House Committee on Oversight and Reform
2154 Rayburn House Office Building
Washington, DC 20515

Via email: oversight_clerks@mail.house.gov

Re: Hearing "Fighting Fire with Fire: Evaluating the Role of Forest Management in Reducing Catastrophic Wildfires," March 16, 2022

March 21, 2022

House Committee on Oversight and Reform:

Friends of the Clearwater respectfully submits this testimony to be submitted into the hearing titled Hearing on "Fighting Fire with Fire: Evaluating the Role of Forest Management in Reducing Catastrophic Wildfires" in the House Committee on Oversight and Reform, Subcommittee on Environment. Because email is the only way to timely introduce material in the record, Friends of the Clearwater was limited in the documents it could send by email. We have linked as many materials as time would allow. If there is any scientific articles or reports that cited below that the committee wants to see as it continues its investigation, please let us know and we would be happy to provide them.

Friends of the Clearwater is a nonprofit with staff in Idaho and Montana who watchdog our mission area, which includes about four million acres in North-central Idaho, mostly comprised of the Nez Perce and Clearwater National Forests, with small parts of our mission area in the Idaho Panhandle National Forest and the Bitterroot National Forest, which straddles Idaho and Montana.

In the opening of this hearing, Representative Ro Khanna noted the importance of a science-based approach to "forest management" as it pertains to wildfire. We certainly hope the Oversight Committee continues its investigation by speaking to more scientists, and hope this is the beginning of several more hearings with different sub-focuses under the gigantic and many-faceted topic of wildfire and forestry. As this testimony cites to articles, the independent scientists who studied what became those articles, are precisely whom this committee should approach for additional testimony. Dr. Dominick DellaSala is a terrific first witness, but this Committee needs to invite more independent scientists like him. The Committee should also consider inviting tribes, organizations, and individuals whose can provide on-the-ground knowledge of the effects of the Forest Service's management of public lands in the name of wildfire.

The first important thing to recognize is that, while Chief Moore corrected the oversight committee that it wasn't "timber harvesting" so much as "managing the forest," Chairwoman

Maloney and Ms. King's exchange on the euphemistic terms utilized by the Forest Service were accurately on point—the Forest Service is taking the most valuable trees. Chief Moore's categorization of "managing the forest," was disingenuous. Chief Moore discusses the confines of working "within the market" that exists with a "viable timber program." That market that exists for *timber harvest* of relatively mature trees. To highlight this point, Friends of the Clearwater provides pictures below of what projects look like after they were completed on the Nez Perce and Clearwater National Forests of North-Central Idaho. Of the pictures below, the Forest Service called one a "timber sale," one a "project," one a "wildfire protection project," and one a "vegetation management project."

See footnote for the answers.[1]

When Chief Moore talks about "managing the forest" as opposed to timber harvest, the Forest Service might call it that, but to organizations that watchdog management of our public lands, "managing the forest" is a euphemism for the substantive act of logging.

We also note that not all ecosystems are the same. There are some dryer places where fire is different and fire-return intervals are different. Cultural burning was historically practiced, but it varied depending on places and was done by a smaller human population and not at a landscape scale. We wouldn't want to see a situation where the US Forest Service ignores the science on when and where this happened and ignores Tribes to culturally appropriate the practice at a landscape scale.

Before moving onto science as it pertains to fire and forestry as it impacts our ecosystems, a short word about grazing because Representative Herrell mentioned this activity. Representative Herrell stated that grazing is both a forest and wildlife management tool and an economic necessity. For forest and wildlife management, the science does not support this. Livestock grazing, permitted on the public's national forests, contribute to greenhouse gases and make the land where they are physically present more vulnerable to the impacts of climate change. Beschta et al. 2012 Livestock grazing reduces carbon storage, Daryanto et al. 2013. Livestock grazing can compound impacts from climate change. For example, one cow may consume up to 30 gallons of water per day, Rasby 2016, which in times of drought can further impact flower production for native bees. Brookshire and Weaver 2015. Livestock trample streambanks, which can widen streams, which cause them to absorb more solar radiation and make them warmer. Nussle et al, 2015 & 2017. Livestock grazing spreads invasive weeds and corresponds with increased instances of cheatgrass. Williamson et al., 2019. ***These invasive grasses can be highly flammable and contribute increased fire risk, including increasing fire occurrence by 230 percent and fire frequency by 150 percent.*** Fusco et al. 2019. Grazing on public lands is like processed sugar in one's diet: the healthier choice is to cut it back.

Representative Tlaib asked about taxpayer subsidies, but because Representative Herrell raised the issue, we point to **taxpayer subsidies for grazing** as well. While Representative Herrell stated that grazing is an "economic necessity," when on public lands, and perhaps it is according to the for-profit ranching industry, grazing is another activity that taxpayers subsidize. Taxpayers help line the pockets of private companies who profit from grazing, grazing that impairs public lands and contributes to global warming.

Taxpayers subsidize logging on public lands. To answer Representative Tlaib's question about who pays for this, taxpayers subsidize the federal logging program. Even though Chief Moore discussed working within the "market available," that market is a money pit that taxpayers fund. Logging, which costs taxpayers biodiversity, poorer water quality, and an increased risk from fires (discussed below) costs the taxpayer between $1.5 to $2.0 billion dollars per year. Talberth

[1] Number 1 is the Adams Camp Wildfire Protection Project. Number 2 is the Iron Mountain Vegetation Management Project. Number 3 is the Little Slate Project. Number 4 is the Cove-Mallard Timber Sale. All of these projects that involved "managing the forest" had regeneration cuts, which is a forestry term for removing most of the trees in the area, whether that be by clearcut, shelterwood cut, or seed tree cut.

and Niemi 2019. **And that cost does not appear to account for timber theft.** Gorte 1995. For example, below is completed work by loggers in the "Adams Camp Wildfire Protection Project." The Forest Service authorized one side of the road to be a "commercial thin," thinning the forest. The other side of the road was slated to be a clear cut, where most-to-all trees are removed. Which is which?

Above, Adams Camp Wildfire Protection Project (taken 2021) from middle of road. Below left, detail of road off of left side of the above picture (the commercial-thin unit). Below right, detail of the right side of the above picture (the clearcut unit).

Finally, much of what we discuss below includes science and materials that we repeatedly provide to the Forest Service for variously named proposed logging projects. We've also provided many of these materials to the US Department of Agriculture (USDA) when it invited the public to comment on the Department's climate strategy, and again to the Council on Environmental Quality when it reviewed the USDA's unscientific climate strategy.

The Forest Service's management on public lands as it relates to fire can be separated into logging before fire in the name of fire risk reduction, logging as a form of fire suppression during a fire, and post-fire logging project meant only to recover the economic value of timber. The testimony we provide here focuses on logging in the name of fire, what drives fire, and the need to preserve the mature trees that now exist as a climate solution (because that relates directly to wildfire). For the during-fire logging, we refer you to the fireline fact sheet that our organization submitted into the record with several other organizations via email on March 17, 2021. For post-fire logging, we refer you to the testimony submitted by the Pacific Northwest Climate Alliance Wildfire Working group and note that we have several examples on the Nez Perce and Clearwater National Forests where we have documented wildfire where it was low-severity (i.e., the type of fire the Forest Service claims it wants), yet the Forest Service has proposed environmentally damaging post-fire logging. These projects include the Sand Mountain Fire Salvage and the Johnson Creek Fire Salvage logging projects on the Clearwater National Forest.

BEFORE FIRE: SCIENCE DOES NOT SUPPORT THE IDEA THAT WE CAN LOG OUR WAY TO LESS FIRE

Before delving into the fire ecology science, FOC first responds to the loaded word "catastrophic" Congress has chosen to use, because words matter. "Catastrophic" is a holdover term from before a body of science that recognized there is an important role that natural high-severity wildfire plays on wild landscapes. High-severity fire on wildlands begets biodiversity.

Fire has varying levels of severity and exists in different regions differently. Forests in the Northern Rockies, for example, have existed for millennia with mixed-severity fire, which includes stand-replacing fires. Mixed severity fire is important for our public forests in this region, and that includes high-severity fire. Mixed severity fire includes patches of natural high-severity fire in addition to low severity fire and unburned pockets. When FOC says "high-intensity fire," we mean stands with over 75 percent tree mortality. High-severity fire in public forests are ecologically important, too—many species evolved with high-severity fire. *See* Bond et al. 2012, Hanson 2010; and Hutto, "Fire Ecology Stories" https://www.youtube.com/watch?v=5EpTncRMbXs. Snag forest habitat "is one of the most ecologically important and biodiverse forest habitat types in western U.S. conifer forests (Lindenmayer and Franklin 2002, Noss et al. 2006, Hutto 2008)." Hanson 2010. "Many plant and animal species are adapted to post-fire conditions, and populations of some (e.g. many bird species; Figure 1) decline after fire exclusion or post-fire logging." Noss et al. 2006. For example, Hutto 2008 found that the black-backed woodpecker is a specialized species on severely burned forests. Hutto found a distribution of black-backed woodpeckers, which "suggests that conditions created by severe fires probably represent the historical backdrop against which this species evolved." And, "[t]he desire to rid our forests of severe fire beyond the urban interface is, for many forest types, not well grounded in ecological science." Hutto 2008. Please also see LeQuire 2009 and Odion et al. 2014.

Repeatedly, from both sides of the aisle, representatives stated this idea that if we can lessen fire risk if only we correct the "fuels" problem that we have on our public land. Even folks who purported to value science repeated concerning misinformation, such as suggesting that the US Forest Service's 10am policy that began in the 1930s, where the agency aimed fires to be out by 10am the next day, has contributed to fuel build-up, and is creating the current problem of "catastrophic" wildfire. Some acknowledged that the climate change complicated the fuels problem or the forest-condition problem. We heard that from Democratic members, Republican members, the minority witness, and even one of the witnesses by the Democrats, who was an engineer. Really, wildfire is primarily a climate and weather phenomenon, not "fuels problem." Dr. DellaSala was the only scientist on this panel who could speak to that, and his testimony did that well.

Dr. DellaSala spoke to a cool wet climate period in the mid-20th Century that was bookended by a warm, dry climate period in the early 20th Century and warmer, dryer weather that climate change is accountable for now. There is an even anecdotal example in the Northern Rockies that fits this explanation and does not fit the "fuel build-up narrative." The Big Burn of 1910 burned three million acres (three times more than the Dixie Fire in 2021) across primarily northern Idaho and western Montana over August 20-23—three days. What caused this? The spring in the region had record-low precipitation, and the summer was remarkably hot. By August of this dry, hot summer, there were many small blazes, from lighting to locomotives. Amidst this backdrop came a weather front on August 20 that created 70+ mile-per-hour winds and whipped the region into a fire storm. The Great Burn of 1910 happened before the US Forest Service's 10am policy and before the agency's suppression tactics for fire, so there was no "fuel build up" at play here, only the forest as it has existed for millennia. The Great Burn of 1910, that firestorm, helped to inspire the 10am policy that so many attribute to large fires now. For a more recent example, the Marshall Fire outside of Boulder, Colorado was a grassland fire—no fuel build-up there. The feature that the Marshall Fire shared with the Great Burn of 1910 was winds—the Marchall fire had 90mph winds.

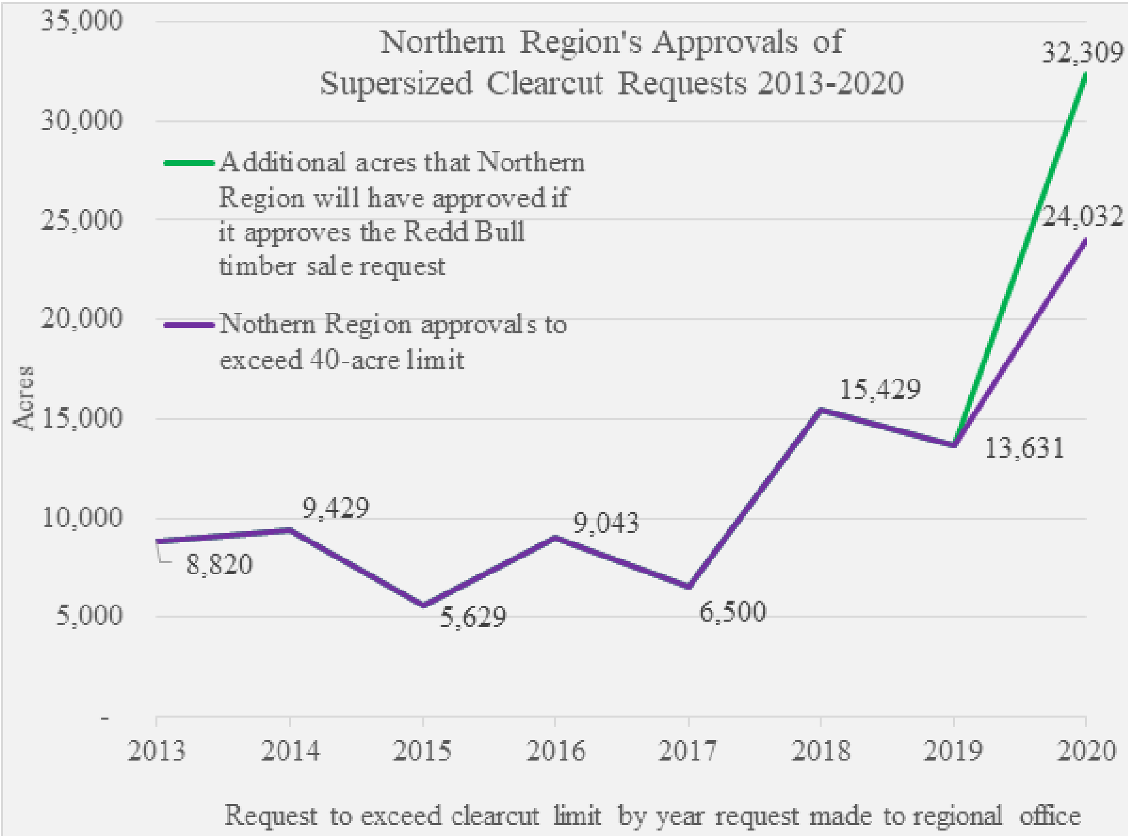
Climate and weather—not fuels—primarily drive fire severity. Global warming is driving the climate and weather that drive the severe wildfires, in part due to more droughts and longer periods of hotter temperatures. *See, e.g.*, Pechony and Shindell 2010; Pierre-Louis and Popovich 2018: Lesmeister 2019. Logging exacerbates the situation driving severe fires because *logging contributes up to three times the carbon emissions that logging purports to save by altering fire behavior*. Campbell et al. 2012.[2] A later study, Harris et al. 2016, found that where some disturbances like insects, disease and fire kill trees and lower carbon sequestration, logging has the greater impact—up to ten times the carbon from forest fires and bark beetles together. **More carbon is lost from logging than from wildfire.** So, contrary to Chief Moore's assertion that fire threatens carbon storage, species habitat, and long-term deforestation, that is actually truer of logging. Logging and contributing to carbon emissions will neither make forests more resilient nor mitigate our contribution to a warming world—logging conversely *contributes* to climate

[2] See also McKinley et al. 2011: "[I]f the starting point is a mature forest with large carbon stocks [], then harvesting this forest and converting it to a young forest will reduce carbon stocks and result in a net increase in atmospheric [CO2] for some time.

change. It is increasingly understood and accepted that reducing fuels does not consistently prevent large fires and does not reduce the outcome of these fires. *See* Lydersen et al. 2014.

Fire severity is not greater where fire has been absent. In a large study across the west, Bradley et al. 2016 found that areas that tend to be more protected had less instances of high-severity wildfire than areas where the Forest Service has "managed" through logging. *See also* Odion et al. 2004. Science suggests that logging tends to exacerbate fire behavior as opposed to an unlogged state. Representative Gibbs referenced seeing a picture of major fires on public land bordered by private land where fire did not impact. This anecdotal observation is countered by scientific analysis, which samples many instances. Dr. DellaSala noted a recent study by Oregon State University that showed that most fires impacting communities have spilled over from private lands that have been logged. There is scientific support for Representative Khanna's statement that clearcutting can put communities at greater risks. Zald & Dunn 2018 found that plantation forestry with young forests and spatially homogenized fuels were more significant in predicting wildfire severity than pre-fire biomass. And this makes more sense. Below are two pictures, taken by FOC staff, from the Nez Perce National Forest approximately 15-20 years after clearcutting swaths of forest:

Clearcutting and regeneration cutting (where most trees are removed and the next generation starts growing at the same time) creates a homogenous forest structure, where all trees are the same height. The branches, which become ladder fuels, are on top of each other. Partly for this reason, intensive regeneration logging can make areas not previous susceptible to high-severity fires more susceptible to them with fuels as a secondary driver when the primary driver (weather and climate) exist. This science suggests that if there is any change in the frequency of fire-severity on the landscape as the secondary driver after weather and climate, it is *likely due to the Forest Service's own forestry practices*. Friends of the Clearwater has found the disturbing practice of supersized clearcuts on the rise in the Northern Region of the Forest Service, where the regional office appears to rubber-stamp every request for an exception to exceed the regulatory 40-acre-limit. Bilodeau and Juel 2021. Below is the chart of supersized clearcuts that Region 1 has approved over the past seven years. These acres only represent clearcuts that are a part of logging units *over* 40 acres in size. This chart does not represent what the Forest Service has authorized for clearcuts *under* 40-acre logging units in the past decade.



Graphic from Bilodeau and Juel, 2021.

"This ceremonial, pro forma request-and-approve routine has impacted national forests of the Northern Region on a large scale. From 2013 until March of 2021, the Northern Region has approved 93,056 acres of supersized clearcuts, about twice the size of the District of Columbia. If the acres were arranged contiguously in a square, a person with an average walking speed of three miles per hour would have to walk two full eight-hour days just to traverse its perimeter." Juel and Bilodeau 2021.

The old and mature forests (that we can protect) play a positive role in countering impacts from high-severity fire. Lesmeister et al. 2019, in looking at fires in southwestern Oregon, mapped northern spotted owl habitat with the 2013 fires in that region. Northern spotted owls are an old-growth-obligate species, meaning they generally only occur in these mature types of forests. Lesmeister et al. 2019 found that the areas of forests that had high habitat suitability for northern spotted owls burned more often at low or moderate severity, while the forests that have been logged were more likely to burn at moderate to high severity. Bradley et al. 2016 had similar findings—protected areas, i.e., older forests, were less likely to burn at high severity. Protecting older, mature forests not only provide increased carbon sequestration, but offer a buffer to high-severity fire. Some even may serve as fire refugia, which are areas disturbed less frequently or severely by wildfire—these areas provide safe havens for wildlife during a fire, and help post-fire recovery of surrounding areas by providing the seed for new vegetation. Meddens et al. 2018. These areas are not always predictable (just as weather sometimes surprises us), and some are created by happenstance. "Treating" areas by logging or prescribed burning can eliminate what would have been natural fire refugia. So again, contrary to what Chief Moore represented, protecting old mature forests are the solution, not a problem to be logged. The science suggests that logging will not "restore" healthy, resilient, fire-adapted forests; forests are better adapted for fire before the Forest Service starts cutting down trees. Cutting down trees will also not protect communities.

Thinning is not the solution for high- or low-severity fire. Representative Khanna acknowledged that "some thinning is necessary" based on science. To what science was he referring? Moore also mentioned in the hearing was that "thinning and treatment works, case after case," citing the Eldorado National Forest in the Tahoe Lake management unit. The whole picture, which is what science examines, tells another story, according to our research. Because weather and climate are the primary drivers of fire severity, thinning will not impact fire weather. For example, the fire that tragically hit Paradise, California, burned through surrounding areas that had been treated and thinned for years. https://www.latimes.com/projects/wildfire-california-fuel-breaks-newsom-paradise/. However, **fuel treatments are unlikely to be effective for low to mid-severity fire as well**. Rhodes and Baker studied fire records and found that, over the 20-year period that fuel reduction is assumed to be effective, approximate 2.0-4.2% of untreated areas would be expected to burn at high or high-moderate severity. Rhodes and Baker 2008. **So, there is over a 90 percent chance that fuel reduction will not influence a fire's behavior.** This, considered with the science above, renders the Forest Service's and many committee members' assumption that logging can satisfy the fuel-reduction purpose and need or that logging won't make a fire risk worse, at best, relies on controversial science.[3]

Protecting people and structures from any wildfire starts with smart zoning and continues with defensible space where it matters the most: right around the house. Dr. Jack Cohen found that the measures taken within the first 130 feet of the house to reduce home ignitability have the most influence on whether a home is lost in a subsequent wildfire. Home ignitions depend upon whether the structure is built with fire-resistant materials and whether there are flammable items on or around the structure: "Because home ignitions depend on home ignitability, the behavior of wildland fires beyond the home or community site does not necessarily correspond to W-UI home loss potential. Homes with low ignitability can survive high-intensity wildland fires, whereas highly ignitable homes can be destroyed during lower-intensity fires." Cohen 2000. **Because home ignitability drive whether a structure is lost, the "fire loss problem can be defined as a home ignitibility issue largely independent of wildland fuel management issues.**" Cohen 2000 (emphasis added). The issue is how people manage the first 130 feet surrounding their structure, not how the US Forest Service manages public lands miles away from houses that are highly ignitable. Because of this science, we agree

[3] We note here that our organization repeatedly gives the Forest Service this science when invited to comment on logging projects to "reduce fuels," and the Forest Service never engages with this science. The Forest Service manages to push through many projects under National Environmental Policy Act tracks that don't require answering comments like ours or engaging with this independent science.

with Dr. DellaSala that there should be more money allocated to defensible space than logging the wildlands.

Beyond that, preparedness planning such as evacuation routes can help. And zoning can help risky development in wildland-urban interfaces. Representative Norman seemed to take issue with providing funds so individuals may “harden homes,” which could help protect property. Idaho County, Idaho, is deeply Republican, and we understand that it has resisted zoning, which could require people building in the middle of the forest to follow firewise landscaping for their own protection. While Representative Norman took issue with spending any money to help homeowners, he seemed unconcerned with subsidizing private industry (either logging or grazing), who only profit with great help from taxpayers.

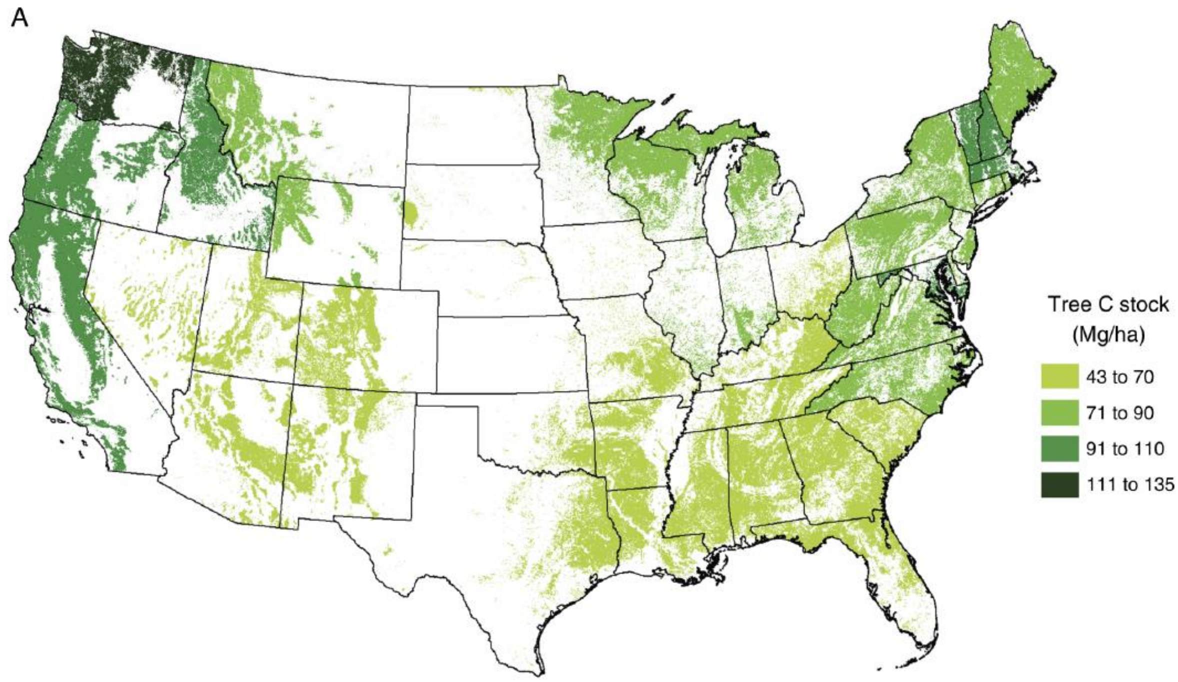
Many of the solutions to protect people from wildfire, like zoning, evacuation routes, and reducing the ignitability of the house and its surrounding 130 feet, simply don’t directly involve the US Forest Service. But, the federal government could provide education. One additional strategy the federal government might consider, however, is acquiring private land next to national forests in places the government believes are risky to inhabit because of wildfire. A Friends of the Clearwater staff member was out on a field trip where the Forest Service was showing a recent “fuels reduction” project, where the Forest Service approved logging in an inventoried roadless area in the middle of the forest to ostensibly “reduce fuels” next to about 20 private structures surrounded by national forest. In the environmental assessment of the project, the Forest Service noted that most of those structures were summer homes, and these private lands were mostly inholdings surrounded by national forest on all sides. There were more than several structures that hadn’t managed the flammability of their home or materials within the first 130 feet of the home in accordance with Cohen’s research (paragraph above). On this field trip, the district ranger said it would have been cheaper to buy out the landowners than cost of doing the project in the first place. Perhaps buying out the private inholdings in national forests is both the safest and most economical thing to do.

OPPORTUNITIES TO ADDRESS WILDFIRE AND CLIMATE CHANGE BASED ON CURRENT CONDITIONS

Protecting our forests—which include drastically reducing logging and roadbuilding, and retiring grazing on public lands (two taxpayer-subsidized environmentally destructive activities), will cheaply and easily contribute to the Administration’s commitment to halving greenhouse gas emissions by 2030.

Forests are carbon sinks

Trees sequester carbon continually throughout their lives. While live trees store that carbon, dead trees also store carbon. And this carbon storage exists throughout forested areas in the United States.



Above: McKinley et al. 2011. "Average statewide forest carbon stocks [in Megagrams of Carbon per hectare] in live and dead trees in the conterminous United States." While the dark green represents the greatest carbon stocks and gain, note how much carbon storage and carbon stocks of forests in the entire United States, when added together, can contribute. All forest lands have the potential to mitigate for global warming in various regions across the United States in both the soils and the vegetation.

While forest lands are carbon sinks, more intact forest lands can be more efficient carbon sinks. For example, larger trees more efficiently store carbon. All parts of the tree—the trunk, the bark, the branches, the leaves or needles, and the roots, is biomass. And scientists have found that the largest one percent of trees in mature and older forests comprised 50 percent of forest biomass worldwide. Lutz, J.A. et al. 2018. Furthermore, larger trees of a species accumulate more carbon on a rate greater than their younger and smaller counterparts; in one year, a large tree species can store carbon equal to a mid-sized tree. Stephenson, N.L. et al. 2014. Large-diameter trees store outsized amounts of above-ground carbon when compared to other trees because the growing up, so to speak, is largely done: "Once trees attain large stature, each additional [diameter at breast height] 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 [diameter at breast height] and [above-ground carbon] results in significant gains." Mildrexler et al. 2020. This potential is impressive: in eastern Oregon, for example scientists found that, while large trees were only three percent of what they inventoried, those same trees stored forty-two percent of the above-ground carbon in the areas inventoried. Mildrexler et al. 2020. While all forests have biomass, Pacific Northwest forests can hold live tree biomass equivalent to or larger than tropical forests. Law and Waring 2015. But all forests store carbon, and we need all national forests involved in climate-change mitigation.

Trees are not the only component that stores carbon in forests. In addition to the living biomass that stores carbon, soils, meadows, and dead trees all store carbon as well. Behind living biomass, soils are the next remarkable carbon sinks in the forest. Pan et al. 2011. And soils are more insulated from the weather extremes that can impact above-ground biomass. Achat et al. 2015. Dead trees not removed from a forest also store carbon, McKinley et al. 2011, emitting it on a more favorable time-delay than human activities that more immediately launch carbon into the atmosphere (discussed below).

Finally, even mountain meadows have the potential to be a carbon sink. Researchers at the University of Nevada Reno found that wet montane meadows, particularly the plants that grow in wetlands and the dense roots that accompany those plants, removed carbon from the atmosphere at a rate comparable to tropical rain forests. They stored carbon in the ground, which again can be less vulnerable to natural ecosystem disturbances. *See* Reed et al. 2020; Wharton 2020.

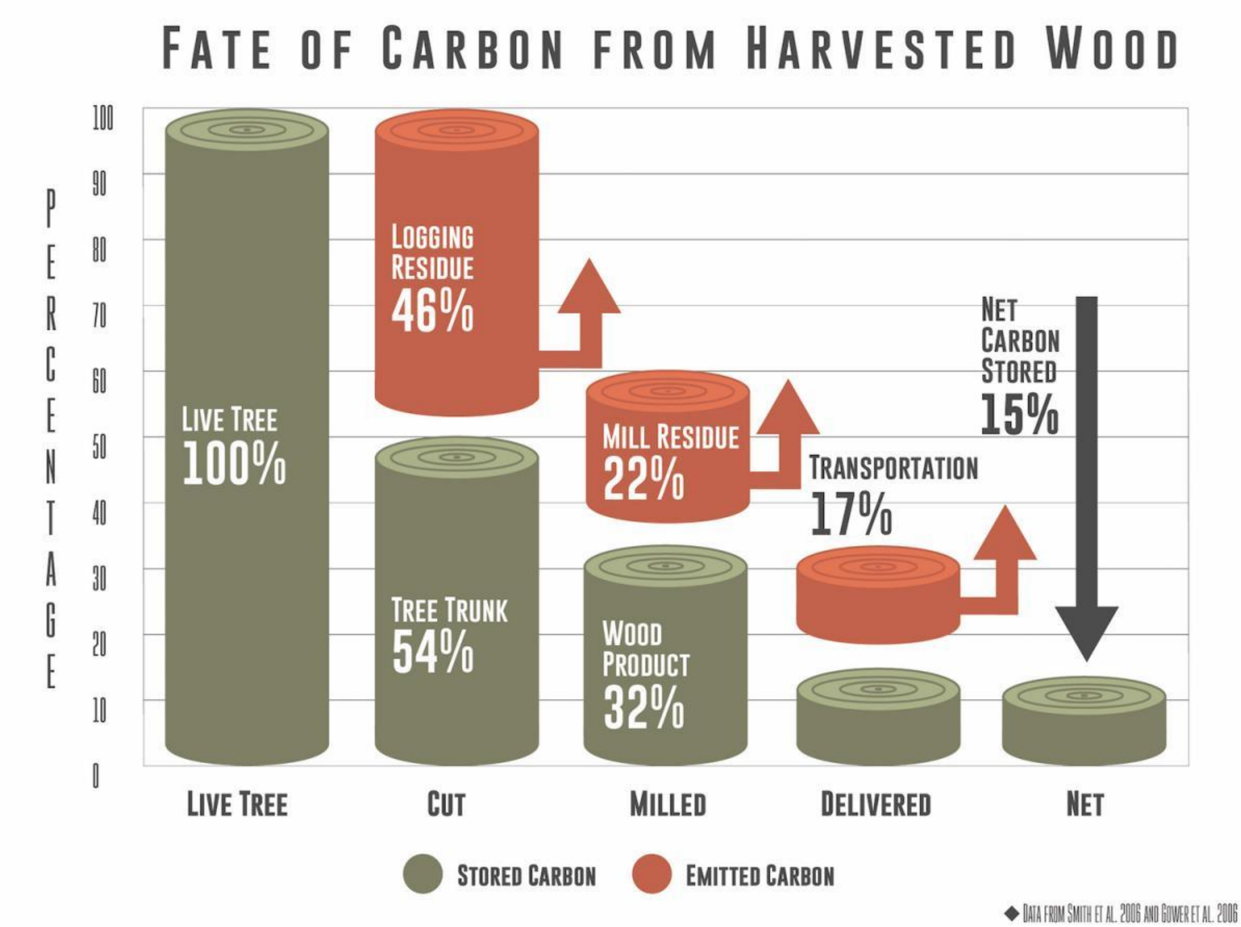
While standing trees, dead trees, soil, and meadows can store carbon, disrupting these areas with active management, including logging, roadbuilding, and grazing, can do just the opposite and emit carbon, contributing to climate change.

Cutting down trees, removing dead wood, and disturbing soils reduce carbon sequestration and also emit carbon

Climate science suggests that cutting down trees and manipulating forest stands does not benefit the climate. Instead, cutting trees and manipulating vegetation by killing and removing it decreases carbon sequestered, decreases carbon stored, and increases carbon emitted.

Carbon is lost to the atmosphere several different ways from harvesting wood. First, cutting down trees reduces a forest’s potential to sequester carbon from the atmosphere. If living trees continually store carbon through the process of sequestration, then it logically follows that killing and removing each tree arrests each tree’s sequestration process, resulting in a net reduction how much carbon a forest sequesters. Even planting new trees cannot fully replace lost sequestration: “[I]f the starting point is a mature forest with large carbon stocks (Cooper 1983, Harmon et al. 1990), then harvesting this forest and converting it to a young forest will reduce carbon stocks and result in a net increase in atmospheric [CO2] for some time (Fig. 8B; Harmon and Marks 2002).” McKinley et al. 2011. Planting replacement trees cannot fully replace the lost carbon sequestration because mature forests with larger trees sequester more carbon than newly planted seedlings. Cutting down trees not only reduces sequestration, but reduces carbon storage.

Not only does cutting down trees reduce the rate of carbon sequestration, but *harvesting wood actively emits carbon*. Disturbing soil, including road construction to logging units and soil disturbance within those units by wheels of machinery and dragging felled trees to where they can be loaded, releases the carbon that soil held into the atmosphere. *See* Pan et al. 2011; Achat et al. 2015. While 100 percent of standing trees store carbon, processing wood does not have this same efficiency.



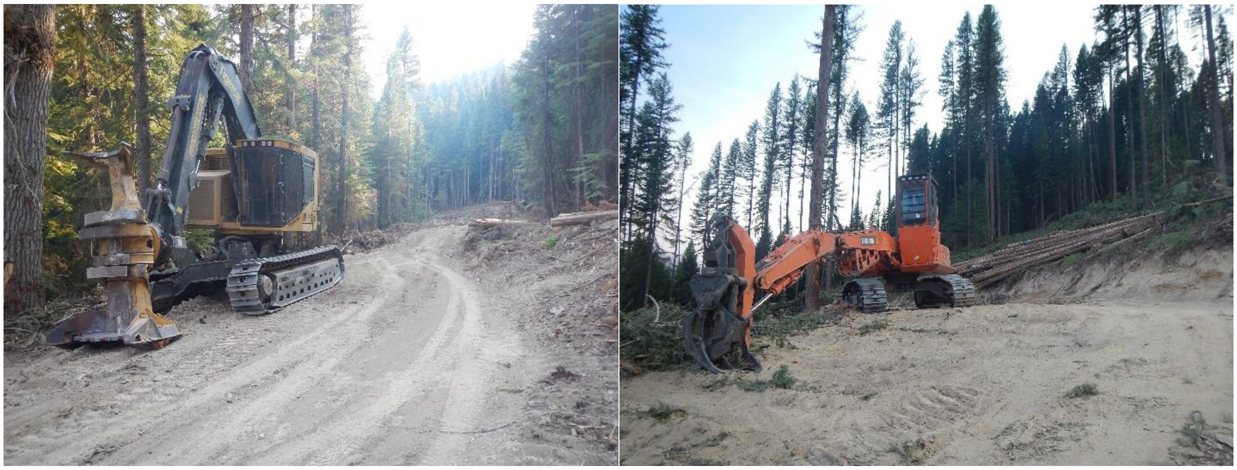
Above: Josephine County Democrats, "Forest Defense is Climate Defense," at https://josephinedemocrats.org/forest-defense-is-climate-defense/ (last visited Apr. 5, 2021), based on data from Gower 2003 and Smith et al. 2006.

Harris et al. 2016 had higher estimates than the above chart: "[Sixty-four] percent of these losses were from logging residues [both above (19%) and below-ground (23%) and mill residues (22%). "The actual carbon stored long-term in harvested wood products represents less than 10 percent of that originally stored in standing trees or biomass." Moomaw and Smith 2017.

Photo: Example of logging residue. Piles are burned when logging operations are complete. Nez Perce-Clearwater National Forests, courtesy of Friends of the Clearwater.

To calculate carbon emissions from logging residue, Harris et al. 2016 used mill surveys, so these concerning percentages do not appear to account for the fossil fuels burned for the power to process the wood.

Logging operations burn fossil fuels. Cutting down trees, dragging logs to trucks, and hauling those logs to mills burn fossil fuels. Below are some examples of the machinery that burn fossil fuels, and all of them are generally involved with logging.



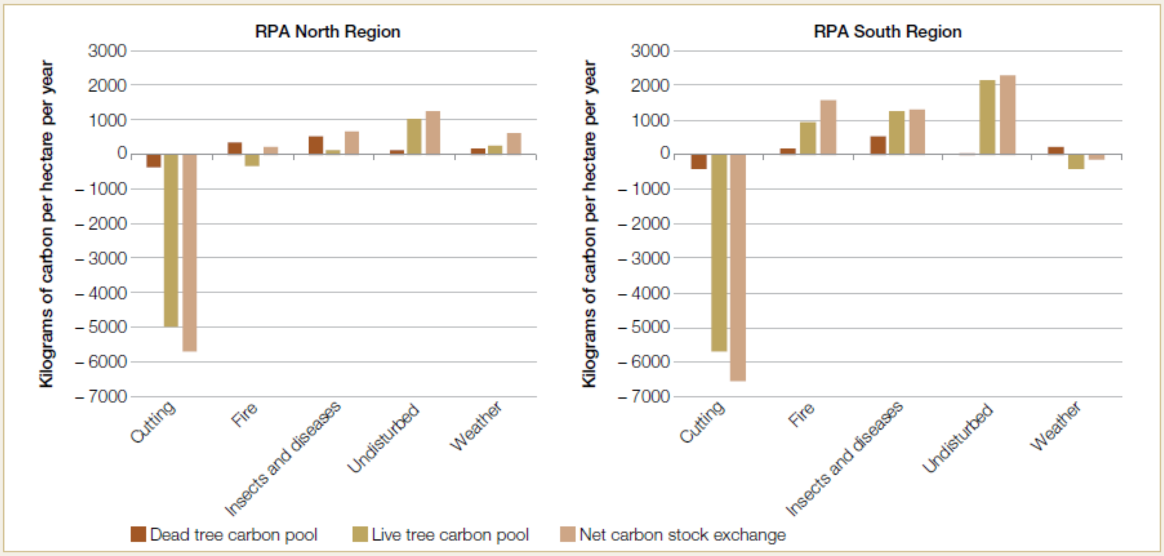
Upper left: feller-buncher; Upper right: swing machine. Photos courtesy of Friends of the Clearwater. This machinery burns fossil fuels and need to be transported to logging sites by trucks that also burn fossil fuels.

Logging truck. Photo Courtesy US Forest Service. https://www.fs.usda.gov/. Logging trucks can haul anywhere from three thousand to six thousand board feet of timber. In 2020, the Nez Perce-Clearwater sold over eighty-four million board feet of timber. It will take over 14,000 truckloads to haul away the timber sold in 2020 once it is cut down.

The most carbon lost in the public's forests is from logging. Even logging to purportedly "reduce fuel" (a strategy largely debunked by science and discussed in further detail below) can emit more carbon than what logging purports to save by altering fire behavior. Harris et a. 2016. And the true emissions associated with logging are sometimes underestimated and not accurately accounted for. *See* Hudiburg et al. 2019.

Even for the emissions we have accounted for, the most carbon lost from forests in the United States is from logging:

Figure 8-4. Carbon accumulation rates (kilogram per hectare per year) resulting from disturbances in the Eastern United States, based on the most recent remeasured Forest Inventory and Analysis data (about a 6-year time step).



Above: USDA Forest Service 2016. Future of America's Forests and Rangelands. Update to the Forest Service 2010 Resources Planning Act Assessment. "Figure 8-4. Carbon accumulation rates (kilogram per hectare per year) resulting from disturbances in the Eastern United States, based on the most recent remeasured Forest Inventory and Analysis data (about a 6-year time step).

Eastern forests of the US are not the only net sinks. Western US forests are, too.

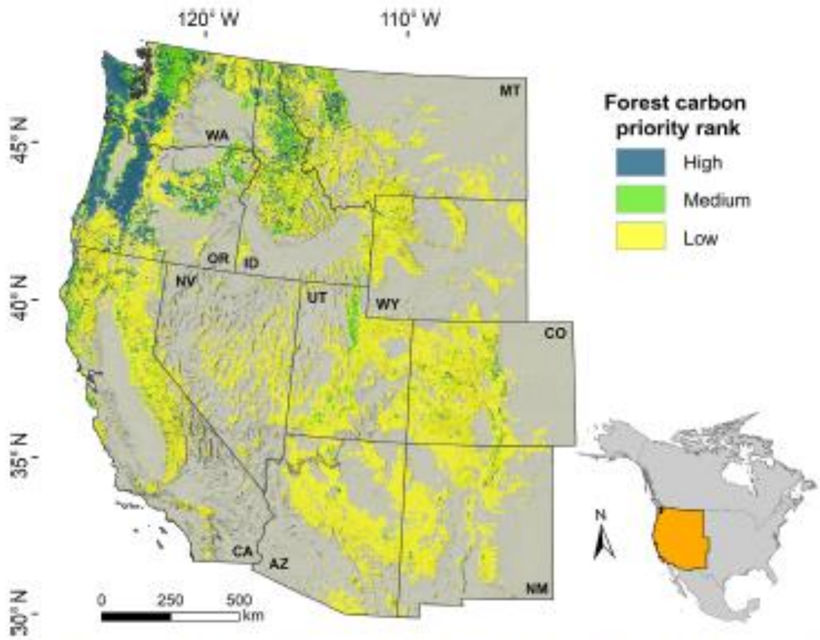


Figure 1, Buotte et al. 2019. While this figure ranks carbon priority, we point out that all of these forests are carbon sinks. Also, this map does not include Alaskan forests.

The sequestration potential is national.

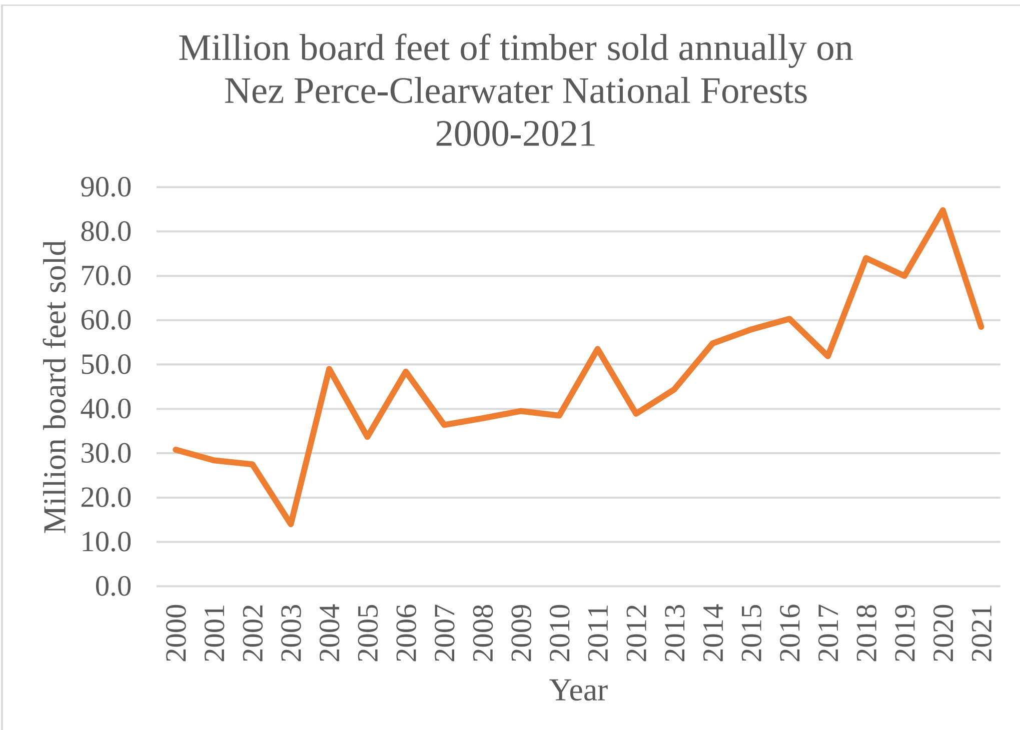
In addition to logging, disturbing other ecosystems with activities like grazing can also emit carbon. For example, the same researchers who found mountain meadows as a potential carbon sink found that disturbing those meadows can be a carbon source. Human activity, like grazing livestock on public lands, has transformed some wet meadows to drier soils with sparser grasses and shrubs, which transforms these same areas to potential sources of carbon emission. *See* Reed et al. 2020; Wharton 2020.

Other benefits of protecting national forests instead of cutting down trees

Intact forests provide benefits to wildlife and biodiversity as well as sequestering carbon for us. Structural diversity begets biodiversity. *See* Moomaw et al. 2019, Buotte et al. 2019. Forest canopies in general can promote cooler microclimates, which buffer warming environments for other living organisms, providing a climate refugia. Intact forests, after a fire, also contain swaths of fire refugia, which are areas that fires miss and provide refugia for animals as well as a source of seed for vegetative regrowth after fire. Meddens et al. 2018. Older forests, including old growth, which is the product of hundreds of years of ecosystem work, are among those cooler microclimates. *See* Davis et al. 2019; Frey et al. 2016. If we see forests as more than just out tree crop to chop, these public lands can provide excellent habitat, coast to coast, for species that are struggling in the face of ever-expanding human development and increasing temperatures. *See* Buotte et al. 2019; Moomaw et al. 2019.

Opportunities to combat climate change through protecting forests

We have an opportunity on our public lands to combat climate change by protecting our forests and drastically reducing logging, and completely eliminating the felling of large trees. Only a small fraction of mature, older areas of national forests is left intact, and the US Forest Service is still logging irresponsible amounts. The US was the forefront timber products in the Obama Administration, Prestemon 2015, and logging levels exploded in our mission area in the administration. In the face of the science that discusses the carbon that large trees sequester and what logging emits, the US Forest Service has *increased* logging. For example, Friends of the Clearwater has noticed this upward creep of timber sales on the Nez Perce and Clearwater National Forests for over a decade now.



Above is a chart of the timber sold off the Nez Perce-Clearwater National Forests. The information is from US Forest Service Region 1's annual reports on what this agency has sold based on logging projects on the Nez Perce and Clearwater National Forests. The units are "thousand board feet," so 84,000 thousand board feet = 84,000,000 board feet = 84 million board feet sold in 2020. ***In five of the past six years, the Forest Service managing this national forest has sold more of the public's trees than any other year since 2000***.[4] For reference, logging trucks on the road can haul about 3,000-6,000 board feet of timber.[5] That means 84 million board feet of timber is equivalent to 14,000 to 28,000 trucks of logs coming off just these two forests for what was sold in 2020.

[4] The Nez Perce and Clearwater National Forests are undergoing a land-management-plan revision, and three of the four action alternatives propose *increasing* logging outputs by at least 50 percent annually from the 84 million board feet sold last year.

[5] A "board foot" of timber is the volume equivalent to one-foot by one-foot of wood that is one inch thick.

The previous administration emphasized increasing logging, so Forest Service's increase in logging projects is not unique to this forest. And on forests like the Nez Perce-Clearwater National Forests, the Forest Service has continued to take bites of out of old growth and mature forests in just about every logging project it approves. Every time the Forest Service "resets" an area that is mature forest or would have become old growth, it eliminates the benefits described above for the rest of the lives of folks living. Living and future generations cannot afford this trend. Where Depro et al. 2008 considers that variations in business-as-usual to increase harvest to previous levels can lead to increasing carbon emissions at least 50 percent, the scientists found that eliminating harvest could tip the carbon balance 50 percent in the other direction: absorbing more carbon from the atmosphere.

Scientists have discussed the opportunities of mitigating global warming and preserving biodiversity by protecting our forests. "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." Law et al., 2018. Scientists have proposed proforestation, which optimizes the trees that currently exists and allows them to grow intact with their natural ecosystems as opposed to disturbing them by logging: "Growing existing forests to their biological carbon sequestration potential optimizes [carbon dioxide removal] while limiting climate change and protecting biodiversity, air, land, and water. Natural forests are by far the most effective." Moomaw et al. 2019 (internal citations omitted). We must stop logging what exists and start protecting it: "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." Mildrexler 2020. Many of these ideas are captured in the Law and Moomaw 2021 article titled, "Keeping trees in the ground where they are already growing is an effective low-tech way to slow climate change."

The present situation reveals the value the US Forest Service still places on logging and industrial exploitation, to the detriment of carbon storage, carbon sequestration, and our future. The current state of things is contributing to the climate crises.

Congress must act immediately to mitigate climate, which is what will mitigate climate change (the primary driver of fire). This action is leaving trees in the ground on public wildlands, and focusing making communities firewise. Anybody who thinks we can *prevent* wildfires are misinformed. But, we can live with them. That involves protecting the carbon sequestering and storage functions that forests already provide us in addition to community preparedness.

We fully agree with Ms. Carole King that Congress should pass the Northern Rockies Ecosystem Protection Act, which will protect ecosystems in the Northern Rockies. Beyond that, however, Congress has before it the Roadless Area Conservation Act. There are current amendments that must happen for H.R. 279 to protect roadless areas, as Friends of the Clearwater has found the Forest Service is currently exploiting the logging exceptions in the roadless rules (in Idaho and Montana at least) to cut trees in roadless areas. *See* Bilodeau and Macfarlane 2020, available at https://www.friendsoftheclearwater.org/the-roadless-report-analyzing-the-impacts-of-two-roadless-rules-on-forested-wildlands/.

The Forest Service's 10-year plan to treat 20 million acres is very concerning, given what the best independent science suggests about climate change as it pertains to fire and forests, as well as carbon sequestration and storage, which logging undermines. The 10-year plan is also very concerning given how we've seen projects implemented in North-central Idaho on the Nez Perce and Clearwater National Forests.

Data-driven policy decisions are inherently rational. Beyond asking Congress generally to act to protect public lands, we specifically ask the Oversight Committee to continue this investigation into Forest Service practices and look for better information, by peer-reviewed science, about the interplay between forests, climate change mitigation, and fire ecology. We ask you to invite independent scientists and economists to testify on some of the issues identified in what we've submitted. We also ask the Oversight Committee to look into fire suppression tactics in accordance with what several environmental organizations have observed in four western states with the practice of bulldozing ecologically destructive and costly firelines where they will not benefit communities. Finally, we ask the Oversight Committee to investigate pre- and post-fire logging practices, which include inviting nonprofits on the ground monitoring the Forest Service's management of our public lands. Logging wastes taxpayer money and worsens our climate crisis at the same time; the Forest Service is disserving the taxpaying public and covering that disservice with euphemistic terms about practices that are grounded in a get-out-the-cut mentality that the agency has espoused for decades.

Our Earth and future generations depend upon rational policy directions, and time is running out.

Thank you for allowing us to submit testimony, and thank you in advance for considering it.

Regards,

[redacted]
Friends of the Clearwater
[redacted]

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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:

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

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]:

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

where p is the annual probability of fire of a specific severity, F is total area burned at any severity within the analysis area over the duration of the data record, 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 fraction 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-

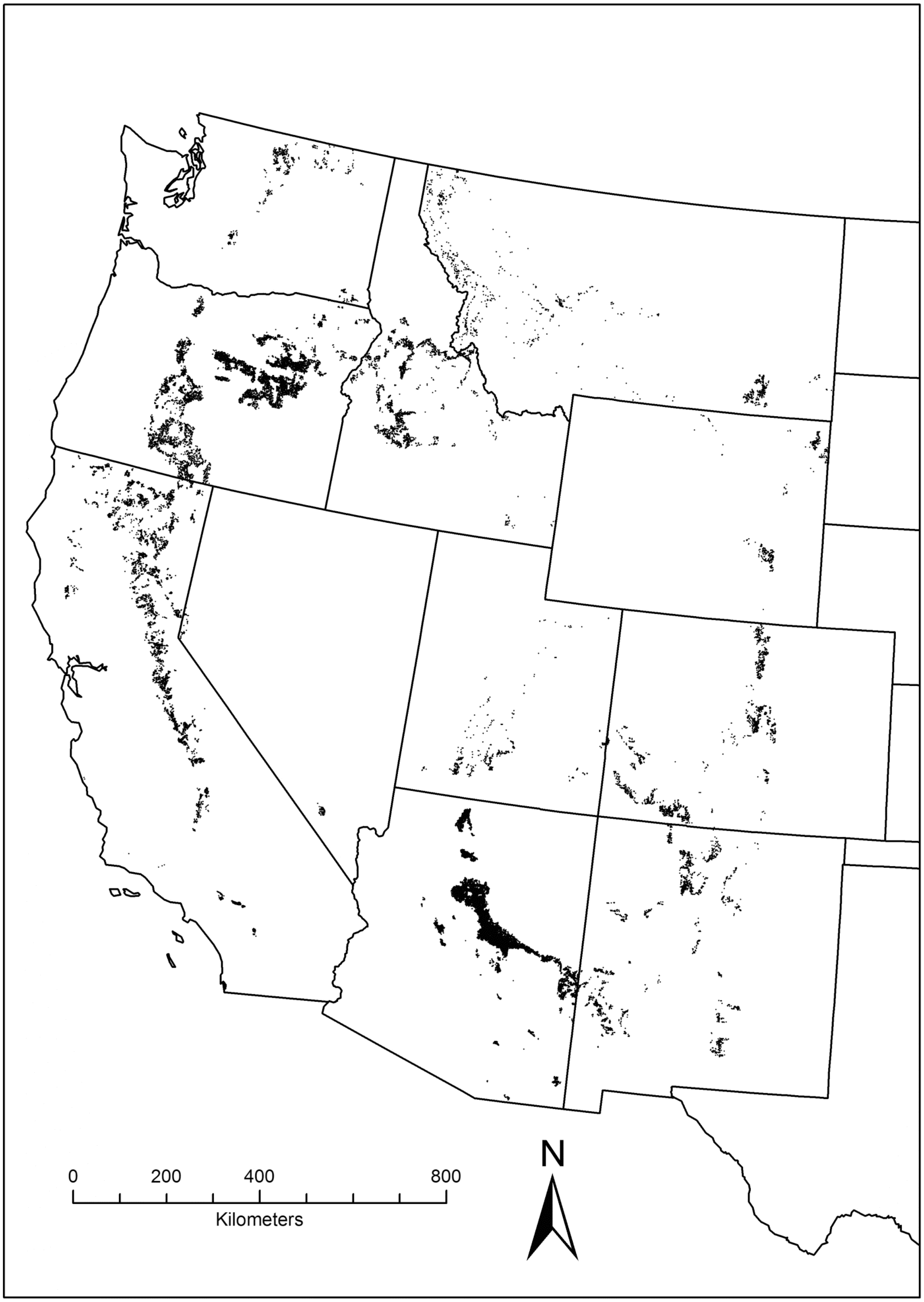

Fig. (1). Ponderosa pine forest fires (n = 40,389) in the western United States from 1980-2003. This is the dataset used in the regional analysis.

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:

$$B = 1/p \tag{3}$$

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 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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ECOSPHERE

Does increased forest protection correspond to higher fire severity in frequent-fire forests of the western United States?

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Abstract. There is a widespread view among land managers and others that the protected status of many forestlands in the western United States corresponds with higher fire severity levels due to historical restrictions on logging that contribute to greater amounts of biomass and fuel loading in less intensively managed areas, particularly after decades of fire suppression. This view has led to recent proposals—both administrative and legislative—to reduce or eliminate forest protections and increase some forms of logging based on the belief that restrictions on active management have increased fire severity. We investigated the relationship between protected status and fire severity using the Random Forests algorithm applied to 1500 fires affecting 9.5 million hectares between 1984 and 2014 in pine (*Pinus ponderosa*, *Pinus jeffreyi*) and mixed-conifer forests of western United States, accounting for key topographic and climate variables. We found forests with higher levels of protection had lower severity values even though they are generally identified as having the highest overall levels of biomass and fuel loading. Our results suggest a need to reconsider current overly simplistic assumptions about the relationship between forest protection and fire severity in fire management and policy.

Key words: biodiversity; climate; fire frequency; fire severity; fire suppression; Gap Analysis Program levels; logging; protected areas.

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Introduction

It is a widely held assumption among federal land management agencies and others that a lack of active forest management of some federal forestlands—especially within relatively frequent-fire forest types such as ponderosa pine (*Pinus ponderosa*) and mixed conifers—is associated with higher levels of fire severity when wildland fires occur (USDA Forest Service 2004, 2014, 2015, 2016). This prevailing forest/fire management hypothesis assumes that forests with higher levels of protection, and therefore less logging, will burn more intensely due to higher fuel loads and forest density. Recommendations have been made to increase logging as fuel reduction and decrease forest protections before wildland fire can be more extensively reintroduced on the landscape after decades of fire suppression (USDA Forest Service 2004, 2014, 2015, 2016). The concern follows that, in the absence of such a shift in forest management, fires are burning too severely and may adversely affect forest resilience (North et al. 2009, 2015, Stephens et al. 2013, 2015, Hessburg 2016). Nearly every fire season, the United States Congress introduces forest management legislation based on this view and aimed at increasing mechanical fuel treatments via intensive logging and weakened forest protections.

However, the fundamental premise for this fire management strategy has not been rigorously tested across broad regions. We broadly assessed the influence of forest protection levels on fire severity in pine and mixed-conifer forests of the western United States with relatively frequent-fire regimes to test this assumption. We used vegetation burn severity data from all fires >405 ha over a three-decade period, 1984–2014, in forests with varying levels of protection.

Study area

Pine and mixed-conifer forests at low/mid-elevations, where historical fires were relatively frequent, are broadly distributed across several ecoregions in the western United States (Fig. 1; Appendix S1: Table S1). Although ponderosa pine often dominates these forests, they can also include Jeffrey pine (*Pinus jeffreyi*), which in places intermix with, and are similar to, ponderosa pine forests, and Madrean pine–oak (*Quercus* spp.) forests with a diversity of pines. Mixed-conifer forests at low/mid-elevations are also broadly distributed across multiple ecoregions (Fig. 1). They can include additional pines (e.g., lodgepole pine, *Pinus contorta*; sugar pine, *Pinus lambertiana*), true firs (*Abies* spp.), Douglas-fir (*Pseudotsuga menzeisii*), and incense-cedar (*Calocedrus decurrens*).

Methods

We used Gap Analysis Program (GAP) protection classes (USGS 2012), as described below, to determine whether areas with the most protection (i.e., GAP1 and GAP2) had a tendency to burn more severely than areas where intensive management is allowed (i.e., GAP3 and GAP4). We compared satellite-derived burn severity data for 1500 fires affecting 9.5 million hectares from years for which there were available data (1984–2014) among four different forest protection levels (Fig. 1), accounting for variation in topography and climate. We analyzed fires within relatively frequent-fire forest types comprised of pine and mixed-conifer forests mainly because these are the predominant forest types at low to mid-elevations in the western United States, there is a large data set on fire occurrence, and they have been a major concern of land managers for some time due to decades of fire suppression. We defined geographic extent of forest types from the Biophysical Settings data set (BpS) (Rollins 2009; *public communication*, http://www.landfire.gov) that derived forest maps from satellite imagery and represents plant communities based on NatureServe's Ecological Systems classification. Baker (2015) noted that some previous work found ~65% classification accuracy of this system with regard to specific forest types and, accordingly, he analyzed groups of related forest types in order to improve accuracy. We followed his approach (see Appendix S1: Table S1). The categories selected from the Biophysical Settings map were ponderosa/Jeffrey pine and mixed-conifer forest types with relatively frequent-fire regimes (e.g., Swetnam and Baisan 1996, Taylor and Skinner 1998, Schoennagel et al. 2004, Stephens and Collins 2004, Sherriff et al. 2014), compared to other forest types with different fire regimes such as high-elevation forests and many coastal forests not studied herein. Forest types in our study totaled 29.2 million hectares (Fig. 1; Appendix S1: Table S1). We used the BpS data to capture areas that were classified as forests before fire, because postfire vegetation maps can potentially show these same areas as temporarily changed to other vegetation types. We sampled our response and predictor variables on an evenly spaced 90 × 90 m grid within these forest types using ArcMap 10.3 (ESRI 2014). This created a data set of 5,580,435 independent observations from which we drew our random samples to create our models. The 90-m spacing was chosen because it was the smallest spacing of points that was computationally practical with which to operate.

Fires

The Monitoring Trends in Burn Severity project (MTBS, *public communication*, http://www.mtbs.gov) is a U.S. Department of Interior and Department of Agriculture-sponsored program that has compiled burn severity data from satellite imagery, which became available in 1984, for fires >405 ha, and was current up to 2014 (Eidenshink et al. 2007). The MTBS Web site allows bulk download of spatial products that include two closely related indices of burn severity: differenced normalized burn ratio (dNBR) (Key and Benson 2006) and relative differenced normalized burn ratio (RdNBR) (Miller and Thode 2007). Both indices are calculated from Landsat TM and ETM satellite imagery of reflected light from the earth's surface at infrared wavelengths from before and after fire to

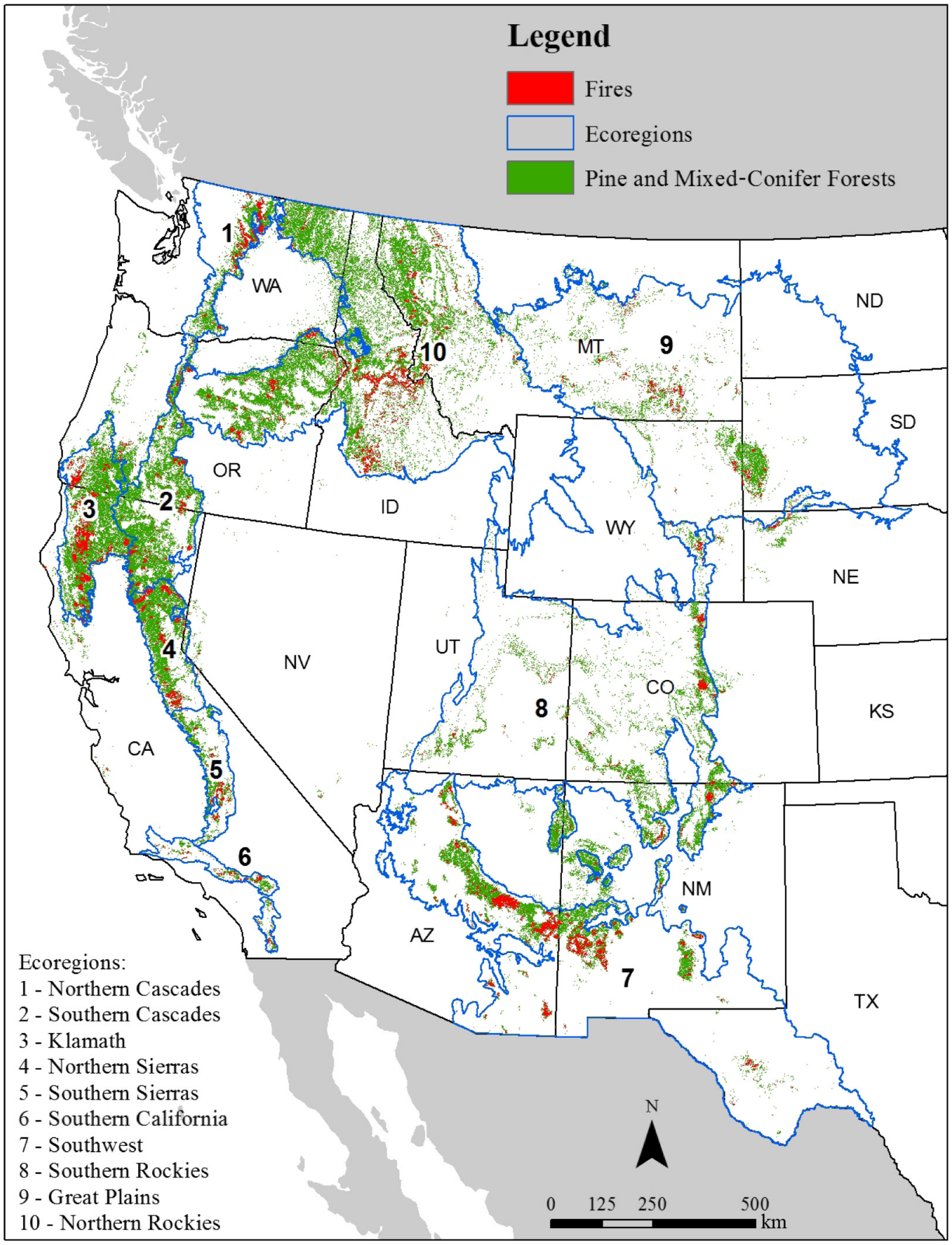

Fig. 1. Pine and mixed-conifer forests, fires, and ecoregions analyzed in this study.

measure associated changes in vegetation cover and soil characteristics. We defined burn severity with the RdNBR index because it adjusts for prefire conditions at each pixel and provides a more consistent measure of burn severity than dNBR when studying broad geographic regions with many different vegetation types (Miller et al. 2009*a*, Norton et al. 2009). RdNBR values typically range from negative 500 to 1500 with values further away from zero representing greater change from prefire conditions. Negative values represent vegetation growth and positive values increasing levels of overstory vegetation mortality. The RdNBR values could be used to classify fires into discrete burn severity classes of low, medium, and high but this was not performed in our study, as we desired to have a continuous response variable in our models.

We intersected forest sampling points with fire perimeters downloaded from MTBS to determine fires that occurred in our analysis area, and censored fires with <100 sampling points (81 ha). The remaining points represented sampling locations from 2069 fires (Fig. 1). We extracted RdNBR values at each sampling point as our response variable as well as predictor variables that included topography, geography, climate, and GAP status. These sampling points were used to investigate the relationship between forest protection levels and burn severity (Appendix S1: Tables S2 and S3). We chose topographic and climatic variables based on previous studies that quantified the relationship between burn severity, topography, and climate (Dillon et al. 2011, Kane et al. 2015).

Topographic and climatic data

To account for the effects of topographic and climatic variability, we derived several topographic indices (Appendix S1: Table S2) from seamless elevation data (*public communication*, http://www.landfire.gov/topographic.php) downscaled to 90-m^2 spatial resolution due to computational limits when intersecting sampling points. These indices capture categories of topography, including percentage slope, surface complexity, slope position, and several temperature and moisture metrics derived from aspect and slope position. We used the Geomorphometry and Gradient Metrics Toolbox version 2.0 (*public communication*, http://evansmurphy.wix.com/evansspatial) to compute these metrics. We also computed several temperature and precipitation variables (Appendix S1: Table S3) by downloading climatic conditions for each month from 1984 to 2014 from the PRISM climate group (*public communication*, http://prism.oregonstate.edu). Climate grids record precipitation and minimum, mean, and maximum temperature at a 4-km grid scale created by interpolating data from over 10,000 weather stations. To determine the departure from average conditions, we subtracted each climate grid by its 30-yr mean monthly value. These "30-yr Normals" data sets were also downloaded from the PRISM Web site and reflected the mean values from the most recent full decades (1981–2010). We determined mean seasonal values with summer defined as the mean of July, August, and September of the year before a given fire; fall being the mean of October, November, and December of the previous year; winter the mean of January, February, and March of the current year of a given fire; and spring the mean of April, May, and June of the current year.

Protected area status and ecoregion classification

We used the Protected Areas Database of the United States (PAD-US; USGS 2012) to determine forest protection status, which is the U.S. official inventory of protected open space. The PAD-US includes all federal and most State conservation lands and classifies these areas with a GAP ranking code (see map at: http://gis1.usgs.gov/csas/gap/viewer/padus/Map.aspx). The GAP status code (herein referred to interchangeably as GAP class or protection status) is a metric of management to conserve biodiversity with four relative categories. GAP1 is protected lands managed for biodiversity where disturbance events (e.g., fires) are generally allowed to proceed naturally. These lands include national parks, wilderness areas, and national wildlife refuges. GAP2 is protected lands managed for biodiversity where disturbance events are often suppressed. They include state parks and national monuments, as well as a small number of wilderness areas and national parks with different management from GAP1. GAP3 is lands managed for multiple uses and are subjected to logging. Most of these areas consist of non-wilderness USDA Forest Service and U.S. Department of Interior Bureau of Land Management lands as well as state trust lands. GAP4 is lands with no mandate for protection such as tribal, military, and private lands. GAP status is relevant to the intensity of both current and past managements.

We made one modification to GAP levels by converting Inventoried Roadless Areas (IRAs) from the 2001 Roadless Area Conservation Rule (S_USA.RoadlessArea_2001, *public communication*, http://data.fs.usda.gov/geodata/edw/datasets.php) to GAP2 unless these areas already were defined as GAP1. We considered most IRAs as GAP2 given they are prone to policy changes and because they allow for certain limited types of logging (e.g., removal of predominately small trees for fuel reduction in some circumstances).

However, we note that very little logging has occurred within IRAs since the Roadless Rule, although there occasionally have been proposals to log portions of some IRAs pre- and postfire, and fire suppression often occurs.

We modified level III ecoregions (U.S. Environmental Protection Agency (EPA) 2013) to create areas of similar climate and geography (Fig. 1). We did this by extracting ecoregions and combining adjacent provinces in our study region.

Random Forests analysis

We investigated the relationship between protection status and burn severity using the data-mining algorithm Random Forests (RF) (Breiman 2001) with the "randomForestSRC" add-in package (Ishwaran and Kogalur 2016) in R (R Core Team 2013). This algorithm is an extension of classification and regression trees (CART) (Breiman et al. 1984) that recursively partitions observations into groups based on binary rule splits of the predictor variables. The main advantage of using RF in our study is that it can work with spatially autocorrelated data (Cutler et al. 2007). It can also model complex, nonlinear relationships among variables, makes no assumption of variable distributions (Kane et al. 2015), and produces accurate predictions without overfitting the available data (Breiman 2001).

Our independent observations were a random subset of our 5.5 million points, from which we drew three random samples of 25,000 points each. Each sample consisted of 500 fires randomly selected without replacement from the pool of 2069 fires. Fifty points were then randomly selected within each of the 500 fires. Our dependent variables were all continuous (Appendix S1: Tables S2 and S3) except for the main variable of interest, protected area status, which included the four GAP levels. The three observation samples were used to create three RF model runs, each consisting of 1000 regression trees. We conducted three RF model runs to assess whether our random samples of 25,000 points produced fairly consistent results.

The RF algorithm samples approximately 66% of the data to build the regression trees, and the remaining data are used for validation and to assess variable importance. We used this validation sample to determine the amount of variance explained and variable importance. The algorithm also produces individual variable importance measures by calculating differences in prediction mean-square-error before and after randomly permuting each dependent variable's values. Variable importance is a measure of how much each variable contributes to the model's overall predicative accuracy.

Unlike linear models, RF does not produce regression coefficients to examine how a change in a predictor variable affects the response variable. The analogy to this in RF is the partial dependence plot which is a graphical depiction of how the response will change with a single predictor while averaging out the effects of the other predictors, such as the climatic and topographic variables (Cutler et al. 2007). We used this approach, in addition to using RF to determine overall variable importance as described above, in order to determine the effect of GAP status, in particular, on fire severity, while averaging out effects of climate and topography.

Mixed-effects analysis

We performed a linear mixed-effects analysis using the "nlme" add-on package in R (Pinheiro et al. 2015). We used a random intercept model and identified year of fire (n = 31) and ecoregion (n = 10) as random effects. Similar to our RF models, our independent observations were a random subset of our 5.5 million points but for these models we drew three random samples of 50,000 points each. Each sample consisted of 500 fires randomly selected without replacement, and within each of those fires, 100 points were randomly selected. Our dependent variables were the same used in our RF models, and we log-transformed the non-normal variables of slope, surface roughness, and topographic radiation aspect index. We removed dependent variables that were correlated with each other (Pearson's r > 0.5), retaining 21 of 45 candidate dependent variables, and centered these on their means. Model reduction was performed in a stepwise process using bidirectional elimination with Bayesian information criterion selection criterion.

Spatial autocorrelation analysis

Spatial autocorrelation (SA) is the measure of similarity between pairs of observations in relationship to the distance between them. Ecological variables are inherently autocorrelated because

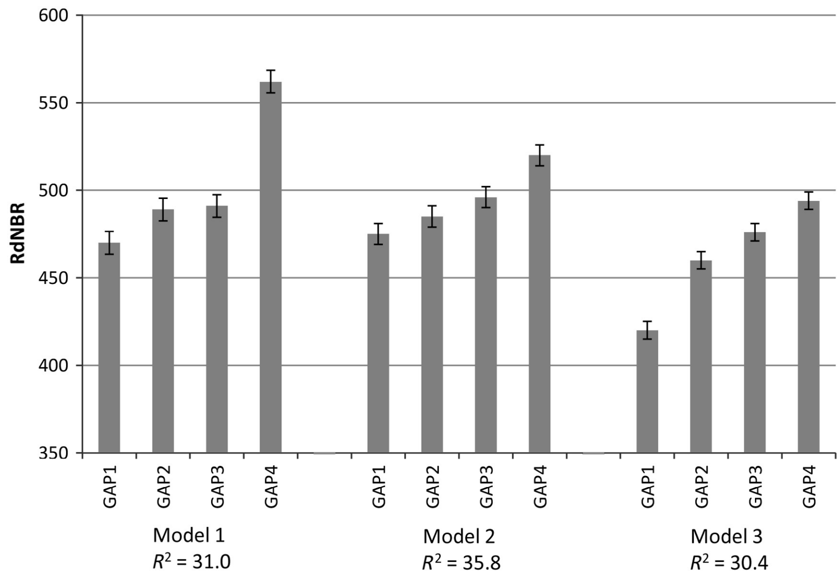

Fig. 2. Random Forests partial dependence of protection status vs. RdNBR burn severity for each model (n = 25,000). The variance explained is shown as pseudo R^2.

landscape attributes that are closer together are often more similar than those that are far apart.

We assessed the SA in the Pearson residuals with inspection of Moran's I autocorrelation index using the "APE" package add-in in R (Paradis et al. 2004) after removing points that shared the same x and y coordinates. Moran's I is an index that ranges from −1 to 1 with the sign of the values indicating strength and direction of SA. Values close to zero are considered to have a random spatial pattern. Our mixed-effects models all had a Moran's I values statistically different from 0 at the 95% confidence level ($P < 0.001$) so we included a spatial correlation structure in our model using the "nlme" package in R. Of Gaussian, exponential, linear, and spherical spatial correlation structures, we determined that the exponential structure produced the lowest Akaike's information criterion (AIC). Despite these additions, our second measurements still found relatively small, but significant, autocorrelation (Moran's I for model runs 1, 2, 3 = 0.10, 0.08, 0.10, all $P < 0.001$).

Results

With regard to ranking of variables in the model runs, variable importance plots from the three RF model runs show that protection status was consistently ranked as one of the 10 most important of the 45 variables in explaining burn severity (Appendix S1: Table S4). The most important variable explaining burn severity was ecoregion for models 1 and 2 and maximum temperature from the previous fall for model 3.

With regard to the GAP status variable in particular, after averaging out the effects of climatic and topographic variables, the RF partial dependence plots show an increasing trend of fire severity with decreasing protection status (Fig. 2). Fires in GAP4 had mean RdNBR values greater than two standard errors higher than all other GAP levels. Fires in GAP3 had mean RdNBR values two standard errors higher than GAP1 in all model runs. GAP3 differences with GAP2 were less pronounced with only one model showing differences greater than two standard errors. Fires in GAP1 were consistently the least severe, being two standard errors less than GAP3 in all model runs and two standard errors less than GAP2 in two of three model runs.

Our mixed-effects models validated these findings with similar results (Fig. 3, Appendix S1: Table S5). Like our RF models, our linear mixed-effects models showed GAP4 fires to have significantly higher RdNBR values and GAP1 fires to have significantly lower RdNBR values when compared to all other GAP classes. Fires in GAP

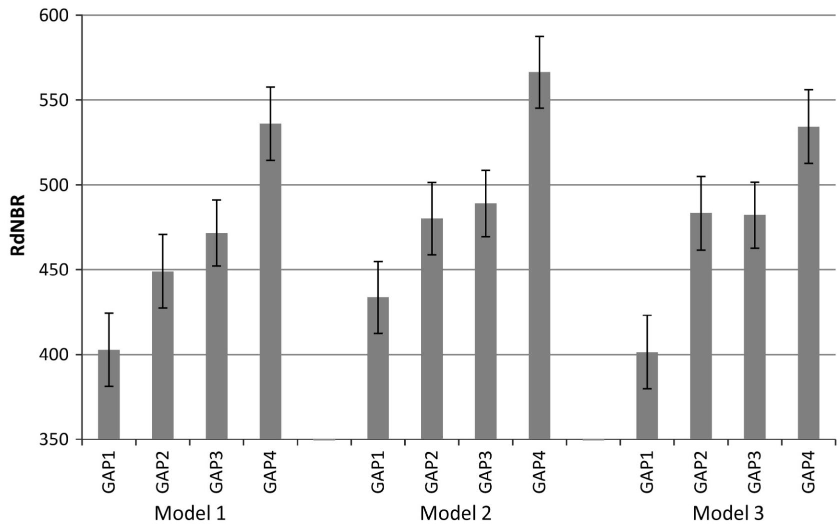

Fig. 3. Linear mixed effects models of protection status vs. RdNBR burn severity (n = 50,000).

status levels 2 and 3 were not significantly different in the mixed-effects models. Although the level of autocorrelation was significant, it was small in our model (Moran's I ~0.1) and not enough to account for such a substantial difference in burn severity among protection classes.

Discussion

Protected forests burn at lower severities

We found no evidence to support the prevailing forest/fire management hypothesis that higher levels of forest protections are associated with more severe fires based on the RF and linear mixed-effects modeling approaches. On the contrary, using over three decades of fire severity data from relatively frequent-fire pine and mixed-conifer forests throughout the western United States, we found support for the opposite conclusion—burn severity tended to be higher in areas with lower levels of protection status (more intense management), after accounting for topographic and climatic conditions in all three model runs. Thus, we rejected the prevailing forest management view that areas with higher protection levels burn most severely during wildfires.

Protection classes are relevant not only to recent or current forest management practices but also to past management. Millions of hectares of land have been protected from logging since the 1964 Wilderness Act and the 2001 Roadless Rule, but these areas are typically categorized as such due to a lack of historical road building and associated logging across patches >2000 ha, while GAP3 lands, for instance, such as National Forests lands under "multiple use management," have generally experienced some form of logging activity over the last 80 yr.

We expect that the effects of historic logging from nearly a century ago to gradually lessen over time, as succession and natural disturbance processes reestablish structural and compositional complexity, but it was beyond the scope of this study to attempt to assess the relative role of recent vs. historical logging. Similarly, industrial fire suppression programs that intensified in the 1940s influenced fire extent across forest protection classes. While more recent let-burn policies have been applied in GAP1 and GAP2 forests in some circumstances, evidence indicates that protected forests nevertheless remain in a substantial fire deficit, relative to the prefire suppression era (Odion et al. 2014, 2016, Parks et al. 2015). Thus, we believe it is unlikely that recent decisions to allow some backcountry fires to burn, largely unimpeded, account for much of the differences in fire severity among protection classes that we found, simply because such let-burn policies have not been extensive enough to remedy the ongoing fire deficit.

While forests in different protection classes can vary in elevation, with protected forests often occupying higher elevations, our results indicate that protection class itself produced notable differences in fire severity after averaging out the effects of elevation and climate (see Fig. 2 and *Results* above). In our study, GAP1 forests were 284 m on average higher in elevation than GAP4 forests, while GAP1 forests experienced lower fire severity. This is the opposite of expectations if elevation was a key influence because higher elevation forests are associated with higher fire severity (see, e.g., Schoennagel et al. 2004, Sherriff et al. 2014). We note that we are not the first to determine that increased fire severity often occurs in forests with an active logging history (Countryman 1956, Odion et al. 2004).

Prevailing forest–fire management perspectives vs. alternative views

An extension of the prevailing forest/fire management hypothesis is that biomass and fuels increase with increasing time after fire (due to suppression), leading to such intense fires that the most long-unburned forests will experience predominantly severe fire behavior (e.g., see USDA Forest Service 2004, Agee and Skinner 2005, Spies et al. 2006, Miller et al. 2009*b*, Miller and Safford 2012, Stephens et al. 2013, Lydersen et al. 2014, Dennison et al. 2014, Hessburg 2016). However, this was not the case for the most long-unburned forests in two ecoregions in which this question has been previously investigated—the Sierra Nevada of California and the Klamath-Siskiyou of northern California and southwest Oregon. In these ecoregions, the most long-unburned forests experienced mostly low/moderate-severity fire (Odion et al. 2004, Odion and Hanson 2006, Miller et al. 2012, van Wagtendonk et al. 2012). Some of these researchers have hypothesized that as forests mature, the overstory canopy results in cooling shade that allows surface fuels to stay moister longer into fire season (Odion and Hanson 2006, 2008). This effect may also lead to a reduction in pyrogenic native shrubs and other understory vegetation that can carry fire, due to insufficient sunlight reaching the understory (Odion et al. 2004, 2010).

Another fundamental assumption is that current fires are becoming too large and severe compared to recent historical time lines (Agee and Skinner 2005, Spies et al. 2006, Miller et al. 2009*b*, Miller and Safford 2012, Stephens et al. 2013, Lydersen et al. 2014, Dennison et al. 2014, Hessburg 2016). However, others have shown that this is not the case for most western forest types. For instance, using the MTBS (www.mtbs.gov) data set, Picotte et al. (2016) found that most vegetation groups in the conterminous United States exhibited no detectable change in area burned or fire severity from 1984 to 2010. Similarly, Hanson et al. (2009) found no increase in rates of high-severity fire from 1984 to 2005 in dry forests within the range of the northern spotted owl (*Strix occidentalis caurina*) based on the MTBS data set. Using reference data and records of high-severity fire, Baker (2015) found no significant upward trends in fire severity from 1984 to 2012 across all dry western forest regions (25.5 million ha), nearly all of which instead were too low or were within the range of historical rates. Parks et al. (2015) modeled area burned as a function of climatic variables in western forests and non-forest types, documenting most forested areas had experienced a fire deficit (observed vs. expected) during 1984 to 2012 that was likely due to fire suppression.

Whether fires are increasing or not depends to a large extent on the baseline chosen for comparisons (i.e., shifting baseline perspective, Whitlock et al. 2015). For instance, using time lines predating the fire suppression era, researchers have documented no significant increases in high-severity fire for dry forests across the West (Williams and Baker 2012*a*, Odion et al. 2014) or for specific regions (Williams and Baker 2012*b*, Sherriff et al. 2014, Tepley and Veblen 2015). Future trends, with climate change and increasing temperatures, may be less simple than previously believed, due to shifts in pyrogenic understory vegetation (Parks et al. 2016).

This is more than just a matter of academic debate, as most forest management policies assume that fire, particularly high-severity fire, is increasing, is in excess of recent historical baselines, and needs to be reduced in size, intensity, and occurrence over large landscapes to prevent widespread ecosystem damages (policy examples include USDA Forest Service 2002, Healthy Forests Restoration Act 2003, USDA Forest Service 2009, HR 167: Wildfire Disaster Funding Act 2015). However, large fires (landscape scale or the so-called megafires) produce myriad ecosystem benefits underappreciated by most land managers and decision-makers (DellaSala and Hanson 2015*a*, DellaSala et al. 2015). High-severity fire patches, in particular, provide a pulse of "biological legacies" (e.g., snags, down logs, and native shrub patches) essential for complex early seral associates (e.g., many bird species) that link seral stages from new forest to old growth (Swanson et al. 2011, Donato et al. 2012, DellaSala et al. 2014, Hanson 2014, 2015, DellaSala and Hanson 2015*a*). Complex early seral forests are most often logged after fire, which, along with aggressive fire suppression, exacerbates their rarity and heightens their conservation importance (Swanson et al. 2011, DellaSala et al. 2014, 2015, Hanson 2014).

Limitations

One limitation of our study is that, due to the coarseness of the management intensity variables that we used (i.e., GAP status), we cannot rule out whether low intensities of management decreased the occurrence of high-severity fire in some circumstances. However, the relationship between forest density/fuel, mechanical fuel treatment, and fire severity is complex. For instance, thinning without subsequent prescribed fire has little effect on fire severity (see Kalies and Yocum Kent 2016) and, in some cases, can increase fire severity (Raymond and Peterson 2005, Ager et al. 2007, Wimberly et al. 2009) and tree mortality (see, e.g., Stephens and Moghaddas 2005, Stephens 2009: Figure 6)—the effects depend on the improbable co-occurrence of reduced fuels (generally a short time line, within a decade or so) and wildfire activity (Rhodes and Baker 2008) and can be over-ridden by extreme fire weather (Bessie and Johnson 1995, Hély et al. 2001, Schoennagel et al. 2004, Lydersen et al. 2014). Empirical data from actual fires also indicate that postfire logging can increase fire severity in reburns (Thompson et al. 2007), despite removal of woody biomass (tree trunks) described by land managers as forest fuels (Peterson et al. 2015). While our study did not specifically test for these effects, such active forest management practices are common on GAP3 and GAP4 lands. Recognizing these limitations, researchers have stressed the need for managers to strive for coexistence with fire by prioritizing fuel reduction nearest homes and allowing more fires to occur unimpeded in the backcountry (Moritz 2014, DellaSala et al. 2015, Dunn and Bailey 2016, Moritz and Knowles 2016).

Follow-up research at finer scales is needed to determine management emphasis and history in relation to fire severity. However, we believe our findings are robust at the subcontinental and ecoregional scales.

Conclusions

In general, our findings—that forests with the highest levels of protection from logging tend to burn least severely—suggest a need for managers and policymakers to rethink current forest and fire management direction, particularly proposals that seek to weaken forest protections or suspend environmental laws ostensibly to facilitate a more extensive and industrial forest–fire management regime. Such approaches would likely achieve the opposite of their intended consequences and would degrade complex early seral forests (DellaSala et al. 2015). We suggest that the results of our study counsel in favor of increased protection for federal forestlands without the concern that this may lead to more severe fires.

Allowing wildfires to burn under safe conditions is an effective restoration tool for achieving landscape heterogeneity and biodiversity conservation objectives in regions where high levels of biodiversity are associated with mixed-intensity fires (i.e., "pyrodiversity begets biodiversity," see DellaSala and Hanson 2015*b*). Managers concerned about fires can close and decommission roads that contribute to human-caused fire ignitions and treat fire-prone tree plantations where fires have been shown to burn uncharacteristically severe (Odion et al. 2004). Prioritizing fuel treatments to flammable vegetation adjacent to homes along with specific measures that reduce fire risks to home structures are precautionary steps for allowing more fires to proceed safely in the backcountry (Moritz 2014, DellaSala et al. 2015, Moritz and Knowles 2016).

Managing for wildfire benefits as we suggest is also consistent with recent national forest policies such as 2012 National Forest Management Act planning rule that emphasizes maintaining and restoring ecological integrity across the national forest system and because complex early forests can only be produced by natural disturbance events not mimicked by mechanical fuel reduction or clear-cut logging (Swanson et al. 2011, DellaSala et al. 2014). Thus, managers wishing to maintain biodiversity in fire-adapted forests should appropriately weigh the benefits of wildfires against the ecological costs of mechanical fuel reduction and fire suppression (Ingalsbee and Raja 2015) and should consider expansion of protected forest areas as a means of maintaining natural ecosystem processes like wildland fire.

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Supporting Information

Additional Supporting Information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/ecs2.1492/full

CENTER FOR SUSTAINABLE ECONOMY
Building the DNA of the New Economy

May 2019 **REPORT**

ENVIRONMENTALLY HARMFUL SUBSIDIES IN THE U.S.
Issue #1: The federal logging program

How damaging logging operations on federal public lands costs taxpayers nearly $2 billion each year

Subsidized commercial logging under the guise of fire risk reduction makes forests hotter, drier, and more susceptible to climate change. Photo credit: US Forest Service.

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CENTER
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ECONOMY
Building the DNA of the New Economy

ENVIRONMENTALLY HARMFUL SUBSIDIES IN THE U.S.
Issue #1 - The federal logging program

By John Talberth, Ph.D. and Ernie Niemi

KEY FINDINGS

"Our analysis finds that the logging program on federal forests continues to lose money for taxpayers in the range of $1.5 to $2.0 billion per year."

"Congress can remedy this situation by restricting use of appropriated funds for vegetation management on national forest and BLM lands to ecological restoration projects that are decoupled from commercial logging."

- Each year, the US Forest Service authorizes logging of roughly 3 billion board feet of timber – equivalent to 650,000 full log truck loads – from national forest lands.
- This logging comes at a steep environmental cost in the form of loss of biological diversity, damage to water supplies, and increased risks from wildfires, flooding, and climate change.
- The federal logging program comes with steep economic and financial costs as well. In particular, national forest logging displaces uses and functions such as carbon storage, recreation and water filtration that are far more valuable than timber. And because the Forest Service sells its timber far below cost, it results in significant taxpayer losses.
- In two previous assessments that applied a methodology reviewed by the Congressional Research Service, the John Muir Project found annual taxpayer losses of nearly $1.2 billion per year between fiscal years 1997 and 2004, $1.7 billion in 2018 dollars.
- This report updates the John Muir Project methodology by comparing timber sale program receipts deposited into the US Treasury with Forest Service logging related expenditures during fiscal years 2013 to 2017.
- Our analysis finds that the logging program on national forests continues to lose money for taxpayers in the range of $1.3 to $1.5 billion per year.
- When additional federal logging subsidies related to fire suppression and BLM losses are included, the total exceeds $1.8 billion per year.
- As such, the federal logging program runs afoul of international agreements and ambitions to phase out environmentally harmful subsidies and make international trade more economically efficient.
- Congress can remedy this situation by restricting use of appropriated funds for vegetation management on national forest and BLM lands to ecological restoration projects that are decoupled from commercial logging.

ABOUT THIS SERIES

"International institutions have active programs to phase out environmentally harmful subsidies and redirect state support, instead, to alternatives that advance triple bottom line goals for sustainable development."

Rescinding and redirecting environmentally harmful subsidies have long been recognized as effective tools for advancing a global sustainable development agenda and making global trade regimes more efficient. Subsidies for fossil fuels, mining, logging, industrial agriculture, factory fishing and other activities that pollute land, air and water and drive climate change run in the trillions of dollars each year. Fossil fuel subsidies by themselves were found to approach $5 trillion annually by a recent International Monetary Fund (IMF) assessment while subsidies for other sectors add at least another $1 trillion.[1] International institutions such as the Organization for Economic Cooperation and Development, Convention on Biological Diversity and IMF have active programs in place to work with governments to identify and phase out these harmful subsidies and redirect state support, instead, to alternatives that advance triple bottom line goals of economic prosperity, equity, and ecological sustainability.

This report series considers environmentally harmful subsidies (EHS) in the United States beginning with an analysis of federal and state-level subsidies for timber. The US is the world's largest producer and consumer of wood products but also has some of the most productive forestlands that can play a major role in mitigating climate change if managed for long term carbon storage and restoration of natural forest conditions. Instead, logging subsidies support carbon intensive forest practices like short rotation clearcutting and the conversion of natural forests into tree plantations. The first two reports in this series consider subsidies for logging on federal public lands and a wide range of state-level tax advantages and subsidies that support industrial forest practices on privately managed timberlands. This issue is devoted to an analysis of logging subsidies on federal public lands, with a focus on national forests managed by the US Forest Service, an agency of the United States Department of Agriculture (USDA).

NATIONAL FORESTS PLAY A UNIQUE ECOLOGICAL ROLE

"These lands play a unique ecological role because they represent islands in a sea of heavily damaged lands managed by states and private landowners."

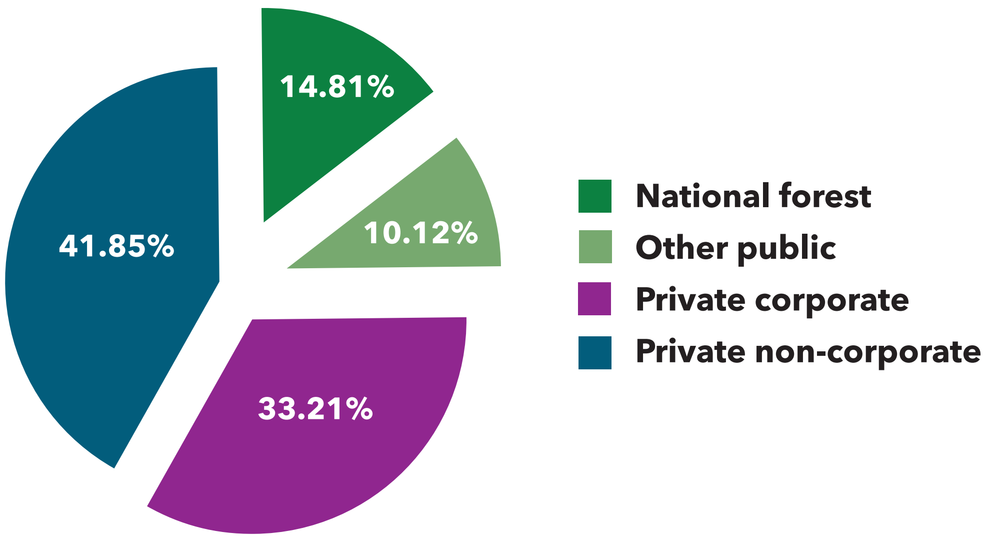
The USDA Forest Service manages 144 million acres of forestland in the US, about 19% of the 765 million acre total. These lands play a unique ecological role because they represent islands in a sea of heavily damaged lands managed by states and private landowners. They support the few remnants of native forest ecosystems that have not been converted to industrial tree plantations or otherwise damaged by logging, grazing, mining, roads, development and other human activities. They are the headwaters of streams and rivers vital for drinking water, irrigation, and industry. They support wildlife, fish, and plant species that provide valuable services to our economy in the form of foods, medicines, and ecosystem services such as pollination. They provide the lion's share of forested recreational opportunities. And they are critical for capturing and storing carbon and helping humanity bend the curve on carbon dioxide concentrations in the atmosphere back below the scientific upper limit safe zone of 350 parts per million.

There are two key reasons why national forests retain their relatively valuable role. The first is that, in general, national forest lands are steeper, higher in elevation, less accessible and less productive than lands managed by states and private landowners. As such, they represent the lands left over after settlers and private industry took the most productive and accessible lands for themselves. This can be seen by considering the distribution of forest productivity among the forestland ownership types. As Figure 1 shows, 85% of the most productive lands – those that have the potential to grow over 120 cubic feet per acre per year – fall outside national forest boundaries. Because national forestlands are less productive, they have been historically less attractive for commercial logging.

"85% of the most productive lands – those that have the potential to grow over 120 cubic feet per acre per year – fall outside national forest boundaries."

Figure 1

Ownership share of high productivity (+120 cubic feet/acre/year) timberlands in the US



As a result, significant tracts of national forestlands have escaped the chainsaw – so far.

The second reason why national forests have remained relatively intact is that the laws governing national forests and other federal forestlands are far more accommodating to non-timber uses such as recreation, hunting, fishing and conservation of wildlife, fish and water quality. Federal laws mandate that significant amounts of the land base be set aside for these resources. In contrast, the laws governing the management of state and private forest lands emphasize timber production above all else and have few requirements for set asides to protect these resources and uses. As a result, a typical landscape within the industrial forestland matrix supports little to no habitat for most native species of wildlife, fish and plants and, instead exists as a sea of industrial tree plantations. The area occupied by timber plantations is growing, threatening more biodiversity loss. One of the key justifications for ending the logging program on national forests is so they can serve as a buttress against this extinction threat.

"One of the key justifications for ending the logging program on national forests is so they can serve as a buttress against the extinction threat posed by industrial tree plantations."

BECAUSE OF THEIR UNIQUE ROLE AND LIMITED SUITABILITY, LOGGING ON NATIONAL FORESTLANDS IS UNECONOMICAL

"As shown by Figure 2, carbon beats out timber on both national forests by a large margin. Even at a high discount rate (which tends to inflate timber values relative to long term carbon storage) management of the land for carbon storage yields two to three times the present value estimate per acre than that same acre managed for timber."

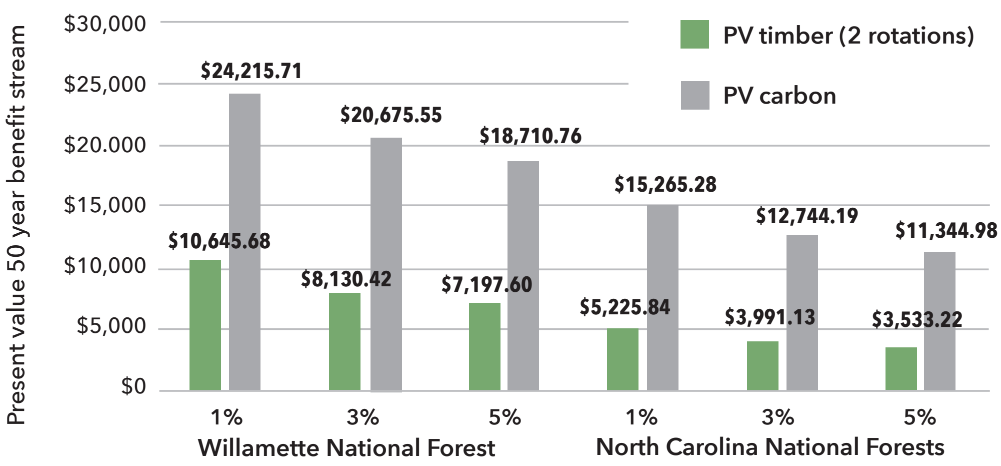
"Adding the value of recreation, water, fish, wildlife, wild pollinators makes the economic case against logging even stronger."

The unique ecological role played by national forest lands coupled with their limitations on suitability render these lands, for the most part, uneconomical to log. There are two economic dimensions to consider: economic and financial feasibility. Economic feasibility is a term used to signify whether or not an economic activity yields benefits in excess of costs for society as a whole, taking into consideration effects measured by market transactions as well as effects that are non-market in nature. Timber and other forest products have established market prices and are measurable with relative ease. Carbon, recreation, and water filtration are examples of non-market goods and services (ecosystem services) that are important economically but harder to measure. There are, nonetheless, established methods for valuing ecosystem services. An entire branch of environmental economics is dedicated to robust methods to do so. And what these studies show is that national forest lands are, in general, far more valuable managed for non-timber ecosystem services.

Figure 2 illustrates this point by comparing the present value of the benefit stream associated with managing a typical acre of land on two national forest areas – the Willamette National Forest in Oregon and the North Carolina national forests – for either carbon or timber. If the acre were managed for timber, the analysis assumes two cutting cycles over 50 years. The first cycle yields income from the sale of timber in year one and then again in year fifty. Timber values are based on actual transaction data for timber sales sold by these national forests in 2018.[2]

Figure 2

Timber vs. carbon values per acre ($2018) at discount rates of 1%, 3% and 5%



If the acre were managed for carbon, the analysis puts value on both the existing carbon stock contained in live trees and vegetation as well as the additional carbon accumulated in trees over the next fifty years. Carbon stock data were drawn from Forest Inventory and Analysis (FIA) program publications. The social cost of carbon as estimated by the EPA and other federal agencies was the basis for valuing these carbon stocks. Think of the value as the avoided costs of emissions associated with logging if the acre were, instead, protected and allowed to grow and accumulate carbon over time. The SCC used in this analysis is $50 per ton of CO_2 which is the midpoint federal estimate at a discount rate of 3% updated to 2018 dollars. Carbon sequestration rates (the annual tons of CO_2 captured) were derived from local estimates of net ecosystem productivity (NEP), which considers all the carbon sequestered by a forest minus what it gives off through natural processes.[3]

"Using a methodology reviewed by the Congressional Research Service and found to be a "reasonable estimate" by CRS Hanson found that the national forest timber sale program lost roughly $1.8 billion ($2018) in FY 1997."

As shown by Figure 2, carbon beats out timber on both national forests by a large margin. Even at a high discount rate (which tends to inflate timber values relative to long term carbon storage) management of the land for carbon storage yields two to three times the present value estimate per acre than that same acre managed for timber. And carbon sequestration is just one ecosystem service provided by forests if allowed to grow and mature. Adding the value of recreation, water, fish, wildlife, wild pollinators makes the economic case against logging more compelling.

And then there is the financial case to consider: the fact that the federal logging program is a big money loser for taxpayers. The issue of below-cost subsidized federal timber first emerged in the 1980s with a series of reports issued by the General Accounting Office and Congressional Research Service and others. Few attempts to actually quantify the annual losses were made until a 1997 study by Chad Hanson with John Muir Project which was then updated in 1999. Using a methodology reviewed by the Congressional Research Service and found to be a "reasonable estimate" of the net cash loss, Hanson found that the national forest timber sale program lost roughly $1.2 billion during FY 1997, or $1.8 billion in 2018 dollars. Receipts generated by timber sales that were not funneled back into logging related expenses did not even come close to covering the Forest Service's logging related costs.

"Replicating most of the earlier methodology, JMP found that the situation did not change much, if at all, during the FY 1998 – FY 2004 period. The logging program on national forests continued to lose money during this period – an average logging subsidy of roughly $1.7 billion per year."

The Hanson (1999) study was followed up in 2005 by an additional analysis by John Muir Project's Rene Voss.[4] Replicating most of the earlier methodology, JMP found that the situation did not change much, if at all, during the FY 1998 – FY 2004 period. The logging program on national forests continued to lose money during this period – an average logging subsidy of roughly $1.2 billion per year, $1.7 billion in 2018 dollars.

Despite being uneconomical and a money loser for taxpayers, the national forest logging program continues, and is expected to grow larger in the coming years. The Forest Service's 2018 Budget Justification states that the agency's goal is to increase the amount of timber sold from 2.94 billion board feet (BBF) in 2016 to 3.8 BBF in the near future. This is equivalent to nearly 650,000 full log truck loads a year. Before we update the subsidy calculations, it is important to explain how the Forest Service justifies losing substantial taxpayer dollars and continuing to allocate lands to timber production when non-timber uses are so much higher.

YET HARMFUL LOGGING ON NATIONAL FORESTS CONTINUES TO BE SUBSIDIZED

"The Forest Service's 2018 Budget Justification states that the agency's goal is to increase the amount of timber sold from 2.94 billion board feet (BBF) in 2016 to 3.2 BBF in the near future. This is equivalent to nearly 650,000 full log truck loads a year."

"The Forest Service contends that timber sales are needed to achieve several important ecological goals."

"The problem is that mixing commercial logging activities in with projects that would otherwise be cleanly focused on ecological objectives results in projects that often do more harm than good as well as projects that would otherwise be unnecessary."

Over the past two decades, the Forest Service – with notable exceptions in places like southeast Alaska – has moved away from justifying its logging program on the grounds that it needs to help sustain the timber industry. With unregulated access to the nation's most productive forestlands and the fact that much of what is cut on non-federal lands is exported, the public is not easily convinced that this industry needs support from taxpayers. So on most national forests, the Forest Service instead contends that timber sales are needed to achieve several important ecological goals. In particular, the agency claims that:

> Timber sales and stewardship projects can both reduce the density of trees and change the type of trees in the forest. This can improve the vigor and health of forests and improve wildlife habitat for multiple species. Additionally, timber sales and stewardship contracts can help with multiple goals, including restoring large-scale watersheds by reducing fuels that create an unacceptable fire risk, recovering timber value following natural disturbances, and preparing sites for vegetation to regenerate. Timber sales and stewardship contracts can also be used to reduce insect and disease infestations, improve resilience to drought, and improve tree growth to produce desirable timber products in the future.[5]

Thus losing money on the timber sale program is justified as necessary in order to carry out ecological restoration projects that would otherwise not occur or have to be financed at full cost. The problem is that mixing commercial logging activities in with projects that would otherwise be cleanly focused on ecological objectives results in projects that often do more harm than good as well as projects that would otherwise be unnecessary. Many thinning projects designed to reduce fire risk have been shown to actually elevate the risk because of logging slash left behind and changes in microclimates that create hotter, drier, and more open forest conditions.[6] Salvage sales ignore the ecological benefits of natural disturbances and result in widespread damage to soils that would otherwise be retained onsite to help the next generation of vegetation grow.[7]

The scientific case against Forest Service timber sales that are purported to help advance ecological goals has, time and time again, landed the agency in court and triggered protests from scientific and conservation organizations. A small sample of recent disputes include:

- **The Crystal Clear Restoration Project, Mt. Hood National Forest:** This project proposes logging of approximately 4,000 acres in order to "improve stand conditions, reduce the risk of high-intensity wildfires, and promote safe fire suppression activities."[8] However, in their comments on the project, a coalition of conservation and scientific organizations refute this claim, noting that the proposed area is in fact at low risk for fires and that logging will pose a threat to critical wildlife habitats.[9] The objectors cite that the project is within Fire Regime Condition Class 1, indicating that this area of forest is closest to its natural vegetation patterns and is of least concern for fuel, fire frequency, severity, and pattern. Nor is it on land that is designated by the Wasco County Community Wildfire Protection Plan as a priority for fuel reduction. According to Forest Service, the project will "downgrade 1,059 acres of suitable nesting, roosting and foraging habitat and remove 895 acres of dispersal habitat" from a northern spotted owl critical habitat area.

- **Crane Point Forest Health Project – Nez Perce-Clearwater National Forest:** The Forest Service proposed to log 1,350 acres on the grounds that doing so would decrease insect and disease levels, decrease the dominance of shade tolerant species of trees, and that harvesting wood products would sustain local and regional economies.[10] However, objectors report that "clearcuts put ecological communities at the forest's edge at risk for disease," and quote the Forest Service itself claiming that "diseases which reduce timber production are certainly damaging in commercial forests…The same diseases, however, may be of little or no consequence in parks or watershed protection areas."[11]

- **French Fire Logging Project – Sierra Nevada National Forest:** The Forest Service is currently proposing to post-fire log most of the complex early seral forest in this fire area—including in Pacific Fisher habitat and occupied California Spotted Owl and Black-backed Woodpecker territories. While Pacific Fishers select dense, old forest for denning and resting, they actively use, and select, higher-intensity fire areas as foraging habitat—especially the females, for which there is the greatest conservation concern.[12]

- **Greenwood Vegetation Management Project – Daniel Boone National Forest:** The Greenwood Vegetation Management Project proposes to log 3,600 acres of the Daniel Boone National Forest to "meet desired future condition for mid-density upland forest" and make forest more resistant to disturbance.[13] The objectors note, however that "the specific justification given for timber harvesting in the Purpose and Need for woodland establishments is to create mid-density forests meeting specific basal area targets that ostensibly do not exist in the project area."[14] Additionally, objectors note that forests are already in the basal range of 30-50 ft2/ac, the goal density of this project.

- **East Side Timber Project – Allegheny National Forest:** The East Side Timber Project in the Allegheny National Forest resulted in 3,000 acres of even-aged logging in 2006, ostensibly, to improve biological diversity through even aged management. However scientific and conservation groups challenged this in court, stating that the proposed project will do the opposite. The plaintiffs cited research showing that "even-aged management would result in the least amount of old growth habitat, the highest amount of soil compaction, the lowest amount of standing dead and lying dead trees for wildlife habitat, the highest acreage of forest with more than 30% stocking of interfering ferns of all alternatives," and that uneven-aged management could "obtain adequate regeneration of diverse tree species."[15]

"What these case studies illustrate is that the Forest Service's ecological justifications for subsidized logging on national forests rests on very shaky footing. Taxpayer losses are not made up for by the purported ecological benefits of logging – to the contrary, taxpayers not only lose money on national forest timber sales but see their lands further degraded."

What these case studies illustrate is that the Forest Service's justifications for subsidized logging on national forests rests on very shaky footing. Scientific information presented in appeals and litigation regularly challenges the idea that commercial logging is compatible with stated goals for fire risk reduction, post fire rehabilitation, biological diversity, watershed integrity and other ecological objectives. Taxpayer losses are not made up for by the purported ecological benefits of logging – to the contrary, taxpayers not only lose money on national forest timber sales but see their lands further degraded.

YET HARMFUL LOGGING ON NATIONAL FORESTS CONTINUES TO BE SUBSIDIZED

This section presents the general methodology we used to update the subsidy calculations for the fiscal years 2013 to 2017. Unless otherwise noted, the calculations replicate the methodology used by John Muir Project (JMP) in their 2005 update. That methodology compares all Forest Service expenditures associated with the timber sale program to timber sale receipts deposited in the US Treasury. Expenditures are identified using the 'but for' criteria – but for the existence of the timber sale program the associated expenditure would otherwise not have been made.

Expenditure data are drawn from annual budget justifications prepared by the Forest Service for each fiscal year.[16] Forest Service timber sale expenditures can be divided into two basic categories: appropriated funds and off budget funds. Appropriated funds are line items authorized by Congress. Off-budget funds are those capitalized by timber sale revenue and spent without the need for additional authorizations from Congress.

"Taxpayer losses are not made up for by the purported ecological benefits of logging – to the contrary, taxpayers not only lose money on national forest timber sales but see their lands further degraded."

Treasury deposits, not timber sale revenues, are the key metric indicating what financial return taxpayers receive because the vast majority of revenues generated by the sale of timber go back into funds that are used to plan, prepare, implement, and clean up after more timber sales. As such, Treasury deposits represent the actual financial benefit to taxpayers. However, the Forest Service does not report Treasury deposits directly. Instead, it reports deposits into the National Forests Fund (NFF), which is then transferred to either the US Treasury or to states (pursuant to 16 USC § 500, states receive 25% of gross receipts for national forest logging projects) for use on roads and schools in the counties where national forests are located. Regardless, payments to states are ostensibly a benefit to taxpayers and so using the NFF deposits in lieu of direct data on Treasury deposits is an acceptable alternative.

Find below a brief description of line items within the two major expenditure categories as well as the methodology used to assign the appropriate portion of the line item to the timber sale program. We also briefly discuss the source of information for the NFF deposits.

National forest timber sale program funds appropriated by Congress

Timber sales and other forest products management (TS): This is the most direct expenditure used to finance planning and preparation of timber sales. However, the line item also includes a small portion used to prepare sales of non-timber forest products sales such as edible and medicinal plants, personal use firewood, posts, and poles, and shrubs

for landscaping. That portion was estimated by JMP to represent about 2% of this line item, and so we follow suit by subtracting that amount from each year's appropriation. As discussed below, we also backed out (deducted) the portion of this line item spent on forest roads, which are reported separately here.

Vegetation and watershed management (VWM): These expenditures are purported to support landscape-level restoration but, in fact, focus on projects to enhance the timber resource including thinning, timber stand improvement, reforestation, pruning, and other tree and nursery improvement projects.[17] In addition, many expenditures not directly related to enhancing the timber resource are made to repair damages from past logging. Following the JMP methodology approved by CRS, we allocate 100% of this line item to the timber sale program.

Reforestation trust fund (RTF): This fund supports reforestation and timber stand improvement activities that would not otherwise be needed but for the timber sale program. So, this line item is allocated as a timber sale program expense in its entirety.

Hazardous fuels (HF): This expenditure supports prescribed fire, mechanical fuels reduction, and thinning activities. While the overall intent of these expenditures is to reduce risks of high intensity fires, much of the spending takes place on lands that are not priorities for fire risk reduction (i.e. wildland-urban interface zones) and supports projects such as the Crystal Clear project on the Mt. Hood National Forest that are less about fire risk reduction and more about generating commercial timber for sale. Following JMP, the share of this expenditure allocated to the timber sale program was calculated in three steps for each of the fiscal years included in our analysis: (1) removing the acres of land treated with prescribed and natural fire based on data reported in budget justifications; (2) assuming that 35% of the remaining treatment acres were treated mechanically to produce wood products, a percentage derived from prior studies, and (3) applying a cost estimate of $400 per acre ($2005) to these treatment acres, but updating the value to reflect current (i.e. current to each fiscal year) dollars. Over the FY 2013 to FY 2017 period, the average allocation to the timber sale program from this line item was just over 35%.

Forest health management – federal lands (FHF): This line item supports projects designed to eliminate or contain invasive species as well as insects and disease that are native, but which pose threats to the timber resource. While controlling invasive species is a desirable and laudable program that has little connection to the timber sale program, management of native insects such as the southern and mountain pine beetles is regularly used to justify timber sales that are difficult to defend ecologically since they involve suppression of a natural disturbance done primarily for the purpose of protecting the timber commodity. JMP backed out expenditures on insects and disease indirectly, however, it is now possible to be more precise since the Forest Service now reports the acreages assigned to invasive species and pathogens in its budget justifications. For each fiscal year in our analysis, we used these figures as a basis for assigning the share of this line item to insects and disease suppression activities. The

share ranged from a low of 53% in FY2014 to a high of 63% in FY 2017. This assumes that the unit cost of each activity – invasive species vs. insect and disease suppression are similar.

Forest roads (FR): The Forest Service provides engineering and other forms of support for road construction by purchasers of national forest timber. This line item captures those expenditures. Beginning in FY 2010, the Forest Service stopped disclosing these expenditures, which are now folded into the larger timber sales and other forest products line item. To compensate, we extracted the latest forest roads expenditure data for the fiscal years 2007 to 2009 and calculated its share of the Timber sales and other forest products line item for those years. That share averaged roughly 15% during this period. We then applied this percentage to the Timber sales and other forest products line items for FY 2013 to FY 2017.

Roads maintenance (RM): There are over 380,000 miles of roads on national forest system lands.[18] The vast majority of these were constructed to support logging operations. Each year, the Forest Service spends roughly $170 million to maintain these roads. The question is what share to allocate to non-timber uses, such as recreation. The JMP methodology does this by multiplying the road maintenance total each year by a ratio that reflects the proportion of direct expenditures on timber sales vs. spending on timber sales plus recreation (R), or: (TS+VWM+SS+OB)/(TS+VWM+SS+OB+R). We made no changes to this method.

Land and resource management planning, inventory, and monitoring (LRMP): A significant share of this line item is spent on delineating and inventorying lands suitable for timber harvest and monitoring timber sale and post-logging activities. JMP calculated this share by dividing total expenditures for logging related activities (TS+VWM+SS+OB) by this sum plus the amount spent on non-logging related programs also addressed by LRMP activities including recreation, grazing, minerals, wildlife and fish. The resultant share to logging varied between 57% and 62% over the FY 2013 to FY 2017 period.

Land ownership management (LO): These funds are used to administer national forest holdings and boundaries, which includes timber sale boundary location. Here we applied the same percentages derived in the previous land management planning, inventory and monitoring (LRMP) estimate and multiplied it by the land ownership management line item total for each year.

Timber research (TR): The Forest Service uses a significant share of funds appropriated for research to support timber sale program activities. The Forest Service's budget justifications break out various subcomponents of the research budget. The line item Resource Management and Use is the most relevant for timber sale program activities, so we include this line item in total. There are no additional sources of information to break down this line item further or assign additional funds from other research programs. The resulting share of the research budget devoted to the timber sale program averaged about 30% during the FY 2013 to FY 2017 period, a bit higher than the previous JMP estimate of 21%.

Support from other budget line items (SP): Many other budget line items contribute to the timber sale program indirectly. For example, many watershed restoration or wildlife habitat improvement projects that include logging activities (see, e.g. the East Side timber sale case study) and generate commercial quantities of timber are paid for out of funds set aside for wildlife and fish. JMP estimated the share of support from other line items to amount to 13.9% of the timber sales line item. We found no reason to adjust this amount in this update and so carried that share forward.

Off-budget expenditures for logging

These are funds that are not appropriated from the general fund of the US Treasury but are nonetheless expended in support of logging activities. Much of the funding comes from timber sale receipts retained by the Forest Service. Unless otherwise noted, each of these line items are included in their entirely since they are exclusively designed to support timber sale program activities. These expenditure line items are also published in each year's budget justification, and include:

Purchaser credit roads: Timber sale purchasers who elect to have the Forest Service build the permanent roads required in the sale contract make deposits to a special account and funds are permanently appropriated to the Forest Service to build the required roads.

Timber pipeline restoration fund: This fund includes receipts from certain canceled-but-reinstated timber sales and from additional sales prepared with the fund. These funds are permanently appropriated to the Forest Service. According to a 2011 CRS analysis, 75% of the funds are used to prepare additional timber sales and the other 25% is for recreation projects.[19] We thus include 75% of this line item as a timber sale program expenditure.

Salvage sales fund: Receipts from the sale of timber salvaged after fires or other disturbances are deposited in this account and permanently appropriated to the Forest Service, primarily to fund additional salvage sales.

Brush disposal fund: Purchasers of national forest timber sales make deposits over and above the stumpage price for the sale into this fund, which is used by the Forest Service to dispose of tree tops, limbs, and other woody debris from timber harvesting. The amount is determined for each sale.

Cooperative work trust funds: Forest Service budget justifications identify two categories of cooperative trust fund work relevant to the timber sale program: (1) Knutson-Vanderberg (KV) related, and (2) 'other.' KV funds are derived from timber sale receipts and are used for reforestation, timber stand improvement, and for protection of other resources affected by timber sales. The 'other' expenditure category includes funds collected directly from timber sale purchasers to finance other special projects within timber sale boundaries purchasers elect not to complete, such as road maintenance.

National Forest Fund deposits

After allocating timber sale program revenues to the line items that support future timber sales, the Forest Service deposits the remainder in the National Forest Fund (NFF). As noted above, these funds are, in turn, either redeposited into the US Treasury or sent back to states to fulfill statutory obligations regarding the sharing of timber sale gross receipts with counties to support roads and schools. As such, they represent the net return taxpayers receive from the timber sale program after all accounting for all costs and diversion of revenue to fund more timber sales.

NFF deposits are reported annually in the Forest Service ASR 04 report series.[20] These reports, which are available by region, by state, and by each national forest disclose NFF deposits from revenues earned through the sale of timber, grazing, land use, recreation, power, minerals and crystals. NFF receipts from each product are reported separately, and so we extracted the relevant data for timber from each fiscal year included in this update (FY 2013 – FY 2017).

RESULTS

Results for FY 2013 through FY 2017 are reported in Table 1, below (page 13). All values are expressed in 2018 dollars using the US consumer price index to account for inflation. Timber sale program expenditures are divided into the two broad groups of appropriated and off budget funds and then totaled. NFF deposits are displayed below the total timber sale program expenditure line. These deposits are subtracted to show the net financial impact to taxpayers.

"…during the fiscal years 2013 to 2017 the timber sale program on national forests was a net cost to taxpayers in the range of $1.34 to $1.51 billion per year, which translates into a subsidy of between $500 and $600 per every thousand board foot (mbf) logged. The average annual taxpayer loss over the five-year period was $1.41 billion."

As shown by Table 1 during the fiscal years 2013 to 2017 the timber sale program on national forests was a net cost to taxpayers in the range of $1.34 to $1.51 billion per year, which translates into a subsidy of between $500 and $600 per every thousand board foot (mbf) logged. The average annual taxpayer loss over the five-year period was $1.41 billion. Previous analyses by Hanson (1999) and Voss (2005) found the average annual losses during the FY 1997 to FY 2004 period to average a bit more - $1.71 billion per year in 2018 dollars. Thus, the timber sale program on national forests is a chronic money loser for taxpayers and continues to be subsidized at roughly the same levels it was since the late 1990s – over $1.4 billion per year. And as noted extensively in the prior Hanson (1999) and Voss (2005) reports these estimates are conservative because they do not include many other expenditures attributable to the logging program, such as the cost of fire suppression and the costs of externalized damages associated with logging such as loss of recreational opportunities, soil erosion, degradation of water quality, loss of game and non-game wildlife and fish species, and a reduction in scenic values.

SUPPLEMENTAL ANALYSIS: BLM AND FIRE SUPPRESSION COSTS

This section provides two supplemental figures expanding on the JMP-based analysis presented in Table 1 – net taxpayer losses from the Bureau of Land Management (BLM) timber sale program and the share of federal firefighting expenditures attributable to the logging program on both national forests and BLM lands. These figures provide

Table 1: Net taxpayer losses from the national forest logging program FY 2013 to FY 2017

Appropriated funds

	2013	2014	2015	2016	2017
Timber sales/ forest products management	$291,802,535	$305,954,877	$305,592,147	$320,183,363	$312,908,622
Vegetation and watershed management	$185,587,235	$195,929,139	$195,696,851	$193,258,866	$188,868,073
Reforestation trust fund	$31,396,325	$31,821,143	$33,902,311	$31,387,460	$32,781,593
Hazardous fuels reduction	$126,866,002	$137,337,660	$110,733,380	$134,740,552	$127,409,850
Forest health management - federal lands	$26,998,370	$33,186,413	$34,412,260	$34,028,152	$38,060,456
Forest roads program	$51,275,097	$53,761,925	$53,698,186	$56,262,133	$54,983,826
Road maintenance	$134,799,344	$129,904,714	$136,762,134	$131,227,742	$130,145,574
LRMP, inventory and monitoring	$122,576,624	$120,619,942	$120,476,939	$118,976,041	$116,494,299
Landownership management	$52,115,048	$47,720,915	$50,944,524	$46,175,107	$46,200,489
Timber research	$97,056,218	$99,050,731	$98,933,300	$93,377,692	$90,844,964
Support from other budget line items	$40,560,552	$42,527,728	$42,477,308	$44,505,487	$43,494,298
Total appropriated for logging	**$1,161,033,350**	**$1,197,815,187**	**$1,183,629,340**	**$1,204,122,595**	**$1,182,192,045**

Off budget expenditures for logging

Purchaser credit roads	$614,409	$1,431,951	$150,442	$1,046,249	$544,994
Timber pipeline restoration fund	$3,318,620	$5,990,330	$3,582,786	$3,923,432	$10,756,460
Salvage sales fund	$25,046,349	$21,683,987	$31,387,183	$31,387,460	$46,099,115
Brush disposal fund	$7,102,358	$9,018,112	$9,667,456	$9,416,238	$20,488,495
Cooperative work trust funds	$297,367,722	$182,519,709	$316,669,831	$119,167,721	$121,906,548
Total off budget expenditures for logging	**$333,449,458**	**$220,644,090**	**$361,457,697**	**$164,941,100**	**$199,795,612**

Net taxpayer losses

Total expenditures for logging	$1,494,482,808	$1,418,459,277	$1,545,087,038	$1,369,063,694	$1,381,987,657
Timber sales receipts deposited in NFF	$34,475,793	$31,863,570	$31,708,209	$31,342,123	$34,035,957
Total net taxpayer losses	**$1,460,007,015**	**$1,386,595,707**	**$1,513,378,829**	**$1,337,721,571**	**$1,347,951,701**

Table 2: Net taxpayer losses - supplemental

Net taxpayer losses - Forest Service, BLM, and fire suppression related to the timber sale program

	2013	2014	2015	2016	2017
Net taxpayer losses - Forest Service (Table 1)	$1,460,007,015	$1,386,595,707	$1,513,378,829	$1,337,721,571	$1,347,951,701
Net taxpayer losses reported by BLM	$68,506,429	$61,264,247	$55,696,240	$48,902,517	$51,010,058
Fire suppression costs related to timber	$273,408,284	$241,183,086	$252,759,685	$259,625,345	$412,593,398
Total net taxpayer losses (supplemental):	**$1,801,921,727**	**$1,689,043,039**	**$1,821,834,754**	**$1,646,249,432**	**$1,811,555,157**

a more expansive estimate of federal logging subsidies; however, the methods have not been peer reviewed or made consistent with the CRS-reviewed JMP methodology, so they should be considered experimental and supplemental to those presented in Table 1.

"The Forest Service is not the only federal agency that manages a logging program. The BLM also supplies timber to private industry, primarily from lands in western Oregon that were formerly granted to the Oregon and California Railroad company but reclaimed by the federal government in 1937. During the FY 2013 to FY 2017 period, the volume of wood extracted from these lands ranged between 200 and 260 million board feet per year."

Taxpayer losses from BLM's logging program

The Forest Service is not the only federal agency that manages a logging program. The BLM also supplies timber to private industry, primarily from lands in western Oregon that were formerly granted to the Oregon and California Railroad company but reclaimed by the federal government in 1937. During the FY 2013 to FY 2017 period, the volume of wood extracted from these lands ranged between 200 and 260 million board feet per year. As with national forest logging projects, the logging program on BLM lands is routinely challenged for its environmental harms, mainly because remnant old growth forests continue to be logged.[21]

The BLM maintains its books in a different manner than the Forest Service, and so it would take quite a bit of cross walking between various expense and revenue categories to make the estimates comparable. Nonetheless, the BLM does maintain its books in a way that facilitates a fairly easy, first pass assessment of net taxpayer costs.

During the FY 2013 to FY 2017 period, the BLM allocated funding for its timber sale program through four separate accounts under the broad category of Western Oregon Resources Management. According to BLM Budget Justifications, "[a]ll of the budget activities provide direct or indirect support for the development or implementation of sustained yield timber production" so it is reasonable to assign all the costs in these accounts to the timber sale program. Timber receipts are tracked closely, but the amount deposited in the US Treasury is not reported.

"In each year of the analysis, BLM reports net taxpayer losses in the range of $50 million to $70 million per year in 2018 dollars, which translates into a subsidy of $200 to $300 per thousand board feet logged."

Table 2 reports the net effect for each fiscal year. In each year of the analysis, BLM reports net taxpayer losses in the range of $50 million to $70 million per year in 2018 dollars, which translates into a subsidy of $200 to $300 per thousand board feet logged. These estimates are conservative, however, because they assume that timber sale receipts are not recycled back into planning for additional timber sales and that the four core accounts through which BLM tracks timber sale program expenses are comprehensive. Neither of these assumptions is likely to be true; however, without a detailed analysis such as we completed for the Forest Service we cannot refine these reported losses any further at this time. Nevertheless, it is clear that the BLM, like the Forest Service, provides a hefty subsidy for a logging program that is regularly challenged for its deleterious effects on climate, biodiversity, water, climate and other public trust resources.

Fire suppression expenditures attributable to the federal logging program

As noted above, the estimates in Table 1 do not include many other expenditures made necessary by the national forest logging program. One of the key expenses involves fire suppression. Each year, the Forest Service and various Department of Interior agencies suppress fires on millions of acres of forests and rangelands. Since 2000, suppression activities have been implemented on a low of 3.4 million acres in 2010 to a high of 10.1 million acres in 2015. Since 2000, fire suppression expenditures on these federal lands has ranged between one and three billion per year.[22]

On forested lands, much of this expense can be attributable to logging activities for three primary reasons. First, many fire suppression activities are carried out to protect timber resources for future timber sales. In past justifications for its firefighting budget, the Forest Service conceded this point.[23] Secondly, many suppression activities are implemented because of past logging practices that have left national forestlands more susceptible to fire. For many fire adapted forest ecosystems in the western United States, logging has created hotter, drier, and more homogenous forest conditions that – whether justified ecologically or not – prompt federal forestland managers to suppress fires rather than let them burn.[24] Third, the vast majority of ignitions are human caused and occur along roads, and many of those roads were built and are now maintained to accommodate logging projects.

"On forested lands, much of this firefighting expense can be attributable to logging activities for three primary reasons. First, many fire suppression activities are carried out to protect timber resources for future timber sales. In past justifications for its firefighting budget, the Forest Service conceded this point. Secondly, many suppression activities are implemented because of past logging practices that have left national forestlands more susceptible to fire."

In order to estimate the share of federal wildland fire suppression expenditures attributable to the federal logging program we partnered with Geos Institute for a GIS analysis of wildfires and fire suppression activities and costs during the 2012 to 2017 period. The basic method was to estimate the acreage on which fire suppression activities were likely related to past and planned logging, and then apply a per-acre cost for firefighting in a given fiscal year.

In particular, from the total area of wildland fires delineated by federal agencies for suppression and other management responses in each year, we removed fires that occurred in three areas: (a) non-forested areas; (b) protected areas, such as designated wilderness, national parks and national monuments, and (c) the wildland-urban interface. The reason for removing these acres is because fire suppression activities here are unlikely to have been related to past or planned logging. The deduction for non-forest areas is self-explanatory. The deduction of protected acres is made because these lands have had little or no past logging activities, and future logging is prohibited by law. The deduction for acres in wildland-urban interface areas is made because suppression activities here have an overriding purpose of saving lives and structures, not protecting the timber resource.

To the residual suppression acres – unprotected forestlands outside the wildland urban interface (WUI) – we applied a per-acre cost figure (updated to $2018) for nationwide fire suppression activities on federal lands reported by federal agencies each year. The results are reported in Table 2. So, for example, in FY 2016, federal agencies reported 67,595 individual fires necessitating fire suppression activities on 5.5 million acres of land at a cost of $1.98 billion. Of these acres, about 12.28% were forested lands outside of protected areas or WUIs. Multiplying these acres (675,998) by the national per-acre firefighting cost of $380.75 implies that over $257 million can reasonably be attributable to past and planned logging activities. While his method is no substitute for a fire-by-fire analysis, a comprehensive analysis of federal logging subsidies would be remiss not to include this line item.

As reported in Table 2, adding BLM losses and these timber sale program related fire suppression costs to the tally pushes our estimate of taxpayer losses from the federal logging program up into the range of $1.7 billion to $1.8 billion each year during the FY 2013 to FY 2017 period.

CONCLUDING THOUGHTS

"Selling timber and other natural resources below cost is one of the classic forms of environmentally harmful subsidies (EHS) opposed by international institutions such as the OECD, IMF, European Union and others."

"Eliminating subsidized logging activities on federal public lands would not only free up taxpayer dollars for use on more socially productive programs but would also reduce environmental costs and make markets and trade more efficient."

Each year, the Forest Service authorizes enough logging on national forest lands to fill over 650,000 log trucks. Most of these commercial logging projects are contested on ecological grounds for their harmful impacts to wildlife, fish, water and increasingly challenged because they represent significant sources of carbon emissions and reduce the ability of the land to adapt to climate change by increasing fire risk, water shortages, and susceptibility to insects and disease. US taxpayers heavily subsidize these projects. As demonstrated in this analysis and previous analyses by Hanson (1999) and Voss (2005), the Forest Service sells this timber far below cost – losses that range between $1.4 and $1.8 billion per year.

Selling timber and other natural resources below cost is one of the classic forms of environmentally harmful subsidies (EHS) opposed by international institutions such as the OECD, IMF, European Union and others.[25] As an OECD member, the US has stated its support for eliminating these subsidies as well. Environmental harmful subsidies distort markets by causing overproduction of a resource – in this case timber – that is connected to one or more adverse impacts and by generating negative externalities that are passed on to the public rather than being absorbed as a cost of doing business.[26] They also distort free trade by creating unfair competition with countries that don't subsidize their timber. As such, eliminating subsidized logging activities on federal public lands would not only free up taxpayer dollars for use on more socially productive programs but would also reduce environmental costs and make markets and trade more efficient.

Congress has at least two options for doing so. The first option is to continue to offer timber for sale from national forests and BLM lands but require that all projects with a significant commercial timber component pay for themselves by ensuring that minimum bid prices reflect all direct and indirect costs to agencies. The second option is to recognize the unique role federal forests play in the forested landscape of the US and do away with the commercial timber sale program on these lands entirely. However, funding for restoration activities that have been linked with commercial logging should continue. Decoupling funding for these restoration activities from commercial logging will greatly bolster their integrity by allowing project managers to focus cleanly on ecological goals.

Offering below cost timber from federal public lands is just one form of environmentally harmful subsidy supporting the US timber industry. There

are many other types of federal subsidies to consider, as well as those implemented by state governments. In the next report in this series, we will consider state-level subsidies by examining a wide range of tax breaks and expenditures made by the State of Oregon but mimicked in many other states where industrial forest practices prevail.

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Severe fire weather and intensive forest management increase fire severity in a multi-ownership landscape

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Abstract. Many studies have examined how fuels, topography, climate, and fire weather influence fire severity. Less is known about how different forest management practices influence fire severity in multi-owner landscapes, despite costly and controversial suppression of wildfires that do not acknowledge ownership boundaries. In 2013, the Douglas Complex burned over 19,000 ha of Oregon & California Railroad (O&C) lands in Southwestern Oregon, USA. O&C lands are composed of a checkerboard of private industrial and federal forestland (Bureau of Land Management, BLM) with contrasting management objectives, providing a unique experimental landscape to understand how different management practices influence wildfire severity. Leveraging Landsat based estimates of fire severity (Relative differenced Normalized Burn Ratio, RdNBR) and geospatial data on fire progression, weather, topography, pre-fire forest conditions, and land ownership, we asked (1) what is the relative importance of different variables driving fire severity, and (2) is intensive plantation forestry associated with higher fire severity? Using Random Forest ensemble machine learning, we found daily fire weather was the most important predictor of fire severity, followed by stand age and ownership, followed by topographic features. Estimates of pre-fire forest biomass were not an important predictor of fire severity. Adjusting for all other predictor variables in a general least squares model incorporating spatial autocorrelation, mean predicted RdNBR was higher on private industrial forests (RdNBR 521.85 ± 18.67 [mean ± SE]) vs. BLM forests (398.87 ± 18.23) with a much greater proportion of older forests. Our findings suggest intensive plantation forestry characterized by young forests and spatially homogenized fuels, rather than pre-fire biomass, were significant drivers of wildfire severity. This has implications for perceptions of wildfire risk, shared fire management responsibilities, and developing fire resilience for multiple objectives in multi-owner landscapes.

Key words: fire severity; forest management; Landsat; multi-owner landscape; Oregon; plantation forestry; RdNBR.

Introduction

The wildfire environment has become increasingly complicated, due to the unanticipated consequences of historical forest management and fire exclusion (Weaver 1943, Hessburg et al. 2005, Fulé et al. 2009, Naficy et al. 2010, Merschel et al. 2014), an increasingly populated wildland urban interface (Haas et al. 2013), and a rapidly changing climate (Westerling and Bryant 2008, Littell et al. 2009, Jolly et al. 2015). These factors are resulting in more intense fire behavior and increasingly negative ecological and social consequences (Williams 2013, Stephens et al. 2014). Fuels reduction via mechanical thinning and prescribed burning have been the dominant land management response for mitigating these conditions (Agee and Skinner 2005, Stephens et al. 2012), although there is an increasing recognition of the need to manage wildfires more holistically to meet social and ecological objectives. (North et al. 2015a, b). However, overcoming these challenges is inhibited by numerous disagreements in the scientific literature regarding historical fire regimes and appropriate policies and management of contemporary fire-prone forests (Hurteau et al. 2008, Hanson et al. 2009, Spies et al. 2010, Campbell et al. 2012, Odion et al. 2014, Collins et al. 2015, Stevens et al. 2016). These factors and others have resulted in a nearly intractable socioecological problem (Fischer et al. 2016); one that is compounded by the fact that many fire-prone landscapes consist of multiple owners and administrative jurisdictions with varying and often conflicting land management objectives.

Developing and prioritizing landscape fire management activities (i.e., thinning, prescribed fire, wildland fire use, and fire suppression) across jurisdictional and ownership boundaries requires landscape-scale assessments of the factors driving fire severity (i.e., the fire behavior triangle of fuels, topography, and weather). Researchers have focused on the influence of bottom-up drivers such as topography (Dillon et al. 2011, Prichard and Kennedy 2014, Birch et al. 2015), and fuels via fuel reduction effects (Agee and Skinner 2005, Raymond and Peterson 2005, Safford et al. 2009, Prichard and Kennedy 2014, Ziegler et al. 2017), as well as the top-down influence of weather on fire severity (Birch et al. 2015, Estes et al. 2017). They have also focused more broadly on how fire severity varies with vegetation and forest type (Birch et al. 2015, Steel et al. 2015, Reilly et al. 2017) and climate (Miller et al. 2012, Abatzoglou et al. 2017). While there is substantial value in further describing how components of the fire behavior triangle influence fire severity, we believe there is a need to account for these known influences on fire behavior and effects to understand

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how different management regimes interact with these controlling factors, so appropriate landscape management strategies can be developed to support social-ecological resilience in fire-prone landscapes (Spies et al. 2014, Schoennagel et al. 2017).

Understanding the relationships between forest management regimes and fire severity is especially important in multi-owner landscapes, where wildfire governance systems concerned about short-term property loss and public safety can reinforce perceptions of wildfire risk and hazard, resulting in individual property owners being less likely to make management decisions that reduce long-term risk exposure (McCaffrey 2004, Fischer et al. 2016). This is particularly important in landscapes that include intensive plantation forestry, a common and rapidly expanding component of forest landscapes at regional, national, and global scales (Cohen et al. 1995, Landram 1996, Del Lungo et al. 2001, Rudel 2009, FAO 2010, Nahuelhual et al. 2012). Researchers have hypothesized that intensive forest management reduces fire behavior and effects (Hirsch et al. 2001, Rodríguez y Silva et al. 2014). However empirical results have been mixed, with evidence that intensive forest management can either reduce (Lyons-Tinsley and Peterson 2012, Prichard and Kennedy 2014) or increase fire severity (Odion et al. 2004, Thompson et al. 2007), and that reduced levels of forest legal protection (a proxy for more active management) have been associated with increased fire severity in the western U.S. (Bradley et al. 2016). These conflicting results further complicate the development of fire governance and management strategies for increasing social-ecological resilience in a rapidly changing fire environment.

The quality, spatial scale, and spatial correlation of explanatory data (i.e., weather, topography, and fuels) are major limitations to empirically understanding how forest management activities influence fire severity across landscapes. Regional studies of fire severity often rely on spatially coarse climatic data (Dillon et al. 2011, Miller et al. 2012, Cansler and McKenzie 2014, Kane et al. 2015, Harvey et al. 2016, Meigs et al. 2016, Reilly et al. 2017), rather than local fire weather that can be a significant driver of fire area and severity (Flannigan et al. 1988, Bradstock et al. 2010, Estes et al. 2017). This is in part because finer-scale fire weather variables are often incomplete across the large spatial and temporal domains of interest. Additionally, regional studies often occur in areas with large elevation relief resulting in strong climatic gradients, while more local studies often have less elevation relief and potentially weaker climatic gradients. Perhaps more importantly, the geographic distribution of different ownership types and management regimes can confound quantification of the drivers of fire severity. For example, high elevation forests in the Pacific Northwest region of the United States are largely unmanaged as National Parks and congressionally designated wilderness areas, compared to intensively managed forests at lower elevations, resulting in differences in topography, weather, climate, forest composition, productivity, and historical fire regimes between ownerships and management regimes. While landscape studies of fire severity and management activities have used a variety of statistical techniques to account for spatial correlation of both response and predictor variables (Thompson et al. 2007, Prichard and Kennedy 2014, Meigs et al. 2016), these techniques may not overcome fundamental differences in response and predictor variables between management and/or ownership types.

In this study, we examined the drivers of fire severity within one large (~20,000 ha) wildfire complex that burned within the Klamath Mountains, an ecoregion with a mild Mediterranean climate of hot dry summers and wet winters in southwestern Oregon, USA. The fire burned within a checkerboard landscape of federal and private industrial forestry ownership. This spatial pattern of contrasting ownership and management regimes provided a unique landscape experiment where we quantified the effects of management regimes after accounting for variation in well-known drivers of fire behavior and effects. Leveraging geospatial data on fire severity, fire progression, fire weather, topography, prefire forest conditions, and past management activities, we asked two questions: (1) What is the relative importance of different variables driving fire severity? And (2) is intensive plantation forestry associated with higher fire severity?

Methods

Study site

In the summer of 2013, the Douglas Complex burned 19,760 ha of forestland in southwestern Oregon, USA (Fig. 1). Starting from multiple lightning ignitions, individual small fires coalesced into two large fires (Dads Creek and Rabbit Mountain) managed as the Douglas Complex.

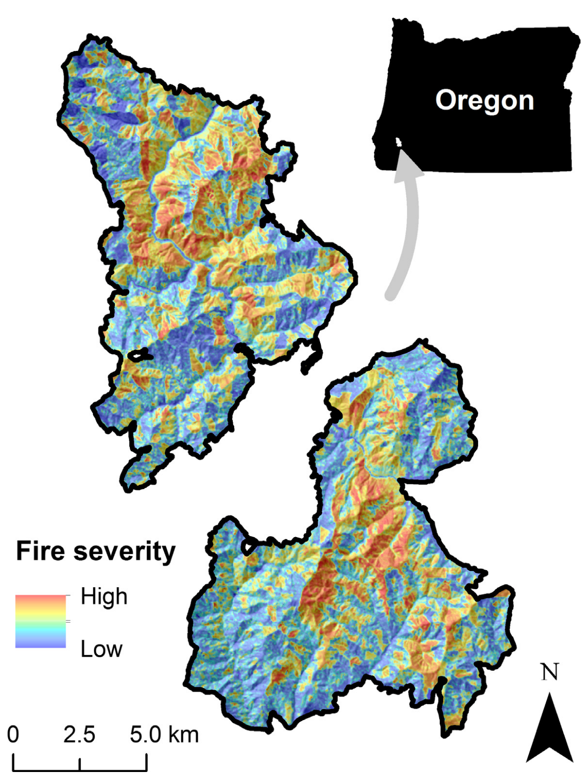

Fig. 1. Location of and fire severity within the Douglas Complex in Oregon, USA. Fire severity quantified using the Relative differenced Normalized Burn Ratio (RdNBR).

This fire burned within the Oregon and California Railroad Lands (hereafter O&C Lands). O&C Lands resulted from 19th century land grants that ceded every other square mile (259 ha) of federally held land to railroad companies along planned routes in Oregon and California to incentivize railroad development and homesteading settlement. The Oregon and California Railroad Company received a total of 1.5 million ha, but failing to meet contractual obligations, 1.1 million ha were transferred back to federal ownership under the Chamberlain-Ferris Revestment Act of 1916. The USDI Bureau of Land Management (BLM) is currently required to manage these lands for sustainable timber production, watershed protection, recreation, and wildlife habitat. Private industrial forestlands dominate the remaining O&C landscape, and are managed intensively as native tree plantations (primarily Douglas-fir, *Pseudotsuga menziesii* var. *menziesii*) for timber production typically on 30–50 yr harvest rotations. The Douglas Complex fires burned 10,201.64 ha of forests managed by the BLM, 9,429.66 ha of private industrial forests, and 129.33 ha managed by the Oregon Department of Forestry (ODF).

The Douglas Complex burned at elevations ranging from 213 to 1,188 m in mountainous terrain of the Klamath Mountains Ecoregion. Climate in the ecoregion is characterized by hot dry summers and wet winters, with greater winter precipitation at higher elevations and western portions of the ecoregion. Vegetation types within the region include oak woodlands and mixed hardwood/evergreen forests at low to mid elevations, transitioning into mixed-conifer forests at higher elevations (Franklin and Dyrness 1988). Forests within the Douglas Complex are dominated by Douglas-fir, ponderosa pine (*Pinus ponderosa*), and white fir (*Abies concolor*). Other conifer tree species present include incense cedar (*Calocedrus decurrens*), sugar pine (*Pinus lambertiana*), Jeffery pine (*Pinus jefferyi*), and knobcone pine (*Pinus attenuata*). Hardwood species include Oregon white oak (*Quercus garryana*), big-leaf maple (*Acer macrophyllum*), Pacific dogwood (*Cornus nuttallii*), Pacific madrone (*Arbutus menziesii*), canyon live oak (*Quercus chrysolepis*), California black oak (*Quercus kelloggii*), golden chinkapin (*Chrysolepis chrysophylla*), and tanoak (*Lithocarpus densiflourus*). Douglas-fir is the primary commercial timber species managed on private and public lands, while fire exclusion and historical management practices have expanded the density and dominance of Douglas-fir across much of the ecoregion (Franklin and Johnson 2012, Sensenig et al. 2013).

Data sources

We analyzed fire severity in relation to eight predictor variables representing topography, weather, forest ownership, forest age, and pre-fire forest biomass (Fig. 2). We quantified fire severity using the Relative differenced Normalized Burn Ratio (RdNBR), a satellite-imagery-based metric of pre- to post-fire change. Cloud-free pre-fire (3 July 2013) and post-fire (7 July 2014) images came from the Landsat 8 Operational Land Imager. Normalized Burn Ratio (NBR), which combines near-infrared and mid-infrared bands of Landsat imagery, was calculated for pre- and post-fire images. Differenced Normalized Burn Ratio (dNBR) was calculated by subtracting $NBR_{\text{post-fire}}$ from $NBR_{\text{pre-fire}}$ values, and RdNBR was then calculated following Miller et al. (2009), where:

$$\text{RdNBR} = \frac{\text{dNBR}}{\sqrt{\text{Absolute Value}\,(\text{NBR}_{\text{pre-fire}}/1,000)}}. \quad (1)$$

We chose RdNBR over dNBR as our fire severity metric because RdNBR removes, at least in part, the biasing effect of pre-fire conditions, improving assessment of burn severity across heterogeneous vegetation and variable pre-fire disturbances (Miller and Thode 2007). We used the continuous RdNBR values as our response variable for fire severity at a 30-m resolution.

Elevation and other topographic variables were derived from the National Elevation Dataset 30 m digital elevation model (Gesch et al. 2002). We generated 30-m rasters of elevation (m), slope (%), topographic position index (TPI), and heat load ($\text{MJ}\cdot\text{cm}^{-2}\cdot\text{yr}^{-1}$). TPI was calculated as the difference between elevation in a given cell and mean elevation of cells within an annulus around that cell, calculated at fine and coarse scales (TPI fine and TPI coarse) with 150–300 m and 1,850–2,000 m annuli, respectively. We also originally considered TPI at a moderate spatial scale (850–1,000 m annuli), but rejected it as an predictor variable due to its high correlation to TPI fine (r = 0.64) and TPI course (r = 0.84). TPI course had strong linear correlations with elevation (r = 0.83) and TPI fine (r = 0.46), so it was also removed to avoid multi-collinearity in statistical analyses. Heat load was calculated by least-squares multiple regression using trigonometric functions of slope, aspect, and latitude following McCune and Keon (2002).

Rasters of daily fire weather conditions were generated by extrapolating weather station data to a daily fire progression map. We obtained hourly weather data for the duration of active fire spread (7 July–20 August 2013) from the Calvert Peak Remote Automatic Weather Station (NWS ID 352919; 42°46′40″ N 123°43′46″ W, 1,165 m), approximately 30 km west-southwest of the Douglas Complex. We then subset each 24-h period of weather data to the daily burn period (10:00 to 18:00) when fire behavior is typically most active. We then calculated the daily burn period minimum wind speed (km/h), maximum temperature (°C), and minimum relative humidity (%). For each daily burn period we also calculated the mean energy release component (ERC), spread component (SC), and burning index (BI) using FireFamilyPlus Version 4.1 (Bradshaw and McCormick 2000). ERC is an index of fuel dryness related to the maximum energy release at the flaming front of a fire, as measured from temperature, relative humidity, and moisture of 1–1,000 h dead fuels. SC is a rating of the forward rate of spread of a head fire, and is calculated from wind speed, slope, and moisture of live fine and woody fuels (Bradshaw et al. 1983). BI is proportional to the flame length at the head of a fire (Bradshaw et al. 1983), calculated using ERC and SC, thus incorporating wind speed and providing more information than ERC and SC individually. ERC, SC, and BI vary by broadly categorized fuel types. We calculated ERC, SC, and BI using the National Fire Danger Rating System Fuel Model G, which represents short-needled

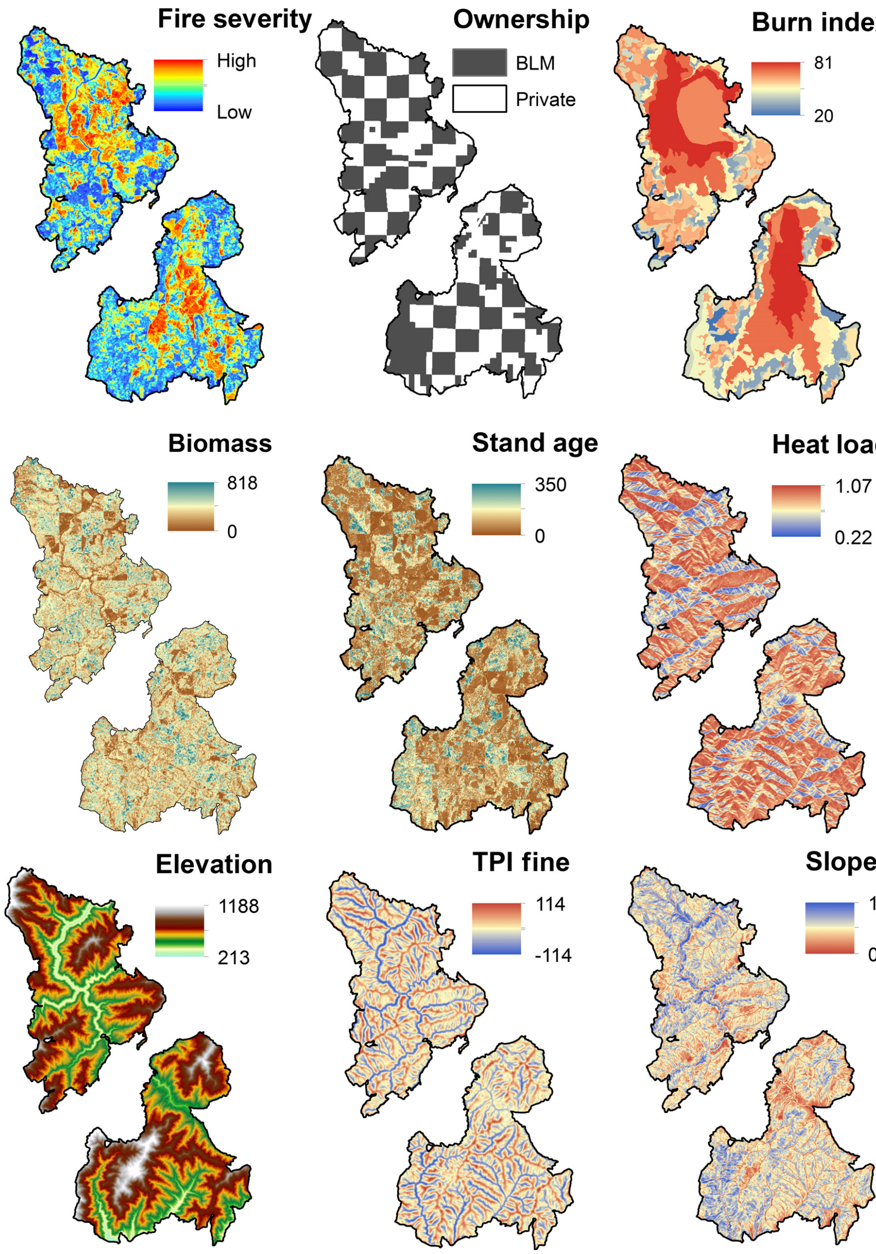

FIG. 2. Maps of response and predictor variables for Douglas Complex. TPI, topographic position index.

conifer stands with heavy dead fuel loads. Daily fire weather variables were then spatially extrapolated to the daily area burned based on daily fire progression geospatial data captured during the fire (GeoMAC 2013).

Forest ownership was derived from geospatial data representing fee land title and ownership in Oregon (Oregon Spatial Data Library 2015). We grouped ODF and BLM lands as a single ownership type, because ODF lands were a small component of the area burned and have management objectives closer to federal vs. private industrial forests (Spies et al. 2007). Pre-fire forest conditions were represented with 30-m rasters of live biomass (Mg/ha) and stand age, derived from a regional 2012 map of forest composition and structural attributes developed for the Northwest Forest Plan Monitoring Program (Ohmann et al. 2012, Davis et al. 2015). These maps were developed using the gradient nearest neighbor method (GNN), relating multivariate response variables of forest composition and structure attributes from approximately 17,000 federal forest inventory plots to gridded predictor variables (satellite imagery, topography, climate, etc.) using canonical correspondence analysis and nearest neighbor imputation (Ohmann and Gregory 2002). Biomass values are directly from the GNN maps, while we quantified forest age as a two-step process. First, we calculated pre-fire forest age in 2013 based on years since each pixel was disturbed in the Landsat time series (1985–2014) from a regional disturbance map generated for the Northwest Forest Plan Monitoring Program using the LandTrendr segmentation algorithm (Kennedy et al. 2010, Ohmann et al. 2012, Davis et al. 2015). Second, for pixels where no disturbance had occurred within the Landsat time series, we amended forest age derived from the Landsat time series using dominant and codominant tree age from the GNN maps.

Statistical analyses

All statistical analyses were conducted in the R statistical environment version 3.3.3 (R Development Core Team 2017). We sampled the burned landscape using a spatially constrained stratified random design, from which response and predictor variables were extracted for analysis. Sample points had to be at least 200 m apart to minimize short distance spatial autocorrelation of response and predictor variables. Our choice of minimum inter-plot distance to reduce spatial autocorrelation was confounded by the dominance of long distance spatial autocorrelation driven by large ownership patches, which would have greatly reduced sample size and potentially eliminated finer scale variability in the sample. For these reasons we based our 200 m minimum inter-plot distance in part on prior research (Kane et al. 2015), that found residual spatial autocorrelation in Random Forest models of fire severity in the Rim Fire of 2013 in the California Sierra Nevada was greatly diminished when inter-plot distances were at least 180 m apart. Additionally, point locations had to be at least 100 m away from ownership boundaries to minimize inter-ownership edge effects. Within these spatial constraints, sample points were located in a stratified random design, with the number of points proportional to area of ownership within the fire perimeter, resulting in 571 and 519 points located in BLM and private industrial forests, respectively. Mean response and predictor variables were extracted within a 90 × 90 m plot (e.g., 3 × 3 pixels) centered on each sample point location to minimize the effects of potential georeferencing errors across data layers and maintain a plot size comparable to the original inventory plots used as source data in GNN maps as recommended by Bell et al. (2015).

We observed high correlation between fire weather variables (mean absolute $r = 0.59$), likely due to their temporal autocorrelation during the fire event, which could result in multi-collinearity in statistical analyses. Therefore, we evaluated the relationships between each fire weather variable and daily mean fire severity, selecting a single fire weather variable as a predictor variable in subsequent analyses. We based our variable selection on visual relationships to daily RdNBR, variance explained in regressions of RdNBR and fire weather variables, and Akaike information criterion (AIC) scores of regressions of RdNBR and fire weather variables following Burnham and Anderson (2002).

The study's strength rests in part on the implicit assumption that the checkerboard spatial allocation of ownership types is a landscape scale experiment, where predictor variables directly modified by management activities (e.g., pre-fire biomass and forest age) are different between ownership types, but fire weather and topographic variables are not. We assessed this assumption by visualizing data distributions between ownerships using boxplots and violin plots, and testing if variables were different between ownership types using Mann–Whitney–Wilcoxon Tests.

To assess the relative importance and relationships between predictor variables and RdNBR, we used Random Forest (RF) supervised machine learning algorithm with the randomForest package (Liaw and Wiener 2002). As applied in this study, RF selected 1,500 bootstrap samples, each containing two-thirds of the sampled cells. For each sample, RF generated a regression tree, then randomly selected only one-third of the predictor variables and chose the best partition from among those variables. To assess the relative importance and relationships of predictor variables on RdNBR across the entire study area and within different ownerships, separate RF models were developed for all 1,090 sample plots across the entire burned area, as well as separately for plots on BLM and private industrial lands. For each of the three RF models, we calculated variable importance values for each predictor variable as the percent increase in the mean squared error (MSE) in the predicted data when values for that predictor were permuted and all other predictors were left unaltered. In addition to variable importance values, we determined which predictor variables should be retained in each RF model using multi-stage variable selection procedures (Genuer et al. 2010). We applied two-stage variable selection for interpretation to each RF model using the VSURF package (Genuer et al. 2016). Final RF models were then run including only the selected variables. Predictive power of the final RF models were assessed by calculating the variance explained, which is equivalent to the coefficient of determination (R^2) used with linear regressions to assess statistical model fit for a given dataset. Last, we visualized the relationships of individual predictor variables on RdNBR in the final RF models using partial dependency plots (Hastie et al. 2001).

Importance values in RF models are not the same as quantifying the fixed effects of predictor variables, nor is RF well suited to explicitly test hypotheses or quantify effects of predictor variables while accounting for other variables in a model. To test if ownership type increased RdNBR, we developed a generalized least squares (GLS) regression model with an exponential spherical spatial correlation structure using the nlme package (Pinheiro et al. 2017). The GLS regression used the distance between sample locations and the form of the correlation structure to derive a variance–covariance matrix, which was then used to solve a weighted OLS regression (Dormann et al. 2007). Using the same response and predictor data as in the RF model for the entire Douglas Complex, and a binary predictor variable for ownership type, we developed a GLS model from which we calculated the fixed effect of ownership on RdNBR. We then predicted the mean and standard error of RdNBR by ownership after accounting for the other predictor variables in the GLS model using the AICcmodavg package (Mazerolle 2017).

Results

Fire weather variables

Regression models of fire weather variables (except maximum temperature) described a significant proportion of the variance in daily mean RdNBR (Table 1; Appendix S1: Fig. S1). SC described the most variance in daily RdNBR, had the lowest AIC score, and was most likely to be the best model of those compared (w_i = 0.8250). However, BI described a comparable amount of the variance in daily RdNBR (R^2 = 0.5815), had a substantial level of empirical support (ΔAIC = 3.3816), was the second most likely model given the data (w_i = 0.1521), and contained additional metrics that influence fire behavior (influence of temperature, relative humidity, and drought on live and dead fuels) not incorporated in SC. For these reasons, we choose to use BI as the single fire weather variable in subsequent analyses, acknowledging that it may describe slightly less variation in RdNBR than SC.

Table 1. Regression models of daily mean Relative differenced Normalized Burn Ratio (RdNBR) in relation to daily burn period fire weather variables.

Models	R^2	AIC	ΔAIC	$L(g_i\|x)$	w_i
RdNBR = SC^2	0.6532	210.0324	0.0000	1.0000	0.8250
RdNBR = BI^2	0.5815	213.4140	3.3816	0.1844	0.1521
RdNBR = min wind speed^2	0.4542	218.1948	8.1624	0.0169	0.0139
RdNBR = log (min relative RH)	0.3800	220.4903	10.4579	0.0054	0.0044
RdNBR = ERC^2	0.3675	220.8497	10.8173	0.0045	0.0037
RdNBR = max wind speed^2	0.2179	224.6700	14.6376	0.0007	0.0005
RdNBR = max temperature^2	0.1069	227.0592	17.0268	0.0002	0.0002
RdNBR = null model	0.0000	228.1855	18.1531	0.0001	0.0001

Notes: R^2, adjusted R squared; AIC$_c$, Akaike information criterion corrected for sample size; ΔAIC$_c$, AIC$_c$ differences; $L(g_i|x)$, likelihood of a model given the data; w_i, Akaike weights; SC, spread component; BI, burn index; RH, relative humidity; ERC, energy release component.

RdNBR and predictor variable differences by ownership

The majority of predictor variables were not statistically different by ownership, as expected given the spatial distribution of ownership. Based on Mann-Whitney-Wilcoxon tests, biomass and stand age were lower on private industrial vs. BLM managed lands (Table 2; Appendix S1: Fig. S2). TPI fine, heat load, slope, and BI were not different between ownership types. Elevation was different between ownership types, but only 44 m higher on BLM land across a range of 875 m for all sample plots. Mean RdNBR was higher (536.56 vs. 408.75) on private industrial vs. BLM lands.

Random forest variable importance values and partial dependency plots

Two-stage variable selection procedures retained seven, five, and six predictor variables in the final RF models for the entire Douglas Complex, BLM, and private forests, respectively (Fig. 3). Across the entire Douglas Complex, BI was the most important predictor variable of RdNBR (increasing MSE by 138.4%), while BI was also the most importance variable separately for BLM (105.4%) and private forests (83.2%). Age and ownership were the next most

Table 2. RdNBR (mean with SE in parentheses) and predictor variables on sampled plots for Bureau of Land Management (BLM) vs. private industrial (PI) ownership.

Variable	BLM	PI	w	P
RdNBR	408.75 (298.53)	536.56 (299.88)	111,124	<0.0001
Biomass (Mg/ha)	234.75 (87.24)	163.88 (74.47)	215,166	<0.0001
Age (yr)	108.81 (55.53)	52.18 (36.78)	236,021.5	<0.0001
BI (index)	62.99 (14.16)	63.64 (14.54)	142,575.5	0.2782
Elevation (m)	653.79 (153.48)	609.46 (161.62)	171,200	<0.0001
TPI fine	0.55 (32.51)	−1.08 (32.12)	152,275	0.4296
Heat load (MJ·cm^{-2}·yr^{-1})	0.77 (0.2)	0.77 (0.2)	150,363	0.6734
Slope (%)	48.4 (13.4)	47.05 (14.01)	156,435	0.1115

Notes: The w values and associated P values are from Mann–Whitney–Wilcoxon tests. TPI, topographic position index.

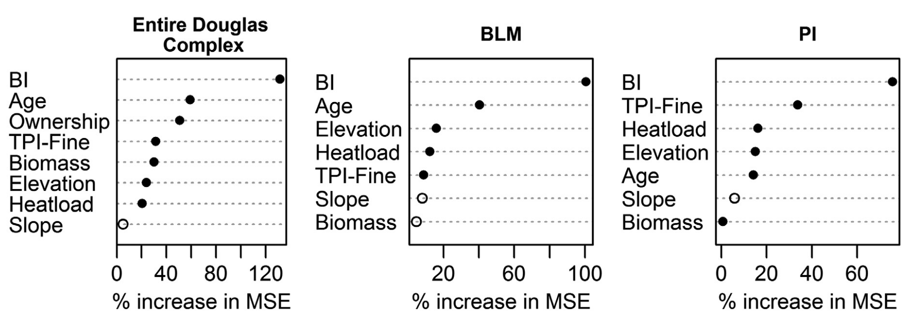

Fig. 3. Variable importance plots for predictor variables from Random Forest (RF) models of RdNBR for 1090 sample plots across the entire Douglas Complex (left panel), 571 plots on Bureau of Land Management (BLM) forests (middle), and 519 plots on private industrial (PI) forests (right). Solid circles denote variables retained in two-stage variable selection, open circles denote variables removed from the final RF models during variable selection. BI, burning index; MSE, Mean Squared Error. [Correction added on May 1st 2018, after first online publication: The x axis label was incorrectly labeled as "MSF"]

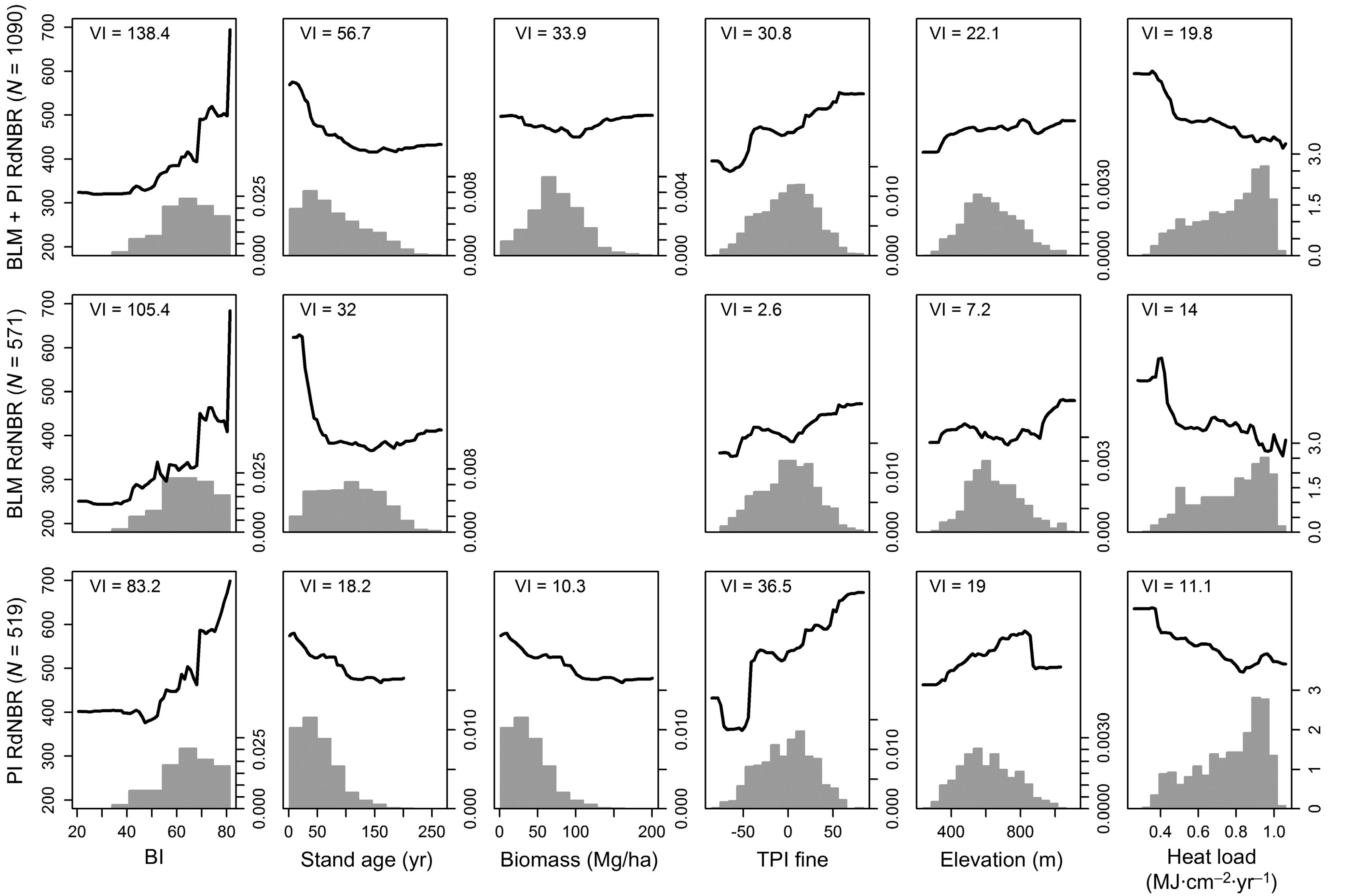

FIG. 4. Partial dependency plots showing relationships between each predictor variable and RdNBR in random forest models for all forests (BLM and PI, top panels), forests on Bureau of Land Management (BLM, middle panels), and private industrial land (PI, bottom panels). Number within each panel shows variable importance (VI; mean squared error increase [%]) of each predictor in the random forest model. Solid lines show trends in RdNBR in response to each predictor, histograms show the distributions of values for each predictor. Note there is no partial dependency plot for the relationship between RdNBR and biomass for BLM forests, as biomass was not a significant predictor variable for BLM forests based on two-stage variable selection procedures.

Table 3. Coefficients of predictor variables in generalized least squares model of RdNBR.

Variable	Parameter estimate	SE	*t*	*P*
Intercept	80.3321	90.4529	0.8881	0.3747
Age	−1.0544	0.2132	−4.9452	<0.0001
BI	6.1413	0.7618	8.0614	<0.0001
Ownership	76.3559	22.1111	3.4533	0.0006
Elevation	0.1179	0.0872	1.3512	0.1769
TPI fine	1.2839	0.2509	5.1169	<0.0001
Heat load	−150.0098	39.5750	−3.7905	0.0002
Slope	1.1321	0.5979	1.8933	0.0586
Biomass	0.1261	0.1194	1.0562	0.2911

important predictor variables, increasing MSE across the Douglas Complex by 56.7% and 53.2%, respectively. Age was the second most important variable in the final RF model for BLM forests (32%), but was the fourth most important variable for private forests (18.2%). Pre-fire biomass was the fourth most importance predictor variable in the RF model of the entire Douglas Complex (33.9%), but was not retained in the final RF model for BLM forests, and was the least important variable (10.3%) in the final RF model for private forests. Overall, topographic variables (TPI fine, heat load, and slope) were less important than BI, ownership, and age, increasing MSE across the Douglas Complex by 2.6–36.5%. RF models described 31%, 23%, and 25% of the variability in RdNBR across the entire burned area, BLM managed forests, and private forests, respectively.

Partial dependency plots displayed clear relationships between RdNBR and predictor variables (Fig. 4). RdNBR increased exponentially with BI across the entire Douglas Complex as well as for BLM and private forests separately, although RdNBR was shifted up by approximately 100 RdNBR on private forests vs. BLM forests for any given BI value. RdNBR was consistently higher in young forests on both ownerships. RdNBR declined rapidly on BLM forests between stand ages of 20 and 80 yr old, and remained roughly level in older forests. In contrast, RdNBR in private forests declined linearly with age across its range, although private lands had few forests greater than 100 yr old. RdNBR on both BLM and private forests increased with higher elevations, higher TPI fine, and steeper slope. Heat load was negatively correlated with RdNBR for all ownerships. Pre-fire biomass was not included in the final RF model for BLM lands, while, for the entire study and private lands, RdNBR appeared to decline slightly in forests with intermediate pre-fire biomass. However, the relationship between RdNBR and pre-fire biomass is more tenuous on private lands because they lacked forests with high pre-fire biomass.

Generalize least squares model

BI, age, ownership, TPI fine, and heat load were all significant predictors of RdNBR in the GLS model (Table 3). Slope had a suggestive relation with RdNBR ($P = 0.0586$), while elevation ($P = 0.1769$) and pre-fire biomass ($P = 0.2911$) were not a significant predictors. Relationships between predictors and RdNBR were consistent with partial dependency plots from RF models, with RdNBR increasing with BI and TPI fine and declining with age and heat load. Ownership had a fixed effect of increasing mean RdNBR by 76.36 ± 22.11 (mean ± SE) in private vs. BLM. Adjusting for all other predictor variables in the model, predicted mean RdNBR was higher on private (521.85 ± 18.67) vs. BLM forests (398.87 ± 18.23).

Discussion

Quantifying fire severity in the unique checkerboard landscape of the O&C Lands, this study disentangled the effects of forest management, weather, topography, and biomass on fire severity that are often spatially confounded. We found daily fire weather was the most important predictor of fire severity, but ownership, forest age, and topography were also important. After accounting for fire weather, topography, stand age, and pre-fire biomass, intensively managed private industrial forests burned at higher severity than older federal forests managed by the BLM. Below we discuss how the different variables in our analysis may influence fire severity, and argue that younger forests with spatially homogenized continuous fuel arrangements, rather than absolute biomass, was a significant driver of wildfire severity. The geospatial data available for our analyses was robust and comprehensive, covering two components of the fire behavior triangle (i.e., topography, weather), with pre-fire biomass and age serving as proxies for the third (fuel). However, we recognize there are limitations to our data and analyses and describe these below. We conclude by suggesting how our findings have important implications for forest and fire management in multi-owner landscapes, while posing important new questions that arise from our findings.

Fire weather was a strong top-down driver of fire severity, while bottom-up drivers such as topography and pre-fire biomass were less important. Across the western United States, evidence suggests bottom-up drivers such as topography and vegetation exert greater control on fire severity than weather, although the quality of weather representation confounds this conclusion (Dillon et al. 2011, Birch et al. 2015). At the same time, it is recognized that bottom-up drivers of fire severity can be overwhelmed by top-down climatic and weather conditions when fires burn during extreme weather conditions (Bradstock et al. 2010, Thompson and Spies 2010, Dillon et al. 2011). Daily burn period BI values were used in our analyses, but it is important to place fire weather conditions for any single fire within a larger historical context. We compared these daily BI values to the historical (1991–2017) summer (1 June–30 September) BI data we calculated from the Calvert RAWS data used in this study (3,296 total days). Within this historical record, mean burn period BI during the Douglas Complex for days with fire progression information was above average (79th percentile), but ranged considerably for any given day of the fire (15th–100th percentile). Fire severity was consistently higher on private lands across a range of fire weather conditions for the majority of days of active fire spread (Appendix S1: Fig. S3), leading us to conclude that while fire weather exerted top-down control on fire severity, local forest conditions that differed between ownerships remained important, even during extreme fire weather conditions.

Variation in pre-fire forest conditions across ownerships were clearly a significant driver of fire severity, and we believe they operated at multiple spatial scales. Private industrial forests were dominated by young trees, which have thinner bark and lower crown heights, both factors known to increase fire-induced tree mortality (Ryan and Reinhardt 1988, Dunn and Bailey 2016). At the stand scale, these plantations are high-density single cohorts often on harvest rotations between 30 and 50 yr, resulting in dense and relatively spatially homogenous fuel structure. In contrast, public forests were dominated by older forests that tend to have greater variability in both tree size and spatial pattern vs. plantations (Naficy et al. 2010), arising from variable natural regeneration (Donato et al. 2011), post-disturbance biological legacies (Seidl et al. 2014), and developmental processes in later stages of stand development (Franklin et al. 2002). Fine-scale spatial patterns of fuels can significantly alter fire behavior, and the effects of spatial patterns on fire behavior may increase with the spatial scale of heterogeneity (Parsons et al. 2017), which would likely be the case in O&C Lands due to the large scale checkerboard spatial pattern of ownership types.

Management-driven changes in fuel spatial patterns at tree and stand scales could also reconcile differences in prior studies that have found increases (Odion et al. 2004, Thompson et al. 2007) and decreases (Prichard and Kennedy 2014) in fire severity with intensive forest management. The two studies that observed an increase in fire severity with intensive forest management were conducted in the Klamath ecoregion of southwestern Oregon and northwestern California, the same ecoregion as this study. In contrast, Prichard and Kennedy (2014) examined the Tripod Complex in north-central Washington State, where harvests mostly occurred in low to mid elevation forests dominated by ponderosa pine, Douglas-fir, lodgepole pine (*Pinus contorta* var. *latifolia*), western larch (*Larix occidentalis*), and Engelmann spruce (*Picea engelmannii*). These forests have lower productivity compared to those studied in the Klamath ecoregion, with more open canopies and longer time periods to reach canopy closure after harvest, which likely results in more heterogeneous within stand fuel spatial patterns. Furthermore, forest clearcut units were relatively small in the Tripod Complex (mean 53 ha; Prichard and Kennedy 2014), and while these harvest units were spatially clustered, they were not large contiguous blocks as found in the O&C Lands. Last, it is unclear if the harvest units evaluated by Prichard and Kennedy (2014) experienced the full distribution of fire weather or topographic conditions compared to unharvested units, as our study does, which may confound their conclusions and our understanding of the relative importance of the factors driving fire behavior and effects.

Limitations

Our study examined a landscape uniquely suited to disentangling the drivers of wildfire severity and quantifying the effects of contrasting management activities. Additionally, we leveraged a robust collection of geospatial data to quantify the components of the fire behavior triangle. However, it is important to recognize the inherent limitations of our study. First, this study represents a single fire complex, instead of a regional collection of fires analyzed to elucidate broader system behaviors (sensu Dillon et al. 2011, Birch et al. 2015, Meigs et al. 2016). However, given the challenges of obtaining high quality fire weather information and accurate daily fire progression maps for fires that have occurred in landscapes with contrasting management regimes, we believe the landscape setting of our study provides key insights into the effects of management on fire severity that are not possible in large regional multi-fire studies. Second, while Landsat imagery is widely used to estimate forest conditions and fire severity, it has specific limitations. The GNN maps used in this study to derive pre-fire biomass and stand age are strongly driven by multi-spectral imagery from the Landsat family of sensors, whose imagery is known to saturate in forests with high leaf area indices and high biomass (Turner et al. 1999). Third, GNN maps of forest attributes used in this study were originally developed for large regional assessments, and as such have distinct limitations when used for analyses at spatial resolutions finer than the original source data (Bell et al. 2015), while application of GNN at fine spatial scales can underestimate GNN accuracy compared to larger areas commonly used by land managers (Ohmann et al. 2014). We addressed potential limitations of using GNN predictions at fine spatial scales in two ways. First, our sample plots are 90-m squares (3×3 30 m pixels) which more closely represents the area of the inventory plots used as GNN source data compared to pixel level analyses (Bell et al. 2015). Second, we visually assessed GNN predictions of live biomass and stand age within the Douglas Complex in relation to high resolution digital orthoimagery collected in 2011 by the USDA National Agriculture Imagery Program. From this qualitative assessment we concluded that GNN predictions characterize both between and within ownership variation in pre-fire biomass and age (Appendix S1: Fig. S4). Fourth and perhaps most fundamentally important, we relied on pre-fire biomass and stand age as proxies for fuel, in part because Landsat and other passive optical sensors have limited sensitivity to vertical and below-canopy vegetation structure (Lu 2006). Accurate and spatially complete quantitative information of forest surface and canopy fuels were not available for the Douglas Complex. More broadly, there are significant limitations to spatial predictions of forest structure and fuels using GNN and other methods that rely on passive optical imagery such as Landsat (Keane et al. 2001, Pierce et al. 2009, Zald et al. 2014), which is why we relied on the more accurately predicted age and pre-fire biomass variables as proxies. Surface and ladder fuels are the most important contributors to fire behavior in general (Agee and Skinner 2005), and surface fuels have been found to be positively correlated to fire severity in plantations within the geographic vicinity of the Douglas Complex (Weatherspoon and Skinner 1995). Yet correlations between biomass and fuel load can be highly variable due to site conditions and disturbance history (i.e., mature forests with frequent surface fires may have high live biomass but low surface fuel loads, while dense young forests that have regenerated after a stand replacing wildfire will have low live biomass but potentially high surface fuel loads as branches and snags fall). Therefore, GNN predicted pre-fire biomass may represent the total fuel load, but not the available surface and ladder fuels that have the potential to burn during a specific fire, and this is supported by the low importance of pre-fire biomass as a predictor of fire severity in our study. Furthermore, it is important to recognize that in addition to total surface and ladder fuels, the spatial continuity of these fuels strongly influences fire behavior (Rothermel 1972, Pimont et al. 2011). Fifth, while private industrial and BLM forests in our study area had very different forest conditions due to contrasting management regimes, ownership alone misses management activities (e.g., site preparation, stocking density, competing vegetation control, partial thinning, etc.) that can influence fuels and fire behavior. Sixth, while our spatial extrapolation of fire weather correlated well with daily fire severity and area burned, it did not account for topographic mediation of weather that can influence fine scale fire behavior, nor did it examine the underlying weather patterns such as temperature inversions that are common to the region and may play a key role in moderating burning index (Estes et al. 2017). Finally, we were unable to discern the effects of fire suppression activities and whether they varied by ownership, since incident documentation of suppression activities are generally not collected or maintained in a manner consistent with quantitative or geospatial statistical analyses (Dunn et al. 2017).

Management Implications

Although only one fire complex, the contrasting forest conditions resulting from different ownerships within the Douglas Complex are consistent with many mixed-ownership or mixed-use landscapes, such that we believe our results have implications across a much broader geographic area. First, it brings into question the conventional view that fire exclusion in older forests is the dominant driver of fire severity across landscapes. There is strong scientific agreement that fire suppression has increased the probability of high severity fire in many fire-prone landscapes (Miller et al. 2009, Calkin et al. 2015, Reilly et al. 2017), and thinning as well as the reintroduction of fire as an ecosystem process are critical to reducing fire severity and promoting ecosystem resilience and adaptive capacity (Agee and Skinner 2005, Raymond and Peterson 2005, Earles et al. 2014, Krofcheck et al. 2017). However, in the landscape we studied, intensive plantation forestry appears to have a greater impact on fire severity than decades of fire exclusion. Second, higher fire severity in plantations potentially flips the perceived risk and hazard in multi-owner landscapes, because higher severity fire on intensively managed private lands implies they are the greater source of risk than older forests on federal lands. These older forests likely now experience higher fire severity than historically due to decades of fire exclusion, yet in comparison to intensively managed plantations, the effects of decades of fire exclusion in older forests appear to be less important than increased severity in young intensively managed plantations on private industrial lands.

Furthermore, our findings suggest challenges and opportunities for managing intensive plantations in ways that reduce potential fire severity. Increasing the age (and therefore size) of trees and promoting spatial heterogeneity of stands and fuels is a likely means to reducing fire severity, as are fuel reduction treatments in plantations (Crecente-Campo et al. 2009, Kobziar et al. 2009, Reiner et al. 2009). The extent and spatial arrangement of fuel reduction treatments can be an important consideration in their efficacy at reducing fire severity at landscape scales (Finney et al. 2007, Krofcheck et al. 2017). However, optimal extent and landscape patterns of fuels reduction treatments can be hampered by a wide range of ecological, economic, and administrative constraints (Collins et al. 2010, North et al. 2015a, Barros et al. 2017). In the past, pre-commercial and commercial thinning of plantations (a potential fuel treatment) in the Pacific Northwest were common, economically beneficial management activities that improved tree growth rates and size, but these practices have become less common with improved reforestation success, alternative vegetation control techniques, and shorter harvest rotations (Talbert and Marshall 2005). This suggests there may be strong economic limitations to increased rotation ages and non-commercial thinning in young intensive plantation forests. More broadly, the development of large-scale forest management and conservation strategies can face legal and equitability challenges in multi-owner landscapes given existing laws constraining planning among private organizations (Thompson et al. 2004, 2006).

We believe two major questions arise from our findings that are important to fire management in multi-owner landscapes, especially those with contrasting management objectives. Plantations burned at higher severity, and this implies they are a higher source of risk to adjacent forest ownerships. However, a more explicit quantification of fire severity and susceptibility is needed to understand how risk is spatially transmitted across ownership types under a variety of environmental conditions. Second, we suggest the need for alternative management strategies in plantations to reduce fire severity at stand and landscape scales. However, the economic viability of such alternative management regimes remains poorly understood. Optimization models integrating spatial allocation of fuel treatments and fire behavior with economic models of forest harvest and operations could be used to determine if alternative management activities in plantations are economically viable. If alternative management activities are not economically viable, but wildfire risk reduction is an important objective on lands adjacent to industrial forestlands, strategic land purchases or transfers between ownership types may be required to achieve landscape level goals. This may be particularly important given the previously stated legal and equitability challenges in multi-owner landscapes. Regardless of the landscape-level objectives and constraints, it is clear that cooperation among stakeholders will be necessary in multi-ownership landscapes if wildfire risk reduction, timber harvesting, and conservation objectives remain dominant yet sometimes conflicting objectives for these landscapes.

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Supporting Information

Additional supporting information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/eap.1710/full

Data Availability

Data available from the Dryad Digital Repository: https://doi.org/10.5061/dryad.3gv5c78

Dear House Committee on Oversight and Reform,

In yesterday's hearing, Representative Ro Khanna and Chairwoman Carolyn Maloney each mentioned the importance of supporting the US Forest Service in its firefighting efforts, whether that be with tools or front-line workers.

Living with wildfire requires smart spending by Congress, which is why Friends of the Bitterroot; Klamath Forest Alliance; and Friends of the Clearwater, who drafted the attached, along with Firefighters United for Safety, Ethics, and Ecology; Wilderness Watch; Blue Mountains Biodiversity Project; and Forest Web; who endorse the attached, implore the Oversight Committee to investigate taxpayer money funding the costly, ineffective, and ecologically destructive firelines that the Forest Service frequently authorizes--many firelines of which significantly damage our public lands while only contributing to the optics of "doing something." Between Friends of the Bitterroot, Klamath Forest Alliance, and Friends of the Clearwater, we have documented this damage in California, Idaho, Montana, and Oregon, suggesting this might be a wider problem.

We ask that you reach out to non-agency experts who can speak to the efficacy of firelines, as well as nongovernmental organizations who watchdog our public lands, who have boots-on-the-ground expertise of old firelines, and who can speak to the immediate and lasting impacts of wasteful fireline construction.

Enclosed is a fact sheet, principally drafted by organizations copied here who are studying and monitoring this increasing problem in their mission areas, and endorsed by the other nonprofit organizations mentioned above. Secondly enclosed is a document of pictures that pertain to each section of the fact sheet--visual examples from across the West to be referenced as you read the facts.

We ask that you include this email and the two attachments into the record for this oversight hearing.

Thank you in advance for giving this the attention it deserves.

Regards,

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[redacted]
Friends of the Clearwater

Friends of the Bitterroot[1] • Friends of the Clearwater[2] • Kamath Forest Alliance[3]
• Firefighters United for Safety, Ethics, and Ecology • Wilderness Watch •
• Blue Mountains Biodiversity Project • Forest Web •

CONGRESS SHOULD INVESTIGATE THE FOREST SERVICE'S WASTEFUL PRACTICE OF BUILDING EXPENSIVE FIRELINES WHERE THEY CANNOT BENEFIT COMMUNITIES AND WHERE THEIR CONSTRUCTION DAMAGES ENVIRONMENTAL AND CULTURAL RESOURCES

1. Firelines damage cultural features, create serious environmental impacts, and impair natural ecosystems on public lands.

1(a). Firelines irreparably damage roadless and wilderness values by disrupting natural ecosystems and leaving permanent scars in wildland habitats. In recent years, the Forest Service has greatly increased wildland and wilderness dozerline* construction that is destructive to public lands. In addition to cutting trees for the dozerlines themselves, backcountry dozerline construction clears vegetation down to bare soil for vehicle access and helicopter landing sites. Parking vehicles and equipment further compacts barren ground, and compacted soil can aggregate with other environmental stressors that cumulatively impair whether and how well native vegetation recovers.

1(b). Firelines damage sensitive areas. In recent years, the Forest Service has increasingly constructed dozerlines in designated Wilderness, Inventoried Roadless Areas, Botanical Areas, Research Natural Areas, Wilderness Study Areas, Riparian Reserves, proposed wilderness areas, old growth forests, wetlands, and other sensitive habitats.

1(c). Firelines leave lasting environmental legacies. Old firelines become ghost roads, and facilitate additional soil erosion, compaction and disturbance, making lands more susceptible to non-native or noxious weed spread and chronic stream-sedimentation impacts. Non-native vegetation can be more fire prone than existing native vegetation. Increased access through illegal off-highway-vehicle use compounds these impacts and increases the harassment of wildlife and illegal poaching. Additionally, access on old dozerlines creates more opportunity for human ignitions in otherwise remote landscapes.

1(d). Firelines desecrate cultural history and archeological sites. For example, the Forest Service authorized bulldozing that injured numerous Native American archeological sites during the 2018 Klamathon Fire in the Soda Mountain Wilderness and Cascade-Siskiyou National Monument. Additional impacts to Native American archeological sites occurred in the 2021 Monument Fire and the 2021 McFarlane Fire on the Shasta-Trinity National Forest.

1(e). Firelines harm recreational resources. The Forest Service bulldozes firelines over trails, damaging the integrity and scenic qualities of Forest System Trails, National Recreation Trails, National Scenic Trails (including the Pacific Crest Trail), and other important recreational resources. Such damage often costs the taxpayer. *See 3(b) below.*

2. Ineffective backcountry firelines cannot replace effective home hardening or firelines close to communities

2(a). The most scientifically supported strategy for protecting homes against wildfire is home hardening. Home hardening means reducing home ignitability and minimizing ignitable features within the 130-foot radius of the house. Highly ignitable structures can be lost in even low-severity wildfires.[4]

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* A "dozerline" is the result of using bulldozers to blade the ground down to bare soil. While dozerlines are generally linear, "dozerlines" in this factsheet include large polygonal areas cleared by bulldozers.

2(b). The Forest Service regularly authorizes the construction of extensive dozerlines that are operationally ineffective and do not contribute to fire containment. This includes building remote backcountry firelines despite a low probability of success and building firelines that cannot be safely held by fire suppression crews.[5] This extensive construction often includes multiple contingency firelines built many miles from the fire perimeter and sometimes also far from homes or communities. As relatively narrow linear features, fires often burn over dozerlines during extreme fire behavior.[6]

3. The Forest Service's current practice of fireline construction wastes taxpayer money.

3(a). Building unnecessary and ineffective firelines is costly. For example, during the 2021 Dixie Fire in Northern California, the fire burned over approximately 600 miles of dozerlines.[7] Regardless, the Forest Service is increasing the scope and scale of dozerline construction, often building ineffective contingency lines, with sometimes a hundred miles or more of contingency lines built on a single large wildfire event.

3(b). Rehabilitating all firelines—including unnecessary and ineffective ones—is also costly. Rehabilitation costs include restoring, repairing, or reversing the damage highlighted in section one of this fact sheet. Rehabilitation includes recontouring and restoring dozerlines in wilderness as well as restoring trails where the Forest Service has bulldozed a fireline over existing trail. Restoring trails has occurred in the Soda Mountain Wilderness following the 2018 Klamathon Fire, the Siskiyou Wilderness after the 2018 Natchez Fire, the Bucks Lake Wilderness after the 2021 Dixie Fire and in numerous locations and on numerous fires along the Pacific Crest Trail. The cost of trail restoration could be avoided by utilizing less damaging fireline construction techniques.

3(c). Firelines create ongoing expenses after rehabilitation. Long-term impacts such as noxious weed control and downstream sedimentation require immediate and long-term mitigation efforts, and such activities must be publicly funded.

3(d). Despite the costs to construct, to rehabilitate, and to mitigate, the Forest Service will allow fireline construction even when the fireline construction becomes obsolete. During the Trail Creek Fire in Montana, for example, the Forest Service allowed private contractors to complete a fireline miles upwind from the fire perimeter, even after the wildfire was no longer a threat to the area.[8]

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[4] Cohen, J.D. 2000. Preventing Disaster: Home Ignitability in the Wildland-Urban Interface, Journal of Forestry, pp. 15-21.
[5] Firefighters United for Safety, Ethics and Ecology 2019. *We Had To Do Something: Futility and Fatality in Fighting the 2018 Mendocino Complex Fire*.
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[8] Sequence of satellite pictures of the Trail Creek Fire activity from beginning to end.pdf.

1(a)i Fireline bulldozed through proposed Pattison Wilderness during Monument Fire, California 2021.
Photo credit: Kent Collard

1(a)iii Large stump on dozerline Kangaroo Inventoried Roadless Area, Fort Complex Fire, Oregon 2012.
Photo credit Luke Ruediger, Klamath Forest Alliance

1(a)v Fireline through the Rackliff-Gedney Inventoried Roadless Area, Idaho (2017). The fire moved in the opposite direction and weeks later, Forest Service used this as a road for a logging project.
Photo credit: Alpha 1 Photography

1(a)ii Diller Canyon dozerline through Mt. Shasta Wilderness during Lava Fire, California 2021.
Photo credit Luke Ruediger, Klamath Forest Alliance

1(a)iv Fireline through Cove Roadless Area, Blue Fire, Idaho 2014. Fire was moving in the opposite direction. Fire managers discussed how to build fireline to accommodate logging trucks. Photo taken 2017.
Photo credit: Friends of the Clearwater

1(a)vi Junction of two firelines in Soda Mountain Wilderness. Klamathon Fire, Oregon 2018.
Photo credit: Luke Ruediger/Klamath Forest Alliance

1(b)i Fireline through riparian area in the Siskiyou Wilderness, Natchez Fire, California 2018.
Photo credit: Luke Ruediger/Klamath Forest Alliance

1(b)iii Fireline through Big Red Mountain Botanical Area, Hendrix Fire, Oregon 2018.
Photo credit: Luke Ruediger/Klamath Forest Alliance

1(b)v Old-growth tree felled along a fireline in the South Kalmiopsis Roadless Area, Buckskin Fire, Oregon 2016.
Photo credit: Luke Ruediger/Klamath Forest Alliance

1(b)ii Fireline bulldozed through wetland area, Davis Fire, Montana 2018.
Photo credit: Yaak Valley Forest Council

1(b)iv Dozerline crossing a half mile above where Lynx Creek becomes a perennial, fishbearing stream, Snow Creek Fire, Idaho 2021.
Photo credit: Friends of the Clearwater

1(b)vi Fireline through headwater stream in the Siskiyou Wilderness, Natchez Fire, California 2018.
Photo credit: Luke Ruediger/Klamath Forest Alliance

1(c)i OHV drives along fireline in Bucks Lake Wilderness during Dixie Fire, California 2021.

Photo credit: Darrel Jury/Friends of Plumas Wilderness

1(c)ii A massive safety zone developed above the Illinois River near the Kalmiopsis Wilderness, Klondike-Taylor Fire, Oregon 2018. *Photo credit: Inciweb*

1(c)iii Dozerline widened a former pack trail over three decades ago, which became a de facto ATV access to a Recommended Wilderness.
Photo credit: Friends of the Clearwater

1(c)iv Fireline in high mountain meadow in Condrey Mountain Inventoried Roadless Area, Abney Fire, California 2017.
Photo credit: Luke Ruediger/Klamath Forest Alliance

1(c)v Dozerline crossing Scotch Creek in Soda Mountain Wilderness, Klamathon Fire 2018.
Photo credit: Luke Ruediger/Klamath Forest Alliance

1(d)i Fireline was bulldozed through numerous archeological sites in the Cascade-Siskiyou National Monument uncovering this ancient mortar for grinding acorns, food or medicine items by Native American tribes. Soda Mountain Wilderness Klamathon Fire, Oregon 2018.
Photo credit: Luke Ruediger/ Klamath Forest Alliance

1(e)i and ii Fireline bulldozed over the Kelsey National Recreation Trail at Siskiyou Wilderness Area boundary, Eclipse Fire, California 2017. *Photo credit: Luke Ruediger/Klamath Forest Alliance.*

1(e)iii Fireline destroys Lone Pilot Trail, Soda Mountain Wilderness Klamathon Fire, Oregon 2018.
Photo credit: Luke Ruediger/Klamath Forest Alliance

1(e)iv 1(e) Fireline miles from fire and built over a narrow trail. Picture and dozerline from 2003. Fireline was recut in 2015. Meadow Creek Roadless Area.
Photo credit: Friends of the Clearwater

2(b)i The Medocino Fire Complex burned over ineffective bulldozed firelines. The fire burned nearly 720 square miles (459,000 acres) and a firefighter was killed. Fire crews and agency personnel predicted that these firelines would have a low probability of success, but they bulldozed them under pressure to "do something".
Photo credit: Klamath Forest Alliance, Environmental Protection Information Center, and FUSEE website

2(b)ii Forest Service cut a fireline up a steep incline above a creek, miles from any community. Fire burned over the fireline, Sand Mountain Fire, Idaho 2021.

Photo credit:
Friends of the Clearwater

3(a)i Dixie Fire Suppression efforts cost 639 millions dollars. Many of the firelines did not see fire. This fireline is in the Bucks Lake Wilderness.
Photo credit: Darrel Jury/Friends of Plumas Wilderness

3(a)ii Fireline leaves stumps along the Pacific Crest Trail, Bucks Lake Wilderness Area during Dixie Fire, California 2021. The fireline did not see fire.
Photo credit: Darrel Jury/Friends of Plumas Wilderness

3(a)iii An ineffectual dozerline along a ridge. This will likely create a legacy sediment site above the creek, Dixie Fire, California 2021.
Photo credit: Darrel Jury/Friends of Plumas Wilderness

3(b)i Firelines were bulldozed right to the lakeshore and now dump sediment into the water. Photo was taken Fall of 2018 when lake level was low. Carr Fire, California 2018.
Photo credit: FUSEE

3(b)ii Firelines leave clearcut corridors that fragment and degrade wildlife habitat. This catline failed to stop the fire from crossing over the ridge, Carr Fire, California 2018.
Photo credit: FUSEE

3(b)iii Fireline running along a ridgeline near Shasta Lake. It did not see fire, Carr Fire, California 2018.
Photo credit: FUSEE

3(c)i Erosion from a steep dozerline Johnson Creek Fire, Idaho 2017. *Photo credit: Friends of the Clearwater*

3(c)iv Fire retardant drop at edge of wet meadow in Kangaroo Inventoried Roadless Area, Creedence Fire, Oregon, 2017. Fire retardant is toxic to fish, amphibians and other wetland inhabiting species. Wetlands, rivers and streams are "avoidance areas," but are often impacted by misapplication.
Photo credit: Luke Ruediger/Klamath Forest Alliance

3(c)ii Unauthorized fireline built in the Kangaroo Inventoried Roadless Area which will bring in invasive fire prone weeds. Creedence Fire, Oregon 2017.
Photo credit: Luke Ruediger/Klamath Forest Alliance

3(c)iii Dozerline following "rehabilitation" in Zane Grey Roadless Area, Big Windy Fire, Oregon 2013.
Photo credit: Luke Ruediger/Klamath Forest Alliance

3(c)v Klamath National Forest dozerline Gap Fire, California 2016. *Photo credit: Luke Ruediger/Klamath Forest Alliance*

3(d)i Fireline slash left over from part of an 8 mile long linear clearcut, upwind, 1.5 miles from fire activity with wetlands in between, Trail Creek fire, Montana 2021.
Photo credit: Michael Hoyt/Friends of the Bitterroot

3(d)ii Forest Service allowed a private contractor to complete fireline upwind from fire perimeter, even after the wildfire was no longer a threat to the area, Trail Creek Fire, Montana 2021.
Photo credit: Michael Hoyt/Friends of the Bitterroot

3(d)iii
Another unnecessary fireline from the Davis Fire, Montana 2018.
Photo credit: Yaak Valley Forest Council