 671; Newcombe and MacDonald 1991, pp. 72, 76, 77; Barrett, Grossman, and Rosenfeld 1992, p. 437; Lake and Hinch 1999, p. 865; Bash et al. 2001n, p. 9; Watts et al. 2003, p. 551; Vondracek et al. 2003, p. 1005; Berry, Rubinstein, Melzian, and Hill 2003."*

D. Large Woody Debris. LWD is an important measure of habitat complexity and is important for sediment retention, pool formation and bank stability (Cordova 1995; Stuart et al. 2007). It is also an important feature in the physical and biological structure of a forested stream (McDade et al, 1990). In general, higher native salmonid densities are found in streams with more LWD. LWD adds to stream complexity and stability and provides habitat for a variety of species and age classes in aquatic communities (Lisle 2002). Most RHCAs on the managed portions of National

Forest lands are deficient in LWD from past and present timber harvest. Roper et al. (2019), comparing actively managed and reference (unmanaged) reaches of streams on national forests in eastern Oregon and Washington, and Idaho and Montana, reported that wood frequency, wood volume, and pool frequencies all increased as the percent of the riparian zone that was forested increased.

IV. Logging-associated roadbuilding and road use also causes degradation of water quality and adversely impacts ESA-listed and aquatic species.

The Eastside forests all have high road densities, a legacy of logging operations. Roads act like dry stream beds, and when rains come, they push significant amounts of sediments and polluted water into streams and rivers. This is true whether roads are functionally open, designated open or designated closed - the roads continue to have some of the greatest impacts contributing sediment to streams and aquatic habitats.

The ongoing existence of a dense road network has already stressed the system and is causing significant effects. The construction and use of new roads in riparian areas to facilitate timber harvest will only increase the significant impact that roads are having on riparian areas, wetlands, seeps, and springs and associated fish and wildlife species. The Forest Service is already proposing new road construction, the reopening of old roadbeds and new stream crossings to facilitate timber harvest. These actions will dramatically increase the impact of and the amount of sediment being pushed into the system. These are not minor, short-term impacts but large-scale, long-term impacts to aquatic systems that are already on the brink.

Most national forests have road density standards that were implemented as part of the 2005 Travel Management Rule (USDA 2005). The 2005 Travel Management Rule was adopted to address environmental harm resulting from unregulated motorized use across National Forest Service lands. For example, the Ochoco Forest has a road density standard of ≤ 3 mi/mi² in general forest (USDA, Forest Service 1989 Ochoco Forest Land and Resource Management Plan), yet in their own analysis of roads in 2003, road densities are as high as 12 mi/mi² (USDA Forest Service, Roads Analysis 2003). Another example is the Malheur National Forest where there are road density standards of 1.5 mi/mi² in big game summer range and 1 mi/mi² in big game winter range (USDA Forest Service 2021b, Cliff Knox Project DEIS) yet the open road density in the project area is 3.98 mi/mi² in the area planned for timber harvest. Since there are many closed and decommissioned roads that are either being driven or still present on the landscape in terms of their effects, the real road density is much higher.

In 2020 and 2021, volunteers with the Great Old Broads for Wilderness conducted surveys of “closed and decommissioned roads” in project areas of the Ochoco Forest. The Black Mountain Vegetation Management project DEIS stated that the existing road density was 2.66 mi/mi² only using ML 2-5 open roads (USDA, Forest Service 2018, Black Mountain DEIS). However, the Broads noted that counting ML 1 closed roads of which most were driven, the road density for M 1-5 roads was 3.66 mi/mi². The Forest Service failed to enumerate the real open road densities on the landscape. The surveys showed that the density was even higher as it found that 63% of the ML-1 “closed” roads, 33% of the decommissioned roads, and numerous user-created roads had unauthorized vehicular use. The surveys noted that 80-90% of the “closed” roads had little or no barriers to restrict access and most had been driven. In another area of the forest, the proposed Mill Creek project area (a project that proposes the removal of large trees greater than 21 inches), the survey crews found a similar level of closed and decommissioned roads actively driven by the public and many user-created roads.

Road density standards and meeting those standards are important because the effects of roads and motorized trails on the landscape concentrate flow and increase runoff and sediment to streams. The increased hydrologic connectivity of flow and sediment seriously degrades instream habitat for aquatic species including macroinvertebrates (aquatic insects), Columbia spotted frog, and a variety of salmonids including native fish such as Chinook salmon, steelhead, bull trout, and redband trout.

Many authors have reported on the consequences of forest management and the attendant proliferation of roads across the landscape. DellaSala et al. (2011) state that *“the roaded, intensively managed landscapes of the other national forest lands have been closely correlated with heavily sediment-laden streams and dramatic changes in flow regimes (Espinosa et al. 1997; Trombulak and Frissell 2000; Center for Biological Diversity et al. 2001; Coffin 2007; Frissell and Carnefix 2007).”*

Elliot et al. (2011) reported that sediment is the greatest pollutant of forest streams, and in the absence of wildfire, a forest road network is usually the main annual source of sediment in forest watersheds. A major factor that increases erosion is traffic. Roads with heavy traffic generate 4 to 5 times the sediment of roads compared to roads with light traffic (Elliot et al. 2010). Carnefix and Frissell (2009) found that there is not a truly “safe” threshold road density for aquatic species, but instead negative impacts begin with the first road segment in a watershed, and highly significant impacts began at road densities as low as 1 mi/mi². Motorized use of unpaved roads is a direct factor in sediment production (Furniss et al. 1991). While several research efforts have attempted to describe and quantify increased sediment production for use of roads, some studies have documented that heavily used roads increase sediment production by 130 times greater than abandoned roads (Reid and Dunne 1984). Other studies have documented that sediment production for roads with traffic deliver 2 to 30 times sediment production compared to unused roads (Luce and Black 2001). Foltz et al. (2009) reported that after Forest Service roads were reopened from being closed or decommissioned, sediment concentrations were significantly higher and were attributed to decreased vegetative cover. In addition, the authors found that 30 years of no traffic and vegetation regrowth was not sufficient to allow recovery of soil infiltration to values similar to an undisturbed forest. In other words, the years of compaction by vehicles effectively and permanently impacted runoff and sedimentation from lack of infiltration. Stream crossings along roads are also a major source of sediment to streams (Furniss et al. 1991).

The small roadless areas that have been left “unmanaged”, with a lack of logging and roads, play a critical role in maintaining high-quality water and protecting aquatic ecosystems. The clean water from remnant roadless areas is important to maintain healthy coldwater salmonid fisheries, sustain viable aquatic ecosystems, and help protect threatened species and ecosystems (Abell et al. 2000; Trout Unlimited 2004). Roadless areas are an important refugia for many salmon and trout populations, as well as for a diversity of endangered freshwater species (Henjum et al. 1994; Huntington 1998; Trombulak and Frissell 2000; Center for Biological Diversity et al. 2001; Strittholt and DellaSala 2001; Oechsli and Frissell 2002; Strittholt et al. 2004; Petersen 2005). DellaSala et al. (2011) state that *“restoration of salmon and trout fisheries in places with high road densities will likely fail without the pivotal role provided by roadless areas as fishery strongholds.”* Further, they state that *“For many major drainages (entire watersheds of major rivers, such as the Columbia River Basin), roadless areas and other wilderness areas represent the last few percentages (typically 1% to 5%) of the landscape with a minimally disturbed, or near natural, hydrology.”*

V. The FS falsely relies on PACFISH and INFISH to protect streams and riparian areas. By failing to disclose all activities and consequences and relying on PACFISH and INFISH to protect riparian areas, the Forest Service disregards impacts to native fish habitat and fish populations.

The Decision Notice and Finding of No Significant Impact (USDA 2021a) for removing the 21-inch rule of the Eastside Screens states that *“the direction of INFISH and PACFISH, which provide standards and guidelines to protect water quality, continue to apply”* while the Decision Summary states that *“all management will remain the same within Riparian Habitat Conservation Areas. PACFISH, INFISH, their objectives, goals, standards and guidelines direct management will still apply.”* Based on this continued management direction, the FS brazenly concludes that the removal of protections for large trees across the Eastside Forests will have “no effect” on ESA-listed aquatic species.

PACFISH and INFISH were designated with the intent to recover highly degraded streams from past and present management activities of timber harvest, roading, mining, and livestock grazing. Timber management activities were only allowed if they were implemented with the intent to move Riparian Management Objectives (RMOs) to a better condition.

However, in the past decade, all forests east of the Cascades have increasingly pursued timber harvest in riparian areas, often under the guise of “restoration” or “wildfire resiliency.” In some cases, forests have created arbitrary designations of a “streamside management zone”, “an inner and outer RHCA zone” and a “limited activity buffer” that allow commercial harvest and other logging operations in a portion of the RHCA. In some cases, RHCA buffers themselves are too narrow to prevent impacts to aquatic ecosystems, and this further narrowing of buffers has introduced still more impacts. This has led to continued timber harvest, roading and livestock grazing and the failure to recover degraded streams and riparian areas.

Yet, the EA and Decision Notice for amending the Eastside Screens failed to provide any numerical data on stream conditions and parameters for meeting or violating RMOs from PACFISH and INFISH. All six forests that were covered under the EA to remove the 21-inch rule have highly degraded watershed function, streams and riparian conditions throughout all actively managed areas. The few areas that may meet RMOs are typically found in wilderness, roadless areas, and undeveloped lands. The EA and Decision Notice fail to look at baseline conditions and the cumulative effect of removing the protections for large trees across the broad landscape.

The EA and Decision document to amend the Eastside Screens (USDA 2021a) state that *“Aquatic and riparian species habitats are protected through the guidance of the PACFISH/INFISH strategies and will not change under this amendment”* and *“Project level analyses will be required to address any site-specific effects that projects may have on riparian and wetland associated species and their habitats.”*

However, at the site-specific project level, the Forest Service fails to acknowledge that increased sediment from the large planning areas for timber sales will increase impacts on ESA-listed and sensitive aquatic species. For example, the Ellis Integrated Vegetation Management project on the Umatilla National Forest encompasses a planning area of over 114,000 acres for commercial timber harvest, thinning, and additional roads to conduct treatment activities. While the DEIS acknowledged that watersheds and streams were in very poor degraded conditions, the only data provided were that the expected sediment loads would increase from all action alternatives (USDA 2021c, Ellis DEIS and Ellis Hydrology Report). The projected sediment increases are alarmingly high, and in most of the subwatersheds, sediment is expected to increase anywhere from 100% to an astonishing almost 400% over the existing condition during treatment activities. Over 85 miles of streams in the project area are identified as designated critical habitat for ESA-listed Mid-Columbia River steelhead. The DEIS states that there are existing conditions of highly degraded watersheds and streams with elevated water temperatures and elevated sediment, and the project will result in increases of sediment from soil disturbance and loss of shade from the project. Then the DEIS asserts that *“Alternatives 2 and 5 [which will have the greatest amount of harvest and roads]*

would not hinder attainment of relevant properly functioning indicators pertaining to sediment and roads.” In other words, the FS found that despite the huge sediment increases caused by the project’s riparian logging and associated roadwork, the project is consistent with the PACFISH management direction.

As discussed above, sediment increases of this magnitude pose significant consequences for aquatic species. High sediment loads and water temperatures have already led to the severe decline of ESA-listed Mid-Columbia steelhead. Clearly in the case of the Ellis Integrated project and other projects, impacts to fish habitat and ESA-listed species will continue to occur, notwithstanding the PACFISH/INFISH management direction. Despite severely degraded riparian and stream habitat conditions and watersheds not in proper functioning condition, and the acknowledgement that timber harvest will dramatically increase sediment, the Forest Service continues to authorize riparian logging activities.

Another example is bull trout in the North Fork of the Malheur River that will be impacted by the Cliff Knox project (USDA 2021b). The Forest Service used a U.S. Fish and Wildlife Service process paper titled “A Framework to Assist in Making Endangered Species Act Determinations of Effect for Individual or Grouped Actions at the Bull Trout Subpopulation Watershed Scale” (USDI FWS 1998) to describe the environmental baseline for the project. The ESA-listed bull trout in this area has a subpopulation of less than 50 adults and are found in an 8-mile segment of designated critical habitat of the Malheur river and tributaries in the project area. The Cliff Knox DEIS cites that *“The Malheur River core area was estimated to have 50-250 individuals; however, the short-term trend was determined to be declining. The threat to the species was considered to be substantial, imminent and the population is at a high risk for extinction (USDI FWS 2008b).”* This high-risk bull trout subpopulation is characterized as in rapid decline or maintaining at alarmingly low numbers and under current management, the subpopulation will not improve with two generations (5 to 10 years). Yet, despite degraded habitat conditions with high water temperatures and sediment loads, the Forest Service downplays the impacts of increased sediment and loss of shade from the project and finds that the project is consistent with the INFISH management direction.

The impacts from riparian logging activities have the potential to increase significantly now that the Forest Service is permitted to cut large trees. As discussed previously, large trees are of outsized importance in terms of the benefits they provide to riparian areas, including provision of LWD, retention of fine sediment, and reductions of temperature. Removal of large trees will deplete already degraded systems. When roads are built in riparian areas and roads are constructed to remove large trees, the cumulative impacts from removing large trees will further aggravate seriously degraded conditions, leading to further declines in ESA-listed fish species. Since many of the ESA-listed fish are in dangerously low numbers, further timber harvest and roading will likely lead to local extinctions. Once gone, they cannot be recovered.

High quality forested riparian habitat for ESA-listed fish species is directly linked to the presence and abundance of large old trees. The combination of shade, soil and bank stability, nutrient cycling, and high-quality fish habitat is dependent on the continued existence of trees in riparian areas, particularly large trees since they have longer retention and durability. Bisson et al. (2003) reported that large trees are important to retain in areas with hydrologic connectivity to streams because they provide longer duration habitat structure in floodplains than small trees. Since large old trees have been depleted for over a century it is important to retain all remaining large trees. Continued removal under the guise of stand improvement is based on a silvicultural construct and not based on the habitat needs of ESA-listed species that are already jeopardized by past logging and roading practices.

In sum, the Forest Service's conclusion that the removal of protections for large trees will have "no effect" on ESA-listed aquatic species, based on the PACFISH/INFISH management direction, is insupportable. PACFISH and INFISH fail to prevent against logging-related impacts, and such impacts are likely to increase in magnitude with the removal of large trees. The continued entry into riparian areas via roads to harvest large trees will further compact and disturb soils, remove trees that are important for shade, sediment retention, nutrient cycling, and water quality, and further harm ESA-listed fish species. Yet the Forest Service is proposing to allow removal of large trees under the guise of restoration and wildfire resiliency contrary to the best available science and facts about large trees. One has only to look and compare wilderness and roadless areas to highly managed forested areas to see the last best chinook, steelhead and trout habitat is in the largely untrammelled areas. The quality of fish habitat and the abundance of ESA species are highly impacted by managed habitats due to the removal of large wood and extensive road systems to remove timber. In my opinion, the loss of the Eastside Screens' protections for large trees is another nail in the future coffin for ESA fish species and will likely lead to their extinction.

VI. Conclusion

Large trees in riparian areas are particularly important for a multitude of reasons. They are critical for a myriad of ecosystem services including filtering and providing clean water, shade to reduce water temperature, nutrient cycling, and future wood recruitment for LWD in streams. LWD provides important functions for sediment retention, hiding cover for aquatic species, a substrate for macroinvertebrates as food sources for fish, pool formation, and bank stability. These are important components for high quality habitat that support ESA-listed and sensitive fish species like Spring Chinook, steelhead, and bull, cutthroat and redband trout. Logging and associated roadbuilding activities adversely impact riparian areas in many ways, and such impacts have continued to occur in the last 25 years despite the PACFISH/INFISH management direction. Allowing the cutting of large trees in and around RHCAs—along with associated roadbuilding activities—will trigger increased impacts for riparian ecosystems and further jeopardize the future existence of ESA-listed fish species.

Sincerely,

*/Amy M. Stuart/*electronic signature

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Education:

M.S. 1980, Wildlife Management, Colorado State University, Ft Collins, CO.

B.S. 1977, Wildlife Biology & Management, Cornell University, Ithaca, NY.

PROFESSIONAL EXPERIENCE:

Oregon Department of Fish and Wildlife (ODFW), 9/2007 to 2/2014, Deschutes Watershed Manager

- ❖ Supervised 60 professional and technical staff.
- ❖ Managed expenditures of a combined budget of \$9 million per biennium for 25 cost centers.
- ❖ Implemented fish, wildlife, research and habitat program goals, objectives and priorities.

ODFW 7/1997 to 8/2007, Hydro Power Program Biologist, High Desert Region

- ❖ Lead for the High Desert Region for re/licensing and compliance for state and federal FERC licensed hydro projects in the Klamath, Deschutes, Hood and Owyhee basins.
- ❖ Lead negotiator and/or technical lead for the Pelton Round Butte Settlement Agreement, Klamath Settlement Agreement, and Powerdale hydro project decommissioning.

ODFW, 1/1990 to 6/997, District Fish Biologist (DB), Supervisory FWB (3 1/2 years) and NRS 3 (4 years), Ochoco District, High Desert Region

- ❖ Fish program manager for the Ochoco District (7½ years). Hired, supervised, and conducted performance reviews of 1 permanent, several seasonal employees and 3 high school interns.
- ❖ Collected and analyzed fish population abundance and distribution, species composition, and harvest; evaluate hatchery stocking, harvest regulations; implement habitat protection/restoration.

ODFW, 2/1988 to 12/1989, Assistant District Fish Biologist II, Ochoco District, High Desert Region

- ❖ Implemented the fish management program. Sole fish biologist and assumed lead responsibility; ½ time Ochoco National Forest Fish Biologist: advised fish issues for timber sales and range allotments; conducted fish, habitat surveys and habitat projects. Authored bull trout plan at Lake Billy Chinook.

ODFW, January 1986 to January 1988, Fish Habitat Biologist II, BPA Program, John Day, NE Region

- ❖ Implemented habitat program using restoration techniques to improve fish and wildlife habitat: bank stabilization and habitat projects. Implemented contracts and landowner lease agreements. Prepared annual reports. Developed, supervised and administered budgets, personnel, equipment, and ODFW staff.

1977-1986. ODFW Ocean Salmon, US Forest Service, US Army Corps of Engineers, Colorado Mined Land Reclamation, and private timber cruiser.

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- Numerous presentations to the Oregon Fish and Wildlife Commission and conservation groups for 20 years on a variety of topics including the Crooked River Basin Fish Management Plan, Annual Fishing Regulations, Hydro Relicensing, anadromous fish reintroduction and several projects in the Hood, Klamath, John Day and Deschutes basins.

November 4, 2016

Paul Dewey

Central Oregon Landwatch

50 SW Bond Street, Suite 4

Bend, Oregon 97702

Re: Review and Comment of the September 2016 Supplemental Final EIS (SFEIS) and Draft Record of Decision (ROD) for the Ochoco National Forest (ONF) proposed Summit Trail System Project (Project) and Forest Plan Amendments.

Dear Mr. Dewey,

I have reviewed the above referenced documents and many other associated documents available on the websites of the USDA Forest Service (USFS) and other research and literature. I also reviewed the 2009 Scoping Letter, 2013 Draft Environmental Impact Statement (DEIS), 2014 Final EIS, and April 2016 Supplemental Draft EIS (SDEIS) for this Project. The following are my comments regarding the planning, analyses and conclusions of the ONF for the Summit Trail Project. In my letter, I will comment on road and trail densities that currently exist and are proposed in the Summit Off-Highway Vehicle (OHV) Project Area, the impacts of OHV trails on hydrology, sediment, and fish, and the adequacy of the environmental documents to fully assess and mitigate for the proposed Project.

I am a fisheries biologist with over 34 years of experience, including 30 years with the Oregon Department of Fish and Wildlife (ODFW) before retiring in February 2014. My work for ODFW included the lands within the Project area. I was the lead author for the ODFW Crooked River Basin Fish Management Plan (Stuart et al. 1996) and was the lead author for a research paper on the redband trout population in this area, "Redband Trout Investigation in the Crooked River Basin" (Stuart et al. 2007). Attached is my resume.

I. Regulatory Background

Regulatory documents for the Summit OHV Project include, among many, the Presidential Executive Orders, numerous federal laws such as the National Forest Management Act (NFMA), the National Environmental Policy Act (NEPA), and the Code of Federal Regulations (CFRs); national Forest Service policy and implementation guidance including the Forest Service Manual (FSM) and Handbook (FSH); Region 6 direction such as the Inland Native Fish Strategy (INFISH), and at the forest level Ochoco Forest Land and Resource Management Plan (LRMP). Other regulatory documents or guidance include the Regional Forester Sensitive Species List, the Aquatic and Riparian Conservation Strategy (ARCS), the 2011 ONF Travel Management Plan (TMP), 2008 Recreation Niche Analysis, 2008 Recreation Facilities Analysis, and 2015 Transportation Roads Analysis.

Executive Order 11644 requires that "areas and trails shall be located to minimize damage to soil, watershed, vegetation or other resources of the public lands." Executive Orders 11989 and 11644 require that the Forest Service must minimize destruction, and loss or degradation of wetlands, including floodplains. Specifically, 11989 requires that, "the respective agency head shall, whenever he determines that the use of off-road vehicles will cause or is causing considerable adverse effects on the soil, vegetation, wildlife, wildlife habitat or cultural or historic resources of particular areas or trails of the public lands, immediately close such areas or trails to the type of off-road vehicle causing such effects, until such time as he determines that such adverse effects have been eliminated and that measures have been implemented to prevent future recurrence".

Executive Order 11988 requires federal agencies to avoid to the extent possible the long and short-term adverse impacts associated with the occupancy and modification of flood plains and to avoid direct and indirect impacts to floodplain development wherever there is a practicable alternative.

Executive Order 11990 has the same direction for wetlands. These orders place a very high standard that the ONF has not demonstrated to avoid, minimize or mitigate sufficiently for impacts caused by the proposed Summit OHV Trail Project.

II. The SFEIS fails to disclose the existing number of roads and trails, including open, temporary, closed and decommissioned, and user-created trails on the landscape in the Summit Trail Project Area. By failing to disclose all road, route, and trail densities on the landscape the SFEIS underestimates impacts to streams and aquatic life.

A. The SFEIS fails to adequately analyze the existing number of roads, routes and trails on the landscape, leading to unjustified conclusions that the road, route and trail system with the addition of an OHV trail system will only cause minor short-term impacts to riparian areas, streams, wetlands and the native species that depend on those habitats.

The fundamental issue is that high road and trail densities on the landscape are not shown and analyzed. The SFEIS inconsistently analyzes road and trail densities, sometimes including Maintenance Level (ML) 1 (closed) roads, and sometimes including ML 2-5 which are open roads. By selecting the information using only open roads and not including closed, decommissioned, temporary, and reconstruction roads and user-created trails, it leads the false conclusion that there is insignificant impact to streams, wetlands, riparian areas and aquatic species. The road density information was cherry-picked to lead the reader and the public to the conclusion that a new OHV area would not harm the environment. The SFEIS contains inconsistent information on roads, with some sections sometimes including closed roads in the analyses, other sections only counting open road densities according to the Motor Vehicle Use Map (MVUM), and in many cases, displaying contradictory information. In most cases, closed, temporary, decommissioned, and user-created trails were not included in the analyses.

Starting at the Project level, the SFEIS reported that there are 1820 miles of roads in the Project area (SFEIS p. 56) which encompasses 301,581 acres (471.2 square miles) for an average of 3.86 miles of road/square mile (this estimate does not include temporary roads, decommissioned roads and user-created trails). Subtracting the Lookout Mountain Roadless Area (14,365 acres), Bridge Creek Wilderness (5,355 acres), Old Growth (7,693 acres),

Rock/Cottonwood Roadless Area (5,098 acres), and Round Mountain National Recreation Trail (1,018 acres), which are all largely unroaded areas, yields a roaded Project area of 268,051 acres (418.8 square miles). The estimated ML 1-5 road density for the Project area is 4.77 miles of road/square mile, again not including temporary and decommissioned roads, and user-created trails. Therefore, the remaining Project area violates the road density standards established by the LRMP without counting decommissioned roads, temporary roads and unauthorized user-created trails.

The following table (Table 1) states the management objectives and seasonal closures by area from the ONF LRMP (USFS 1989) and the 2003 Roads Analysis (Table 2-13) (USDA 2003):

Table 1. Road density standards from the ONF LRMP and 2003 Roads Analysis

Road Densities	Ochoco NF
Winter Range (LRMP 4-255)	1 m/m ² During closure (12/1 – 5/1) 3 m/m ² Outside closure (5/1 – 11/30)
Summer Range (LRMP 4-259)	3 m/m ² (all year)

The 2003 Roads Analysis stated in Chapter 2 that “In most areas existing open road densities are higher than the identified objectives. The Trail System and Off-Highway Vehicle Management and Development EIS completed by the Ochoco N.F. in 1996 found road densities ranged from 0 to > 12 m/m². Average road densities in General Forest and Winter Range designations were 4.01 m/m² and 3.14 m/m² respectfully.” This standard was to be applied to General Forest and General Forest Winter Range management areas, respectively. So, for example, it obviously would not apply to roadless area such as the Lookout Mountain Roadless management area.

In the Hydrology and Aquatics section of the SFEIS, part of the analyses disclosed road densities that included closed roads (ML 1) but did not include temporary roads, decommissioned roads, and user-created trails, while other parts of the analyses only included open roads. For example, the SFEIS used road/trail densities within 300 feet of Riparian Habitat Conservation Areas (RHCAs), and stream crossing densities to evaluate flow and sediment regimes and potential impacts on streams and aquatic life (SFEIS Table 116, p. 221). This information is important because the effects of roads and trails on the landscape concentrate flow and increase runoff and sediment to streams. The increased hydrologic connectivity of flow and sediment seriously degrades instream habitat for aquatic species including macroinvertebrates (aquatic insects), Columbia spotted frog, and native redband trout.

The SFEIS reported that “A road/trail density rating with less than one mile per square mile was considered low, between 1 and 2.4 miles per square mile was considered moderate, and greater than 2.4 miles per square mile was considered high” (SFEIS p. 139) and used ML 1-5 roads, but did not include temporary, decommissioned and user-created trails. These ratings were based on a USFS Watershed Condition Classification Guide and seem reasonable. However, stream crossings, which have a high potential to deliver sediment had an arbitrary rating with less than 2.5 crossings per square mile as “low”, 2.5 to 3.5 crossings per square mile as “moderate” and greater than 3.5 crossings per square mile as “high”. There is no logical explanation or basis for describing these stream crossing densities as low, moderate and high other than looking at the overall ranges of stream crossings for the subwatersheds, and randomly picking from low to high ranges of stream crossing densities. The SFEIS

further did not disclose whether these stream crossings used in the analyses were open roads or all roads including closed roads.

The ONF indicated that approximately 700 miles of user-created OHV routes are in the Project area (SFEIS p. 79) but failed to disclose those estimated numbers of miles by watershed or subwatershed, again making the analysis subjective and without disclosing those impacts. The SFEIS also notes that *“Although the Forest Service does not yet have statistically collected use data for OHVs on the district, forest personnel and the public have observed and reported an increase in the number of riders and a growing network of user-created trails”* (SFEIS p. 70). This discrepancy is further cause for concern since the SFEIS admits that *“Since miles of existing user created trails by subwatershed are unknown, these are not accounted for in road/trail density estimates and likely would affect these results”* in the flow/sediment analysis and conclusion (SFEIS p. 237).

The fundamental assertion in the Hydrology and Aquatics section is that roads on the ONF have an impact on streams and aquatic life, but then chooses to implement an OHV trail system over the No Action alternative, despite essentially increasing the motorized road/trail densities. The No Action alternative provides the most protection of streams, wetlands and native species that rely on riparian and wetland ecosystems. Numerous authors have documented the impacts of roads and motorized OHV trails on streams, riparian areas and wetlands. The following is a brief summary of the impacts of roads and motorized trails on runoff and sediment that degrade aquatic habitat.

A Brief Summary of Impacts to Aquatic Habitats from Roads and OHVs

Elliot et al. (2011) reported that sediment is the greatest pollutant of forest streams, and in the absence of wildfire, forest road networks is usually the main annual source of sediment in forest watersheds. The authors also reported that a relatively new source of sediment in forested watersheds is from recreation particularly all-terrain vehicle trails. Foltz et al. (2009) reported that roads can become overgrown if not used, but if cleared and used for logging traffic, erosion rates can increase 100 fold. A major factor that increases erosion is traffic. Roads with heavy traffic generate 4 to 5 times the sediment of roads compared to roads with light traffic (Elliot et al. 2010). Carnefix and Frissell (2009) found that there is not truly “safe” threshold road density for aquatic species, but instead negative impacts begin with the first road segment in a watershed, and highly significant impacts began at road densities as low as 1 mi/mi².

Motorized use of unpaved roads is a direct factor in sediment production (Furniss et al. 1991). While several research efforts have attempted to describe and quantify increased sediment production for use of roads, some studies have documented that heavily used roads increases sediment production by 130 times greater than abandoned roads (Reid and Dunne 1984). Other studies have documented that sediment production for roads with traffic deliver 2 to 30 times sediment production from unused roads (Luce and Black 2001). Foltz et al. (2009) reported that after Forest Service roads were reopened from being closed or decommissioned, sediment concentrations were significantly higher and were attributed to decreased vegetative cover. In addition, the authors found that 30 years of no traffic and vegetation regrowth was not sufficient to allow recovery of soil infiltration to values similar to an undisturbed forest. In other words, the years of compaction by vehicles effectively and permanently impacted runoff and sedimentation from lack of infiltration. Stream crossings along with roads are also a major source of sediment to streams (Furniss et al. 1991).

Williams (1999) gave an excellent summary of how roads (and therefore by the same mechanisms, OHVs) impact aquatic habitat: Increased sediment in streams connected with roads causes increased aggradation of stream beds, filling of pools, enlarged channel widths and widening width-to-depth ratios (Jackson and Beschta 1984; Lisle 1982). Lee et al. (1997) (referenced in Carnefix and Frissell 2009) reported as road density increases, there is a clear decline in pool frequency and frequency of large pools, both of which are essential requirements for high-quality fish habitat. Lee et al. (1997) reported that increasing road densities are correlated with declines in the four non-anadromous salmonid species including bull trout, westslope cutthroat trout, Yellowstone cutthroat trout, and redband trout. All species are less likely to use streams in highly roaded areas for spawning and rearing, and if found are less likely to be at strong population levels. As pools are filled in by sediment they support fewer fish and they suffer higher mortality. Elevated levels of fine sediment also adversely affect salmonid embryo survival (Bjornn and Reiser 1991; Jensen et al. 2009) and have been linked to decreased fry emergence and juvenile densities, loss of winter carrying capacity, and increased predation (Chapman 1988; Everest et al. 1987; Scrivener and Brownlee 1989; Magee et al. 1996; Weaver and Fraley 1993; Young et al. 1991). Furthermore, higher fine sediment load reduces intragravel dissolved oxygen, increases metabolic waste concentrations, decreases intergravel space for aquatic life, and restricts movements of alevins (young fry) (Bjornn and Reiser 1991; Chapman 1988; Everest et al. 1987; Baird et al. 2012).

Stream temperature can also be impacted by roads, routes and trails. Increases in stream temperature have been strongly correlated with road constructed along valley bottoms and next to stream channels and from loss of riparian vegetation and canopy cover. Higher stream temperature impacts aquatic species by raising stream temperatures above the tolerable range for rearing, and increases vulnerability to disease, reduces metabolic efficiency, shifts fish species assemblages, and inhibits upstream migration (Beschta et al. 1987; Hicks et al. 1991).

The effects of motorized [OHV] trails on watershed processes are similar to roads although they may be lower if the footprint is smaller (Welsh 2008). Erosion rates on trails are in part determined by slope gradient and length, and amount and type of trail use with direct mechanical impacts occurring in the form of abrasion, compaction, shearing and displacement (Meyer 2002). Indirect effects occur in the form of erosion, mostly by water from rainfall or runoff from snowmelt. The SFEIS acknowledged the research done on OHV impacts to trails by Foltz (2006) and Meadows et al. (2008) but then incorrectly stated that “both of these studies need to be qualified with the fact that this study is for unmanaged ATV use” (SFEIS 238). While the Foltz (2006) did look at unmanaged OHV trail use, the Meadows et al. (2008) study was a very thorough research project that looked at how OHVs degrade natural resources from the starting point of no OHV activity, then assessed through carefully controlled experiments of repeated use of an all-terrain vehicle (ATV) on soils on 7 different national forests around the country.

The Meadows et al. (2008) study started each test site with unmotorized use areas. The two national forests that most closely fit the ONF are the Wenatchee Forest in central eastern Washington and the Beaverhead-Deerlodge in western Montana. In Montana site, loop trails were laid out on slopes ranging from 16-25%. The study site on the Wenatchee in Washington was in a planned and yet to be developed area on abandoned logging roads that had been closed to vehicle access for 20-30 years, and were selected for conversion to OHV trails. While there were some differences in soil types and terrains than the ONF, the authors had several revealing conclusions

about impacts of OHV use on natural resources for all 7 forests. In summary, Meadows et al. (2008) noted the following observations and impacts to forest resources:

- 1) At all 7 forests, natural resources were affected by ATV traffic. Some portions of the previously unused trails changed from low to medium disturbances in 20 to 40 vehicle passes, while high disturbance occurred in 40 to 120 passes. While the number of days required achieving this number of passes varied by location, 20 to 40 passes can be achieved in one weekend from a moderate-sized ATV group, while a high level of disturbance from 40 to 120 passes can be achieved in less than a month, depending on how and how many OHVs use the trails.
- 2) Following any level of disturbance, runoff and sediment generated on the ATV trails increased by 56 and 625%, respectively, compared to the undisturbed forest floor. ATV trails are “high-runoff, high-sediment producing strips on a low-runoff, low sediment-producing landscape”. The authors acknowledged that levels of disturbance can be reduced by proper trail design and maintenance but it was only a reduction and not elimination of impacts.
- 3) Increased use of ATVS on public lands means that even a small percentage of riders who desire to travel off trails and roads can have considerable effects on natural resources.
- 4) Lands may be able to rehabilitate after the effects from a few ATV rides across a meadow, but multiple passes across the same area often results in reduced or complete loss in the capacity for natural rehabilitation.
- 5) ATV use compacted soils, reducing infiltration and increasing runoff, which are the same phenomena that occur on unsurfaced forest roads with motorized use. While vehicle weights and number of passes are different, the mechanisms of traffic causing compaction and increased runoff are similar. Sediment concentrations tended to increase with increasing disturbance levels.
- 6) ATV trails adversely affected vegetation, soil, water and air by reducing or destroying vegetation, including forest floor litter, exposed and destroyed plant root networks, exposed bare soil, caused soil erosion, and dust migration as a result of bare soils. Soils were compacted, displaced or loosened, and in cases of high intensity rainfall simulation [such as can occur on the ONF during normal wet spring and fall periods and thunderstorms in the summer], freshly disturbed soils on ATV trails produce an average of 10 times more sediment than undisturbed soils.
- 7) The study results also indicated that regulating ATVs to trails is not enough to protect natural resources. While trail planning and design are key considerations to reduce impacts, it is not possible to completely eliminate impacts, and areas to completely avoid are riparian areas, ridge tops and sensitive wildlife habitats (which has not done for the Summit OV Project).

The discussion about roads and OHV impacts to streams and sediment is really important to understand because OHVs have some of the greatest impacts contributing sediment to streams and aquatic habitats. The SFEIS assertion that impacts will be minimal from the proposed OHV trail system is false. The implementation of any action alternative will significantly impact riparian areas, wetlands, seeps, and springs and associated fish and wildlife species by construction by increasing road/trail densities over the existing condition on the forest. The construction and seasonal use of an additional 100 plus miles (new construction and reopening old road beds) of an OHV trail system and putting in new stream crossings will dramatically increase the risk of further changing the

hydrologic connectivity, adding sediment, and further altering aquatic sediment. These are not minor short term impacts but large scale, long term impacts that will be added to a forest that has severely degraded streams, riparian areas and wetlands.

B. The ONF justifies using only “open” roads from the Motor Vehicle Use Maps as the basis for analyzing road and trail densities. Many, if not most closed roads are driven by the public throughout the recreation season despite the TMP (2011) decision on the ONF.

Motor Vehicle Use Maps (MVUM) are for the public to use to travel on open roads identified in the 2011 Travel Management Plan (TMP), and are just that, maps. They identify for the forest user the “open” roads, but maps do not prevent or enforce users from traveling on closed, decommissioned, user-created roads and trails, particularly with the low level of enforcement (1 law enforcement officer for approximately 1 million acres). As a result, many casual forest users unknowingly travel on “closed” roads (i.e., roads that are not designated “open” on the MVUM) because there are no physical barriers or signs indicating that the road is closed. Adding additional confusion to the situation, another map available to the public, the Ochocho National Forest user map (colored half inch to the mile map), shows roads that are open for traffic but are closed via the MVUM map.

For example the following picture is of a road near Little Summit Creek in the Project area that is “closed” on the MVUM map (Figure 5). This road is shown as an open road available for public use on the ONF user map. There is no gate or sign or any evidence that this road is closed. A forest user is supposed to know this road is closed by having their MVUM map with them in the vehicle while driving on forest and making sure that each road they drive is open. It was noted on a site visit in September 2016 that the 400 spur sign was removed and lying broken on the ground, so a casual forest visitor probably would not know the road number as well.



Figure 5. A “Closed” road in the Project area less than 150 feet from Little Summit Creek. No gate or no obstruction is apparent. Only the MVUM map discloses that this is a “closed” road.

Another example of a “closed” road on the MVUM map is signed with a green dot sign that states the road is closed during the hunting season (Figure 6), yet it’s closed year round on the MVUM map.



Figure 6. A “closed” road on the MVUM map with a sign stating that it’s closed from 3 days prior to the deer rifle season through the elk rifle season, but is technically closed year round.

These photos illustrate that there is no consistency between the MVUM maps and what roads are on the landscape. How is the casual forest user supposed to know which it is? They don’t, so people drive these “closed roads”. On the color ½ inch to the mile forest map, there are roads shown on the map that are closed according to the MVUM. A forest user sees these roads on the colored map and is led to believe that they are open when in reality they are closed.

An example that illustrates the lack of objectivity in conducting the analyses in the SFEIS is a comparison of the maps of the Project area. Below are three pictures (Figures 7) of the same area located in T14S, R23E from the north corner of Little Summit Prairie in the lower right part of the picture to the road along Deep Creek on the left side of the picture. The first map is the MVUM map showing only open roads, the second is the “administrative use map” that shows most of the administrative roads on the landscape (sometimes called the firemen’s map), while the third map shows the designated trail overlaid with the “open roads” from the MVUM map.

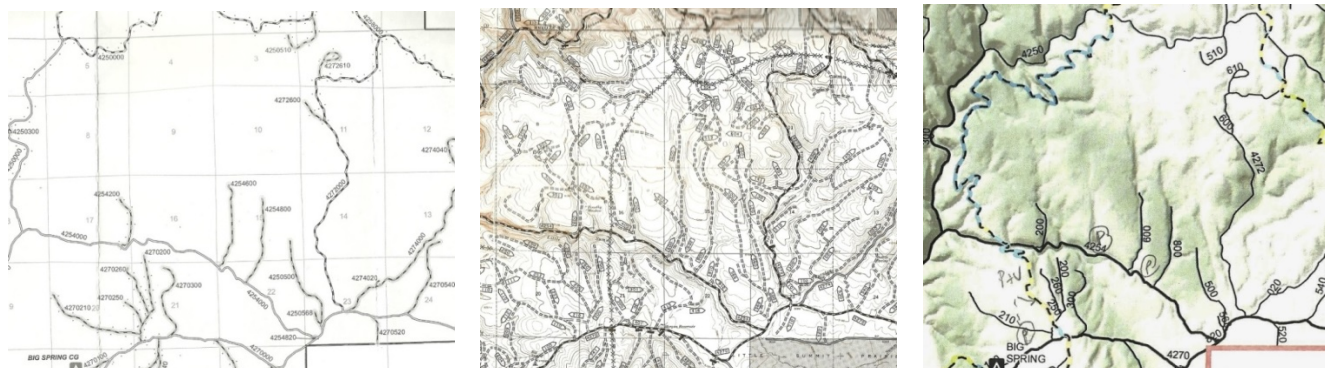


Figure 7-9. Three views of the same area. The first is the MVUM map with “open” roads, the second is the administrative use map with most of the roads, and the third is proposed OHV trail map overlaid on the open road MVUM map.

Figure 7 demonstrates that there are 1) a lot more roads on the landscape than were analyzed (compare sections 9 and 10 that show no roads on the MVUM map while the administrative use map shows many miles, and 2) the ONF chose to build an entirely new trail, designated in blue, along the east boundary on the proposed trail map, causing far more disturbance, rather than using the existing road system, some of which are “closed” or “decommissioned”.

The USFS Aquatic and Riparian Conservation Strategy (ARCS) (USFS 2008a) set objectives for restoring key watersheds:

Developing Objectives for Key Watersheds:

Key watersheds should be the highest priority for active, aquatic and riparian restoration. As part of plan revision, each Forest will identify which key watersheds are priorities for active restoration during the life of the plan. Restoration objectives will be prepared for key watersheds identified as high priority by the Forest plan. These objectives will complement programmatic objectives contained in the Regional Aquatic Restoration Strategy (USDA Forest Service, 2005). They will identify priority treatment activities (e.g. improved fish passage, instream habitat improvement, riparian/floodplain vegetation treatments, road stabilization and/or decommissioning), the extent of treatment areas (miles, acres) and timing for completion within individual key watersheds, and blocking up of federal ownership within key watersheds. The objectives should specifically contribute to the maintenance or restoration of desired conditions for each watershed. Source information for development of objectives include watershed analyses and/or plans for the recovery of fish, water or other riparian dependent plant and animal species. Integration and coordination with other Forest Service resource management programs, state and Federal agencies, tribal governments, local communities, land owners and partners is strongly encouraged.

Then, ARCS cites “standards” for roads and hydroelectric projects (USFS 2008 a, p. 26);

Standard KW-1. There shall be no net increase in the mileage of Forest roads, in any key watershed unless doing so results in a reduction in road-related risk to watershed condition. No net increase means that for each mile of new road constructed, at least one mile of road must be decommissioned to hydrologically stable, self-maintaining conditions. Priority should be given to roads that pose the greatest relative ecological risks to riparian and aquatic ecosystems.

While ARCS may be a strategy document and may not be a regulatory document, it sets a standard that the ONF violates in the Summit Trail Project. In all action alternatives, road/trail densities are increased, leading to the risk of further degradation of riparian areas, wetlands, streams and aquatic habitat.

The Summit OHV Project proposes to rely on roads that are or were closed by other project decisions such as vegetation and fuel treatment projects as “compensation” for high road densities. The SFEIS (p. 73) states that *“With closing and/or decommissioning these roads as authorized in previous NEPA decisions that overlap this analysis area, the road density would be decreased, which would in some areas compensate for the increase in proposed trails”*. But by failing to disclose all of the roads and trails on the landscape including open, closed, decommissioned, temporary, and user-created, the impacts to a variety of resources including streams, fish, and wildlife are severely underestimated. The road pictured in Figure 5 is a “closed” road, so the SFEIS assumes in all action alternatives and analyses that no one drives on closed roads and it no longer affects sediment transport, runoff and stream habitat below, which is patently false.

ODFW conducted a recent survey for a 3 week period in late May and early June of 2016 to identify if either wildlife or vehicles were using the roads. Many of the roads surveyed were “closed” per the MVUM map. The survey revealed that 60% of the transects (a subset of segments of roads) on closed roads (via the MVUM map) had vehicle use (Greg Jackle, district wildlife biologist, pers. comm.). ODFW observations as well as my own observations after working and recreating on this forest since the late 1980s discredit the comment in the SFEIS (p. 57) that *“most roads see little use other than administrative traffic and range permittees through the course of the spring and summer...the bulk of use for the majority of roads comes in the late summer and fall, with the commencement of deer and elk hunting seasons”*. The implication of the comment from the ONF is that hunters are the majority of people that drive on the forest and they are the law breakers driving closed roads when in fact these closed roads are driven throughout the spring, summer and fall by forest users. The public begins driving all the roads on the forest soon after the snow melts and when they can start punching vehicles through the last remaining snow drifts. It starts with turkey hunters and wood cutters in the spring, and then numerous people travel to the forest for day trips or camping all summer long for hiking, fishing, and wildlife and wildflower viewing, and in the fall there are hunting seasons and more wood cutters.

On a personal note, for the past 3 springs, since the TMP was implemented, my family has hunted much of the Rager area for wild turkeys and we regularly see and hear vehicles illegally driving roads allegedly closed by the TMP MVUM maps. When we encountered a man hunting for turkeys via his OHV, we told him he was driving on closed roads. At first, he stated that it was only closed during the green dot closure, and then admitted that he knew he was driving on closed roads and was going to continue to do so. By late May, virtually every closed road we encountered had been driven by various forest users. My family and I also regularly use the forest through the summer and fall for hiking, camping, wood cutting, viewing wildflowers and wildlife, and fishing and continue to see violations of road closures.

In addition, this past September, the Great Old Broads for Wilderness, of which I am a member, conducted a survey of “closed roads” of several areas in the Summit OHV Project area, including the Deep Creek watershed and the Howard Elliot Johnson area north of Big Summit Prairie. While we have not had time to compile the data yet, we noted that most roads had little or no barriers to restrict access and most closed roads had been driven. In addition, we found new roads that were not on the administrative use map, had no signage, and were not connected to any timber operation, further confirming that most “closed” roads are not effectively blocked or enforced, and new unauthorized roads were created by the public. Approximately 80-90% of the “closed” roads we investigated were not physically blocked in any way.

The ONF, by restricting analysis of roads, routes and trails on the landscape, and then assuming that users only use the MVUM “open” roads, fails to analyze the true impacts of these roads and trails to streams, riparian areas, wetlands and associated species. As the above section illustrates, there are roads everywhere on the forest, and forest users use them whether it’s for OHV riding, hunting, woodcutting, or out of just plain ignorance.

C. The ONF uses an unjustified leap of logic that supports implementation of the OHV trail system, despite analyses showing that road/trail densities and road/trail densities within 300 feet of Riparian Habitat Conservation Areas will increase risk of hydrologic connectivity to streams, which further impairs aquatic life.

The SFEIS (p. 58) states that ML1 (closed roads) *“while being maintained at level 1, they are physically closed due to vehicular traffic, but may be open and suitable for non-motorized uses”*. Clearly, most closed roads on the forest and in the Project area, are not physically closed to vehicles and *are* being used by forest users. In several places in the Hydrology and Aquatics section of Chapter 3, the following statement is made (example Lower Deep Creek watershed, SFEIS p. 175) *“Based on Forest Service data layers, the Lower Deep Creek Subwatershed was found to have poor total road densities, but fair open road densities. This indicates that road densities are high, but some of these roads are closed and are producing less sediment than the open road system because they are not currently being driven”*. So the analysis to use only open road densities is not supported by the fact that many closed roads are driven frequently by forest users. The assertion that they are producing less sediment is untrue.

Further, that statement above about closed roads not being driven is contradicted later in the SFEIS (p. 248) that states *“Given the traditional pattern of “open unless designated closed” on this forest, there are currently large areas with moderate to high levels of human disturbance and predator access”*.

In many cases, the SFEIS reports that total (ML 1-5) and open (ML 2-5) road densities within close proximity (300 feet) to stream channels and wetlands are in “poor” condition. For example, the Lower Deep Creek Subwatershed open road densities with the proposed trail were estimated at 4.3 mi/mi² while total road densities (the addition of ML 1 closed roads to the open road estimate) were 5.8 mi/mi², a 1.5 mi/mi² increase with the addition of the OHV trail. Meanwhile, the Lower Deep Creek Subwatershed (SFEIS p. 175), which is part of the Deep Creek “focus” watershed, has aquatic habitat that is described as “poor”... This is especially true for shade, large woody debris, pool frequency and pool quality. According to Table 116 (SFEIS p. 221), Alternative 5 will increase total road/trail densities in the following subwatersheds: Elliot Creek (4.2 mi/mi²), Howard Creek (7.5 mi/mi²), Jackson Creek (5.6 mi/mi²), Little Summit Creek (7.1 mi/mi²), Lower Deep Creek (5.8 mi/mi²), North Wolf Creek (6.5 mi/mi²), Porter Creek (6.3 mi/mi²), and Upper Marks Creek (6.4 mi/mi²). All of these proposed road/trail densities within 300 feet of the RHCA *fail to meet the LRMP road standards and guidelines* for road densities of no greater than 3 mi/mi². They are also 2-3 times the criteria of “high” rating of 2.4 mi/mi² so perhaps these road densities should be called “extremely high”. With these very high road densities, it substantially increases flow and sediment regimes, further impacting aquatic life. The ONF concludes that the OHV trail system will have a minor “short-term impact” despite analyses that shows the ONF is not meeting INFISH standards and guidelines as amended by INFISH, and also violates the described parameters of a “high” rating. Fundamentally, these streams throughout the forest are in terrible shape, and the watersheds are crisscrossed with very high road densities.

After extensive descriptions of poor stream quality and aquatic habitat parameters throughout the Hydrology and Aquatic section for all subwatersheds (SFEIS p. 131-178), the ONF concludes that implementing any of the action

alternatives does not harm aquatic streams, wetlands and aquatic habitat (SFEIS p. 222). The SFEIS (p. 222-223) states that *"The effects to aquatic species are less than what is displayed in Alternative 2 but greater than those displayed for the No Action Alternative and have adequately been displayed in the EIS...Short-term sediment inputs from trail construction, especially at stream crossings, may have negative effects during construction when turbidity is high. These negative effects may include fish displacement during times of high turbidity, limited feeding, and possible mortality"*. Then, the SFEIS asserts that the Alternative addresses some key hydrologic concerns but is overall more beneficial than Alternative 2, but does not compare Alternative 5 with the No Action Alternative. The ONF makes an illogical conclusion that despite the abysmal conditions of the streams, and very high road densities, that the addition of an OHV trail will have little to no impact. Despite acknowledging short-term impacts during trail construction and stream crossings, the ONF relies on "Project Design Criteria" and the false assumption that forest users will only drive on open roads and not drive closed roads to compensate for OHV impacts to soil erosion, sediment and increased hydrologic connectivity. These conclusions defy logic.

Another problem is that the SFEIS makes contradictory statements such as *"Overall, this alternative [no action] would have the least adverse impacts to streambanks across all Ochoco Summit Project Area subwatersheds. Implementation of a cessation of cross-country travel across the Ochoco National Forest would facilitate a more rapid increase than is presently occurring in bank stability and prevent loss of bank stabilizing riparian vegetation"*(SFEIS p. 181). That comment is contradicted by statement later in the SFEIS (p. 219) that says *"Overall, this alternative [alternative 5] would reduce impacts relative to water quality, in particular sediment and turbidity. In the long term, relative to OHV cross-country travel, sediment and turbidity related water quality would be maintained or slightly improved through an overall reduction in the number of stream and wetland crossing locations, improved stream crossing types, improved trail drainage, reduction in the miles of trail in the vicinity of perennial and intermittent streams, and anticipated decreased peak flow magnitudes"*. The SFEIS makes these contradictory statements about impacts of roads and trails on streams, riparian areas, wetlands and aquatic habitat and then again, leaps to conclusions that are irrational.

In the section Action Alternatives, Flow and Sediment Regime, the ONF uses its analyses using only open road densities and combining subwatersheds into watersheds to dilute the impact of the high road/trail densities at the subwatershed level (SFEIS Table 122, p. 232). The case is made that cumulative road/trail densities are below the 3 mi/mi² LRMP standard and guideline with the implementation of MVUM maps, and therefore justifies the OHV trail system. This is a great example of the ONF obfuscating the data by throwing out relevant total road densities counting on vegetation and fuels treatment projects to close roads, which may take 10-20 or longer to occur (if ever), and relies on the presumption that forest users will stay off of closed roads. For example, the Canyon Fuels and Vegetation Management Project (FEIS 2010), HEJ Fuels and Vegetation Management (FEIS 2011) and Jackson Fuels and Vegetation Management (2012) are all predicted to take up to 10 years to complete (SFEIS p. 226-227). The failure to use closed and decommissioned roads and user-created trails in the analyses, and the assumption that forest users don't drive closed roads are an arbitrary and capricious thought process and inappropriately leaves out important information to justify the conclusion to choose an action alternative.

D. The ONF has inconsistently applied recent decision documents for land management activities to support conclusions to implement the Summit Trail system. The results are illogical and capricious since the public cannot discern what the intent of the multiple decision documents are, and what the significances are to

streams, road densities, crossings and therefore, subsequent impacts to hydrology, sediment, and aquatic habitats and resources.

There are numerous decision documents from the past 16 years and into the future that are described in the SFEIS (p. 223-230) that cumulatively could impact flow and sediment to the Summit OHV Project (SFEIS p. 231). These are described in Table 7 (SFEIS p. 53) and in the Cumulative Effects for Hydrology and Aquatic Species description of future and ongoing projects (SFEIS p. 223-230). They include a variety of projects such as Howard Elliot Johnson (FEIS 2011), Canyon (FEIS 2010), Jackson (FEIS 2012), Spears (FEIS 2007), Wolf (FEIS 2014), and Gap (2016 FEIS) Fuels and Vegetation Treatment projects, the 2011 Travel Management Plan, the Deep Creek Restoration Project (Environmental Assessment 2004), recent fires (Maxwell 2006, Bailey Butte 2014, Ochoco Complex 2014), allotment management plans, and other projects. The assumption in the SFEIS is made that some of these projects, such as vegetation and fuels treatment, over time, will compensate for the OHV trail and improves watersheds by closing some roads, despite the fact that many of the same projects are treating acres with commercial harvest, pre-commercial harvest and prescribed fire. These same projects will require new road construction, new temporary road construction and reconstruction, building temporary roads on existing disturbance and in some cases decommissioning or closing roads. The problem is the ONF uses some of the same roads for both open and closed roads in multiple decision documents.

A good example is the Howard Elliot Johnson Fuels and Vegetation Management Project (HEJ) (FEIS May 2011). The following figure (Figure 8) shows road status changes under the HEJ Selected Alternative with roads highlighted in yellow and red, as roads that are to be decommissioned and closed, respectively. Many of these were the same roads that were recommended from a Howard Elliot Johnson Porter Roads analysis (2010), which was used as the basis for road status changes for the Project.

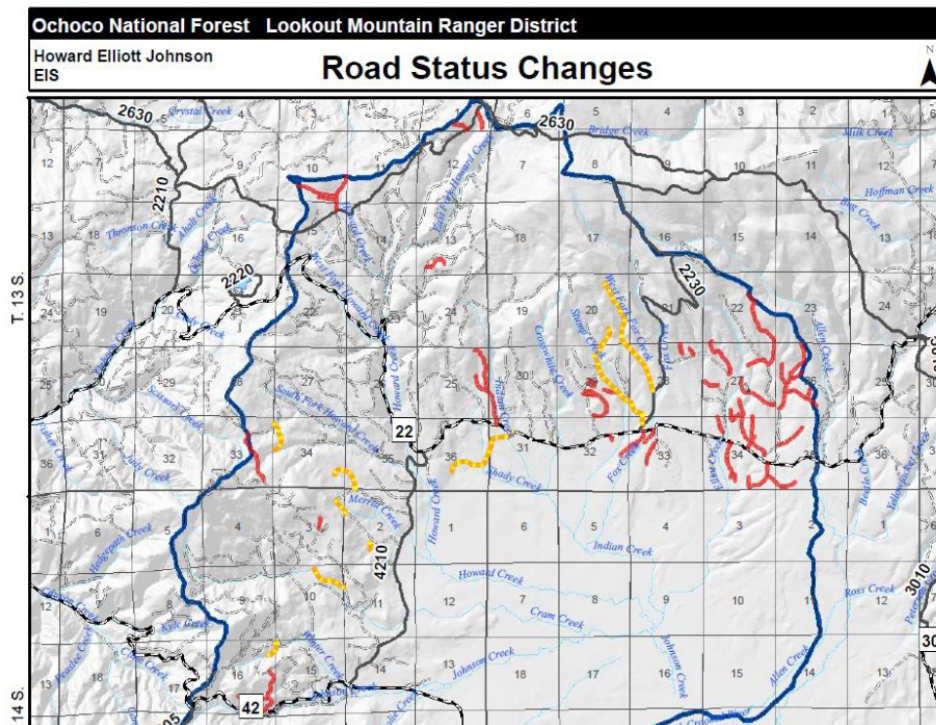


Figure 8. Roads highlighted in yellow and red are proposed for decommissioning and closing in the HEJ FEIS (May 2011 decision).

Subsequent to the HEJ FEIS in May 2011, the TMP FEIS decision was released in August 2011, only 3 months later. Many of these same roads under the 2011 TMP were then left open on all MVUM maps issued from 2012 to 215 (Figure 9). The first decision document intended to close the roads and then was contradicted by a second decision only 3 months later to leave those same roads open.

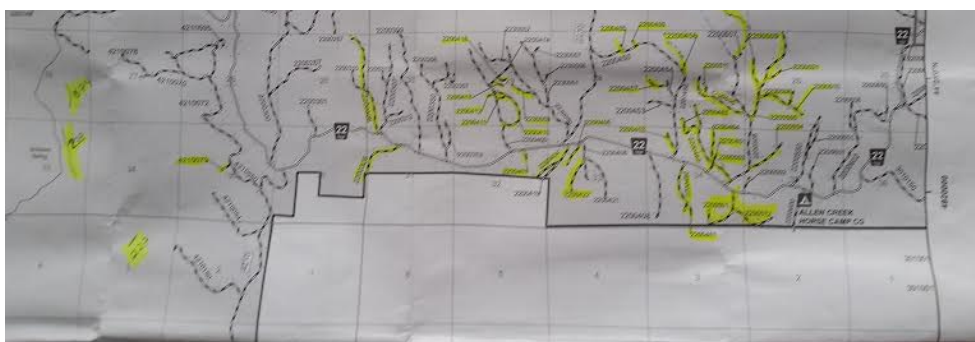


Figure 9. HEJ roads open in the Travel Management Plan MVUM. Roads highlighted in yellow are proposed for closing in the Summit OHV Project.

Then, 5 years later in the Summit OHV Project as a potential cumulative effect, the Summit OHV identifies many of the same roads to be closed or decommissioned in order to move the area toward meeting the LRMP road density of 3 mi/m² (see Table 1). These alleged road closures/decommissioning are described as “compensation” in the SFEIS (p. 73) “With closing and/or decommissioning these roads as authorized in previous NEPA decisions that overlap this analysis area, the road density would be decreased, which would in some areas compensate for the increase in proposed trails.” Tables 2 and 3 are lists of roads proposed for decommissioning or closing, respectively, as “compensation” in the Summit Trails Project in order to reduce open road densities below the LRMP standard and guideline. The tables were taken from Tables 10 and 12 from the Summit Trail Project SFEIS (p. 61-62). Most of the roads that are proposed for decommissioning and closing in the Summit Project are the same roads highlighted in the map for Road Status Changes (Figure 12 above) are for closure/decommissioning under the April 2011 HEJ FEIS.

Table 2. Roads proposed for decommissioning under Howard Johnson Elliot and Porter Creek Subwatershed Roads Analysis (from Table 10 (SFEIS, p. 61).

Creek	Road Number	Segment Length (miles)	Current Maintenance Level	Reason	Recommended Maintenance Level
Elliot Creek	2200-308	0.25	2	Wildlife	Decommission

Elliot Creek	2200-511	0.14	2	Aquatic/Wildlife	Decommission
Johnson Creek	4200-170	0.29	2	Cercla – Amity Mine	Decommission
Johnson Creek	4200-203	0.24	2	Botany/Wildlife	Decommission
Porter Creek	4200-451	0.10	2	Botany/Wildlife	Decommission
Porter Creek	4200-423	0.27	2	Botany/Wildlife	Decommission
Porter Creek	4200-463	0.66	2	Botany/Wildlife	Decommission
Howard Creek	4210-079	0.10	2	Wildlife	Decommission
Total		2.05			

Table 3. Roads proposed to change from Operation Maintenance Level 2 to Operational Maintenance Level 1 – closed under the Howard Johnson Elliot and Porter Subwatersheds Road Analysis (from Table 12 (SFEIS, p. 61-62)).

Creek	Road Number	Segment Length (miles)	Current Maintenance Level	Reason	Recommended Maintenance Level
Howard Creek	2200-274	.35	1	Aquatic	1
Elliot Creek	2200-306	1.2	2	Aquatic, Wildlife/Botany	1
Elliot Creek	2200-401	0.08	2	Botany	1
Elliot Creek	2200-404	0.5	2	Botany	1
Elliot Creek	2200-405	0.2	2	Botany	1
Elliot Creek	2200-410	0.68	2	Wildlife	1
Elliot Creek	2200-411	0.13	2	Wildlife	1
Elliot Creek	2200-412	0.04	2	Wildlife	1
Elliot Creek	2200-417	0.11	2	Wildlife	1
Elliot Creek	2200-418	0.10	2	Wildlife	1
Elliot Creek	2200-422	0.34	2	Botany	1
Elliot Creek	2200-451	0.22	2	Aquatic	1
Elliot Creek	2200-452	0.24	2	Botany	1
Elliot Creek	2200-455	0.21	2	Aquatic	1 (2 segments)
Elliot Creek	2200-457	0.21	2	Aquatic	1
Elliot Creek	2200-458	0.70	2	Wildlife/Aquatics	1 (2 segments)
Elliot Creek	2200-459	0.10	2	Wildlife/Aquatics	1 (2 segments)
Elliot Creek	2200-460	0.54	2	Wildlife/Aquatics	1
Elliot Creek	2200-461	0.13	2	Aquatic/Botany	1
Elliot Creek	2200-462	0.66	2	Aquatic/Botany	1
Elliot Creek	2200-463	0.36	2	Wildlife/Botany	1
Elliot Creek	2200-464	0.33	2	Aquatic	1
Elliot Creek	2200-500	0.30	2	Wildlife	1
Elliot Creek	2200-501	0.63	2	Wildlife/Botany	1
Elliot Creek	2200-503	0.27	2	Botany	1
Elliot Creek	2200-504	0.42	2	Aquatic	1
Elliot Creek	2200-505	1.04	2	Aquatic	1
Elliot Creek	2200-506	0.70	2	Aquatic	1
Elliot Creek	2200-507	1.18	2	Aquatic/Wildlife	1 (2 segments)
Elliot Creek	2200-508	0.49	2	Aquatic	1
Elliot Creek	2200-509	0.32	2	Aquatic/Wildlife	1

Elliot Creek	2200-510	0.10	2	Wildlife	1
Elliot Creek	2200-512	0.48	2	Wildlife	1
Elliot Creek	2200-521	0.18	2	Wildlife	1
Elliot Creek	2230-058	0.21	2	Wildlife	1
Howard Creek	2630-358	0.52	2	Botany	1
Howard Creek	2630-368	0.25	2	Botany	1
Howard Creek	2630-369	0.56	2	Botany	1
Howard Creek	2630-403	0.24	2	Wildlife	1
Howard Creek	2630-408	0.25	2	Wildlife	1
Porter Creek	3000-550	4.78	2	Aquatic	Trail
Porter Creek	3010-095	0.05	2	Wildlife	1
Johnson Creek	4200-153	0.84	2	Aquatic	1
Johnson Creek	4200-175	0.46	2	Aquatic	1
Johnson Creek	4200-227	0.16	2	Wildlife	1
Howard Creek	4200-233	0.56	2	Aquatic	1
Total		22.46			

Essentially, the roads highlighted in yellow in the 2 previous tables are slated for closure/decommissioning in the HEJ project, but are identified as “open” in the MVUM. To further confuse the reader, these same roads are now proposed for closing to compensate for the addition of the Summit OHV trails. Roads highlighted in yellow in the 2 previous tables are the roads proposed for decision documents. The road highlighted in green is in the Porter Creek subwatershed, is not part of the HEJ project, is intended to be a trail in the Summit Trail system but is counted as a closed road in the mileage of the HEJ table to reduce road densities in the Howard Elliot and Johnson subwatersheds.

The SFEIS further noted that (SFEIS p. 64) Road Analyses are not decisions, but only provide recommendations. The Ochoco Summit Project states that it “*does not propose to close any roads regardless of road analysis recommendations*” (SFEIS p. 64). But in fact, this is a defacto claim for compensation for the Summit Trail Project for intended road closures under the HEJ FEIS decision as it further states in the SFEIS (p. 73) that “With closing and/or decommissioning these roads as authorized in previous NEPA decisions that overlap this analysis area, the road density would be decreased, which would in some areas compensate for the increase in proposed trails.”

Clearly, the ONF is muddying what is going on with roads. The significance of this road closing and decommissioning, then opening via the MVUM maps, then closing/decommissioning exercise via the Summit OHV Project, is that there are 3 different NEPA decision documents with different conclusions on whether these roads are supposed to be open or closed. How is the public supposed to know whether these roads are open, closed or decommissioned? The ONF cannot legitimately claim these roads for lowering road densities in order to “compensate” for the OHV trail.

I did this same exercise for the Deep Creek Watershed, looking at other decision documents and found the same inconsistencies in designating open, closed and decommissioned roads. The ONF has no believability in identifying open or closed roads, because there is no consistency between multiple NEPA decision documents. The ONF cannot convincingly discount closed or decommissioned roads from other projects to compensate for

the Summit Project, especially when they are open on the MVUM maps. The ONF is inconsistent in its analyses on the landscape and inconsistent in multiple decision documents.

Due to impacts of high road densities and user-created trails the ONF cannot objectively evaluate and account for all impacts to the hydrology and aquatic resources. They cannot adequately assess impacts to flow and sediment regimes, as well as impacts to streams, riparian areas, wetlands and aquatic habitat. Further, the ONF cannot justify using roads that were intended for closure/decommissioning in the HEJ FEIS, then designated as open roads on the MVUM, and then justify these same roads for closure/decommissioning to add the Summit Trail system. In all cases, this smokescreen of the roads analysis leads to underestimating impacts to hydrologic connectivity, runoff, erosion, sediment and subsequent alteration of stream flows and sediments, which in turn, lead to significant degradation of water quantity, water quality and aquatic habitat.

III. OHV Impacts on Soils and Erosion, Sediment, Hydrology, Channel Morphology, Riparian Areas and Wetlands and Fish.

The following discussion summarizes the impacts of OHVs on soils and erosion, sedimentation, hydrologic connectivity, and resultant impacts to channel effects and aquatic habitat.

Impacts to Soils and Erosion

OHVs weigh several hundred pounds and compress and compact soils, and thereby reduces soil absorption, its ability to retain soil moisture (Brooks and Lair 2005; Jones 1999; Wilshire and Nakata 1976), decreases soil fertility by harming microscopic organisms that provide nutrients and increases overland surface flow across the ground (Sack and da Luz 2003, Meadows et al. 2008). Increased compaction of soils causes decreased soil permeability, resulting in high surface flows across the ground and reduced absorption into the soil. Increased flows concentrate water and increases erosion (Wilshire et al. 1978; Webb 1983).

Webb (1983) used a “dirt bike” to evaluate soil conditions after multiple passes over a test plot. One, ten, 100, and 200 passes were conducted to evaluate impacts of each level of use. The author noted that that compaction usually occurred just below the surface and ranged up to one meter deep. Multiple passes showed that compaction increased and infiltration decreased with increased number of passes. At the beginning of the experiment, tire tracks were observed after just one pass, while most annual vegetation was removed after ten passes. Berms of soils formed at the trail sides by the 100- and 200-passes and the center was rutted.

In a controlled study on ATV effects on soil disturbance on 7 different national forests (Meadows et al. 2008), the authors reported that “some portion of the trail transitioned from low to medium disturbance class in 20 to 40 passes. While the number of days required to achieve this number of passes varies from location to location, this level of ATV traffic could be achieved in one weekend from a moderate-sized ATV group”. Similarly, all seven forests had test sites that some portion of the trails transitioned from medium to high disturbance in 40 to 120 passes, which the authors estimated would occur in less than a month, depending upon trail usage.

The knobby tires that help OHVs climb hills increase direct erosion of soils (Switalski and Jones 2012). The surface flow increases soil erosion and adds sediment to streams. When an OHV enters a stream crossing, no matter the size of the stream, from ephemeral to perennial flow, there is mechanical removal of soils from the bank that are

transported into the stream whenever there is flow. OHV use on trails is a significant source of fine sediment in streams (Chin et al. 2004, Ayala et al. 2005).

The following is a summary of the impacts of OHVs to soil and erosion from Switalski and Jones (2012): In areas with OHV use, OHVs primarily accelerate soil erosion and remove vegetation, and secondarily increase precipitation runoff of precipitation. OHVs also produce large amounts of dust and airborne particles and lead to wind erosion and air quality problems. The knobby, cup-shaped tires that help OHVs climb slopes cause major erosion and soil loss. As the tires dig into the soil, the mechanical forces far greater than the soil strength result in a “rooster tail” of soil and plants thrown behind the OHV. In an Ohio forest, Sack and da Luz (2003) measured erosion soil losses in high-use ORV areas as high as 209 kg/m² while Meadows et al. (2008) reported that ATV trails on Forest Service lands produced an average of 10 times more sediment than undisturbed soils. Foltz (2006) demonstrated in an experiment that sediment loss increased with increased ORV traffic, with the greatest losses when trails were wet.

Mechanical erosion on steep slopes can cause soft soils to erode as much as 40 tons/mi. (Wilshire 1992; reference from Switalski and Jones 2008). On moderate slopes, erosion rates on OHV trails can surpass natural erosion rates by 10 to 20 fold (Iverson et al. 1981, Hinckley et al. 1983; both referenced in Brooks and Lair 2005), while trails on steep slopes can completely remove the entire soil layer, exposing bedrock. Several authors measured erosion losses in high use ORV areas. They ranged from 1.4-242 lbs./ft.² (Wilshire 1977) to 102-614 lbs./ft.² (Webb et al. 1978; referenced in Switalski and Jones 2008), while Sack and da Luz (2003) reported soil erosion losses of more than 200 lbs. of soil for every 100 ft. of trail each year.

(Schubert and Associates 1999; referenced in Switalski and Jones 2008) analyzed more than 500 soils at more than 200 sites and found that essentially all types of soils are susceptible to OHV impairment. Clay-rich soils, while less susceptible to direct mechanical dislodgment by OHVs have greater rates of erosion than most other soil types. When compacted, soils with high clay content produce a tough surface seal that intensifies runoff and gully erosion. Conversely, sand and gravel soils are more subject to direct excavation by OHVs, and when stripped of vegetation are more prone to rapid erosion by rilling and gully erosion.

Sediment

Several recent studies have documented ORV use on trails to be a significant source of fine sediment in streams (Chin et al. 2004, Ayala et al. 2005, and Welsh 2008).

Foltz (2006) concluded from his study of ATV impacts on sediment production that there was no statistical difference between levels of disturbance, in other words, a trail section was either undisturbed or disturbed. Marion et al. (2014) found that OHV impacts were spread downstream of stream channel crossings. Most fine sediment was deposited in the immediate vicinity of stream crossings, but deposits of fine sediment were found at least 100 to 200 m downstream, sometimes continuing into receiving streams. Reidel (2006) studied designated OHV trails and found that suspended sediment concentrations and transport on OHV-impacted streams was many times greater than the control stream. Exiting the stream, OHV wheels spin into the bank causing erosion and further increasing sediment loading.

Meadows et al. (2008) documented that *any* level of disturbance by OHVs increased runoff and sediment by 56% and 625%, respectively, compared to an undisturbed forest floor. Using Foltz' data (Foltz 2006), Tom Davis, a central Oregon retired hydrologist and water resources engineer, took the high values of erosion observed on six national forests from the Missouri westward, and graphed the results, demonstrating the high impact of OHVs on disturbed forest lands (Figure 10) (T. Davis, retired hydrologist, pers. comm.):

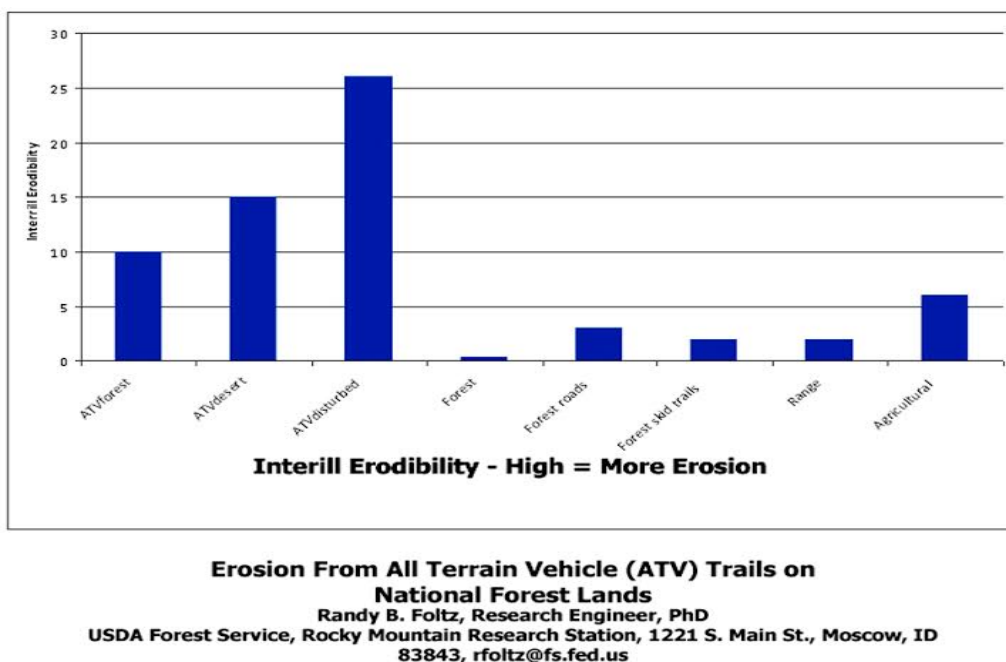


Figure 10. Comparison of interill erodibility on forest lands. Interill erodibility is a measure of raindrop soil erosion with a unit of measure as kg s m^{-4} . High values are for the combined observations of 6 national forests. Higher erosion values occurred on ATV-disturbed sites compared to sites without ATV use (Foltz 2006).

In addition, sediment delivery to streams can occur long after the original impact, and even up to 100 years after the original impact occurred. The following is a statement to Tom Davis, PE; from a Rocky Mountain Research Station Team Leader (Trimble 2009), US Forest Service:

"Sediment coming into and going out of a stream system is cited in Stan Trimble's paper: <http://www.sciencemag.org/cgi/content/full/285/5431/1244>. This is a classic paper about sediment processes, sediment routing equilibrium and massive deposition. Even though it is information from an agriculture watershed in Wisconsin, the principles are the same. When there is a large risk of erosion in a watershed, like a wildfire or logging practices 100 years ago, massive amounts of sediment get deposited in the watershed. Caspar Creek in N. CA is a forest example of this when redwoods were first removed a century ago. It is still delivering unusually large amounts of sediment even though the forest is now regrown. In forests, roads tend to buck the natural trend by delivering small amounts of sediment EVERY year, rather than the sediment pulses that characterize the disturbance driven forested ecosystems."

Hydrology and Channel Morphology

OHVs cause serious damage to streams by increasing erosion and essentially creating new “channels” of bare dirt that deliver runoff and sediment (Switalski and Jones 2008, Foltz 2006, Meadows et al. 2008). The most serious impact is in the immediate vicinity of stream crossings but the impacts can occur from streamside to hundreds of feet upslope from streams. As riparian vegetation is removed by vehicles, plant root networks are destroyed, bare soils are exposed and eroded, and the hydrologic stream network increases (Meadows et al. 2008). Subsequently, stream channels widen and sediment is deposited into stream reaches below the points of delivery. Road networks and stream crossings increase the drainage or hydrologic network and erosion, while the downstream effects are sediment and alteration of peak flow event (Forman and Alexander 1998). Even if OHVs are not directly crossing streams, sediment is delivered to the streams by the increased network of delivery caused by the OHV trails.

Stream crossing cause changes in channel width and slope both upstream and downstream of the crossings (Heede 1980). These alterations in channel morphology can continue for long period of time. Roads [and by virtue of similar mechanics] concentrate and divert flow in headwater areas and cause incisions in areas of the landscape not previously channelized (Montgomery 1994).

Recent research demonstrated that on a designated trail system that had been in place since 1992 OHVs crossings significantly widen stream channels by obliterating the banks and typically the channel widths of the cross section immediately upstream of the OHV crossings (Marion et al. 2014). Overall, stream crossings ranged from 1.15 to 5.45 times wider than the upstream section. A third of the sites also had increases or very large increases in downstream bank erosion. Analysis of the fords showed geomorphic impacts resulting from OHV use including channel erosion and widening,

Riparian Areas and Wetlands

Some ecosystems such as riparian areas and wetlands are rare and disproportionally ecologically and socially important (Switalski and Jones 2008). These special ecosystems are especially important for management since they are more sensitive and more vulnerable to destruction from OHVs, and/or because their impairment is more important by virtue of their scarcity. While riparian areas and wetlands make up just a small percentage of forestlands, they are generally more productive in plant and animal biomass and higher in species biodiversity than the surrounding areas. Good riparian habitat conditions result in cool water, and directly influence instream habitat by maintaining stable streambanks, good water quality, and late season flows (Stuart et al. 1996).

Wetlands also serve important ecosystem functions, such as improving water quality by filtering sediments, and nutrients and contaminants from the water column (Environmental Protection Agency 2002). Wetlands are also critically essential habitats for a wide variety of plants and animals, including many federally threatened and endangered species, and amphibian populations such as the Columbia spotted frog, a highly aquatic federal species of concern.

Aquatic Habitat

River and streambanks in many places of the Crooked River basin including headwater tributaries on the ONF lack riparian vegetation, cover, and large woody debris and are actively eroding or unstable (Stuart et al. 1996). Effects on fish habitat from loss of riparian vegetation include increased stream temperature, loss of cover, increased erosion, a widening and shallowing of the stream channel, and reduction or loss of perennial flow. Degraded habitat is characterized by increased sediment, increased water temperatures, a decline in pool depth, quality and frequency, reduced large woody material (LWM), increased cutbanks and bank instability, and high width/depth ratios. Water quantity and quality problems, primarily flow reduction or loss, temperature, sedimentation, and turbidity, limit fish distribution and production (Bottom et al. 1985; referenced in Stuart et al. 1996). If an illegal OHV user violates a trail system and crosses perennial or intermittent streams that are fish-bearing, they have the potential to drive over redds (salmonid spawning nests built in the substrate), crush or disrupt incubating eggs and increase sediment transport in the water.

Fine sediments and turbidity reduce spawning habitat, egg survival, and food production of insects and plankton (Stuart et al. 1996). Sedimentation affects quality of fish habitat increasing the amount of fines and causing embeddedness of gravel and substrate. This in turn reduces spawning habitat by making gravel difficult to lift, oxygenation to incubating eggs, and cover for rearing fish. Turbidity affects fish production by reducing production of aquatic insects and plankton, and foraging efficiency of fish. Sediment fills interstitial spaces between rocks in the substrate, reduces habitable area in streams, and when it exceeds 20 percent of the total area on the substrate, can smother fish and frog eggs and increase mortality (Bjornn and Reiser 1991). Increased turbidity reduces the ability of fish to feed on aquatic insects and reduces aquatic insect and plant production.

The additional bedload of both fine sediments and coarser sediment delivered to streams fills in the channel, losing pool depth, channel complexity, and changing the channel profile to a shallow, wide profile (Chin et al 2004; May and Lee 2004). Chin et al. (2004) conducted a study of OHV effects on streams, comparing pool characteristics in two watersheds with OHV trails to those in two control watersheds. The OHV-impacted streams had pools with higher percentages of sand and fine sediments. They also had shallower pools with reduced pool volumes, along with higher turbidity, all of which are signs of degraded habitat quality. The second study used tracers and found that an OHV trail had massive impacts on water quality, sediment yield and stream bed sedimentation, and resultant impacts on the stream ecosystem.

LWM is an important measure of habitat complexity and is often associated with pool formation and bank stability (Cordova 1995; Stuart et al. 2007). It is also an important feature in the physical and biological structure of a forested stream (McDade et al, 1990). In general, higher native trout densities are found in streams with more LWM. LWM adds to stream complexity and stability and provides habitat for a variety of species and age classes in aquatic communities (Lisle 2002).

After flow, high summer water temperatures are probably the second largest factor affecting fish in the Crooked River basin (Stuart et al. 1996). Optimal stream temperatures for rainbow trout are 54° to 64°F. Water temperatures in headwater streams on the ONF often exceed 70°F and have been recorded as high as 83°F, well

in excess of state water quality standards for the Deschutes Basin. High water temperatures result in stress or direct mortality to cold water fish species and increases competition from nongame species such as suckers, chiselmouth, and northern pikeminnow, which can tolerate higher temperatures.

The SFEIS failed to properly analyze impacts caused by OHVs

SFEIS states that *“Implementation of the no action alternative would provide for the greatest improvements to streambank alteration and vegetative conditions in the shortest time frame. There would be no all-terrain vehicles crossing streams, directly impacting streambanks at crossing locations. There would be no further effects to width-to-depth ratio, entrenchment, or sediment yield than what currently exists. In addition, by reducing the density of user-created roads and stream crossings, peak flows would not be affected or would slightly decrease as a result, in turn maintaining or slightly decreasing bank erosion of stream crossings”* (SFEIS p. 180).

All action alternatives, however, increase the number of stream crossings and road/trail densities and claim that while the new trails may temporarily increase sediment production, that in the long term, the implementation of Project Design Criteria (SFEIS p. 29-30 and p. 31-33) and rehabilitation of user-created trails in each action alternative would have the result of reducing sediment delivery to streams. For example, the SFEIS (p. 218) concludes that *“In the short-term with the implementation of Alternative 5, impacts of the addition of trails on the flow and sediment regimes would increase. However, in the long-term, once most user-created trails have revegetated (or are rehabilitated/restored (as prioritized), impacts to watersheds flow and sediment regimes should decrease to levels below the existing condition. Proposed trails in Alternative 5 would increase the overall road/trail density and stream crossing density. Increased road/trail densities have the potential to increase drainage density, resulting in increased connectivity and peak flow magnitudes. However, the trail system, as designed in Alternative 5, would not exacerbate existing water quality issues, because of appropriate design criteria and proper management and maintenance.”*

The ONF failed to adequately compare the no action alternative to the proposed action alternatives that implement varying lengths of the new OHV trail system and fail to disclose the real impacts. The Alternative 5 displays open road and total road densities comparing alternative 5 with alternative 2 (SFEIS Table 114, p. 218), and another table displays miles of existing roads and proposed trails within 300 feet of streams, comparing alternatives 3 and 5 (SFEIS Table 116, p. 221). But there is no comparison of road/trail densities in the action alternatives with the No Action Alternative, so the public cannot discern the original road/trail densities in each subwatershed or in each subwatershed within 300 feet of streams, and therefore cannot compare impacts of each action alternative OHV trail system.

Also, while the Hydrology and Aquatics section compares increases in road and trail densities including ML 1 (closed roads) in Table 16 (SFEIS p. 221), other roads such as user-created trails, temporary roads from vegetation and fuels treatment projects, reconstructed roads, and decommissioned roads are not accounted for in the analyses. These other types of trails and roads such as user-created trails, temporary roads and decommissioned roads have the same effect on the landscape of channeling water and delivering sediment to streams, although magnitudes differ by how actively these roads and trails are driven. The SFEIS acknowledges that an *“Increase in overall subwatershed road/trail densities and stream crossing densities have the potential to increase connectivity between roads, trails and streams resulting in higher peak flows and changes to peak flow timing...Since miles of*

existing user-created trails by subwatershed are unknown, these are not accounted for in road/trail density estimates and likely would affect these results” (SFEIS p. 236-7), but there is no attempt to estimate these impacts.

Then, the SFEIS goes on to state that *“Additionally, a prioritization of unauthorized trails would be stabilized or restored as funds allow, which would be beneficial to aquatic species and water resources” ... “Alternative 5 has the least number of site-specific concerns, and all trails proposed within areas of concern are Class III trail motorcycle trails, which are expected to have the least impact of all three trail classes”* (SFEIS p. 238). There are two flaws in these last two statements. First, the action alternatives are not compared to the No action alternative in any numerical or tabular way, they are just statements telling the public to trust them, and all will be well. In the absence of displaying action alternatives with the no action alternative with numerical information such as total road/trail densities, the public is not allowed to view the real impacts of each action alternative. The second problem is that the ONF claims that public funds will provide the Band-Aids to restore unauthorized user-created trails. The latter is a speculative assumption since grant funding is competitive and grant awards are unpredictable.

The SFEIS (p. 237) makes the assumption that with *“all action alternatives include hydrologically stabilizing and closing unauthorized trails that are not part of the Ochoco Summit trail system, which would help decrease the amount of time for recovery and eliminate sediment and other stream impacts once implemented”*. This is a frivolous assumption contradicted by research and literature (e.g. Meadows et al. 2008). Once a motorized trail is developed, it is irreversible and there will be increased hydrologic connectivity to streams and increased sediment input, no matter how well the trail is constructed.

The SFEIS assumes that user-created trails will become naturally revegetated or actively restored, the latter which is contingent upon uncertain grant funding. The SFEIS states that *“erosion and sediment delivery to the stream channel is mitigated by the design criteria requiring outsloping and hardening approaches to stream crossing, installing rolling dips to disperse water from the trail, by reducing the number of locations where OHVs can cross intermittent and perennial streams, and by reducing the density of trails within the sediment delivery zone (300 feet)”* (SFEIS p. 220). However, Foltz (2006), Meadows et al. (2008) and other authors have demonstrated that while a forest may be able to rehabilitate itself after a few passes by an ATV, multiple passes result in the *permanent* loss of natural rehabilitation ability. Once an OHV trails system is implemented, there will be many miles of new disturbance and impacts to streams and wetlands that may never be mitigated.

The SFEIS also makes the case that *“As the effect of the Travel Management Plan are realized, impacts to Groundwater-Dependent Ecosystems (wetlands) in the form of wetlands, wet meadows, and springs would be reduced even with the designation of the trail network in Alternative 5. Through the implementation of the design criteria, user-created trails across wetlands would be hydrologically stabilized and would naturally revegetate over time, and trails that cross wetlands would have minimal effect”* (SFEIS p. 219). The use of Project Design Criteria and putting puncheon stringers over wetlands to elevate the trails are assumed to resolve the impacts of OHV trails. Rather than choosing to go around these sensitive habitats, the ONF is proposing risky Band-Aids that are assumed to avoid or mitigate the impact.

Switalski and Jones (2008) recommended all OHV trails be required to use Best Management Practices (BMPS). BMPS for OHV trails were that all should be located a minimum distance 300 feet from fish-bearing streams and lakes, 150 feet from permanently flowing non-fish-bearing streams, and 150 feet from ponds, reservoirs, and wetlands greater than one acre – 150 ft.

The SFEIS also incorrectly assumes that with the development of an OHV area, all users will automatically behave themselves and stay on trails. This is refuted by nearby OHV trails systems. For example, on the Crooked River National Grassland (CRNG), where an OHV system has been in place for over 24 years, there are numerous illegal user-created OHV and 4 Wheel drive trails and roads throughout. These have largely not revegetated and have received little and relatively ineffective restoration or rehabilitation for the past 24 years, despite the designation of the CRNG as a motor vehicle use area and the nearby 18 mile Henderson Flat OHV area.

IV. Redband trout is listed as a federal and state sensitive species, is a “management indicator species” (MIS), is at risk in the Crooked River Basin, and will be negatively impacted by the Summit OHV Project. Redband trout populations on the ONF are highly fragmented and appear to be largely declining in abundance on ONF streams. Climate change was not considered as a foreseeable cumulative impact in the effects analysis.

FSM 2672.41 is the Forest Service policy implementing the NFMA and establishes objectives and direction to ensure that actions on National Forest System (NFS) lands do not contribute to trends toward Federal listing or loss of viability of any native or desired non-native species.

Redband trout are on the Region 6 Regional Forester sensitive species list (SFEIS p. 124). They are also a “management indicator species” (MIS) for the ONF by virtue of their being a subspecies of rainbow trout (*Oncorhynchus mykiss*), which is a management indicator species (MIS). They are called MIS because when the LRMP was written, *“a group of wildlife species were identified as Management Indicator Species (MIS). These species were selected because their well-being could be used as an indicator of other species dependent upon similar habitat conditions. Indicator species can be used to assess the impacts of management actions on a wide range of other wildlife with similar habitat requirements”* (SFEIS p. 274). Their designation gave them essentially a “bellwether” status for other aquatic life in the basin.

The SFEIS (p.124) failed to state that redband trout is a MIS because apparently they are unaware that redband trout are a subspecies of rainbow trout (Ashbrook et al. 2009, Rhew 2007; Thurow et al. 2007). Redband trout are the native resident rainbow trout in the Columbia Basin east of the Pacific Northwest’s Cascade Crest and are typically referred to as redband trout or inland trout (*O. mykiss gairdneri*). There were three petitions in the mid-1990s to list redband trout under the Endangered Species Act but with changes in procedures for listing, redband trout were dropped from consideration but are still regarded as a species of concern under the ESA (Rhew 2007). The principal reasons for concern are habitat loss, fragmentation of current habitat, isolation of existing populations, and hybridization with coastal rainbow trout and cutthroat trout. Recently, in 2014, a Conservation Agreement was signed by 6 states, 4 federal agencies (including the US Forest Service), one non-governmental organization and 5 tribal governments (California Department of Fish and Wildlife (CDFW) et al. 2014). The intent of the Conservation Agreement is to protect and conserve native redband trout, reduce potential threats to the species, and potentially preclude future needs for listing under the ESA as threatened or endangered.

While the species as a whole is sensitive, Crooked River redband trout are particularly sensitive to further habitat degradation due to loss of functional habitat throughout much of the Crooked River Basin (Stuart et al. 2007). Because redband trout are extremely depressed or absent in streams on much of the private lands in the Crooked River Basin due to irrigation withdrawal and fragmentation of fish populations by dams and impoundments, the remaining populations on the forest are the last stronghold for the species survival in the basin for native redband trout, despite generally poor to fair habitat conditions. Stuart et al. (2007) found that most Crooked River redband trout populations are highly fragmented, many of which are isolated from each other. Streams with low population abundance, marginal habitat and low exchange of genetic material have a high risk of extinction (Rieman and McIntyre 1993).

With the adoption of the Wild Fish Management Policy by the Oregon Department of Fish and Wildlife (ODFW) in 1992, which was later revised to the Oregon Native Fish Conservation Policy (2002), managers discontinued stocking hatchery fish in Marks Creek in 1992, the last stream that had a hatchery program on the ONF. The new management direction was to emphasize the protection and restoration of redband trout and native fish habitat. Surveys conducted jointly by ODFW and ONF personnel indicated that while redband trout were present in watersheds or areas of watersheds that represented 45% of the Crooked River basin, relatively strong populations were located in only 7% of the basin, primarily in small headwater streams on ONF lands and in short reaches of the mainstem Crooked River below Bowman Dam and above Lake Billy Chinook (Stuart et al. 2007). However, habitat conditions on ONF lands indicated that many streams were in poor condition with high summer water temperatures, suboptimal shading and lack of large wood, due to past and present land management activities.

Stuart et al. (2007) reported that many streams on the ONF had high temperatures. Their data indicated that redband trout are highly susceptible to increasing water temperatures. Figure 11 (from Stuart et al. 2007, Figure 7) demonstrates the swift decline in redband trout abundance as stream temperatures increase, one of the parameters for redband survival and persistence.

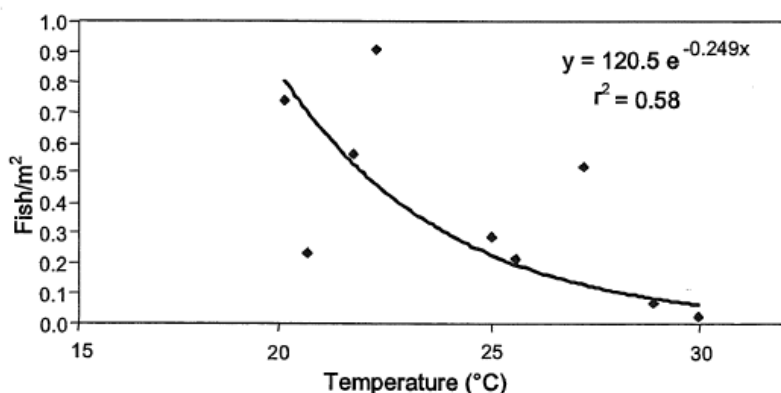


FIGURE 7.—Relationship between density (fish/m²) of redband trout (all age classes) and maximum water temperatures (°C) in small streams of the Crooked River basin, 1994–1995.

Figure 11. Relative abundance of redband trout declines many fold as stream temperatures increase (from Stuart et al. 2007).

Changes in stream temperatures have been implicated with the construction and maintenance of roads, which by inference includes OHV trails since they have the same mechanism for impacts. Williams (1999) reported that “Roads directly affect natural sediment and hydrologic regimes by altering streamflow, sediment loading, sediment transport and deposition, channel morphology, channel stability, substrate composition, stream temperatures, water quality, and riparian conditions within a watershed” (Lee et al. 1997; referenced in Williams 1999). So, one mechanism to potentially affect temperatures is by interrupting hillslope drainage patterns. By intercepting surface and subsurface flow, and increasing the channels on the landscape, it alters the magnitude of peak flows and lowers base stream discharge (Furniss et al. 1991). Another mechanism that could alter stream temperatures is caused by loss of riparian cover (e.g. for OHV trail clearing across streams and through riparian areas). A third mechanism that has been explored recently is by conversion of groundwater to surface water (Williams 1999). When water is changed from subsurface (e.g. rheic to hyporheic flow) to surface water, as streams degrade, it is exposed to solar insolation (Fissekis and Hersh-Burdick 2007; Wonzell et al. 2009). The hyporheic zone is the area under or beside a stream channel or floodplain that contributes water to the stream, and has been found to have a significant role in the supporting salmonid fish populations. Bowerman et al. (2014) reported that fine sediments reduced percolation of water through salmonid redds (fish nests) and impacted both egg and early life survival. The authors also reported that management actions that restore hydraulic complexity in the form of pool-riffle sequences, point bars, and large wood enhance hyporheic exchange and support higher salmonid survival. May and Lee (2004) also stated that “increases in coarse sediment supply can be associated with sharp reductions in salmonid habitat and productivity. Land-use practices, such as timber harvest and road construction, are often associated with large influxes of coarse sediment in forested mountain streams. They also reported that streams with discontinuous flow reduced survival by crowding fish into smaller areas with reduced food availability.

Redband trout abundance surveys were conducted on Brush, Porter and Roba creeks, all located in the proposed Project area, and correlated habitat conditions with fish abundance (Dambacher and Jones 2007). Brush Creek with lower water temperatures, low intensity of cattle grazing, and low intensity of timber harvest had the highest fish densities while Porter and Roba creeks successively had lower fish densities that correlated with higher water temperatures, high levels of livestock grazing and moderate to high levels of timber harvest. Although available habitat for redband trout in the three streams differed by no more than a factor of 1.6, there was a 20-fold difference in the density of fish, and age 1+ fish in Brush Creek were significantly higher.

At the Crooked River basin level, the OHV Project could significantly move redband trout towards a loss of population viability and localized extinction. Virtually the entire Project area is in the Crooked River Basin. Fish populations were sampled by ODFW biologists for upper limits distribution in 1990 and 1991 on many ONF streams due to concern for effects of past and proposed timber harvest projects in riparian areas and along Class I and II streams (maps are available at the ODFW Prineville office). At the time, sampling was conducted to discern fish bearing streams that had greater setbacks, aka “buffers” from timber harvest projects. During the course of the redband trout investigations, it was determined that drought conditions from 1991-1994 inevitably led to the

loss of isolated redband trout populations. Some “perennial” streams on the south side of the Maury Mountains of the ONF went dry. Streams such as Double Cabin Creek that formerly had redband populations went extinct.

With physical barriers on private lands below from irrigation dams, these isolated populations on the forest went extinct because there was no opportunity for fish to reoccupy these habitats (A. Stuart and S. Thiesfeld, pers. comm.). With climate change trending towards warmer and dryer conditions in the Pacific Northwest, “Summer base flows will be lower, and the network of perennially flowing streams in a drainage system will shrink during the summer dry period, forcing fish into smaller wetted channels and less diverse habitats” (Battin et al. 2007). Bisson (2008) and the Independent Scientific Advisory Board (ISAB) (2007) predicted that “Trout and salmon within the interior Columbia River Basin may be especially sensitive to climate change... Although the intensity of the effects will vary spatially, climate change will alter virtually all streams and rivers in the basin. Current predictions suggest that temperature increases alone will render 2 to 7 percent of headwater trout habitat in the Pacific Northwest unsuitable by 2030, 5 to 20 percent by 2060, and 8 to 33 percent by 2090.” Climate change, particularly in the light of already poor stream conditions of high sediment, poor bank stability, low pool frequency and high summer temperatures on ONF lands may cause redband trout to go extinct in the Crooked River basin (USDA 2015d). With highly disconnected habitats between streams, like Double Cabin Creek in the Maury Mountain, there is no opportunity for redband trout to reoccupy habitats once the population has been lost.

Most of the proposed OHV trails will cross highly fragmented and disconnected redband trout populations from Marks Creek on the west end to North Wolf Creek on the east end. Most of the redband trout populations have been disconnected by irrigation dams and diversions on private lands. Even streams that are separated by only a few miles on the landscape such as Crosswhite, Peterson and Porter Creek are completely disconnected and fish cannot move freely between these small populations. The proposed OHV system will further stress these redband trout populations by adding over 100 miles of new trails or reopening of abandoned road beds, which will further change runoff patterns, increase sediment, decrease riparian vegetation, increase water temperature, and compromise the hydrology and stability of the stream systems.

Construction of bridges, culverts, fords and miles of trails both inside and outside of riparian areas will severely impact redband trout throughout the Project area. The cumulative impacts of the extensive road network (open, closed and decommissioned), illegal motorized user trails and livestock grazing already have compromised Deep Creek and its tributaries with high sediment loads, poor riparian conditions and high summer temperatures.

Lee et al. (1998) reported that redband trout were impacted by high road densities. They observed that pool frequency was inversely correlated with road density and land management emphasis. Streams in wilderness and roadless areas generally retained pool habitat in the 50-60 year period from the 1930s to the 1980s and 1990s. They also noted that an important aspect of habitat quality that is fine sediments which were caused by a direct and significant correlation with road densities.

Crooked River redband trout are a subset of the Columbia Basin distinct population segments (DPS) identified (Thurow et al. 2007). Like many of the other DPS, Crooked River redband trout are remnants of what were once larger, more complex, and diverse and connected populations. Redband trout in Crooked River tributaries are in patches scattered in watersheds (Stuart et al. 2007). Most populations are poorly connected and are fragmented by dams, irrigation diversions, temperature and flow barriers, and other migration barriers. Remaining

populations have been progressively isolated into smaller and smaller patches of habitat. Remaining habitat and corridors, such as streams like Deep Creek that have no significant barriers and support redband trout, are critical habitat for migration, rearing, and overwintering and are critical for the conservation of the species.

Further, these remnant populations and habitat are critical for survival of the species in the Crooked River basin with impending climate change. Loss of spatial diversity, population structure, productivity and stability threaten long term persistence of redband trout (Lichatowich and Mobrand 1995; referenced in Thurow et al. 2007). This is particularly true in the Crooked River where few strong populations are left and in highly fragmented habitat.

Deep Creek

A good example of where the OHV trail puts redband trout at risk is Deep Creek and its tributaries located in the heart of the Eastside Implementation Area. It has the largest connected population in the Project area and is the principal tributary of the North Fork Crooked River. Deep Creek itself is one of the last best remaining streams with reasonably fair redband habitat on the ONF. Numerous OHV trails are proposed in the Deep Creek watershed throughout the area and that cross Deep, Little Summit, Jackson and Crazy creeks. The SFEIS (p. 137) states that “The Deep Creek watershed represents one of the most interconnected habitats for trout...Deep Creek and its tributaries provide some of the premier trout fishing opportunities in the region due to its diverse, relatively intact and abundant cold water...” Yet, the FEIS proposes to put a trail across Deep Creek at an area with relatively good habitat and is one of the locations that is popular for fishing. The OHV system will essentially further degrade habitat conditions for native trout in one of the last best places for trout fishing on the forest.

SFEIS (p. 194) states that *“Some of the perennial streams that have been identified by Forest fish biologists and the Oregon Department of Fish and Wildlife to be critical refuges for redband trout in the Deep Creek Watershed are Deep Creek, Little Summit Creek, Happy Camp Creek and Double Corral Creek. However, many of these systems have been impacted by past management and are not as effective as they once were in sustaining redband trout. ODFW estimates that these systems are occupied at approximately 10% of historic levels. In addition to redband trout, Columbia spotted frogs are present throughout the area.”* Yet, most of the watershed is largely in very poor habitat conditions for shade, water temperature, LWM, pool frequency and pool quality (SFEIS p. 173). The OHV trail will increase the total road/trail density to an 5.6 mi/mi² in the Jackson subwatershed, 7.2 mi/mi² in the Little Summit watershed, and 5.8 mi/mi² in the Lower Deep watershed, generally 2 or more times the Forest standard (SFEIS Table 116, p. 221). Meanwhile fish populations in Deep Creek have plummeted since 1997. All 3 reaches surveyed by ODFW from 1997 to 2012 showed 2 to 5 fold declines in abundance (SFEIS p. 171-172) (Figures 12-14).

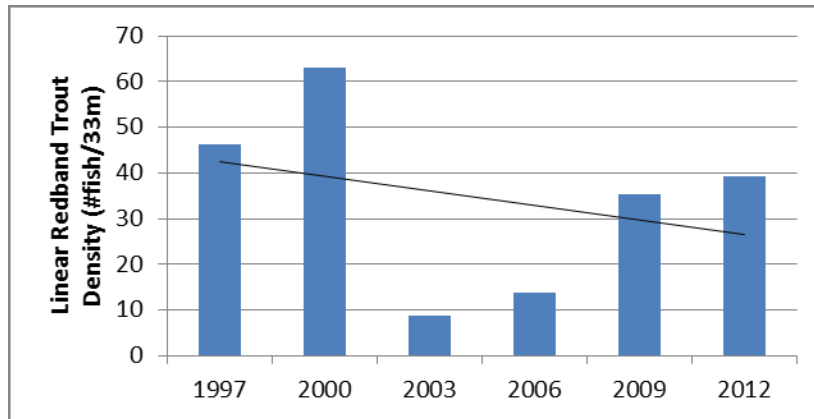


Figure 12. Linear fish densities (fish/33 m) for upper Deep Creek, 1997-2012 in 3 year incremental surveys. From SFEIS Figure 19, SFEIS p. 171.

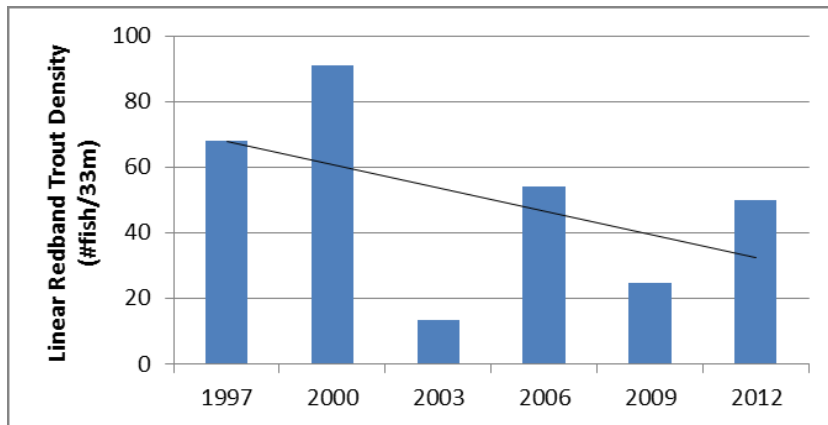


Figure 13. Linear fish densities (fish/33 m) for middle Deep Creek, 1997-2012 in 3 year incremental surveys. From SFEIS Figure 20, SFEIS p. 172.

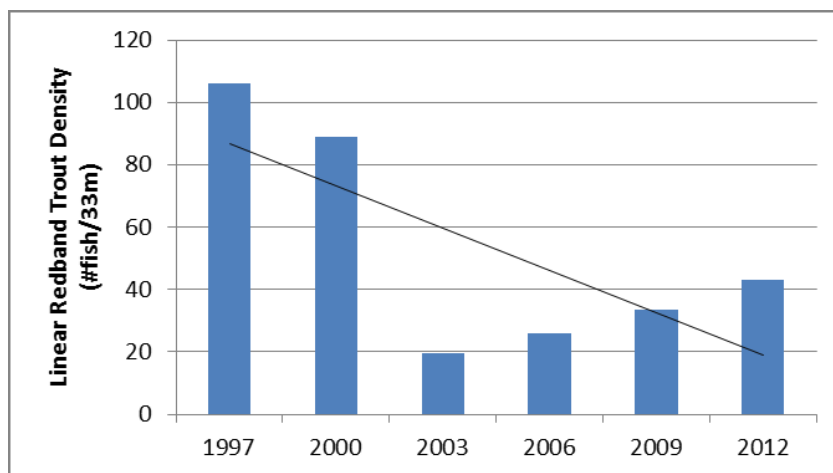


Figure 14. Linear fish densities (fish/33 m) for lower Deep Creek, 1997-2012 in 3 year incremental surveys. From SFEIS

The SFEIS (p. 225) states that “*Deep Creek is a focus watershed as defined by the Forest Service Region 6 Aquatic Restoration Strategy.*” The 2004 Deep Creek Restoration Plan was to implement 1) Replace 35 culverts and 3 culverts with rock fords to allow passage of flood flows and improve upstream access for redband trout; 2) create 7 water developments, 227 acres of grazing enclosures, 342 acres of riparian pasture, and place large wood along 3.6 miles of stream to better distribute livestock; 3) repair 37 headcut complexes and 18 exposed cutback; close, decommission, and reconstruct 47 miles of roads; and reconstruct 0.25 miles of channel reaches to reduce instream sedimentation; and 4) construct 6 in-stream structures to promote pool habitat for fisheries. In 12 years, the only accomplishments to date are installation of 2 gates, 2.8 miles of road decommissioned in the Lower Deep Creek Subwatershed, and 1.2 miles of road decommissioned in Little Summit Prairie Creek Subwatershed. This is a glacial pace of restoration while fish populations are declining precipitously.

V. The proposed OHV Project impedes meeting the standards for the Ochoco LRMP (FEIS p. 115) and goals and objectives for INFISH (INFISH, Appendix E).

The desired future condition when the LRMP was signed in 1989 was that in “ten years individual watersheds on the Forest that are currently in excellent condition are expected to remain so and those not presently in good condition would be given first priority for improvement. Fifty years and beyond, it is expected that 90 to 95% of the riparian areas on the Forest will be in “excellent condition” by the year 2040” (SFEIS p. 123).

Given that 27 years (54% of the time) has elapsed, and the stated expectations for excellent stream conditions by year 2040 are likely to not be met, the new OHV trail system further will inhibit the ONF from meeting INFISH standards in streams that are largely still in poor condition. Most of the streams in the OHV Summit Project area continue to not meet LRMP standards for an “excellent condition” or INFISH standards for sedimentation, pool frequency, large woody material, and temperature.

ONF goals for water in the LRMP are to “maintain or improve water quality, quantity, and timing of run-off” and to comply with the objectives of the “Clean Water Act” and Oregon State water quality standards” as described in the MOU between the Oregon Department of Environmental Quality (ODEQ) and the Forest Service. The long-term objective was to maintain or improve all riparian areas to “excellent condition.” In 1989, it was estimated that approximately ½ of 9,100 of the Forest riparian areas were in excellent condition, although this estimate was largely overestimated and refuted by field sampling (Dambacher and Jones 2007; Stuart et al. 2007). The remaining areas were scheduled for improvement over time. It was intended that a large portion of the needed work to improve streams would occur in the first decade of implementation, although full biological potential could require 20 to 60 years.

Other relevant LRMP standards and guidelines were 1) maintain contacts with Federal, State, and other agencies, groups, and individuals concerned with the management of threatened, endangered, and sensitive species, and 2) consult with the Oregon Department of Fish and Wildlife, Oregon Natural Heritage Data Base and the US Fish and Wildlife Service for technical assistance in developing species management guides and in determining viable

population levels. When suitable habitats or reported locations were suspected to occur in the area of influence of the project, a field reconnaissance would be performed to more precisely verify the presence, abundance, and distribution of the sensitive species.

INFISH (1995) set riparian goals to include: maintain or restore water quality, stream channel integrity and channel processes, and instream flows to support healthy riparian and aquatic habitats. Interim riparian management objectives include bank stability greater than 80%, and interim pool frequency objectives (Appendix E). INFISH stated that “Actions that reduce habitat quality, whether existing conditions are better or worse than objective values, would be inconsistent with the purpose of this interim direction. Without the benchmark provided by measureable RMO [riparian management objectives], habitat suffers a continual erosion” (INFISH Appendix E-3). Standards and guidelines for all Riparian Habitat Conservation Areas (RHCAs) dictate that “where existing roads, facilities, and other improvements found to be causing an unacceptable risk cannot be relocated, eliminated or reconstructed, those improvements would be closed” (INFISH E-6).

The FEIS describes several streams with empirically collected data from the Bottom Line Surveys, Level II and “PIBO” and indicates that many of the streams on the ONF are compromised for sedimentation, habitat and bank stability, and do not meet RHCA objectives. For example, reach 2 of the Elliot Creek subwatershed in the Project Area is described in the 2014 FEIS (p. 132) during the 1996 Bottom Line Survey: “Cattle damage is extreme on the creek. Banks have sloughed off into the channel. Banks are trampled and lack of definition causing heavy sediment to the stream. There is a fence ¼ mile upstream from FR 22 and a cattle crossing that is also causing heavy sediment into the creek due to the heavy cattle traffic through the stream. This area needs restoration to improve fish habitat”. Reach 1 is described as “Cattle impact is extreme in this reach... banks are sloughed and trampled, and there is very minimal hardwood such as alders...Overall, this stream offers fish habitat, yet it is in very poor condition due to grazing and past timber harvests.” This description was left out of the SFEIS.

Field observations in August 2010 for Elliot Creek on Reach 2, 14 years after the original survey, for Elliot Creek reported virtually the same conditions and no improvement. “Streambanks were impacted by grazing with postholing, hedged alder, and removal of riparian vegetation. The cut streambank appears to be sheered by livestock recognized by their hoof prints. The stream is downcut from loss of vegetation...Sediment deposition on the substrate indicates continuous sediment inputs from cut streambanks” (SFEIS p. 147). Clearly, the stream had not improved from surveys conducted in 1996 to 2010. Adding the impact of the proposed Summit OHV system is inconsistent with the requirement of INFISH of **“Actions that reduce habitat quality, whether existing conditions are better or worse than objective values, would be inconsistent with the purpose of this interim direction”** as the Project will further compromise degraded stream habitat.

Streambank cutbanks are another bank stability parameter that empirically demonstrates poor stream condition on ONF lands. Streams with higher levels of bank instability have higher amounts of fine sediment. The ONF LRMP and Region 6 INFISH standards are for less than 20% cutbank or stream instability. For example, bank stability at Peterson Creek where a Class II trail overlaps the FS 2630 road showed 36% cutbanks.

Turbidity and sediment fines from soil erosion impact water quality and fish habitat. A 20% embeddedness or sediment fines on a substrate is a threshold well established in the literature (Bjornn and Reiser 1991, Rhodes et al. 2000, Rowe et al. 2003). The INFISH standard for channels and sediment is to “maintain or restore the stream

channel integrity, channel processes, and the sediment regime (including the elements of timing, volume and character of sediment input and transport) under which the riparian and aquatic ecosystems developed” and “help achieve rates of surface erosion, bank erosion, and channel migration characteristic of those under which the community developed”. Total suspended solids (TSS) and turbidity are key parameters indicative of land disturbance and water quality degradation. ODEQ’s standard for total suspended solids or settleable solids (essentially fine sediments) is a narrative and states that the standard is for the stream not to exceed deposits deleterious to fish or aquatic life or injurious to public health. The turbidity standard is also important because fine particles from soils can clog gills of fish, promote excessive algae growth, and reduce dissolved oxygen in waters. Furthermore, it fills in gravel beds and destroys spawning habitat. The ODEQ’s antidegradation policy (Oregon Administrative Rule 340-041-0004) is designed to protect water from further degradation from new or increased sources of pollution and protects, maintains, and enhances surface water quality to protect existing beneficial uses.

The following table is taken from data from the SFEIS Hydrology and Aquatics section in Chapter 3 (p. 144-177). The fine sediment data is from the 2014 FEIS as it was not provided in the SDEIS or SFEIS. These two important parameters and ODFW fish sampling data reveal the generally poor habitat conditions and declining fish populations on ONF streams (Table 4). Stream temperature and its effects on redband trout were discussed in the previous section IV. Redband Trout. The temperature data is shown because it is a long term and more rigorous data set and is critical for all life stages of native trout. Sediment data is also included in the table because it is another important parameter that affects salmonids, reflects watershed condition and stream productivity. The high level of embeddedness on many ONF streams impacts native fish and aquatic insects, their important food source. Although the ODFW linear fish sampling data was not as rigorous, it provides a fairly accurate picture about the status and trend of native redband trout in sampled reaches. It is noteworthy that of the 14 sites sampled, only 2 showed an increasing trend, while the others trended downward.

Table 4. Comparison of stream temperature and sediment conditions and redband trout abundance on ONF streams.

Stream Name	Average Temperature Rating	Temperature Trend	Fine Sediment Subwatershed Average	Fine Sediment	Fish Transect Trend # reaches/direction
North Wolf	Poor	No Change 1995-2011	39%	Poor	NA
E Fork N. Wolf	Poor (2013)	NA		Poor	NA
Elliot	Poor	No Change 1996-2008	28%	NA	NA
Fox	Poor	No Change 1994-2002		Fair	NA
W Fork Fox	Poor	NA		NA	NA
Stump and tributaries	Poor	NA		NA	NA
Indian and tributaries	Poor	NA		NA	NA
Howard and tributaries	Poor	No Change 1994-2001		NA	2/Down

Allen and tributaries	Poor	No Change 1994-2013		Fair-Poor	1 Up/1 Down
Peterson	Poor	No Change 1994-2001	45%	Fair	NA
Porter	Poor	No Change 1994-2001	22%	Fair	NA
Jackson	Poor	No Change 1992-2010	21%	Fair-Poor	1/Down
Derr	Poor	No Change 1998-2001		Poor-Fair	NA
Happy Camp	Poor	No Change 1998-2005		Mostly Poor	1/Up
Double Corral	Poor	No change 1992-1994		Fair	2/Down
Little Summit	Poor	No Change 1995-2005		Fair	3/Down
Toggle	Poor	NA		Poor	
Chamberlain	Poor	NA		Poor	
Deep Creek	Poor	No Change 1995-2013		Poor	3/Down
Crazy Creek	Poor	No change 1994-2005	21%	Poor	NA
Crazy tributaries	Poor	No Change 1994-1999		Poor-Fair	NA
Marks	Fair	NA	29%	NA	NA
Crystal	Poor-Fair	No Change 1996-2004	29%	Poor	NA

All of the streams measured were greater than the 20% fines standard, exhibited sedimentation violating Oregon's water quality standards, and are already compromised by past and present land use practices with an extensive road network, timber harvest, and livestock grazing. Again, the proposed OHV Project violates the non-degradation standards of INFISH and State of Oregon water quality standards and its implementation will further compromise streambank stability, fish habitat and native fish populations by subjecting these sensitive areas.

The SFEIS (p. 222-223) does not compare Alternative 5 to the No Action Alternative and irrationally concludes that *"Alternative 5 would have no measurable effect on redband trout and Columbia spotted frog long-term in all project subwatersheds. Short-term sediment inputs from trail construction, especially at stream crossings, may have negative effects during construction when turbidity is high. These negative effects may include fish displacement during times of high turbidity, limited feeding, and possible mortality... From an aquatic species standpoint, Alternative 5 addresses some key hydrologic concerns that were brought up in comments and should, overall, be more beneficial to aquatic species than Alternative 2."*

This summary is specious and misleading. The SFEIS disregards the increased hydrologic connectivity created by many miles of the proposed OHV trail system, disregards that most closed and decommissioned roads are still driven, disregards that the analyses does not account for unauthorized user-created trails, disregards the incredibly high road densities in the numerous subwatersheds, disregards the data that show most of the parameters for INFISH standards such as stream temperature and fine sediments violate standards, and disregards that the majority of fish populations sampled showed declining trends in the past 15-20 years. It is lunacy to use this kind of faulty analyses that lead to erroneous conclusions

VI. The FEIS is inconsistent with the requirements of the Forest Service Manual by proposing an OHV trail that will degrade streams, failing to assist the State of Oregon in meeting conservation goals and failing to avoid or minimize impacts.

This Summit OHV system adds significant stream degradation to streams that are already compromised for high stream temperatures, high sediment loads, low woody material, riparian condition and bank stability and has the potential to lead to local population extinctions and loss of species viability in the Crooked River Basin. The following streams and their numerous tributaries were reported as not meeting one or more stream standards for shade, cutbank, pool frequency, and sediment: North Wolf, Deep, Porter, Peterson, Allen, Howard, Elliot, and Marks creeks. Virtually all of the streams in the proposed OHV area have been affected by past and present land management activities that have degraded stream conditions.

FSM Policy 2670.22 - Sensitive Species objectives requires that the USFS: 1. Develop and implement management practices to ensure that species do not become threatened or endangered because of Forest Service actions, 2. Maintain viable populations of all native and desired nonnative wildlife, fish, and plant species in habitats distributed throughout their geographic range on National Forest System lands, and 3. Develop and implement management objectives for populations and/or habitat of sensitive species.

Policy 2670.32 - Sensitive Species requires that the USFS 1. Assist states in achieving their goals for conservation of endemic species... 3. Avoid or minimize impacts to species whose viability has been identified as a concern, 4. Analyze, if impacts cannot be avoided, the significance of potential adverse effects on the population or its habitat within the area of concern and on the species as a whole, and 5. Establish management objectives in cooperation with the states when projects on NFS lands may have a significant effect on sensitive species population numbers or distributions.

The SFEIS proposes to implement the OHV Summit Project and disregarded the results of the Bottom Line, Level and PIBO aquatic surveys conducted that showed streams were already compromised. These included sediment loading, pool frequency, summer stream temperatures, lack of LWM, and bank instability. Somehow, the SFEIS concludes that subwatersheds that violate road/trail density standards and streams that already violate LRMP and INFISH standards will not be furthered impaired by a 137 mile trail system. This view of the world is based on the assertion that 1) grant funding will be used to restore and rehabilitate the numerous user created trails or they will magically heal themselves, 2) all OHV users will magically become responsible, well-behaved users that only drive on the designated trails system, and 3) Project Design Criteria will be the cure all Band-Aids that will mitigate for construction and extensive riding use by upwards of 15,000 riders per year.

The first assertions that user created trails will heal is refuted by Foltz et al. (2009) who found that when abandoned roads were reopened for use, runoff and sedimentation increased dramatically. The authors reported that “Runoff was lowest on the non-track portion of the abandoned road and highest on the reopened road. Sediment concentrations were significantly higher on the reopened road....Thirty years of no traffic and vegetation regrowth was not sufficient to allow recovery of infiltration to values similar to an undisturbed forest.” Similarly, Meadows et al. (2008) reported in experiments on 7 national forests that were tested for designated trail, that freshly disturbed soils on ATV trails produce an average of 10 times more sediment than undisturbed soils.

The second assertion is refuted by the section below on law enforcement (see XI. Law Enforcement). Further, two examples demonstrate continuous examples of OHV and off road violations: 1) the ONF closed a majority of the forest to cross country by the 2011 TMP and the 2) CRNG has a designated motorized travel system with an OHV area (Henderson Flat) that closed cross-country travel and limited seasonal in 1992. Pictures of unauthorized roads and trails abound on the forest and provide visual evidence of the difficulty of the ONF to control illegal use. The following is a picture of a road that doesn’t exist on the CRNG but is clearly being used by vehicles (Figure 15). Figure 16 is a picture of the same area and shows some rocks recently were added at the bottom to block vehicles from using this unauthorized trail, but appear to be largely ineffective.



Figure 15. An unauthorized road on the CRNG, which has had a “wheeled motorized restriction area” in place since the early 1990s. Photo taken in April, 2014.



Figure 16. Both roads beyond the gate are unauthorized roads. Road on the right with boulders is the same road as in Figure 18. Note rocks at bottom to “deter” use on the illegal road on the right, although no rocks are blocking the illegal road on the left. Photo taken October, 2016.

The 2005 Travel Management Rule requires that the Forest Service consider in designating NFS trails and areas on NFS lands, the responsible official shall consider effects on the following, with the objective of minimizing:

- (1) Damage to soil, watershed, vegetation, and other forest resources” (36 CFR 212.55(b)(2)).

As part of the SFEIS analysis, all action alternatives increase impacts to the hydrologic network, soils, riparian and upland vegetation communities, stream habitat and fish populations. The SFEIS gave very marginal consideration to the major alteration of the hydrology of the landscape and adding further impacts to streams already compromised that largely fail to meet criteria for good or excellent condition. The SFEIS proposes to reopen approximately 50 miles of closed or decommissioned roads and construct somewhere between 51 and 58 miles of new trails (it varies depending on which page of the SFEIS one is reading), which further exacerbates fragmentation. The proposed Project is still proposing to add essentially over 100 miles of new disturbance that will add substantially to the road network, and cause all the same impacts that roads cause. This is not an insignificant impact to hydrology, streams and fish, sediment load and fish habitat.

In addition, while the USFS informed ODFW and Oregon Department of Environmental Quality and invited comments, agency goals and standards were disregarded. ODFW recommended the Alternative 1. No Action due to the concern of a variety of impacts that would affect important fish and wildlife species and their habitats, necessary to meet their *management objectives*. ODFW’s information was largely disregarded other than to move a few proposed trails out of RHCAs in Alternative 5.

In essence, the FEIS disregards the standards and guidelines for the FSM and state of Oregon standards for fish, wildlife and water quality protection, and proposes to overlay an OHV system on aquatic habitats that have already been degraded.

VII. The FEIS arbitrarily sets stream crossing densities in riparian, wetland, seeps and springs, and wet meadows with ratings of “low”, “moderate” or “high”. The FEIS marginalizes the impacts of the OHV area by defining the foot print of disturbance as only the trails while the magnitude is much larger in its effects on natural resources.

The ONF developed a parameter called “stream crossing density” to describe potential effects of the number of times a stream was crossed by a road or trail with the general thought that the more a stream is crossed, the more sediment is put into the stream. The SFEIS states (p. 139) that *“Stream crossings (mainly culverts) can cause a large amount of sediment to be mobilized when hydrologic capacity is exceeded, or when the culvert becomes plugged and the road fills are overtopped. When road fills are breached, sediment loading can greatly increase. Interception of interflow by cut slopes can generate substantial amounts of runoff converting subsurface flow into surface flows (Furniss et al. 2000).”*

Then, the SFEIS set subjective ratings for subwatersheds based on the overall ranges observed in the Project area with more than 3.5 crossings per square mile of drainage area were considered “high”, between 2.5 and 3.5 crossings per square mile were considered “moderate”, and less than 2.5 crossings per square miles were considered “low”. These ratings seem arbitrary and not based on any other logic other than the observed range, probably in GIS.

In addition, the SFEIS (p. 139) reported that *“A road/trail density rating with less than one mile per square mile was considered low, between 1 and 2.4 miles per square mile was considered moderate, and greater than 2.4 miles per square mile was considered high”* and used ML 1-5 roads (but failed to include temporary, decommissioned and user-created trails). These ratings were based on a USFS Watershed Condition Classification Guide and seem rational. However, decommissioned roads, temporary roads and user-created trails that add to stream degradation were arbitrarily and inappropriately left out of analyses for important impacts because these roads and trails are driven by forest users.

By designating such a large Project area and spreading out the number of trails over the landscape, the SFEIS draws the conclusion that the impacts on natural resources will be minor. All the analyses of impacts on soils, hydrology, sediment, fish and wildlife habitats are marginalized by the claim that the Project affects a relatively small footprint on the landscape (an estimated 200 acres of actual trail on a large Project area (301,580 acres). A good example is the analysis of impacted riparian habitat. The SFEIS (Table 160, p. 367) shows that Alternative 5 while only 4.76 miles of new trail and trails on reopened roads will occur in riparian habitats, with only 1 10 foot assumption on the width of the trail, the estimated area impacted will be 5.72 acres. There will be additional impacts from stream crossings as well. The SFEIS (p. 366) states that *“These acreages are small when put in context of the overall amount of land in the project area due to the linear nature of trails. This is especially true considering the vast amount of the planning area where no trails are proposed - the southern half of the project area.”*

The arbitrarily small impact zone is based on the fact that the Forest ID Team was told to assume all riders will stay on the designated trail and to base the analysis of project effects to the narrow linear trail feature (Barb Franano, retired Ochoco Forest Fish Biologist, pers. comm.). OHV monitoring results, OHV videos on the internet and first-hand experience show that OHV riders prefer to ride off-trail (See XI. Law Enforcement section). Given

the sensitivity and scarcity of riparian and wetlands habitats and given that a percentage of riders will “play” off trail, even if it’s a few riders, many acres of these sensitive habitats will be destroyed.

This is a case of “mission creep”. A major change slowly occurs over a long period of time but is accepted as the norm because it happens slowly in small and unnoticed increments. Visitors and forest staff alike slowly adjust to the gradual decline of fish and wildlife populations and sensitive habitats as road densities slowly increase over time. Rather than moving toward the LRMP and Roads Analysis standards, there is gradual acceptance, that the situation normal.

VIII. The ONF failed to share important data on location of trails through sensitive habitats and failed to meet criteria to minimize impacts to riparian areas and aquatic resources.

Central Oregon Landwatch requested through the Freedom of Information Act (FOIA) the GIS data layers for the Summit Trail Project. Most of the base layers were not provided timely such as GIS information requested on all of the fuels and vegetation treatment projects and associated information vegetation treatments and roads for each project such as temporary roads, new construction and reconstruction, and user-created OHV trails. In a few cases, we some of the treatment areas were received but only the areas that had been treated to date, not the entire proposed project treatments that have yet to be completed. For example, for the HEJ project, we received polygons representing the outline of activity units but not the data specific to the project treatments. For trails we received one endurance trail (horse) in the NW corner of the project area, but no data on user-created trails. We did not receive any data for Alternative 1 which is the No Action Alternative and the basis for comparing action alternatives. We are unsure why this data was not provided for all public users to view in the SFEIS as this is important to understand the details and impacts to natural resources for each trail location. Without the site-specific data and analysis, we have no understanding of the existing condition and how trails will impact resources such as Class I-IV streams, riparian habitats, wetlands, seeps, and springs.

One of the bits of information that was received was the location of the Alternative 5 trails with respect to streams, topography and type of trail (new, mixed use open and roads proposed for reopening from decommissioned and closed) in the Deep Creek watershed. Figure 20 shows an area west of Little Summit Prairie on the Eastside Implementation Area and with proposed crossings for Deep and Little Summit creeks.

While there are a lot of problems with this trail, two items are particularly noteworthy from Figure 17. First is the trail north of Deep Creek that is proposed on an old road bed that travels a mile up a Class IV stream before reaching the top of the ridge on the scablands. The second is the amount of new OHV trail construction north of Little Summit Creek on scablands. Figure 18 is the Deep Creek area photo from Google Earth and depicts the Class IV stream where the proposed trail will be located on an old road bed. Measurements indicate that the old road bed is located from approximately 30 to 150 feet from the Class IV stream, well within a sediment delivery zone.

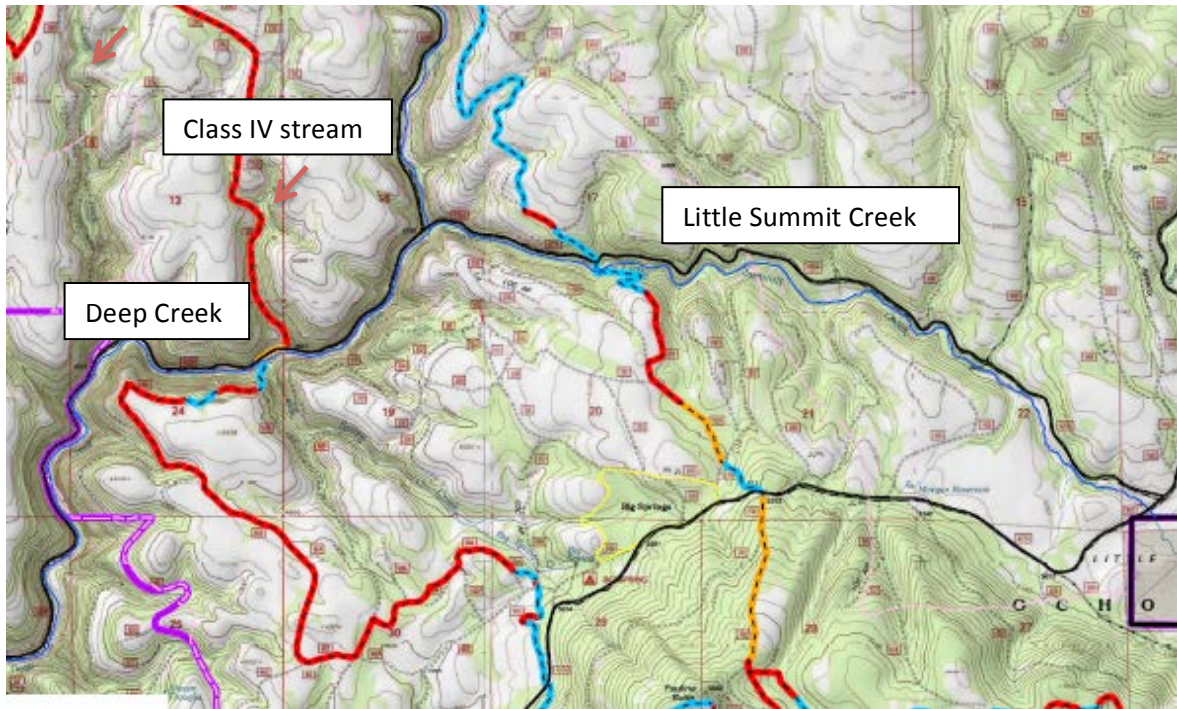


Figure 17. Proposed OHV trails west of Little Summit Prairie and crossing Deep and Little Summit creeks. Note the trail to the right of the “Deep Creek” text box in red, which is an old roadbed that is either closed or decommissioned and will be converted to an OHV trail. Note the new construction trail colored in blue north of Little Summit Creek. Orange are “open” roads that are mixed use for both OHVs and highway-legal vehicles.



Figure 18. Google Earth photo with Deep and Little Summit Creeks. The proposed OHV trail will be located in the Class IV drainage in the upper center portion of the photo.

Per Executive Orders 11644 and 11989 in designating NFS trails and areas on NFS, the responsible official (ONF – Forest Supervisor) shall consider effects on the following, with the objective of minimizing:

1. Damage of soil, watershed, vegetation, and other forest resources;
2. Harassment of wildlife and significant disruption of wildlife habitats;

Therefore, these Executive Orders guide the US Forest Service to apply minimization criteria in a meaningful way to each trail segment and minimize impacts to sensitive soils, vegetation and aquatic habitats. While most of the information is presented in such a broad way that it's difficult to see actual site impacts, Figures 17 and 18 reveal the lack of minimization criteria that the ONF used to place these trail segments.

The ONF proposes to reopen an abandoned road that has trees growing on the old road bed and convert it to a trail, and the abandoned road is located immediately adjacent to a Class IV stream. So there are combined impacts of crossing a Class I fish-bearing stream (Deep Creek) and then have a trail that is easily less than 50 to 100 feet from a Class IV stream for approximately 1 mile. This is a high risk scenario that will put sediment into the Class IV stream and then down into Deep Creek.

In the second case, most of the trail is new construction up a steep grade to a scabland on top (Figure 17). Despite the comments from the SFEIS that states (p. 440) *"The proposed amendments [for scablands] are site-specific and apply only to proposed segments of connector trail, most of which are located on existing disturbance"*, this trail highlighted in blue is obviously new construction on a large scabland area. The LRMP reports that "Scablands are recognized as among the most fragile ecosystems on the Ochoco National Forest. Damage to the soil and vegetation as a result of management activities is nearly impossible to mitigate. This is a result of their having very shallow soils which are subject to severe water saturation and frost heaving during winter, thus making revegetation virtually impossible." (LRMP 4-197). Scablands have highly erodible soils and the likelihood of sediment reaching streams from these sensitive shallow soils from OHV trails is very high. The SFEIS (p. 376) also noted that *"A project design criteria was developed to protect TES plant habitat by closing the trail system when adverse conditions exist that may result in resource damage (rutting, runoff, etc.), however, in scabland habitat, once the damage is done, it is permanent"*. This scenario is another high risk scenario to both the scabland area on the ridge and the high likelihood for both Deep and Little Summit Creeks to be receive large amounts of highly erodible fine clay sediment from the scablands.

In both examples, the ONF failed to meaningfully apply 36 CFR 212.55 (b) (2) minimization criterion to each trail being considered for designation to reduce the effects of the trail system on riparian areas, stream crossings, scablands and aquatic habitat. We were unable to obtain this type of site-specific data for most of the trail system. While there was limited data provided to Central Oregon Landwatch despite the FOIA request for all the base layers and the proposed trails, it is clearly apparent that the ONF failed to locate OHV trails away from these sensitive habitats and it will result in substantial erosion on the scablands and ridge area, and deposition of fine sediments in two Class I fish-bearing streams. Despite the verbiage in the SFEIS that proposes to use Project Design Criteria for Water Quality and Aquatic Organisms (SFEIS p. 31-33), such as using the least grade for stream approaches, reverse grades, hardening with concrete blocks, these are largely construction Band-Aids. There is a

high risk that the locations of these proposed trails will increase runoff and sedimentation from sensitive scabland soils and cause further impacts to already degraded streams, riparian areas, and aquatic habitats.

IX. The FEIS failed to recognize important collaborative and coordinated planning strategies by ODFW, the USFS, and other organizations to protect much of the proposed OHV area from impacts of development.

Numerous entities including the USFS, ODFW, The Nature Conservancy, American Fisheries Society, the Interior Columbia Basin Ecosystem Management Project, and others designated large areas around the North Fork Crooked River including Big Summit Prairie and the tributaries draining into the prairie, and the entire Deep Creek watershed as critical for protection and conservation. Development and implementation of an OHV trail system in this area is inconsistent with goals to protect and restore this sensitive area.

The proposed Project is inconsistent with the USFS Region 6 Terrestrial Resource and Conservation Strategy Areas (TRACS) which was developed as a framework for the region and forests to identify key habitats and species to conserve, restore and enhance habitats for these landscapes (USFS 2012a). The TRACS theme is “protect the best and restore the rest”. The TRACS states in the section on Minimum Roads Analysis that “TRACS should be a primary consideration when completing a Minimum Roads Analysis. Roads are a major threat to priority species and habitats. For example, open roads cause disturbance to 52% of the TRACS Priority Vertebrate Species. Information in the TRACS databases can be used to identify areas that are most important to species and habitats affected by roads, as well as areas where access is important to conduct restoration efforts” (USFS 2012a).

In addition, the priority area that includes the Upper North Fork Crooked River watershed includes the North Fork Crooked River/Big Summit and tributary areas also include an overlapping Aquatic Restoration and Conservation Strategy (ARCS) area (USDA 2008a). The SFEIS failed to acknowledge the importance of redband trout and conserve redband trout as an Oregon Conservation Strategy (OCS) species or the Blue Mountain BM-04 Conservation Opportunity Area as an important strategy habitat to be conserved and protected (ODFW 2006, 2016) (Figures 19-21). Planning and implementation of the Summit OHV Project is inconsistent with protection and restoration of key strategy species and habitats in the North Fork Crooked River and Big Summit areas.

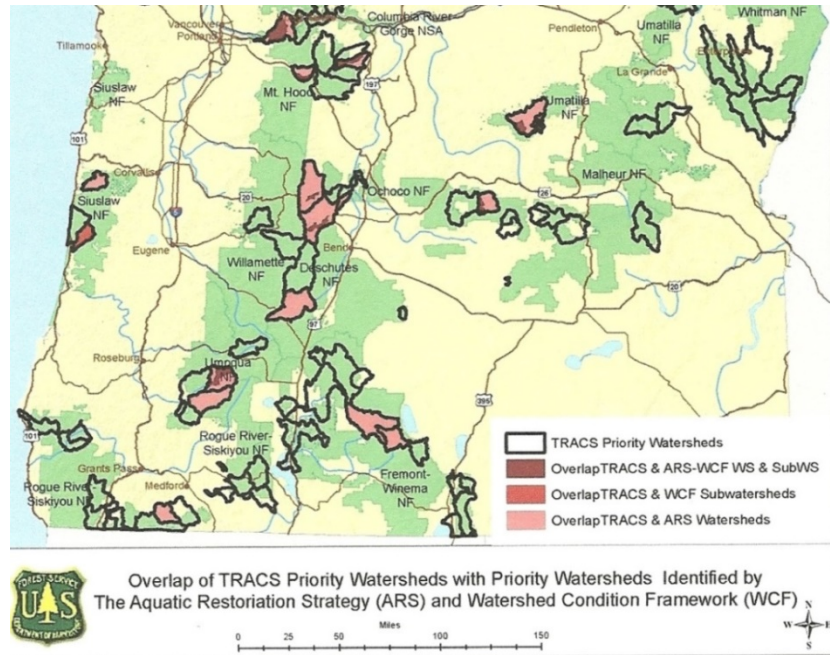


Figure 19. TRACS. Note the overlap of the ARS and WCF with TRACS identifying the Big Summit Prairie and North Fork Crooked River tributaries including the Deep watershed as a priority area for conservation.

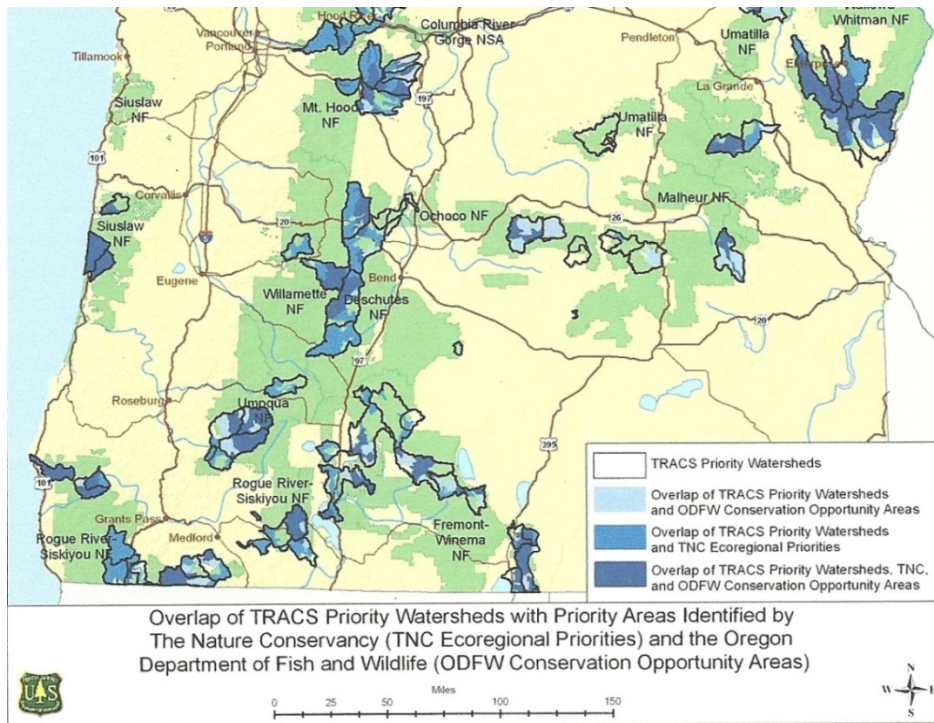


Figure 20. TRACS priority watersheds. Note the overlay with the ODFW Conservation Opportunity Area and TNC Ecoregion Priorities.

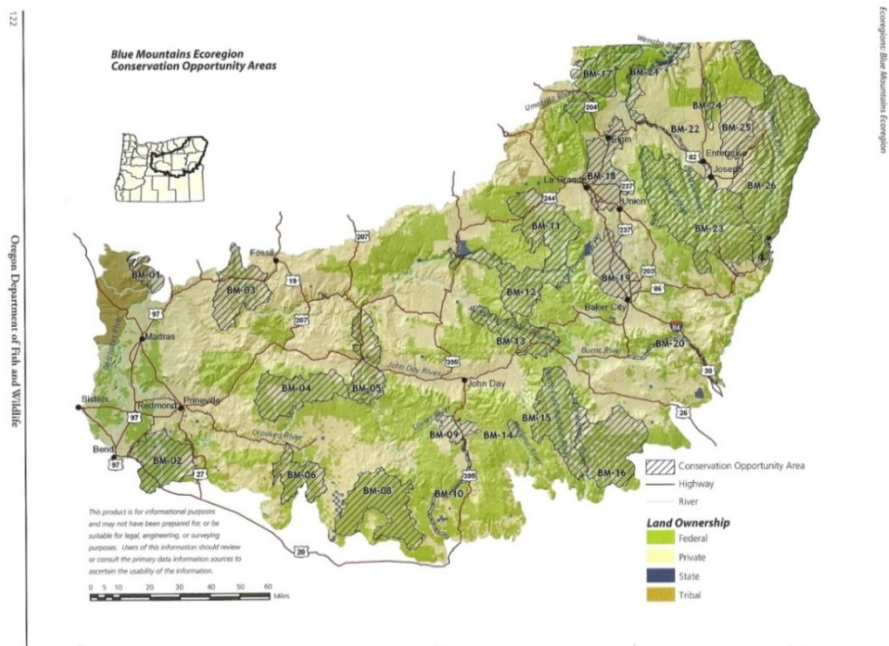


Figure 21. The ODFW OCS Blue Mountain-04 Strategy Area covers large areas and tributaries of the Big Summit Prairie and the North Fork Crooked River tributaries, essentially much of the east and west proposed OHV areas.

The Oregon Conservation Strategy (OCS) is ODFW’s equivalent of the TRACS and is an overarching conceptual framework for long term conservation of fish and wildlife and their habitats using the best available science. Proactive conservation and restoration supports healthy populations of fish and wildlife and avoids costly and controversial regulations that accompany federal listing of species under ESA. USFS staff participated on the Technical Advisory Committee and reviewed the strategy as it went through development and a public review process. One action specifically listed in the OCS is to “work cooperatively with land managers and off-highway vehicle groups to direct use to maintained trails and to improve enforcement of existing off-highway vehicle rules” (ODFW 2006, 2016). ODFW recommended comments to the DEIS (2013), FEIS (2014), and SDEIS (2016) that the USFS abandon this OHV Project, and their comments were largely ignored.

In the Blue Mountains Ecoregion, riparian areas are a strategy habitat for a high level of protection while Inland Columbia Basin redband trout are a strategy species. Increasing recreational pressure and invasive species are major impacts. Even more specifically, OHVs are singled out as a factor causing impacts to riparian, aquatic, and other sensitive habitats, a source for spread of invasive plants, increased fire danger and impacts to wildlife behavior and distribution especially during critical breeding and wintering periods. Use of forest roads also affects wildlife behavior and distribution depending on road type and traffic levels (ODFW 2006, 2016).

BM-04 Ochoco Mountains is a Conservation Opportunity Area in the OCS emphasized for protection and restoration. It includes the parts of the North Fork Crooked River designated as a Wild and Scenic River, the Big Summit Prairie, one of the largest montane wetlands in eastern Oregon, streams throughout the area provide habitat for inland Columbia Basin redband trout and an area with a high potential for breeding sandhill cranes. Key habitats include aquatic, wetlands and wet meadows and key species include Columbia Basin redband trout and sandhill crane. This same area is also recommended for protection in the American Fisheries Society Aquatic

Diversity Areas, the Eastern Oregon Bird Conservation Plan, and the Interior Columbia Basin Ecosystem Management Project plant biodiversity area.

Numerous entities including the USFS, ODFW, The Nature Conservancy, American Fisheries Society, the Interior Columbia Basin Ecosystem Management Project, and others have designated large areas around the North Fork Crooked River including Big Summit Prairie and the tributaries draining into the prairie, and the entire Deep Creek watershed as critical for protection and conservation. Development and implementation of an OHV trail system in this area is inconsistent with goals to protect and restore this sensitive area.

X. The proposed mitigation measures are contradictory and insufficient to mitigate for impacts caused by the Summit OHV system on soils and stream sedimentation.

The SFEIS (p. 31-33) proposes “Resource Protection Measures” (SFEIS, p. 29) and then cites “Project Design Criteria” (DPC) for action alternatives that are intended as *“sideboards and assumptions that were used in development of specific alternatives”* (SFEIS p. 29). The water quality and aquatic life PDCs on pages 31-33 are supposed to avoid, minimize or mitigate the Project but many of these are largely ineffective. The SFEIS (SFEIS p. 33) estimates that *“Annual precipitation ranges from 15 to 31 inches per year”*, which at the higher elevations in the Project area, would average around 30 inches. The SFEIS further describes the climate with respect to vegetation and sediment delivery to streams (SFEIS p. 33) *“Semi-arid climates, such as the climate found in Central Oregon, are expected to be elevated relative to other climates as a results of these climates having sufficient precipitation to drive erosion, but limited amounts of vegetation that stabilize hillslopes from erosion (Goode et al. 2012). Generally, vegetated filter strips between 200 to 300 feet from the stream are considered to be effective in controlling sediment that is not channelized depending upon the riparian buffer slope (Belt et al. 1992).”*

One problem is the season of use (SFEIS Table 27, p. 72). Alternative 5 has a 4 month operating season proposed from June 1 to September 30, however, much of the trails are in high elevations and both construction activities and OHV riders are likely to encounter wet soils particularly in June when snow is receding and rainfall is more frequent. May and June are when redband trout are spawning and young of the year are rearing in headwater tributaries from early July to mid-August (Stuart et al. 2007).

Most of the DPCs are largely construction measures but don’t mitigate for the ongoing and continuous loss of vegetation and soil disturbance, and soil ruts, and subsequent sediment into streams every time it rains or snowmelt occurs. In fact, it’s hard to determine what structures will be implemented let alone whether they are effective at reducing sediment into streams at crossings. The SFEIS has a confusing mixture of bridges, culverts and fords that vary depending on the location of the information. For example, the total numbers of crossing types are either 79 or 96 (SFEIS p. xiv, p. 107). Data on the types of stream crossings are with a total of 9 bridges, 39 culverts, 7 puncheon stringers, and 24 fords in Table 26 (SFEIS p. 71), a total of 23 bridges, 73 culverts, and no fords and puncheon stringer in Table 53 (SFEIS p.107). There are more inconsistencies so which is it? This conflicting data shows how the ONF cannot possibly even begin to minimize OHV trail impacts to aquatic resources when they have not completed field surveys and don’t even know what’s on the ground. It leads to confusion and makes comparison of alternatives difficult, and calls into question the accuracy of all data presented throughout the SFEIS. Regardless of the conflicting information, merely adding a bridge or culvert on

most of the stream crossings does not alleviate impacts from 1) hardened fords, 2) a largely expanded network of roads and new trails within 300 feet of streams, and 3) crossings on ephemeral streams.

The FEIS proposes puncheon stringers for wet meadows, hardening stream approaches, crossing drainage structures, and water drainage facilities (SFEIS p. 32) for all trails. These mitigation measures are also insufficient to impact streams already compromised by past and present land use practices. When OHVs ford streams, they release sediment into streams ((Switalski and Jones 2012) by other mechanisms: 1)surface runoff concentration from wheel ruts, 2) exposed soil surfaces from tracks, 3) increased runoff from soil compaction, 4)vehicle backwash, and 5) undercutting of banks from waves (Switalski and Jones 2012). While the proposed culverts may alleviate impacts from items 4 and 5 above, they cannot alleviate sediment sources from the first 3 mechanisms.

The SFEIS also alleges that *“physical closure would offset the development of new designated trails by physically limiting unauthorized access on these routes”* (FEIS p. 343). There are many areas such as scablands that are large and relatively flat and easily accessible to almost any motorized vehicle. It is specious to assert that physical closures could possibly limit off road use, since it is already occurring on many “closed” areas that are physically blocked with a gate, large rocks, or ripping. The following is a photo on a scabland where the proposed OHV area will be on the dirt road straight ahead while the road on the left will remain “closed” (Figure 22).



Figure 22. Scablands with “closed” roads. Road straight ahead is proposed as part of the Summit OHV system while the road to the left will remain “closed” and according to FEIS p. 343, “physically closed”.

XI. The FEIS fails to address the impacts of illegal OHV use. With one law enforcement officer (LEO), adding an additional OHV area substantially reduces the effectiveness of law enforcement on the landscape. Grant or “soft” dollars are inappropriately relied on to fund restoration and mitigation requirements.

The SFEIS fails to adequately assess the impacts of the proposed OHV system and fails to assess impacts of resulting illegal use. The SFEIS did not include many of the closed and decommissioned roads and user created trails in the analysis on impacts to hydrology, streams and sediment. With an estimated 700 additional miles or more of known unauthorized user-created trails, the SFEIS failed to disclose the harm occurring to soils and streams. Meadows et al (2008) reported that increased use of ATVs on public lands means that even a small percentage of riders who travel off roads/trails cause considerable damage, and because the motorized vehicles are powerful, can travel many miles quickly and damage sensitive resources easily.

One compelling example is Henderson Flat, an OHV area on the CRNG, administered by the ONF and which has been in place for over 20 years. The CRNG has numerous illegal roads and trails, most of which have been created in the past 20 years. Because much of the Grassland is relatively flat with easily accessible grades, vehicles continue to drive off road causing much damage to natural resources (Figures 23-29).



Figure 23. Motorized Restrictions on the Grassland



Figure 24. Posted regulations on the Grassland.



Figure 25. “Closed” OHV road; note the broken sign



Figure 26. Illegal OHV trails in Skull Hollow



Figure 27. Illegal 4WD road



Figure 28. Illegal motorcycle trail

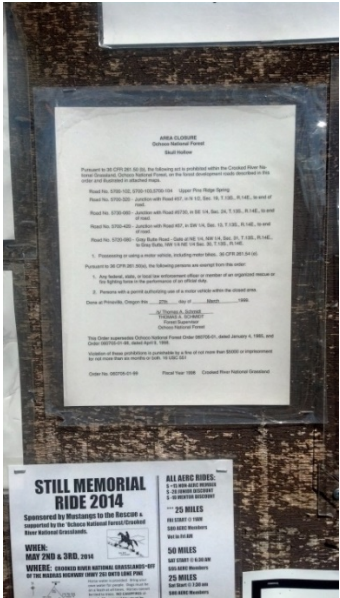


Figure 29. Road closure sign

Figures 23-29 through depict the problem of ongoing illegal use on a restricted motorized area that has been in place since the early 1990s. The SFEIS argues that the Summit OHV Project would better distribute OHV use but in fact, these illegal roads and trails have never been restored or ineffectively at best, are rarely if ever enforced, and continue to degrade water quality, increase sedimentation, and impact important year-round and big game winter range for mule deer and antelope. A likelier scenario is that the proposed Summit OHV trail would create even more illegal activity and access to more areas, creating more damage to sensitive habitats and fish and wildlife resources.

The addition of a new OHV Project would take the already limited law enforcement (one law enforcement for the approximately 1 million acres of ONF and CRNG) and direct it at these new trails, further shifting resources away from the illegal trails, rampant throughout the CRNG and Forest.

The following is a summary of observations of off-trail and off-road use from other locations around western states:

- A Utah report reveals that a high percentage of riders prefer to ride “off established trails” and did so on their last outing. 49.4% of ATV riders surveyed prefer to ride off established trails. 39% rode off trail on their most recent trip. 38.1% of motorized dirt bike riders prefer to ride off established trails. 50% rode off established trails on their most recent trip (Fisher et al. 2002).
- More than nine out of ten (91%) of respondent rangers from the Bureau of Land Management (BLM) and the U.S. Forest Service (USFS) agree that off road vehicles represent “a significant law enforcement problem” in their jurisdictions (Rangers for Responsible Recreation 2007).
- According to one BLM respondent, “90% of ORV users cause damage every day they ride. Most will violate a rule, regulation or law daily.” (Rangers for Responsible Recreation 2007)
- A 2006, Montana Fish, Wildlife and Parks study of OHV users surveyed 446 owners of registered off-road vehicles. 23% of respondents said that they “always or sometimes” ride cross-country even though off-route riding is against the rules in Montana and has been since 2001. Over 28% “sometimes or never” avoid riparian areas and wetlands, in violation of rules for federal and state public lands in Montana. 64% have used an off-road vehicle while hunting and the majority of this hunting subset admits to riding cross-country. Over 58% traveled off legal routes to retrieve downed game. (Lewis and Paige 2006).
- A Colorado study found that the majority of off-roaders understand that staying on designated routes is “fundamental trail etiquette” and that going off trail is not “correct” off-road vehicle behavior. The survey revealed, however, that regardless of this knowledge “as many as two-thirds of adult users go off the trail occasionally.” 15-20% of the riders admitted to frequently breaking the rules and riding off of legal routes often. Participants also stated that “others” ride off-route and cause most of the damage (Frueh et. al. 2001; referenced in Kiely and Kassir 2007).

Kiely and Kassir (2007) assessed attitudes and behaviors of OHV drivers. The fairytale assumption is that a well-designed trail will keep riders on OHV systems when riders are provided a well-maintained designated trail system (SFEIS p. 80). Kiely and Kassir (2007) discredited this with a review of studies and demonstrated that OHV riders have a distinct predilection to travel cross-country and ride off of legal routes. They also reported that public land managers tend to assume that designating additional OHV trails and roads leads to improved compliance with regulations, reduced illegal riding, and, therefore, less resource damage and fewer conflicts with non-motorized

users. They stated that some public land managers believe that OHV users will not create illegal routes once more trails and routes are designated “open” and OHV users are educated to proper OHV etiquette. Their research demonstrated that damage and conflicts are not just caused by a small percentage of OHV riders that law enforcement was essential to manage lawless behavior, and education and information needed to be accompanied by strong law enforcement to change off-road behavior.

In central Oregon, there are almost 1,000 miles signed routes that OHV users may choose to use. The proposed Summit OHV area is estimated to have 700 miles of illegal user created trails. However, there is no assurance that federal agencies will provide sufficient on-the-ground enforcement to keep OHVs on designated routes. It hasn’t happened yet, so why give credence to empty promises? Given the preference of OHV users for going off trail, enforcing existing designated trail systems are already an enormous challenge for law enforcement, let alone adding a new system in the Ochoco Mountains for the sole law enforcement officer.

The TMR indirectly addresses enforcement, by stating that "In designating NFS roads, NFS trails, and areas on NFS lands for motor vehicle use, the responsible official shall consider effects on NFS natural and cultural resources, public safety, provision of recreational opportunities, access needs, conflicts among uses of NFS lands, the need for maintenance and administration of roads, trails, and areas that would arise if the uses under consideration are designated; and the availability of resources for that maintenance and administration." While the SFEIS reports that COHVOPS will be the entity that is responsible for maintenance and enforcement, it has largely been ineffective in places like the CRNG near Henderson Flat OHV area. Three years ago at a 2013 OPRD ATV Grants Committee meeting, it was reported that COHVOPS already has a total of 1,121 miles to patrol, operate and maintain. More OHV areas have come online or been expanded since then including the Three Trails Area on the Deschutes National Forest and Cline Buttes on the BLM lands west of Redmond.

The SFEIS commits to funding the OHV Project, which includes road closures and restoration along with construction, operation, and law enforcement, using “soft dollars” from grant sources. Most of the proposed funding for the OHV Project is from COHVOPS competing for state ATV grant funds from the Oregon Parks and Recreation Department. The 2014 FEIS admitted (p. 498) that *“Competition for state ATV grant funds is challenging”....“It is possible the State could divert gasoline tax revenue (which composes the bulk of the ATV grant funds) towards other uses”*. With a history of declining federal dollars, the USFS has a long history of building and not maintaining facilities.

The ATV Grants Committee reported funding shortfalls for OHV Project requests starting in 2011 (April 2011 Meeting) and declined to fund the request from the ONF for the Summit Planning Grant for \$36,500.00. Table 5 shows the general declining support for law enforcement for OHV trail systems while COHVOPS has experienced some grant fund fluctuations for operations, maintenance and compliance in the 3 recent bienniums.

Table 5. Grant funds from the ATV Committee to Central Oregon from the 2011-13 biennium to 2015-17 biennium.

Entity/Biennium	Crook County Sheriff	BLM Law Enforcement	COHVOPS O&M
2011-13	\$145,890	\$161,104	\$936,298

2013-15	\$96,433	\$122,536	\$880,414
2015-17	\$50,050	\$0	\$1,035, 781

Declining funds for recreation facilities and roads and increasing costs for wildfires mean less funds for managing a proposed OHV trail system. The 2006 ONF Recreation Facility Analysis projected a decline in funds to maintain recreation sites on forest (USDA 2006). The same decline in funding for transportation was reported in the 2015 Forest Wide Transportation Analysis Report (USDA 2015b). The SFEIS (p. 57) stated that *“It would also be anticipated, however, that the selection and implementation of an action alternative in this analysis would result in an increase in recreational traffic during the summer months, as a result of increased OHV use.”* The 2015 Forest-Wide Travel Analysis Report (TAR) (USDA 2015c) indicated that *“Forest Service road budgets have been steadily declining for the past 20 plus years. Region-wide, the amount of funding for road work including both appropriated funding and work contributed by commercial users is less than 20 percent of what it was 20 years ago. Appropriated road funds to the Pacific Northwest Region (Region 6) have been reduced 40% in the past 5 years alone”* (TAR, USDA 2015c). With the projected decline in funds, maintenance has been deferred for much of the forest roads. Meanwhile, wildfires have taken a steadily increasing portion of Forest Service funding (USDA 2015a). In 1995, fire made up 16 percent of the Forest Service’s annual appropriated budget, made up over 50% in 2015, and is projected to reach 67% by 2025.

Adding the OHV system to a transportation and recreation that is already overcommitted and underfunded, and relying on soft grant dollars for mitigation and restoration violates the 2011 Council of Environmental Quality (CEQ) guidance for monitoring and mitigation (CEQ 2011). The regulation states that “agencies should not commit to mitigation measures considered in an EIS or EA absent the authority or expectation of resources”. The CEQ guidance states that “if it is reasonably foreseeable that funding for implementation of mitigation may be unavailable at any time during the life of the project, the agency should disclose in the EA or EIS the possible lack of funding and assess the resultant environmental effects”. This guidance states that if mitigation such as closing roads cannot be done because of lack of funding, then the FEIS must disclose the environmental consequences. The proposed grant funds are soft dollars and should not be relied upon to implement and therefore the Project should not happen due to the environmental harm it would do to fish and wildlife and their habitat.

XII. The FEIS and associated documents are poorly written, do not summarize the information well and make the public reader search for information.

According to 40 C.F.R. § 1500.2(d), a critical part of the obligation for public review and disclosure is to present the data and analysis that will enable the public to thoroughly review and understand the analysis of environmental consequences. NEPA requires the use of high quality data and the disclosure of the methodology underlying proposed decisions, and also explicitly requires that an EIS “be written in plain language” and presented in a way that “the public can readily understand”. The regulation at 40 C.F.R. § 1502.8 states that these requirements are particularly true for an EIS; the “primary purpose” of this document is “to allow for informed public participation and informed decision making” so its language must be “clear” and “supported by evidence that the agency has made the necessary environmental analyses.”

A careful review of data, analyses and the conclusions of the FEIS was challenging at best, and misleading or disingenuous at worst. The effects analysis for watersheds and hydrology used data that was in many cases outdated or not well explained. Assumptions were not well documented or highly speculative. There was little if any ground truth surveys. It was difficult to analyze existing conditions since the information was scattered throughout the document. For example, all of the streams should have been summarized in a table to demonstrate the existing condition for important elements such as temperature, bank stability, sediment, and pool frequency. The SFEIS cannot adequately demonstrate analyses unless there is a complete picture of the existing condition. Much of the literature cited in the document was not listed in the Literature Cited section. So once again, the reader had be a detective and look for the documentation on line.

If you have any questions about these comments, please call me.

Sincerely,

/s/ Amy M. Stuart (verifiable signature available upon request)

Amy M. Stuart, Fish Biologist

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PROFESSIONAL EXPERIENCE:

Oregon Department of Fish and Wildlife (ODFW), 9/2007 to 2/2014, Deschutes Watershed Manager

- ❖ Supervised 60 professional and technical staff.
- ❖ Managed expenditures of a combined budget of \$9 million per biennium for 25 cost centers.
- ❖ Implemented fish, wildlife, research and habitat program goals, objectives and priorities.

ODFW 7/1997 to 8/2007, Hydro Power Program Biologist, High Desert Region

- ❖ Lead for the High Desert Region for re/licensing and compliance for state and federal FERC licensed hydro projects in the Klamath, Deschutes, Hood and Owyhee basins.
- ❖ Lead negotiator and/or technical lead for the Pelton Round Butte Settlement Agreement, Klamath Settlement Agreement, and Powerdale hydro project decommissioning.

ODFW, 1/1990 to 6/1997, District Fish Biologist (DB), Supervisory FWB (3 1/2 years) and NRS 3 (4 years), Ochoco District, High Desert Region

- ❖ Fish program manager for the Ochoco District (7½ years). Hired, supervised, and conducted performance reviews of 1 permanent, several seasonal employees and 3 high school interns.
- ❖ Collected and analyzed fish population abundance and distribution, species composition, and harvest; evaluate hatchery stocking, harvest regulations; implement habitat protection/restoration.

ODFW, 2/1988 to 12/1989, Assistant District Fish Biologist II, Ochoco District, High Desert Region

- ❖ Implemented the fish management program. Sole fish biologist and assumed lead responsibility; ½ time Ochoco National Forest Fish Biologist: advised fish issues for timber sales and range allotments; conducted fish, habitat surveys and habitat projects. Authored bull trout plan at Lake Billy Chinook.

ODFW, January 1986 to January 1988, Fish Habitat Biologist II, BPA Program, John Day, NE Region

- ❖ Implemented habitat program using restoration techniques to improve fish and wildlife habitat: bank stabilization and habitat projects. Implemented contracts and landowner lease agreements. Prepared annual reports. Developed, supervised and administered budgets, personnel, equipment, and ODFW staff.

1977-1986. ODFW Ocean Salmon, US Forest Service, US Army Corps of Engineers, Colorado Mined Land Reclamation, and private timber cruiser.

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 - * Oregon Chapter American Fisheries Society. 1999. Hydro Relicensing in the State of Oregon, the Good, the Bad, and the Ugly. Sunriver, Or.
- Numerous presentations to the Oregon Fish and Wildlife Commission and conservation groups for 20 years on a variety of topics including the Crooked River Basin Fish Management Plan, Annual Fishing Regulations,

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Oct. 31, 2016

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Re: Review of Supplemental Final Environmental Impact Statement (FSEIS) and appendices for the Ochoco Summit Trail System Off Highway Vehicle (OHV) Trail Project (Project) on the Ochoco National Forest (ONF), dated September 2016

Dear Mr. Dewey,

I reviewed the aforementioned FSEIS, including appendices, particularly with respect to its assessment of watershed and aquatic impacts and compliance with the ONF Land and Resource Management Plan (LRMP) standards. These standards include those in the 1995 “Inland Native Strategies for Managing Fish-producing Watersheds in Eastern Oregon and Washington, Idaho, Western Montana, and Portions of Nevada” (INFISH) amendment to the LRMP. As part of my review, I also reviewed:

- My previous 2014 review and discussion of the numerous inadequacies in the FEIS for the Project, which also describes the material I reviewed at that time;
- The 2016 draft reports on botany, geology, and aquatics for the FSEIS;
- Salient scientific literature, which is listed in the literature cited section at the end of this report.

My comments explain how the FSEIS failed to properly assess the Project’s aquatic impacts and compliance with required standards and failed to substantively respond to most of my previous comments on the FEIS’s defects.

**THE FSEIS FAILS TO PROPERLY DESCRIBE EXISTING CONDITIONS THAT
CONTRIBUTE TO CUMULATIVE EFFECTS**

The FSEIS failed to disclose the extent of grazing in the Project area at the scale of watersheds and subwatersheds.

Most of the Project area is subject to grazing as the FSEIS acknowledges (p. 388). This is significant because, as discussed in greater detail in following sections, it is well-documented that livestock grazing contributes to the degradation of several stream conditions that are INFISH Riparian Management Objectives (RMOs). The ONF is required to avoid degradation of RMOs and not allow activities to retard their attainment.

Studies have repeatedly documented that grazing contributes increases water temperatures by reducing stream shading and widening streams via trampling and sedimentation (USFS and USBLM, 1997; Cowley, 2002; Beschta et al., 2013; Nusslé et al., 2015). As a USBLM publication (Cowley, 2002) noted:

“[i]t is well documented that large herbivores such as cattle, horses, sheep, bison, elk, and moose can alter the physical dimensions (e.g., increasing the bankfull width) of stream channels by bank trampling and shearing...Increasing the bankfull width makes the stream shallower, increases sediment, decreases the floodplain, increases temperature, and increases the adverse [effects on] the physical functioning of a stream, its associated riparian area, and aquatic habitat...”

Water temperature is an INFISH RMO. As will be discussed in greater detail, the action alternatives will adversely affect water temperatures in numerous subwatersheds in the Project Area, many of which already have severely degraded water temperatures.¹ Water temperatures are cumulatively affected by watershed scale conditions (Allen and Dietrich, 2005), including those affected by grazing. Thus, grazing and the action alternatives will have combined temperatures impacts that must be assessed in order to:

- a) reasonably assess compliance with INFISH RMO for water temperature;
- b) adequately determine compliance with the state water temperature standard;
- c) properly estimate and make known cumulative effects on water temperature;
- d) adequately determine impacts on redband trout which are already adversely affected by degraded water temperatures (McCullough, 1999) due to the combined degradation of streams and riparian systems in the Project area caused by existing roads, grazing, logging, and OHV use.

As discussed in more detail in a following section, the FSEIS failed to reasonably assess these combined impacts on water temperatures. Nonetheless, at a minimum, the FSEIS needed to make known the fraction of Project subwatersheds that are subjected to grazing in order to provide at least some index of water temperature impacts due to grazing combined with trail impacts under the action alternatives.

Stream channel width/depth ratio is another INFISH RMO which is greatly affected by grazing (Rhodes et al., 1994; Magilligan and McDowell, 1997; USFS and USBLM, 1997; Nusslé et al., 2015) and would be significantly affected by the impacts of the action alternatives, although the FSEIS fails to disclose the latter. Grazing not only directly widens channels, but also increases width/depth ratios via greatly accelerated sedimentation, which the action alternatives would also increase. As discussed in more detail in a following section, the FSEIS acknowledges (pp. A-46, A-47, A-116) that it completely failed to assess grazing impacts on sediment delivery and sedimentation. In so doing, the FSEIS also failed to assess the combined impact of grazing and action alternatives on cumulative sediment delivery, resulting sedimentation, and the INFISH RMO for width/depth ratio. Especially due to these major failures, the FSEIS needed to make

¹ According to the FSEIS (p. 178), Project area streams listed as Water Quality Limited Waterbodies by the of Oregon due to degraded water temperatures “...include Double Corral Creek, Fox Creek, Happy Camp Creek, Howard Creek, Indian Creek, Jackson Creek, Little Summit Creek, Marks Creek, North Wolf Creek, Ochoco Creek, Peterson Creek, Porter Creek, and Toggle Creek.”

known the fraction of the area of each of the Project subwatersheds subjected to grazing in order to provide at least some index of width/depth ratio impacts due to grazing combined with trail impacts under the action alternatives.

Pool frequency is also an INFISH RMO which is strongly affected by grazing and cumulative sediment delivery at the scale of subwatersheds and watersheds (Rhodes et al., 1994; USFS and USBLM, 1997; McIntosh et al., 2000; Buffington et al., 2002; Beschta et al., 2013). Thus, the impacts of the action alternatives would combine with those from grazing to cumulatively affect pool conditions. The FSEIS admittedly failed to include grazing-caused sediment delivery (pp. A-46, A-47, A-116) in combination with those under the action alternatives, which are necessary to reasonably assess the cumulative impacts on the INFISH RMO for pools under the action alternatives. This defect made it even more critical for the FSEIS at least reveal the fraction of the area subjected to grazing in all of the Project subwatersheds to provide at least some index of pool impacts due to grazing combined with trail impacts under the action alternatives.

Studies have consistently documented that grazing greatly reduces bank stability (Rhodes et al., 1994; USFS and USBLM, 1997; Cowley 2002; Beschta et al., 2013; Batchelor et al., 2015), which is an INFISH RMO. This, too, makes it critical error that FSEIS failed to at least provide an index of cumulative bank stability impacts under the action alternatives by making known the area subjected to grazing in all Project subwatersheds and watersheds.

Grazing also contributes significantly to cumulative sediment delivery, which significantly degrades fine sediment levels in streams, which significantly reduces redband trout survival and production, as discussed in a following section. As with other sediment-related conditions affected by cumulative sediment delivery from grazing and action alternatives, the FSEIS's failure to evaluate sediment delivery from grazing made it crucial for the FSEIS to disclose the area subjected to grazing in all Project subwatersheds and watersheds.

For these reasons, the FSEIS's failure to make known area subjected to grazing in all Project subwatersheds and watersheds is a major defect.

The FSEIS failed to make known the extent and locations of user-created trails

The FSEIS (p. A-69) admits that it is devoid of information on the extent, location, and density of user-created trails at the scale of watersheds and subwatersheds, despite repeated comments on the FEIS that this was crucial to disclose (FSEIS, p. A-69). This information is critical to assess and make known for two other reasons.

First, these routes unquestionably contribute to cumulative sediment delivery at the watershed scales, affecting a host of RMOs and extremely important stream conditions, including pools, width/depth ratio, water temperature,² and fine sediment levels in streams. The FSEIS failed to make any quantitative estimate of sediment delivery from the user-created routes. Hence, it was critical for the FSEIS to make this information known at the scale of watersheds, to at least provide an index of their impacts.

² Increases in width/depth caused by elevated sediment delivery increases water temperatures even in the absence of shade loss (Rhodes et al., 1997; USFS and USBLM, 1997; Bartholow, 2000; Cowley, 2002)

Second, the FSEIS repeatedly asserts (e.g., p. 181) without any sound basis, that the action alternatives will ultimately result in sediment delivery that is lower than existing conditions due to reductions in sediment delivery from user-created trails. While this is a flawed assumption for several reasons, as will be discussed in greater detail, a primary one is that it is impossible for this to be sound absent information on the extent, density, and location of these trails at the scale of watersheds and subwatersheds, all of which exert a profound influence on sediment delivery from these trails. Simply enough, the FSEIS cannot assert that sediment delivery reductions from existing user-created routes will be greater than that generated from OHV trails under the action alternatives unless it knows the magnitude of sediment delivery from the latter, which requires complete information on the location, extent, and density of user-created trails at the scale of watersheds and subwatersheds.

The FSEIS attempts to have it both ways: a) that it is not a fatal flaw to quantitatively ignore sediment-related impacts from user-created trails; and, b) that reductions in sediment delivery from these trails will more than offset increases in sediment delivery under the action alternatives. Instead, the facts are that both of these contentions in the FSEIS are false.

For these reasons, the FSEIS has no sound basis for the notion that sediment delivery will decrease below existing conditions under the action alternatives (FSEIS, p. 181) in the absence of information on the extent, location and density of user-created trails at watershed scales. The failure to analyze and make this information known is a severe defect.

The FSEIS did not reasonably evaluate and make known existing stream conditions, and existing cumulative effects, because the vast majority of the stream and riparian data is exceedingly stale.

Much of the data in the FSEIS on instream conditions in watersheds and subwatersheds in the Project area are quite stale (FSEIS, pp. 145- 178). In fact, a considerable amount of this data is for historical conditions, because much of the data is not even from this decade. For instance, there is no data from this decade on stream conditions that would be affected by the action alternatives in the Elliot Creek subwatershed (FSEIS, Tables 74-75, pp. 147-148), Howard Creek subwatershed (FSEIS, Tables 77-78, pp. 151-152), Peterson Creek subwatershed (FSEIS, Tables 80 and 81, pp. 155-156) and Upper Marks subwatershed (FSEIS, Tables 97 and 98, p. 177). The bulk of the instream data for other subwatersheds are more than 10 years old, as is all of the subwatershed scale water temperature data in the FSEIS (pp. 145-178).

This is wholly inadequate because these conditions, that would be affected by action alternatives, can be rapidly degraded, due to the combination of existing management activities, such as grazing, logging, and roading, the pervasively poor riparian and watershed conditions throughout the project area, and/or triggering by natural disturbances, such as floods and fire. Affected conditions subject to relatively rapid degradation include fine sediment, width/depth ratio, pool frequency, water temperature, and bank stability. For instance, a single season of grazing can greatly reduce bank stability and increase width/depth ratio (Rhodes et al., 1994) and most of the Project area is annually afflicted by grazing.

Notably, the data on the density of redband trout in affected subwatershed corroborates that instream habitat conditions may have continued to deteriorate. This data is far more recent and is ample for assessing trends and it demonstrates that redband trout density has declined significantly in most subwatersheds for which data are available (FSEIS, Figures 8-10, 14-21, pp. 150-172).

While it is likely that stream conditions have deteriorated over time in most of the affected streams in the Project area, it is highly unlikely that there has been any appreciable improvement in these conditions. This is because the watersheds are in exceedingly poor condition due to high road density combined with pervasive grazing, as well as logging.

The FSEIS notes that *existing* conditions are used in the FSEIS as a proxy for existing cumulative effects. Because the majority of data in the FSEIS on fundamentally important stream conditions are stale, the FSEIS fails to reasonably assess the cumulative effects of existing conditions in combination with impacts of the action alternatives.

Further, much of the stale data is for INFISH RMO conditions. Thus, the FSEIS also failed to reasonably assess existing INFISH compliance and existing cumulative effects on RMOs.

THE FSEIS IMPROPERLY ADOPTED A SCIENTIFICALLY UNSOUND RMO FOR WIDTH/DEPTH RATIOS

Instead of using the INFISH RMO for width/depth ratios, the FSEIS adopted an RMO for this stream attribute (FSEIS, Table 64, p. 131) in a manner inconsistent with INFISH direction. INFISH (p. E-3) does allow modification of the interim RMOs in the absence of watershed analysis by amendment where "...watershed or stream reach specific data support the change." However, the adoption of the width/depth ratio RMO in the FSEIS was not by amendment and is not based on watershed or stream reach specific data that support the change made in the FSEIS. Further, the FSEIS provides no data to support the modification. Instead, the FSEIS only avows that it lacks data on width/depth conditions in reference reaches in the Project area and then adopts a width/depth RMO that is based on broad criteria used in a stream classification system (Rosgen, 1996) that are **not specific to the Project area.**

INFISH (p. E-3) also notes that RMO refinement should be based on "...local geology, topography, climate, and potential vegetation." In contrast, the RMO adopted for width/depth ratio in the FSEIS is not based on these local factors, but instead comes from broad classification system that is not based on local or even regional factors.

INFISH (p. E-3) directs that if INFISH RMOs are modified, "In all cases, the rationale supporting the RMOs and their effects would be documented." The FSEIS includes absolutely no assessment of the effect of this RMO modification.

The rationale for modifying this modification in the FSEIS is unsound, as is the adopted RMO. INFISH notes that RMOs should represent healthy stream and aquatic habitat conditions. In contrast, the width/depth RMO adopted in the FSEIS are width/depth criteria used in the Rosgen classification system. Notably, nowhere does the Rosgen classification system (Rosgen, 1994)

suggest that these criteria for channel type delineation be set as ecological goals, or that they represent healthy stream conditions for stream types. Instead, the classification system uses the criteria as thresholds breaks for categorizing streams into different types.

These classification thresholds that are adopted as RMOs by channel type do not represent healthy stream conditions. For instance, Rosgen (Figures 8-9, 1994) discuss E-type channels with a width/depth ratio of 0.5-2, which are fairly normal for undegraded E-type channels. In contrast, the RMO adopted in the FSEIS (p. 131) for this channel type is width/depth ratio < 12 . This is not sound, because a width/depth ratio of 10 for an E-type channel represents a degraded condition that would be considered satisfactory under the unsound RMO adopted in the FSEIS.

The RMO adopted in the FSEIS also leads to preposterous results. For instance, a C-type channel with a highly degraded width/depth ratio of 100 would be considered to meet the adopted RMO because the adopted RMO for this channel type is width/depth ratio > 12 with no upper bound.

For these reasons, the modification of the RMO does not meet INFISH guidance for such a change, is not site-specific, and is not scientifically sound.

THE FSEIS FAILS TO PROPERLY MAKE KNOWN IMPORTANT ASPECTS OF PROJECT ALTERNATIVES

The FSEIS did not make known the number of stream crossings by OHV routes on currently closed or decommissioned roads.

Stream crossings by OHV routes are a major vector for degradation of a host of riparian and stream conditions, including large woody debris (LWD), pools, fine sediment, water temperature, runoff regimes, and noxious weed spread in riparian areas. There is no question that re-opening decommissioned and closed routes for OHV use will increase sediment delivery at crossings to a far greater degree than from allowing OHV use on existing open roads. However, the FSEIS fails to make known the number of stream crossings under the action alternatives that would be on currently decommissioned and/or closed routes opened for OHV use. Instead, the FSEIS only attempts to make known the number of crossings on newly created or existing routes. Due to the nature of impacts at stream crossings, this renders the FSEIS defective.

The FSEIS asserts that some undisclosed length of user-created routes in undisclosed locations would be closed and rehabilitated under the action alternatives but not under the No Action alternative.

The FSEIS repeatedly asserts that some user-created routes would have closure and possibly rehabilitation treatments under the action alternatives. Notably, the FSEIS provides no specificity whatsoever regarding the magnitude or location of these activities under the alternatives. They are also clearly contingent on funding that currently doesn't exist and potential for which the FSEIS (p. 57) states is “**outside the scope of this project.**”

To recap, much of the FSEIS assessment of cumulative effects analysis relies on the notion that an unspecified amount of user-created roads in unspecified locations in unspecified watersheds will be treated by unspecified remedial activities of unknown effectiveness at an unspecified rate under the action alternatives with funding that is not only uncertain, but that the probability of obtaining is "...outside of the scope of project." There is not much reality to this scenario but nonetheless, this spun cloth is repeated in mantra-like fashion in the FSEIS.

Further, doing something to reduce damage from user-created trails, especially those within riparian habitat conservation areas, is not discretionary. INFISH requires that it be done. It is not something that can be ignored under the No Action alternative, if the ONF is to comply with INFISH.

The FSEIS's attempts at disclosing existing conditions and important aspects of the action alternatives at the watershed and subwatershed scales are so riddled with error that it is not possible to reliably ascertain the impacts and nature of the action alternatives at the watershed and subwatershed scales.

The FSEIS's discussion of aspects of the action alternatives fails to clearly convey aspects of the alternatives that are critical to describing the nature of the alternatives due to numerous errors. Examples of such problems include the following, which is not an exhaustive list of erroneous information in the FSEIS:

- The FSEIS (p. 128) states "From Table 65, the overall ratings for the Ochoco Summit Project area came out as 'properly functioning' for both the Upper and Lower Big Marsh Creek subwatersheds." However, Table 65 (p. 141) only contains hydrologic unit codes for some watersheds and subwatersheds in the Project area and table contains no condition ratings. Further, neither Upper nor Lower Big Marsh Creeks are listed anywhere in the FSEIS as being within the Project area.
- The first row of in Table 75 (FSEIS, p. 148) which provides water temperature data in the Elliott Creek subwatershed does not add up in terms of sample numbers. This row in the table indicates that 113 seven-day averages of water temperature data were calculated, but the categorical data in the row adds up to 123 seven day averages. The sixth row in this table indicates that 121 seven day averages of water temperature were calculated, but the categorical data adds up to only 120 seven day averages. A reader is unable to discern which parts of these data on water temperature are correct.
- The FSEIS (p. 191) states that under Alt. 2 "[a]pproximately **1.3 miles of new trail** would be added within 300 feet of an intermittent or perennial stream..." in the Peterson Creek subwatershed. However, the following paragraph states that under the same alternative "approximately **3.3 miles of new trail** would be added within 300 feet of an intermittent or perennial stream..." in the Peterson Creek subwatershed. These are substantial differences with substantially different impacts on riparian areas and aquatic systems. Due to these contradictions, it is entirely unclear which statement is correct, or given the contradictions, if either statement is correct.
- The FSEIS (p. 194) states that under Alt. 2 proposed trails would "...cross streams at **ten new locations** ..." in the Jackson Creek subwatershed. However on this same page another paragraph states that Alt. 2 adds "...nine new intermittent or perennial stream

crossings to the existing number of stream crossings by roads in the Jackson Creek Subwatershed...” in the Jackson Creek subwatershed. These are substantial differences with substantially different impacts on riparian areas and aquatic systems. Due to these contradictions it is entirely unclear which statement is correct, or given the contradictions, if either statement is correct.

- The FSEIS (p. 199) states that under Alt. 2 proposed trails in the **Upper Ochoco Creek Watershed** include “...**approximately 3 miles of trail on closed roads, 0.1 mile on decommissioned road, and 8 miles on new routes.**” However on this same page in another paragraph, the FSEIS states that under Alt. 2 “...**approximately 8 miles of trail are proposed on closed roads, decommissioned, and open roads...**” and **about 5.3 miles of new trail** are proposed to be constructed in the Upper Marks Creek subwatershed, which is in the Upper Ochoco Creek watershed. These are substantially different with substantially different impacts on riparian areas and aquatic systems. Both statements cannot be correct. Due to these contradictions it is entirely unclear which statement is correct, or given the contradictions, if either statement is correct.
- The FSEIS (Table 101, p. 201) the first two columns of data indicate that under Alt. 2 there would be **25 crossings** by new routes and existing roads for OHV in the Elliott Creek subwatershed. However, in the same table, the columns of data for stream crossings by stream type indicate there would **29 crossings** of perennial, ephemeral, and intermittent streams by OHV routes in the Elliott Creek subwatershed. Further, in the same table the data for crossing type for the subwatershed only adds up to **24 crossings**. Crossings are points of major stream and riparian damage by a host of impacts, so these substantial differences in crossing numbers in the subwatershed would cause substantially different impacts. However, due to these contradictions it is not possible to know which, if any, of these crossing numbers are correct for the Elliott Creek subwatershed.
- The FSEIS (Table 101, p. 201) the first two columns of data indicate that under Alt. 2 there would be **22 stream crossings** by new routes and existing roads for OHV in the Jackson Creek subwatershed. However, in the same table, the data for the type of stream crossed indicates that there would be **31 stream crossings of perennial, ephemeral and intermittent streams** under Alt. 2 in the Jackson Creek subwatershed. These same problems occur for other subwatersheds listed in Table 101: the number of stream crossings by new and existing routes is consistently and significantly less than the number of crossings of perennial, ephemeral and intermittent streams provided in the table. Crossings are points of major stream and riparian damage by a host of impacts, so these substantial differences in crossing numbers in the subwatershed would cause substantially different impacts. Due to these contradictions in the data, it is not possible to know which, if any, of these crossing numbers in Table 101 are correct for the affected subwatersheds.
- The caption for Table 115 (FSEIS, p. 219) states that the table provides a comparison of road plus trail stream crossing densities by subwatershed for **Alt. 3 and Alt. 5**. However, in this table, the column header for the data on differences in crossings states that the data are for **Alt. 2 and Alt. 5**. Due to these contradictions it is impossible to ascertain which alternative the data corresponds to.
- The caption for Table 116 (FSEIS, p. 221) states that the table provides a comparison of miles of existing roads and proposed trails within 300 feet of streams for **Alt. 3 and Alt. 5**. However, in the Table the column headers indicate that mileage data for these routes

by subwatershed are for **Alt. 2 and Alt. 5**. The column header for the data on differences in this mileage indicates that the differences are for **Alt. 2 and Alt. 5**. Due to these contradictions it is impossible to ascertain which alternative the data corresponds to.

- The caption for Table 117 (FSEIS, p. 223) states that the table provides a comparison of stream crossings for **Alt. 3 and Alt. 5**, but the column headers indicate that the crossing data are for **Alt. 2 and Alt. 5**. Due to these contradictions it is impossible to ascertain which alternative the data is for. Due to the impacts of crossings on aquatic systems this ambiguity is highly significant.
- The narrative describing Table 118 (FSEIS, p. 222) states that the table "...displays estimated sediment contribution from road/trail crossings and road/trail segments within 300 feet of streams in project area subwatersheds for **Alternative 3 and Alternative 5**. The final column in the table compares the difference in sediment produced in tons/year between **Alternative 3 and Alternative 5**." (emph. added). However, the columns in Table 118 indicate that the data are for **Alt. 2 and 5**. This contradictory information creates confusion about significant aspects of the Project alternatives where clarity is absolutely essential for understanding the nature of the alternatives and their aquatic impacts.
- The FSEIS (p. 216) states: "For trail miles and crossings in RHCAs by watershed and by stream class, see Appendix D." However, the FSEIS has no Appendix D. As of 10/28/16, there was no Appendix D on ONF's online collection of environmental analyses associated with the Project: http://data.ecosystem-management.org/nepaweb/nepa_project_exp.php?project=26807 This is critical information related to the nature of the alternatives and their impacts, so this is a major error.

The foregoing is not an exhaustive listing of the major and pervasive contradictions in data regarding the location and nature of trail impacts associated with the action alternatives. However, it does amply indicate that these contradictions create confusion rather than clarity regarding these important aspects of the Project alternatives. As a result, the FSEIS has failed to clearly convey the nature of the action alternatives.

SEDIMENT-RELATED IMPACTS OF THE ACTION ALTERNATIVES

The FSEIS's assessment of sediment-related impacts of the action alternatives and cumulative effects of these alternatives and existing conditions remains extremely defective in several ways. These defects are significant, because the cumulative effect of sediment delivery from roads, routes, and livestock grazing on the ONF has adverse effects on host of stream conditions, including those set as Riparian Management Objectives (RMOs) in INFISH. RMOs affected by sediment delivery include those for pools, stream width/depth ratio, and water temperature. Therefore, the FSEIS fails to properly assess compliance with these INFISH RMOs and related INFISH standards, due to the defects in the FSEIS's assessment of cumulative sediment delivery under the action alternatives. Due to the manifold defects related to sediment impacts, the FSEIS fails to properly disclose the alternatives' effects on stream conditions, water quality, and aquatic biota.

The sediment-related degradation of pools, width/depth ratio, and water temperature adversely affects redband trout in several ways. Elevated sediment delivery also adversely affects fine sediment levels, which profoundly reduces the survival and production of redband trout. However, the FSEIS does not properly assess the alternatives' direct and cumulative impacts on fine sediment levels and resulting impacts on redband trout. In so doing, the FSEIS failed to properly assess the action alternatives' inconsistency with INFISH standard RM-1 which requires the avoidance of adverse impacts from recreation activities and facilities on inland fish. These numerous, serious defects persist despite my previous comments which clearly identified these defects.

The FSEIS did not properly assess cumulative sediment delivery under the action alternatives.

Livestock grazing

An overarching flaw in the FSEIS's assessment of sediment-related impacts is that the FSEIS did not assess the cumulative impacts of sediment delivery under the action alternatives. The FSEIS's repeatedly admits (e.g., pp. A-46, A-47, A-116) that its estimates of sediment delivery at the scale of watersheds and subwatersheds completely ignore sediment delivery from livestock grazing. This fatal flaw with respect to cumulative impacts on aquatic systems persists despite my previous comments and those of many others, including ODFW. This is major failure in the FSEIS is a severe defect for several reasons.

First, it is extremely well-established that livestock grazing greatly elevates sediment delivery to streams via several mechanisms, including direct bank damage from trampling, loss of bank stability, gullying, loss of soil cover, increased runoff due to soil compaction, and vastly elevated levels of channel erosion, which deposits sediment directly into streams (Rhodes et al., 1994; USFS and USBLM, 1997; Beschta et al., 2013). It is well-documented that grazing impacts that increase are particularly profound in riparian areas and that grazing greatly increases bank erosion.

Second, the failure to assess management-induced sediment delivery from grazing is a major flaw due to the extent of livestock grazing within the Project area. There are portions of 25 active allotments within the Project area (FSEIS, p. 52). Although the FSEIS fails to disclose the total area subjected to grazing within the Project area and affected subwatersheds,³ the FSEIS states (p. 388) "Cattle grazing occurs on most of the project area." Based on the extent of grazing and its innate impacts, it is likely that accelerated sediment delivery from grazing is a highly significant fraction of the total management-induced sediment delivery under the action alternatives in all watersheds and subwatersheds. Therefore, the FSEIS clearly failed to properly determine cumulative sediment delivery under the action alternatives and related aquatic impacts due to the FSEIS's failure to include management-induced sediment delivery from grazing impacts.

Third, sediment-related channel conditions, such as pools, width/depth ratio, and fine sediment levels are affected by cumulative sediment delivery. The increased sediment delivery to streams

³ This failure persists despite my comments on the FEIS, which discussed the necessity of disclosing and assessing it at the scale of subwatersheds and the Project area.

from grazing is significant and has many negative impacts on aquatic conditions, including those that are INFISH RMOs and adversely affect redband trout. As the Forest Service acknowledged in USFS and USBLM (1997):⁴

“Grazing is a major nonpoint source of channel sedimentation (Dunne and Leopold 1978; MacDonald and others 1991; Meehan 1991; Platts 1991). Grazed watersheds typically have higher stream sediment levels than ungrazed watersheds (Lusby 1970; Platts 1991; Rich and others 1992; Scully and Petrosky 1991). Increased sedimentation is the result of grazing effects on soils (compaction), vegetation (elimination), hydrology (channel incision, overland flow), and bank erosion (sloughing) (Kauffman and others 1983; MacDonald and others 1991; Parsons 1965; Platts 1981a; 1981b; Rhodes and others 1994). Sediment loads that exceed natural background levels can fill pools, silt spawning gravels, decrease channel stability, modify channel morphology, and reduce survival of emerging salmon fry (Burton and others 1993; Everest and others 1987; MacDonald and others 1991; Meehan 1991; Rhodes and others 1994)...Compared to ungrazed sites, aquatic insect communities in stream reaches associated with grazing activities often are composed of organisms more tolerant of increased silt levels, increased levels of total alkalinity and mean conductivity, and elevated water temperatures (Rinne 1988).”

Available scientific evidence clearly demonstrates that elevated cumulative sediment delivery, including that from grazing, degrades width/depth ratios (Rhodes et al., 1994; Beschta et al., 2013), the frequency, volume, and quality of pools (USFS et al., 1993; Rhodes et al., 1994; USFS and USBLM, 1997; McIntosh et al., 2000), both of which are RMOs. Therefore, it is impossible for the FSEIS to assess compliance with INFISH RMO standards affected by sediment delivery in the absence of a thorough assessment of sediment delivery from grazing.

Increased sediment delivery from grazing also degrades water quality by elevating suspended sediment levels and turbidity. Elevated suspended sediment adversely affects fish, water quality, and water resources (Rhodes et al., 1994). Elevated turbidity can contribute to violations of state water quality standards for turbidity.

Numerous lab and field studies have repeatedly documented that increases in cumulative sediment delivery increase fine sediment levels in streams (Rhodes et al., 1994; USFS and USBLM, 1997; Buffington and Montgomery, 1999; Hassan and Church, 2000; Kappesser, 2002; Cover et al., 2008). Thus, sediment delivery from grazing, which is a significant fraction of cumulative sediment delivery in affected subwatersheds, must be assessed in order to reasonably evaluate cumulative effects on fine sediment levels in those watersheds.

For these reasons, credible assessment of impacts on turbidity, pools, width/depth ratio, fine sediment levels, and other sediment-related impacts on fish require the assessment of the impacts of cumulative sediment delivery on these stream conditions.

⁴ USFS and USBLM (1997) assessed aquatic conditions and the effects of management activities on USFS and USBLM lands in the interior Columbia basin, including the ONF. This salient finding by the USFS is never disclosed in the FSEIS or any of the previous EIS's for the Project, although it was provided to the ONF in my previous comments.

However, the FSEIS repeatedly acknowledges that it fails to include any quantitative estimate of cumulative sediment delivery, including that from grazing. Instead, the FSEIS only provides some estimates of sediment delivery from roads and trails, which are not a surrogate for assessment of cumulative management-induced sediment delivery, because it may not even constitute the majority of management-induced sediment delivery. Therefore, as will be discussed in greater detail, the FSEIS's assessment of the action alternatives' compliance with INFISH is completely flawed due to the FSEIS's failure to properly evaluate the effects of cumulative management-induced sediment delivery under the action alternatives, including that from pervasive grazing impacts, on RMO standards for width/depth ratio and pools.

The FSEIS's complete failure to assess cumulative sediment delivery to streams, including that from livestock grazing, also renders the FSEIS's assessment of the fine sediment impacts highly defective. The FSEIS incorrectly assumed (e.g., pp. 137) that sediment delivery to streams that is below an assumed threshold of 35%⁵ over the estimated natural sediment delivery background does not increase fine sediment levels. While available scientific information, including that in the model that the ONF used incorrectly to assess in-channel sediment impacts (Stowell et al., 1983), indicates that this impact threshold is not valid, the FSEIS failed to reasonably assess if the magnitude of cumulative sediment delivery in Project watersheds and subwatersheds is greater than 35% over natural background, because the FSEIS ignored major and pervasive contributions to sediment delivery from grazing. This is highly significant because, as previously noted, livestock grazing, even in the absence of other impacts, elevates fine sediment levels in streams (Rhodes et al., 1994; USFS and USBLM, 1997). It is highly likely that if management-induced sediment delivery from grazing had been properly assessed and included in a cumulative sediment delivery estimates, it would show that cumulative management-induced sediment delivery under the action alternatives in Project area watersheds and subwatersheds is more than 35% over estimated natural background levels. Therefore, the failure to include sediment delivery from grazing impacts is a severe defect with respect to the FSEIS's assessment of impacts on fine sediment levels under the action alternatives.

Proposed OHV routes

The FSEIS estimates of sediment delivery from OHV routes is also defective because these estimates do not reflect that available data has consistently shown that sediment delivery from OHV routes is far greater than from roads. The FSEIS (p. A-46) relied on erosion rates for roads to estimate sediment delivery from OHV routes within 300 feet of streams and at stream crossings.⁶ This results in considerable and consistent underestimation of sediment delivery from OHV routes because field studies have consistently demonstrated that sediment delivery

⁵ The significant problems with the assumed fine sediment impact threshold in the FSEIS is discussed in greater detail in these comments. This was also communicated to the ONF in my comments on the FEIS. Despite this information, the FSEIS still grossly mischaracterizes the information in Stowell et al. (1983) in ways that distort the information in Stowell et al. (1983) regarding sediment delivery and fine sediment impacts.

⁶ The FSEIS (p. A-46) incorrectly asserts that the modeling approach is conservative because OHVs cause less damage to road surfaces than heavier log trucks for which road erosion rates are derived. However, Meadows et al. (2008) which is cited in FSEIS clearly documented that lighter OHVs caused as much damage to road surfaces and elevated erosion than much heavier vehicles. Therefore, the FSEIS once again distorts available information rather than reporting it properly.

from OHV routes is several times higher than from roads (Foltz, 2006; Meadows et al., 2008; Welsh, 2008). Welsh (2008) documented that mean sediment production from roads was **six times higher** than that from roads and that the WEPP model used in the FSEIS failed to accurately estimate sediment production from OHV routes, because it consistently underestimated sediment production from these routes. For these reasons, it the FSEIS significantly underestimated sediment delivery from OHV routes, and, hence impacts on sediment-related conditions, including turbidity, width/depth ratios, pools, and fine sediment levels in streams.

Notably, the FSEIS (p. 238) incorrectly asserts that the documentation of far greater erosion and sediment delivery from OHV routes relative to roads (Foltz, 2006; Meadows et al., 2008; Welsh, 2008) is not relevant to the Project because the results are from “unmanaged” OHV use. However, this is a baseless assumption in the FSEIS. None of these studies indicate that the results were from uncontrolled, unmanaged use. Therefore, the FSEIS assumption is baseless, as is its effort to ignore available scientific evidence documented by USFS research on OHV impacts on erosion and sediment delivery.

The FSEIS (A-63) further obfuscates these documented scientific findings by incorrectly characterizing the discussion of these results in my FEIS comments as being from “illegal OHV trails⁷ that have no drainage features or other water quality design criteria applied to them.” There is absolutely no information in these studies comments indicating that the studied trails had no drainage features. These study results were certainly not from “illegal trails,” but rather trails open to OHV use on several forests. These incorrect characterizations in the FSEIS further occlude, rather than properly disclose, the documented effects of OHVs on erosion and sediment delivery from OHV routes proposed under the Project’s action alternatives. Further, the FSEIS is devoid of an iota of scientific documented information indicating that “managed” OHV use results in fundamentally different levels of erosion and sediment from OHV trails than those documented in these studies (Foltz, 2006; Meadows et al., 2008; Welsh, 2008).

Existing user-created OHV routes

The FSEIS concedes that it did not estimate sediment delivery from existing user-created routes. The FSEIS (p. A-69) admits that FSEIS is does not even include information on the extent, location, and density of user-created trails at the scale of watersheds and subwatersheds, despite repeated comments on the FEIS that this was crucial to disclose (FSEIS, p. A-69). Absent information on extent and location of the routes, the FSEIS does not even have an index of the sediment-related impacts of these existing routes. This is a fatal error with respect to impact assessment of the action alternatives for two major reasons.

First, several RMOs, including pools, width/depth ratio, and water temperature are affected by *cumulative* sediment delivery at the scale of subwatersheds and watersheds. Sediment delivery from user-created routes contributes to cumulative sediment delivery. Therefore, by failing to assess sediment delivery from these routes, the FSEIS fails to properly assess cumulative impacts

⁷ It is ironic that FSEIS baselessly assumes the trails in the cited research are “illegal” because the FSEIS repeatedly notes that routes are not illegal if they predate a signed Travel Management Plan (e.g. FSEIS, pp. A-129 A-144). The cited research contains no information that the trails studied came after a signed Travel Management Plan.

on RMOs under the action alternatives. Thus, the FSEIS does not reasonably assess compliance with INFISH, due to the failure to incorporate sediment delivery from these routes in its analysis of RMO impacts.

The FSEIS indicates that the ONF believes that sediment delivery from these user-created routes is substantial because FSEIS repeatedly states that sediment delivery in the Project area will be reduced as sediment delivery from user-created routes abate, despite increased erosion and sediment delivery from OHV trails under the action alternatives. Taking this at face-value clearly indicates that the FSEIS omits a major source of management-induced sediment delivery in its purported cumulative effects analysis for the alternatives, because FSEIS's estimates of the sediment delivery do not include that from user-created routes.

Similarly, cumulative sediment delivery also affects turbidity and fine sediment levels, which affect the survival and production of redband trout. Thus, by failing to include sediment delivery from user-created routes, the FSEIS fails to properly ascertain the cumulative effects of the action alternatives on fine sediment levels, red band trout, and turbidity. Because the FSEIS does not properly assess cumulative impacts on turbidity, it has failed to reasonably determine compliance with state water quality standards for turbidity.

Second, the FSEIS repeatedly asserts, without any sound basis, that cumulative sediment delivery under the action alternatives will ultimately decrease below existing conditions, due to remediation measures on user-created trails under the action alternatives. Plainly, this assertion is baseless without a quantitative notion of the magnitude of existing sediment delivery from user-created routes. Therefore, this repeated assertion in the FSEIS that cumulative sediment delivery will eventually be lower than under the no-action alternative has absolutely no merit.

However, there are several other severe problems with the FSEIS's baseless assertion that the action alternatives would significantly reduce sediment delivery in the Project area due to remedial treatments on user-created trails. First, the FSEIS fails to provide any estimate of the magnitude of these trails would be treated under the action alternatives. Absent such a credible estimate, it is impossible to reasonably estimate any magnitude of sediment reduction on user-created trails in the Project area.

Second, the FSEIS provides no notion of the location of these trails that might be treated under the action alternatives. Similarly the FSEIS provides no information on stream proximity of these trails. Both factors strongly influence sediment delivery to streams. Thus, due to the absence of this information the FSEIS's assertions regarding sediment delivery from user-created routes under the action alternatives has no merit.

Third, the FSEIS provides no specifics on treatments that would be used to attempt to reduce sediment delivery from user-created roads. Absent such specifics, effectiveness is unknown.

Fourth, the FSEIS suggests that additional efforts will be made to effectively close these routes to use. However, information in the FSEIS (p, 186) indicates that route closures have not been effective at preventing OHV use of closed routes.

Fifth, even trail obliteration and ripping, which are relatively expensive treatments, are not particularly effective at reducing erosion and runoff, even over the course of several years. USFS research has shown that such treatments result in nominal improvement in conditions that affect erosion and sediment delivery from the vehicular routes, even after several years and that data indicates that it is unclear if complete recovery would ever occur even with these treatments (Foltz et al., 2007). Hence, it is unlikely that even with relatively costly treatments that sediment delivery from treated user-created routes would ever be appreciably reduced.

Sixth, funding for these treatments under the action alternatives is not reasonably certain to occur. In fact, the FSEIS considers the probability of obtaining such funding to be outside of the scope of the Project. Thus, the unspecified funding-dependent treatments are far from reasonably certain to occur.

In aggregate, these problems fully undermine the FSEIS's repeated assertion that under the action alternatives sediment delivery in the Project area would eventually be reduced relative to existing conditions. Therefore, the FSEIS misleads rather than making known the likely impacts of the alternatives on sediment delivery and sediment-related impacts on RMOs, fine sediment levels, redband trout, and water quality.

Logging

It is extremely well-documented that logging elevates sediment delivery for several years (USFS, 1993; Rhodes et al., 1994; USFS and USBLM, 1997). Copious amounts of logging have occurred in many watersheds based on the Equivalent Harvest Acres (EHA) estimates in the FSEIS (p. 231). There is no indication that the FSEIS included any effort to estimate sediment delivery from logging and include it in estimates of cumulative sediment delivery under the action alternatives at the scale of watersheds and subwatersheds.

This is a major defect for the same reasons as for the omission of sediment delivery from other sources in the FSEIS. Cumulative sediment delivery affects several RMOs, fine sediment levels in streams, redband trout survival and production, and water quality. Thus, by failing to estimate sediment delivery from logging as part of cumulative sediment delivery under the action alternatives the FSEIS fails to properly assess compliance with INFISH and effects on RMOs, redband trout, and water quality.

Conclusion

The FSEIS clearly did not properly estimate management-induced sediment delivery under the action alternatives because it omitted major contributions from pervasive grazing, user-created routes, and logging. Due to the extent and nature of these sources of management-induced sediment delivery, it is likely that the sediment delivery estimates in the FSEIS for OHV trails only constitute a minor fraction of management-induced sediment delivery in affected watersheds and subwatersheds. For these reasons, the FSEIS fails to properly assess cumulative effects of sediment delivery under the action alternatives on a host of important conditions, including RMOs, INFISH compliance, redband trout, and water quality.

The FSEIS did not properly assess the adverse cumulative impacts on fine sediment levels in streams and resulting adverse effects on redband trout under the action alternatives.

The FSEIS failed to properly determine that the cumulative effects under the action alternatives will increase fine sediment levels and thereby adversely affect redband trout. This failure is due to several severe defects in the FSEIS's assessment of cumulative sediment delivery, its effects on fine sediment, and resulting impacts on redband trout.

First, the FSEIS (e.g., pp. 137, 185, 189, 193) consistently and incorrectly assumed that increases in watershed and subwatershed sediment delivery that are less than 35% over natural background levels have no significant effect on fine sediment conditions in C-type channels. This assumption is not only unwarranted, but in direct conflict with available scientific information, including the very approach the FSEIS incorrectly cites as the basis for this claim (Stowell et al., 1983). Contrary to the FSEIS's unsound assumption regarding sediment delivery and fine sediment impacts, numerous field and laboratory studies have consistently demonstrated that increases in sediment delivery increase fine sediment in streams, especially when the sediment delivery is primarily comprised of fine sediment (Rhodes et al., 1994; Buffington and Montgomery, 1999; Hassan and Church, 2000; Cover et al., 2008). Notably, management-induced sediment delivery from roads, OHV trails, and grazing is primarily comprised of fine sediment. Therefore, contrary to the unsound assumption in the FSEIS, increases in sediment delivery, both cumulative and from the OHV trails, will contribute to increases in fine sediment.

The FSEIS relies solely Stowell et al. (1983) for the incorrect notion that management-induced sediment delivery less than 35% over natural background levels has no significant effect on fine sediment conditions and stream aggradation. This is a gross mischaracterization of the explicit information in Stowell et al. (1983). The narrative in Stowell et al. (1983) clearly states that the model is based on the well-documented result that as sediment delivery increases, so to does fine sediment and cobble embeddedness in streams.

Stowell et al. (p. 24, 1983) clearly provide an equation relating levels of cumulative management-induced sediment delivery to increases fine sediment levels in a C-type channel, as follows:

$$FS_f = FS_e + 0.12*(x)$$

where: FS_e = existing fine sediment levels

x = **cumulative management-induced** sediment delivery as estimated by the **R1-R4 sediment delivery** model expressed in terms of percent estimated natural sediment delivery

FS_f = future fine sediment level caused by cumulative management-induced sediment delivery

Casual inspection of this equation clearly indicates that for **any** increment of increased sediment delivery, the model predicts an increase in fine sediment levels, which is in direct conflict with the FSEIS's spurious characterization of the results in Stowell et al. (1983). This response equation for fine sediment and sediment delivery indicates that threshold for zero increase in fine

sediment levels is zero management-induced sediment delivery, contrary to the gross misinterpretation in the FSEIS.

Further, Stowell et al. (p. 24, 1983) provides an explicit calculation of increased fine sediment using the foregoing equation in an example where existing fine sediment is initially 26.9% in a C-type channel and cumulative management-induced sediment delivery from the R1-R4 model is estimated to be **32%** over natural (well below the incorrect and mischaracterized ‘threshold’ the FSEIS ascribes to Stowell et al. 1983). This explicit example in Stowell et al. (1983) increases fine sediment levels from 26.9% to 30.7%. Here’s the excerpted calculation from a scan of Stowell et al. (1983):

$$Y_2 = \underline{\underline{26.9}} + 0.12(32) = 30.7\%$$

where: $Y_2 = FS_f$ = future fine sediment level caused by cumulative management-induced sediment delivery = 30.7% in this example

Plainly, this directly conflicts with the FSEIS’s fabrication that Stowell et al. (1983) indicates that there are no increases in fine sediment levels when management-induced sediment delivery is less 35% over natural sediment delivery.

Importantly, based on the survival curves in Stowell et al. (1983), this resulting increase in fine sediment levels from 26.9% to 30.7% would significantly reduce salmonid survival. Thus, the FSEIS relies on a complete distortion of the findings and methods of Stowell et al. (1983).

Stowell et al. (1983) indicates even larger increases in cobble embeddedness, which significantly impair salmonid survival and production, per increment of increase in cumulative management-induced sediment delivery. Stowell et al. (p. 24, 1983) clearly provide an equation relating levels of cumulative management-induced sediment delivery to increases in cobble embeddedness in a C-type channel, which is as follows:

$$CE_f = CE_e + 0.16*(x)$$

where: CE_e = existing cobble embeddedness

x = **cumulative management-induced** sediment delivery as estimated by the **R1-R4 sediment delivery** model expressed in terms of percent estimated natural sediment delivery

CE_f = future fine sediment level caused by cumulative management-induced sediment delivery.

Again, casual inspection of this equation by even a pedestrian analyst clearly indicates that for **any** increment of increased sediment delivery, the model predicts an increase in cobble embeddedness, which is in direct conflict with the FSEIS’s spurious characterization of the results in Stowell et al. (1983). This response equation for cobble embeddedness and sediment delivery does not have any threshold for management-induced sediment delivery below which an increase in cobble embeddedness is not predicted to occur, other than zero management-induced sediment delivery.

Stowell et al. (p. 24, 1983) provides an explicit calculation of increased cobble embeddedness using the foregoing equation in an example where existing cobble embeddedness is initially 36.4% in a C-type channel and cumulative management-induced sediment delivery from the R1-R4 model is estimated to be **32%** over natural (below the incorrect and mischaracterized ‘threshold’ the FSEIS ascribes to Stowell et al. 1983). This explicit example in Stowell et al. (1983) increases cobble embeddedness from 36.4% to 41.5%. Here’s the excerpted calculation from a scan of Stowell et al. (1983):

$$Y_2 = \underline{\underline{36.4}} + 0.16(32) = 41.5\% \text{ embeddedness}$$

$Y_2 = CE_f$ = future cobble embeddedness caused by cumulative management-induced sediment delivery = 41.5% in this example.

Importantly, based on the information Stowell et al. (1983), this resulting increase in cobble embeddedness from 36.4% to 41.5 % would significantly reduce the salmonid rearing capacity of aquatic habitat. Thus, the FSEIS relies solely on a distortion of the findings and methods of Stowell et al. (1983).

The foregoing clearly demonstrates that the FSEIS’s characterization of the information in Stowell et al. (1983) is fabricated and directly conflicts with the actual information in Stowell et al. (1983) regarding expected changes in substrate conditions due to management-induced sediment delivery. Because this fabrication is relied upon in the FSEIS for its assessment of sediment-related impacts on fine sediment conditions and redband trout, the FSEIS’s assessment of the cumulative sediment-related impacts on redband trout under the action alternatives is terminally defective.

I am quite familiar with Stowell et al. (1983), having used it many times over decades of work, as well as having evaluated it in detail at the behest of NMFS in a peer-reviewed report produced under contract with NMFS (Rhodes, 1996) during my tenure with the Columbia River Inter-Tribal Fish Commission. I also recently conferred about the model with Al Espinosa, retired USFS fish biologist, who performed most of the statistical analysis in Stowell et al. (1983), and is a primary author of the publication.

There are several other problems with the way that the FSEIS grossly misused the approach in Stowell et al. (1983). First, the documentation in Stowell et al. (1983) clearly state that it is **only** for use with the sediment delivery estimation methods in the R1-R4 model. The method of Stowell et al. (1983) can only be used with the estimates of cumulative management-induced sediment delivery from the R1-R4 sediment model because all of the response equations for fine sediment and sediment delivery were developed from calibrations with the R1-R4 output (*pers. comm.*, A. Espinosa, 10/23/16). However, the FSEIS used “WEPP” to estimate sediment delivery from proposed OHV routes (FSEIS, p. 140), which is not valid for use with Stowell et al. (1983).

Second, the method of Stowell et al. (1983) is based on estimating sediment response to **cumulative management-induced sediment delivery**, as corroborated by a primary author of Stowell et al. (1983) (*pers. comm.*, A. Espinosa, 10/23/16), because cumulative sediment

delivery affects fine sediment response in streams. As previously discussed, the FSEIS is devoid of a credible assessment of cumulative management-induced sediment delivery, because it ignored sediment delivery from grazing, logging, and user-created roads. It is not valid to use the model to estimate potential changes in fine sediment conditions based on only consideration of a fraction of the cumulative sediment delivery in watersheds (*pers. comm.*, A. Espinosa, 10/23/16), as was done in the FSEIS.

Notably, the WEPP model used in the FSEIS to estimate sediment delivery from routes is an inappropriate for estimating cumulative sediment delivery in the Project area. This is because the WEPP model is incapable of estimating accelerated channel erosion and it is well-documented that one of the principal ways that grazing elevates sediment delivery is via accelerated channel erosion (Rhodes et al., 1994; USFS and USBLM, 1997; Cowley, 2002; Beschta et al., 2013).

Third, Stowell et al. (1983) very clearly note that the method should use existing substrate conditions (cobble embeddedness and fine sediment levels) as the starting place for estimating impacts, as shown in the equations previously discussed. The FSEIS clearly did not do this. Instead, it repeatedly substitutes irrelevant statements about how Stowell et al. (1983) indicates that fine sediment levels of less than 20% have limited adverse impacts on redband trout, which steadfastly ignores that the available data in the FSEIS indicate that many streams in the project area have fine sediment levels that are well in excess of 20% (FSEIS, pp. 147, 155, 158, 163-165, 169, 173, 176) which Stowell et al. (1983) clearly indicates are reducing salmonid survival and production.

Fourth, the FSEIS never actually applied any of the equations in Stowell et al. (1983) for changes in fine sediment in response to sediment delivery. Instead, the FSEIS only repeats fabricated mischaracterizations of the information in Stowell et al. (1983) regarding stream substrate response to management-induced sediment delivery.

For these combined reasons, the FSEIS's use of Stowell et al. (1983) is invalid because it conflicts with the directions for how to use the model and the model itself. Due to the reliance on a complete misunderstanding of Stowell et al. (1983) the FSEIS's assessment of cumulative impacts on redband trout under the action alternatives is bankrupt.

Fine sediment conditions in streams in several subwatersheds and watersheds analyzed in the FSEIS conclusively demonstrate that assumption that management-induced sediment delivery as estimated in the FSEIS must exceed 35% over natural background levels to have a negative effect on fine sediment conditions is wholly unsound. Several of these streams have elevated levels of fine sediment (>20%) that harm redband trout survival and production even though the FSEIS erroneously estimates that sediment delivery is far below the assumed adverse effect threshold of 35% over natural background (FSEIS pp. 147-201). This site-specific data from the ONF clearly demonstrates that FSEIS's assumptions regarding the threshold of sediment delivery magnitude that causes elevated fine sediment levels that are injurious to redband trout is wholly unsound. For these combined reasons, the use of this threshold, which is in conflict with available scientific literature, including Stowell et al. (1983), and site-specific evidence from streams in the project area, is not sound. As a result of this unsound assumption, the FSEIS fails

to properly assess that the increases in sediment delivery caused by the action alternatives will elevate fine sediment levels that harm redband trout.

The FSEIS incorrectly assumes that fine sediment added to redband trout streams under the action alternatives do not harm redband trout and that fine sediment levels less than 20% do not reduce redband survival and production. Research has demonstrated that *any* increase in fine sediment significantly impairs steelhead growth and production (Suttle et al., 2004), which likely applies to redband trout.⁸ Bryce et al. (2010) demonstrated that fine sediment levels in excess of 13% cause significant alterations of aquatic food webs upon which redband depend. Therefore, this information strongly demonstrates that the FSEIS's assumptions regarding the threshold of fine sediment levels that adversely affect redband trout are unsound. Salient scientific information indicates that the additional sediment delivery generated by the action alternatives will adversely affect redband trout.

Rather than properly incorporating this information on fine sediment impacts on salmonids, the FSEIS (p. 236) incorrectly characterizes these findings by stating (emphasis added): "There are several pieces of literature (Suttle et al. 2004 and Bryce et al. 2010) that conclude any increase in fine sediment impairs steelhead (and similarly redband) growth and production or aquatic food webs. **However, these pieces of literature do not state that they may significantly impact steelhead growth and production or aquatic food webs.**" But, here's what Suttle et al. (2004) actually states (emphasis added)

"The linear relationship between deposited fine sediment and juvenile steelhead growth suggests that there is no threshold below which exacerbation of fine-sediment delivery and storage in gravel bedded rivers will be harmless,...It is widely known that salmonid stocks decline when land use increases fine-sediment delivery to gravel-bedded rivers...By confining juvenile steelhead over discreet patches of riverbed with experimentally imposed fine-sediment concentrations, we were able to investigate the mechanisms underlying previously observed patterns of salmonid declines in response to fine-sediment loading and storage... The decreases in steelhead growth and survival we observed with increasing fine-sediment deposition were associated with lower prey availability and higher activity, aggression, and risk of injury. Declines in growth rates lower survival of salmonids and other fishes...Differences in growth and survival imposed by fine sediment could therefore have important population-level impacts...If land uses that increase loading or decrease transport of fine sediments continue unabated, however, areas of formerly suitable juvenile rearing habitat may be lost from the riverbed long enough to cause irreversible population declines in resident salmonids. This concern is particularly important for juvenile salmonids, whose territoriality limits their ability to crowd into shrinking areas of good habitat... Our experiment demonstrates that fine-sediment deposition, even at low concentrations, can decrease growth and survival of juvenile salmonids. We find

⁸ Notably, the FSEIS relied on information for relationships between steelhead and fine sediment levels in Stowell et al. (1983). This indicates that ONF considers the response of redband and steelhead trout to fine sediment levels to be similar. Therefore, the findings of Suttle et al. (2004) are applicable: any additional increment of added fine sediment will adversely affect the survival and production of redband trout.

no threshold below which fine-sediment addition is harmless. These results suggest that any augmentation of fine-sediment deposition in steelhead bearing rivers in this region will further impair this potentially population-limiting life stage...”

Thus, the FSEIS grossly mischaracterized the findings of Suttle et al. (2004) and failed to properly incorporate it into the FSEIS analysis of the cumulative effects of the action alternatives on redband survival and production.

In contrast to the FSEIS’s mischaracterization, Bryce et al. (2010) states:

“Sediment has long been recognized as a leading cause of biological impairment in rivers and streams of the US... Fine sediments affect fish food sources, growth rates, migration, and reproduction...we focused on 4 sediment-sensitive fish species in the Mountains region—Chinook salmon (*Oncorhynchus tshawytscha*), bull trout (*Salvelinus confluentus*), rainbow trout (*Oncorhynchus mykiss*), and cutthroat trout (*Oncorhynchus clarkii*)—and a group of 8 sediment-sensitive aquatic macroinvertebrates ... Reiser and White (1988) also tested the survival of newly fertilized Chinook salmon eggs ... They found that declines in survival were delayed as fines were progressively added until % fines in the incubation gravel was 10% by volume, after which declines began and continued unabated. As injected sediments (0.85–4.6 mm) increased from 10 to 30%...survival declined from 63% to 25%... At reference (0%), low (4.6%), and medium (8.2%) levels of sediment, [rainbow trout] total emergence was between 85% and 90%. Emergence decreased to 78.5% when sediment was increased to 11.2% by volume. The sediment loadings also resulted in a number of nonemergent fry that were trapped beneath the sediment. The numbers of nonemergent (trapped) fry were significantly greater ($p < 0.05$) at medium and high amounts of sediments (8.2% and 11.2%, respectively) than they were at low and reference levels... using our results for sand and fines, regional managers might propose a 13% criterion for sand and fines, but apply a more restrictive 11% criterion to critical spawning areas for the more sediment-sensitive bull trout or Chinook salmon... After evaluating the available information, we concluded that minimum-effect bedded surficial sediment levels for aquatic vertebrates (fish and amphibians) were 5% and 13%, respectively, for % fines (≤ 0.06 mm) and % sand and fines (≤ 2 mm)”

Notably, these serious defects in the FSEIS persist despite information provided in comments on the FEIS demonstrating that the overall approach for assessing the cumulative sediment-related impacts on redband trout under the action alternatives was terminally defective. Instead of attempting to cure these defects, the FSEIS merely repeats them.

For these combined reasons, the FSEIS did not properly determine that the additional sediment delivery generated by the action alternatives will harm redband trout. As will be discussed, this caused the FSEIS to fail to properly determine that the action alternatives violate INFISH requirements.

THE FSEIS FAILED TO REASONABLY EXAMINE AND MAKE KNOWN THE VERY LIMITED EFFECTIVENESS OF BEST MANAGEMENT PRACTICES AND PROJECT DESIGN CRITERIA (PDC)

The state-of-the-art review of best management practices (BMPs) for roads done for the USEPA (GLEC, 2008) clearly concludes, based on a thorough examination of available scientific literature, that efforts to disconnect routes from streams are often ineffective in areas, particularly in areas with steep slopes and in close proximity to streams, particularly in the Pacific Northwest. It has long been known that it is not possible to eliminate greatly elevated sediment delivery and runoff impacts from routes at stream crossings (Kattelman, 1996; Beschta et al., 2004; Rhodes and Frissell, 2015). 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 routes near streams.

The extensive analysis of sediment data 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 forest removal elevated turbidity levels, which are an index of suspended sediment levels, despite modern BMPs (Klein et al., 2012). While these results were for areas dominated by rainfall, rather than snowmelt, they strongly indicate that BMPs do not eliminate sediment-related impacts of forest removal.

Notably, despite the FSEIS repeated assertions that BMPs and PDCs will minimize stream crossing impacts, the FSEIS is devoid of any scientific literature that supports this notion. It is long-established that the most effective way to minimize impacts from disturbance is to **avoid** locating disturbances in sensitive areas, such as RHCAs and stream crossings. The action alternatives do not do this.

While there is a considerable amount of scientific literature documenting the limited effectiveness of BMPs, the FSEIS steadfastly ignored this critically important information. In so doing, the FSEIS failed to reasonably assess and make known the impacts of the action alternatives on aquatic systems, including RMOs, redband trout, and water quality.

THE FSEIS FAILED TO PROPERLY DETERMINE THAT THE ACTION ALTERNATIVES VIOLATE INFISH REQUIREMENTS

The FSEIS did not reasonably determine that the action alternatives, including Alt. 5, do not avoid adverse impacts on redband trout, due to sediment-related impacts, which violates INFISH requirements.

As previously discussed, available information demonstrates that the increased sediment delivery under each of the action alternatives would harm redband trout. The FSEIS failed to properly assess these adverse impacts due to scientifically unsound assumptions combined with the failure to reasonably assess the cumulative impacts on sediment delivery.

Nonetheless, the FSEIS repeatedly acknowledges that the action alternatives would have significant negative effects on redband trout within the Project area. For instance, the FSEIS (p.

178) correctly notes that the consequences common to all action alternatives include: “Construction of culverts can increase sedimentation in streams that may reduce overall pool volume. Reducing pool volume for fish reduces hiding cover, resting and feeding areas and increases stream temperature. Construction of trails within the 300-foot sediment delivery zone would continue to deliver sediment into streams. In Class I and II streams, increased sediment would affect fish habitat by filling pools and increasing fine sediment affecting fish and frog reproduction success.” Thus, the action alternatives would clearly harm redband trout.

The FSEIS (p. 222) notes that the effects of Alternative 5 would “...essentially be the same for aquatic species as described in each subwatershed under Alternative 2.” The FSEIS (p. 187) notes that under Alternative 2 “Short-term sediment inputs from trail construction, especially at stream crossings, may have negative effects during construction when turbidity is high. These negative effects may include fish displacement during times of high turbidity, limited feeding, and possible mortality.”

These acknowledged impacts are certainly both adverse and serious, clearly indicating that adverse impacts on redband trout have not been avoided under any of the action alternatives.

The adverse impacts of the action alternatives on redband trout directly conflict with INFISH standard RM-1 which unambiguously requires the **avoidance of adverse impacts of recreation activities and facilities on inland fish, including redband trout**. Notably, as discussed, salient scientific information, which was arbitrarily ignored in the FSEIS, indicates that the adverse impacts on redband trout under the action alternatives are likely to be far greater than acknowledged in the FSEIS.

The FSEIS did not reasonably determine that sediment delivery under the action alternatives, including Alt. 5, will retard attainment of the width/depth ratio RMO in INFISH, which violates INFISH requirements.

Although stream channel width/depth ratio RMO adopted in the FSEIS is not sound, many streams in the Project area still do not meet this RMO standard (pp. FSEIS, pp. 145- 178). Available scientific information clearly indicates that increases in sediment delivery retard recovery of width/depth ratio (See: review in Rhodes et al., 1994). There is no question that the action alternatives will elevate sediment delivery, as previously discussed. Therefore, the action alternatives will retard the recovery of width/depth ratios in affected streams, which is impermissible under INFISH standard RM-1. This standard requires that the operation and design of recreation facilities, including trails, do not retard attainment of RMOs.

The FSEIS did not reasonably determine that impacts of the action alternatives, including Alt. 5, will retard attainment of the INFISH water temperature RMO, which violates INFISH requirements.

The FSEIS fails to credibly assess how the proposed action alternatives cumulatively affect water temperatures and attainment of the water temperature RMO in INFISH. This is a serious defect because many of the affected streams within the Project area do not meet state water temperature standards (FSEIS, p. 145-178) or the INFISH RMO for water temperature and are so thoroughly

degraded with respect to water temperature that many are listed as water quality limited under the Clean Water Act. Rather than reasonably assess the impacts of the routes on water temperatures, the FSEIS only asserts that the removal of shade-providing vegetation will be minimal. This is extremely inadequate for several reasons.

First, the loss of stream shade inexorably contributes to stream warming and retards the attainment of the INFISH water temperature RMO. The loss of stream shading is inevitable at newly constructed stream crossings at locations that provide stream shade. It is also inevitable with newly constructed routes that are proximate to streams. These impacts not only remove stream shade, but prevent the recovery of vegetation that provides shade and, thus, contribute to the recovery of thoroughly degraded water temperatures in the Project area. Therefore, the numerous new stream crossings and routes in RHCAs proposed under the action alternatives, will contribute to exacerbating stream temperature problems and retard attainment of the INFISH water temperature RMO. Importantly, the stream crossings and routes in RHCAs have long term negative impacts on water temperatures RMO attainment because OHV routes permanently prevent the recovery of vegetation that can provide stream shading and contribute to the attainment of the water temperature RMO in INFISH. Therefore, the action alternatives plainly retard water temperature recovery and thereby violate INFISH standard RM-1, although the FSEIS fails to make known this obvious impact of the action alternatives.

Second, stream crossings and routes in riparian areas contribute to water temperatures elevation and retard INFISH water temperature RMO attainment in several other ways besides the loss of stream shading. Stream crossings elevate water temperature via warmed route runoff from summer rain events that is delivered at stream crossings and other points that are hydrologically connected to streams (National Research Council (NRC), 2008). Runoff from routes occurs in response to even small precipitation events and this runoff is heated by warm road surfaces during summer. Notably, this thermal pollution from routes occurs when streams are already warm due to seasonal effects, elevating the adverse impacts on salmonids (Rhodes et al., 1994; McCullough, 1999).

Routes also elevate water temperatures due to their effects on subsurface runoff, by preventing infiltration of precipitation and shunting it to streams as surface runoff, thereby reducing subsurface flows to streams. Subsurface flow contributions are typically cooler than streamflows and, are critical to moderation of summer high water temperatures (Rhodes et al., 1994, Pollock et al., 2009). Thus, routes proximate to streams contribute to water temperature elevation and retard attainment of the water temperature RMO.

As previously discussed, OHV routes under the action alternatives will elevate sediment delivery, which contributes to increases in stream width and width/depth ratio. It is well-documented that stream widening, even in the absence of shade loss, elevates summer water temperatures (Rhodes et al., 1994; Bartholow, 2000; Pollock et al., 2009; Beschta et al., 2013). The increased sediment delivery under the action alternatives will retard the recovery of stream width/depth ratios, which will contribute to retarding the attainment of the INFISH water temperature RMO, although this obvious impact is never made known in the FSEIS.

OHV routes located in riparian areas under the action alternatives impair the microclimatic functions of riparian areas, which help moderate water temperatures in streams (USFS et al., 1993). Therefore, these proposed routes would retard the attainment of the water temperature RMO in INFISH.

Research in multiple watersheds has verified that stream temperatures tend to increase with increasing density of routes and stream crossings (Nelitz et al., 2005). This is likely due to the combined impacts of routes and route crossings on water temperatures, including shade loss, subsurface flow disruption, channel widening, and warmed runoff contributions. The action alternatives all increase the number and density of stream crossings, which available information indicates will retard the attainment of the INFISH water temperature RMO.

For these combined reasons, the FSEIS failed to properly assess the action alternatives' effects on the attainment of the INFISH water temperature RMO. In so doing, the FSEIS also failed to properly determine that the combined effects of these alternatives will retard attainment of the water temperature RMO in direct conflict with INFISH standard RM-1.

The FSEIS did not properly determine that impacts of the action alternatives, including Alt. 5, will retard attainment of the INFISH large woody debris RMO, which violates INFISH requirements.

The FSEIS fails to acknowledge that the action alternatives would reduce large woody debris (LWD), especially in the long-term, by permanently impairing LWD recruitment from routes that cross streams and are in RHCAs. USFS researchers (Meredith et al., 2014) have documented this long-term impairment of LWD levels due to routes near streams in the Columbia basin, although this is completely ignored in the FSEIS.

The loss of LWD is an extremely long-term impact, lasting more than a century in areas where trees are able to grow back after removal (Rhodes et al., 1994; McIntosh et al., 2000). However, the loss of LWD from route construction is even more persistent, because trees that could ultimately be recruited to streams would not be able to begin to grow on the routes for as long as the routes exist. This long-term loss of LWD under the action alternative will retard attainment of the INFISH LWD RMO, although the FSEIS fails to make known this obvious impact.

The FSEIS's argument that such reductions have no effect because they are localized and minimal is absurd. A local effect is still an effect that retards attainment of the LWD RMO. Such an effect is both easily measured and assessed. The counting of LWD is simple and straightforward. The FSEIS has no basis for the assumption that the effect would be minimal: the FSEIS is devoid of any quantitative assessment of the magnitude of the reduction of LWD recruitment under the action alternatives.

Regardless of the magnitude of the loss of LWD under the alternatives, the loss will inexorably retard the attainment of the INFISH LWD RMO and degrade wood levels in affected streams. Both impacts will be extremely persistent and irreversible. Therefore, the impacts of the action alternatives clearly retard the attainment of the INFISH LWD RMO in direct violation of INFISH standard RM-1.

The FSEIS failed to reasonably determine that impacts of the action alternatives, including Alt. 5, will retard attainment of the INFISH pool frequency RMO which violates INFISH requirements.

Studies have repeatedly shown that increased sediment delivery and the loss of LWD both contribute to the loss of pool frequency (USFS et al., 1993; Rhodes et al., 1994; McIntosh et al., 2000; Buffington et al., 2002). McIntosh et al. (2000) and USFS et al., (1993) noted that elevated sediment delivery and loss of LWD are the major causes of the documented loss of pool frequency in watersheds with logging, grazing, and roads. Meredith et al. (2014) documented that routes within about 280 feet of streams cause long-term LWD depletion and the action alternatives would include many routes within this distance. Thus, the action alternatives will contribute to pool loss by increasing sediment delivery and causing long-term loss of LWD. The FSEIS (p. 236) acknowledges that the action alternatives effects on sediment delivery could result in pool infilling by sediment, yet fail to address the obvious: this will retard attainment of the INFISH RMO for pools.

These impacts of the action alternatives will certainly retard the attainment of the pool RMO in INFISH, due to these alternatives' impacts on sediment delivery and LWD. As McIntosh et al.⁹ (2000) clearly noted in their regional assessment of pool loss: "The restoration of pool habitats will take even longer to recover if management activities that forestall recovery remain the status quo (e.g., road construction, riparian timber harvest, and livestock grazing leading to degraded riparian vegetation, low recruitment of large woody debris, and increased sediment delivery)." This is clearly applicable to the action alternatives, which would construct and reconstruct routes in riparian areas, remove trees from riparian areas, degrade riparian vegetation, and reduce LWD recruitment in a persistent fashion, while elevating sediment delivery. Therefore, the action alternatives will retard attainment of the INFISH RMO for pools in direct conflict with INFISH standard RM-1.

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⁹ Three of the co-authors of McIntosh et al. (2000) were USFS researchers.

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60-Day Notice of Intent to Sue

EXHIBIT A

Letter from Jonathan Rhodes, Hydrologist



Jonathan J. Rhodes
Hydrologist
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April 28, 2022

Oliver Stiefel
Staff Attorney, Crag Law Center
3141 E Burnside Street
Portland, OR 97214

Re: Review and evaluation of how the removal of trees greater than 21" dbh will significantly and persistently harm aquatic systems on USFS lands east of the Cascades.

Dear Mr. Stiefel,

I am a hydrologist with more than 35 years of experience, with a B.S. in hydrology from University of Arizona, a M.S. in hydrology and hydrogeology from University of Nevada-Reno and I finished all required academic work toward a Ph.D. in forest hydrology at the University of Washington. Since 2001, I worked as a consulting hydrologist for a variety of clients, including county and tribal governments in Oregon, Washington, and Idaho. Prior to that I worked for more than 12.5 years at the Columbia River Inter-Tribal Fish Commission (CRITFC), where I served as Senior Scientist-Hydrologist. My professional experience includes work with tribal, federal, state, county, and city governments, universities, and non-profit groups in eight western states. A copy of my curriculum vitae is attached.

Most of my work has focused on the effects of current and proposed uses of land and water on water quality, streams and habitats for native trout and salmon. I am quite familiar with the management direction in PACFISH and INFISH, including how the USFS has implemented activities on its lands in Eastern Oregon and Washington. Our work at the CRITFC provided much of the foundation of PACFISH. I am also quite familiar with riparian and stream conditions on these USFS lands.

I have co-authored numerous papers on the effects of logging (including thinning), roads, and other activities on watersheds and aquatic systems published in peer-reviewed scholarly

journals. I've served as a peer-reviewer for scholarly forestry and fishery journals, including the North American Journal of Fisheries and the Open Forest Science Journal, as well as the Proceedings of Forest-Fish Conference: Land Management Affecting Aquatic Ecosystems.

My evaluation of how the removal of trees greater than 21" dbh will significantly and persistently harm aquatic systems on USFS lands east of the Cascades is based on my education and experience. It is also based on my considerable experience with forest removal activities on those lands implemented under PACFISH, INFISH and other USFS management direction. **The allowance to log larger trees in Inland USFS lands will cause significant long-term harm to already highly degraded aquatic systems and imperiled fish.**

There are several reasons that aquatic systems and imperiled fish will be harmed by the allowance to remove larger trees. Most significantly, this loss of larger trees will contribute to the loss of downed large woody debris (LWD) in streams and riparian habitats. This will persistently degrade fish habitats and riparian functionality.

The allowance for removal of larger trees is likely to occur within Riparian Habitat Conservation Areas (RHCAs) established under PACFISH and INFISH, based on my reviews of numerous forest removal activities (hereinafter: "logging") implemented in the more than two decades since the adoption of PACFISH and INFISH on numerous inland National Forests. These projects have commonly involved logging within designated RHCAs, despite the guidelines and standards contained in PACFISH and INFISH.

A significant number of these larger trees in RHCAs, if left unmolested by the depredation from logging, would ultimately be recruited into streams via various means (windthrow, bank erosion, death from old age, etc), providing instream LWD. Larger trees within a given distance from streams in RHCAs are more likely to land in streams other factors being equal. Notably, larger trees are farther along the trajectory to age-related mortality, which is a significant LWD recruitment factor.

The loss of larger trees in and near RHCAs also contributes to long-term LWD losses from adjacent trees in RHCAs. This is because, when larger trees fall, they often knock down other trees that provide LWD to streams and riparian areas. Thus, logging of larger trees is likely to ultimately reduce instream LWD in perennial streams via several means.

The removal of larger trees outside of RHCAs on non-fish-bearing non-perennial streams will also reduce LWD levels in these streams. This is because the widths of RHCAs along these streams are not adequate to retain unimpaired levels of LWD recruitment, even if all trees in these RHCAs are retained. The full retention of all trees within one site potential tree height from streams is required to ultimately provide unimpaired levels of LWD recruitment (Rhodes et al., 1994), but RHCA widths on non-perennial streams without fish are much narrower than that under PACFISH and INFISH.

The loss of LWD in streams is extremely significant for several reasons. LWD is vital to the survival and production of imperiled salmonids and the functionality of riparian ecosystems.

As we noted in Karr et al., (2004): “...there is no debate about the key role that large trees play in aquatic systems...”

LWD is the primary pool-forming element in salmonid habitats (Buffington et al., 2002; Pollock and Beechie, 2014). Pools, particularly, large, deep pools, are essential to salmonid production and survival. They provide essential refugia and rearing and resting habitats (McIntosh et al., 2002). Instream LWD provides cover, a vital aspect of pool quality affecting salmonid production and survival (Pollock and Beechie, 2014). It also aids in moderating water temperature (Pollock and Beechie, 2014).

The largest pieces of LWD, which can only come from larger trees, are the most effective agent of pool formation. Larger LWD pieces serve as critical anchors for LWD jams which are important pool forming agents (Buffington et al., 2002) that are particularly important to aquatic systems due to multiple benefits they convey (Pollock and Beechie, 2014). The effectiveness of LWD in pool formation tends to increase with the size of LWD (Buffington et al., 2002). These larger pieces of LWD also have the greatest longevity in stream channels and provide longer duration of instream functionality. Larger trees are older, thus, the recovery time from LWD lost due to logging increases as the size of tree logged increases. The loss of LWD is an extremely persistent impact, lasting more than a century in areas where trees are able to grow back after removal (Rhodes et al., 1994; McIntosh et al., 2000).

Loss of LWD from large tree logging negatively impacts salmonids. Available data indicate that salmonid production increases as LWD increases (Rhodes et al., 1994).

LWD loss contributes to the loss of pool quality, volume, and frequency (McIntosh et al., 2000; Pollock and Beechie, 2014). Salmonid numbers tend to increase with pool volume, quality, and frequency (Rhodes et al., 1994). Pool loss reduces the production capability of salmonid habitat (Rhodes et al., 1994). Thus, the loss of LWD from larger trees incurs long-term significant impacts on salmonids and their habitats (McIntosh et al., 2004; Pollock and Beechie, 2014). The loss of salmonid survival and production caused by loss of larger LWD in streams is not reversible and is irretrievable.

LWD aids in reducing channel erosion and buffering sediment inputs by providing sediment storage in headwater streams (Rhodes et al., 1994). This is an important function even in non-perennial streams because these streams comprise the vast majority of the stream network and provide most of the sediment to salmonid habitats.

LWD provides important sources of organic matter and nutrients in soils, which are vital to the long-term maintenance and protection of soil productivity (USFS and USBLM, 1997a; Karr et al., 2004). Soil productivity is critical to forest vegetation and a host of forest functions.

LWD in riparian areas aids in limiting stream damage from livestock grazing. This is not trivial, because livestock grazing is a major source of damage to salmonid habitats in the affected area (USFS and USBLM, 1997a; b). LWD helps to limit this damage by impeding livestock access to streams. The largest pieces of downed LWD in riparian areas provide the most effective impediment to livestock access to streams.

Any loss of LWD is significant due to its importance to stream habitats and salmonids coupled with their existing conditions. Pool loss in salmonid habitats has been extensive and severe over the past several decades in logged and roaded watersheds draining USFS lands in the interior Columbia basin (USFS and USBLM, 1997a; McIntosh et al., 2000). LWD loss has been a major contributing cause of this pool loss. The loss of pools and LWD has been repeatedly implicated as a significant factor in the widespread decline in the range and numbers of salmonid populations (Rhodes et al., 1994; McIntosh et al., 2000; Pollock and Beechie, 2014). Past logging and roading has greatly depleted existing LWD sources in many riparian areas throughout the affected region. This is extremely significant because one of most important roles of riparian forests with respect to aquatic biodiversity is production of LWD from large trees big enough to anchor wood jams and form pools (Pollock and Beechie, 2014). Instream LWD levels are widely depleted, and the allowance to log larger trees will exacerbate this long-term aquatic affliction. Salmonid populations throughout the affected area are imperiled (USFS and USBLM, 1997a). Additional degradation due to LWD loss contributes to the likelihood of increased extirpation of these populations and is inimical to the recovery of these salmonid populations. The time needed to recover from the loss of LWD caused by large tree logging is on the order of century or more. For these combined reasons, the loss of any sources of LWD from larger trees is ecologically significant.

It is extremely unlikely that the removal of larger trees in riparian areas would convey any potential benefits to aquatic systems. Logging in riparian areas under the rubric of “thinning” has been shown to reduce LWD (Pollock and Beechie, 2014). Similarly, such logging has been repeatedly shown to be highly unlikely to alter fire behavior, because it is extremely unlikely that fire will encounter treated areas during the short period of time that fuels have been temporarily reduced by treatments aimed at fuel reduction (Rhodes and Baker, 2008; Price et al., 2015).

My decades of experience with forest management activities on lands implemented under PACFISH and INFISH amply demonstrate that riparian logging has often been approved under the guise of restoring forest structure and/or fire behavior. While PACFISH and INFISH contain subjective direction that projects are not to retard or prevent attainment of riparian management objectives, in practice, this has not meant that that riparian logging is prohibited, or that negative impacts from such logging have not occurred. Instead, riparian logging activities approved under PACFISH and INFISH have frequently impacted aquatic ecosystem functions. This trend will continue, based on the track record of the USFS over the more than two decades since the adoption of PACFISH and INFISH. In my professional opinion, based on abundant scientific information, these impacts will be more severe and long-lasting now that large tree logging is allowed.

Please don't hesitate to contact me with any questions or follow-up related to any aspect of this work.

Sincerely,

Jonathan J. Rhodes
Hydrologist

Literature Cited

Buffington, J.M., Lisle, T.E., Woodsmith, R.D., and Hilton, S., 2002. Controls on the size and occurrence of pools in coarse-grained forest rivers. *River Res. Applications*, 18: 507-531.

Karr, J.R., Rhodes, J.J., and five others, 2004. Postfire salvage logging's effects on aquatic ecosystems in the American West. *BioScience*, 54: 1029-1033.

McIntosh, B.A. and four others, 2000. Historical changes in pool habitats in the Columbia River Basin. *Ecological Applications*, 10: 1478-1496.

Pollock, M. M. and Beechie, T. J., 2014. Does Riparian Forest Restoration Thinning Enhance Biodiversity? The Ecological Importance of Large Wood. *JAWRA*, 50: 543-559.

Price, O.F., Pausas, J.G., and five others, 2015. Global patterns in fire leverage: the response of annual area burnt to previous fire. *International Journal of Wildland Fire*, 24: 297-306.

Rhodes, J.J., McCullough, D.A., and Espinosa Jr., F.A., 1994. A Coarse Screening Process for Evaluation of the Effects of Land Management Activities on Salmon Spawning and Rearing Habitat in ESA Consultations. CRITFC Tech. Rept. 94-4, Portland, Or.

Rhodes, J. J., and Baker, W. L., (2008). Fire probability, fuel treatment effectiveness and ecological tradeoffs in western US public forests. *The Open Forest Science Journal*, 1(1).

USFS and USBLM, 1997a. The Assessment of Ecosystem Components in the Interior Columbia Basin and Portions of the Klamath and Great Basins, Volumes I-IV. PNW-GTR-405, USFS, Walla Walla Washington.

USFS and USBLM (Bureau of Land Management), 1997b. The DEIS for the "Eastside" Planning Area. USDA Forest Service, Walla Walla, Washington

Curriculum Vitae: Jonathan J. Rhodes

Hydrologist

EDUCATION

1989: Doctoral candidacy degree in forest hydrology at the Univ. of Wash. Completed all requirements but dissertation.

1985: M.S. in Hydrology and Hydrogeology at the Univ. of Nev.-Reno. Thesis topic: The influence of seasonal stream runoff patterns on water quality.

1981: B.S. in Hydrology and Water Resources at the Univ. of Ariz.

PROFESSIONAL HISTORY

Sept. 2001 -- present. Principal Hydrologist, Planeto Azul Hydrology. Main duties: Analysis of water and land use effects on streams and aquatic resources, including native salmonids and their habitats; diagnosis of watershed and stream conditions; stream monitoring; development of programmatic and site-specific watershed and stream protection measures; project management. Some projects (and clients): Analysis of potential effects of groundwater pumping on streamflow (Conf. Tribes of the Umatilla Indian Reservation, OR); diagnosis of watershed and stream conditions in an urbanized watershed (West Multnomah Soil and Water Conservation District, OR); analysis of data on sediment effects on ESA-listed salmon in the South Fork Stillaguamish River, WA (Snohomish County, WA).

Aug. 1990 -- Sept. 2001. Consulting hydrologist for non-profit organizations. Past projects (and clients) include: hydrologic characterization of remnant marsh proposed as urban wildlife refuge/greenspace (Multnomah Co. Parks Dept, OR); review of aquatic effects of: quarry expansion (Friends of Forest Park, OR), urban construction (homeowners consortium, W. Linn, OR); forest manipulations on streamflow (Pacific Rivers Council).

Apr. 1989 -- Sept. 2001. Senior Fishery Scientist-Hydrologist, Columbia River Inter-Tribal Fish Commission. Main duties: Administration and implementation of projects monitoring channel change from land use; development of programmatic and site-specific land management plans to ensure protection of watershed integrity, water quality and aquatic resources; development of restoration plans for watersheds degraded by grazing, roads, logging, and mining; design of plans for monitoring watershed and stream erosion, sedimentation, water quality, and habitat conditions; review of land management plans for adequacy of protection of aquatic resources; field evaluation of watershed and channel conditions throughout the Columbia Basin; expert witness testimony; development of technical recommendations for policy staff for protection of natal habitat for anadromous fish; review of state and federal aquatic resource monitoring plans; report and proposal writing; and, participation in various state and federal technical work groups.

Aug. '84 -- Apr. '89. Research assistant, College of Forestry, Univ. of Wash. Main duties: analysis and interpretation of water quality-quantity data; technical report writing; design and maintenance of water chemistry and quantity monitoring network in a coastal forested watershed; training in data acquisition techniques; public presentation of findings.

July -- Oct. 1987 and May -- Oct. 1988. Consulting hydrologist, Tahoe Regional Planning Association, CA and NV. Main duties: field delineation and mapping of riparian zones, wetlands, and erosion-prone areas.

June -- Sept. 1985 and July 1986. Research assistant, Dept. of Geophysics, Univ of Wash. Main duties: operation of field station for glacier research on Mt. Olympus, Wash.; measurement of snow and glacier melt rates; mapping of supra- and extra- glacial streams contributing to basal sub-glacial flow rates on surging and non-surging glaciers in the Alaska Range, Alaska.

Jan. 1984. Consultant with C.M. Skau, Reno, NV. Main duties: field evaluation of logging roads for erosion potential and sedimentation risk; recommendations for placement of future roads to minimize erosion and sediment delivery to fish-bearing streams in coastal Northern California.

Oct. 1983 -- June 1984. Hydrologic Tech., USGS, Carson City, NV. Main duties: aid in development and calibration of predictive water quality model for the Truckee River; statistical analysis of water quality data; identification and quantification of non-point sources of nutrients to Truckee River, NV.

Aug. 1981 -- Sept. 1983. Research Assistant, Univ. of Nev.-Reno. Main duties: design and installation of instrument network to monitor water chemistry and quantity in a small, forested alpine watershed in the Sierra Nevada; water quality sampling; data interpretation and management; preparation of reports, grant proposals, and publications, computer programming for data reduction and storage; mapping of geology, soils and runoff-producing areas; and, training of field technicians.

Feb. -- May 1981. Water Quality Intern, Pima Assoc. of Gov'ts., Tucson, AZ. Main duties: water quality sampling of agricultural production wells; mapping of groundwater levels; and, coordination of sampling efforts.

PROFESSIONAL SERVICE

May 2009 – present. Peer Reviewer for the scholarly journal, Open Forest Science Journal, for papers related to hydrology and forest and watershed responses to disturbance.

Feb. 2010. Invited Guest Lecturer, Lewis and Clark School of Law course on public lands law: “PACFISH and INFISH and Imperiled Salmonids on Public Lands” Portland, OR.

Feb. 2009. Invited Guest Lecturer, Lewis and Clark School of Law course on public lands law: “PACFISH and INFISH and Imperiled Salmonids on Public Lands” Portland, OR.

Feb. 2008. Invited Guest Lecturer, Lewis and Clark School of Law course on public lands law: “PACFISH and INFISH and Imperiled Salmonids on Public Lands” Portland, OR.

Mar. 2007. Invited Panel Speaker, International Environmental Law Conference: “Fuel Treatments & Thinning: Its Impacts and Low Priority Relative to Other Needed Restoration Measures,” Univ. of OR, Eugene, OR.

Mar. 2007. Invited Panel Speaker, International Environmental Law Conference: “The Impacts of Livestock Grazing on Water Quality and Trout Habitats,” Univ. of OR, Eugene, OR.

Feb. 2005. Invited Guest Lecturer, Lewis and Clark School of Law course on public lands law: “Postfire Watershed Management on Western Public Lands” Portland, OR.

Mar. 2004. Invited Panel Speaker, International Environmental Law Conference: “Postfire Watershed Restoration,” Univ. of OR, Eugene, OR.

April 2002. Invited Speaker, Restoring Public Lands Conference: Reclaiming the Concept of Forest Restoration, "Watersheds and Fisheries: Restoration Needs for Trout Habitats," Univ. of CO, Boulder, CO

Mar 2002. Invited Panel Speaker, International Environmental Law Conference: "Soils, Impacts and Effects on Trout Habitat," Univ. of OR, Eugene, OR

Mar. 2001. Invited Panel Speaker, International Environmental Law Conference: "NFMA and Salmon Habitat Protection," Univ. of OR, Eugene, OR.

May 2000. Invited speaker, 5th National Tribal Conf. on Environmental Management: "Federal Land Management's Effects on Critical Habitat for Endangered Salmon," Lincoln City, OR

July 1998-2000. Peer Reviewer for the scholarly journal, N. Amer. J. Fish, for papers related to the sedimentation of fish habitat in response to erosion from land uses and fire.

Feb. 1998. Invited Speaker, Oregon AFS Annual meeting: "Adaptive management: Is it really adaptive?" Sunriver, OR

May 1996-2000. Guest lecturer, Oregon State Univ. graduate course on riparian and wetland ecology, Corvallis, OR

Apr.-May 1996. Peer-reviewer for Proceedings of Forest-Fish Conference: Land Management Affecting Aquatic Ecosystems, Proc. Forest-Fish Conf., May 1-4, 1996, Calgary, Alberta, Canada. Nat. Resour. Can., Can. For. Serv. Nort. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Apr. 1995. Invited speaker, Pacific Rivers Council Workshop on Watershed Analysis and Salvage Logging, Wenatchee, Wash.

Apr. 1995. Invited speaker, Oregon State Univ. Dept of Fisheries and Wildlife Seminar, Corvallis, OR

Apr. 1995. Invited speaker, American Fisheries Society North Pacific International Chapter, Annual Meeting, Vancouver B.C., Can.

Mar. 1995. Invited speaker, American Fisheries Society Idaho Chapter Annual Meeting, Boise, ID.

Nov. 1994. Invited speaker, President's Council on Sustainable Development Workshop, Yakima, WA.

Sept. 1994. Invited speaker, Oregon Water Resources Research Institute Streambank Restoration Conference: "Biological Methods to Stabilize Streambanks--From Theory to Practice," Portland, OR.

Mar.-April, 1994. Peer-reviewer for Henjum et al., 1994. Interim Protection for Late Successional Forests, Fisheries, and Watersheds: National Forests East of The Cascade Crest, Oregon and Washington. The Wildlife Soc., Bethesda, MD.

Jan. 1993-Sept. 1995. Member, Oregon Department of Environmental Quality's (ODEQ) Technical Advisory Committee for Triennial Review of the State Water Temperature Standard.

Mar. 1993. Invited speaker, Northwest Scientific Association Symposium: "Cumulative Effects of Land

Management Practices on Anadromous Salmonids," La Grande, OR.

Curriculum Vitae: **J.J. Rhodes**

page 4 **Professional Service** (cont'd)

Aug. 1992 - Sept. 1992. Member, Ad Hoc Consultant Selection Committee for Portland Water Bureau Study of Future Water Supply Needs.

May 1992. Invited Speaker, US Forest Service, Pacific Northwest Region, Regional Workshop on Monitoring Soil and Water Resources, Bend, OR.

May 1992. Invited Speaker, Northern Arizona University, School of Forestry, Graduate Seminar Series, Flagstaff, AZ.

Jan. 1991 - Mar. 1995. Member, Technical Work Group: Upper Grande Ronde River Anadromous Fish Habitat Protection, Restoration and Monitoring Plan.

Aug. 1989 - Feb. 1990. Member, Technical Advisory Committee to ODEQ for development of definitions for level of beneficial use impairment by nonpoint sources.

May 1989 - Jan. 1991. Member, Nonpoint Source Technical Advisory Committee to Idaho Department of Environmental Quality: Coordinated Nonpoint Source Monitoring Program For Idaho.

PUBLICATIONS

Peer-Reviewed:

Rhodes, J.J., C.M. Skau, and W.M. Melgin, 1984. Nitrate-nitrogen flux in a forested watershed -- Lake Tahoe, USA. In: Recent Investigations in the Zone of Aeration, Proc. of Inter. Symp., Munich, West Germany, 1984, P. Udluft, B. Merkel, and K. Prosl (Eds), pp. 671-680.

Rhodes, J.J., 1985. A Reconnaissance of Hydrologic Transport of Nitrate in An Undisturbed Forested Watershed Near Lake Tahoe. M.S. thesis, Univ. of Nev. Reno, 254 pp.

Rhodes, J.J., C.M. Skau, and J.C. Brown, 1985. An areally intensive approach to hydrologic nutrient transport in forested watersheds. In: The Forest-Atmosphere Interaction, B.A. Hutchison and B.B. Hicks (Eds), pp. 255-270.

Rhodes, J.J., C.M. Skau, D. Greenlee, and D.L. Brown, 1985. Quantification of nitrate uptake by riparian forests and wetlands in an undisturbed headwaters watershed. US Forest Service Gen. Tech. Rept. RM-120.

Rhodes, J.J., C.M. Skau, and D. Greenlee, 1986. The role of snowcover on diurnal nitrate concentration patterns in streamflow from a forested watershed in the Sierra Nevada, Nevada, USA. In: Proc. of AWRA Symposium: Cold Regions Hydrology, Fairbanks Alaska, 1986, D.L. Kane (Editor), pp. 157-166.

Rhodes, J.J., R.L. Armstrong, and S.G. Warren, 1987. Mode of formation of "ablation hollows" controlled by dirt content of snow. J. Glaciology, **33**: 135-139.

Edmonds, R.L., T.B. Thomas, and J.J. Rhodes, 1991. Canopy and soil modification of precipitation chemistry in a temperate rain forest. Soil Soc. of Amer. J., **55**: 1685-1693.

Rhodes, J.J., McCullough, D.A., and Espinosa Jr., F.A., 1994. A Coarse Screening Process for Evaluation of the Effects of Land Management Activities on Salmon Spawning and Rearing Habitat in ESA Consultations. CRITFC Tech. Rept. 94-4, Portland, OR <http://www.critfc.org/tech/94-4report.htm>

Rhodes, J.J. 1995. A Comparison and Evaluation of Existing Land Management Plans Affecting Spawning and Rearing Habitat of Snake River Basin Salmon Species Listed Under the Endangered Species Act. CRITFC Tech. Rept. 95-4, Portland, OR <http://www.critfc.org/tech/95-4report.htm>

Rhodes, J.J. 1996. Description and Evaluation of Some Available Models for Estimating the Effects of Land Management Plans on Sediment Delivery, Channel Substrate, and Water Temperature, CRITFC, Portland, OR

Espinosa, F.A., Rhodes, J.J., and McCullough, D. A. 1997. The failure of existing plans to protect salmon habitat on the Clearwater National Forest in Idaho. J. Env. Management **49**: 205-230.

Rhodes, J.J., and Purser, M.D., 1998. Overwinter sedimentation of clean gravels in simulated redds in the upper Grande Ronde River and nearby streams in northeastern Oregon, USA: Implications for the survival of threatened spring chinook salmon, Forest-Fish Conference: Land Management Affecting Aquatic Ecosystems, Proc. Forest-Fish Conf., May 1-4, 1996, Calgary, Alberta, Canada. Nat. Resour. Can., Can. For. Serv. Nort. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356, pp: 403-412.

Beschta, R.L., Rhodes, J.J., Kauffman, J.B., Gresswell, R.E, Minshall, G.W., Karr, J.R, Perry, D.A., Hauer, F.R., and Frissell, C.A., 2004. Postfire Management on Forested Public Lands of the Western USA. Cons. Bio., 18: 957-967.

Karr, J.R., Rhodes, J.J., Minshall, G.W., Hauer, F.R., Beschta, R.L., Frissell, C.A. Perry, D.A, 2004. Postfire Salvage Logging's Effects on Aquatic Ecosystems in the American West. BioScience, 54: 1029-1033.

Rhodes, J.J. and Odion, D.C., 2004. Comment Letter: Evaluation of the Efficacy of Forest Manipulations Still Needed. BioScience, 54: 980.

Rhodes, J.J., 2005. Comment on "Modeling of the interactions between forest vegetation, disturbances, and sediment yields" by Erkan Istanbuluoglu et al. J. Geophys. Res. Earth Surf., Vol. 110, No. F1, F01012 10.1029/2004JF000240

Rhodes, J.J., 2007. The Watershed Impacts of Forest Treatments to Reduce Fuels and Modify Fire Behavior. Pacific Rivers Council, Eugene, OR

Rhodes, J.J. and Baker, W.L., 2008. Fire probability, fuel treatment effectiveness and ecological tradeoffs in western U.S. public forests. Open Forest Science Journal, 1: 1-7.

Lewis, J., Rhodes, J.J., and Bradley, C. 2018. Turbidity responses from timber harvesting, wildfire, and post-fire logging in the Battle Creek watershed, Northern California. Env. Mgmt, 1-17.

Technical Reports:

1986. Annual Report on Watershed Studies at Olympic National Park. College of Forestry, Univ. of Wash., Seattle, Wash. (Co-authors: R.L. Edmonds, T.B. Thomas, T.W. Cundy)

1987. Annual Report on Watershed Studies at Olympic National Park. College of Forestry, Univ. of Wash., Seattle, Wash. (Co-authors: R.L. Edmonds, T.B. Thomas, T.W. Cundy)

1988. Annual Report on Watershed Studies at Olympic National Park. College of Forestry, Univ. of Wash., Seattle, Wash. (Co-authors: R.L. Edmonds, T.B. Thomas, T.W. Cundy)
1989. Annual Report on Watershed Studies at Olympic National Park. College of Forestry, Univ. of Wash., Seattle, Wash. (Co-authors: R.L. Edmonds, T.B. Thomas, T.W. Cundy)
1990. Coordinated Nonpoint Source Monitoring Program For Idaho. Idaho Dept. of Environmental Quality, Boise, Idaho. (Co-authors: B. Clark, D. McGreer, W. Reid, T. Burton, W. Low, I. Urnovitz, D. McCullough, T. Litke)
1992. The Upper Grande Ronde River Anadromous Fish Habitat Protection, Restoration and Monitoring Plan. Wallowa-Whitman National Forest, Baker, OR (Co-authors: M. Purser, P. Boehne, R.E. Gill, R.L. Beschta, J.R. Sedell, B. McIntosh, J. Zakel, J.W. Anderson, D. Bryson, S. Howes, R. George).
1992. Salmon Recovery Program for the Columbia River Basin: An Advisory Report for the US Congress, Col. Riv. Inter-Tribal Fish Comm., Portland, OR (Co-authors: P.R. Mundy, D.A. McCullough, M.L. Cuenco, T.W. Backman, D. Dompier, P. O'Toole, S. Whitman, E. Larson, B. Watson, G. James).
1993. A comprehensive approach to restoring habitat conditions needed to protect threatened salmon species in a severely degraded river--The Upper Grande Ronde River Anadromous Fish Habitat Protection, Restoration and Monitoring Plan. USFS Gen. Tech. Rept RM-226, pp. 175-179. (Co-authors: J.W. Anderson, R.L. Beschta, P. Boehne, D. Bryson, R.E. Gill, S. Howes, B. McIntosh, M.D. Purser and J. Zakel).
1993. Dante's Video Guide to Habitat Conditions for Wild Spring Chinook Salmon, Steelhead and Bull Trout in the John Day Basin, Oregon. (Video) Presented at AFS National Meeting, Portland, Or, Aug. 29-31. (Co-authors: R. Taylor and M. Purser).
1995. Wildfire and Salvage Logging: Recommendations for Ecologically Sound Post-Fire Salvage Logging and Other Post-Fire Treatments on Federal Lands in the West. Pacific Rivers Council, Portland, OR (Co-authors: R. Beschta, C. Frissell, R. Gresswell, R. Hauer, J. Karr, G. Minshall, D. Perry).
1998. Adaptive management: Is it really adaptive? Abstracts: Oregon AFS Annual Meeting, Feb. 11-13, 1998, p. 31.
1998. Thinning For Increased Water Yield in the Sierra Nevada: Free Lunch or Pie in the Sky? Pacific Rivers Council, Eugene, OR. (Co-author: M. Purser)
1999. Annual Project Report: Watershed Evaluation and Aquatic Habitat Response to Recent Storms. Bonneville Power Administration (BPA), Portland, OR. (Co-author: C. Huntington)
1999. Annual Project Report: Monitoring Fine Sediment in Salmon Habitat in John Day and Grande Ronde Rivers. BPA, Portland, OR (Co-author: M. Purser)
2000. Annual Project Report: Watershed Evaluation and Aquatic Habitat Response to Recent Storms. BPA, Portland, OR. (Co-author: C. Huntington)
2000. Annual Project Report: Monitoring Fine Sediment in Salmon Habitat in John Day and Grande Ronde Rivers. (Co-author: M. J. Greene)

2001. Annual Project Report: Monitoring Fine Sediment in Salmon Habitat in John Day and Grande Ronde Rivers. BPA, Portland, OR. (Co-author: M. J. Greene)

2001. Imperiled Western Trout and the Importance of Roadless Areas. Western Native Trout Campaign, Center for Biological Diversity, Tucson, Az. (Co-authors: J. Kessler, C. Bradley, and J. Wood)

2002. Tryon Creek Watershed: Overview of Existing Conditions, Data Gaps, and Recommendations for the Protection and Restoration of Aquatic Resources. West Multnomah Soil and Water Conservation District, Portland, OR

2002. An Analysis of Trout and Salmon Status and Conservation Values of Potential Wilderness Candidates in Idaho and Eastern Washington. Western Native Trout Campaign, Center for Biological Diversity, Tucson, AZ. (Co-authors: C. Bradley, J. Kessler, C. Frissell)

2003. Stream and Fish Habitat Conditions in Tryon Creek: Their Likely Causes and Ramifications for Salmonids. Proceedings of Urban Ecology and Conservation Symposium, January 24, 2003, Portland, OR. Portland State University, Environmental Sciences and Resources, Portland, OR

2008. Primary Sources of Fine Sediment in the South Fork Stillaguamish River. Interim progress report for Washington State Salmon Recovery Funding Board, Olympia, WA. Snohomish County Public Works Surface Water Management, Everett, WA. (Co-authors: M. Purser, B. Gaddis, S. Britton, T. Coburn, and M. Rustay)

2009. Primary Sources of Fine Sediment in the South Fork Stillaguamish River. Project completion report for Washington State Salmon Recovery Funding Board, Olympia, WA. Snohomish County Public Works Surface Water Management, Everett, WA. (Co-authors: M. Purser, B. Gaddis)

2015. The High Costs and Low Benefits of Attempting to Increase Water Yield by Forest Removal in the Sierra Nevada. 108 pp. Environment Now, 12400 Wilshire Blvd, Suite 650, Los Angeles, CA 90025. (Co-author: CA. Frissell)

Semi-Technical Publications:

1993. Dam the analysis--heal streams instead. The Assoc. of Forest Service Employees for Env. Ethics Inner Voice, **5**(6): 1, 4-5.

1994. Invited Preface to Northwest Science Special Issue--Environmental History of River Basins in Eastern Oregon and Washington. Northwest Sci., **68**.

PROJECT MANAGEMENT

1993-1996. Technical Assistance Contract with NMFS to produce technical guidance for ESA consultations for effects of land management on critical habitat for Columbia basin salmon. Main duties: Primary Investigator; primary author of peer-reviewed reports including proposed guidelines for salmon habitat effects assessment (Rhodes et al., 1994), evaluation of land management plans' protection of critical salmon habitat (Rhodes, 1995), and evaluation of models for estimating land management effects on salmon habitat (Rhodes, 1996); available scientific literature review; budget tracking; project coordination. Total budget: \$230,000.

1998-2000. Watershed Evaluation and Aquatic Habitat Response to Recent Storms. Main duties: Primary Investigator; design and implementation of monitoring methods, coordination of technical staff in 10 watersheds with differing levels of grazing and logging in 3 subbasins in Idaho, Washington, and Oregon; technical training; data analysis; contract administration; proposal development; report preparation; budget development and tracking; coordination with grantor representatives. Total budget: \$164,000.

1998-2000. Evaluation of Effects of Grazing on Rate of Salmon Habitat Recovery. Main duties: Primary Investigator; design and implementation of monitoring methods, training of field technician; data analysis and synthesis; proposal development; preparation of progress reports; budget development and tracking; coordination with grantor representatives. Total budget: \$73,000.

1998-2001. Monitoring Fine Sediment Levels in Salmon Habitat in Grande Ronde and John Day Rivers. Main duties: Primary Investigator; design and implementation of methods for monitoring fine sediment levels in four rivers; field technician training; data analysis and synthesis; subcontract administration; proposal development; progress and technical report preparation; budget development and tracking; coordination with grantor representatives. Total budget: \$128,000.

2001-2002. Western Native Trout Campaign, Aquatic Scientist and Coordinator. Main duties: Oversight and assurance of scientific integrity of all reports and work products; coordinate conservation efforts among campaign member organizations and other groups working to protect and restore trout habitats and populations; budget tracking; technical and progress report preparation.

HONORS AND AWARDS

1996. Leadership and Excellence. Col. River Inter-Tribal Fish Comm., Portland, OR

1991. Employee of the Year. Col. River Inter-Tribal Fish Comm., Portland, OR

1984. Academic Recruitment Scholarship for Outstanding Graduate Prospect. Univ. of Wash, Seattle, Wash.

1982. Maxey Award -- Outstanding Graduate Student Paper in Hydrology. Univ. of Nev.-Reno.

1980. Winslow and Myron Reuben Scholarship for Outstanding Undergraduate in the Earth Sciences. Univ. of Ariz., Tucson, Az.

ADDITIONAL TRAINING

1993. USFWS Water Temperature Modeling via SNTMP

1991. USFWS Introduction to IFIM Investigations

October 27, 2016

Paul Dewey
Central Oregon Landwatch
50 SW Bond Street, Suite 4
Bend, Oregon 97702

Re: Review and Comment on the 2009 Scoping Letter, 2013 Draft Environmental Impact Statement (DEIS), 2014 Final Environmental Impact Statement (FEIS), April 2016 Supplemental DEIS (SDEIS), September 2016 Supplemental Final EIS (SFEIS) and Draft Record of Decision (ROD) for the Ochoco National Forest (ONF) proposed Ochoco Summit Trail System Project (Project) and Forest Plan Amendments.

Dear Mr. Dewey,

I have reviewed all Project documents for the proposed Ochoco Summit Trail System, numerous other documents available on the USDA Forest Service (Forest Service) website, and Project documents from the Ochoco National Forest (ONF) along with relevant research and literature. Below are my comments on the ONF Summit Trail System Project regarding the planning, analysis and conclusions of the ONF.

I am a wildlife biologist with over 34 years of experience, including 25 years with the Forest Service, 5 years with the Oregon Department of Fish and Wildlife (ODFW), and 4 years as a consulting terrestrial resource specialist with the Forest Service. Much of my work for the Forest Service was spent in central Oregon on the Deschutes National Forest and as a hydropower program terrestrial biologist for many projects east of the Cascade Mountains. I also recreate extensively in the proposed Summit Trail System Project area for camping, hiking, fishing, hunting and mountain biking. I have authored numerous documents on wildlife species and their habitats while working with the Forest Service, and impacts of Project developments on those species. Attached is my resume.

Wildlife Habitat Primer:

Wildlife species require a variety of habitat types by season to meet basic needs of food, water, shelter, and space. Habitat quality is dependent on a habitat's structural components in conjunction with an environment where those structural components can function. In other words the structural components that provide adequate food, water, shelter and space are needed in time and space relatively free of disturbance. Functional habitat needs to be of adequate size and arrangement on the landscape to meet an animal's requirement for any particular season, such as breeding, wintering, calving, lactating/rearing, summering, and migration.

For example, in the Project Wildlife report (p. 45): elk calving structural components are described as: *"Aspen stands and wet meadow complexes interspersed in coniferous cover and riparian woodlands serve as calving grounds..."* These components are sought out by elk to birth their calves. However, without the habitat being functional and relatively free of disturbance, elk use of this habitat will be diminished or negated. For example, the noise of motor vehicles is a disturbance that affects the functional quality of wildlife habitat. As a general rule one can get a snapshot of noise disturbance by looking at its intensity (decibel level), frequency (number of times per hour), and duration (the length of time the disturbance lasts before conditions return to normal). For a more complete understanding of wildlife and human

disturbance see Chapter 7. Monitoring Human Disturbance for Management of Wildlife Species and Their Habitats in USDA Forest Service GTR WO-89 2013 (Rowland and Vojta 2013).

A summary of motorized impacts to wildlife and their habitat also is included in the Wildlife Section Chapter 3 of the 2011 FEIS for the Travel Management Project Deschutes National Forest Ochoco National Forest Crooked River National Grasslands (2011 Travel Management FEIS) and SFEIS (p. 243-247). The summary in the Travel Management Plan looks at general information about traffic/ road and landscape influences, species behavior and ecology, amphibian/ reptiles, carnivores, road construction, road disturbance, decreased sensitivity to sound, communication confusion, physiological effects, stress, habitat loss, access for predators, connectivity, permeability, and fragmentation, followed by species specific information (2011 Travel Management FEIS p. 302-311).

Following is my biological review of the Project.

1. The ONF Fails to Show Impacts to Wildlife Species and Habitats.

1A. Rocky Mountain Elk Calving and Elk Wallow-Habitat

The ONF fails to document how it minimized Project effects to Rocky Mountain elk calving areas and elk wallows during the LRMP restriction period. The ONF LRMP Wildlife and Fish – Forest-wide Standards and Guidelines for Rocky Mountain Elk and Mule Deer (LRMP 4-246) state “Protect the character of elk calving sites. Minimize disturbance from human activity during calving season (approximately May 15 to June 30). Also protect wallows during rutting season (September 1 to October 15).”

Elk calving and wallow habitat are important to ensure the reproductive viability of the local elk populations (Oregon Forest Resources Institute (OFRI) 2013; Washington Department of Fish and Wildlife (WDFW) 2005). Elk calving (and deer fawning) areas, with low disturbance, are needed for birthing to ensure population recruitment. Lehman et al. (2015) found that “At the fine scale in forests and grasslands, female elk selected sites in areas with intermediate slope (19%), closer to water (355–610 meters), and far from roads (541–791 meters)”. Likewise, bull elk create and use wallows, with low disturbance, to cover themselves in mud laced with their urine and feces (their perfume) to attract and breed with cow elk in the fall of the year. Both life history functions breeding and reproduction are essential to the health, productivity and vitality of an elk herd.

All action alternatives in the SFEIS place OHV trails in close proximity to elk calving and deer fawning habitat during the Ochoco Forest Land and Resource Management Plan (LRMP) restriction period. Impacts range from 44.5 miles to 70.4 miles of trail not on open roads (SFEIS Table 146, p. 309) and impact between 7,120 to 11,264 acres of effective calving and fawning habitat (SFEIS p. 305). The SFEIS (p. 305) assumes a reduction in effective habitat of 200 meters (approximately .25 miles) on each side of a trail (160 acres per mile). None of the action alternatives address impacts to elk wallows by OHV riders.

Phillips and Alldredge (2000) studied reproductive success of elk following disturbance by humans during calving season and showed that with only 10 disturbance periods above ambient noise levels, their model showed no recruitment. The Summit Project, if implemented with any of the action alternatives, has the ability to significantly affect the ODFW Ochoco management unit elk population objectives. The

LRMP cites the Ochoco management unit elk management objective as 2600. In 2004-2005, the ONF persuaded ODFW to increase the elk population objectives to 4500. Population trends show that elk numbers and elk harvest are decreasing (Figure 1). (Greg Jackle, ODFW district wildlife biologist, pers. comm.).

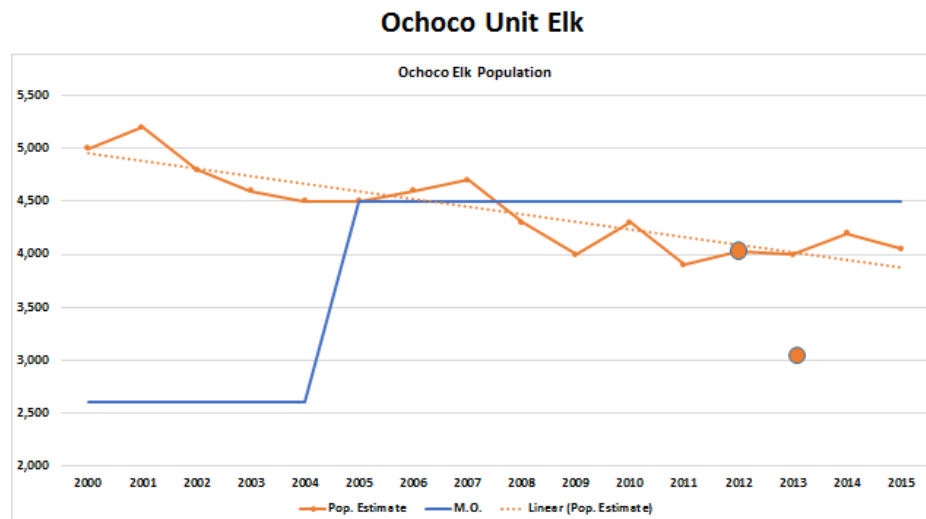


Figure 1. Ochoco Management Unit elk population estimates, 2000-2015. The orange line is the annual population estimate for elk while the “management objective” (MO) is the blue line (data from Greg Jackle, ODFW District Wildlife Biologist, pers. comm.). The MO was raised at the request of the forest in a public process.

All action alternatives propose a trail-riding season ranging from May 1 through November 1 of each year for alternatives 2 and 4, and June 1 through September 30 of each year for Alternatives 3 and 5. All of the action alternatives will impact elk calving during the months of May-June and elk wallows used for breeding during the months of September-October. Even the shorter proposed seasons of motorized recreational use for action alternatives 3 and 5 will impact the months of June and September each year.

The only recommended action by the ONF is to restrict “project construction, re-construction, decommissioning and maintenance activities from May 15 through June 30 with elk calving areas...and restrict project construction, re-construction, decommissioning and maintenance activities from September 1 through October 15 within 0.25 mile of known elk wallows” (SFEIS p.36). The ONF did not implement measures to protect these sensitive habitats during the allowed season of use for general riding by OHV recreationists.

1B. Elk Security Habitat

Elk occupy a mix of open areas for feeding and need both hiding and thermal cover (ODFW 2003). Hiding cover or security habitat is essential for protection from predators and human disturbance (ODFW 2003; WDFW 2005, Wyoming Game & Fish Bulletin 27). Security cover may consist of thick brush, conifers or heavy timber, and topography that allow them to hide. Lack of security cover and disturbance may force elk away from critical areas or stress animals. Cover also is used in extreme weather, to avoid hunters or other human disturbances, and conceal newborn calves from predators.

Elk security habitat is composed of structural and functional cover such as thick timber that an elk will seek out to hide from perceived threats (ODFW 2003). Oregon's principal elk ranges lack the large blocks of unroaded wilderness that are present in other western states. The farther an elk has to travel from security habitat to obtain food, the less those needs are met and the habitat may not be used, depending on the level and degree of potential threats. Where there are few threats, such as in National Parks, elk are often seen in the open; while areas of greater threats such as national forests where there are frequent disturbance from roads or recreational activities or poaching is common, elk stay close to security cover. The arrangement of security habitat with foraging habitat along with the amount of disturbance generally defines the area of functional foraging habitat. For example, adequate forage quantity and quality is necessary for a lactating cow to feed herself and her newborn calf. The amount of disturbance she is subject to affects how far she will travel from security habitat to forage, and may affect her and her calf's nutritional needs.

Many studies have demonstrated the huge effect that roads and motorized OHVs have on elk (Lyon 1979; Gaines et al. 2003; Rowland et al. 2000, 2005; Wisdom et al. 2005, 2006; McCorquodale et al. 2010; McCorquodale 2013; Proffitt et al. 2012; Ranglack et al. 2014). Rowland et al. (2005) described numerous effects of motorized vehicles on elk. These ranged from direct impacts to individual animals and populations such as increased vulnerability to harvest from legal and illegal hunting and vehicle collisions. The largest impact is the indirect impact caused by fragmentation of habitat with heavily roaded areas leaving too few areas for animals to escape, and displacement or avoidance, leaving less functional habitat for elk to use. Roads have been documented to cause reduced habitat availability and carrying capacity, movement of elk from public to private lands, abandonment of entire herd ranges when roads exceed some threshold, higher stress levels and increased energetic costs that reduce productivity (Millspaugh et al. 2001), and reduced local and regional populations (Forman et al. 2003). Increasing growth of motorized and non-motorized recreational pursuits is believed by state elk managers to threaten some herds and have contributed to shifts of elk from some public lands onto adjacent private lands (ODFW 2003).

According to the SFEIS (p. 284), the 301,580 acre Project area provides both summer and winter range habitat for mule deer and Rocky Mountain elk. Within the project area there are 17,828 acres in Winter Range, 25,183 acres in General Forest Winter Range, and 195,735 acres in General Forest (summer range) allocations, so the approximately 65% of the Project area is summer range. The higher elevations in the Ochoco Summit project area become snowbound in the winter and deer [and elk] are known to concentrate in foothills and valley bottoms along the Crooked and John Day Rivers and their tributaries during the winter (SFEIS p. 285). Both mule deer and elk are listed as Management Indicator Species for the Ochoco National Forest and elk in particular can act as a barometer for species sensitive to human disturbance. The Forest Plan provides habitat management direction for big game animals (SFEIS p.287).

The analyses by the ONF defined elk security habitat (SFEIS p.287) as forested habitat at least 250 acres in size and at least ½ mile from any motorized route. (See Item 2 below regarding motorized route density) The SFEIS used distance banding to describe open road effects to big game. Included in the distance banding analysis was the defined elk security habitat. Table 144 (SFEIS p. 296) shows acres of elk security area greater than ½ mile from open motorized routes for the no action and all action alternatives by 5th field watersheds.

Table 144 (SFEIS p. 296) shows a decrease in elk security habitat acres for all action alternatives in each of the 9 - 5th field watersheds. The following table (Table 1) is another way to show the data displayed in Table 144 in the SFEIS.

Table 1. Change in elk security habitat by watershed and action alternative.

5 th Field Watershed	Watershed Acres (ONF)	Current Elk Security % (Acres)	Alternative 2 % / (acres) Impacted	Alternative 3 % / (acres) Impacted	Alternative 4 % / (acres) Impacted	Alternative 5 % / (acres) impacted
Bridge Creek	15,595	27.4% (4,274)	0.3% (14)	0.3% (14)	0.3% (14)	0.3% (14)
Deep Creek	54,561	6.9% (3,745)	90.0% (3,367)	73.7% (2,721)	99.9% (3,744)	73.5% (2,753)
Lower Beaver Cr	23,611	17.6% (4,176)	18.1% (754)	18.1% (754)	18.1% (754)	17.6% (733)
Lower NF Crooked R	21,458	22.6% (4,868)	5.6% (275)	5.6% (275)	5.6% (275)	5.9% (285)
Mountain Creek	19,244	36.3% (6,986)	0.4% (27)	0	0.4% (27)	0.1% (9)
Paulina Creek	16,093	10.0% (1,609)	0.4% (7)	0.4% (7)	0.4% (7)	0.5% (9)
Rock Creek	14,181	24.4% (3,466)	1.6% (57)	1.6% (57)	1.6% (57)	1.7% (59)
Upper NF Crooked R	89,005	10.6% (9,433)	3.5% (329)	3.4% (325)	3.9% (366)	3.5% (331)
Upper Ochoco Creek	47,832	19.7% (9,412)	7.2% (678)	13.9% (1,304)	17.0% (1,593)	14.3% (1,346)
Total Project Area	301,580	15.9% (47,969)	11.5% (5,509)	11.4% (5,455)	13.6% (6,537)	11.5% (5,538)

* The estimate for Lower NF Crooked River watershed is misleading since there are no OHV trails in this watershed.

Table 1 and the SFEIS (Table 144) show significant elk security habitat impacts to the Deep Creek 5th field watershed from the proposed OHV trail system. Much of the Deep Creek watershed is comprised of stringer stands of timber in the bottom of canyons along streams and open ridge areas of shrub scablands. Even without the road system, the area was historically comprised of open ridges with most hiding cover associated with timber stands along stream bottoms. Much of the forest road system was developed for timber harvest in the 1960s and 1970s (SFEIS p. 119). The SFEIS (p. 56) reported that “*The majority of roads within the project area have been in existence for more than 40 years*”. Since much of the timber in the eastern portion of the forest occurred in stream bottoms, there are high road densities in the subwatersheds along streams where timber was extracted for wood products. Since that’s where the hiding cover is primarily located, these roads have impacted elk security habitat.

With the existing road system of open roads that were developed for timber harvest, less than 7% elk security habitat remains in the 54,561-acre watershed in the No Action Alternative. The

“Environmentally Preferable Alternative” (Alternative 5 plus portions of Alternative 2) impacts are not available, as the ONF did not provide any environmental analysis on its selected alternative. However, Alternative 5, which represents the majority of the “Environmentally Preferable Alternative”, leaves only 992 acres (1.8%) of elk security habitat left in this 54,561 acre watershed. The watershed had little elk security habitat left because of roads and past timber harvest, and with the addition of the OHV trail it removes 73.5% of the remaining elk security habitat that is greater than 250 acres in the Deep Creek watershed. This estimate also underestimates the real impact because it only includes the open road density, and does not account for closed (ML 1), decommissioned, and temporary roads, and user-created OHV trails. So, the situation is even more ominous than the analysis shows.

The Deep Creek watershed is in the heart of the Ochoco Mountains and is key summer range for elk populations. ODFW conducted an elk telemetry study starting in 1989 (ODFW 1998). A total of 39 elk were radio-collared and monitored until June 1994. Nineteen elk were captured in the Deep Planning Area and both elk (and uncollared deer) were recorded using the area during the snow free months (summarized in the Deep Vegetation Management Project, DEIS April 2001). In winter, animals moved to lower elevations generally either north of the district, east toward the South Fork of the John Day River, or south to the Crooked River valley. Locations of elk during the calving season in the Deep watershed were associated with riparian areas. Wildlife surveys in the Deep Vegetation Management Planning area recorded sightings of calves along riparian areas in the northern part. While specific sites of wallows were not recorded, they were thought to be present in the Deep Creek watershed and were associated with seeps, springs and other riparian habitats.

In a letter to the ONF on December 8, 2009 regarding the Travel Management DEIS (2011 Travel Management Plan FEIS, Appendix E Letters from Agencies, p. 488-497), ODFW noted a substantial increase in big game damage complaints from private landowners in the Ochoco Wildlife district between 1989 and 2008 from 7 to 55 complaints, respectively, and that elk had moved off of Forest lands and damage had increased considerably on private lands. The letter further noted that local research by ODFW in central Oregon had shown that illegal harvest of deer was greater than the legal harvest of deer during the regulated season. Because most of the illegally harvested deer were does, the problem was compounded since does are the recruitment component of the herd. Most poached collared deer were harvested within 300 meters of a road based on estimations by the biologists retrieving the collars. The Department requested in the letter that the ONF leave higher levels of hiding cover and close roads to provide more security to big game from poachers and other motorized disturbance.

The conflict essentially continues today because roads are left open and vegetation projects are implemented and continue to reduce hiding cover for big game. The 2004 Deep Vegetation Management Final Supplemental EIS (FSEIS) decision, which encompassed the entire Deep Creek watershed, was to implement Alternative C. That alternative includes 6393 acres of commercial timber harvest, 10,042 acres of precommercial thinning, and 8741 acres of prescribed burning outside of the treatment areas for a total of 25,176 acres. The Deep Creek FEIS Alternative C has/will treat 46% of the watershed.

Central Oregon Landwatch, per a Freedom of Information Request, asked the ONF for the GIS base layers and action alternative layers for the Summit OHV trail Project and information on vegetation and fuels treatment projects in the Project area in order to conduct their own analyses. Most of the

information was not supplied but there was one piece of information about elk security habitat for Alternative 5 that was provided that was relevant for Project impacts.

The following 2 figures are the Deep Creek watershed. Figure 2 depicts the harvest and precommercial treatment areas for the Deep Vegetation project (FEIS 2004). Figure 3 is from the GIS layers received in response to the FOIA request from Landwatch and depicts the remaining elk security cover for Alternative 5 and they are all areas with elk security habitat, not just the 250 acre “elk security habitat” areas defined in the Summit SFEIS.

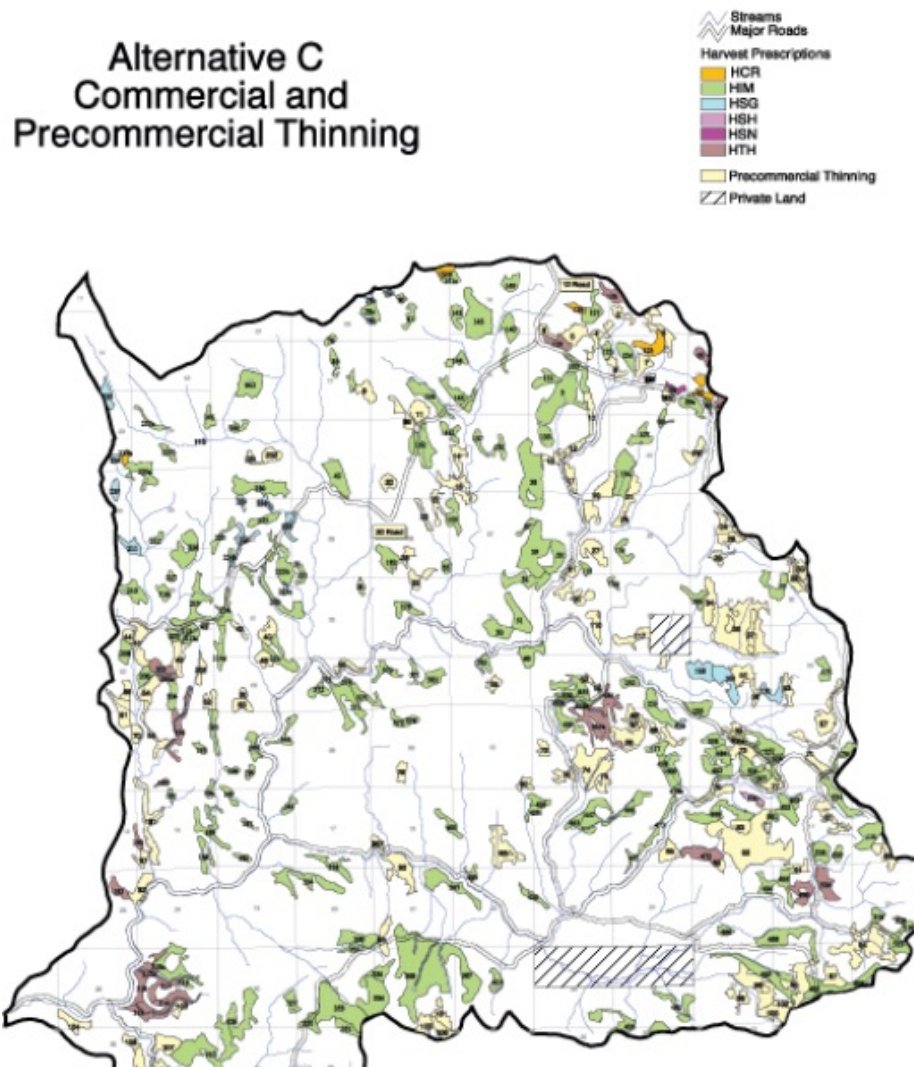


Figure 2. Deep Vegetation Treatment FSEIS harvest and precommercial treatments excluding the prescribed burning. Areas in color are harvest and precommercial treatments.

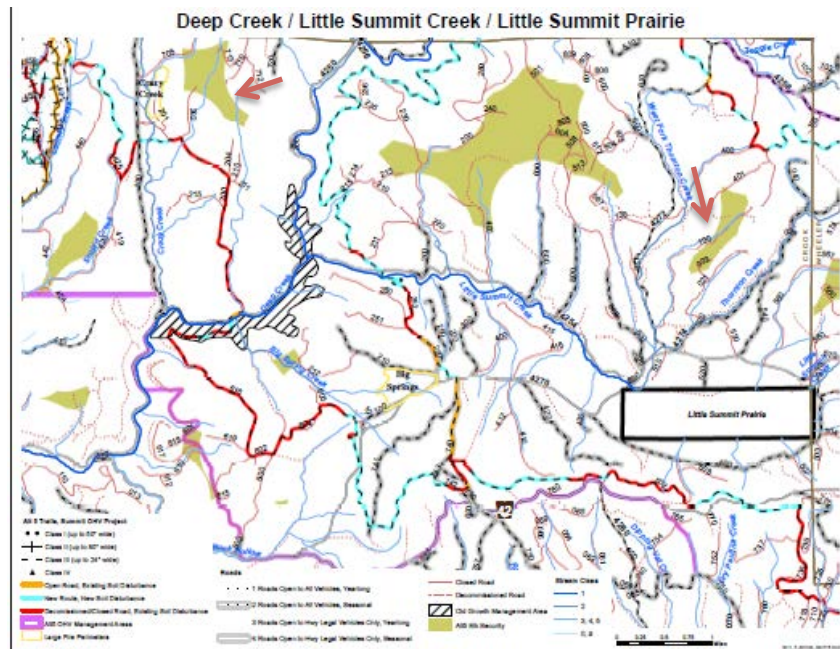


Figure 3. Summit Project Alternative 5. Shaded green areas are elk security habitat of any size, no minimum acre size. Information received by FOIA and on elk security habitat for action Alternative 5.

Reviewing the Deep Vegetation FEIS, there were no analyses to determine if security habitat would be impacted by the project, which would affect 46% of the entire watershed. Therefore, none was analyzed or subsequently protected by the Deep Vegetation project. Comparing the 2 maps of the Deep Vegetation Treatment areas and the proposed OHV trails above, it appears that there was elk security habitat that are/were treated with timber harvest. There may have been more in the existing condition of the No Action Alternative; we will never know how much was impacted since those units have likely been treated and the existing condition in 2004 no longer exists today. Now, the ONF proposes to further impact the Deep Creek watershed that has less than 8% elk security habitat left with implementing an OHV trail, and further compound the harm to important elk habitat.

By implementing any of the action alternatives, there will be significant impacts to elk security habitat, and the estimates above do not include the effects of all roads, routes and trails currently driven on the landscape, only open roads (Maintenance Level (ML) ML-2-5). This will significantly degrade elk security habitat and further move elk to exterior boundaries of the forest, and off forest onto private lands during the spring and summer when elk are normally on the forest, calving and rearing young. The Ochoco Mountains are characterized by big game that summer on the forest and then move downslope to winter range that is primarily on private land that circles the forest at lower elevations. Most private landowners tolerate current winter elk use, but an increase in use during the spring and summer, when the big game are historically on the forest, would likely require ODFW to significantly reduce the elk [and deer] populations. This in turn would significantly reduce ODFW's management objectives for big game species and hunting opportunity and associated economic benefits.

The elk security habitat acres shown in Table 144 and the impacts shown above are predicated on the use of the 2011 Travel Management Plan and the annual Motor Vehicle Use Map showing *only open roads*

(Management Level (ML) 2-5) which ONF used to calculate roads/routes per square mile). As described in Item 2 below, the ONF landscape is covered with roads, trails and other routes that are physically open for motor vehicle use. While the analysis only included open roads, if the analysis were properly conducted using all roads, trails and other routes physically open on the landscape that the public drives for recreational pleasure or work, it would demonstrate even greater impacts to elk security habitat.

All action alternatives would allow OHV trail use through current elk security habitat. However, impacts to the Deep Creek watershed disproportionately affect 73.7 % to 99.9% of the remaining functional elk security habitat by action alternatives. Elk security habitat in Lower Beaver and Upper Ochoco Creeks is also greatly impacted by proposed OHV trail routes. As one can see, the functional habitat remaining in these watersheds, primarily due to ongoing disturbance from timber harvest and roads, is a fraction of what was available historically. Further impact results in less available habitat to support elk on the ONF, which will reduce the potential to meet LRMP and ODFW elk management objectives.

More broadly, over the entire Project area, the SFEIS (p. 308) states that *“Distance banding analysis of the combination of existing open roads (based on operational maintenance level) and proposed trails indicates that there would be a total of 44,944 acres of National Forest lands greater than 800 m (1/2 mile) from an existing road (elk security habitat). This represents approximately 15% of the project area (a 2% reduction compared to existing condition).”* Each incremental change from vegetation treatment project and now the proposed OHV project chips away at elk security habitat and therefore, over time, impacts long term productivity of elk on the forest.

1C. Scablands

Another important special habitat that will be impacted by an OHV system is scablands. These open meadow or shrub-like areas have very shallow soils, and support associated plants that are adapted to such unique habitat. Gast et al. (1991) reported that scabland vegetation, located in dry, hot, thin soils are very vulnerable to high levels of disturbance. Shrub-scabland ecosystems have rocky, shallow soils and are commonly found on topographically high sites such as ridges. Dominated by scattered sagebrush, bunchgrasses and forbs these sites are particularly exposed to summer heat and dryness, winter cold and spring frost heaving.

The LRMP recognized the fragile nature of this community and designated protection standards that included “ORV use on scablands will be limited to snowmobiles operating on an adequate snow base.” “Scablands are recognized as among the most fragile ecosystems on the Ochoco National Forest. Damage to the soil and vegetation as a result of management activities is nearly impossible to mitigate. This is a result of their having very shallow soils which are subject to severe water saturation and frost heaving during winter, thus making revegetation virtually impossible.” (LRMP 4-197). The LRMP standard and guidelines required that “Road construction on scablands will be limited to long-term collector, arterial and local roads. Temporary or short-term roads or trails will not be constructed across scablands unless there is no other feasible alternative.” (LRMP 4-226). Damage to the soil and vegetation as a result of management activities, particularly motorized vehicles, is nearly impossible to mitigate.

Biological soil crust is another fragile component of the shrub-scabland ecosystem. It functions as living mulch by retaining soil moisture, discouraging annual weed growth, and influences soil stability and

fertility. Once disturbed, natural recovery rates are found to be very slow and re-establishment can be extremely difficult, if not impossible (Belnap et.al. 2003).

Ouren et al. (2007) conducted a literature synthesis of the effects of OHVs on natural resources including soils. OHVs impact soils and overall watershed function by altering via soil compaction and destroying soil crusts (biotic and abiotic) causing increased soil bulk density and decreased permeability. As soil compaction increases, its ability to support vegetation diminishes because the resulting loss of porosity inhibits the growth of root systems and reduces infiltration of water. As vegetative cover, water infiltration, and soil stabilizing crusts are diminished or disrupted, the precipitation runoff rates increase, which further accelerates soil erosion. Jones (1999) reported that “Loss of the insulating effects of plant [from OHV impacts] cover and changes in the heat capacity of compacted soils causes soil temperatures to increase by as much as 10°C in daytime and decrease by as much as 3°C in nighttime”, which in turn, further inhibits the ability of native plants to re-establish.

Switalski and Jones (2008) reported on impacts of OHVs to forest soils but the concepts apply to scabland soils as well. Healthy forest soils provide nutrients and are the physical base for plants. Since OHVs can weigh several hundred pounds, they compress and compact soil reducing its ability to absorb and retain water, decreasing soil fertility by impacting microscopic organisms that are important for nutrient cycling.

The SFEIS (p. 392) and LRMP (4-197) states that “*Scablands are recognized as among the most fragile ecosystems on the Ochoco National Forest. Damage to the soil and vegetation as a result of management activities is nearly impossible to mitigate. Revegetation is nearly impossible due to water saturation and frost heaving.*” However, despite recognizing the fragile nature of scabland, the ONF proposes to amend the 1989 LRMP to allow OHV trails and use on the shrub-scabland ecosystem. The SFEIS (p. 440) states that “*The proposed amendments are site-specific and apply only to proposed segment of connector trail, most of which are located on existing disturbance.*” Contrary to the preceding claim by the ONF, all action alternatives have new trail construction and conversion of closed/decommissioned roads to OHV trails. These are not largely connector trails as stated, but have disturbance from new construction and conversion from closed and decommissioned roads that ranges from 63% to 79% of the miles of OHV trail proposed on scablands (SFEIS, Table 161, p. 368). We find it disingenuous to state that these proposed trails are “*most of which are located on existing disturbance*” (SFEIS p. 440) when the vast majority of trails are new construction and re-opening of closed and decommissioned roads.

The ONF claims that the LRMP amendment applies to proposed segments of “connector trail” is also frankly misleading. The OHV trail routes in the east implementation area are loop trails not simple connector trails. The SFEIS (p. 440) further states that “*trail segments are proposed in order to allow orderly and logical trail system by linking existing roads...*” There are approximately 26 miles of new construction in the east implementation area. Alternative 5 proposes 8.5 miles of new construction and 7.8 miles of conversion from closed/decommissioned roads, respectively. This many miles of new construction and reopening of closed and decommissioned roads is not just “linking existing roads”. It’s developing an outdoor play area for motorized recreationists.

The only action proposed to minimize impacts to scablands is the broad statement in the PDCs “*where trails cross scablands treatments to prevent erosion will be implemented. Techniques may vary by trail type and may include but are not limited to rock surfacing or soil amendments*” (SFEIS p. 33), which has

already been demonstrated in preceding statements is almost impossible to do. The SFEIS (p. 376) notes that *“A project design criteria was developed to protect TES plant habitat by closing the trail system when adverse conditions exist that may result in resource damage (rutting, runoff, etc.), however, in scabland habitat, once the damage is done, it is permanent”*. Given the likelihood that rutting and runoff will significantly occur during the runoff periods, such as during spring runoff and summer thunderstorms on the ONF, the effects of OHV trails are impossible to repair. Once the damage is done, it’s irreversible.

The SFEIS justifies the impacts as insignificant by comparing the minimum number of acres of disturbance from trails to the total numbers of acres of scablands in the project area, thereby dismissing the impacts. The SFEIS also fails to disclose the total number of acres of shrub scabland already impacted in the project area by existing roads on the landscape including open, closed, decommissioned, temporary, and user-created trails. Further, the SFEIS (p. 454) not only fails to address minimizing impacts as required under the minimization criteria, but also claims that *“Proposed trail segments in all action alternatives are in ecologically suitable locations”* (SFEIS p. 454). We’re not sure what the definition of an “ecologically suitable location” is but it’s not defined in the SFEIS. In the vegetation section, the SFEIS does not address minimization to shrub-scablands. All action alternatives propose to re-construct closed and decommissioned roads, and construct new trails affecting a total new disturbance of 6.45 to 21.73 miles of the shrub-scabland ecosystem.

The ONF did not provide any maps showing the location of scablands and how proposed OHV trails might impact, much less be located to avoid and minimize impacts to these special habitats. Despite acknowledging the fragility of scablands, Alternative 5 as modified includes at least 23.6 miles of trail and one staging area in scabland habitat. Instead of designing a project to protect these fragile landscapes, the ONF proposes Forest Plan amendments so it can authorize new motorized trails in scablands (SFEIS p. 28 and 441-444). In conclusion, the ONF failed to meaningfully apply 36 CFR 212.55 (b) (1) and (2) minimization criterion to each trail being considered for designation in these special habitats. For all action alternatives, the ONF failed to locate OHV trails away from scablands and designate effective site-specific actions to negate the impacts of the proposed trail system.

In addition to the direct impact to scablands, there is a Regional Forester’s Sensitive Plant species located on scablands known as Henderson’ needlegrass. Dewey (2013) reported that principal threats to this species include 1) invasive plant species, 2) livestock grazing, 3) timber and fuels management activities and 4) off-road vehicle use. In Oregon, nearly all documented sites of this species occur fully on federal lands. Twenty-six populations of Henderson’s needlegrass occupy 144 acres in the Project area. It occurs on shallow scablands with little vegetation; the greatest potential for impacts comes from soil disturbance. All action alternatives propose OHV trails within sensitive plant populations, with proposed trail distances within the plant populations’ that range from 538 feet to 1,836 feet.

Antelope (Pronghorn) also occupy the dry open shrub-scabland ecosystem when not disturbed for many of their life history requisites. Pronghorn use their eyesight and hearing to detect any threat from predators hence their preference for wide-open expansive terrain (O’Gara and Yoakum 2004). SFEIS Table 148 (p. 315) and Table 169 (p. 395) show that of the 32.4 miles of OHV trail in pronghorn habitat, 13.35 miles occur in the shrub-scabland ecosystem.

OHV trails through the shrub-scabland on the ONF would increase soil compaction, damage biological soil crust, affect native plants, and displace pronghorn antelope. Because the shrub-scabland is very open there is a significant risk of OHV riders riding off the designated route to “play”, increasing the amount of impacts to this fragile habitat. A good example of scablands that are already impacted by OHV users is in the Rager country in the Eastside Implementation Area where much of the forest consists of stringer stands of timber along stream bottoms and in draws, while scablands dominate the ridges. While hiking and hunting the Rager country, we frequently find illegal OHV trails running across scablands. We have also repeatedly encountered OHV users driving closed and decommissioned roads. As an example, Figure 4 is a road on a scabland with fresh tracks visible.



Figure 4. A scabland with a road in the Deep Creek watershed. Note the broad expanse that is easily driven by any OHV.

Under all action alternatives, the ONF failed to locate OHV trails away from the fragile shrub-scabland ecosystem, and the ONF failed to demonstrate site-specific actions to negate the impacts of the proposed trail system on the shrub-scabland ecosystem. The ONF failed to disclose how it minimized Project effects to shrub-scablands, the Regional Forester Sensitive plant Henderson’s needlegrass, and pronghorn antelope.

1D. Late Successional Old Growth

The ONF failed to disclose how it minimized Project effects to late successional old growth located in Old Growth Management Areas (MA - F6 Old Growth).

The ONF LRMP recognized the importance of retaining late successional old growth stands for dependent and associated wildlife species. The LRMP established Old Growth Management Areas (MA-F6) to provide habitat for the pileated woodpecker and other old growth associated species as part of the Regional Foresters Minimum Management Requirement (MMR). The LRMP designated management areas specifically for old growth, setting aside approximately 2% of the ONF into MA-F6 Old Growth. It used the pileated woodpecker as the ONF Management Indicator Species (MIS) to monitor the health of

the old growth areas (LRMP 4-176). Further, the ONF delineated supplemental pileated woodpecker feeding areas of similar size immediately adjacent to the old growth areas.

Generally, each old growth area is approximately 300 acres in size providing the primary nesting habitat plus 300 acres of supplemental feeding habitat immediately adjacent. Given the landscape configuration of forested and non-forested plant association groups, many of the ONF MA-F6 Old Growth areas in the Project area are not blocks of interior habitat, but rather they conform to the northern aspects of topographic features (i.e. buttes, canyons and draws).

As the MA-F6 Old Growth areas are limited to just barely 2% of the entire ONF. Previous timber management practices have reduced much of the late successional old growth stands in the Blue Mountain forests including the ONF (Powell 2011). The intent of the LRMP guidelines was to designate old growth areas to maintain the biological integrity and ensure functional habitat for old growth dependent wildlife species. Wildlife species that are dependent on old growth structural components tend to be specialists. Wildlife specialists are species that require unique habitat types versus those that can use multiple habitat types or wildlife generalists (i.e., coyotes). Specialists also tend to be federal or state listed species as sensitive, threatened or endangered due to the significant habitat changes that have taken place as a result of past management practices. Old growth habitat is very limited today compared to historic conditions as portrayed (Powell 2011). Wildlife species dependent on old growth habitats on the ONF will be affected by changes to the existing old growth characteristics proposed by this Project.

The ONF LRMP Transportation section for the Management Area Standard and Guideline for Off-road vehicles stated that “No motorized use allowed” in MA-F6 Old Growth (LRMP 4-233/234). Further, in the ONF LRMP Fish and Wildlife section, the Management Area Standard and Guideline for Pileated Woodpecker Habitat, which applies to MA-F7 Old Growth area states “*Vegetation management (except prescribed livestock use) will not be allowed, until further research is available on the needs of the dependent species.*” (LRMP 4-251). In other words, vegetation management or removal to accommodate OHV trails is not allowed.

The ONF essentially dismissed the impact to old growth as insignificant by using the small footprint of the project compared to the large size of the project area. Appendix A of the Final Wildlife Report assumes 10 feet of forest clearing along proposed trails, resulting in vegetation clearing in both the old growth areas and the supplemental feeding areas. The vegetation clearing for each action alternative is Alternative 2 (6.5 acres), Alternative 3 (2.6 acres), Alternative 4 (6.5 acres) and Alternative 5 (4.4 acres). The SFEIS (p. 454) states that “Designation of a motorized trail system, including construction of new trail segments, is unlikely to affect forest vegetation (i.e. timber stands) in any measurable way.” Appendix A of the Final Wildlife Report estimates trail (vegetation) clearing for all action alternatives range from 2.6 to 6.5 acres and OHV trail lengths range from approximately 2.14 to 5.39 miles.

The SFEIS (p. 454-455) fails to provide any discussion regarding disturbance of wildlife and disruption of wildlife habitats by the trail designation for old growth dependent and associated wildlife, including the pileated woodpecker. The 2011 Travel Management FEIS used a distance of 200 meters (660 feet) each side of a motorized route as a metric to determine general wildlife disturbance, yet in the analysis for this project, the disturbance is only a 10 foot trail width. We don’t understand the discrepancy other than the intent of the Project is to dismiss impacts to wildlife species by using such a small footprint.

A good example where the ONF failed to consider alternate routes to avoid Old Growth areas is the proposed OHV trail inside and adjacent to the old growth area OG-D1-08 near Indian Butte north of Big Summit Prairie (SFEIS p. 441). Much of the area around this old growth area has been treated with vegetation and fuels projects over the past several decades, with approximately 7,400 acres scheduled for commercial harvest and precommercial thinning as part of the recent Howard Elliot Johnson Fuels and Vegetation Management Project (May 2011 FEIS). This old growth area is an island of habitat surrounded by a sea of treated areas and roads, left on the landscape to provide biological diversity and habitat for dependent Old Growth species. There were many choices to locate this trail outside of the Old Growth area but the proposed trail travels through and then adjacent to this island of unique habitat. There were plenty of opportunities to place the trail on mixed use roads to completely avoid the old growth area that were not chosen in any action alternatives.

Another example is the proposed OHV trail in the Deep Creek area OG-D2-02 (SFEIS p. 444). Because of the poor quality of maps in the SDEIS and SFEIS and lack of disclosure of proposed lengths of trail in each Old Growth area it's difficult to estimate the impact of specific OHV trails in each Old Growth area. A best guestimate using the extremely poor maps is that there are approximately 0.35 miles of new construction and 2.5 miles of conversion of closed and decommissioned roads to OHV trail in the Deep Creek Old Growth area. Much of the Eastside Implementation Area is stringer stands of timber along streams and in draws, while scablands occupy much of the uplands and ridge areas. The choice to place a motorized trail that impacts many resources including crossing a Class I fish bearing stream (Deep Creek) and traveling up a steep slope into the Deep Creek Old Growth area is an example of how the ONF failed to avoid important fish and wildlife habitat on the landscape. In order to implement the trail, vegetation will have to be removed further violating LRMP standards and guidelines.

Under all action alternatives, the ONF failed to locate OHV trails away from MA-F6 Old Growth areas and the pileated woodpecker supplemental feeding areas. The ONF also failed to demonstrate site-specific actions to negate the impacts of the proposed trail system on the MA-F6 Old Growth areas.

1E. Gray Wolf and Use of the ONF

The gray wolf is federally listed endangered west of Hwy 395 (ODFW 2010, 2016; US Fish and Wildlife Service 2016). It is also under the jurisdiction of ODFW's Wolf Management Plan. Both federal and state fish and wildlife agencies have an interest in the recovery and management of the Gray Wolf. The most recent 2015 ODFW summary report of wolves with telemetry collars shows expanding distribution as wolves have moved from occupied habitat in northeast Oregon and, with increasing regularity, moving through the ONF on its way to occupying portions of the southern Cascades, Summer Lake area, and the Siskiyou Mountains (Figure 5).

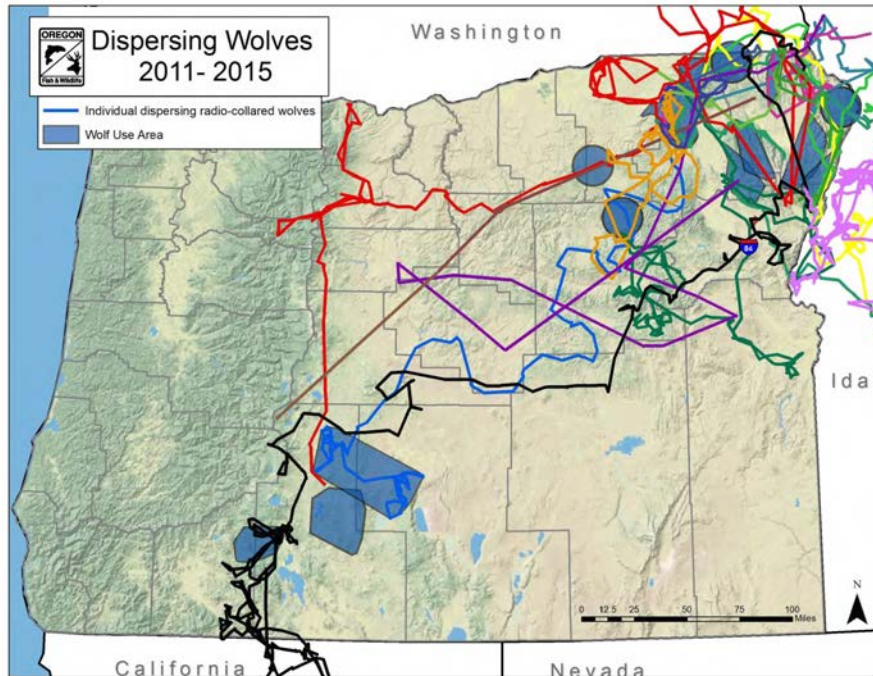


Figure 5. Dispersal of radio-collared wolves from northeast Oregon to new habitats in Oregon.

Since the compilation of this data, there have been reputable reports of observed wolves in a small pack on Lookout Mountain in the fall of 2015. A wolf track was observed in the South Fork Crooked River (south and east of the ONF) drainage in the fall of 2015, and a wolf was observed traveling through Big Summit Prairie (middle of the Summit Trail Project Area) in January 2016 (Greg Jackle, ODFW District Wildlife Biologist, pers. comm). In the past few years, three collared wolves were documented crossing through the Ochoco Mountains (Figure 5) indicating that the ONF has become a likely migratory path for large carnivores and an important connective corridor between the Blue Mountains and the Cascades. The presence of large blocks of public land is correlated with wolf habitat (Mladenoff et al. 1995; Larsen and Ripple 2006). In a recent conversation with John Stephenson, wolf biologist with the U.S. Fish and Wildlife Service, it is expected that a pack could become established on the ONF as packs are now established throughout northeast Oregon, near Summer Lake, and in the southern Cascades and Siskiyou Mountains of Oregon (Stephenson, pers. communication, 2016).

The SFEIS (p. 255) states that “*the potential for disturbance from motorized recreation would not be very different from disturbance associated with the existing open road system; the project is not expected to affect distribution and population size of prey species for wolves. The selection of any alternative would have no effect on potential wolf denning, foraging or dispersal habitat in the project area. Because the project would not affect wolves, no cumulative effects are anticipated*”. This information disregards the latest information regarding wolves moving through Oregon and reoccupying historic habitat. The comment also discounts the increase in motorized use of the 659 miles of mixed-use roads that will be caused by the Project if it is implemented.

Wolf prey is usually large ungulates such as deer, elk and moose (Pletscher et al. 1997). Since elk and deer are the most abundant ungulates in the Ochocos, they are the likely primary prey species in the Ochocos. Therefore, based on current science and literature of big game species and carnivores, this

Project will affect both elk [which are the primary prey species for the gray wolf in northeast Oregon] and the wolf itself by 1) increasing open road and trail densities and 2) increasing road and trail use rates [frequency], both of which will cause shifts in elk distribution. In regards to the wolf's primary prey species, elk are displaced by motor vehicles, and the Project may cause a temporary or permanent reduction in functional habitat. Numerous authors and research at Starkey Research Station have demonstrated that elk in areas with higher road densities and areas with higher human disturbance from motor vehicles have increased stress levels and increased movement rates and energetic costs (Rowland et al. 2000, 2005; Wisdom et al. 2004, 2005). These, in turn, reduce body condition, which is a primary factor affecting pregnancy rates and can influence juvenile growth and survival. Therefore, this Project is highly likely to affect performance of individual elk and have subsequent population level effects.

Beyond impacting the wolf's primary prey species (elk population and distribution), the OHV Project could directly influence and reduce the probability of wolves using the area for denning, foraging or dispersal. Wolf habitat suitability is strongly and positively correlated with public ownership (Mladenoff et al. 1995, Carroll et al. 2003, Larsen and Ripple 2006), and by moving the wolf's primary prey species to private lands, this Project has the potential to shift wolves to private lands. Since prey availability and distribution are a strong determinant of wolf habitat suitability, this shift in prey distribution associated with the OHV Project could cause wolves to forage on private property where wolves are less likely to survive, due to increased conflict with humans. This Project will increase motorized route densities and human use, which will decrease the suitability of habitat in the Project area for wolves (Belongie 2008). Concluding that this Project will have no effect on wolf habitat in the Project area ignores the best available science. This disregard by the ONF of recent gray wolf movement and expansion through the forest is particularly concerning as the projected increased use on both trails and mixed-use roads and will diminish prospects for re-establishment of the gray wolf in historic habitat on the forest.

Gray wolves are both a federally listed species and state managed species that the ONF must adequately consider for potential impacts in the proposed Project area. Failure to consider impacts to wolves by an OHV trail system will impede efforts by both federal and state agencies for conservation, recovery and management of this species. Further, wolves will follow their prey base down to private lands, as deer and elk move off the forest from being pushed by the development of an OHV trail system (ODFW December 8, 2009 letter to the Ochoco National Forest for the Travel Management Plan FEIS).

ODFW (2010) reported that wolves historically were common in Oregon and when a bounty was offered in 1913, within 8 months, 6 wolf payments were made for wolves killed in Crook County. While there are no current known wolf packs currently occupying ONF lands, as demonstrated by collared wolves in Figure 5, the species is increasing use of the forest as a migratory corridor for reoccupying habitat in other areas of Oregon and have a reasonably high likelihood of establishing a wolf pack in the ONF vicinity.

The ONF failed to meet its obligations to support federal and state agencies in their recovery goals for the gray wolf. It also failed by potentially causing increases in landowner damage by wolves to livestock by pushing big game prey species off the forest with an OHV trail system. Under all action alternatives, the ONF failed to locate OHV trails away from relatively undisturbed habitats such as Old Growth areas and Bridge Creek Wilderness and the ONF fails to demonstrate site-specific actions to negate the impacts of the proposed trail system on the gray wolf. In addition, the ONF fails to use current science and assist

federal and state managers achieve recovery goals for the Endangered Species Act listed endangered gray wolf (ODFW 2010; US Fish and Wildlife Service 2016).

1F. Noxious and Invasive Plant Species

The spread of exotic and noxious weeds is one of the most pervasive threats to public lands and degrades native plant communities that support important fish and wildlife habitat. Invasive and noxious plant species colonize disturbed sites where native vegetation has been disturbed, reduced or removed. OHVs disturb and destroy native vegetation, create bare soils, and facilitate the spread of noxious weeds. Switalski and Jones (2008) reported that OHVs “are a major vector for non-native invasive plant species. With knobby tires and large undercarriages, ORVs can unintentionally transport invasive non-native species deep into forestlands. For example, one study found that in a single trip on a 10 mile course in Montana, an ORV dispersed 2,000 spotted knapweed (*Centaurea stoebe*) seeds (Montana State University 1992). In Wisconsin, a survey of seven invasive plant species along ORV routes found at least one of these exotic plant species on 88% of segments examined (Rooney 2005).”

Executive Order 13112 directs federal agencies to reduce the spread of invasive plants, which has been identified as one of the four major threats to ecosystem health. One Project Design Criteria is “Do not route trails, place staging or camping areas, or other disturbance within areas containing invasive plants” (SFEIS p. 33). Yet that is exactly what all action alternatives propose to do. Alternative 5 will have 3.2 acres trails (estimated trail width of 10 feet X the number miles of trail) in areas infested with houndstongue, Canada thistle, medusahead, spotted knapweed, and diffuse knapweed (SFEIS p. 387). It will also have staging areas at Walton Lake and Cottonwood Pit that have infestations of spotted knapweed, Canada thistle, and Russian knapweed (SFEIS p. 387).

Another PDC is that all trail construction and maintenance equipment be cleaned of all dirt and plant parts before entering NFS lands (SFEIS p. 34), but this standard is not held for the OHV riders who should be cleaning their equipment before and after riding Summit OHV trails. The SEIS (p. 384) reports that *“OHVs brought to the project area that are not clean may harbor invasive species seed and plant parts that can introduce new infestations. OHVs can also run through existing invasive species infestations, further spreading the population by creating additional disturbance for a seed bed, or picking seed and carrying it to another portion of the area. There are several areas open to cross country travel (such as rock pits) that have invasive species populations.”* The following statement summarizes the expected effect of the no action alternative. SFEIS (p. 384) *“OHV use would continue to occur under the No Action alternative... Less soil disturbance and fewer vehicles results in less risk of invasive species introduction and spread in this alternative.”*

However, under all action alternatives, there will be a very high risk of spread of noxious and invasive weed species. SFEIS (p. 385) states that *“When tires of motor vehicles displace topsoil and damage vegetation, the resulting bare soil becomes potential sites for invasive plant establishment. A Montana State University Extension Service Bulletin (MSU 1992) explains that rapid dispersal of invasive plants is characteristic of motorized routes where “ORVs can quickly spread 2,000 knapweed seeds over a ten mile area.” Sheley et al. (1998) state, “Protecting and conserving the surface soil are critical to the long-term sustainability of healthy, functioning ecosystems. In one study, runoff was 1.5 times higher and sediment yield was 3 times higher on spotted knapweed-dominated plots than on plots dominated by the*

native bluebunch wheatgrass. Loss of soil because of invasive plants may have very serious consequences in the future.” and “Alternatives that propose more area open to motor vehicle travel and that have higher densities of designated routes, staging areas, trailheads and play areas pose a higher risk of invasive plant spread, introduction into new sites, and create greater challenges for detecting and controlling new invasive plant sites” and “All [action] alternatives propose trails circling around Little Summit Prairie that was designated as a sensitive resource area by the Paulina Ranger District, and who recommended to prohibit off-road OHV use. This area contains important resource attributes including a major population center for Peck’s mariposa lily, a Regional Forester’s sensitive plant species; a high density of prehistoric archaeological sites; soils that have a high-risk for erosion; and one of the most interconnected habitats for redband trout, another Regional Forester’s sensitive species” (SFEIS p. 386).

The SFEIS (p. 387) states “All action alternatives would result in an increase of invasive plants across the project area. This in turn would result in additional expense for controlling invasive plants on the Forest. The nature of the proposal involves a constantly disturbed environment due to continued use of the trail system by vehicles coming from different parts of the state or country with a high probability of bringing in invasive plant seed and plant parts... Using the low end of estimated [OHV] increase in the first half of the decade (due to the current recession), and using the high end on the second half of the decade, it is estimated there may be 15,000 riders using the Ochoco Summit trail system by 2020. Concentrated and increased use in the area will result in a constant need to monitor the trail system for invasive plant presence, and an increased need for treatment.” SFEIS (p. 388) asserts “All action alternatives result in a high risk, with differences due to magnitude.”

These assessments of invasive species impacts indicate that expanded OHV use in the proposed project area will exacerbate the spread of noxious and invasive species in the Project area and that violates Executive Order 13112.

2. The ONF Failed to Conduct an Adequate Cumulative Effects Analysis on Wildlife and Elk Security Habitat.

The Council on Environmental Quality (CEQ) regulations at 40 CFR 1508.7 and 1508.8 define cumulative effects and the required analysis. The Forest Service Handbook 1909.15: National Environmental Policy Act Handbook, Chapter 10: Environmental Analysis: Section 15.3 adopted the CEQ regulation and defined the framework in development of meaningful cumulative effects analyses for project proposals. The Forest Service is to describe the effects to each resource that overlap in time and space for past, present and reasonably foreseeable future actions regardless of ownership, that may combine with effects of the proposed action and result in cumulative effects. Cumulative effects can be additive or synergistic resulting in effects greater than the sum of the parts.

The SFEIS failed to conduct a full cumulative effects analysis from recent fires, and vegetation and fuels treatment projects on the forest that also add new road construction, and temporary roads. There are two essential issues with inadequately assessing these large area fires and projects. The first is not adequately including all the new road construction, reconstruction and temporary roads, which as explained above have impacts on both fish and wildlife habitats, and the second is failing to account for changes in elk security habitat with these fires and project.

The 2016 SFEIS (p. 306) in the Wildlife Section, states “Approximately 81,000 acres of vegetation and fuels management are proposed for treatments including commercial thinning, small diameter thinning and prescribed burning in these projects. The primary effect of these vegetation management projects on big game is related to roads used for access to treatment areas, roads closed as part of these projects, and changes in cover as a result of proposed vegetative treatments.” Later, the Botany section (SFEIS p. 389) reports conflicting information that states that “Approximately 130,000 to 150,000 acres of vegetation and fuels management are proposed for treatments including commercial thinning, small diameter thinning and prescribed burning.”

The 2011 Howard Elliot Johnson Vegetation and Fuels Treatment Project One will treat a large area north of Big Summit Prairie with over 13,000 acres of commercial harvest, precommercial thinning, and prescribed fire. The following two figures display how much of the project area was or will be treated (Figures 6 and 7).

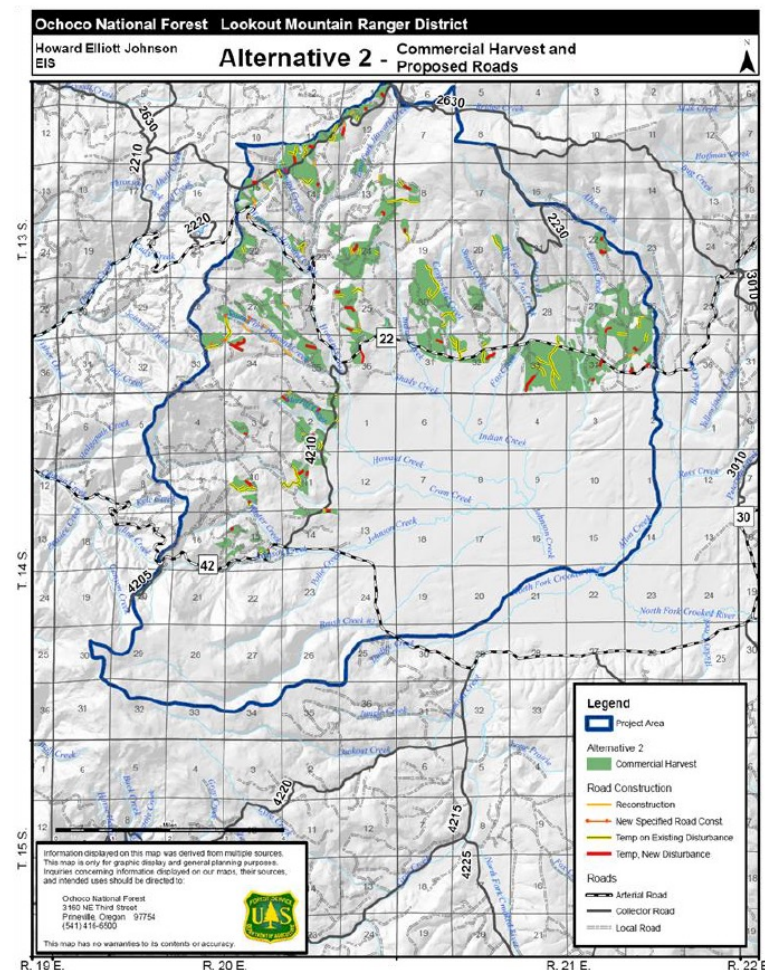


Figure 4. Map of Alternative 2: Commercial Harvest and Proposed Roads.

Figure 6. Howard Elliot Johnson Commercial Harvest and Proposed Roads; areas in green are commercial harvest; from Howard Elliot Johnson Vegetation and Fuels Management FEIS.

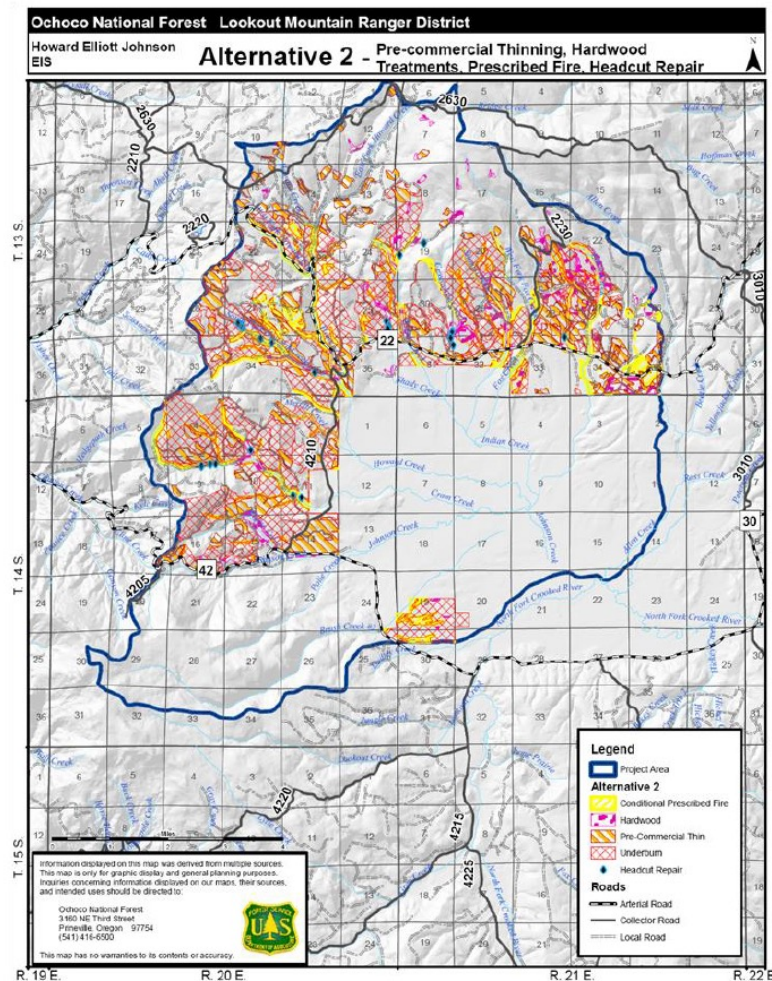


Figure 5. Map of Alternative 2: Pre-commercial Thinning, Hardwood Treatments, Prescribed Fire & Headcut Repair.

Figure 7. Howard Elliot Johnson Pre-commercial Thinning, Hardwood Treatments and Prescribed Fire. Colored areas depict treatment areas. From Howard Elliot Johnson Vegetation and Fuels Treatment FEIS.

The two figures above (Figures 6 and 7) demonstrate the large area destined for treatment that will impact elk security habitat just for one project.

SFEIS Table 7 (p. 53-55) lists the “connected actions” of past, present and future actions include recent wildfires, vegetation and fuels treatment projects and other activities. I looked up the projects for which there was information online and summarized areas to be treated either by timber harvest, precommercial thinning or prescribed fire for each project, along with estimates of acres burned by wildfires. Table 2 is a synthesis of the information I could find on these past, present and foreseeable future projects. Contrary to the ONF estimate of “81,000 acres” in the Wildlife Section, there are almost 133,000 acres that will have been burned by wildfires or treated with vegetation and fuels treatment projects and affect a much

larger area that was stated in the SDEIS or SFEIS in the wildlife analysis of impacts on elk security habitat.

Table 2. Recent fires and past, present and foreseeable future vegetation and fuels projects.*, **, ***.

Project/Fire	Type	Year	Harvest	Precommercial	Prescribed Burn	Total Acres
Ochoco Complex	Wildfire	2014				2956
Bailey Butte	Wildfire	2014				7789
Maxwell	Wildfire	2006				2317
Total Fires						13062
Spears	Veg & Fuels	2007	4935	10935	15162	21567***
Canyon	Veg & Fuels	2010	4099	5362	2038	11499
Howard Elliot Johnson	Veg & Fuels	2011	4370	3007	6247	13624
Jackson	Veg & Fuels	2012	6303	7779	7485	21567
Wolf	Veg & Fuels	2014	4706	988	5000	10694
Gap	Veg & Fuels	2016	14037	2027	5587	21651
Black Mtn	Veg & Fuels	proposed	5364	1040	3234	9638
Total Veg & Fuels						119705
Total Fires & Veg & Fuels						132767

* Vegetation and Fuels treatment acres exclude juniper and hardwood treatment areas.

** Excludes Blue Mountain Resiliency project since no acre estimates were available.

*** Portion of Spears is outside the Summit Trail Project area.

Clearly, the number of acres in Table 2 is substantially greater than the estimate in the analysis completed in the wildlife section on elk security habitat. The SFEIS fails to display on a project-by-project basis the acres of treatment, and new and temporary roads and how they affect elk security habitat. Data should have been displayed in a table by subwatershed and watershed for treatment acres and roads. It is critical to display both the road information with the vegetation and fuels treatment project information to understand cumulative effects.

The SFEIS failed to analyze and disclose the cumulative effects to elk security habitat by fully accounting for the cumulative effects of wildfires, vegetation and fuels treatment projects; along with the addition of new roads, temporary roads, user-created trails and the Summit trail system. The ONF should have employed the distance banding analysis for all roads on the landscape, not just the “Motor Vehicle Use Mans” (MVUM) which designates open roads. By not analyzing all roads, not just open roads, the ONF underestimates the effects of the roads and the vegetation and fuels treatment projects to elk security habitat. The addition of the OHV trail system along with high road densities –open, closed, decommissioned, proposed temporary - and numerous treatment projects would reveal why elk are increasingly moving from summer range on the ONF to the forest fringe and adjacent private lands earlier and earlier in the season.

A cumulative effects analysis that assesses the loss of significant acres of security cover from past, present, and foreseeable future projects and includes all motorized routes (open, closed, decommissioned, temporary, and user created) would provide a much more sound and realistic baseline from which to

assess Project effects. This baseline would provide a more accurate picture of existing conditions relative to past projects and determine actions that would better meet LRMP standards and guidelines, yet the ONF failed to do so. A more realistic understanding of existing conditions relative to LRMP goals and objectives and standards and guidelines would assist decision makers and the general public to better understand how much of the wildlife habitat has been affected by past decisions. The Ochoco Summit Trail system construction along with the projected increase in use of motorized designated trails and mixed-use roads will significantly increase impacts from existing conditions, thereby significantly impacting wildlife resources as described previously in this report.

The Ochoco Summit Project does not include a substantial amount of vegetation management, with only limited clearing and hazard tree abatement along trails and staging areas, so this Project does not tell the real story of cumulative impacts to structural habitat components when considered in combination with past, present and reasonably foreseeable salvage, hazard tree removal, commercial harvest, thinning and prescribed burning treatments.

3. The ONF SFEIS Failed to Analyze Motorized Vehicle Use on All Roads, Routes and Trails on the Landscape to Determine Project Effects to Wildlife Species and Their Habitat.

Habitats for native plants and animals, including endangered and threatened species, are impacted by motor vehicles in several ways. A significant impact is habitat fragmentation and reduced habitat connectivity as OHV roads and trails proliferate across the landscape. Reduced habitat connectivity may disrupt plant and animal movement and dispersal; lack of connectivity results in altered population dynamics and reduced potential for recolonization if a species is extirpated from a given habitat (Haddad et al. 2015). Wildlife is also directly affected by excessive noise (decibel levels/noise durations well above those of typical background noise) and other perturbations associated with OHV activities. Disturbance effects range from physiological impacts—including stress and mortality due to breakage of nest-supporting vegetation, collapsed burrows, inner ear bleeding, and vehicle-animal collisions—to altered behaviors and population distribution/dispersal patterns, which can lead to declines in local population size, survivorship, and productivity (see discussion #4 below).

The SFEIS fails to analyze all roads, routes and trails physically occupying the landscape to assess the Summit Trail System impacts on all terrestrial resources. Therefore, it underestimates the real impact because it only includes the open road density, and does not account for closed (ML 1), decommissioned, and temporary roads, and user-created OHV trails. The ONF SFEIS reports that there are 1820 miles of roads in the Project area (SFEIS p. 56) across all jurisdictions. The SFEIS further refines miles of roads in the Project area in Table 8 (SFEIS p. 59) for 1738 miles, encompassing 301,580 acres or 471.22 square miles. Using these figures, the average road density across the Project area is 3.69 miles of road per square mile. The 3.69 miles of road per square mile product does not include any user-created trails, temporary roads associated with vegetation management projects, and decommissioned roads.

The SFEIS (p. 60) identifies 188 miles of ATV trail, 87 miles of motorcycle and 159 miles of shared use of which approximately 40% are on existing roads. Using this last data set, user-created trails would add another 260 miles of routes accessed by motor vehicles increasing the road density in the Project area to 4.24 miles per square mile. This figure does not include any temporary roads associated with vegetation management projects and decommissioned roads.

The following LRMP Management Areas within the Project area are generally roadless: Lookout Mountain Recreation Area (14,365 acres), Bridge Creek Wilderness (5,355 acres), Rock/Cottonwood Roadless Area (5,098 acres), and Round Mountain National Recreation Trail (1,018 acres). Combined these management areas contain approximately 33,529 acres of largely un-roaded habitat. Subtracting the management area acres from the total Project acres equates to 268,051 acres or 418.8 square miles.

Using the road miles plus the user-created miles, $(1,738 + 260 = 1,998)$ to calculate the miles per square mile in the Project area less the above generally roadless management areas yields 4.77 miles of roads, routes and trail per square mile in the Project area. Again, the density figure still underestimates the real density of roads and trails because it does not include temporary roads for vegetation management or decommissioned roads.

The ONF LRMP Management Area map (SFEIS Appendix C, Map 2), show that General Forest, General Forest Winter Range, Winter Range, Lookout Mountain Recreation Area, Old Growth, and Wilderness - Bridge Creek provide the vast majority of management area acres within the Project area. ONF LRMP Standards and Guidelines state that in General Forest, road densities will be less than or equal to 3 miles per square mile (LRMP 4-259). In General Forest Winter Range and Winter Range, Standards and Guidelines state “except for constant service through routes, use will be restricted during the period of December 1 to May 1. Access routes will be limited to 1 mile per square mile during that period, and 3 miles/section on the average, the remainder of the year” (LRMP 4-257).

According to the ONF LRMP, the maximum allowed roads per square mile within the Project area would be 3 miles per square mile. Using the above calculations, the existing roads/routes per square mile currently being used by OHVs is greater than LRMP standards and guidelines by 1.77 miles per square mile in General Forest MA, which still doesn’t include decommissioned and temporary roads.

The ONF motorized vehicle use policy for the Forest prior to 2011 was “open unless designated closed” (SFEIS p. 248). Forest users essentially had motorized access to anywhere they could drive as long as they weren’t causing resource damage (i.e., not causing wildlife disturbance, damage to meadows and scablands, spread of weeds, etc.). With implementation of the 2011 Travel Management Plan (TMP), roads and off-road use was closed unless designated as open per the annual MVUM maps. Compliance with the 2011 TMP and MVUM maps across the ONF is low. The SFEIS states that vehicle use of the closed roads and user-created trails receive moderate to high levels of unauthorized motorized vehicle use (SFEIS p. 248). Personal observation and experience support this lack of compliance of motorized vehicles using closed roads, trails and routes. ODFW conducted a motorized vehicle survey on randomly sampled closed roads per the 2015 MVUM for 3 weeks from mid-May to early June, 2016. By June 3, 2016, the last day of the survey, 60% of the closed roads had received motorized vehicle use (Greg Jackle, ODFW District Wildlife Biologist, pers. comm.)

An example of a “closed” road is in the Little Summit Creek area and is not on the MVUM map (Figure 8). However, at the site, there is no gate or sign or any evidence that this road is closed. A forest user is supposed to know this road is closed by having their map with them in the vehicle while driving on the forest. Further, the 2015 MUVUM map conflicts with the ONF color ½ inch to the mile forest-users map. The ONF forest-users map shows many 2015 MUVUM closed roads as open roads, leading the forest user to believe all shown roads are open for their use. Many forest users do not know of the MVUM maps.

The ONF generates substantial non-compliance by forest-users by not funding and creating one forest-user map, combining the MVUM and the standard ½ mile forest-user map.



Figure 8. “Closed” road in the Project area less than 150 feet from Little Summit Creek. No gate or no obstruction is apparent. Only the MVUM map discloses that this is a “closed” road. The forest users map shows this road is available to the public.

The ONF only analyzed open roads from the 2011 TMP and MVUM maps to calculate road density per square mile even though they recognize that closed roads and user created trails continue to be used (SFEIS Alternative 1, p. 248)). ML1 “closed roads”, user-created trails, unclassified roads, vegetation treatment unit temporary roads and decommissioned roads are not used in the calculation for road density per square mile. Many of the ML 1 roads, user-created trails, temporary roads and decommissioned roads are currently being driven by OHV users, forest visitors and other recreationist simply because these roads are physically open. There are no road closure signs on the vast majority of these roads, and there is little to no law enforcement present to educate the public. ONF forest users see roads wide open and they drive them.

All roads, routes and trails occupying the landscape that receive motorized use biologically impact wildlife and their habitat by providing access to predators (SFEIS p. 243), fragmentation of habitat (SFEIS p. 244), behavioral changes (SFEIS p. 245), impacts due to noise (SFEIS p. 245) and physical alternation of habitat (SFEIS p. 247). The SFEIS (p. 249) states that “*For an estimate of acres affected due to fragmentation of habitat by the network of existing open roads and by the proposed activities refer to the column labeled “Acres within 660’ (200 m)” in Table 131.*” Therefore, the ONF underestimates the impact to wildlife and their habitat by only using open roads in their analysis (SFEIS p. 249).

In particular, Project impacts to wildlife resources will be greater when considering expected increases in frequency of use on all motorized routes: open, closed, trail, mix-use, decommissioned, temporary.

Failure to do this more complete analysis on all routes gives a false view of the current wildlife habitat condition within the Project area, and leads to the erroneous conclusion that the ONF is meeting their LRMP wildlife goals and objective and standards and guidelines (3 mi/mi²). Adding a designated trail system with the promise of better controlling unmanaged OHV use to improve wildlife habitat conditions is questionable and misleading given current conditions due to past decisions and projected outcomes.

Open roads are defined as all roads that the public is able to drive and are generally Maintenance Level (ML) 2-5 roads. The SFEIS (p.57) states that “*Operational Maintenance Levels define the degree of maintenance required for a specific road and the level of service which that road provides, consistent with road management objectives and maintenance criteria (FSH 7709.58, Transportation System Maintenance Handbook).*” Per SFEIS (p. 58), a ML 1 road is “*closed to vehicular traffic...while being maintained at level 1, they are physically closed to vehicular traffic, but may be open and suitable for non-motorized uses*”. ML 2 are “*open for use by high clearance vehicles*” while ML 3 are “*roads open and maintained for travel by a prudent driver in a standard passenger car*” and generally not maintained for comfort or convenience. ML 4 roads “*provide a moderate degree of user comfort and convenience at moderate travel speeds*” while ML 5 roads are “*Assigned to roads that provide a high degree of user comfort and convenience. Normally, roads are double-lane, paved facilities.*”

To illustrate the road density difference between what the ONF used in their analysis and what the actual road density is on the landscape, the following two maps show the most recent MVUM map of open roads and the Forest Service Administrative Use map (Figure 9), the former which is free, and the latter which can be purchased from the Forest. Both maps are of the same area located west and north of Little Summit Prairie. Neither map shows user-created trails and temporary roads associated with vegetation management projects and decommissioned roads.

The first map is the 2015 MVUM map that shows only open roads that one can legally drive on the forest, and can be anything from a rough 4 wheel drive road to a high speed paved road. The second map is the Administrative Use map that shows the vast majority of roads on the landscape and includes all of the closed ML1 roads and the open roads (ML 2-5). Note the dramatic difference in road density when ML 1 roads are added to the map.

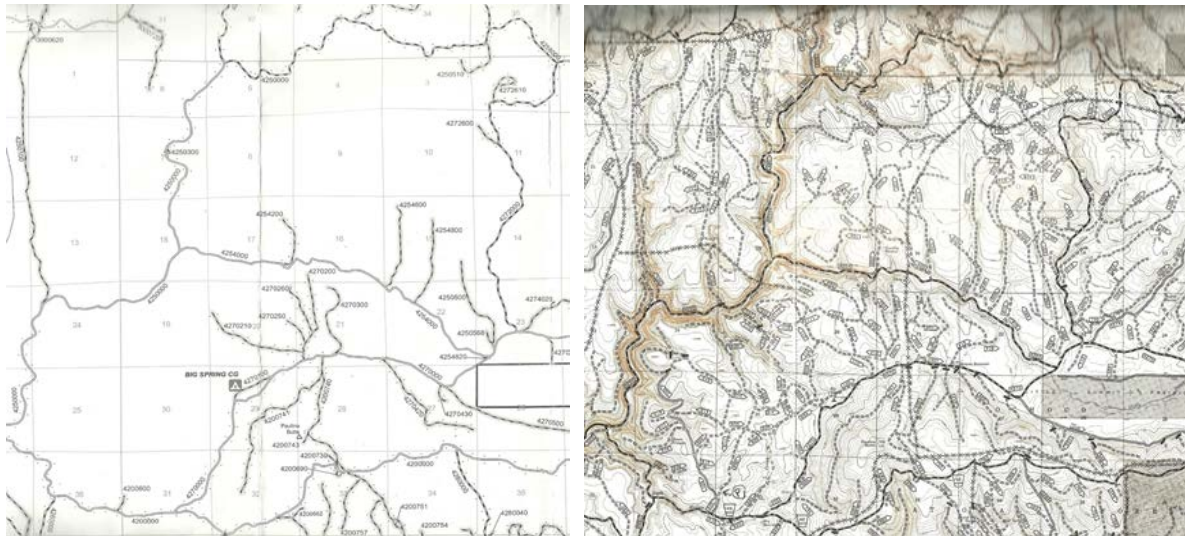


Figure 9. The first map is the MVUM (open) roads while the second map is the Administrative Use map showing all the open and “closed” (ML 1 roads) for the same area.

Currently, the vast majority of roads, routes and trails on the landscape, whether open, closed, decommissioned or user-created are used both legally and illegally. The public is frequently unaware that the 2011 Travel Management Plan limited driving on all roads or they don’t have their MVUM maps with them when driving on the forest. Forest users drive these roads, including legal and illegal, and those many users are unaware that some roads are illegal to drive. When forest users see a road without any noticeable restrictions (i.e., gates, signs, berms), they consider it available to drive.

Therefore, to accurately display environmental impacts due to road, route and trail densities, all roads, routes and trails on the landscape must be included in the analysis. Because the ONF only used the 2015 MVUM maps of open roads and proposed Project trails, they significantly underestimated the road density, therefore under estimating the current and projected environmental impacts to wildlife species and their habitat caused by motor vehicle use on all roads, routes and trails on the landscape.

The following table (Table 3) shows the management area and road density standards, and seasonal closures for different big game life history requisites.

Table 3. Data from the ONF LRMP (1989). Closures by Area. (m/m² = miles per square mile).

LRMP Management Area	Road Density
Winter Range (LRMP 4-255)	1 m/m ² During closure (12/1 – 5/1)
General Forest Winter Range	3 m/m ² Outside closure (5/1 – 11/30)
General Forest (LRMP 4-259)	3 m/m ² (all year)
Big Game Seasonal Restrictions	Ochoco NF
Winter Range (LRMP 4-255)	Dec. 1 - May 1
Calving Areas (LRMP 4-246)	May 15 - June 30
Rutting (Wallows) Areas (LRMP 4-246)	Sept. 1 - October 15

In numerous cases, the SFEIS road densities violate the open road density standards stated in the ONF LRMP. In addition, the proposed Project operating period is June 1 to September 30 are in direct conflict with the closure dates for elk calving and wallow rutting areas.

To further validate road densities, the 1996 ONF Trail System and Off-Highway Vehicle Management and Development EIS and the subsequent 2003 Roads Analysis for the forest found that road densities ranged from 0 to > 12 miles/miles². At the time, average road densities in General Forest and Winter Range designations were 4.01 miles/miles² and 3.14 miles/miles² respectively. Since then, more roads and trails have been added as part of vegetation and fuels treatment projects.

In the Wildlife Section, the SFEIS (p. 248) states “*ML1 closed roads allow for administrative use but these roads are not included in the disturbed habitat analysis*” (not used to calculate road density). Further, it states there are 140 miles of open ML 3 to 5 roads that are not legal for OHV use, 270 miles of unclassified or other jurisdiction roads, and 669 miles of closed roads (ML 1), **many of which are currently receiving unauthorized OHV use**. The SFEIS (p. 248) states that “*Given the traditional pattern of “open unless designated closed” on the ONF there are currently large areas with moderate to high levels of human disturbance*”. Further, the SFEIS (p. 237) states, “*since miles of existing user-created trails by subwatershed are unknown, these are not accounted for in road/route density estimates and likely would affect flow and sediment regimes.*”

The above discussion means that the ONF SFEIS only used open roads from the MVUM map (TMP 2011) to calculate road density per square mile. The ONF failed to discharge its duty to fully disclose and analyze all roads, routes and trails on the landscape, which significantly biased the analysis and underestimated potential Project effects on wildlife species and their habitat. Project impacts to wildlife resources will be greater when considering expected increases in frequency of use on all motorized routes: open, closed, trail, mix-use, decommissioned, temporary. Failure to do this more complete analysis on all routes and increased use with the addition of the Project, gives a false view of present and projected wildlife habitat condition in the Project area and demonstrates that the ONF fails to meet their LRMP wildlife goals and objectives, and standards and guidelines. Adding a designated trail system with the promise of better controlling unmanaged OHV use to improve wildlife habitat conditions is questionable and misleading given current conditions due to past decisions and project outcomes. When reviewing the above Figure 9, it is obvious that the existing landscape road system let alone the user-created trails impacts all wildlife that rely on the variety of habitats within the Project area.

4. The ONF Failed to Adequately Assess Impacts on Wildlife Populations and Their Habitats.

Wildlife populations are impacted by projects that fragment habitat, sever connectivity between important habitats, and eliminate habitat. In a review of forest fragmentation effects, Reed et al. (1996) found that roads add to forest fragmentation more than clear-cutting by dissecting large patches into smaller pieces and by converting forest interior habitat into edge habitat. The total landscape area affected by clearcuts and roads was 2.5-3.5 times the actual area occupied by these disturbances (Reed et al. 1996). The SFEIS discusses impacts of OHV trails on wildlife including fragmentation of habitat patches (SFEIS p. 244), behavioral changes in response to human use (SFEIS p. 245), and impacts due to noise (SFEIS p. 245).

Habitat fragmentation alters the distribution of wildlife species across the landscape and affects many functions such as feeding, courtship, breeding, and migration. In fact, fragmentation from roads and other human infrastructure has been identified as one of the greatest threats to biological diversity worldwide (Noss 1987, Wilcove 1987, Noss and Cooperrider 1994). Climate change further compounds the threats of habitat fragmentation and biodiversity loss. As animals migrate due to changing climate, landscape connectivity will be increasingly important to best ensure the survival of many species (Hansen et al. 2001; Kettunen et al. 2007).

The proposed Summit Project would cause numerous impacts to wildlife species and their habitats. Species that would be impacted include small mammals, birds, and big game including mule deer and Rocky Mountain elk. In addition, there are important sensitive habitats that would be impacted including old growth areas that provide habitat for species that are dependent on old growth forest habitat characteristics and scablands with very shallow soils and associated plants that have adapted to such unique habitat. OHVs cause both direct and indirect impacts, causing mortality and disturbance to animals, and cumulative loss of habitat for wildlife species (Switalski and Jones 2012). While direct mortality through vehicle collisions is likely low, the far greater impacts are caused by indirect mortality, reduced productivity from disturbance, and habitat loss. OHV use also increases access to more remote habitats for illegal harvest of wildlife (poaching) and is often in areas that are difficult for law enforcement officers to patrol.

Types of OHV impacts to wildlife include increased expenditure of energy due to stress from disturbance, destruction of vegetation that supports wildlife habitat and food, lower reproductive success due to disturbance during critical mating and reproductive periods, greater competition for resources due to population concentrations, and disturbance and displacement of animals. Species with greater sensitivity to motorized disturbance cannot use habitats associated with these areas (physically it's there, but functionally it's unavailable). Species such as big game can partially habituate to predictable activities in certain habitats, but sudden encounters increase stress and cause animals to move to other areas. This movement may cause greater risk of predation (Lyon 1979). Perry and Overly (1976) found in the Blue Mountains of Washington that deer reduced use in habitat 1/8 mile from roads and elk reduced use 1/2 mile from roads.

As an example, Wisdom et al. 2006 evaluated a variety of recreational activities on elk movement. Elk were displaced the furthest by OHVs compared to hiking, mountain biking and horseback riding. Figure 10 displays mean elk movement in response to activity type, with elk moving the greatest distance in response to disturbance from OHVs (Wisdom et al. 2006).

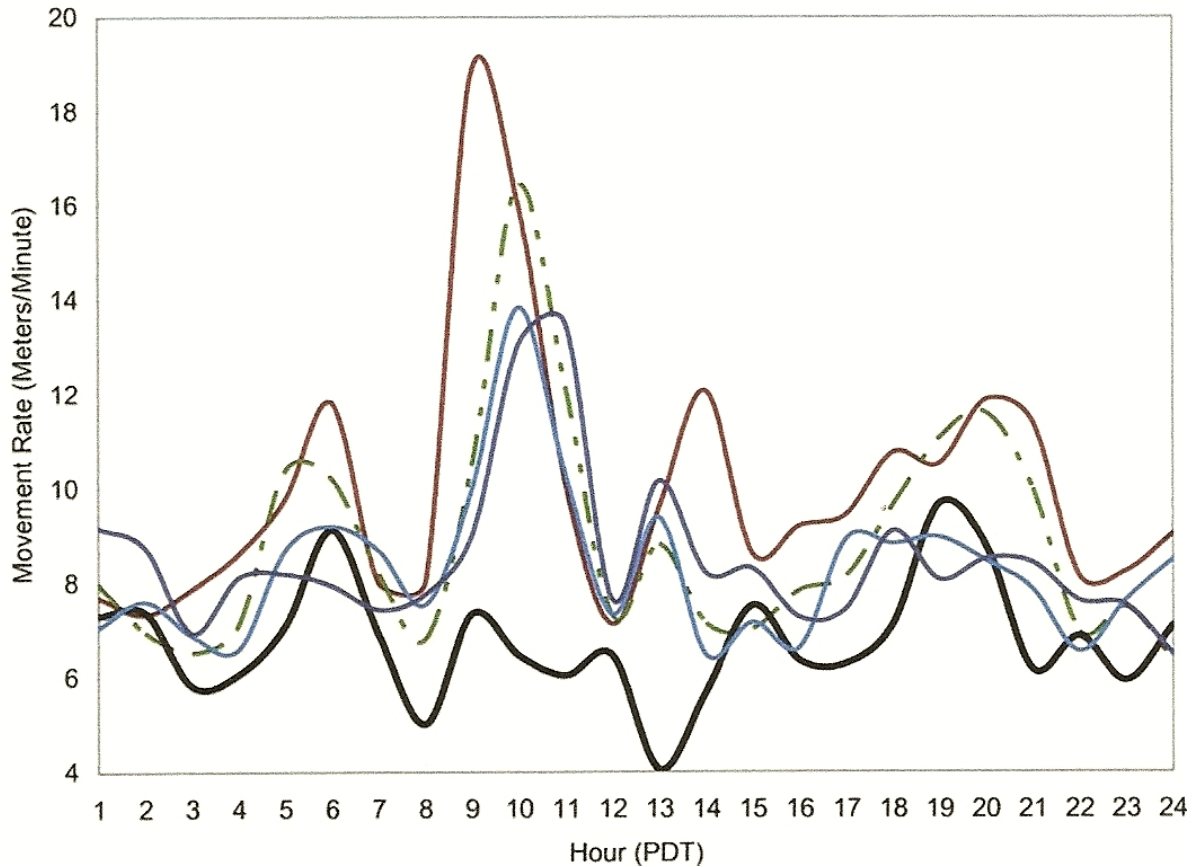


Figure 10. Elk mean movement - no human activity (black), OHV (red), hiking (solid blue), mountain biking (green dot dash) and horseback riding (solid purple)

Rowland et al. (2005) reported that road effects are far more pervasive than originally believed and include such disparate consequences as population and habitat fragmentation, accelerated rates of soil erosion, and invasion of exotic plants along roadways. One author reports that “road systems are the largest human investment and the feature most damaging to the environment” (Gucinski et al. 2001). Summaries of the effects of roads on wildlife habitats and biological systems in general have been compiled by Forman and Alexander (1998), Trombulak and Frissell (2000), Gucinski et al. (2001), Forman et al. (2003), and Gaines et al. (2003).

The ONF analyzed using distance bands to estimate the amount of acres impacted by various distances to motorized routes for each action alternative (SFEIS Tables 131 and 132, p. 250) demonstrating the amount of decreased habitat quality and avoidance due to roads. The SFEIS (p. 249) states that “*For an estimate of acres affected due to fragmentation of habitat by the network of existing open roads and by the proposed activities refer to the column labeled “Acres within 660’ (200 m)” in Table 131.*” The 200-meter (660 feet) distance from an open road is disturbed habitat for all species. While the analysis provides an initial basis, it is limited as the ONF underestimates the acres of fragmented habitat within the distance bands because it did not include all roads and trails, including those open, closed, and user created, on the landscape that are accessible and driven. Therefore, disturbance from motorized roads and trails and subsequent fragmentation of habitat is even more severe than the SFEIS displays.

The SFEIS estimated noise effects to wildlife from a literature review (SFEIS at 245-247). Because the SFEIS did not analyze all roads on the landscape and it assumed all users would stay on the designated OHV trails, the SFEIS (Tables 131 and 132, p. 250) also underestimated the impacts of noise and disturbance to wildlife. The analysis assumed that OHVs would be at their legal limit of 99 dB. Over 97% of the habitat within 0.89 mile of all routes on the landscape would be impacted by noise by any type of vehicle, thereby reducing habitat quality. At approximately 0.89 mile depending on topography, vegetation, slope wind and other environmental variables, the SFEIS estimates that OHV noise would be reduced to 45 decibels, which the SFEIS considers non-invasive to the casual observer (SFEIS p. 95). “*A decibel level of 30 is characterized as somewhere between ‘quiet’ and ‘very quiet’ and equates to a quiet rural area*” (SFEIS p. 95).

Even with just using only open road density calculations, all action alternatives increased impacts to big game, other wildlife species, and the functional availability of their habitats.

5. Rocky Mountain Elk and Mule Deer are ONF LRMP Management Indicator Species and will be negatively impacted by the Ochoco Summit Trail Project.

The SFEIS analyzes information for the Project due to impacts caused by road density and disturbance. It further addresses elk security habitat based on distance from a motorized route of more than a ½ mile and habitat blocks of at least 250 acres in size. The distance banding analysis is only based on the 2015 MVUM open road map and proposed Project trails. As stated earlier, the banding analysis should have included all roads and trails, including open, closed, decommissioned, user-created, and temporary access roads due to vegetation management projects to fully assess the effects and harm to big game caused by the motorized use of roads, routes and trails. In fact, the SFEIS (p. 310) acknowledges that “*this project would increase motorized route density and would reduce the effectiveness of security cover within the Project area*”.

In contrast, the Hydrology and Aquatics resource section (SFEIS p. 218) reported total road densities within subwatersheds exceeding the LRMP standards and guidelines. For example, both open and total road densities were compared between Alternatives 2 and 5 (Table 114) for 9 subwatersheds, but neither were compared to the existing condition of the No Action Alternative. Also not included in the Table 114 estimate was total road density for closed, decommissioned, user-created trails and temporary roads due to vegetation management projects. Table 116 (SFEIS p. 221) compares Alternatives 2 and 5 (but we’re not sure since the label is Alternatives 3 and 5 but the table shows 2 and 5) of open, total and open with the OHV trail and total with the OHH trail road densities within 300 feet of streams for the same 9 watersheds. All action alternatives violate the LRMP road density for total road densities and range from the lowest road/trail density of 4.2 mi/mi² in the Elliot Creek drainage in Alternative 5 to 8.8 mi/mi² in the Lower Deep Creek subwatershed in Alternative 2.

Data in the SFEIS is not consistent and or well defined within resource areas and between resource areas (e.g. Hydrology and Aquatics versus Wildlife), thereby making it very difficult to understand existing conditions and Project effects. For example, open road densities, total road densities, and open and total road densities within 300 feet of streams were calculated sometimes at the subwatershed level and sometimes at the watershed level, and in many cases did not show the existing condition for the No Action Alternative. Reviewing Tables 9, 114, 116, and 143 (SFEIS p. 59, 218, 221 and 294 respectively)

leave the reviewer confused on why subwatershed data is not displayed at the subwatershed level while at the watershed level in Table 143, only open road densities are shown and includes the No Action Alternative. What is obvious though, are the high levels of total road densities in all subwatersheds that violate LRMP standards and guidelines.

This discussion on road densities and the confusion of estimating impacts at the subwatershed level is important because of the huge impacts that roads and motorized trails have on Rocky Mountain elk and mule deer. They are called “management indicator species” (MIS) because when the LRMP was written, *“a group of wildlife species were identified as Management Indicator Species (MIS). These species were selected because their well-being could be used as an indicator of other species dependent upon similar habitat conditions. Indicator species can be used to assess the impacts of management actions on a wide range of other wildlife with similar habitat requirements”* (SFEIS p. 274). In other words, they are supposed to be the “bellwether” species that indicate how the rest of the wildlife species are doing that occupy similar habitats.

Numerous authors have reported on the effects of roads to Rocky Mountain elk, one of the MIS. Rowland et al. (2005) summarized the following impacts of roads on Rocky Mountain elk:

1. Elk avoid areas near open roads. A plethora of studies have demonstrated an increasing frequency of elk occurrence or indices of elk use, such as pellet groups, at greater distances from open roads (defined here as any road where motorized vehicles are allowed). This response varies in relation to traffic rates (frequency) (Johnson et al. 2000, Ager et al. 2003), the extent of forest canopy cover adjacent to roads (Perry and Overly 1977, Lyon 1979, Wisdom et al. 2004b), topography (Perry and Overly 1977, and type of road (e.g., improved versus primitive; Perry and Overly 1977, Lyon 1979, Marcum and Edge 1991, Rowland et al. 2000, Lyon and Christensen 2002, Benkobi et al. 2004), which also correlates with traffic rates (frequency). Responses may also differ between sexes, with bull elk demonstrating a stronger avoidance of areas close to roads than do cow elk (Marcum and Edge 1991). On a larger scale, entire ranges can be abandoned if disturbance from traffic on roads and the associated habitat loss and fragmentation exceed some threshold level. The ultimate effect of displacement of elk, by motorized traffic as well as other disturbances, is a temporary or permanent reduction in effective habitat for elk. Concomitant with loss of effective habitat are reduced local and regional populations (Forman et al. 2003).
2. Elk vulnerability to mortality from hunter harvest, both legal and illegal, increases as open road density increases. Survival rates of elk are reduced in areas with higher road density (Leege 1984, Gratson and Whitman 2000a, Weber et al. 2000, Hayes et al. 2002, McCorquodale et al. 2003). Closing roads offers more security to elk and may decrease hunter densities (fewer hunters may be willing to hunt without vehicle access). Poaching losses may decrease when roads are closed (Cole et al. 1997).
3. In areas of higher road density, elk exhibit higher levels of stress and increased movement rates. Higher levels of physiological indicators of stress, such as fecal glucocorticoids, have been observed in elk exposed to increased road density and traffic on roads (Millspaugh et al. 2001). In Rowland et al. (2005), the energetic costs of moving away from disturbance associated with roads may be substantial (Cole et al. 1997).

The SFEIS used distance banding to evaluate elk security habitat impacted by motorized disturbance. While distance banding is an appropriate method in evaluating elk response to disturbance, the ONF used an estimate of elk security habitat by Hillis et al. 1991. That definition of elk security habitat may be an inappropriate and out of data extrapolation. The ONF did not explain why the definition of elk security habitat used by Hillis et al 1991 would be applicable to the Project area, or why the agency chose to use Hillis et al. (1991) over other more recent studies of elk security habitat. Hillis et al. 1991 (referenced in Proffitt et al. 2012; Ranglack et al. 2016) defined elk security habitat as 250 acres or more > .5 miles from a road in western Montana in an area with contiguous conifer forest. When Hillis et al (1991) conducted their study roads were not differentiated whether they were open, closed, or decommissioned. Back then a road was a road if it was driven and travel management planning had not yet occurred.

With many current forests attempting to manage off road use and develop and implement travel management plans, it adds complexity on interpreting how animals respond to motorized routes. More recent studies of elk movement in response to motorized routes (Proffitt et al. 2012; Ranglack et al. 2016) used open road densities to evaluate elk movements during archery and rifle seasons in Montana. They evaluated other variables including canopy cover and how intermixed private and public lands were in the study areas. These two recent studies showed that motorized routes were the strongest variable that determined elk movement, even more so than canopy cover during hunting seasons. Ranglack et al. (2016) recommended significantly greater habitat protection for elk in order to maintain elk on public lands with the following variables and to manage for areas 1) that are at least 5,000 acre blocks of elk security habitat, 2) greater than 13% canopy cover, and 3) greater than 1,535 m from motorized routes. The ONF Project area is a mixed conifer forest located generally in canyons and draws associated with Class I through IV streams and having shrub-scablands occupying the ridge areas between forested areas, and more closely approximates habitat studied by Proffitt et al. (2012) and Ranglack et al. (2016). The one main difference is that topography in the Ochocos is gentler and has higher road densities that are driven by forest users, than the areas studied in Montana. Based on numerous trips to the region, in my professional judgment, southwest Montana has a similar mosaic of habitat more like the ONF of open shrub-grasslands and coniferous forest, but has generally much lower road densities than the ONF.

Applying the Hillis et al. 1991 definition of elk security habitat definition to this area significantly underestimates the impacts to elk and elk habitat availability. Data can be manipulated to either emphasize or minimize the apparent effects of a project. Data and reports can be “cherry-picked” to emphasize an argument that the proponent wants to make. This Project is an example of where information was presented to look like the analysis is objective but in fact fails to demonstrate how the Project minimizes impacts of the proposed OHV trail system on elk security habitat. For example, Table 144 (SFEIS p. 296) shows acres of elk security area greater than ½ mile from any open motorized routes for the no action and all action alternatives by 5th field watersheds. When these action alternatives estimates are lumped into the larger watersheds, the impacts seem relatively minor.

Even with only analyzing open road density impacts on elk, Table 144 (SFEIS p. 296) showed a decrease in elk security habitat acres for all action alternatives in each of the 9 - 5th field watersheds. Table 1 on p. 5 of this document is another way to show the data displayed in SFEIS Table 144. These tables showed a significant decline in elk security habitat in the Deep Creek 5th field watershed. Currently, less than 7% elk security habitat remains within the 54,561-acre watershed. The “Environmentally Preferable Alternative” (Alternative 5 plus portions of Alternative 2) impacts are not available, as the ONF did not

provide any environment analysis on its selected alternative. However, Alternative 5, which represents the majority of the Environmentally Preferable Alternative, shows a decrease of 73.5% in available elk security habitat. There were also 14.6% and 17.3% declines in elk security habitat in Upper Ochoco Creek and Lower Beaver Creek subwatersheds, respectively.

The elk security habitat acres shown in Table 144 and the impacts shown above are predicated on the use of the 2011 Travel Management Plan and the annual Motor Vehicle Use Map showing only open roads (ML 2-5), which ONF used to calculate roads/routes per square mile. As described above, the landscape is covered with roads, trails and other routes that are physically open for motor vehicle use. A more appropriate analysis is to include all roads, trails and other routes that are physically open and the public drives in order to estimate impact on elk security habitat, and would show much more degradation of elk security habitat from baseline conditions to all action alternatives.

The SFEIS (p. 260-261) discusses the amount of elk security cover and shows the amount of acres greater than ½ mile from a road and greater than 250 acres in size. Even with these much smaller blocks of elk security habitat compared to the recommendations by studies in Montana for larger blocks, all action alternatives decrease the amount of elk habitat from the No Action alternative. What is not shown is that, already 87% of the Project area, even with the No Action Alternative, which is entirely in elk summer range, has been compromised by the “open” roads, and fails to account for the ongoing impacts of closed, decommissioned and user-created roads/trails. If all roads were used in the elk habitat analysis for the action alternatives, like those road densities identified above (SFEIS p. 221), there would be a much greater impact to habitat than in the existing condition. Further compounding the existing condition with a new OHV trail system will further diminish and harm elk security habitat and elk populations.

Given the extrapolation of the Hillis et al (1991) paradigm of a heavily forested area and the new information from habitats more similar to the ONF, in my professional opinion, it is more appropriate to refine the elk security habitat definition to a banding to at least an interval of > .75 miles (3,960 feet) and > 250 acres. If these new estimates for elk security habitat were evaluated, the existing condition would likely show that there is significantly less elk security habitat. Using the above-suggested banding interval would better evaluate impacts to elk security habitat given the lack of contiguous conifer forest. Furthermore, if all roads, routes and trails on the landscape, that are currently being driven were incorporated and modeled using the suggested distance banding criteria, the effects to elk security habitat would be even more significant.

Wisdom et al (2006) reported that the more frequent vehicles in elk habitat in a 12 hour period, the greater distance elk moved in response to the vehicle disturbances (Table 4).

Table 4. Elk Movement in Response to Vehicle Trips in a 12 Hour Period

Traffic Volume	Vehicles Per Time	Mean Distance of Elk Movement
Low	< 2 vehicles per 12 hours	869-880 m
Moderate	2-4 vehicles per 12 hours	909-1032 m
High	> 4 vehicles per 12 hours	1103-1560 m

The SFEIS estimated 5,000 visitors (SFEIS p. 290) participating in the OHV trails in the Recreation section, and 15,000 visitors (SFEIS p. 387) by 2020 in the Botany section. Assuming an average of the two estimates yields an estimated 10,000 visitors during one season of use, and dividing by the number of days in the riding season of 122 days (June 1 to September 30), results in an average of 82 visitors per day. While seasonal use may be higher on weekends and lower on weekdays, regardless, a mean of 82 visitors per day is a lot of motorized activity. Fisher et al. (2002) reported that the median distance traveled (including motorcycles, ATVs and 4x4) of miles traveled on OHVs by users during their last outing was 50 miles. Therefore, on any average day on the ONF, there could be at least 40 vehicles on each of the 2 implementation areas. An OHV trail system that produces that level or more of motorized vehicles will significantly alter the available habitat and displace elk.

The SFEIS does not comply with the General Forest road density standard of 3 miles per square mile and therefore continues to degrade and fragment big game wildlife habitat, failing to assist the State of Oregon in meeting management objectives and goals for big game, and fails to avoid or minimize impacts by development and implementation of the OHV Project. Further, it appears that the above rough calculation of visitor days would decrease and possibly even eliminate elk of the Project area.

6. The FEIS Failed to Incorporate and Acknowledge the Importance and Protection of Strategy Habitats or Species in 1) the Forest Service Terrestrial Resource and Conservation Area Strategy (TRACS) and 2) With Partner Agencies such as the ODFW Oregon Conservation Strategy (ODFW 2006), including the Blue Mountain BM-04 Conservation Opportunity Area as Important Strategy Habitat and Species to be Conserved, Restored and Enhanced.

The Project is inconsistent with the protection and restoration of key strategy species and their habitats in the North Fork Crooked River basin and Big Summit areas in the Region 6 TRACS and the ODFW Oregon Conservation Strategy (OCS). These same overlapping conservation and restoration priorities are also on The Nature Conservancy Ecoregion Priorities, the Eastern Oregon Bird Conservation Plan, and the Interior Columbia Basin Ecosystem Management Project plant biodiversity area.

The USDA Forest Service PNW Region 6 developed the TRACS as a framework for the region and forests to identify key habitats and wildlife species to protect, conserve, restore and enhance habitats for these landscapes (USDA 2012) (Figure 11). The TRACS theme is “protect the best and restore the rest”. It’s a comprehensive planning tool to identify the most important terrestrial species, habitats and watersheds by R6 ecoregion, and provides examples how this information can be used to integrate restoration projects and accomplish multiple objectives simultaneously. The vision of TRACS is to strategically conserve, restore and enhance high-quality habitat and improve less-healthy ecosystems with a general theme of “Protect the best and restore the rest”. It includes collaboration with other entities and partners, and coordinated actions to benefit priority species and habitats in key watersheds. It was developed by Region 6 staff and biologists, botanists, and other natural resource and planning staff.

In the section on Minimum Roads Analysis, the guidance to the forests in Region 6 was that TRACS should be a primary consideration when completing a Minimum Roads Analysis. Roads are a major threat to priority species and habitats, and cause disturbance to 52% of the TRACS Priority Vertebrate Species. Information in the TRACS databases is to identify species and areas that are most important to protect with respect to roads, as well as areas where access is important to conduct restoration efforts

(USDA 2012). The priority area on the ONF includes the Upper North Fork Crooked River watershed including Big Summit Prairie and tributary areas and includes an overlapping Aquatic Restoration and Conservation Strategy (ARCS) area. For example, TRACS identifies the Upland Sandpiper as a species of concern because it historically nested in the Blue Mountains Ecoregion in montane meadows from 1,000 to 30,000 acres at 3,400 to 5,060 in elevation and is a Region 6 sensitive species. Both the Upper and Lower North Fork Crooked River watersheds were identified as important habitats to protect.

TRACS also identified elk as a priority vertebrate species in the Blue Mountain and East Cascade ecoregions. TRACS recommended that the Blue Mountains Elk Model (the joint ODFW- Forest Service habitat model) be used to identify areas of the highest value for habitat enhancement or restoration. TRACS also recommended that areas where elk populations numbers are below state game management unit objectives (i.e. the Ochoco Wildlife Management Unit where the Summit OHV Project is proposed), that projects should be implemented to reduce open road densities where they are excessive and maintain adequate levels of cover (USDA 2012).

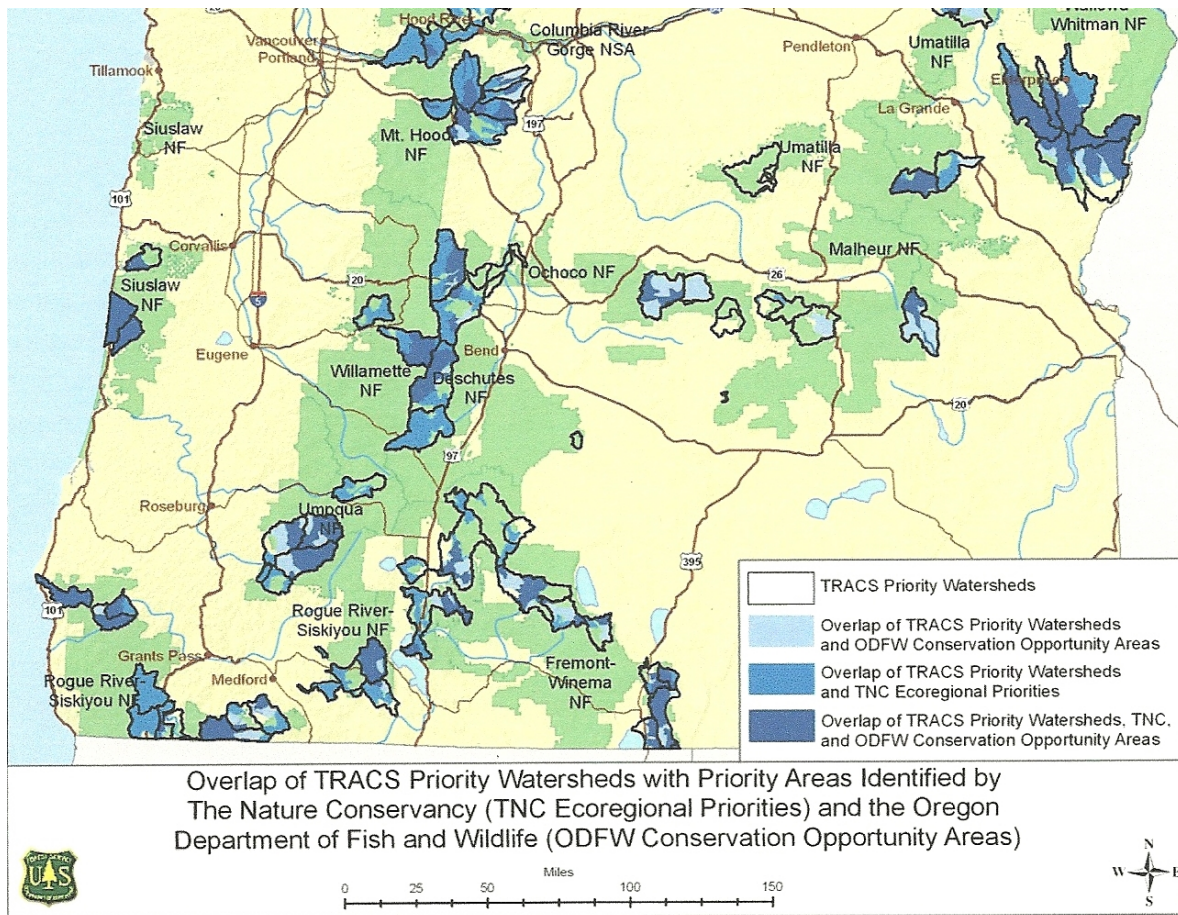


Figure 11. The OHV Project overlays a key TRACS priority watershed, ODFW's conservation opportunity areas, and The Nature Conservancy Ecoregion priority area for protection (USDA 2012).

TRACS also identifies mule deer as a priority vertebrate species in all ecoregions east of the Cascade crest (Forest Service 2012). Both Deep Creek and the Upper North Fork Crooked River were cited as

priority watersheds for protection and restoration. Threats to mule deer include roads (mortality, disturbance, barrier), poaching, recreation (motorized and non-motorized), fire exclusion and habitat loss due to invasive species (USDA 2012).

Similar to the elk population decline seen in Figure 1 (page 3 of this document), mule deer in the Ochoco Wildlife Management unit are also on the decline and are well below the ODFW management objective (Figure 12).

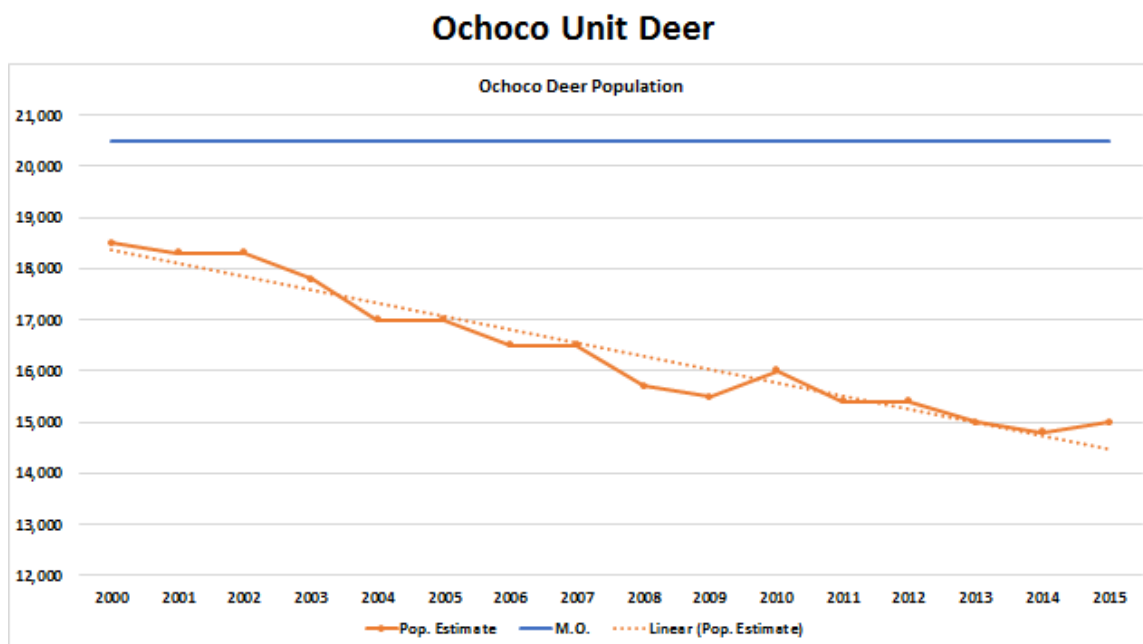


Figure 12. Ochoco Management Unit mule deer population estimates, 2000-2015. The orange line is the annual population estimate for elk while the “management objective” is the blue line (data from Greg Jackle, ODFW District Wildlife Biologist, pers. comm.).

TRACS states that mule deer are “especially sensitive to disturbance and direct mortality from roads. Impacts from different levels of access and recreational use should be considered alongside the need for hiding cover. Researchers have established that unlike elk, mule deer do not flee long distances...Permanent or seasonal road closures may be needed to protect key winter range and fawning areas...New recreation developments may impact mule deer, especially those that might influence movement patterns across the landscape such as trails...Off-highway vehicles trails and cross-country trail have increased exponentially in the past few decades, and can displace mule deer into unfamiliar or less productive habitat. High open road densities also provide increased opportunity for poaching” (USDA 2012).

TRACS identified several priority habitats: aspen stands, springs and seeps in all ecoregions. TRACS specifically identified priority watersheds in the Crooked River and Deep Creek, and late-seral ponderosa pine habitat with Deep Creek and Upper North Fork Crooked River as priority watersheds. Elk and mule deer were also identified as priority social/economic vertebrates for late seral ponderosa pine habitat.

In addition, the Blue Mountain 04 (BM-04) of the OCS is a Conservation Opportunity Area that recognizes the importance of the North Fork Crooked River (a Wild and Scenic River) and its tributaries, and the Big Summit Prairie, one of the largest montane wetlands in eastern Oregon (ODFW 2006, 2016). Key habitats include aquatic/riparian areas, wetlands and wet meadows. Further, this same area is also recommended for protection in the American Fisheries Society Aquatic Diversity Areas, the Eastern Oregon Bird Conservation Plan, the Interior Columbia Basin Ecosystem Management Project plant biodiversity area, and The Nature Conservancy Ecoregion Priorities.

The fundamental issue is that the USFS TRACS, ODFW OCS and others have designated this area of the Ochocos as a priority area for protection, conservation and restoration. The ONF failed to consider and logically apply Regional direction to use TRACS and ARCS when proposing and locating projects. The analyses to locate and implement the Summit Trail project using TRACS would have quickly revealed how inconsistent the OHV project is with Region 6 direction for conservation, restoration and enhancement. This area should receive a higher level of protection and conservation in recognition of the important wildlife species and habitats that occupy this special area, rather than less protection. By failing to apply the TRACS strategy and working with partners such as ODFW, the ONF misses an important opportunity to protect and restore the Big Summit Prairie area and North Fork Crooked River and its tributaries. Given the potential damage by motorized recreation, plans for the area should be to de-emphasize motorized recreation and implement a more appropriate strategy that is consistent with TRACS to protect, conserve and restore fish and wildlife. For example, other projects that emphasize recreational pursuits such as wildflower photography, bird watching, nature study, hiking, dispersed camping, and horseback riding would be more aligned with conservation of these important resources. Development of this OHV system will harm and displace fish and wildlife resources and is inconsistent with Region 6 TRACS and ARCS guidance to protect and restore this unique and sensitive area.

7. The ONF Designates a Large Project Area that Dilutes Project Effects.

The Summit Trail Project area of 301,580 acres covers such a vast area of the Forest that it dilutes and disguises the numerous effects to fish and wildlife species and their habitat. When the Deschutes Provincial Advisory Committee reviewed potential areas that could be used to analyze potential OHV trail sites, working group members never intended for such a large area to be developed anywhere on the ONF unless the ONF realistically could significantly improve wildlife habitat as outlined in the Working Group Recommendations (Glen Ardt, retired ODFW biologist and former Deschutes PAC member, pers. comm.) One of the agreements in the PAC was that the forests were expected to meet LRMP standards and guidelines for road densities by effectively closing roads (physically block, not just a paper exercise). The large size of a community consensus area was to provide flexibility to find a place to put in a smaller scale OHV trail system, not encompass such a huge geographic area. By doing this approach of a large OHV trail system over a huge area, the ONF misused the guidance from the PAC and used inappropriate analyses to conclude that there is a small impact to resources.

The Project area is further divided into the OHV Management Area (MA) that encompasses the Implementation Area (IA) for each action alternative (SFEIS p. 28). The SFEIS, while providing small-scale maps for each action alternative MA and IA, did not provide acreages for any action alternative MA or IA (SFEIS Maps). The OHV MA would serve as the overall management area for motorized travel. The IA would serve as the focus area for trails, education, law enforcement, and monitoring.

The ONF cites 40 CFR 1508.25 “Scope” which states the EIS must consist of a range of actions, a range of alternatives (including no action), and impacts to be considered. The CFR does not define the geographic scale of the Project area. There is no regulation that states that a project has to use a huge watershed area to analyze Project effects.

The OHV MA for Alternative 5 encompasses approximately 35% of the total Project area (our best estimate given the small scale map provided in the SFEIS Maps) that is predominately the General Forest MA. If the ONF used GIS to conduct its resource analysis of Alternative 5 OHV MA within portions of the 9-watersheds, the analysis results would show significant impacts to fish and wildlife species and their habitat simply because the impact analysis would be restricted to approximately 100,000 acres instead of the vast Project area acreage of 301,580.

For example, the ONF relies on the MVUM maps, the Project trails and road densities, and the 5th field watersheds to display Project effects to elk security habitat. Table 144 (SFEIS p. 296) shows the 9 watershed effects by alternative. It shows that the Deep Creek watershed is significantly impacted by all action alternatives plus Lower Beaver Creek and Upper Ochoco Creek watersheds are moderately impacted by the action alternatives. In applying the above premise where the OHV MA is analyzed within portions of the 9-watersheds and analyzing all roads, routes and trails on the landscape, the results would show significant impacts to elk security habitat across all 9-watersheds thereby revealing far more accurate impacts to elk security habitat associated with implementing the Summit Trail Project. We do not have access to GIS to display the above describe analysis method to show actual Project impacts.

Another example is how shrub-scabland habitat and impacts caused by the Summit Trail Project are marginalized by the large size of the Project area. A small impact zone was utilized to determine Project effects. The SFEIS (p. 375) states there are 23.6 miles of trail proposed in Alternative 5 in shrub-scabland habitat type. By using only a 10 foot width for trail disturbance, only about 28.6 acres of the 48,447 acres will be impacted in scabland habitat (less than 1%). Yet, the LRMP describes scablands “*among the most fragile ecosystems on the Ochoco National Forest. Damage to the soil and vegetation as a result of management activities is nearly impossible to mitigate. This is a result of their having very shallow soils which are subject to severe water saturation and frost heaving during winter, thus making revegetation virtually impossible*”(LRMP 4-197). Figure 13 depicts a scabland area with an old roadbed that is proposed to become an OHV trail. The road to the left is supposed to stay “closed”.



Figure 13. Scablands with “closed” roads. The road straight ahead is a proposed OHV trail while the road to the left is supposed to remain “closed”.

A project design criteria was developed to protect TES plant habitat, by closing the trail system when resource damage occurs such as rutting and runoff. However, once the damage is done it is permanent.

The small impact zone is based on the direction given to the Forest IDT, which was to assume all riders will stay on the designated trail and to base the analysis of Project effects to the narrow linear trail feature (Barb Franano, Ochoco Forest retired fish biologist, pers. comm.). OHV monitoring results, OHV videos on the internet, research (Kiely & Kassar 2007), and first-hand experience show that OHV riders prefer to ride off-trail. Given the openness of the shrub-scrubland habitat and the many acres of trail through this type habitat, it is a given that a percentage of riders will “play” off trail. Even if it is only a few riders, many acres of this habitat type will be destroyed.

These are just a couple of examples of how a huge project area is used to display a small impact on a linear feature that is only assumed to be 10 feet in width. It leads to a biased conclusion that the OHV trails will have little impact.

8. The Project Design Criteria and Mitigation Measures are Insufficient to Mitigate for Impacts Caused by the Summit OHV Trail System to Sensitive Wildlife and Their Habitats, and Botanical Resources.

The Project Design Criteria (PDC) (SFEIS p. 29-37) elements do not comply with LRMP Standards and Guidelines and are insufficient to mitigate the impacts of the Summit Project.

For example, as discussed in 1A. Elk Calving and Wallow areas, the LRMP standards and guidelines state that projects are to:

1. Protect the character of elk calving sites. Minimize disturbance from human activity during calving season (approximately May 15 to June 30).
Protect wallows during rutting season (September 1 to October 15) (LRMP 4-246).
2. Antelope: Manage antelope habitat on the Forest in accordance with ODFW population objectives (LRMP 4-248). While ODFW does not have specific population management objectives, they monitor for abundance annually and set harvest tags based on monitoring.

The SFEIS (p. 36) states the PDC for big game does not allow construction, de-construction and maintenance activities for May 15-June 30 within elk calving areas. However, the PDC does not restrict OHVs riders using trails during the month of June when elk and other big game species are dropping their young. Riding OHVs during the calving, fawning and kidding period will highly impact these game species by displacing them from high quality birthing sites due to disturbance from OHV activities, into less functional sites (Lehman et.al. 2015), and will likely result in reduced productivity of the population. The same is true for timing restrictions for known elk wallows. Restrictions for trail construction, reconstruction, decommissioning and maintenance activities are applied from September 1 to October 15 for known elk wallows, but do not prohibit OHV trail use from September 1-30, allowing disturbance impacts to elk during the mating season by OHV riders.

The PDCs for TES plants states that the Forest Service will conduct surveys prior to trail construction and disturbance (SFEIS at 33). Since the release of the 2013 DEIS, the ONF has known the identified OHV routes. As is common with vegetation management projects, site-specific surveys are completed before

release of the environmental documents. Between issuance of the DEIS and SFEIS (a period of 3 1/2 years), these trails could have been surveyed to display Project effects for public review and comment. By delaying surveys until after the NEPA process is completed, the survey routes could substantially change, depending on species and habitats encountered, thereby changing Project design and effects without public review and comment.

Mitigation measures are implemented to compensate for Project impacts. Mitigation is the so-called “last defense” or “last resort” when and where there is no opportunity to avoid, minimize, or reduce the impact. The agency failed to consider Best Management Practices (BMPs) identified in OHV Best Management Practices for Forestlands (Switalksi and Jones 2012). Only once the impacts have been avoided or minimized should the agency establish site-specific management actions to reduce or mitigate remaining impacts.

For example, the mitigation measure to protect sensitive plant habitats with a split rail fence (SFEIS p. 38) is insufficient. It is unclear in the SFEIS if the purpose of the fence is to protect the plants or to contain the OHV riders. The companion element for monitoring is absent. Given that there are Regional Forester sensitive plant populations, it is more appropriate to avoid and eliminate the OHV trail and staging areas to meet Forest Service management direction. All action alternatives will increase the spread of noxious and invasive weed species, leading to further deterioration of native plants, and fish and wildlife habitat (see discussion in #1F).

The ONF fails to sufficiently provide adequate PDC and mitigation measures, let alone responsible monitoring elements to meet CEQ regulations for minimizing Project affects to forest natural resources.

9. The SFEIS Fails to Address the Impacts of Illegal OHV Use.

The SFEIS (p. 60) states that there are “*user-created motorized trails within the 33 subwatersheds, but it was not calculated for each subwatershed. According to the “COMAC” inventory there are potential routes identified; 188 miles of ATV trails, 87 miles of motorcycle and 159 miles of shared use. Approximately 40% of the routes identified are on existing open roads.*” Therefore, there is an estimated 435 miles of trails, of which approximately 260 miles are cross-country user-created trails. The SFEIS (p. 79) then contradicts the previous statement and reports that “*approximately 700 miles of user created OHV routes, including open and closed roads and user-created trails have been identified in the Project area*”. If we assume that 40% are on existing open roads, then approximately 420 miles are cross-country user-created trails. That is a substantial number of motorized trail miles that will impact habitat and wildlife populations.

There is no analysis anywhere in the SFEIS of user-created trails and their effects to natural resources. See Discussion in Issue #2 for lack of disclosure of open, closed, decommissioned and user-created trails on the landscape and their effect to all natural resources.

One study of OHV use on public lands reported that increased use of All Terrain Vehicles (ATVs), even with a small percentage of riders who travel off roads and trails, can cause considerable detrimental impacts on the natural resources around them (Meadows et al. 2008). These effects from irresponsible OHV users are impossible to contain these OHVs are powerful machines and can travel many miles quickly and damage sensitive resources easily.

As an example, the ONF has a “managed” motorized travel area on the Crooked River National Grassland that has been in place since 1992 at the Henderson Flat OHV area. Illegal trail and road development by OHV and 4-wheel drive (4WD) users has been rampant in the last 24 years and has negatively impacted deer and antelope winter range, and damaged soils, vegetation and water quality on the Grassland (Figures 14-17). The same is true for McKay Creek on the forest, despite efforts by fisheries staff of the forest to restore the drainage.



Figure 14. “Closed” OHV road



Figure 15. Illegal OHV trails in Skull Hollow



Figure 16. Illegal 4WD road near Skull Hollow



Figure 17. Illegal motorcycle trail Lone Pine



Figure 18 and 19. Soil erosion and loss of habitat near McKay Creek from illegal OHVs. Photos by Wayne Elmore.

Adding a new OHV area of this magnitude will further stretch the very limited law enforcement resources already in place (One Law Enforcement Officer oversees one million acres).

The SFEIS argues that the Summit Trail Project would better distribute OHV use but in fact, these illegal roads and trails have never been restored, and are rarely if ever enforced, the latter of which was the intent of the 2011 Travel Management Plan. These illegal trails continue to degrade water quality, increase sedimentation, damage sensitive riparian and wetland habitats, spread noxious and invasive weed species, and impact important year-round and big game winter range for elk, mule deer and antelope. In fact, the new OHV Project would take the already limited law enforcement and direct it at these new trails, further shifting resources away from the illegal trails, rampant throughout the remainder of the Forest and Grassland.

Conclusion

All the actions associated with the Summit Trail Project are discretionary, including the choice to implement the Project, to use MVUM roads as trails or construct new trails or convert closed and decommissioned roads into OHV trails, and to locate proposed trails through or away from sensitive habitats. By selecting Alternative 5 modified by Alternative 2, the ONF has chosen to not avoid or minimize impacts to many sensitive habitats including elk security habitat, elk calving and wallows, shrub scablands, late successional old growth areas, Class I - IV streams, riparian areas, wetlands, wet meadows and many others. In fact, by not avoiding or minimizing impacts to shrub scablands and old growth, the ONF must do Forest Plan Amendments to allow new motorized trails in these sensitive habitats, another discretionary choice that fails to avoid or minimize impacts to important natural resources.

As a professional wildlife biologist for 26 years with the USDA Forest Service, 4 years with the Oregon Department of Fish and Wildlife and 4 years as a Professional Consulting Wildlife Biologist, I have had ample opportunity to draft environmental documents for public review and comment. I have always strived to make my data, its analysis and conclusions, clear and unbiased for public understanding.

My review of the ONF Summit Trail System Project was challenging at best as the basic data used between resource areas was inconsistent and in numerous cases misleading. Most of the analyses were completed with GIS and desktop computers, and with little or no field verification, and surveys. Presentation of the contents of the SFEIS was NOT in a clear and open fashion so the public reader, let alone a Professional Wildlife Biologist, could readily understand and find logical conclusions. I had to have numerous documents open to follow the data, analyses and discussion that may or may not have led to a rational conclusion.

Finally, I find it unconscionable that the Forest Service did not fully disclose the entire Project record at the SFEIS release so the public would have the full 45 days to review and provide timely objections. I also find it inexcusable that the SFEIS was full of inconsistent disclosures and data within and across resource discussions in Chapter 3. In an EIS, Chapter 3 is the most important section of the environmental document, in that the existing condition is analyzed and objectively compared to all action alternatives to fully disclose direct, indirect and cumulative effects of the proposed Project.

The Summit Trail System SFEIS fails to fully disclose effects per NEPA to all natural resources. Instead, it seems the document was crafted to facilitate a decision to implement an OHV trail system on the ONF.

Respectfully,

/s/ Michael G. Gerdes (verifiable signature available upon request)

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Qualifications:

- Extensive knowledge of ecosystem management principles including a comprehensive understanding of land and resource management principles and practices
- Significant knowledge & experience in settlement negotiations with proponents and all stakeholders
- Highly effective in managing staff, administering fiscal resources and information technology to accomplish agency mission & objectives
- Effective skills, ability & knowledge to handle multiple resources, balance many programs and projects, set priorities, organize personnel, make decisions and keep to time schedules to meet program goals, all of which are necessary for agency accountability
- Highly effective oral and written communication skills
- Significant knowledge & experience in all hydropower (including transmission line) resource (aquatic, terrestrial, recreation, aesthetic, land use, cultural, etc) issues, study design & methodology development, study implementation, FS independent analysis of study results, mitigation development and defining rationale for substantial evidence supporting FS 4e conditions, critical review and comment on license applications, draft & final environmental assessments and/or impact statements, and new license review, comment and rehearing submittals if appropriate

Education: A.S. Wildlife Mgmt 1978 College of the Siskiyous Weed, CA
 B.S. Wildlife Mgmt 1982 Humboldt State University Arcata, CA

Professional Profile:

4/12 - 12/15	Consulting Wildlife Biologist
10/02 - 4/12	Zone Terrestrial Resource Specialist, GS-401-12 Regional Hydropower Assistance Team Regional Energy Team
11/98 - 10/02	Wildlife & Terrestrial Resource Specialist, GS-401-13 National Hydropower Assistance Team
2/98 - 11/98	Wildlife & Terrestrial Resource Specialist, GS-486-12 Regional Hydropower Assistance Team
2/96 - 2/98	Forest Wildlife Biologist, GS-486-12
11/92 - 1/96	Forest Wildlife Biologist, GS-486-11
3/90 - 10/92	Spotted Owl Coordinator, GS-486-9
5/89 - 2/90	OR Fish & Wildlife Biologist
3/87 - 4/89	OR Fish & Wildlife Tech
4/86 - 3/87	OR Fish & Wildlife Tech
3/85 - 10/85	Wildlife Biologist Supervisor
2/82 - 6/84	Wildlife Biologist, GS-486-7
7/81 - 2/82	Biological Tech, GS-404-5