A carcass of an elephant that succumbed to drought is seen under a tree in Hwange National Park, in Zimbabwe, on 12 November 2019.

PHOTO: ZINYANGE AUNTONY/AFP VIA GETTY IMAGES

REVIEW

Hanging by a thread? Forests and drought

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Trees are the living foundations on which most terrestrial biodiversity is built. Central to the success of trees are their woody bodies, which connect their elevated photosynthetic canopies with the essential belowground activities of water and nutrient acquisition. The slow construction of these carbon-dense, woody skeletons leads to a slow generation time, leaving trees and forests highly susceptible to rapid changes in climate. Other long-lived, sessile organisms such as corals appear to be poorly equipped to survive rapid changes, which raises questions about the vulnerability of contemporary forests to future climate change. The emerging view that, similar to corals, tree species have rather inflexible damage thresholds, particularly in terms of water stress, is especially concerning. This Review examines recent progress in our understanding of how the future looks for forests growing in a hotter and drier atmosphere.

No tree species can survive acute desiccation. Despite this unambiguous constraint, predicting the death of trees during drought is complicated by the process of evolution, whereby the fitness of tree species may benefit equally from traits that either increase growth or enhance drought resilience. Complexity arises because improving either of these two beneficial states often requires the same key traits to move in opposite directions, which leads to important trade-offs in adaptation to water availability. This conflict promotes strategic diversity in different species' adaptations to water availability, even within ecosystems. Understanding how the diversity of tree species will be affected by future droughts requires a detailed knowledge of how the functions of different species interact with their environment. Temperature and atmospheric CO_2 concentration are fundamental elements that affect the water relations of all tree species, and the rapid rise in both of these potent environmental drivers has the potential to markedly change the way trees behave during drought. The future of many forest systems will be dictated by how these atmospheric changes interact with tree function.

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Is rising CO_2 good for trees?

A primary example of conflicting selection pressures on trees can be seen in the basic operation of photosynthesis. Achieving a higher photosynthetic rate requires higher leaf porosity to CO_2, but a higher leaf porosity causes a parallel increase in water loss, which is detrimental during an environmental water shortage. This trade-off plays a fundamental role in structuring terrestrial plant evolution and ecology (*1*), emphasizing the potential for rising CO_2 levels and temperatures to affect forests during drought conditions. There has been a change in perspective over the past 10 years, from expectations of enhanced forest growth under enriched atmospheric CO_2 to the more sobering prospect of damage or decimation of standing forest caused by an increase in the drying rates of leaves and soil in a hotter climate (*2*).

Early discussions of plant responses to rising atmospheric CO_2 (*3*) focused largely on CO_2 fertilization, a concept that refers to the potentially beneficial effects of atmospheric CO_2 enrichment on plant growth. Under controlled conditions, elevated CO_2 can theoretically increase plant growth by stimulating photosynthesis or by increasing the water use efficiency (WUE) of plants (the ratio of carbon intake to water lost by leaves). Both of these behaviors depend on the active response of stomata (microscopic valves on the leaf surface that regulate gas exchange) to CO_2 (*4*). Long-term studies of tree growth under artificially enhanced atmospheric CO_2 suggest that improved photosynthetic performance at elevated CO_2 can translate into increased growth (*5*, *6*), but there is little evidence of any CO_2-associated growth enhancement in natural forest conditions (*7*, *8*). This is thought to be either because of colimiting resources for plant growth, such as water and nitrogen (*9*–*11*), or because stomatal closure in response to rising CO_2 increases WUE (*12*, *13*) at the cost of enhanced assimilation and growth. Controversially, it has been suggested that the impacts of future drought stress may be ameliorated by higher atmospheric CO_2 if WUE is sufficiently enhanced (*14*, *15*). The validity of this concept depends largely on the effects of rising temperature on WUE and plant survival during extended rainfall deficits.

Rising temperature and drought

Ultimately, the impact of elevated CO_2 on forest trees is likely to come down to the intensity of the CO_2-associated temperature rise and its effect on trees' water use. This is because the distributions of tree species, in terms of water availability, broadly reflect their intrinsic tolerance of water stress (*16*–*18*). In other words, species from rainforests to arid woodlands face similar exposure to stress or damage during periods of drought (*19*). Hence, any increase in the rate of soil drying caused by elevated temperatures is likely to lead to increasing damage to standing forests during drought. Improved tree WUE could ameliorate the temperature effect, but this argument remains highly debatable because most reports of improvements in tree WUE with rising atmospheric CO_2 refer to intrinsic WUE, a value that converts to real plant water use only with a knowledge of leaf temperature and atmospheric humidity (*20*). Thus, rising atmospheric temperature and the associated increase in evaporative demand is likely to reverse the improvements in tree WUE that are proposed to result from higher CO_2. Recent evidence suggests that this is the case, with observations of reduced global tree growth and vegetation health associated with enhanced evaporative gradients and warming temperatures (*21*, *22*).

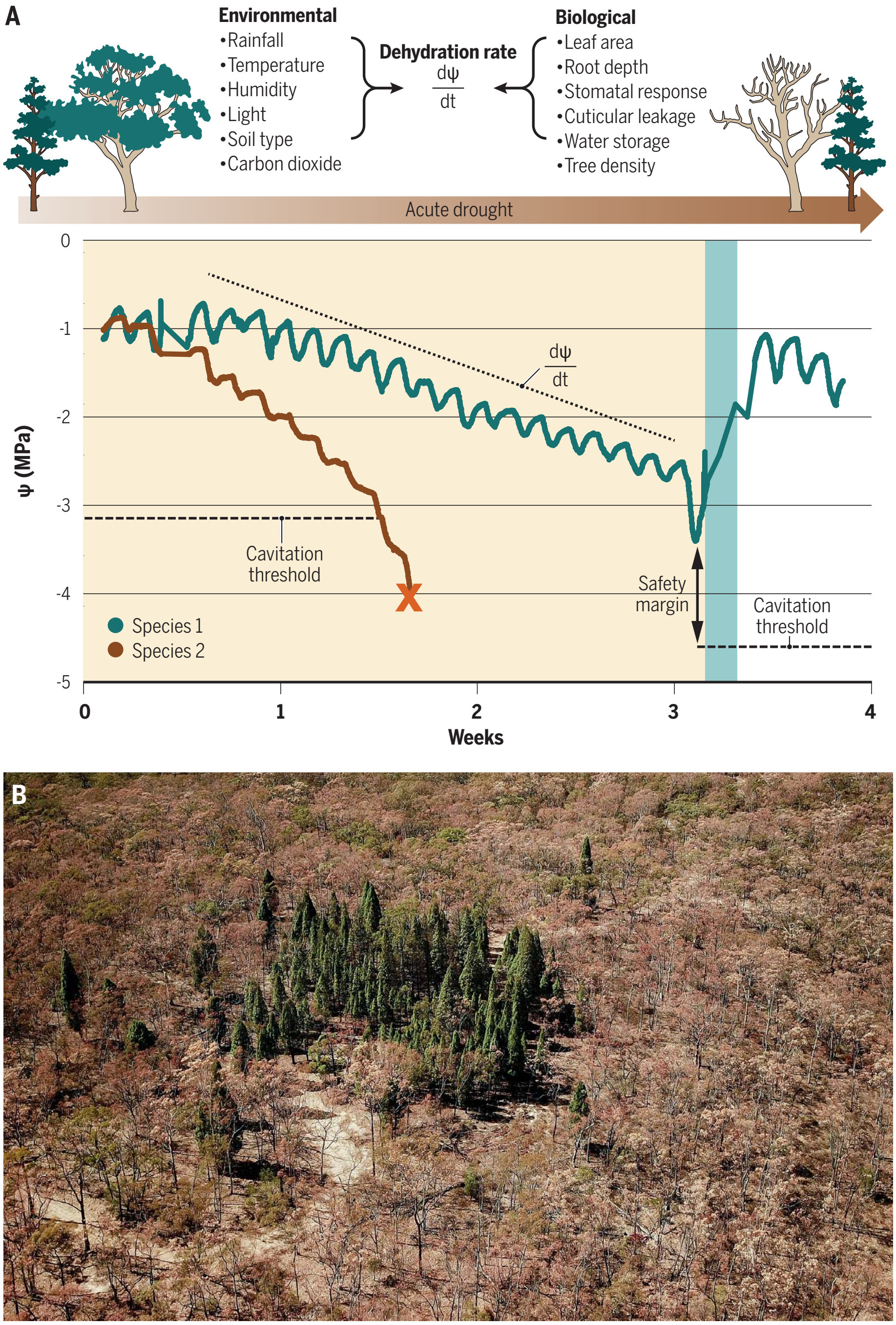

Fig. 1. Theoretical and observed impacts of drought on co-occurring tree species. (**A**) A representation of the impact of drought on two tree species with different thresholds for drought-induced vascular damage. Different xylem cavitation thresholds determine the water potential (Ψ: water stress intensifies as water potential becomes more negative) causing tree mortality. Two lines indicate the oscillating water stress between day and night as the two species (indicated by small tree icons) dehydrate after the cessation of rainfall (data are from two trees from a dry forest site in Tasmania, Australia). The cavitation threshold and the rate of drying (dΨ/dt) both determine how many days into an acute drought each species will die. The taller species, which is more vulnerable to cavitation and faster drying, dies (indicated by an orange X) in week 2, whereas the shorter species survives until rainfall (indicated by the blue rectangle in week 3), enabling the tree to recover hydration. The proximity between the cavitation threshold and the lowest water potential during drought is known as the hydraulic safety margin. The dehydration rate is a product of a set of environmental and biological factors, many of which interact. Increasing temperature increases the rate of drying both directly and by interaction with biological factors, whereas CO_2 has the potential to reduce dehydration by its biological interaction with stomata and the photosynthetic rate. (**B**) Recent (2019) drought-induced mortality of native forest in eastern Australia. Large-scale mortality of *Eucalyptus* trees (seen as recently killed dry canopies) contrast with the more cavitation-resistant conifer species (*Callitris*). The observed pattern of mortality can be explained by the processes described in (A).

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In combination with the size and allometry of trees, the dynamic behavior of stomata and their regulation of water loss from tree canopies largely dictates the course of plant and soil dehydration. During atmospheric or soil water deficit, stomatal closure limits transpiration, preserving water content in the soil and tree (*23*). However, this well-characterized behavior becomes unpredictable when leaf temperatures are substantially elevated, with stomata permitting greater water loss than expected during both day (*24*, *25*) and night (*26*–*28*). Additionally, plants continue to lose some residual water after the stomatal valves are closed, and this residual leakiness also appears to increase with elevated temperatures (*29*–*31*). Herein lies perhaps the greatest threat for forests subjected to warming atmospheric temperature, because warmer plants not only consume water faster when soils are hydrated, but they also have a diminished capacity to restrict water loss during drought, thereby exhausting soil water reserves.

Tree mortality is most commonly observed when drought and high temperature are combined (*32*–*34*), likely owing to the compounding effects of the increased evaporative gradient and the increased porosity of leaves at high temperature. The inevitable rise in the intensity and/or frequency of such events as global temperatures climb (*35*) has already been associated with an increase in tree mortality globally (*36*), especially in larger trees (*37*), which raises a grave concern about the capacity of existing forests to persist into the future. Establishing the magnitude of this threat is an important challenge that requires a fundamental understanding of how water deficit leads to tree mortality.

Much research has focused on the possible mechanisms behind tree death during drought. Possible mechanisms primarily include vascular damage, carbon starvation, and enhanced herbivory (*38*–*42*). These studies reveal the complex nature of tree death, where the moment of death is difficult to pinpoint or even define (*43*). Although it remains difficult to connect cause and effect at the point where drought injury becomes lethal, strong and consistent correlational data from trees suffering mortality or growth inhibition across the globe point unequivocally to the plant water transport system as a fundamental axis dictating the long-term survival of trees (*44*–*47*).

Forests on a thread

The massive woody structure of trees provides mechanical support for their photosynthetic crowns; however, the matrix of microscopic threads of water that is housed within the porous woody cells of the xylem is even more fundamental to tree survival. These liquid threads provide a highly efficient mechanism to transport large quantities of water over long distances under tension, from the roots to the leaves. Relying on this passive pathway to replace the water transpired by leaves has the major drawback that the internal water column in trees becomes increasingly unstable during times of water stress, as the tension required to draw water from the soil increases. Rising xylem water tension (conventionally described as an increasingly negative water potential) during intensifying soil water deficit exposes a universal vulnerability in trees to xylem cavitation during drought (*48*). This occurs when the water potential in the xylem becomes sufficiently negative to draw minute bubbles through the cell wall into the lumen of the xylem cells, at which point the small bubbles trigger a very rapid formation of voids (in a process termed xylem cavitation), which subsequently become air bubbles or embolisms that block water flow. The vulnerability of a species to cavitation is conventionally quantified as a P50, which is the water potential that causes 50% of the xylem to cavitate. The most extreme form of xylem damage occurs when a feedback develops, as increasing xylem water tension caused by soil water deficit leads to xylem cavitation and blockage, further exacerbating the tension in the xylem, and ultimately killing the plant by completely severing the connection between soil and leaves. This process is likely to occur under acute water shortage (*49*, *50*), killing plants (*51*) before the return of rainfall. Although this type of acute drought-induced mortality may not describe all instances of tree death during water shortage, the existence of quantifiable biophysical thresholds defining specific survival limits for different tree species has greatly enabled our capacity to understand tree mortality and distribution (*42*) and provides a robust basis for modeling future effects of drought (*52*, *53*). Many aspects of the xylem cavitation process remain uncertain because of difficulties associated with measuring water flow in a system that operates under high tension (*54*); however, new methods are providing more clarity and confidence to our understanding of the critical sensitivity of plant vascular systems to damage under water stress (*55*, *56*).

The water transport system in plants lies at the center of interactions between rainfall, soil water, carbon uptake, and canopy dehydration, which makes xylem hydraulics an obvious focus for understanding and predicting the thresholds between tree death or survival during exposure to drought and heat stress. Xylem vulnerability to cavitation varies markedly among species (*19*), not only indicating sensitivity to water deficit but also enabling the quantification of functional impairment if trees are not immediately killed by drought (*43*, *50*). Although a knowledge of cavitation thresholds informs the triggering of tree damage, the rate of tree dehydration indicates how quickly that damage threshold is approached during drought. The characteristics of tree species that are classically associated with adaptation to water availability—such as rooting depth, water storage, stomatal behavior, root and canopy area, and leaf phenology—can be predictably integrated to determine how plant water content will respond to environmental conditions. The combination of environmental conditions with biological attributes results in a highly tractable framework (Fig. 1) for understanding the dynamics of mortality or survival during slow dehydration (*57*).

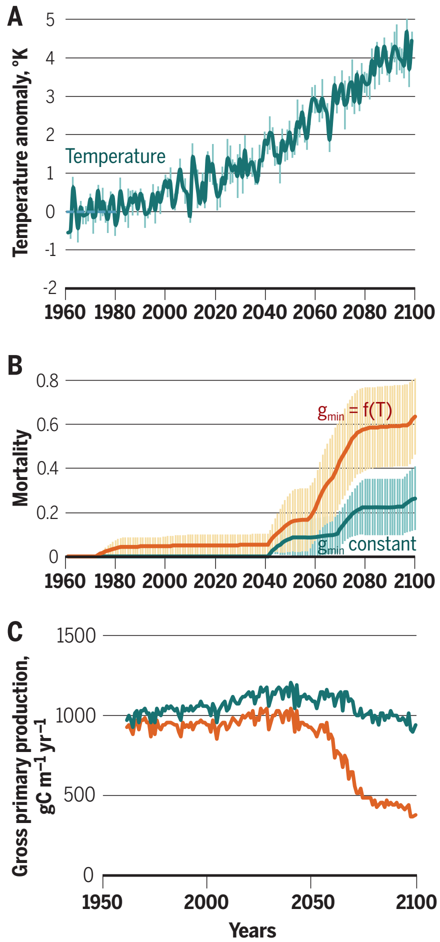

Fig. 2. A mechanistic hydraulic model of future drought-induced tree mortality. (**A** to **C**) Sensitivity of a process-based hydraulic model to predict tree mortality and gross primary production (GPP) under the representative concentration pathway (RCP) 8.5 climatic scenario. The model was parametrized with data for a population of a typical temperate coniferous tree, displaying a Gaussian distribution of cavitation resistance (mean xylem vulnerability of P50 = –3.5 MPa, variance = 0.3). Daily climatic data from five Eurocodex climate models were used to simulate tree transpiration, soil water content, xylem water tension, and xylem cavitation. The lethal threshold of cavitation was set to 88%. The model forecasts an increase in tree mortality with the rise of temperature caused by predicted climate change. The predicted collapse of the tree population and forest GPP was more drastic when a more realistic temperature-dependent increase in the cuticular leakage (g_{min}) (*108*) was implemented in the model [g_{min} = f(T); orange line] compared with a static cuticular leakage [g_{min} constant (gC); green line]

Despite the existence of sharp xylem cavitation thresholds, post-drought legacies of damage and mortality of trees are often protracted over months or years after peak drought intensity (*58*), which implies that more-complex interactions between plant water and carbon status are also important in the recovery process. Post-drought rainfall enables trees that have not suffered catastrophic xylem failure to replace drought-damaged xylem by woody regrowth (*50*), but this is highly costly and can lead to rapid depletion of tree carbon reserves (*59*), leaving them vulnerable to insect attack [although insect interactions remain unpredictable (*60*)] unless conditions remain favorable. Recovering, drought-damaged trees may invest disproportionately in new leaves rather than xylem growth (*61*), potentially making them more sensitive to subsequent water shortage because of reduced xylem water delivery. Although much remains to be learned about the physiology of plant hydraulics, the principles of hydraulic failure provide a solid framework for understanding and predicting mortality, damage, and recovery under a diversity of drought scenarios.

Modeling forest mortality in the future

Diverse approaches have been employed to predict how forests are likely to respond to hotter and potentially drier and more-variable conditions in the future. Progress toward understanding the mechanisms that lead to tree mortality has seen a movement away from traditional correlative niche models (*62*) in favor of more process-based modeling. Incorporation of theoretically derived mortality modules into dynamic vegetation models has the potential to capture drought mortality, but these models are currently rather unsophisticated and unreliable, particularly when applied outside the domain of calibration (*63*, *64*). At the more functional end of the modeling spectrum are recent attempts to explicitly model drought mortality triggered by hydraulic failure (or associated carbon starvation) (*52*). In particular, the combination of tree hydraulics with the principles of stomatal optimization (assuming that stomatal behavior regulates assimilation and transpiration to achieve a maximum difference between photosynthetic gain and the risk of hydraulic damage) is emerging as a promising structure for models of land surface gas exchange (*65*–*67*). Although the mathematical rendering of physiological processes to predict forest productivity and tree survival provides a powerful approach for modeling the performances of species or genotypes in a range

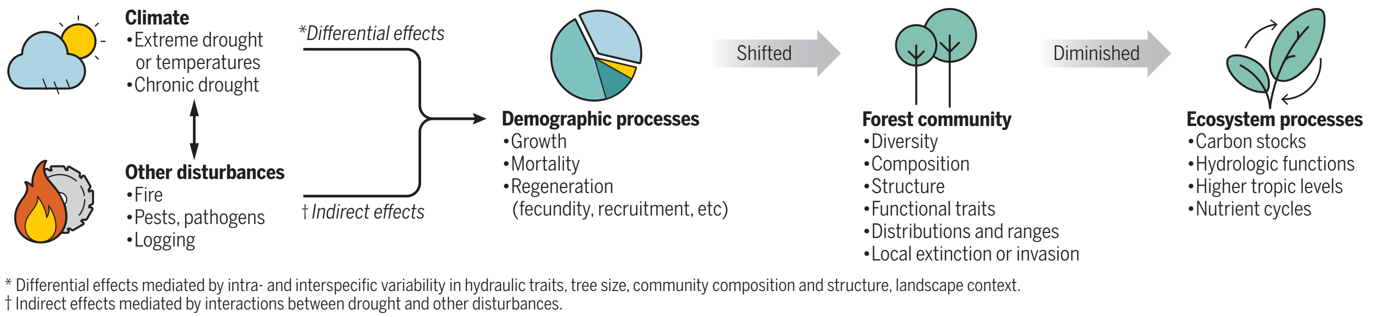

Fig. 3. Interactions between climate and forest community. Schematic of how climatic variability interacts with disturbance to affect tree demographic processes, which may result in shifted community diversity and species distributions as well as ecosystem processes.

of future climates, a limitation in using these mechanistic formulations is that relatively small changes in parameterization or biological assumption can substantially change predictions (Fig. 2). To capture this uncertainty, recent studies have spanned a range of assumptions, particularly with regard to how trees might acclimate to drought, in order to reveal a range of possible scenarios (*15*, *68*).

Modeling provides the most credible view of how forests may cope with different intensities of future global warming, with most models suggesting large-scale mortality, range contraction, and productivity loss through this century under the current warming trajectories (Fig. 2). Greater precision as to the nature and pace of forest change is urgently needed, requiring dedicated work on key knowledge gaps (*69*) that limit model precision accuracy. These gaps are apparent in even the basic physiological processes of trees, such as stomatal behavior, tree water acquisition (*70*), and interactions between water and carbon stores in trees (*67*). Critical components such as the dynamic connection between trees and the soil are highly simplified in models owing to a lack of knowledge about water transfer and storage in the roots under conditions of water stress. The triggering of mortality is also highly oversimplified because the negative feedbacks likely to operate during acute tree stress are difficult to capture in a model. Avoiding this complexity, a commonly used proxy for lethal water stress is the point of 50% xylem cavitation in stems (Fig. 2). Although this threshold is not strictly correct (because trees can survive with a 50% impairment of water transport capacity), it does provide a readily measurable indication of rapid vascular decline incipient to complete failure of the vascular connection between roots and leaves. More-precise understanding of the post-drought transition to recovery or tree death is needed to accurately represent the legacy effects of drought in large-scale models.

Acclimation of forest in situ

The long generation time and slow growth of trees present a formidable challenge to survival in the face of rapid environmental change, particularly increases in aridity and the frequency of extreme-drought events. Avoidance of local extinction (extirpation) in tree species is possible by two non–mutually exclusive mechanisms: (i) migration tracking the ecological niches to which they are adapted or (ii) adaptation and acclimation to novel climate conditions and persistence within their current range. Species distribution models based on climatic envelopes have predicted pronounced range shifts in tree populations over the next century; however, this mechanism of survival is contingent on the capacity of species to achieve rapid migration (*71*), and few tree species are likely to disperse rapidly enough to keep pace with the current rate of climate warming (*72*). The persistence of tree populations exposed to increased aridity in their current range will depend on adaptation and acclimation to higher intensities of plant water stress. Given the rapid pace of climate change, adaptation of organisms with such long generation times appears unlikely to enable persistence in most species.

The potential for rates of adaptation to keep pace with environmental change depends on a number of factors, including the levels of genetic diversity present in critical traits, differentiation between leading and trailing edge populations, and gene flow between populations. Very few studies have examined the genetic diversity present in important plant hydraulic traits, with the most-comprehensive studies focused on temperate deciduous and conifer species (*73*–*75*). The results of these studies suggest that genetic diversity of traits, such as cavitation resistance, is low in pine species (*74*) but may be higher in temperate angiosperms such as beech (*73*, *76*). Overall, genetic diversity in hydraulic traits appears to be limited relative to the changes in intensity of water stress that are expected over the coming decades. This lack of genetic diversity across populations may limit the capacity for adaptation to increasing aridity in current distributions.

Acclimation by means of phenotypic plasticity presents another mechanism by which trees may adjust to novel climate regimes (*77*). Acclimation is dependent on trait plasticity in individuals and may occur over much shorter time scales than evolutionary processes such as adaptation. The acclimation of some physiological and morphological traits in response to changes in temperature and drought stress is well documented. This includes the acclimation of photosynthesis, respiration, and leaf thermal tolerance to temperature (*24*, *71*) and changes in resource allocation, such as sapwood-to-leaf ratio (*78*). For example, leaf shedding allows trees to rapidly reduce the leaf surface area available for transpiration and is a primary mechanism limiting water loss during drought. Studies examining intraspecific variation across precipitation gradients have shown that populations adjust to greater aridity through increasing sapwood-to-leaf ratios (*79*–*81*), increasing hydraulic capacity relative to leaf area deployed.

Acclimation in physiological traits related to drought tolerance is less well studied. However, the available data suggest that there is limited plasticity in key mechanistic traits. This is borne out in common-garden and reciprocal transplant experiments as well as throughfall exclusion experiments and studies of natural populations growing across aridity gradients (*80*, *82*, *83*). Low plasticity in hydraulic safety has also been observed with tree size (*84*), although the behavior of seedlings remains unknown. Pine species exhibit particularly low variation in cavitation resistance, with available evidence suggesting canalization of hydraulic traits, which constrains the capacity of pines to acclimate or adapt to drier conditions (*74*). Common-garden studies suggest that traits associated with hydraulic safety (Fig. 1) appear to be under strong genetic control (*16*, *81*). This may be one reason why partial leaf shedding is a commonly observed response to drought, because higher plasticity in leaf area may assist trees in maintaining levels of water stress within the functional limits set by inflexible hydraulic failure thresholds. However, reducing leaf area comes at the cost of lowered productivity and growth rates, and it may adversely affect survival in trailing edge populations exposed to intense interspecific competition.

Communities and consequences

Although hydraulic failure may be sudden and pronounced, predicting the consequences of drought for tree populations and communities is more challenging than simply extrapolating from models of hydraulic processes. This is because drought may also affect demographic processes beyond tree mortality and may interact with other disturbances. Stand-level interactions among individuals and species may attenuate or exacerbate drought impacts, and landscape-scale variations in topography, edaphic conditions, or forest-patch characteristics can modulate drought effects (Fig. 3). Moreover, current forest communities are responding to both extreme events, such as El Niño–Southern Oscillation (ENSO)–related droughts (*85*), and to directional changes in rainfall, such as decadal-long decreases in rainfall (*86*). What does seem certain is that these changes in forest composition and tree species distributions will have important consequences for the diversity and structure (*69*), hydrologic function (*87*), and carbon-storage potential (*88*) of future forests.

Interspecific variation in hydraulic and other traits is clearly linked to differential damage and mortality rates during extreme drought (*47*, *89*, *90*). However, other demographic processes or life history stages—such as fecundity, seedling recruitment, and tree growth—may also be affected, and species- or functional group–specific responses to drought may change community composition and functional traits over decadal time scales or even result in shifts among biomes, such as forests being replaced by shrublands (*91*). Regeneration dynamics are especially critical in mediating shifts between vegetation types or biomes (*91*), but, at this point, the data are too limited to generalize about how the likelihood of such shifts differs among forest types. For example, an extreme drought during the 2015 ENSO reduced seed rain of drought-deciduous tree species relative to evergreen trees and lianas in a seasonally dry tropical forest in Costa Rica (*92*). By contrast, in a semimoist tropical forest in Panama, a 30-year record of leaf and fruit production showed elevated seed production during ENSO years that mirrors seasonal patterns, suggesting that the sunnier conditions that accompany ENSO favor fruit over leaf production (*93*).

Predicting or modeling the impacts of drought on forest communities is also complicated by interactions between changes in climate and interactions with other disturbance agents, such as fire (*94*), insects and pathogens (*95*), or logging (*96*). The catastrophic wildfires that have affected Australia in 2019 and 2020, after years of extreme drought, is just one such example of drought-fire interactions. Such interactions are also affecting forests in North America (*97*), Amazonia (*94*), and elsewhere (*98*). Increases in vapor-pressure deficit and temperature during drought dry out fuel, thereby increasing fire activity and the area that is burned (*97*). Drought-fire interactions may also cause tipping points and shifts among vegetation types in areas such as the southwestern Amazon (*94*). There, tree mortality is elevated during intense fires experienced in drought years (*94*), resulting in altered microclimatic conditions and grass invasion into the understories, which further increases flammability and fire risk (*94*).

The identification of which trees and species within stands are most vulnerable to drought (*37*, *99*) and of the factors that render certain stands within landscapes more susceptible to changing climates (*100*, *101*) may inform both basic science and management strategies (*69*). Meta-analysis and theoretical models suggest that large trees are more likely than smaller trees to die during and after drought (*37*, *59*). However, simple predictions of which size classes of trees die during drought may not hold in mixed-species forests, where different sizes of drought-weakened trees experience different levels of attack by host-specific bark beetles in idiosyncratic ways (*102*). Additional knowledge of community composition beyond tree size—i.e., size-species distributions—may help bridge predictions from the individual to the stand scale (*69*). Forest density may be an indication of competition for water, and trees growing at low densities may experience lower mortality rates (*101*) and less-pronounced reductions in growth during drought compared with those in higher density stands (*103*).

Advances in the remote sensing of proxies of plant stress, like canopy water content, may help us to monitor and map patterns at coarse geographic scales (*104*). These findings may guide silvicultural actions, such as selective thinning to reduce vulnerability to drought in managed forests (*103*). Finally, the diversity of hydraulic traits in forests has emerged as a property that helps explain ecosystem responses to climatic variability (*105*). Ecosystem fluxes inferred from eddy covariance measurements of forests with higher trait diversity of hydraulic traits appear more buffered against changes in soil water and vapor-pressure deficit compared with forests with low trait diversity (*105*), presumably because catastrophic failures of canopy dominants (Fig. 1B) are reduced. This underscores the idea that building large databases of hydraulic traits, rather than morphological traits such as specific leaf area and wood density, is a high priority to advance our understanding of forest vulnerability to drought (*106*).

Outlook

Drought is a natural phenomenon that plays a major role in limiting the distributions of species. However, the extremely rapid pace of climate change appears to be introducing enormous instability into the mortality rates of global forests (*107*). Instability and unpredictability are intrinsic aspects of the physiological processes that are linked to the drought-induced mortality process, whereby vascular damage is prone to failure and positive feedback, leading to tree death. Most models predict major damage to forests in the next century if current climate trajectories are not ameliorated. Debate still remains as to the magnitude of stabilizing forces, such as tree acclimation and positive CO_2-associated effects on water use, but most observational data suggest that forest decline is well under way. Future improvements in physiological understanding and dynamic monitoring are needed to improve the clarity of future predictions; however, changes in community structure and ecology are certain, as are extinctions of tree species by the direct or indirect action of drought and high temperatures.

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Science

Hanging by a thread? Forests and drought

Timothy J. Brodribb, Jennifer Powers, Hervé Cochard and Brendan Choat

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FEATURE | **September 10, 2019**

Satellite Data Record Shows Climate Change's Impact on Fires

Fires are a natural part of the ecosystem in North American forests. However, their size and intensity is shaped by climate. Credit: NASA

By Ellen Gray,

NASA's Earth Science News Team

Hot and dry. These are the watchwords for large fires. While every fire needs a spark to ignite and fuel to burn, the hot and dry conditions in the atmosphere determine the likelihood of a fire starting, its intensity and the speed at which it spreads. Over the past several decades, as the world has increasingly warmed, so has its potential to burn.

Since 1880, the world has warmed by 1.9 degrees Fahrenheit (1.09 degrees Celsius), with the five warmest years on record occurring in the last five years. Since the 1980s, the wildfire season has lengthened across a quarter of the world's vegetated surface, and in some places like California, fire has become nearly a year-round risk. The year 2018 was California's worst wildfire season on record, on the heels of a devasting 2017 fire season. In 2019, wildfires have already burned 2.5 million acres in Alaska in an extreme fire season driven by high temperatures, which have also led to massive fires in Siberia.

Whether started naturally or by people, fires worldwide and the resulting smoke emissions and burned areas have been observed by NASA satellites from space for two decades. Combined with data collected and analyzed by scientists and forest managers on the ground, researchers at NASA, other U.S. agencies and universities are beginning to draw into focus the interplay between fires, climate and humans.

"Our ability to track fires in a concerted way over the last 20 years with satellite data has captured large-scale trends, such as increased fire activity, consistent with a warming climate in places like the western U.S., Canada and other parts of Northern Hemisphere forests where fuels are abundant," said Doug Morton, chief of the Biospheric Sciences Laboratory at NASA's Goddard Space Flight

Center in Greenbelt, Maryland. "Where warming and drying climate has increased the risk of fires, we've seen an increase in burning."

A Hotter, Drier World

High temperatures and low humidity are two essential factors behind the rise in fire risk and activity, affecting fire behavior from its ignition to its spread. Even before a fire starts, they set the stage, said Jim Randerson, an Earth system scientist at the University of California, Irvine who studies fires both in the field and with satellite data.

He and his colleagues studied the abundance of lightning strikes in the 2015 Alaskan fire season that burned a record 5.1 million acres. Lightning strikes are the main natural cause of fires. The researchers found an unusually high number of lightning strikes occurred, generated by the warmer temperatures that cause the atmosphere to create more convective systems — thunderstorms — which ultimately contributed to more burned area that year.

Hotter and drier conditions also set the stage for human-ignited fires. "In the Western U.S., people are accidentally igniting fires all the time," Randerson said. "But when we have a period of extreme weather, high temperatures, low humidity, then it's more likely that typical outdoor activity might lead to an accidental fire that quickly gets out of control and becomes a large wildfire."

For example, in 2018 sparks flying from hammering a concrete stake into the ground in 100-degree Fahrenheit heat and sparks from a car's tire rim scraping against the asphalt after a flat tire were the causes of California's devastatingly destructive Ranch and Carr Fires, respectively. These sparks quickly ignited the vegetation that was dried out and made extremely flammable by the same extreme heat and low humidity, which research also shows can contribute to a fire's rapid and uncontrollable spread, Randerson said. The same conditions make it more likely for agricultural fires to get out of control.

A warming world also has another consequence that may be contributing to fire conditions persisting over multiple days where they otherwise might not have in the past: higher nighttime temperatures.

"Warmer nighttime temperature allow fires to burn through the night and burn more intensely, and that allows fires to spread over multiple days where previously, cooler nighttime temperatures might have weakened or extinguished the fire after only one day," Morton said.

Climate Systems at Work

Hot and dry conditions that precede fires can be tempered by rain and moisture circulating in the atmosphere. On time scales of months to years, broader climate patterns move moisture and heat around the planet. Monitoring these systems with satellite observations allows researchers to be able to begin to develop computer models for predicting whether an upcoming fire season in a given region will be light, average or extreme. The most important of these indicators are sea surface temperatures in the Pacific Ocean that govern the El Niño Southern Oscillation (ENSO).

"ENSO is a major driver of fire activity across multiple continents," Randerson said, who along with Morton and other researchers have studied the relationship between El Niño events and fire seasons in South America, Central America, parts of North America, Indonesia, Southeast Asia and equatorial Asia. "The precipitation both before the fire season and during the fire season can be predicted using sea surface temperatures that are measured by NASA and NOAA satellites."

An ongoing project, Randerson said, is to now extend that prediction capability globally to regions that are affected by other ocean-climate temperature changes and indicators.

The Human Factor

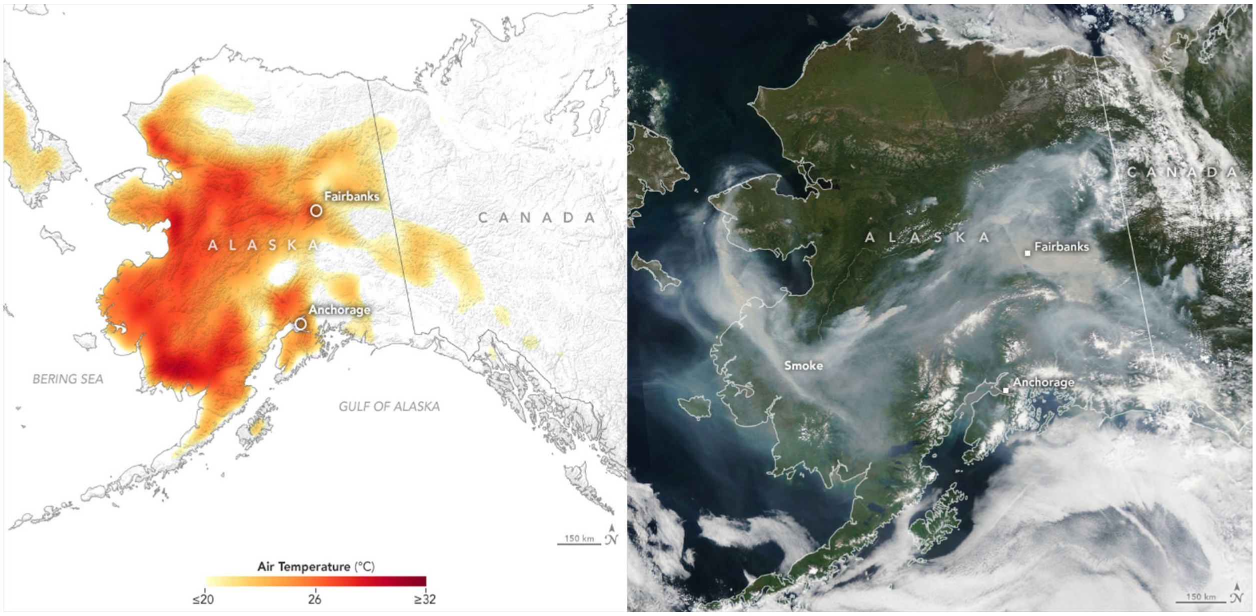
In studying the long-term trends of fires, human land management is as important to consider as any other factor. Globally, someplace on Earth is always on fire — and most of those fires are set by people, either accidentally in wildlands, or on purpose, for example, to clear land or burn agricultural fields after the harvest to remove crop residues.

But not all fires behave the same way. Their behavior depends on the fuel type and the how people are changing the landscape. While fire activity has gotten worse in northern latitude forests, research conducted by Randerson and Morton has shown that despite climate conditions that favor fires, the number of fires in grassland and savanna ecosystems worldwide are declining, contributing to an overall decline in global burned area. The decline is due to an increased human presence creating new cropland and roads that serve as fire breaks and motivate the local population to fight these smaller fires, Morton said.

"Humans and climate together are really the dual factors that are shaping the fires around the world. It's not one or the other," Randerson said.

Fire Feedbacks

Fires impact humans and climate in return. For people, beyond the immediate loss of life and property, smoke is a serious health hazard when small soot particles enter the lungs. Long-term exposure has been linked to higher rates of respiratory and heart problems. Smoke plumes can travel for thousands of miles affecting air quality for people far downwind of the original fire. Fires also pose a threat to local water quality, and the loss of vegetation can lead to erosion and mudslides afterwards, which have been particularly bad in California, Randerson said.



In June and early July 2019, a heat wave in Alaska broke temperature records, as seen in this July 8 air temperature map (left). The corresponding image from the Moderate Resolution Imaging Spectroradiometer (MODIS) instrument on the Aqua satellite on the right shows smoke from lightening-triggered wildfires. Credit: NASA Earth Observatory

For the climate, fires can directly and indirectly increase carbon emissions to the atmosphere. While they burn, fires release carbon stored in trees or in the soil. In some places like California or Alaska, additional carbon may be released as the dead trees decompose, a process that may take decades because dead trees will stand like ghosts in the forest, decaying slowly, Morton said. In addition to releasing carbon as they decompose, the dead trees no longer act as a carbon sink by pulling carbon dioxide out of the atmosphere. In some areas like Indonesia, Randerson and his colleagues have found that the radiocarbon age of carbon emissions from peat fires is about 800 years, which is then added to the greenhouse gases in that atmosphere that drive global warming. In Arctic and boreal forest ecosystems, fires burn organic carbon stored in the soils and hasten the melting of permafrost, which release methane, another greenhouse gas, when thawed.

Another area of active research is the mixed effect of particulates, or aerosols, in the atmosphere in regional climates due to fires, Randerson said. Aerosols can be dark like soot, often called black carbon, absorbing heat from sunlight while in the air, and when landing and darkening snow on the ground, accelerating its melt, which affects both local temperatures — raising them since snow reflects sunlight away — and the water cycle. But other aerosol particles can be light colored, reflecting sunlight and potentially having a cooling effect while they remain in the atmosphere. Whether dark or light, according to Randerson, aerosols from fires may also have an effect on clouds that make it harder for water droplets to form in the tropics, and thus reduce rainfall — and increase drying.

Fires of all types reshape the landscape and the atmosphere in ways that can resonate for decades. Understanding both their immediate and long-term effects requires long-term global data sets that follow fires from their detection to mapping the scale of their burned area, to tracing smoke through the atmosphere and monitoring changes to rainfall patterns.

"As climate warms, we have an increasing frequency of extreme events. It’s critical to monitor and understand extreme fires using satellite data so that we have the tools to successfully manage them in a warmer world," Randerson said.

Bloomberg Green, available at https://www.bloomberg.com/news/articles/2020-04-18/warmest-oceans-on-record-could-set-off-a-year-of-extreme-weather
(last accessed April 20, 2020)

Climate Adaptation

Warmest Oceans on Record Adds to Hurricanes, Wildfires Risks

By
Brian K Sullivan
April 18, 2020, 5:30 AM PDT *Updated on April 20, 2020, 8:03 AM PDT*

Pacific, Atlantic and Indian Oceans have reached record highs

Hurricanes, wildfires and severe thunderstorms all affected

The world's seas are simmering, with record high temperatures spurring worry among forecasters that the global warming effect may generate a chaotic year of extreme weather ahead.

Parts of the Atlantic, Pacific and Indian Oceans all hit the record books for warmth last month, according to the U.S. National Centers for Environmental Information. The high temperatures could offer clues on the ferocity of the Atlantic hurricane season, the eruption of wildfires from the Amazon region to Australia, and whether the record heat and severe thunderstorms raking the southern U.S. will continue.

In the Gulf of Mexico, where offshore drilling accounts for about 17% of U.S. oil output, water temperatures were 76.3 degrees Fahrenheit (24.6 Celsius), 1.7 degrees above the long-term average, said Phil Klotzbach at Colorado State University. If Gulf waters stay warm, it could be the fuel that intensifies any storm that comes that way, Klotzbach said.

"The entire tropical ocean is above average," said Michelle L'Heureux, a forecaster at the U.S. Climate Prediction Center. "And there is a global warming component to that. It is really amazing when you look at all the tropical oceans and see how warm they are."

The record warm water in the Gulf of Mexico spilled over into every coastal community along the shoreline with all-time high temperatures on land, said Deke Arndt, chief of the monitoring section at the National Centers for Environmental Information in Asheville, North Carolina. Florida recorded its warmest March on record, and Miami reached 93 degrees Wednesday, a record for the date and 10 degrees above normal, according to the National Weather Service.

While coronavirus has the nation's attenton right now, global warming continues to be a threat. Sea water "remembers and holds onto heat" better than the atmosphere, Arndt said.

Overall, the five warmest years in the world's seas, as measured by modern instruments, have occurred over just the last half-dozen or so years. It's "definitely climate-change related," said Jennifer Francis, a senior scientist at the Woods Hole Research Center in Massachusetts. "Oceans are absorbing about 90% of the heat trapped by extra greenhouse gases."

Worldwide, sea temperatures were 1.49 degrees Fahrenheit above average in March. That's the second highest level recorded since 1880 for the month of March, according to U.S. data. In 2016, temperatures were 1.55 degrees above average.

The first of Colorado State's 2020 storm reports, led by Klotzbach, forecast this year that eight hurricanes could spin out of the Atlantic with an above-average chance at least one will make landfall in the U.S. during the six-month season starting June 1. The U.S. is set to issue its hurricane forecast next month.

Arctic Systems

The searing global temperatures this year can also be traced back to intense climate systems around the Arctic that bottled up much of that region's cold, preventing it from spilling south into temperate regions.

Combined with global warming, this was a one-two punch for sea temperatures that's brought them to historic highs.

One of the best-known examples of how oceans drive global weather patterns is the development of the climate system known as El Nino. It occurs when unusually warm waters in the equatorial Pacific interact with the atmosphere to alter weather patterns worldwide. In the Atlantic, for instance, El Ninos can cause severe wind shear that can break up developing storms with the potential to become dangerous hurricanes.

This year, the chance of an El Nino developing is small, and scientists are theorizing one reason could be that climate change is warming all the world's oceans. El Nino "depends on contrasts, as well as absolute values of sea-surface temperatures," according to Kevin Trenberth, a scientist at the National Center for Atmospheric Research.

Strengthening Their Fury

Meanwhile, if the Atlantic stays warm through the six-month storm season that starts June 1, the tropical systems can use it as fuel to strengthen their fury. In 2017, a small storm called Harvey actually fell apart as it crossed Mexico's Yucatan Peninsula into the Gulf, but once it got there it reformed and grew into a Category 4 monster that went on to flood Texas, killing at least 68 people, and caused about $125 billion in damage.

If the Gulf stays record warm "then it raises the risk of another Harvey type storm perhaps," Trenberth said.

The oceans also play a role in setting the stage for wildfires. In the case of Australia and the Amazon, really warm areas of the ocean can pull rain away from the land, causing drier conditions and, in extreme cases, drought. Last year, for instance, the Indian Ocean was really warm off Africa, so that is where all the storms went. Australia was left high and dry.

Back in the Atlantic, research by Katia Fernandes, a geosciences professor at the University of Arkansas, has also shown a correlation between sea surface temperatures in the northern tropical Atlantic and drought and wildfires in the Amazon. The warmer the water, the further north rainfall is pulled across South America.

According to the Fernandes model, even Atlantic temperatures in March can serve to predict if the Amazon will be dry and susceptible to fires.

For California, the outlook isn't as clear. Wildfires there depend as much on how well vegetation grows, providing fuel for the flames, as it does on the weather conditions coming off the Pacific.

"Tricky question," said Mike Anderson, California state climatologist. "Our weather outcomes are influenced by sea-surface temperatures in the Pacific, but it depends on where and when the warm waters appear and how long they persist. In the end we have a highly variable climate that doesn't map in a statistically convenient way to patterns of sea-surface temperatures."

Climate-Driven Megadrought Is Emerging in Western U.S., Says Study

Warming May Be Triggering Era Worse Than Any in Recorded History

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BY KEVIN KRAJICK IAPRIL 16, 2020

With the western United States and northern Mexico suffering an ever-lengthening string of dry years starting in 2000, scientists have been warning for some time that climate change may be pushing the region toward an extreme long-term drought worse than any in recorded history. A new study says the time has arrived: a megadrought as bad or worse than anything even from known prehistory is very likely in progress, and warming climate is playing a key role. The study, based on modern weather observations, 1,200 years of tree-ring data and dozens of climate models, appears this week in the leading journal *Science.*

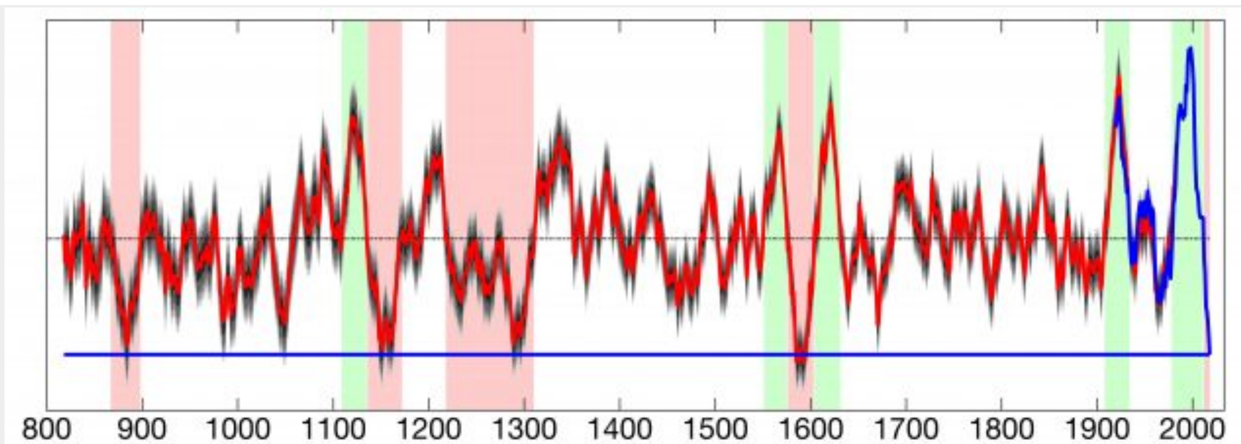
“Earlier studies were largely model projections of the future,” said lead author Park Williams, a bioclimatologist at Columbia University’s Lamont-Doherty Earth Observatory. “We’re no longer looking at projections, but at where we are now. We now have enough observations of current drought and tree-ring records of past drought to say that we’re on the same trajectory as the worst prehistoric droughts.”

Reliable modern observations date only to about 1900, but tree rings have allowed scientists to infer yearly soil moisture for centuries before humans began influencing climate. Among other things, previous research has tied catastrophic naturally driven droughts recorded in tree rings to upheavals among indigenous Medieval-era civilizations in the Southwest. The new study is the most up-to-date and comprehensive long-term analysis. It covers an area stretching across nine U.S. states from Oregon and Montana down through California and New Mexico, and part of northern Mexico.

Using rings from many thousands of trees, the researchers charted dozens of droughts across the region, starting in 800 AD. Four stand out as so-called megadroughts, with extreme aridity lasting decades: the late 800s, mid-1100s, the 1200s, and the late 1500s. After 1600, there were other droughts, but none on this scale.

The team then compared the ancient megadroughts to soil moisture records calculated from observed weather in the 19 years from 2000 to 2018. Their conclusion: as measured against the worst 19-year increments within the previous episodes, the current drought is already outdoing the three earliest ones. The fourth, which spanned 1575 to 1603, may have been the worst of all — but the difference is slight enough to be within the range of uncertainty. Furthermore, the current drought is affecting wider areas more consistently than any of the earlier ones — a fingerprint of global warming, say the researchers. All of the ancient droughts lasted longer than 19 years — the one that started in the 1200s ran nearly a century — but all began on a similar path to to what is showing up now, they say.

Nature drove the ancient droughts, and still plays a strong role today. A study last year led by Lamont's Nathan Steiger showed that among other things, unusually cool periodic conditions over the tropical Pacific Ocean (commonly called La Niña) during the previous megadroughts pushed storm tracks further north, and starved the region of precipitation. Such conditions, and possibly other natural factors, appear to have also cut precipitation in recent years. However, with global warming proceeding, the authors say that average temperatures since 2000 have been pushed 1.2 degrees C (2.2 F) above what they would have been otherwise. Because hotter air tends to hold more moisture, that moisture is being pulled from the ground. This has intensified drying of soils already starved of precipitation.



Varying soil moisture in southwestern North America, 800-2018. The straight horizontal center line indicates average moisture; blue line at bottom shows 2000-2018 mean. Green bars indicate abnormally wet periods, pink ones abnormally dry. The fluctuating red moisture line is based on tree-ring data until it converts to blue at the start of modern instrumental observations. (Adapted from Williams et al., Science, 2020)

All told, the researchers say that rising temperatures are responsible for about half the pace and severity of the current drought. If this overall warming were subtracted from the equation, the current drought would rank as the 11th worst detected — bad, but nowhere near what it has developed into.

"It doesn't matter if this is exactly the worst drought ever," said coauthor Benjamin Cook, who is affiliated with Lamont and the Goddard Institute for Space Studies. "What matters is that it has been made much worse than it would have been because of climate change." Since temperatures are projected to keep rising, it is likely the drought will continue for the foreseeable future; or fade briefly only to return, say the researchers.

“Because the background is getting warmer, the dice are increasingly loaded toward longer and more severe droughts,” said Williams. “We may get lucky, and natural variability will bring more precipitation for a while. But going forward, we’ll need more and more good luck to break out of drought, and less and less bad luck to go back into drought.” Williams said it is conceivable the region could stay arid for centuries. “That’s not my prediction right now, but it’s possible,” he said.

Lamont climatologist Richard Seager was one of the first to predict, in a 2007 paper, that climate change might eventually push the region into a more arid climate during the 21st century; he speculated at the time that the process might already be underway. By 2015, when 11 of the past 14 years had seen drought, Benjamin Cook led a followup study projecting that warming climate would cause the catastrophic natural droughts of prehistory to be repeated by the latter 21st century. A 2016 studycoauthored by several Lamont scientist reinforced those findings. Now, says Cook, it looks like they may have underestimated. “It’s already happening,” he said.

The effects are palpable. The mighty reservoirs of Lake Mead and Lake Powell along the Colorado River, which supply agriculture around the region, have shrunk dramatically. Insect outbreaks are ravaging dried-out forests. Wildfires in Californiaand across wider areas of the U.S. West are growing in area. While 2019 was a relatively wet year, leading to hope that things might be easing up, early indications show that 2020 is already on a track for resumed aridity.

In the Catalina Mountains in southern Arizona, forests struggle to keep up with recent increases in drought and wildfire activity, which are expected to continue due to human-caused climate change. (Park Williams/Lamont-Doherty Earth Observatory)

“There is no reason to believe that the sort of natural variability documented in the paleoclimatic record will not continue into the future, but the difference is that droughts will occur under warmer temperatures,” said Connie Woodhouse, a climate scientist at the University of Arizona who was not involved in the study. “These warmer conditions will exacerbate droughts, making them more severe, longer, and more widespread than they would have been otherwise."

Angeline Pendergrass, a staff scientist at the U.S. National Center for Atmospheric Research, said that she thinks it is too early to say whether the region is at the cusp of a true megadrought, because the study confirms that natural weather swings are still playing a strong role. That said, "even though natural variability will always play a large role in drought, climate change makes it worse," she said.

Tucked into the researchers' data: the 20th century was the wettest century in the entire 1200-year record. It was during that time that population boomed, and that has continued. "The 20th century gave us an overly optimistic view of how much water is potentially available," said Cook. "It goes to show that studies like this are not just about ancient history. They're about problems that are already here."

The study was also coauthored by Edward Cook, Jason Smerdon, Kasey Bolles and Seung Baek, all of Lamont-Doherty Earth Observatory; John Abatzaglou of the University of Idaho; and Andrew Badger and Ben Livneh of the University of Colorado Boulder.

The study:

Large contribution from anthropogenic warming to an emerging North American megadrought

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A trend of warming and drying

Global warming has pushed what would have been a moderate drought in southwestern North America into megadrought territory. Williams *et al.* used a combination of hydrological modeling and tree-ring reconstructions of summer soil moisture to show that the period from 2000 to 2018 was the driest 19-year span since the late 1500s and the second driest since 800 CE (see the Perspective by Stahle). This appears to be just the beginning of a more extreme trend toward megadrought as global warming continues.

Science, this issue p. **314**; see also p. **238**

Abstract

Severe and persistent 21st-century drought in southwestern North America (SWNA) motivates comparisons to medieval megadroughts and questions about the role of anthropogenic climate change. We use hydrological modeling and new 1200-year tree-ring reconstructions of summer soil moisture to demonstrate that the 2000–2018 SWNA drought was the second driest 19-year period since 800 CE, exceeded only by a late-1500s megadrought. The megadrought-like trajectory of 2000–2018 soil moisture was driven by natural variability superimposed on drying due to anthropogenic warming. Anthropogenic trends in temperature, relative humidity, and precipitation estimated from 31 climate models account for 47% (model interquartiles of 35 to 105%) of the 2000–2018 drought severity, pushing an otherwise moderate drought onto a trajectory comparable to the worst SWNA megadroughts since 800 CE.

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Fire weather drives daily area burned and observations of fire behavior in mountain pine beetle affected landscapes

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Keywords: wildfire, dendrocotnus ponderosae, disturbance interactions, bark beetle, lodgepole pine, fire weather

Supplementary material for this article is available online

Abstract

In the western United States, mountain pine beetles (MPBs) have caused tree mortality across 7% of the forested area over the past three decades, leading to concerns of increased fire activity in MPB-affected landscapes. While fire behavior modeling suggests MPB-associated changes in fuels may influence fire behavior, retrospective studies have generally found negligible or weak effects of pre-fire MPB outbreak on fire activity. This apparent disagreement may arise from differences in fire weather, fuels, or scale and highlights the need for empirical studies that examine the influence of MPB outbreak on fire activity at finer spatiotemporal scales. Here we use a novel combination of geospatial data and firefighter observations to test the relative influences of red and gray stage MPB outbreak on two measures of daily wildfire activity—daily area burned (DAB) and observed fire behavior. We analyzed 2766 large wildfires that burned in the West over the 2003–2012 period. We found 329 fires that intersected prior MPB outbreak, however most burned in areas affected by MPB for only a few days (median = 4 d). We modeled DAB and the occurrence of observation of high-extreme fire behavior in 57 large (>1140 ha) wildfire events that burned for long time periods (>10 d) in landscapes affected by MPB. Under these conditions, we found no effect of red or gray stage MPB outbreak on either DAB or observed fire behavior. Instead, greater DAB and observations of high-extreme fire behavior occurred during warmer, drier, and windier weather conditions and where pre-outbreak fuels were characterized by lower canopy base heights and greater canopy bulk densities. The overriding influence of weather and pre-outbreak fuel conditions on daily fire activity observed here suggest that efforts to reduce the risk of extreme fire activity should focus on societal adaption to future warming and extreme weather.

1. Introduction

Wildfire and bark beetle outbreaks are key disturbances affecting forests across the western United States, where they have respectively affected 6.3% and 7.1% of the forested area over the past three decades (Hicke *et al* 2016). The coincidence of extensive wildfire and bark beetle outbreak has led to widespread concern that bark beetle-induced tree mortality may exacerbate fire activity (Hicke *et al* 2012, Jenkins *et al* 2014). These concerns are driving costly federal policy decisions and forest management actions across millions of acres of National Forest System lands (Agricultural Act of 2014). Yet at the broad spatial scales most relevant to forest management and policy, empirical evidence for increased fire activity in bark beetle-affected forests is lacking (Harvey *et al* 2014b, Hart *et al* 2015, Meigs *et al* 2016).

Across the Western US, the mountain pine beetle (*Dendroctonus ponderosae*; hereafter 'MPB') has caused most of the recent bark beetle-induced tree mortality, primarily in lodgepole pine (*Pinus contorta)* and ponderosa pine (*Pinus ponderosa*) (Negrón and Fettig 2014). Adult MPBs inhabit a tree's inner bark, where they oviposit and the resulting larvae feed on phloem tissues. Extensive colonization and reproduction by MPBs lead to tree death. MPBs typically attack weakened trees, however outbreaks may occur when beetle populations rapidly grow in response to favorable effects of temperature on beetle development rates and drought stress in host trees (Raffa *et al* 2008, Chapman *et al* 2012).

Tree mortality caused by MPB infestation is expected to alter fire activity by changing the flammability, continuity, and structure of fuels (Hicke *et al* 2012, Jenkins *et al* 2014). Initially following death, trees enter the '*red stage*', which is characterized by declines in leaf moisture content and changes in chemical composition that increase flammability (Jolly *et al* 2012, Page *et al* 2012). Predictive models of fire behavior suggests these changes may promote crown fire (Page and Jenkins 2007, Schoennagel *et al* 2012, Hoffman *et al* 2012, 2013, 2015). About three years following initial attack, needles drop, and trees move into the '*gray stage.*' During the gray stage, canopy bulk density and continuity are reduced (Klutsch *et al* 2009, Simard *et al* 2011, Schoennagel *et al* 2012), fuels build up on the forest floor due to falling needles, branches, and/or release of understory vegetation (Klutsch *et al* 2009), and the forest microclimate changes from the loss of live tree crowns (Simard *et al* 2011). In fire behavior models these changes generally lead to decreased crown fire potential (Page and Jenkins 2007, Klutsch *et al* 2011, Simard *et al* 2011, Schoennagel *et al* 2012, Hoffman *et al* 2015). However, fire behavior modeling also shows that pre-outbreak stand structure and fuels, weather conditions, temporal and spatial patterns of tree mortality, and surface fireline intensity exert an important influence on fire behavior in MPB affected stands (Page and Jenkins 2007, Klutsch *et al* 2011, Simard *et al* 2011, Schoennagel *et al* 2012, Hoffman *et al* 2012, 2013, 2015).

Retrospective studies have generally found negligible or weak effects of pre-fire MPB outbreak on fire occurrence (Kulakowski and Jarvis 2011, Mietkiewicz and Kulakowski 2016), area burned (Hart *et al* 2015, Meigs *et al* 2015) and fire severity (Harvey *et al* 2014a, 2014b, Meigs *et al* 2016, Talucci and Krawchuk 2019). Effects of pre-fire MPB outbreak are also thought to depend on daily weather conditions (Harvey *et al* 2014a, 2014b, Agne *et al* 2016). However, there are few empirical studies of the effects of MPB outbreak on daily fire activity, largely due to difficulties associated with collecting data in active wildfire. Perrakis *et al* (2014) observed greater rates of spread in red and gray stage MPB than unaffected forests, however analyses were limited to 16 observations. Results from nine paired experimental burns in control and simulated MPB-killed stands revealed no effect of simulated red-stage MPB outbreak on fire behavior (Schroeder and Mooney 2012). Yet, in interviews with 28 wildland firefighters, Moriarty *et al* (2019) found amplified fire behavior in the red stage of outbreak. These uncertainties highlight the need for a broad-scale analysis of the relative influence of MPB outbreak on daily wildfire activity.

We used a novel approach that combined geospatial data and wildland firefighter observations. Specifically, we used satellite data to generate maps of daily fire progression and linked these data to daily Incident Status Summary reports (ICS-209 reports) filed by wildland firefighters. ICS-209 reports are filed for all large wildfires on lands managed by the US Federal agencies and detail the daily fire size and observed fire behavior. Reports provide coarse-grain daily 'snapshots' of the management situation and are critical in determining resource allocation (National Intelligence Working Group 2017). ICS-209 reports have been used to make inferences about containment probability (Finney *et al* 2009) and biomass emissions (Pouliot *et al* 2008), but to our knowledge have not been used to analyze descriptions of observed fire behavior. We focused on three questions: (1) At the daily timescale, how frequently did fires burn in areas affected by MPB? (2) In large wildfire events, is MPB outbreak associated with greater daily area burned (DAB) and/or the occurrence of days with high-extreme fire behavior as reported by wildland firefighters? and (3) Do the potential effects persist after controlling for biophysical factors known to affect fire activity? Additionally, we examined the relationship between remotely sensed DAB and wildland firefighter observations of fire behavior.

2. Methods

2.1. Data

We first obtained spatial data on large wildfire (>405 ha) extent for all fires that burned during the 2003–2012 period in the contiguous western United States (figure 1) from the Monitoring Trends in Burn Severity Project (MTBS Project 2017). To characterize daily fire growth, we obtained fire progression polygons produced by incident management teams from the Geospatial Multi-Agency Coordination group (Geospatial Multi-agency Coordinating Group GeoMAC 2019). Consistent with previous retrospective studies of wildfire (e.g. Collins *et al* 2009, Parks *et al* 2015), we found data on fire growth from GeoMAC was characterized by many temporal gaps, thereby limiting our ability to quantify DAB. We therefore followed methods outlined in Parks (2014) and created maps of day-of-burning (DOB) for each fire by spatially interpolating Moderate Resolution Imaging Spectrometer (MODIS) active fire-detection data (NASA MCD14ML product, Collection 5, Version 1), which depicts the location of actively burning 1 × 1 km MODIS pixels (figure S1 is available online at stacks.iop.org/ERL/15/054007/mmedia). We used the high temporal resolution of MODIS to estimate DOB at higher spatial resolutions (30 × 30 m) using a weighted by mean and distance spatial interpolation approach (Parks 2014). Finally, we converted gridded

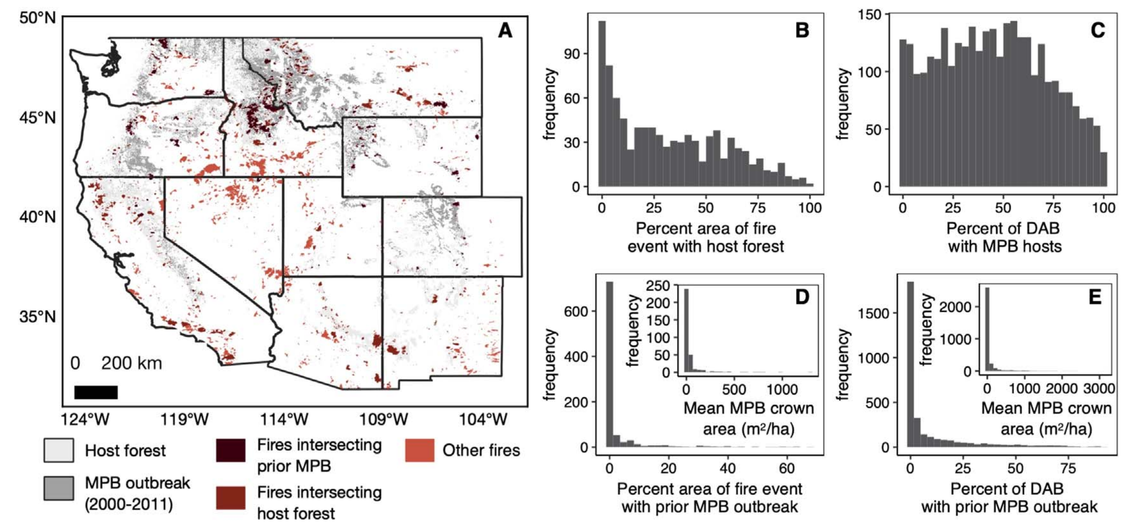

Figure 1. The study area (A) and frequency distributions of the percent area of the fire event and daily fire progression polygon that intersected mountain pine beetle (MPB) host forest (B) and (C) and prior MPB outbreak (D) and (E). The insets show frequency distribution for mean crown area of prior MPB outbreak within an event and daily fire progression polygon. Note the area of mean crown area of prior MPB outbreak represents a conservative estimate.

day-of-burning data to polygons depicting the daily fire progression.

Maps of DOB are difficult to validate because of the lack of ground-truthed data, particularly in large wildfires. Parks (2014) found GeoMAC and MODIS-based maps of DOB, matched exactly for 46.1% of cases, while 75% of cases were within 1 d. However, disagreement arises from errors in both datasets. To determine if our results were sensitive to the use of MODIS-based maps of fire progression, we compared our findings to results obtained using the subset of wildfires with both GeoMAC- and MODIS-based maps of daily fire progression (*Supplement*).

To characterize daily fire behavior, we obtained daily ICS-209 reports from the National Fire and Aviation Management (2019) Web Application. ICS-209 reports are filed daily by incident management teams, who record the cumulative area burned, observed fire behavior, and projected fire behavior, among other things. For each day, we extracted the observed fire behavior descriptions which were classified following the Fire Behavior Reference Guide (NWCG 2017), where possible values included (1) smoldering, (2) creeping, (3) running, (4) torch/spot, (5) crowning, and (6) erratic and extreme (table 1). To do so, we compiled a list of terms associated with each category from the Fire Behavior Reference Guide (NWCG 2017) (tables 1 and S3) and classified the text in R (R Core Team 2017). In the dataset examined here, the frequency of these six groups was extremely unequal (figure 2). Therefore, we combined smoldering, creeping, running, and torch/spot classes into a low-moderate fire behavior category. The crowning and erratic and extreme classes were grouped into a high-extreme category (table 1). Where observations indicated varied fire behavior, we assigned the highest category.

To characterize MPB outbreak, we first defined the presence of MPB host-species using three forest cover-type spatial datasets: Landfire Existing Vegetation Type (LANDFIRE 2001a), National GAP Landcover data (LANDFIRE 2011), and a map of US forest types produced by Zhu and Evans (1994). We created 30×30 m grids of the presence of MPB host tree species (see table S2) by defining presence as any cell where two or more datasets were in agreement (e.g. Preisler *et al* 2012). We then aggregated the data by calculating the percent of host cover within a 990×990 m cell, a spatial grain chosen to match the coarse-scale of forest disturbance data from the United States Forest Service (USFS) Aerial Detection Survey (ADS) program (Johnson and Ross 2008). We acquired all ADS describing the extent and estimated severity (e.g. damaged trees per hectare) of MPB infestation across the western US over the 2000–2011 period (USFS and its partners 2017). Approximating the methods outlined by Meddens *et al* (2012), we first converted annual (2000–2011) ADS polygon data to a 990×990 m raster by first calculating the percent of each pixel that intersected an MPB damage polygon. We then constrained the percent MPB damage of each pixel so that it could not exceed the percent MPB host forest by overlaying the percent host and percent MPB damage rasters. Thus, the rasters created here are conservative estimates of where outbreak is most likely to have occurred. Next, we multiplied the percent MPB damage rasters by the estimated number of affected trees per hectare to generate grids of the number of damaged trees per hectare. These grids were converted to crown area per hectare by multiplying by the average tree crown diameter for each host tree species, which we obtained from Meddens *et al* (2012).

Table 1. Categories, descriptions and search terms for categorizing observed fire behavior descriptions from ICS-209 reports.

Fire behavior class	General fire behavior[a]	Description[a]	Example search terms[b]
Low-moderate	Smoldering	• White smoke • Smoldering ground fire • No open flame	• 'smoldering' • 'no open flame'
	Creeping	• Visible open flame (1–4 ft.) • Surface fire only • Unorganized flame front • Little or no spread	• 'creeping' • 'surface' • 'minimal spread'
	Running	• Organized surface flame front (4–8 ft.) • Moderate rate of spread • Torching and short-range spotting • Some candling	• 'running' • 'moderate fire behavior' • 'moderately active'
	Torch/spot	• Organized surface flame front (8–12 ft.) • Moderate to fast rate of spread • Gray to black smoke Torching • Short range spotting	• 'spotting' • 'torching' • 'fast rate of spread'
High-extreme	Crowning	• Organized crown fire front • Moderate to long range spotting • Fast rate of spread • Independent spot fire growth • Black to copper smoke	• 'Long distance spotting' • 'Long range spotting' • 'Crown' • 'Active fire behavior'
	Extreme and erratic	• Independent spot fire growth • Development of pyrocumulus clouds • Presence of fire whirls • Violent fire behavior	• 'Extreme' • 'Erratic' • 'Plume' • 'Fire whirl'

[a] Categories of fire behavior and their descriptions are adapted from the Fire Behavior Reference Guide (NWCG 2017).
[b] For a full list of search terms, see table S1.

Finally, we created rasters depicting the stage of outbreak, where we defined red-stage outbreak as the total MPB crown area mapped within two years of the fire year and gray-stage outbreak as the total MPB crown area within three or more years prior to the fire year (e.g. Hart *et al* 2015). We assume the resulting datasets describe the relative amount of MPB activity within a cell, but caution that aerial sketchmapping is subjective and often underestimates the severity of tree mortality (Johnson and Wittwer 2008, Meddens *et al* 2012, Hicke *et al* 2016).

To determine the relative influence of MPB outbreak on DAB and observed fire behavior, we acquired spatial data for eleven independent biophysical predictors (table S3). We selected variables that are known to influence fire behavior, including fuels, topography, and weather (Agee 1993). To characterize daily fire weather, we used daily weather data (collected hourly) from the nearest Remote Automated Weather Station (table S5) describing the daily maximum temperature, average relative humidity, and wind gust speed. To describe fuel moisture, we obtained 4×4 km daily grids of Energy Release Component (ERC), a composite fuel moisture index that integrates the effects of weather conditions on the focal day and preceding seven days (Abatzoglou 2013). We characterized fuel characteristics using a 30×30 m raster of pre-outbreak canopy bulk density (CBD) and canopy base height (CBH) (LANDFIRE 2001b, 2001c), factors that are known to affect crowning. Finally, to control for potential differences in fire activity between MPB hosts, we used 30×30 m rasters listing the presence/absence of lodgepole pine and ponderosa pine. To represent topography, we obtained 30×30 m rasters of slope and elevation (LANDFIRE 2013). We assigned values of each predictor to each daily fire progression polygon by calculating the mean value of the raster cells overlapping the DOB polygon. Prior to modeling, we used pairwise correlations to detect potential multicollinearity issues associated with the predictor variable set (figure S2).

2.2. Analyses

To better understand how frequently wildfires intersected prior MPB outbreak, we first used GIS to determine the number of wildfires that intersected MPB host forest and prior MPB outbreak for all wildfires. For wildfires that intersected MPB outbreak, we also calculated the number of daily fire progression polygons that intersected prior MPB outbreak and the mean MPB crown area.

To determine if prior MPB outbreak was associated with increased DAB or high-extreme observed fire behavior, we selected the fires where more than 50% of the daily fire progression days intersected prior MPB outbreak. To determine if the 50% threshold for

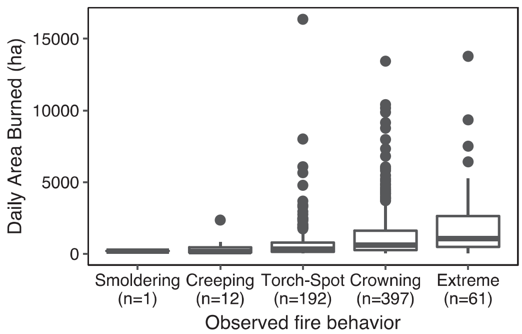

Figure 2. Boxplots displaying the DAB in each observed fire behavior class. Data shown are from the 41 fires and 663 d with coincident MODIS and ICS-209 data. The horizontal line is the median, 25th and 75th percentiles are box edges, vertical lines show 1.5 times the interquartile range, and points are outliers.

selecting fires influenced our results, we additionally performed all analyses using both a 25% and 75% threshold (see Supplement). We further constrained our analysis to only wildfires with ⩾10 d with fire progression data, which allowed for enough replication within each fire to treat fire identity as a random effect in analyses (described below) (Bolker *et al* 2009). We examined the effect of pre-fire MPB outbreak on observed fire behavior for the subset of these burning days where data on observed fire behavior was recorded and could be classified for both the current and proceeding day, which allowed for the inclusion of a temporal autocovariate term (table S4). As a consequence, our analysis focuses on large wildfires that burned over long time periods, which tend to have the greatest social and ecological consequences.

To understand if fire growth was associated with firefighter observations, we first compared DAB with classified observed fire behavior. Specifically, we modeled the occurrence of high-extreme fire versus low-moderate fire behavior as function of DAB using a generalized linear model (GLM) with a binomial distribution and logit link function. To account for differences between fire events and temporal autocorrelation, we also included an autocovariate term—the prior day's observed fire behavior. We assessed significance using a likelihood ratio test that compared the model with DAB as predictor to the intercept only model (Zuur *et al* 2009).

To determine if MPB outbreak was associated with either increased DAB or occurrence of high-extreme rather than moderate-low fire behavior, we used two approaches. First, we constructed univariate models, where the only predictor was either the crown area of red or gray stage MPB outbreak. As above, models of observed fire behavior were constructed using a GLM with a binomial distribution (logit link) and included an autocovariate term. We used a linear mixed effects (LME) model with a random intercept term for fire identity and a first order autocorrelation structure nested within fire identity to model DAB (log transformed to improve normality). We fit LME models using the R package 'nlme' (Pinheiro *et al* 2017).

Next we constructed multivariate models using the same modeling frameworks as in the bivariate analyses and including the eleven other biophysical predictors of fire activity. We used a model selection approach to determine the influence of MPB outbreak relative to other drivers of daily fire activity. Prior to model construction, all predictor variables were z-score transformed to allow for comparison. We started with a model with all possible predictors and used likelihood ratio tests to evaluate the effect of dropping each variable from the model (Zuur *et al* 2009). Non-significant ($p > 0.05$) terms were removed until all remaining variables were significant. Variable selection removed both red and gray stage MPB outbreak from models of observed fire behavior and DAB, but we forced these variables into the final model to evaluate their relative influence as indicated by the model coefficients. Because the effects of MPB outbreak on fire activity have been hypothesized to only occur during specific conditions (Harvey *et al* 2014a), we also tested for interactions between MPB outbreak variables and significant biophysical variables using likelihood ratio tests. Overall explanatory power of the most parsimonious model was evaluated using the marginal coefficient of determination (R^2), which reflects the variance explained by the fixed effects (Nakagawa and Schielzeth 2013).

3. Results

We examined 2766 large wildfires that burned in the western US during the 2003–2012 period (figure 1(A)). Of these fires, 916 (33%) burned in MPB host forest and 329 (12%) burned in areas with prior MPB outbreak. When fires burned in areas with MPB host forest, typically less than 30% of the total fire area was MPB host forest (median = 27%) and less than 3% of the total fire area was affected by prior MPB outbreak (figure 1). At the daily scale, we were able to use MODIS data to produce at least one fire progression perimeter for 92% of the fire events that intersected MPB affected forest ($n = 302$). The resulting dataset consisted of 3501 daily fire progression polygons, of which 91% ($n = 3191$) intersected MPB host forest and 58% ($n = 2030$) burned in areas with prior MPB outbreak (figures 1(C) and (E)). Of the 302 fires intersecting prior MPB outbreak, 60% ($n = 192$) had a fewer than 10 daily fire progression polygons (median = 8) and only a few days (median = 4 d) intersected prior MPB outbreak. Of the fires intersecting prior MPB outbreak that also burned for at least 10 d, 43% ($n = 57$) burned in MPB-affected landscapes for at least half of the burning days ($n = 1318$ daily fire progression polygons) (table S4).

Coincident ICS-209 data and MODIS-based fire progression polygons were available for 41 fires and

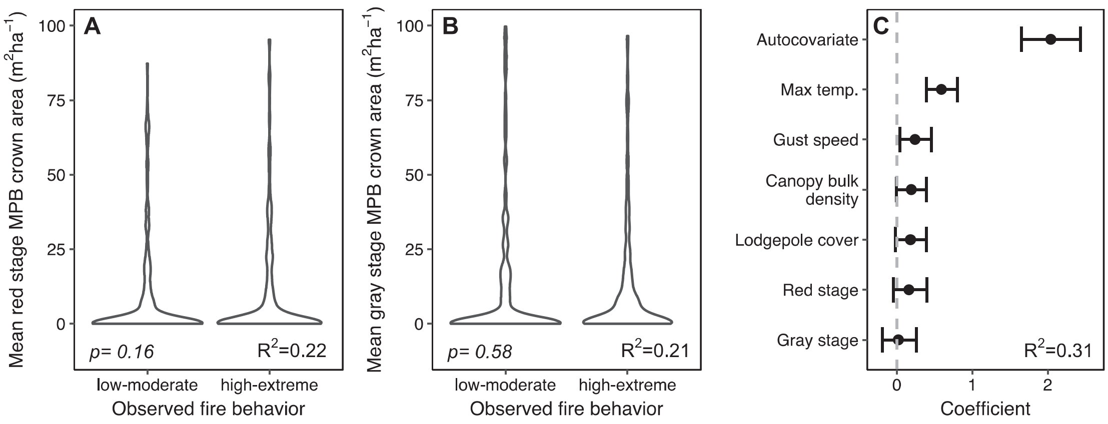

Figure 3. The effect of MPB outbreak on observed fire behavior. Density plots illustrate the univariate associations between observed fire behavior and mean red stage MPB (A) and gray stage MPB (B). The right panel shows regression estimates for the top performing multivariate model of observed fire behavior (C). Whiskers illustrate the 95% confidence intervals for the regression estimates. Note that confidence intervals for MPB associated coefficients overlap with zero. Data shown are from the 41 fires and 663 d with coincident MODIS and ICS-209 data.

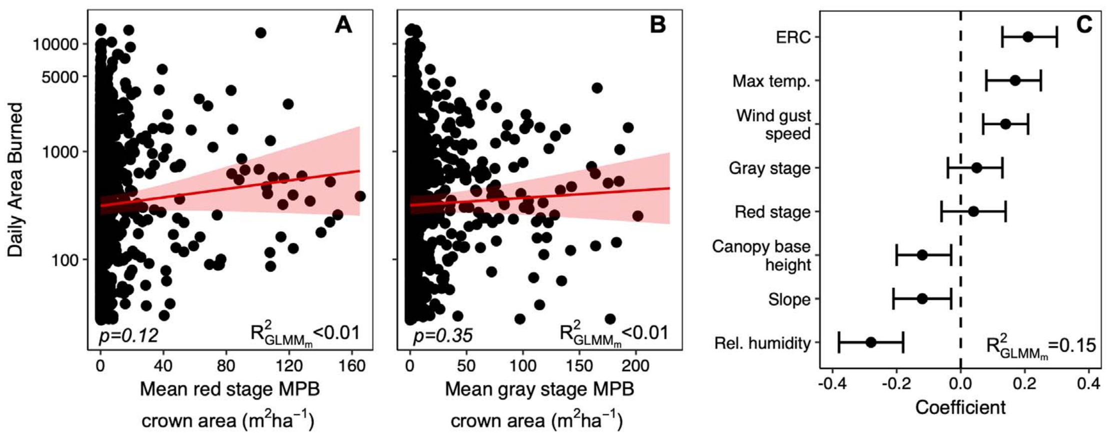

Figure 4. The effect of MPB outbreak on daily area burned (DAB). Scatterplots illustrate the univariate associations between DAB and mean red stage MPB (A) and gray stage MPB (B). The solid red line illustrates predicted values for the population based on the fixed effect estimates. Red shading shows the 95% confidence interval for the population prediction conditional on estimates of the random effects. *P*-values are from likelihood ratio tests. $R^2_{GLMM_m}$ and $R^2_{GLMM_c}$ are the marginal and conditional coefficients of determination, respectively. The right panel shows regression estimates for the top performing multivariate model of observed fire behavior (C). Whiskers illustrate the 95% confidence intervals for the coefficient estimates. Data shown are for the 57 wildfires that burned in MPB-affected landscapes for at least half of the burning days ($n = 1318$ daily fire progression polygons).

663 d (table S4). Data was generally available in both datasets when fires were large (>500 ha) and observed fire behavior was more extreme (figure 2). Consistent with the expectation of more rapid growth during periods of more extreme fire behavior, increases in DAB were associated with the occurrence of high-extreme (i.e. crowning or extreme) fire behavior.

We found observations of high-extreme fire behavior in daily ICS-209 reports and greater DAB often occurred in areas with greater red stage MPB mortality area (figures 3–4). However, neither red nor gray stage MPB were significantly related to either the occurrence of high-extreme fire behavior or DAB ($p > 0.15$; figures 3–4). Around 20% of the variance in observed fire behavior was explained by the MPB variables and autocovariate combined. Less than 1% of the variance in DAB was explained by MPB variables ($R^2_{GLMM_m} \leqslant 0.01$; figures 4(A)–(B)). These results were not sensitive to either the source of daily fire progression data nor the choice of threshold used to select fires for our analyses (figures S2–S7).

The top performing multivariate model of observed fire behavior explained about 30% of the variance ($R^2 = 0.31$) and included maximum temperature, gust speed, pre-outbreak canopy bulk density, and percent lodgepole pine cover (figure 3(C), tables S6–S7). More extreme fire behavior occurred when maximum daily temperatures were warmer, gust speeds were higher, and average pre-outbreak canopy bulk density was greater (figure 3(C), table S6). We found the top performing model of DAB explained about 30% of the variance ($R^2_{GLMM_c} = 0.30$) about half of which was attributed to the predictor variables ($R^2_{GLMM_m} = 0.15$). Greater DAB was associated with higher ERC, warmer temperatures, windier conditions, lower canopy base height, flatter terrain, and lower relative humidity (figure 4(C), table S6). The inclusion of interaction terms between MPB outbreak and biophysical variables did not significantly improve performance of either observed fire behavior or DAB models (table S8). Similarly, average red or gray stage MPB outbreak were not important predictors of DAB or observed fire behavior when the threshold used to select fires was higher or GeoMAC data was used to map fire progression (figures S3–7).

4. Discussion

Using a combination of geospatial data and firefighter observations, we found no effect of prior MPB outbreak on DAB or the occurrence of observed high-extreme fire behavior in 1318 burning days occurring in 57 large wildfire events. Instead, the occurrence of high-extreme fire behavior and greater DAB were associated with the burning conditions of the previous day, weather conditions, pre-outbreak fuel conditions, and to a lesser extent, topography. These findings are consistent with experimental burning research that showed fire behavior increased with fire weather but not simulated MPB-kill conditions (Schroeder and Mooney 2012). Our results suggest that during drought conditions that promote extensive wildfire across the western US (Dennison *et al* 2014), short-term fluctuations in weather are likely more important than MPB-alterations to fuels at moderate spatiotemporal (daily) scales.

Consistent with previous work showing negligible effects of prefire-MPB on area burned (Hart *et al* 2015, Meigs *et al* 2015), our results indicate that few wildfires burned extensively in MPB-affected landscapes during the a decade of widespread fire and MPB outbreak. The limited overlap between MPB-affected forest and wildfire reflects the heterogenous environments where western wildfires burn—even when fires intersected MPB host forest, most fires burned in areas composed of diverse plant communities. Moreover, here we show that when wildfires intersected prefire-MPB, the area burned in MPB outbreak accounted for a small proportion of the total area burned. At the daily scale, the overlap of MPB outbreak and wildfire disturbance typically occurred for a small proportion of the DAB and a relatively low crown area of MPB mortality. Given the limited spatial and temporal coincidence of MPB and wildfire, post-outbreak fuels treatments may be of limited efficacy.

We found firefighter observations of daily fire behavior and fire growth agree with MODIS-based reconstructions of DAB. Additionally, we found similar models of DAB using maps of fire progression from incident management teams (e.g. GeoMAC) and MODIS active fire detection data. However, because MODIS data does not lack temporal gaps it provides key information about daily fire activity (Parks 2014). These findings support the use of MODIS daily fire activity data in reconstructing daily fire growth. However, we note that MODIS data is biased toward differences days of large fire growth and results should be interpreted in this context.

Here we examined the effect of MPB outbreak on two measures of daily fire activity in large wildfire events. At finer spatial and temporal scales, anecdotal observations and qualitative interviews with firefighters suggest that outbreaks may affect fire behavior (Moriarty *et al* 2019). Subsequent analyses where the spatial and temporal heterogeneity of MPB outbreak is better characterized may provide critical insights into this apparent incongruity. Additionally, our analyses use average conditions to characterize fuels, weather, and topography, the role of variability and extreme values at the daily scale requires more research. Further our analysis targeted large wildfires and days of notable fire growth (minimum daily area burned = 26.7 ha), when the potential effects of MPB on fuels are expected to be less important. Under more moderate weather conditions and finer spatiotemporal scales, the potential effects of bark beetle outbreaks on fire behavior may be important and warrant caution from a firefighting perspective (Jenkins *et al* 2014).

In addition to finding no significant effect of prefire MPB outbreak on observed fire behavior, we found strong influences of daily weather variability on observed fire behavior and daily fire growth (DAB). Weather and climate variability strongly influence fire occurrence (Dennison *et al* 2014), annual area burned (Littell *et al* 2009, Abatzoglou and Kolden 2013), fire severity (Holden *et al* 2007), and average fire size (Miller *et al* 2012) across the western United States. Yet the effects of climate conditions on individual fire sizes are less clear. For instance, Harvey *et al* (2016) found fire size in the Northern Rockies was weakly associated with cumulative moisture deficit at the time of fire and moisture deficit during the burning period. Yet Riley *et al* (2013) found individual fire sizes across the contiguous western United States were not strongly related to long- (>6 months) or moderate-term (7 d to 3 months) drought indices. Here, we found daily fire growth was sensitive to both daily weather variability (maximum temperature, gust speed, and relative humidity) and moderate-term drought (ERC), supportive of the idea that variation in weather is a key driver of fire size. Thus, predictions of future wildfire should incorporate both the effects of slowly changing broad-scale climate, which promote periods of widespread wildfire, and extreme weather events, which lead to rapid periods of fire growth.

5. Conclusion

At a moderate spatiotemporal scale, both daily fire growth (DAB) and observed fire behavior, as recorded in ICS-209 reports, were driven by fire weather, not

MPB outbreak in 56 large wildfire events that burned across the West during the 2003–2012 period. Given the relative rarity of wildfire burning in MPB-affected forests and negligible effects on daily fire activity, post-outbreak management strategies should emphasize mitigation of other negative effects on socioecological systems, including diminished tourism, tree-fall hazards, and effects on wildlife habitat (Morris *et al* 2018). In general, efforts to reduce the risk of extreme fire behavior should focus on societal adaption to future warming and extreme weather events.

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Data availability statement

The data that support the findings of this study are openly available. ADS data is available from the USFS and its partners (2017) at https://fs.fed.us/foresthealth/applied-sciences/mapping-reporting/detection-surveys.shtml. LANDFIRE vegetation (LADNFIRE 2001a), canopy base height (LANDFIRE 2001c), canopy bulk density (LANDFIRE 2001b), and elevation (LANDFIRE 2013) data are available online at www.landfire.gov. GAP vegetation data are available from the USGS (2011) at 10.5066/F7ZS2TM0. RAWS data is available from the Western Regional Climate Center (2019) at https://raws.dri.edu/. The ERC data (Abatzoglou 2013) is at http://climatologylab.org/gridmet.html. Fire perimeter data from the MTBS project (2017) and Geospatial Multi-Coordination Agency (GeoMac) (2019) are available online at www.mtbs.gov/direct-download and https://rmgsc.cr.usgs.gov/outgoing/GeoMAC/, respectively. ICS-209 reports can be downloaded from the National Fire and Aviation Management (2019) at fam.nwcg.gov/fam-web/. MODIS active fire location data is available from NASA (2019) at https://earthdata.nasa.gov/active-fire-data.

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United States Department of Agriculture

RESOURCE UPDATE FS-227

Greenhouse Gas Emissions and Removals from Forest Land, Woodlands, and Urban Trees in the United States, 1990-2018

Introduction

As a signatory to the United Nations Framework Convention on Climate Change (UNFCCC), the United States has been reporting an economy-wide Inventory of greenhouse gas (GHG) emissions and removals since the mid-1990s (US EPA 2020). Forest land, harvested wood products (HWPs), and urban trees within the land sector collectively represent the largest net carbon (C) sink in the United States, offsetting more than 11 percent of total GHG emissions annually (US EPA 2020). Estimates of GHG emissions and removals are compiled by U.S. Department of Agriculture (USDA) Forest Service researchers and are based primarily on National Forest Inventory (NFI) data collected and maintained by the Forest Inventory and Analysis (FIA) program within the USDA Forest Service. This report—the second in a new series of annual updates—provides an overview of the status and trends of GHG emissions and removals from forest land, woodlands in the grassland category, HWPs, and urban trees in settlements in the United States from 1990 to 2018. The estimates for the United States summarized here are based on the compilation reported in the *Land Use, Land-Use Change, and Forestry* chapter of the US EPA (2020) submission to the UNFCCC. New in this report, most of the national scale estimates are also reported by individual U.S. state (Fig. 1) and are available online for the entire 1990-2018 time series (see appendix).

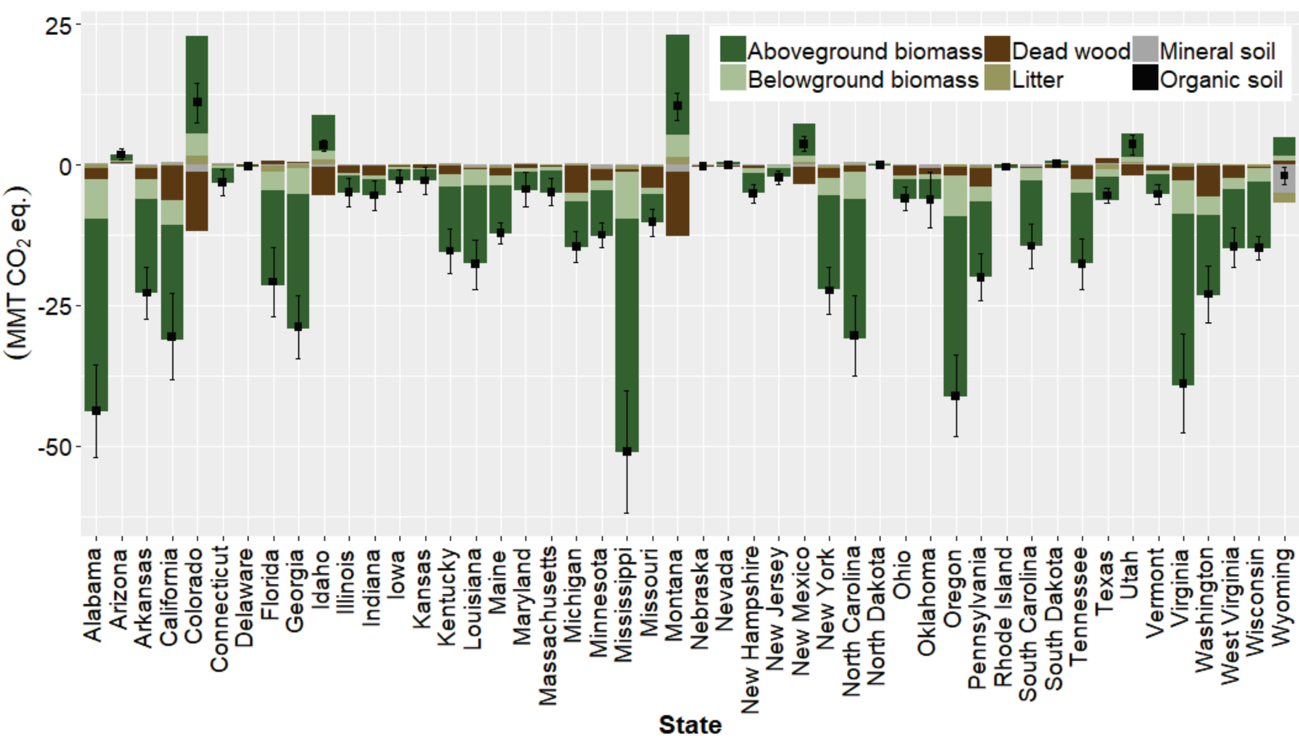

Figure 1.—Estimated annual emissions and removals from forest land remaining forest land by carbon pool for each of the conterminous 48 states in 2018 (MMT CO_2 Eq.). Note that points and confidence intervals (95 percent) reflect net flux for all carbon pools in each state. Negative estimates indicate net C uptake (i.e., a net removal of C from the atmosphere).

Forest Carbon Cycle

Carbon is continuously cycled among ecosystem pools and the atmosphere as a result of biogeochemical processes in forests (e.g., photosynthesis, respiration, decomposition, and disturbances such as fires or pest outbreaks) and anthropogenic activities (e.g., harvesting, thinning, and replanting). As trees photosynthesize and grow, C is removed from the atmosphere and stored in living tree biomass. As trees die and otherwise deposit litter and debris on the forest floor, C is released to the atmosphere and is also transferred to the litter, dead wood, and soil pools by organisms that facilitate decomposition.

The net change in forest C is not equivalent to the net flux between forests and the atmosphere because timber harvests do not result in an immediate flux of all harvested biomass C to the atmosphere. Instead, following harvesting a portion of the C stored in wood is transferred to a "product pool." Once in a product pool, the C is emitted over time as carbon dioxide (CO_2) from decomposition and as CO_2, methane (CH_4), nitrous oxide (N_2O), carbon monoxide (CO), and other nitrogen oxides (NO_x) when the wood product combusts. The rate of emission varies considerably among different product pools.

Carbon Pools

For estimating C stocks or stock change (flux), C in forest ecosystems can be divided into the following five storage pools (IPCC 2006):

- Aboveground biomass—all living biomass above the soil including stem, stump, branches, bark, seeds, and foliage. This pool includes live understory.
- Belowground biomass—all living biomass of coarse living roots greater than 2 millimeters (mm) diameter.
- Dead wood—all nonliving woody biomass either standing, lying on the ground (but not including litter), or in the soil.
- Litter—the litter, fumic, and humic layers, and all nonliving biomass with a diameter less than 7.5 centimeters (cm) at transect intersection, lying on the ground.
- Soil organic C (SOC)—all organic material in soil to a depth of 1 meter but excluding the coarse roots of the belowground pools.

In addition, two harvested wood pools are included when estimating C flux:

- Harvested wood products (HWP) in use.
- HWP in solid waste disposal sites (SWDS).

Total Emissions and Removals

Forest land, HWPs, woodlands, and urban trees in settlements collectively represent a net GHG sink over the UNFCCC reporting period, with interannual variability driven, in large part, by natural and anthropogenic disturbances (e.g., wildfire, harvesting), land conversions, and changes in HWPs in use (Table 1.; US EPA 2020). In 2018, forest land, HWPs, woodlands, and urban trees in settlements collectively represented an estimated net uptake of 752.9 million metric tons of carbon dioxide equivalent (MMT CO_2 Eq.). The category “forest land remaining forest land” was the largest net sink in the land sector, with an estimated uptake of 564.5 MMT CO_2 Eq. Conversions from forest land were the largest source of emissions within the categories included in this report, with estimated emissions of 127.4 MMT CO_2 Eq. (Table 1; US EPA 2020).

Table 1.—Emissions and removals (net flux) from land use, land-use change, and forestry (MMT CO_2 Eq.)

Emissions and Removals Category[a]	1990	1995	2000	2005	2010	2016	2017	2018
Forest land remaining forest land[b]	(610.1)	(598.7)	(572.1)	(572.6)	(556.2)	(565.5)	(552.0)	(564.5)
Non-CO_2 emissions from fire	1.5	0.6	2.9	8.2	4.6	5.6	18.8	18.8
N_2O emissions from forest soils	0.1	0.3	0.5	0.5	0.5	0.5	0.5	0.5
Non-CO_2 emissions from drained organic soils	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Forest land converted to non-forest land[b]	119.1	120.8	122.5	124.4	126.0	127.4	127.4	127.4
Non-forest land converted to forest land[b]	(109.4)	(109.7)	(109.9)	(110.2)	(110.4)	(110.6)	(110.6)	(110.6)
Harvested wood products	(123.8)	(112.2)	(93.4)	(106.0)	(69.1)	(92.4)	(95.7)	(98.8)
Woodlands remaining woodlands[c]	5.0	4.9	4.8	4.6	4.4	4.1	4.0	4.0
Urban trees in settlements[d]	(96.4)	(103.3)	(110.4)	(117.4)	(124.6)	(129.8)	(129.8)	(129.8)
Total Emissions and Removals	**(813.9)**	**(797.2)**	**(755.0)**	**(768.4)**	**(724.7)**	**(760.6)**	**(737.3)**	**(752.9)**

[a] For details on how estimates were compiled see US EPA 2020.
[b] Estimated emissions and removals include the net changes to C stocks stored in all ecosystem pools.
[c] Estimates for woodlands, which are included in the grassland land use category, were compiled using the same methods and models as those in the forest land category.
[d] Estimates of emissions and removals from urban trees in settlements were compiled using percentage tree cover in carbon sequestration density per unit of tree cover.
Notes: Totals may not sum due to independent rounding. Parentheses indicate net C uptake (i.e., a net removal of C from the atmosphere).

Forest Land Remaining Forest Land and Harvested Wood Products

Within the "forest land remaining forest land" category, aboveground live biomass is the largest contributor to the net uptake over the reporting period, followed by belowground live biomass and dead wood (Table 2). Harvested wood products in use and in solid waste disposal sites (SWDS) are also an important contributor to the net sink in the land sector, and 2018 estimates for both pools increased from previous years.

Table 2.—Emissions and removals (net flux) from forest land remaining forest land and harvested wood pools (MMT CO_2 Eq.)

Carbon Pool[a]	1990	1995	2000	2005	2010	2016	2017	2018
Forest ecosystem	**(610.1)**	**(598.7)**	**(572.1)**	**(572.6)**	**(556.2)**	**(565.5)**	**(552.0)**	**(564.5)**
Aboveground biomass	(425.1)	(416.1)	(392.7)	(391.3)	(391.3)	(397.0)	(381.2)	(385.2)
Belowground biomass	(98.6)	(96.6)	(91.5)	(90.8)	(90.3)	(91.1)	(87.6)	(88.6)
Dead wood	(81.9)	(82.8)	(82.7)	(84.1)	(83.4)	(87.6)	(83.1)	(86.4)
Litter	(5.0)	(3.5)	(4.5)	(5.2)	(1.4)	(0.9)	(3.5)	(3.1)
Soil (mineral)	0.3	(0.1)	(1.0)	(1.8)	4.6	8.2	1.4	(3.3)
Soil (organic)	(0.6)	(0.5)	(0.3)	(0.1)	4.9	2.3	1.4	1.4
Drained organic soil	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8
Harvested wood	**(123.8)**	**(112.2)**	**(93.4)**	**(106.0)**	**(69.1)**	**(92.4)**	**(95.7)**	**(98.8)**
Products in use	(54.8)	(51.7)	(31.9)	(42.6)	(7.4)	(27.8)	(30.3)	(31.5)
SWDS	(69.0)	(60.5)	(61.5)	(63.4)	(61.7)	(64.6)	(65.5)	(67.2)
Total Net Flux	**(733.9)**	**(710.9)**	**(665.5)**	**(678.6)**	**(625.3)**	**(657.9)**	**(647.7)**	**(663.2)**

[a] For details on these estimates and how they were compiled see US EPA 2020.
Notes: Totals may not sum due to independent rounding. Parentheses indicate net C uptake (i.e., a net removal of C from the atmosphere).

Carbon stock estimates for forest ecosystem and harvested wood C storage pools are presented in Table 3. Together, the estimated aboveground biomass and soil C pools account for a large proportion of total forest ecosystem C stocks. By maintaining current harvesting practices and regeneration activities on these forested lands, along with continued input of harvested products into the HWP pool, C stocks in forests are likely to continue to increase in the near term, though possibly at a lower rate. Because most of the timber harvested from U.S. forest land is used in wood products and many discarded wood products are disposed of in SWDS rather than by incineration, significant quantities of C in harvested wood are transferred to these long-term storage pools rather than being released rapidly to the atmosphere (Skog 2008).

Table 3.—Carbon stocks in forest land remaining forest land and harvested wood pools (MMT C)

Carbon Pool[a]	1990	1995	2000	2005	2010	2017	2018	2019
Forest	**51,527**	**52,358**	**53,161**	**53,886**	**54,663**	**55,746**	**55,897**	**56,051**
Aboveground biomass	11,833	12,408	12,962	13,484	14,020	14,780	14,884	14,989
Belowground biomass	2,350	2,483	2,612	2,734	2,858	3,033	3,056	3,081
Dead wood	2,120	2,233	2,346	2,454	2,568	2,731	2,753	2,777
Litter	3,662	3,670	3,676	3,647	3,646	3,639	3,640	3,641
Soil (mineral)	25,636	25,636	25,637	25,639	25,641	25,637	25,637	25,638
Soil (organic)	5,927	5,928	5,928	5,929	5,929	5,926	5,926	5,926
Harvested wood	**1,895**	**2,061**	**2,218**	**2,353**	**2,462**	**2,616**	**2,642**	**2,669**
Products in use	1,249	1,326	1,395	1,447	1,471	1,505	1,513	1,521
SWDS	646	735	823	906	991	1,112	1,129	1,148
Total stocks	**53,423**	**54,419**	**55,380**	**56,239**	**57,124**	**58,362**	**58,539**	**58,720**

[a] For details on these estimates and how they were compiled see US EPA 2020.
Notes: Totals may not sum due to independent rounding. Forest C stock estimates include all forest land remaining forest land in the conterminous 48 states and Alaska. Forest ecosystem C stocks do not include U.S. Territories because managed forest land for U.S. Territories is not currently included in Section 6.1 Representation of the U.S. Land Base. Forest ecosystem C stocks also do not include Hawaii because there is not sufficient NFI data to support inclusion at this time. Forest ecosystem C stocks on managed forest land in Alaska were compiled using the gain-loss method as described in Annex 3.13. Harvested wood product stocks include exports, even if the logs are processed in other countries, and excludes imports. Harvested wood estimates are based on results from annual surveys and models. Totals may not sum due to independent rounding. Population estimates compiled using FIA data are assumed to represent stocks as of January 1 of the inventory year. Flux is the net annual change in stock. Thus, flux estimates for 2018 require C stocks for 2018 and 2019.

Forest Land Conversions

Land use conversions to and from forest land result in substantial emissions and removals each year. In this section all emissions and removals included for land conversions to and from forest land, as reported in US EPA (2020), are included in Table 4. Forest land conversion to settlements was the largest source of emissions in the conversion categories while cropland conversion to forest land resulted in the largest annual uptake. Considering all forest land conversions included in the US EPA (2020) report, over the reporting period there have been net emissions each year, with estimated net emissions of 16.7 MMT CO_2 Eq. for the most recent year.

Table 4.—Emissions and removals (net flux) from conversions to and from forest land (MMT CO_2 Eq.)

Land Conversions[a]	1990	1995	2000	2005	2010	2016	2017	2018
Forest land converted to cropland	48.6	48.7	48.5	48.4	48.3	48.7	48.7	48.7
Forest land converted to grassland	15.9	15.8	16.0	16.0	16.0	15.9	15.9	15.9
Forest land converted to settlements	54.6	56.3	58.0	59.9	61.6	62.9	62.9	62.9
Cropland converted to forest land	(45.9)	(45.9)	(46.0)	(46.1)	(46.2)	(46.3)	(46.3)	(46.3)
Grassland converted to forest land	(9.8)	(9.7)	(9.7)	(9.6)	(9.6)	(9.7)	(9.7)	(9.7)
Other land converted to forest land	(14.3)	(14.5)	(14.6)	(14.8)	(14.9)	(14.9)	(14.9)	(14.9)
Settlements converted to forest land	(38.6)	(38.6)	(38.7)	(38.7)	(38.8)	(38.9)	(38.9)	(38.9)
Wetlands converted to forest land	(0.9)	(0.9)	(0.9)	(0.9)	(0.9)	(0.9)	(0.9)	(0.9)
Net Emissions and Removals	**9.6**	**11.2**	**12.6**	**14.3**	**15.6**	**16.7**	**16.7**	**16.7**

[a] For details on these estimates and how they were compiled see US EPA 2020.

Notes: Totals may not sum due to independent rounding. Parentheses indicate net C uptake (i.e., a net removal of C from the atmosphere). Emissions and removals from forest land converted to other lands are currently not included in US EPA (2020). Forest land converted to wetlands estimates were not compiled by the Forest Service.

Land Area

The land area included in the US EPA (2020) report includes lands directly influenced by human intervention. Direct intervention occurs mostly in areas accessible to human activity and includes altering or maintaining the condition of the land to produce commercial or noncommercial products or services; to serve as transportation corridors or locations for buildings, landfills, or other developed areas for commercial or noncommercial purposes; to extract resources or facilitate acquisition of resources; or to provide social functions for personal, community, or societal objectives where these areas are readily accessible to society. Forest Inventory and Analysis data from each of the conterminous 48 states and Alaska comprise an estimated 280 million hectares (ha) of forest land that are considered managed and are included in this report along with an additional 10 million ha of non-forest land converted to forest land. Some differences exist in forest land area estimates in the latest update to the Resources Planning Act Assessment (Oswalt et al. 2019) and the forest land area estimates included in the US EPA (2020) report, which are based on annual FIA data through 2018 for all states (USDA Forest Service 2019). These differences are due, in large part, to the separation of land categories and the managed land definition used in the US EPA (2020) report (Nelson et al. 2020). Sufficient annual inventory data are not yet available for Hawaii, but estimates of these areas are included in Oswalt et al. (2019). Even though Hawaii and U.S. Territories have relatively small areas of forest land that may not substantially influence the overall C budget for forest land, these regions will be added to the forest C estimates as sufficient data become available. Agroforestry systems that meet the definition of forest land are also not currently included in the US EPA (2020) report since they are not explicitly inventoried (i.e., they are classified as agroforestry system) by either the FIA program or the Natural Resources Inventory of the USDA Natural Resources Conservation Service. Woodland area is included in the "grassland remaining grassland" and "land converted to grassland" categories and is not explicitly separated in the US EPA (2020) report as a subcategory of grasslands. Combined, forest land and woodland area accounts for more than 311 million ha (Table 5).

Table 5.—Annual estimates of forest land and woodland area (1000 ha)

Land Area Category[a]	1990	1995	2000	2005	2010	2018	2019
Forest land remaining forest land	279,748	279,840	280,025	279,749	279,918	279,787	279,682
Non-forest land converted to forest land	9,622	9,654	9,689	9,725	9,761	9,796	9,796
Woodland remaining woodland[b]	19,891	19,669	19,255	18,630	17,733	16,000	15,776
Non-woodland converted to woodland[b]	5,782	5,702	5,552	5,322	4,994	4,607	4,607
Total Area	**315,043**	**314,865**	**314,521**	**313,426**	**312,405**	**312,209**	**311,880**

[a] For details on these estimates and how they were compiled see US EPA 2020.

[b]Woodland area is included in the "remaining grassland" and "land converted to grassland" categories and is not explicitly separated in the US EPA (2020) report.

Notes: Totals may not sum due to independent rounding. The estimates reported here may differ from the Land Representation section of US EPA (2020) but are consistent with estimates used to compile emissions and removals in these categories. See Annex 3.13 in US EPA (2020) for more details.

Planned Improvements

Planned improvements to estimation and reporting include the following general topics: development of a more robust estimation and reporting system, individual C pool estimation, coordination with other land-use categories, and annual inventory data incorporation. Research is underway to leverage auxiliary information (i.e., remotely sensed information) to operate at finer spatial and temporal scales. As in past submissions, emissions and removals associated with natural (e.g., wildfire, insects, and disease) and human (e.g., harvesting) disturbances are implicitly included in the report given the design of the annual NFI, but are not explicitly estimated. In addition to integrating auxiliary information into the estimation framework, alternative estimators are also being evaluated that will eliminate latency in population estimates from the NFI, improve annual estimation and characterization of interannual variability, facilitate attribution of fluxes to particular activities, and allow for easier harmonization of NFI data with auxiliary data products. There are also investments being made to leverage state-level wood products and harvest information to allow for the disaggregation of HWPs estimates at the state level. Collectively these improvements are expected to reduce uncertainties in the estimates at the national and state scales and facilitate entity-level estimation and reporting.

2020 Estimates at a Glance

Below are summary statistics from the compilation of the forest land, woodlands, HWPs, and urban trees in settlements in the US EPA (2020) report.

- Forest land, HWPs, and urban trees in settlements collectively offset more than 11 percent (752.9 MMT CO_2 Eq.) of total GHG emissions annually, or 14 percent of CO_2 emissions.
- Forest land accounts for more than 95 percent of the net C sink within the land sector.
- Live vegetation in forests and urban trees account for nearly 80 percent of the C sink strength.
- Land conversions to and from forest land continue to result in net emissions (16.7 MMT CO_2 Eq.).
- More than 56 percent of all carbon in forest ecosystems is stored in the soil with small stock changes annually.
- Carbon storage in HWPs continues to increase annually since the Great Recession.
- Forests uptake averages 0.6 metric tons of C per hectare per year (MT C ha^{-1} yr^{-1}) with live vegetation accounting for more than 85 percent (0.5 MT C ha^{-1} yr^{-1}) of the uptake.

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The Environmental Consequences of Forest Roads and Achieving a Sustainable Road System

March 2020

Table of Contents

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Cover photos credits: Mt. Baker-Snoqualmie National Forest, 2016. WildEarth Guardians.

Introduction

The Forest Service faces many challenges with its vastly oversized, under-maintained, and unaffordable transportation system. With 370,643 miles of system roads and 137,409 miles of system trails (USDA Forest Service 2019), the network extends broadly across every national forest and grassland and through a variety of habitats, ecosystems and terrains. An impressive body of scientific literature addresses the various effects of roads on the physical, biological and cultural environment. Numerous studies demonstrate the harmful environmental consequences to water, fish, wildlife, and ecosystems.

In recent years, the scientific literature has expanded to address the effects of roads on climate change adaptation and conversely the effects of climate change on roads, as well as the multiple benefits of road removal on the physical, biological and cultural environments.

The first section of this paper provides a literature review summarizing the most recent science related to the environmental impacts of forest roads and motorized trails. The second section focuses on climate change effects and strategies to address the growing ecological consequences to forest resources. The third section provides background and specific direction for the Forest Service to provide for an ecologically and economically sustainable road system, including recommendations for future action.

I. Impacts of Transportation Infrastructure and Access to the Ecological Integrity of Terrestrial and Aquatic Ecosystems and Watersheds

It is well understood that transportation infrastructure provides access to national forests and grasslands and also harms aquatic and terrestrial environments at multiple scales. In general, the more roads and motorized trails the greater the impacts. Since its emergence, the field of road ecology and the resulting research has proven the magnitude and breadth of ecological issues related to roads; entire books have been written on the topic (e.g., Forman et al. 2003, van der Ree et al. 2015), and research centers continue to expand their case studies, including the Western Transportation Institute at Montana State University and the Road Ecology Center at the University of California - Davis.[1]

Below, we provide a summary of the current understanding of the impacts of roads and motorized access on terrestrial and aquatic ecosystems, supplementing long-established, peer-reviewed literature reviews on the topic, including Gucinski et al. (2000), Trombulak and Frissell (2000), Coffin (2007), and Robinson et al. (2010). More targeted reviews have been published on the effects of roads on insects (Munoz et al. 2015), vertebrates (da Rosa 2013), and animal abundance (Fahrig and Rytwinski 2009, Benítez-López et al. 2010). Literature reviews on the ecological and social impacts of motorized recreation include Gaines et al. (2003), Davenport and Switalski (2006), Ouren

[1] See http://www.westerntransportationinstitute.org/programs/road-ecology and http://roadecology.ucdavis.edu/

et al. (2007), Switalski and Jones (2012), and, more recently, Switalski (2017). In addition to the physical and environmental impacts of roads, increased visitation has resulted in intentional and unintentional damage to many cultural and historic sites (Spangler and Yentsch 2008, Sampson 2009, Hedquist et al. 2014).

A. Impacts on geomorphology and hydrology

The construction and presence of forest roads can dramatically change the hydrology and geomorphology of a forest system leading to reductions in the quantity and quality of aquatic habitat (Al-Chokhachy et al. 2016). While there are several mechanisms that cause these impacts (Wemple et al. 2001, Figure 1), most fundamentally, compacted roadbeds reduce rainfall infiltration, intercepting and concentrating water, and providing a ready source of sediment for transport (Wemple et al. 2001). In fact, roads contribute more sediment to streams than any other land management activities on Forest Service lands (Gucinski et al. 2000). Surface erosion rates from roads can be up to three orders of magnitude greater than erosion rates from undisturbed forest soils (Endicott 2008).

Erosion and sediment produced from roads occur both chronically and catastrophically. Every time it rains, sediment from the road surface and from cut-and fill-slopes is picked up by rainwater that flows into and on roads (fluvial erosion). The sediment that is entrained in surface flows are often concentrated into road ditches and culverts and directed into streams. The degree of fluvial erosion varies by geology and geography, and increases with increased motorized use (Robichaud et al. 2010). Closed roads produce significantly less sediment than open drivable roads (Sosa Pérez and Macdonald 2017, Foltz et al. 2009).

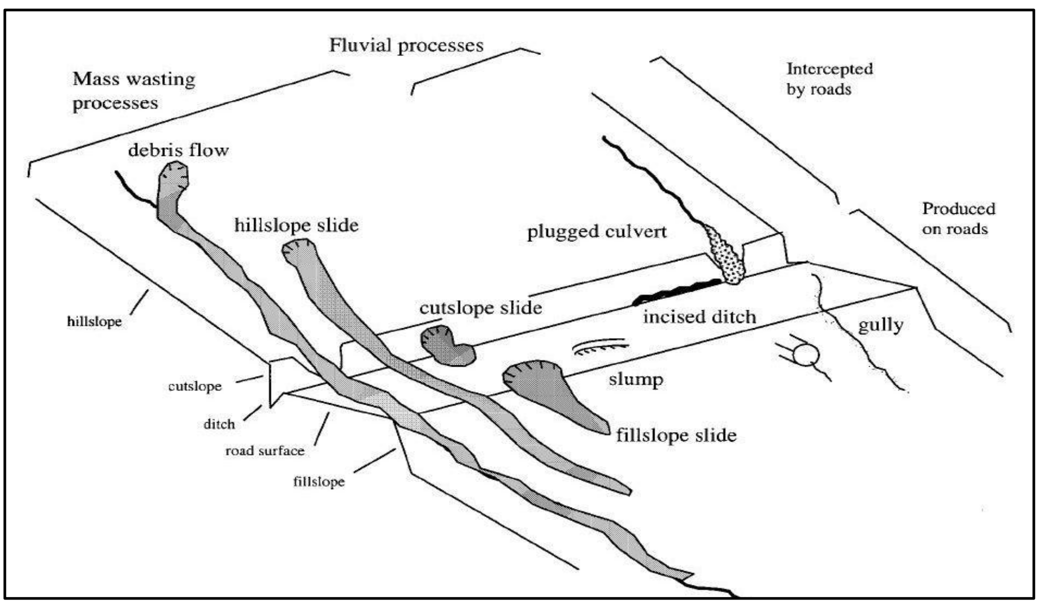

Figure 1: Typology of erosional and depositional features produced by mass-wasting and fluvial processes associated with forest roads (reprinted from Wemple et al. 2001).

Roads also precipitate catastrophic failures of road beds and fills (mass wasting) during large storm events leading to massive slugs of sediment moving into waterways (Gucinski et al. 2000, Endicott 2008). This typically occurs when culverts are undersized and cannot handle the volume of water funneled through them, or they simply become plugged with debris and sediment. The saturated roadbed can fail entirely and result in a landslide, or the blocked stream crossing can erode the entire fill down to the original stream channel.

The erosion of road- and trail-related sediment and its subsequent movement into stream systems affects the geomorphology of the drainage system in a number of ways. It directly alters channel morphology by embedding larger gravels as well as filling pools. It can also have the opposite effect of increasing peak discharges and scouring channels, which can lead to disconnection of the channel and floodplain, and lowered base flows (Gucinski et al. 2000). The width/depth ratio of the stream changes can trigger changes in water temperature, sinuosity and other geomorphic factors important for aquatic species survival (Trombulak and Frissell 2000).

B. Impacts on aquatic habitat and fish

Roads can have dramatic and lasting impacts on fish and aquatic habitat. Increased sedimentation in stream beds has been linked to decreased fry emergence, decreased juvenile densities, loss of winter carrying capacity, increased predation of fish, and reductions in macro-invertebrate populations that are a food source to many fish species (Gucinski et al. 2000, Endicott 2008). Roads close to streams reduce the number of trees available for large wood recruitment, and reduce stream-side shade (Meredith et al. 2014.) On a landscape scale, these effects add up to: changes in the frequency, timing and magnitude of disturbance to aquatic habitat and changes to aquatic habitat structures (e.g., pools, riffles, spawning gravels and in-channel debris), and conditions (food sources, refugia, and water temperature; Gucinski et al. 2000).

River fragmentation

Roads also act as barriers to migration and fragment habitat of aquatic species (Gucinski et al. 2000). Where roads cross streams, road engineers usually place culverts or bridges. Undersized culverts interfere with sediment transport and channel processes such that the road/stream crossing becomes a barrier for fish and aquatic species movement up and down stream (Erikinaro et al. 2017). For instance, a culvert may scour on the downstream side of the crossing, actually forming a waterfall up which fish cannot move. Undersized culverts can infringe upon the channel or floodplain and trap sediment causing the stream to become too shallow and/or warm such that fish will not migrate past the structure. Or, the water can move through the culvert at too high a gradient or velocity to allow fish passage (Endicott 2008).

River fragmentation is problematic for many aquatic species but especially for anadromous species that must migrate upstream to spawn. Well-known native aquatic species affected by roads include salmon such as coho (*Oncorhynchus kisutch*), Chinook (*O. tshawytscha*), and chum (*O. keta*); steelhead

(*O. mykiss*), a variety of trout species including bull trout (*Salvelinus confluentus*) and cutthroat trout (*O. clarki*), as well as other native fish and amphibians (Endicott 2008). The restoration and mitigation of impassable road culverts has been found to restore connectivity and increase available aquatic habitat (Erikinaro et al. 2017), and the quality of aquatic habitat (McCaffery et al. 2007).

C. Impacts on terrestrial habitat and wildlife

Roads and trails impact wildlife through a number of mechanisms including: direct mortality (poaching, hunting/trapping), changes in movement and habitat-use patterns (disturbance/avoidance), as well as indirect impacts including altering adjacent habitat and interference with predator/prey relationships (Coffin 2007, Fahrig and Rytwinski 2009, Robinson et al. 2010, da Rosa and Bager 2013). Some of these impacts result from the road itself, and some result from the uses on and around the roads (access). Ultimately, numerous studies show that roads reduce the abundance, diversity, and distribution of several forest species (Fayrig and Ritwinski 2009, Benítez-López et al. 2010, Munoz et al. 2015).

Abundance and distribution

The extensive research on roads and wildlife establish clear trends of wildlife population declines. Fahrig and Rytwinski (2009) reviewed the empirical literature on the effects of roads and traffic on animal abundance and distribution looking at 79 studies that addressed 131 species. They found that the number of documented negative effects of roads on animal abundance outnumbered the number of positive effects by a factor of 5. Amphibians, reptiles, and most birds tended to show negative effects. Small mammals generally showed either positive effects or no effect, mid-sized mammals showed either negative effects or no effect, and large mammals showed predominantly negative effects. Benítez-López et al. (2010) conducted a meta-analysis on the effects of roads and infrastructure proximity on mammal and bird populations. They found a significant pattern of avoidance and a reduction in bird and mammal populations in the vicinity of infrastructure. Muñoz et al. (2015) found that many insect populations have declined as well.

Direct mortality, disturbance, and habitat modification

Road and motorized trail use affect many different types of species. For example, trapping, poaching, collisions, negative human interactions, disturbance and displacement significantly impact wide ranging carnivores (Gaines et al. 2003, Table 1). Hunted game species such as elk (Cervus canadensis), become more vulnerable from access allowed by roads and motorized trails resulting in a reduction in effective habitat among other impacts (Rowland et al. 2005). Slow-moving migratory animals such as amphibians, and reptiles who use roads to regulate temperature, are also vulnerable (Gucinski et al. 2000, Brehme et al. 2013). Roads and motorized trails also affect ecosystems and habitats because they are major vectors of non-native plant and animal species (Gelbard and Harrison 2003). This can have significant ecological and economic impacts when aggressive invading species overwhelm or significantly alter native species and systems.

Table 1: Road- and recreation trail-associated factors for wide-ranging carnivores (Reprinted from Gaines et al. (2003)[2]

Focal species	Road-associated factors	Motorized trail-associated factors	Nonmotorized trail-associated factors
Grizzly bear	Poaching Collisions Negative human interactions Displacement or avoidance	Poaching Negative human interactions Displacement or avoidance	Poaching Negative human interactions Displacement or avoidance
Lynx	Down log reduction Trapping Collisions Disturbance at a specific site	Disturbance at a specific site Trapping	Disturbance at a specific site
Gray wolf	Trapping Poaching Collisions Negative human interactions Disturbance at a specific site Displacement or avoidance	Trapping Disturbance at a specific site	Trapping Disturbance at a specific site
Wolverine	Down log reduction Trapping Disturbance at a specific site Collisions	Trapping Disturbance at a specific site	Trapping Disturbance at a specific site

Habitat fragmentation

At the landscape scale, roads fragment habitat blocks into smaller patches that may not be able to support interior forest species. Smaller habitat patches result in diminished genetic variability, increased inbreeding, and at times local extinctions (Gucinski et al. 2000; Trombulak and Frissell 2000). For example, a narrow forest road with little traffic was a barrier in Arizona to the Mt. Graham red squirrel (*Tamiasciurus hudsonicus grahamensis*; Chen and Koprowski 2013). Fragmentation intensifies concerns about grizzly bear population viability, especially since roads increase human/bear interactions exacerbating the problem of excessive mortality (Proctor et al, 2012)

Roads also change the composition and structure of ecosystems along buffer zones, called edge-affected zones. The width of edge-affected zones varies by what metric is being discussed; however, researchers have documented road-avoidance zones a kilometer or more away from a road (Robinson et al.2010; Table 2). In heavily roaded landscapes, edge-affected acres can be a significant percentage of total acres. For example, in a landscape where the road density is 3 mi/mi^2 and where the edge-affected zone is estimated to be 500 ft from the center of the road to each side, the edge-affected zone is 56% of the total acreage.

[2] For a list of citations see Gaines et al. (2003).

Table 2: A summary of some documented road-avoidance zones for various species (adapted from Robinson et al. 2010).

Species	Avoidance zone m (ft)	Type of disturbance	Reference
Snakes	650 (2133)	Forestry roads	Bowles (1997)
Salamander	35 (115)	Narrow forestry road, light traffic	Semlitsch (2003)
Woodland birds	150 (492)	Unpaved roads	Ortega and Capen (2002)
Spotted owl	400 (1312)	Forestry roads, light traffic	Wasser et al. (1997)
Marten	<100 (<328)	Any forest opening	Hargis et al. (1999)
Elk	500–1000 (1640-3281)	Logging roads, light traffic	Edge and Marcum (1985)
Grizzly bear	3000 (9840)	Fall	Mattson et al. (1996)
	500 (1640)	Spring and summer	
	1122 (3681)	Open road	Kasworm and Manley (1990)
	665 (2182)	Closed road	
Black bear	274 (899)	Spring, unpaved roads	Kasworm and Manley (1990)
	914 (2999)	Fall, unpaved roads	

Migration disruption

Roads disrupt migration of large ungulates, such as elk, impeding travel at multiple scales, including seasonal home range use and migration to winter range (Buchanan et al. 2014, Prokopenko et al. 2017). For example, a recent study found migrating elk changed their behavior and stopover use on migration routes that were roaded (Paton et al. 2017). The authors suggest this disturbance may lead to decreased foraging, displacement of high-quality habitat, and affect the permeability of the migration route. In addition, roads disrupt grizzly bear movements influencing dispersal away from the maternal home range and ultimately influencing population-level fragmentation." (Proctor et al. 2018).

Oil and gas development (and associated roads) reduced the effectiveness of both mule deer and pronghorn migration corridors in western Wyoming. (Sawyer et al. 2005). Multiple studies found that mule deer increased their rate of travel during migrations, reducing stop over time and their use of important foraging habitats (Sawyer et al. 2012, Lendrum et al. 2012; Ledrum et al. 2013;). A study in Colorado found that female mule deer changed their migration timing which may change alignment with vegetative phenology and potentially result in energetic and demographic costs (Lendrum et al. 2013).

D. Road density thresholds for fish and wildlife[3]

It is well documented that, beyond specific road density thresholds, certain species will be negatively affected, and some risk being extirpated (Robinson et al. 2000, Table 3). Most studies that look into the relationship between road density and wildlife focus on the impacts to large endangered carnivores or hunted game species, although high road densities certainly affect other species. Grizzly bears have been found to have a higher mortality risk as road density increases (Boulanger and Stenhouse 2014). Gray wolves (*Canis lupus*) in the Great Lakes region and elk in Montana and Idaho also face increased mortality risk, and have undergone the most long-term and in-depth analysis. Forman and Hersperger (1996) found that in order to maintain a naturally functioning landscape with sustained populations of large mammals, road density must be below 0.6 km/km² (1.0 mi/mi²).

A number of studies show that higher road densities also impact aquatic habitats and fish (Table 3). Carnefix and Frissell (2009) provide a concise review of studies that correlate cold water fish abundance and road density, and from the cited evidence concluded that:

> 1) no truly "safe" threshold road density exists, but rather negative impacts begin to accrue and be expressed with incursion of the very first road segment; and 2) highly significant impacts (e.g., threat of extirpation of sensitive species) are already apparent at road densities on the order of 0.6 km/km² (1.0 mi/mi²) or less, (Carnefix and Frissell (2009), p. 1).

Cold water salmonids such as threatened bull trout, are particularly sensitive to the impacts of forest roads. The U.S. Fish and Wildlife Service's Final Rule listing bull trout as threatened (USDI Fish and Wildlife Service 1999) addressed road density stating:

> … assessment of the interior Columbia Basin ecosystem revealed that increasing road densities were associated with declines in four non-anadromous salmonid species (bull trout, Yellowstone cutthroat trout, westslope cutthroat trout, and redband trout) within the Columbia River Basin, likely through a variety of factors associated with roads (Quigley & Arbelbide 1997). Bull trout were less likely to use highly roaded basins for spawning and rearing, and if present, were likely to be at lower population levels (Quigley and Arbelbide 1997). Quigley et al. (1996) demonstrated that when average road densities were between 0.4 to 1.1 km/km² (0.7 and 1.7 mi/mi²) on USFS lands, the proportion of subwatersheds supporting "strong" populations of key salmonids dropped substantially. Higher road densities were associated with further declines (USDI Fish and Wildlife Service (1999), p. 58922).

Anderson et al. (2012) showed that watershed conditions tend to be best in areas protected from road construction and development. Using the U.S. Forest Service's Watershed Condition Framework assessment data, they showed that National Forest lands protected under the Wilderness Act tend to have

[3] We intend for the term "road density" to refer to the density of all roads within national forests, including system roads, closed roads, non-system roads, temporary roads and motorized trails, and roads administered by other jurisdictions (private, county, state).

the healthiest watersheds. In support of this conclusion, McCaffery et al. (2005) found that streams in roadless watersheds had less fine sediment and higher quality habitat than roaded watersheds. Miller et al. (2017) showed that in 20 years of monitoring forests managed by the Northwest Forest Plan there were measurable improvements in watershed conditions as a result of road decommissioning, finding "...the decommissioning of roads in riparian areas has multiple benefits, including improving the riparian scores directly and typically the sedimentation scores."

Table 3: A summary of some road-density thresholds and correlations for terrestrial and aquatic species and ecosystems (reprinted from Robinson et al. 2010).

Species (Location)	**Road density (mean, guideline, threshold, correlation)**	**Reference**
Wolf (Minnesota)	0.36 km/km2 (mean road density in primary range); 0.54 km/km² (mean road density in peripheral range)	Mech et al. (1988)
Wolf	>0.6 km/km² (absent at this density)	Jalkotzy et al. (1997)
Wolf (Northern Great Lakes re-gion)	>0.45 km/km² (few packs exist above this threshold); >1.0 km/km² (no pack exist above this threshold)	Mladenoff et al. (1995)
Wolf (Wisconsin)	0.63 km/km² (increasing due to greater human tolerance	Wydeven et al. (2001)
Wolf, mountain lion (Minne-sota, Wisconsin, Michigan)	0.6 km/km² (apparent threshold value for a naturally functioning landscape containing sustained popula-tions)	Thiel (1985); van Dyke et al. (1986); Jensen et al. (1986); Mech et al. (1988); Mech (1989)
Elk (Idaho)	1.9 km/km² (density standard for habitat effectiveness)	Woodley 2000 cited in Beazley et al. 2004
Elk (Northern US)	1.24 km/km² (habitat effectiveness decline by at least 50%)	Lyon (1983)
Elk, bear, wolverine, lynx, and others	0.63 km/km² (reduced habitat security and increased mortality)	Wisdom et al. (2000)
Moose (Ontario)	0.2-0.4 km/km2 (threshold for pronounced response)	Beyer et al. (2013)
Grizzly bear (Montana)	>0.6 km/km²	Mace et al. (1996); Matt-son et al. (1996)
Black bear (North Carolina)	>1.25 km/km² (open roads); >0.5 km/km2 (logging roads); (interference with use of habitat)	Brody and Pelton (1989)
Black bear	0.25 km/km² (road density should not exceed)	Jalkotzy et al. (1997)
Bobcat (Wisconsin)	1.5 km/km² (density of all road types in home range)	Jalkotzy et al. (1997)
Large mammals	>0.6 km/km² (apparent threshold value for a naturally functioning landscape containing sustained popula-tions)	Forman and Hersperger (1996)
Bull trout (Montana)	Inverse relationship of population and road density	Rieman et al. (1997); Baxter et al. (1999)

Fish populations (Medicine Bow National Forest)	(1) Positive correlation of numbers of culverts and stream crossings and amount of fine sediment in stream channels (2) Negative correlation of fish density and numbers of culverts	Eaglin and Hubert (1993) cited in Gucinski et al. (2001)
Macroinvertebrates	Species richness negatively correlated with an index of road density	McGurk and Fong (1995)
Non-anadromous salmonids (Upper Columbia River basin)	(1) Negative correlation likelihood of spawning and rearing and road density (2) Negative correlation of fish density and road density	Lee et al. (1997)

E. Roads and Fires

Wildland forest fire plays an essential role in many forest ecosystems, and with climate change, fire will increasingly shape National Forest lands. Humans have made fire more common on the landscape, and studies have found that forest roads can affect fire regimes and localized fuel regimes. Changes in the timing and location of fire can alter the natural fire regime and has negative, cascading effects in ecological communities. For example, a change in timing and frequency of fire can result in habitat loss and fragmentation, shift forest composition, and affect predator-prey interactions (DellaSalla et al. 2004). Following a fire, exposed bare ground on roads can result in chronic erosion, catastrophic culvert failures, and noxious weed invasion.

Forest roads can increase the occurrence of human-caused fires, whether by accident or arson, and road access has been correlated with the number of fire ignitions (Syphard et al. 2007, Yang et al., 2007, Narayanaraj and Wimberly 2012, Nagy et al. 2018). A recent study found that humans ignited four times as many fires as lightning. This represented 92% of the fires in the eastern United States and 65% of the fire ignitions in the western U.S. (Nagy et al. 2018). Another study that reviewed 1.5 million fire records over 20 years found human-caused fires were responsible for 84% of wildfires and 44% of the total area burned (Balch et al. 2017).

In addition to changes in frequency, human-caused fires change the timing of fire occurring when fuel moisture is significantly higher than lightning-started fires (Nagy et al. 2018.). Forest roads may also limit fire growth acting as a fire break and providing access for suppression (Narayanaraj and Wimberly 2011, Robbinne et al. 2016). The result is a spatial and temporal distribution of fire that differs from historical fire regimes.

Roaded areas create a distinct fire fuels profile which may influence ignition risk and burn severity (Narayanaraj and Wimberly 2013). Forest roads create linear gaps with reduced canopy cover, and increased solar radiation, temperature, and wind speed. Invasive weeds and grasses common along roadsides also create fine fuels that are highly combustible. These edge effects can change microclimates far into the forest (Narayanaraj and Wimberly 2012, Ricotta et al. 2018). While there is little definitive research on roads and burn severity, an increase in the prevalence of lightning-caused fires in roaded areas may be due to roadside edge effects (Arienti et al 2009, Narayanaraj and Wimberly 2012). Furthermore, watersheds that have been heavily roaded have typically received intensive management in the past leaving forests in a condition of high fire vulnerability (Hessburg and Agee 2003).

Roadless areas are remote and secure from many human impacts such as unintentional fire starts or arson. A forest fire is almost twice as likely to occur in a roaded area than a roadless area (USDA Forest Service 2000). In fact, human-ignited wildfire is almost five times more likely to occur in a roaded area than in a roadless area. (USDA Forest Service 2000). Higher road density correlates with an increased probability of human-caused ignitions. (Syphard et al. 2007).

After a forest fire, roads that were previously well vegetated often burn or are bladed for fire suppression access or firebreaks leaving them highly susceptible to erosion and weed invasion. Roads are a source of chronic erosion following a fire, and pulses of hillslope sediment and large woody debris can result in culvert failures (Bisson et al. 2003). Fine sediment is frequently delivered to streams and reduces the quality of aquatic habitat. Noxious weeds are established on many forest roads, and post-fire weed invasion can be facilitated by creating a disturbance, reducing competition, and increasing resource availability (Birdsaw et al. 2012).

II. Climate Change and Transportation Infrastructure

Before the Trump administration took office, the Forest Service recognized the importance of considering and adapting to changing climate conditions. The USDA Strategic Plan for Fiscal Years 2014-2018 set a goal to: "Ensure our national forests and private working lands are conserved, restored, and made more resilient to climate change, while enhancing our water resources." (USDA 2014, p 3). As climate change impacts grow more profound, forest managers must consider the impacts *on* the transportation system as well as *from* the transportation system. In terms of the former, changes in precipitation and hydrologic patterns will strain infrastructure, resulting in damage to streams, fish habitat, and water quality as well as threats to public safety and loss of access. As to the latter, the fragmenting effect of roads on habitat will impede the movement of species which is a fundamental element of adaptation. Through planning, forest managers can proactively address threats to infrastructure, and can actually enhance forest resilience by removing unneeded roads to create larger patches of connected habitat.

A. Climate change, forest roads, and fragmented habitat

It is expected that climate change will be responsible for more extreme weather events, leading to increasing flood severity, more frequent landslides, changing hydrographs, and changes in erosion and sedimentation rates and delivery processes (Schwartz et al. 2014, USDA FS 2018). The Forest

Service Office of Sustainability and Climate has compiled climate change vulnerability assessments for several regions of the Forest Service discussing near-term consequences for managers to consider. (Halofsky et al. 2017, 2018a, 2018b, 2019, with additional vulnerabilities displayed below in Table 4).

> Warmer locations will experience more runoff in winter months and early spring, whereas colder locations will experience more runoff in late spring and early summer. In both cases, future peakflows will be higher and more frequent, (Halofsky et al. 2018b at ii).
>
> The frequency and extent of midwinter flooding are expected to increase. Flood magnitudes are also expected to increase because rain-on-snow-driven peak flows will become more common," (*Id.* at 83).
>
> Roads and other infrastructure that are near or beyond their design life are at considerable risk to damage from flooding and geomorphic disturbance (e.g., debris slides). If road damage increases as expected, it will have a profound impact on access to Federal lands and on repair costs, (*Id.* at viii).

Magnifying these consequences is the fact that roads, culverts and trails in national forests were designed for storms and water flows typical of past decades, and may not be designed for the storms in future decades. Hence, climate driven changes may cause transportation infrastructure to malfunction or fail (USDA Forest Service 2010, ASHTO 2012). The likelihood is higher for facilities in high-risk settings—such as rain-on-snow zones, coastal areas, and landscapes with unstable geology. The following consequences may occur (USDA Forest Service 2010):

- access to national forests will be interrupted temporarily or permanently as roads wash-out due to landslides or blown-out culverts during events of heavier precipitation or flooding;
- public safety will be compromised as roads, trails and bridges become unstable due to landslides, undercut slopes, or erosion of water-logged slopes due to heavy rainfall; and
- infrastructure may be compromised or abandoned along coastal areas or low-lying estuaries when inundated during high tides and coastal storms as sea-levels rise.

Forests fragmented by roads will likely demonstrate less resistance and resilience to stressors, like those associated with climate change (Noss 2001, see also Table 4. below). First, the more a forest is fragmented (and therefore the higher the edge/interior ratio), the more the forest loses its inertia characteristic, and becomes less resilient and resistant to climate change. Second, the more a forest is fragmented, characterized by isolated patches, the more likely the fragmentation will interfere with the ability of species to track shifting climatic conditions over time and space.

> Hence, roads may impede the movement of many species in response to climate change. Closing unnecessary roads and providing wildlife crossings on roads with heavy traffic might mitigate some of these effects (Noss 1993; Clevenger & Waltho 2000), (Noss (2001) p. 584).

Watershed types within national forests may change which will impact hydrology and when high streamflows occur (Halofsky et. al. 2011). A study in Washington's Mt. Baker-Snoqualmie National

Forest (MBSNF) shows that currently 27% of the roads are in watersheds classified as rain-dominated but that will increase to 75% by 2080 - increasing risk of damage to infrastructure (Strauch 2014). By 2040, 300 miles of forest roads in this forest will be located in watersheds that are projected to see a 50% increase in 100-year floods. Landslide risk will be higher during the winter and spring and decline during summer and autumn. These changes reinforce the importance of transportation analysis that incorporates the impacts of climate change.

Earlier snowmelt may open previously snow-closed roaded areas for a greater portion of the year. While this may appear to benefit visitors that wish to access trails and camps early in the spring, this may also put them in harm's way with melting snow-bridges, avalanche chutes and flooding events (Strauch 2015). Wildlife historically protected by snow-closed roads would be more vulnerable.

B. Modifying infrastructure to increase resilience

To prevent or reduce road-triggered landslides and culvert failures, and other associated hazards, forest managers will need to take a series of actions. In December 2012, the USDA Forest Service published a report entitled, *Assessing the Vulnerability of Watersheds to Climate Change* (USDA FS 2012) which reinforces that forest managers need to be proactive in reducing erosion potential from roads:

> Road improvements were identified as a key action to improve condition and resilience of watersheds on all the pilot forests. In addition to treatments that reduce erosion, road improvements can reduce the delivery of runoff from road segments to channels, prevent diversion of flow during large events, and restore aquatic habitat connectivity by providing for passage of aquatic organisms. As stated previously, watershed sensitivity is determined by both inherent and management-related factors. Managers have no control over the inherent factors, so to improve resilience, efforts must be directed at anthropogenic influences such as instream flows, roads, rangeland, and vegetation management…. [Watershed Vulnerability Analysis (WVA)] results can also help guide implementation of travel management planning by informing priority setting for decommissioning roads and road reconstruction/maintenance. As with the Ouachita NF example, disconnecting roads from the stream network is a key objective of such work. Similarly, WVA analysis could also help prioritize aquatic organism passage projects at road-stream crossings to allow migration by aquatic residents to suitable habitat as streamflow and temperatures change, (USDA Forest Service 2012a, p. 22-23).

Other Forest Service reports support road-related actions to increase climate resilience including replacing undersized culverts with larger ones, prioritizing maintenance and upgrades, and restoring roads to a natural state when they are no longer needed and pose erosion hazards (USDA Forest Service 2010, USDA Forest Service 2011a USDA Forest Service 2012a, USDA FS 2018, Halofsky et al. 2018a).

The Forest Service has developed several resources to identify and mitigate climate change impacts on forests and infrastructure. The aforementioned climate change vulnerability assessments for each region focus on causes, consequences, and options to address them. For example, Halofsky et al.

(2018a) reviews the effects and adaptation options for Region 1 (Northern Region) of the Forest Service, and identifies the increased magnitude of peak streamflows as a primary impact to road infrastructure. Adaptation strategies identified in the report include:

> ...increasing the resilience of stream crossings, culverts, and bridges to higher peakflows and facilitating response to higher peakflows by reducing the road system and disconnecting roads from streams. Tactics include completing geospatial databases of infrastructure (and drainage) components, installing higher capacity culverts, and decommissioning roads or converting them to alternative uses. (Halofsky et al. 2018a)

U.S. Forest Service Transportation Resiliency Guidebook provides a review of the impacts of climate change on Forest Service infrastructure, and a process to assess and address climate change impacts at local and regional levels (USDA FS 2018; Table 4). Included in the guidebook is a step-by-step guide for identifying vulnerabilities and preparedness planning within their transportation network (USDA FS 2018). In addition, the guidebook recommends using the forest plan revision process as "an opportunity to analyze baseline conditions and climate change vulnerabilities and to develop climate resilient strategies for the future." (USDA FS 2018). The Forest Service should use the transportation resilience guidebook to inform forest plan revision analysis and plan components to address climate change in the context of the forest's transportation system.

Table 4. Role of adaptation strategies in reducing climate change impacts of Forest Service lands (reprinted from USDA FS 2018).

	Impacts on Transportation	**Example Strategies to Reduce Impacts**
Heavy Precipitation / Flooding	Flooded roadways interrupting service Damage/destruction of roads and bridges Pavement buckling Erosion comprising soil stability and transportation assets Slope failures Landslides damaging and disrupting routes Plugged or blown out culverts	Retrofit facilities Relocate facilities Upgrade culverts and drainage facilities Build new facilities to climate ready standards Protect existing infrastructure Divest in assets
Wildfires	Additional woody debris that plug culverts Reduced slope stability causing increased landslides Increased heavy vehicle traffic wear and tear on FS roadways	Sustain forest ecology Protect forests from severe fire and wind disturbance
Tree Mortality	Fallen trees disrupt access along transportation routes Increased need for clearing hazard trees along roadways Provide forest fuel for wildfire	Facilitate Forest community adjustments through species transitions

Individual forests have also drafted climate mitigation strategies. The Olympic National Forest in Washington, has developed documents oriented at protecting watershed health and species in the face of climate change, including a 2003 travel management strategy and a report entitled, *Adapting to Climate Change in Olympic National Park and National Forest* (USDA FS 2011a). The report calls for road decommissioning, relocation of roads away from streams, enlarging culverts as well as replacing culverts with fish-friendly crossings (Table 5). In the travel management strategy, Olympic National Forest recommended that one third of its road system be decommissioned and obliterated. In addition, the plan called for addressing fish migration barriers in a prioritized and strategic way – most of these are associated with roads.

Table 5: Current and expected sensitivities of fish to climate change and associated adaptation strategies and action for fisheries and fish habitat management and relevant to transportation management at Olympic National Forest and Olympic National Park (reprinted from USDA Forest Service 2011a).

Current and expected sensitivities	**Adaptation strategies and actions**
Changes in habitat quantity and quality	Implement habitat restoration projects that focus on re-creating watershed processes and functions and that create diverse, resilient habitat.
Increase in culvert failures, fill-slope failures, stream adjacent road failures, and encroach-ment from stream-adjacent road segments	Decommission unneeded roads. Remove sidecast, improve drainage, and increase culvert sizing on remaining roads. Relocate stream-adjacent roads.
Greater difficulty disconnecting roads from stream channels	Design more resilient stream crossing structures.
Major changes in quantity and timing of streamflow in transitional watersheds	Make road and culvert designs more conservative in transitional watersheds to accommodate expected changes.
Decrease in area of headwater streams	Continue to correct culvert fish passage barriers. Consider re-prioritizing culvert fish barrier correction projects.
Decrease in habitat quantity and connectivity for species that use headwater streams	Restore habitat in degraded headwater streams that are expected to retain adequate summer streamflow (ONF).

C. Reducing fragmentation to enhance aquatic and terrestrial species adaptation

Reconnecting fragmented forests has been shown to benefit native species (e.g., Damschen et al. 2019). Decommissioning and upgrading roads can reduce fragmentation of both aquatic and terrestrial systems. For example, reducing the amount of road-generated fine sediment deposited on salmonid nests can increase the likelihood of egg survival and spawning success (Switalski et al. 2004, McCaffery et al. 2007). Strategically removing or mitigating barriers such as culverts has been shown to restore aquatic connectivity and expand habitat (Erkinaro et al. 2017). Decommissioning roads in riparian areas may provide further benefits to salmon and other aquatic organisms by permitting reestablishment of streamside vegetation, which provides shade and maintains a cooler, more moderated microclimate over the stream (Battin et al. 2007, Meridith et al. 2014). Coordinating the repair of an aging road system with the mitigation of aquatic organism passage may allow for restoring connectivity while improving infrastructure (Nesson et al. 2018).

One of the most well documented impacts of climate change on wildlife is a shift in the ranges of species (Parmesan 2006). As animals migrate, landscape connectivity will be increasingly important (Holman et al. 2005), and restoring and mitigating migration routes in key wildlife corridors will increase wildlife resiliency. Access management in important elk migration sites would reduce disturbance and improve connectivity (Parton et al. 2017). Similarly, a recent study found grizzly bear population density increased 50 percent following the restriction of motorized recreation (Lamb et al. 2018). Decommissioning roads in key wildlife corridors will also reduce the many road-related stressors. Road decommissioning restores wildlife habitat by providing security and food such as grasses, forbs, and fruiting shrubs (Switalski and Nelson 2011, Tarvainen and Tolvanen 2016).

Forests fragmented by roads and motorized trail networks will likely demonstrate less resistance and resilience to stressors, such as weeds. As a forest is fragmented and there is more edge habitat, Noss (2001) predicts that weedy species with effective dispersal mechanisms will increasingly benefit at the expense of native species. However, decommissioned roads when seeded with native species can reduce the spread of invasive species (Grant et al. 2011), and help restore fragmented forestlands. Off-road vehicles with large knobby tires and large undercarriages are also a key vector for weed spread (e.g., Rooney 2006). Strategically closing and decommissioning motorized routes, especially in roadless areas, will reduce the spread of weeds on forestlands (Gelbard and Harrison 2003).

D. Transportation infrastructure and carbon sequestration

The relationship of road restoration and carbon has only recently been explored. There is the potential for large amounts of carbon (C) to be sequestered by restoring roads to a more natural state. When roads are decompacted during reclamation, vegetation and soils can develop more rapidly and sequester large amounts of carbon. Research on the Clearwater National Forest in Idaho estimated total soil C storage increased 6-fold compared to untreated abandoned roads (Lloyd et al. 2013). Another study concluded that reclaiming 425 km (264 miles) of logging roads over the last 30 years in Redwood National Park in Northern California resulted in net carbon savings of 49,000 Megagrams (54,013 tons) of carbon to date (Madej et al. 2013, Table 5). A further analysis found that recontouring roads had higher soil organic carbon than ripping (decompacting) the roads (Seney and Madej 2015). Finally, a recent study in Colorado found that adding mulch or biochar to decommissioned roads can increase the amount of carbon stored in soil (Ramlow et al. 2018).

Kerekvliet et al. (2008) used Forest Service estimates of the fraction of road miles that are unneeded, and calculated that restoring 126,000 miles of roads (i.e. 30% of the road system) to a natural state would be equivalent to revegetating an area larger than Rhode Island. In addition, they calculate that the net economic benefit of road treatments are always positive and range from US $0.925-1.444 billion.

Table 6. Carbon budget implications in road decommissioning projects (reprinted from Madej et al. 2013).

Road Decommissioning Activities and Processes	Carbon Cost	Carbon Savings
Transportation of staff to restoration sites (fuel emissions)	X	
Use of heavy equipment in excavations (fuel emissions)	X	
Cutting trees along road alignment during hillslope recontouring	X	
Excavation of road fill from stream crossings		X
Removal of road fill from unstable locations		X
Reduces risk of mass movement		X
Post-restoration channel erosion at excavation sites	X	
Natural revegetation following road decompaction		X
Replanting trees		X
Soil development following decompaction		X

E. The importance of Roadless Areas and intact mature forests

Undeveloped natural lands provide numerous ecological benefits. They contribute to biodiversity, enhance ecosystem representation, and facilitate connectivity and provide high quality or undisturbed water, soil and air (Strittholt and Dellasala 2001, DeVelice and Martin 2001, Crist and Wilmer 2002, Loucks et al. 2003, Dellasalla et al. 2011, Anderson et al. 2012, Selva et al. 2015). They can also serve as ecological baselines to help us better understand our impacts to other landscapes, and contribute to landscape resilience in the face of climate change.

Forest Service roadless lands, in particular, are heralded for the conservation values they provide. The benefits are described at length in the preamble of the Roadless Area Conservation Rule (RACR)[4] as well as in the Final Environmental Impact Statement (FEIS) for the RACR[5], and include: high quality or undisturbed soil, water, and air; sources of public drinking water; diversity of plant and animal communities; habitat for threatened, endangered, proposed, candidate, and sensitive species and for those species dependent on large, undisturbed areas of land; primitive, semi-primitive non- motorized, and semi-primitive motorized classes of dispersed recreation; reference landscapes; natural appearing landscapes with high scenic quality; traditional cultural properties and sacred sites; and other locally identified unique characteristics (e.g., include uncommon geological formations, unique wetland complexes, exceptional hunting and fishing opportunities).

The Forest Service, National Park Service, and the U.S. Fish and Wildlife Service recognize that protecting and connecting roadless or lightly roaded areas is an important action agencies can take to enhance climate change adaptation. For example, the *Forest Service National Roadmap for Responding to Climate Change* (USDA Forest Service 2011b) establishes that increasing connectivity and reducing fragmentation are short- and long-term actions the Forest Service should take to facilitate adaptation

[4] Federal Register, Vol. 66, No. 9. January 12, 2001. Pages 3245-3247.
[5] Final Environmental Impact Statement, Vol. 1, 3–3 to 3–7

to climate change. The National Park Service also identifies connectivity as a key factor for climate change adaptation along with establishing "blocks of natural landscapes large enough to be resilient to large-scale disturbances and long-term changes," and other factors. The agency states that: "The success of adaptation strategies will be enhanced by taking a broad approach that identifies connections and barriers across the landscape. Networks of protected areas within a larger mixed landscape can provide the highest level of resilience to climate change."[6] Similarly, the *National Fish, Wildlife and Plants Climate Adaptation Partnership's Adaptation Strategy* (2012) calls for creating an ecologically-connected network of conservation areas.[7]

Crist and Wilmer (2002) looked at the ecological value of roadless lands in the Northern Rockies and found that protection of national forest roadless areas, when added to existing federal conservation lands in the study area, would 1) increase the representation of virtually all land cover types on conservation lands at both the regional and ecosystem scales, some by more than 100%; 2) help protect rare, species-rich, and often-declining vegetation communities; and 3) connect conservation units to create bigger and more cohesive habitat "patches."

Roadless lands also are responsible for higher quality water and watersheds. Anderson et al. (2012) assessed the relationship of watershed condition and land management status and found a strong spatial association between watershed health and protective designations. Dellasalla et al. (2011) found that undeveloped and roadless watersheds are important for supplying downstream users with high-quality drinking water, and developing these watersheds comes at significant costs associated with declining water quality and availability. The authors recommend a light-touch ecological footprint to sustain the many values that derive from roadless areas including healthy watersheds.

Allowing roadless and other intact forested areas to reach their full ecological potential is an effective and crucial strategy for atmospheric carbon dioxide removal. Moomaw et al (2019) termed this approach as "proforestation" and explained,

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[6] National Park Service. Climate Change Response Program Brief. http://www.nature.nps.gov/climatechange/adaptationplanning.cfm. Also see: National Park Service, 2010. Climate Change Response Strategy. http://www.nature.nps.gov/climatechange/docs/NPS_CCRS.pdf. Objective 6.3 is to "Collaborate to develop cross-jurisdictional conservation plans to protect and restore connectivity and other landscape-scale components of resilience."

[7] See http://www.wildlifeadaptationstrategy.gov/pdf/NFWPCAS-Chapter-3.pdf. Pages 55- 59. The first goal and related strategies are:

> Goal 1: Conserve habitat to support healthy fish, wildlife, and plant populations and ecosystem functions in a changing climate.
>
> Strategy 1.1: identify areas for an ecologically-connected network of terrestrial, freshwater, coastal, and marine conservation areas that are likely to be resilient to climate change and to support a broad range of fish, wildlife, and plants under changed conditions.
>
> Strategy 1.2: Secure appropriate conservation status on areas identified in Strategy 1.1 to complete an ecologically-connected network of public and private conservation areas that will be resilient to climate change and support a broad range of species under changed conditions.
>
> Strategy 1.4: Conserve, restore, and as appropriate and practicable, establish new ecological connections among conservation areas to facilitate fish, wildlife, and plant migration, range shifts, and other transitions caused by climate change.

> [f]ar from plateauing in terms of carbon sequestration (or added wood) at a relatively young age as was long believed, older forests (e.g., >200 years of age without intervention) contain a variety of habitats, typically continue to sequester additional carbon for many decades or even centuries, and sequester significantly more carbon than younger and managed stands, (Luyssaert et al., 2008; Askins, 2014; McGarvey et al., 2015; Keeton, 2018).

The authors recommend "scaling up" proforestation, which includes both protecting and expanding designations of intact forested areas, as a cost-effective means to increase atmospheric carbon sequestration.

III. Achieving a Sustainable Minimum Road System on National Forest Lands

A. Background

For two decades, the Travel Management Rule, 36 C.F.R. Part 212, has guided Forest Service road management and use by motorized vehicles. It is divided into three parts: Subpart A, the administration of the forest transportation system; Subpart B, designation of roads, trails, and areas for motor vehicle use; and Subpart C, use by over-snow vehicles. *See* 36 C.F.R. Part 212.

Table 7. Travel Management Rule Subparts – Objectives, Requirements & Products

36 C.F.R. §212	Objective:	Requires:	Product(s):
Subpart A; Roads Rule 2001	To achieve a sustainable national forest road system.	Use a science-based analysis to identify the minimum road system and roads for decommissioning	- Travel Analysis Report - Map with roads identified as "likely needed" and "likely unneeded"
Subpart B; Travel Management Rule 2005	To protect forests from unmanaged off-road vehicle use by ending cross-country travel and ensuring the agency minimizes the harmful effects from motorized recreation.	Designating a system of roads, trails and areas available for off-road vehicle use according to general and specific criteria.	- Motor Vehicle Use Maps that indicate what roads/trails are open for motorized travel
Subpart C; Travel Management Rule	To protect forests from unmanaged over-snow vehicle use in a manner that minimizes their harmful effects.	Designating specific roads, trails and/or areas for oversnow vehicle use according to the criteria per Subpart B.	- Oversnow vehicle maps designating trails and areas for winter motorized recreation

This broad-based national rule is needed because at over 370,000 miles, the Forest Service road system is long enough to circle the earth over 14 times and it is over twice the size of the National Highway System.[8] It is also indisputably unsustainable from ecological, economic and management perspectives. The majority of the roads were constructed decades ago when design and management techniques did not meet current standards (Gucinski et al. 2000, Endicott 2008), making them more vulnerable to erosion and decay. Further, current design standards and best management practices have not been updated to address climate change realities. Exacerbating the problem are massive Forest Service road maintenance backlogs that forces the agency to forego actions necessary to ensure proper watershed function, such as preventing sediment pollution and sustaining aquatic organism passages. Nationally, the total deferred maintenance backlog reached $5.5 billion in FY 2019 of which $3.1 billion is associated with roads.[9] As a result, the road network is not only a massive economic liability, it is also actively harming National Forest System lands, waters, fish and wildlife.

Over the past two decades the Forest Service - largely due to the Travel Management Rule - has made some limited efforts to identify and implement a sustainable transportation system. Yet, overall the agency has yet to meet the requirements of Subpart A. The challenge for forest managers is figuring out what is a sustainable road system and how to achieve it – a challenge exacerbated by climate change. It is reasonable to define a sustainable transportation system as one where all the roads and trails are located, constructed, and maintained in a manner that minimizes harmful environmental consequences while providing social benefits and within budget constraints. This could potentially be achieved through the use of effective best management practices. However, the reality is that even the best transportation networks can be problematic simply because they exist and usher in land uses that, without the access, would not occur (Trombulak and Frissell 2000, Carnefix and Frissell 2009, USDA Forest Service 1996), and when they are not maintained to the designed level they result in environmental problems (Endicott 2008; Gucinski et al. 2000). Moreover, what was sustainable yesterday may no longer be sustainable under climate change realities since roads designed to meet older climate criteria may no longer hold up under new scenarios (USDA Forest Service 2010, USDA Forest Service 2011b, USDA Forest Service 2012a, AASHTO 2012, Schwartz et al. 2014, USDA FS 2018, Halofsky et al. 2018a, 2018b).

Given consistent budget shortfalls and increasing risks from climate change vulnerabilities, it is clear the agency has an urgent need to both identify and implement a minimum road system, one that will ensure the protection of all Forest Service system lands. However, without specific direction from the Forest Service's Washington D.C. office or Congress, it is reasonable to expect the agency will continue to rely on piecemeal, project-level analyses to identify the minimum road system. Such an approach is inefficient, and insufficient to achieve a sustainable road system forestwide.

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[8] USDOT Federal Highway Administration, Office of Highway Policy Information.
https://www.fhwa.dot.gov/policyinformation/pubs/hf/pl11028/chapter1.cfm
[9] USDA Forest Service. 2019. FY2020 Budget Justification. p.83.

Further, where the Forest Service does act to comply with Subpart A, it typically fails to consider shortcoming in its previous travel analysis processes. In fact, an independent review of 38 Travel Analysis Processes and corresponding reports conducted in 2016 by the U.S. Department of Transportation John A. Volpe National Transportation Systems Center found three overarching concerns:

- A lack of clarity regarding the process;
- Failure to follow 36 CFR 212.5(b) direction and Washington Office guidance; and
- Omission of required documents, referenced appendices, or key supporting materials.

Compounding these concerns is the fact that not only do project-level NEPA analyses fail to account for the TAP shortcomings, they also fail to consider real road/motorized densities when identifying the minimum road system. Moreover, these analyses erroneously assume best management practices and project-specific design features will be effective when the Forest Service authorizes actions to achieve a sustainable road system. Finally, if the project-level decision includes actual road decommissioning, the analysis typically fails to consider or specify treatments, resulting in a legacy of ghost-roads persisting on the landscape. The following sections expand on these shortcomings, which the Forest Service must consider in all project-level analyses, and when revising its land and travel management plans.

B. Using Real Road and Motorized Trail Densities to Identify a Minimum Road System

As the Forest Service works to comply with Subpart A, it is crucial that the agency incorporate the true road and motorized trail densities in both its travel analysis process and NEPA-level analyses. Further, the agency must establish standards in land management plan revisions and amendments to ensure each forest achieves an ecologically sustainable minimum road system. Road density analyses should include closed roads, non-system roads, temporary roads, and motorized trails. Typically, the Forest Service calculates road density by looking only at open system road density. From an ecological standpoint, this is a flawed approach since it leaves out the density calculations of a significant percent of roads and motorized trails on the landscape. These additional roads and motorized trails impact fish, wildlife, and water quality, and in some cases, have more of an impact than open system roads. In this section, we provide justification for why a road density analyses should include more than just open road density whenever the Forest Service evaluates the ecological health of an area during NEPA-level analysis or other processes such as for watershed assessments, forest plan revisions or during travel analysis.

Impacts of closed roads

It is crucial to distinguish the density of roads physically present on the landscape, whether closed to vehicle use or not, from "open-road density." An open-road density of 1.5 mi/mi^2 has been established as a standard in some national forests as protective of some terrestrial wildlife species. However, many areas with an open road density of 1.5 mi/mi^2 often have more miles of closed roads which are still hydrologically connected and negatively affecting aquatic and wildlife habitat. This higher density occurs because many road "closures" may block vehicle access, but do nothing to mitigate the hydrologic alterations the road causes. The problem is often further compounded by the existence of "ghost" roads that are not captured in agency inventories, but that are nevertheless physically present and causing hydrologic alteration (Pacific Watershed Associates 2005).

Closing a road to public motorized use can mitigate the impacts on water, wildlife, and soils only if proper closure and storage techniques are followed. Flow diversions, sediment runoff, and illegal incursions will continue unabated if the road is not hydrologically stabilized and adequately blocked from motorized traffic. The Forest Service's National Best Management Practices for non-point source pollution recommends the following management techniques for minimizing the aquatic impacts from closed system roads: eliminate flow diversion onto the road surface, reshape the channel and streambanks at the crossing-site to pass expected flows without scouring or ponding, maintain continuation of channel dimensions and longitudinal profile through the crossing site, and remove culverts, fill material, and other structures that present a risk of failure or diversion (USDA Forest Service 2012b).

As noted above, many species benefit when roads are closed to motorized use. However, the fact remains that closed system roads are often breached resulting in impacts to fish and wildlife. A significant portion of gates and closure devices are ineffective at preventing motorized use (Griffin 2004, USFWS 2007). For example, in a legal decision from the Utah District Court, *Sierra Club v. USFS*, Case No. 1:09-cv-131 CW (D. Utah March 7, 2012), the court found that, as part of analyzing alternatives in a proposed travel management plan, the Forest Service failed to examine the impact of continued illegal use. In part, the court based its decision on the Forest Service's acknowledgement that illegal motorized use is a significant problem and that the mere presence of roads is likely to result in illegal use.

In addition to the disturbance to wildlife from motorized use, incursions and the accompanying human access can also result in illegal hunting and trapping of animals. The Tongass National Forest refers to this in its EIS to amend the Land and Resources Management Plan. Specifically, the Forest Service notes in the EIS that Alexander Archipelago wolf mortality due to legal and illegal hunting and trapping is related not only to roads open to motorized access, but to all roads, and that *total road densities* of 0.7-1.0 mi/mi^2 or less may be necessary (USDA Forest Service 2008).

Impacts of unauthorized (non-system) roads

As of 1998, there were approximately 130,000 miles of non-system roads in national forests (USDA Forest Service, 1998). However, the creation of unauthorized roads continues to be a problem as the Forest Service struggles to properly enforce travel management plans protecting areas from motorized travel. No requirements are in place directing the agency to track or inventory unauthorized roads, therefore currently their precise number is unknown. These roads contribute significantly to the environmental impacts of the transportation system on forest resources, just as forest system roads do. Because the purpose of a road density analysis is to measure the impacts of roads at a landscape level, the only way to do this is for the Forest Service to include all roads, including non-system roads, when measuring impacts. An all-inclusive analysis will provide a more accurate representation of the environmental impacts of the road network within the analysis area.

Impacts of temporary roads

Temporary roads are not considered system roads. Most often they are constructed in conjunction with timber sales. Temporary roads have the same types of environmental impacts as system roads, although at times the impacts can be worse if the road persists on the landscape because they are not built to last. It is important to note that although they are termed temporary roads, their impacts are not temporary. According to Forest Service Manual (FSM) 7703.1, the agency is required to "Reestablish vegetative cover on any unnecessary roadway or area disturbed by road construction on National Forest System lands within 10 years after the termination of the activity that required its use and construction."

Regardless of the FSM 10-year direction, temporary roads often remain for much longer because timber sale contracts typically last 3-5 years or more. If the timber purchaser builds a temporary road in the first year of a five-year contract, its intended use may not end until the full project is complete, which can include post-harvest actions such as prescribed burning. Even though the contract often requires the purchaser to close, obliterate and seed the roadbed with native vegetation, this work typically occurs after a few years of treatment activities. The temporary road, therefore, could remain open for 7-8 years or longer before the FSM ten-year clock starts ticking. Therefore, temporary roads can legally remain on the ground for up to 20 years or more, yet they are constructed with fewer environmental safeguards than modern system roads. Exacerbating the problem is the rise of landscape-scale projects that last between 10-20 years. Unless there is explicit direction requiring temporary road removal within a certain time after treatment activities, it is likely these roads could persist for decades.

Impacts of motorized trails

Motorized use on trails has serious harmful effects similar to roads, and it is crucial for the Forest Service to include motorized trails in its density calculations. As we note several times in Section I above, scientific research and agency publications find similar impacts between motorized trails and roads. Off-road vehicle (ORV) use on trails impact multiple resources, resulting in soil compaction and erosion, trampling of vegetation, as well as wildlife habitat loss, disturbance, and direct mortality. Many of these impacts increase on trails not planned or designed for vehicles, as is often the case when the Forest Service designates ORVs on trails built for hiking or equestrian uses. In many instances the agency designates motorized use on unauthorized trails created through illegal use or from a legacy of unmanaged cross-country travel, further exacerbating the related harmful effects. For a full review of the environmental and cultural impacts on forest lands see Switalski and Jones (2012), and for a review of impacts in arid environments see Switalski (2018).

C. Using Best Management Practices to Achieve a Sustainable Road System

Numerous Best Management Practices (BMPs) were developed to help create a more sustainable transportation system and identify restoration opportunities. BMPs provide science-based criteria and direction that land managers follow in making and implementing decisions about human uses and projects that affect natural resources. Several states have developed BMPs for road construction, maintenance, and decommissioning practices (e.g., Logan 2001, Merrill and Cassaday 2003). The report entitled, National Best Management Practices for Water Quality Management on National Forest System Lands, includes specific road BMPs for controlling erosion and sediment delivery into waterbodies and maintaining water quality (USDA FS 2012b). These BMPs cover road system planning, design, construction, maintenance, and decommissioning as well as other transportation-related activities.

Forest Service BMPs - Implementation and Effectiveness

While national BMPs have been established, the effectiveness of individual BMPs, and whether they are implemented at all, is in question. Furthermore, design features are increasingly replacing BMPs for project-level mitigation of road-related environmental impacts. These design features are not consistent among projects, but rather adapted from forest plans and state BMPs, rather than national Forest Service guidelines. Design features need to be standardized, and their rate of implementation and effectiveness systematically reviewed.

When considering how effective BMPs are at controlling nonpoint pollution on roads, both the rate of implementation, and their effectiveness should both be considered. The Forest Service tracks the rate of implementation and the relative effectiveness of BMPs from in-house audits. This information is summarized in the *National BMP Monitoring Summary Report* with the most recent data being the fiscal years 2013-2014 (Carlson et al. 2015). The rating categories for implementation are "fully implemented," "mostly implemented," "marginally implemented," "not implemented," and "no BMPs." "No BMPs" represents a failure to consider BMPs in the planning process. More than a hundred evaluations on roads were conducted in FY2014. Of these evaluations, only about one third of the road BMPs were found to be "fully implemented" (Carlson et al. 2015, p. 12).

The monitoring audit also rated the relative effectiveness of the BMP. The rating categories for effectiveness are "effective," "mostly effective," "marginally effective," and "not effective." "Effective" indicates no adverse impacts to water from project or activities were evident. When treated roads were evaluated for effectiveness, almost half of the road BMPs were scored as either "marginally effective" or "not effective" (Carlson et al. 2015, p. 13). However, BMPs for completed road decommissioning projects showed approximately 60 percent were effective and mostly effective combined, but it was unclear what specific BMPs account for this success (Carlson et al.

2015, p. 35). As explained below, road recontouring that restores natural hillside slopes is a more effective treatment compared to those that leave road features intact.

A recent technical report by the Forest Service entitled, *Effectiveness of Best Management Practices that Have Application to Forest Roads: A Literature Synthesis* summarized research and monitoring on the effectiveness of different BMP treatments for road construction, presence and use (Edwards et al. 2016). They found that while several studies have found some road BMPs are effective at reducing delivery of sediment to streams, the degree of each treatment has not been rigorously evaluated (Edwards et al. 2016). Few road BMPS have been evaluated under a variety of conditions, and much more research is needed to determine the site-specific suitability of different BMPs (Edwards et al. 2016, also see Anderson et al. 2011).

Edwards et al. (2016) cites several reasons for why BMPs may not be as effective as commonly thought. Most watershed-scale studies are short-term and do not account for variation over time, sediment measurements taken at the mouth of a watershed do not account for in-channel sediment storage and lag times, and it is impossible to measure the impact of individual BMPs when taken at the watershed scale. When individual BMPs are examined there is rarely broad-scale testing in different geologic, topographic, physiological, and climatic conditions. Further, Edwards et al. (2016) observes, "The similarity of forest road BMPs used in many different states' forestry BMP manuals and handbooks suggests a degree of confidence validation that may not be justified," because they rely on just a single study. Therefore, BMP effectiveness would require matching the site conditions found in that single study, a factor land managers rarely consider.

Climate change will further put into question the effectiveness of many road BMPs (Edwards et al. 2016). While the impacts of climate will vary from region to region (Furniss et al. 2010), more extreme weather is expected across the country which will increase the frequency of flooding, soil erosion, stream channel erosion, and variability of streamflow (Furniss et al. 2010). BMPs designed to limit erosion and stream sediment for current weather conditions may not be effective in the future. Edwards et al. (2016) states, "More-intense events, more frequent events, and longer duration events that accompany climate change may demonstrate that BMPs perform even more poorly in these situations. Research is urgently needed to identify BMP weaknesses under extreme events so that refinements, modifications, and development of BMPs do not lag behind the need."

The uncertainties about BMP effectiveness as a result of climate change, compounded by the inconsistencies revealed by BMP evaluations, suggest that the Forest Service cannot simply rely on them, or design features/criteria, as a means to mitigate project-level activities. This is especially relevant where the Forest Service relies on the use of BMPs instead of fully analyzing potentially harmful environmental consequences from road design, construction, maintenance or use, in studies and/or programmatic and site-specific NEPA analyses.

D. Effectiveness of Road Decommissioning Treatments

In order to truly achieve a sustainable minimum road system, the Forest Service must effectively remove unneeded roads. According to the Forest Service, the objective of road decommissioning is to "stabilize, restore, and revegetate unneeded roads to a more natural state to protect and enhance NFS lands" (FSM 7734.0). However, rather than actively removing roads, the Forest Service is increasingly relying on abandoning roads to reach decommissioning treatment objectives (Apodaca et al.2018). Simply closing or abandoning roads will lead to continued resource damage. Other treatments such as ripping the roadbed or installing drainage such as waterbars or dips, have limited and often short-term benefits to natural resources (e.g., Luce 1997, Switalski et al. 2004, Nelson et al. 2010). Recontouring roads is the only proven method to attain the intended outcome of road decommissioning.

Several studies have documented the benefits of fully recontouring roads for ecological restoration. Lloyd et al. (2013) found that rooting depths were much deeper in recontoured roads than in abandoned roads in Idaho, and soil organic matter was an order of magnitude higher on recontoured roads than abandoned roads. Further studies show that soil carbon storage is much higher on recontoured roads as well. A study in Northern California found that recontouring roads resulted in higher soil organic carbon than ripping the roads (Seney and Madej 2015). Higher tree growth and wildlife use has also been found on and near recontoured roads than ripped or abandoned roads (Kolka and Smidt 2004, Switalski and Nelson 2011). Switalski and Nelson (2011) found increased use by black bears on recontoured roads than closed or abandoned roads due to increased food availability and increased habitat security. In addition, removing culverts at stream crossings results in restoring aquatic connectivity and expanding habitat (Erkinaro et al. 2017).

Legacy Roads Monitoring Project

Since 2008, the Forest Service Rocky Mountain Research Station has conducted systematic monitoring on the effectiveness of decommissioned roads in reducing hydrologic and geomorphic impacts from the Forest Service road network. One intent of the monitoring project was to gauge the success of the Legacy Roads and Trails Program that Congress established to provide dedicated funding for the treatment and removal of unnecessary forest roads. The monitoring found that recontouring roads and restoring stream crossings results in dramatic declines in road-generated sediment. Storm-proofing treatments lead to fewer benefits, and on control sites (untreated or abandoned roads), high levels of sediment delivery continued, and the risk of culvert failures remained. For example, a study on the Lolo Creek Watershed on the Clearwater National Forest found a 97% reduction in road/stream connectivity following road recontour (Cissel et al. 2011). Using field observations and the Geomorphic Roads Analysis and Inventory Package (GRAIP), they found a reduction of fine sediments from 38.1 tonnes/year to 1.3 tonnes/year along 3.5 miles of road. Furthermore, they found that restoring road/stream crossings eliminated the risk of culverts plugging, stream diversions, and fill lost at culverts (Table 8).

On the other hand, monitoring conducted on the Caribou-Targhee National Forest found only a 59% reduction of fine sediment delivery from a combination of storm proofing (installation of drain dips), ripping, tilling, and outsloping techniques. There was a reduction of 34.9 tons/year to 14.1 ton/year – leaving a significant amount of sediment continuing to be delivered to streams. Additionally, some stream crossing culverts were not treated and the risk of plugging remained leaving 330 m^3 of fill material at risk. While trail conversion and decommissioning treatments reduced slope failure risks, in some cases storage treatments actually increased the risk of failure (Nelson et al. 2010). Additional monitoring studies conducted in Montana, Idaho, Washington, Oregon, and Utah have similar results.[10]

Table 8. Summary of GRAIP road risk predictions for a watershed on the Clearwater National Forest road decommissioning treatment project (reprinted from Cissel et al. 2011).

IMPACT/RISK TYPE	EFFECT OF TREATMENT: INITIAL GRAIP PREDICTION
Road-stream hydrologic connectivity	-97%, -2510 m
Fine Sediment Delivery	-97%, -36.8 tonnes/yr.
Landslide Risk	Reduced to near natural condition
Gully Risk	Reduced from very low to negligible
Stream Crossing Risk -plug potential -fill at risk -diversion potential	 -100% eliminated at 9 sites -100%, 268 m^3 fill removed -100%, eliminated at 3 sites
Drain Point Problems	17 problems removed, 4 new problems

The Forest Service recognizes that fundamental to road decommissioning is revegetating the roadbed. FSM 7734 states, "Decommission a road by reestablishing vegetation and, if necessary, initiating restoration of ecological processes interrupted or adversely impacted by the unneeded road." However, roads are inherently difficult to revegetate because of compaction, lack of soil and organic material, low native seedbank, and presence of noxious weeds (Simmers and Galatowitsch 2010, Ramlow et al. 2018). Many recently acquired industrial timberlands (e.g. Legacy Lands) have

[10] For reports visit https://www.fs.fed.us/GRAIP/LegacyRoadsMonitoringStudies.shtml

road systems with limited canopy cover, little woody debris available, and a large weed seedbank. Thus, revegetation is going to be particularly challenging on these lands.

Consistent application of BMPs that direct recontouring roads for decommissioning will be essential to ensure the treatments best achieve improvements in ecological conditions. More than any other treatment, road recontouring ensures complete decompaction of the roadbed, incorporates native soils that were side-cast during construction, and prevents motorized use. This in turn increases plant rooting depths, soil carbon storage, tree growth, and wildlife use. Any earth disturbing activity can create conditions favorable to noxious weeds, so treating weeds before any treatment and ensuring quick revegetation can limit weeds spread. Applying road recontour BMPs that also mitigate risks associated with noxious weed expansion will help prevent their spread

Conclusion

Numerous studies show that roads and motorized trails negatively impact the ecological integrity of terrestrial and aquatic ecosystems and watersheds. There is ample evidence to confirm the harm to wildlife, aquatic species, water quality, and natural processes from forest roads and motorized use. In addition, the evolving science surrounding roads and wildfire demonstrate a direct link between access and human-caused ignitions, and also suggests that land managers must consider how roads affect fire behavior. Minimizing these impacts by reducing road densities could be an effective solution.

An increasing body of literature exists demonstrating that not only is the Forest Service's transportation infrastructure highly vulnerable to climate change, but also that roads exacerbate climate change's harmful effects to other resources. The agency itself has published multiple reports and guidelines for adaptation, yet few forests are fully translating the information into tangible actions. The Forest Service must implement climate change adaptations as soon as possible, including protecting and expanding intact forests as part of a growing effort to promote natural climate change solutions. Opportunities exist to reduce fragmentation, sequester carbon, and expand roadless areas by implementing a minimum road system.

The Forest Service must fulfil its mandate to achieve an ecologically and economically sustainable forest road system by fully complying with the Roads Rule's requirement to identify a minimum road system. Inconsistent policy interpretations, inadequate travel analysis reports and lack of accountability has largely left this goal wholly out of reach. Yet this work remains vitally important, especially in the context of climate change. The Forest Service should reinvigorate its efforts to comply with the rule's requirements. Towards this end, the agency must include current science, particularly related to future climate conditions. All road and motorized trail densities should be included in the analysis. When the agency actually does identify a minimum road system and proposes to remove unneeded roads, it must carefully evaluate the effectiveness of all proposed BMPs and design features, and fully implement the most effective decommissioning treatments to maximize restoring ecological integrity to the area. These actions will ensure the Forest Service finally achieves its goal to establish a truly sustainable forest road system.

Recontoured road, Olympic National Forest - Skokomish Watershed, 2017. By WildEarth Guardians

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PERSPECTIVE

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

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

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

Wildfire is a key driver of ecosystem change that increasingly poses a significant threat and cost to society. In western North America (hereafter, the West), warming, frequent droughts, and legacies of past management combined with expansion of residential development have made social–ecological systems (SESs) more vulnerable to wildfire. As the annual area burned has increased over the past three decades, we are confronting longer fire seasons (1, 2), more large fires (3, 4), a tripling of homes burned (5), and more frequent large evacuations. In 2016, the Fort McMurray Fire in Alberta, Canada and the Blue Cut Fire in southern California prompted evacuation orders for a combined total of more than 160,000 people. The costs of wildfire have also risen substantially since the 1990s. The US Congress appropriated $13 billion for fire suppression and $5 billion for fuels management in fiscal years 2006–2015 (6). Other societal costs, including real estate devaluation, emergency services, and postfire rehabilitation, total up to 30 times the direct cost of firefighting (7).

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

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

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

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

Why Has Coping with Wildfire Become Such a Challenge?

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

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

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

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

Potential Consequences of Future Wildfire

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

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

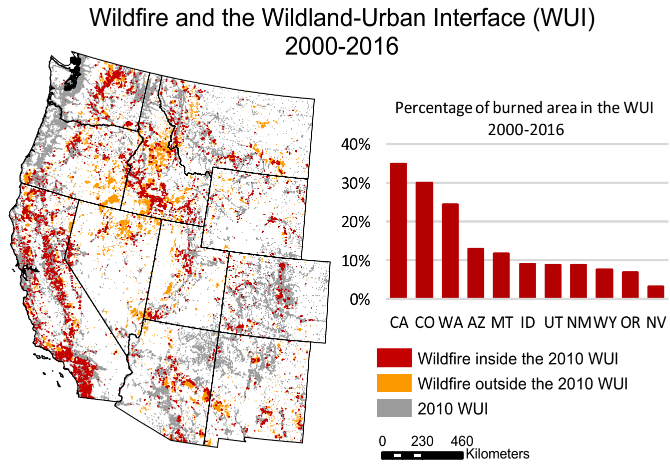

Fig. 1. **(*Left*) Area burned by wildfires between 2000 and 2016 across the western United States inside and outside the 2010 WUI including a 2.5-km community protection zone (27). (*Right*) About 15% of the WUI burned during this period, with largest proportions of the WUI burning in California, Colorado,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

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

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

Specified, Adaptive, and Transformative Resilience to Wildfire

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

Defining resilience to wildfire in an SES is especially challenging in the WUI, where people, ecosystems, and wildfire interact over multiple spatial and temporal scales (12). An SES is the intersection and interdependence of biophysical units and associated people and institutions. Resilience in an SES generally has been defined as the capacity to absorb disturbance so as to retain essential structures, processes, and feedbacks and to adapt to and reorganize following disturbance (45). These perspectives of resilience, absorbing versus adapting to disturbance, offer different guiding principles for policy and management in responding to wildfire and measuring success over different planning timelines (44). Here we outline a consistent framework that defines resilience to wildfire in coupled SESs based on the concepts of specified resilience and general resilience, the latter of which includes adaptive and transformative approaches (Table S1) (15, 16, 44).

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

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

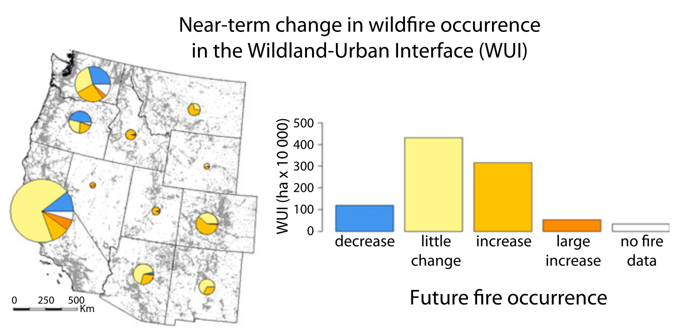

Fig. 2. **(*Left*) Area of the WUI in the conterminous western United States, classified according to projected near-term changes in fire occurrence. The size of each pie is scaled relative to the area of the WUI (both intermix and interface) in each state, based on data from Martinuzzi, et al. (27).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

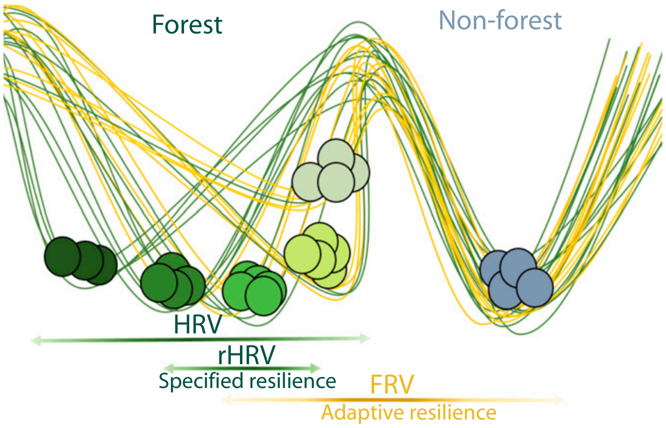

Fig. 3. Conceptual ball-and-basin representation of specified and adaptive resilience across a forested landscape. Lines defining basins depict the ranges of variation in fire regimes across forest types. Sets of green balls reflect the variation in abundance and composition within different forest types,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

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

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

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

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

How Policy and Management Can Promote Adaptive Resilience to Wildfire

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

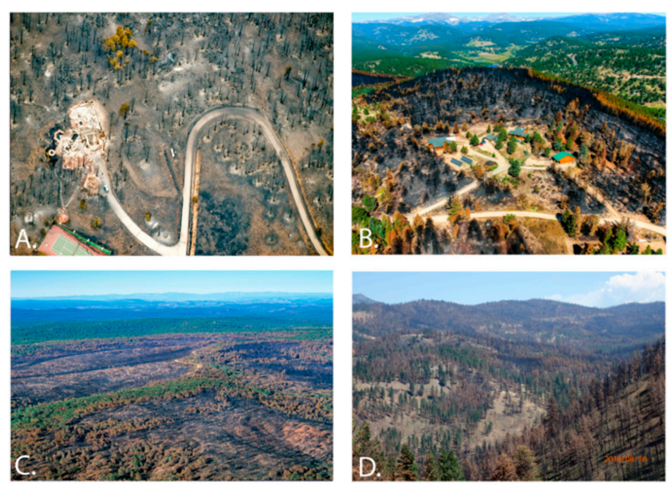

Fig. 4. Wildfires are catalysts of change that promote adaptive resilience by communities and ecosystems to future wildfires. (*A* and *B*) Example of adaptation in communities. (*A*) A home burned in the 2010 Fourmile fire, Boulder County, CO, which at the time was the most destructive fire in Colorado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

Managing Wildfire

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

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

Challenges to increasing use of managed and prescribed fires vary from the public's limited experience with smoke and wildfire to significant direct health impacts of smoke on vulnerable populations, including children, the elderly, and low-income communities (40, 70, 71). Some smoke hazards can be reduced through careful planning and management of fire, public health monitoring, and provisioning of health services for vulnerable populations. Public perceptions of fire are also an important hurdle, given the success of Smokey Bear's fire-prevention campaign and because most urban and suburban residents have very limited experience with wildfire compared with rural residents of the early 20th century. Therefore, public education programs that demonstrate the inevitability of wildfire will be a key aspect of living with increasing fire in the West. We need to develop a new fire culture. Despite these and various legal and operational challenges (58), the benefits of prescribed fire and managed wildfires to ecosystems and communities are high (72). Promoting more wildfire away from people and prescribed fires near people and the WUI are important steps toward augmenting the adaptive resilience of ecosystems and society to increasing wildfire.

Managing Fuels

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

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

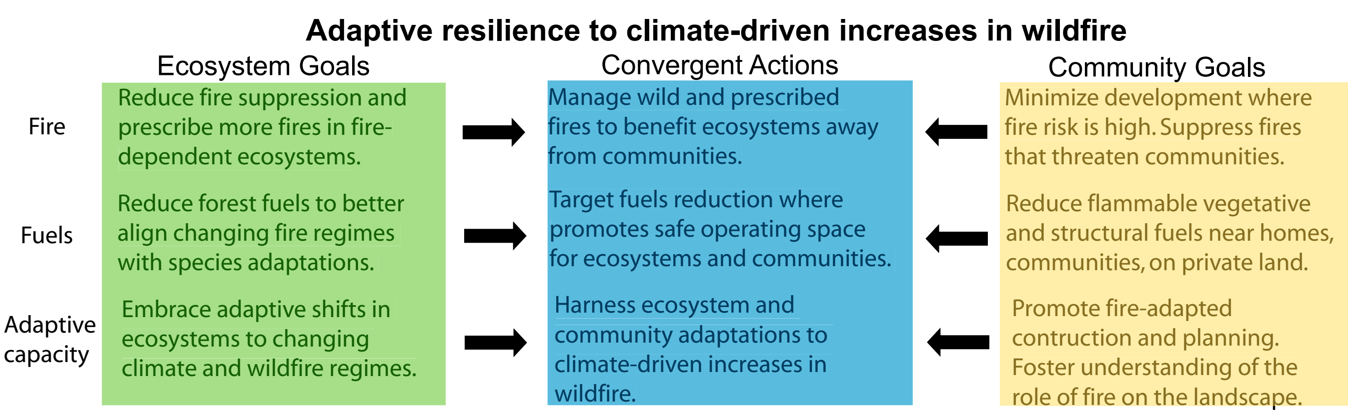

Fig. 5. Convergent actions that promote adaptive resilience to climate-driven increases in wildfire in the West by ecosystems and communities, based on goals related to management of fire, fuels, and adaptive capacity.

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

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

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

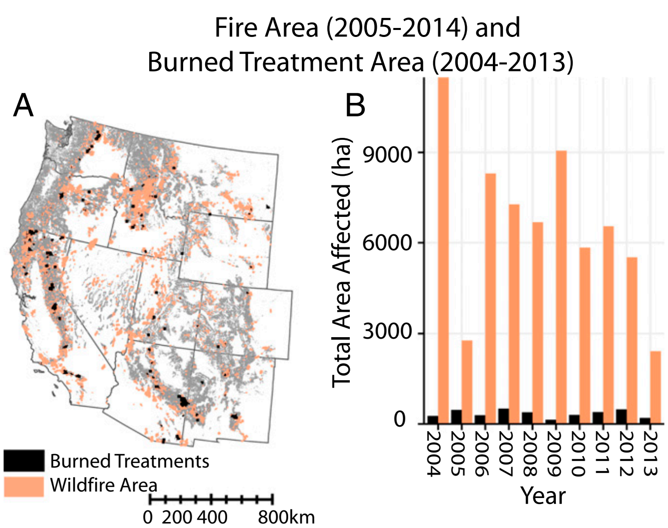

Fig. 6. (*A*) Spatial distribution and area of US Forest Service fuels treatments from 2004–2013 and wildfire from 2005–2014 across forests and woodlands in the western United States. About 3% of the total treated area and 10% of the total number of treatments burned in the period 2005–2014. (*B*) Annual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

Promoting Adaptive Capacity

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

Creating Fire-Adapted Communities. The majority of home building on fire-prone lands occurs in large part because incentives are misaligned, where risks are taken by homeowners and communities but others bear much of the cost if things go wrong. Therefore, getting incentives right is essential, with negative financial consequences for land-management decisions that increase risk and positive financial rewards for decisions that reduce risk. For example, shifting more of the wildfire protection cost and responsibility from federal to state, local, and private jurisdictions would better align wildfire risk with responsibility and provide meaningful incentives to reduce fire hazards and vulnerability before wildfires occur. Currently, much of the responsibility and financial burden for community protection from wildfire falls on public land-management agencies. This arrangement developed at a time when few residential communities were embedded in fire-prone areas. Land-management agencies cannot continue to protect vulnerable residential communities in a densifying and expanding WUI that faces more wildfire (12). The US Government Accountability Office questioned the US Forest Service's prioritizing protection of WUI communities that lie under private and state jurisdictions and has argued for increased financial responsibility for WUI wildfire risk by state and local governments (93). This shift in obligation would enhance adaptive governance and could increase the motivation to pursue adaptive resilience of WUI communities to increasing wildfire (94).

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

Strengthening and expanding programs such as Fire Adapted Communities, Fire Adapted Communities Learning Network, Firewise Communities USA, and FireSmart Canada will also help communities become more fire-adapted. Capacities to assume these responsibilities will vary significantly among homeowners, communities, and local jurisdictions with markedly different risks and resources (99–101). For example, home hazard mitigation programs and community planning tools are more successful in communities at the fringe of urban areas that have more financial resources and often have a greater trust in government than in more isolated, resource-dependent WUI communities, immigrant non–English-speaking communities, or tribal and First Nations communities (101). Although some tax incentives and rebates are available for wildfire risk mitigation on and around homes, more comprehensive programs that include broader incentives and support are needed for meaningful and widespread impacts. Efforts that combine wildfire-specific efforts with other community capacity-building efforts may leverage the networks that enable communities to act on shared notions of risk (102).

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

Conclusions

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

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

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Beyond Fuel Treatment Effectiveness: Characterizing Interactions between Fire and Treatments in the US

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Beyond Fuel Treatment Effectiveness: Characterizing Interactions between Fire and Treatments in the US

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Abstract: In the United States, fuel reduction treatments are a standard land management tool to restore the structure and composition of forests that have been degraded by past management. Although treatments can have multiple purposes, their principal objective is to create landscape conditions where wildland fire can be safely managed to help achieve long-term land management goals. One critique is that fuel treatment benefits are unlikely to transpire due to the low probability that treated areas will be burned by a subsequent fire within a treatment's lifespan, but little quantitative information exists to corroborate this argument. We summarized the frequency, extent, and geographic variation of fire and fuel treatment interactions on federal lands within the conterminous United States (CONUS). We also assessed how the encounters between fuel treatments and fires varied with treatment size, treatment age, and number of times treated. Overall, 6.8% of treatment units evaluated were encountered by a subsequent fire during the study period, though this rate varied among ecoregions across the CONUS. Larger treatment units were more likely to be encountered by a fire, and treatment units were most frequently burned within one year of the most recent treatment, the latter of which is likely because of ongoing maintenance of existing treatments. Our results highlight the need to identify and prioritize additional opportunities to reduce fuel loading and fire risk on the millions of hectares of federal lands in the CONUS that are in need of restoration.

Keywords: encounter rate; treatment maintenance; treatment longevity; MTBS; LANDFIRE; wildland fire

PACS: J0101

1. Introduction

Interactions between historical fire exclusion, land use changes, and a warming climate have increased fuel loading and fire hazard across millions of hectares of federal forested lands in the United States [1]. Fuel reduction treatments, whereby surface and canopy fuels are removed through mechanical thinning and/or prescribed fire, are a standard management tool to reduce fire risk and restore the vegetative structure of ecosystems that have been degraded by past management and fire suppression [2,3]. Fuel treatments can moderate subsequent fire behavior [4,5], mitigate fire severity [6,7], and increase forest resilience to subsequent disturbances [8,9]. At the stand level, fuel treatment effects vary according to treatment type, size, and age [10], while their spatial arrangement and rate of implementation can affect outcomes at the landscape level [11,12]. One principal critique of fuel treatments is that their benefits are rarely realized because of the low likelihood that an unplanned fire will encounter a previously treated area during its effective lifespan [13–15], though the rate and extent to which this occurs remains largely unknown.

Myriad economic and operational constraints to fuel treatment implementation on federal lands in the United States make it unlikely that treatments alone can achieve forest restoration goals at landscape scales [16]. Recognizing this limitation, several calls have been made to expand the use of unplanned fire to accelerate the pace of forest restoration [17,18]. Managing fire in fire-adapted ecosystems is challenging given the current social and institutional constraints to managing fire for resource benefits [19,20]. However, low-risk opportunities to use unplanned fire to achieve land management goals can be expanded when tied into existing fuel treatment networks or previously burned areas [21]. Treated areas can serve as "anchor points" [2] during incident management to facilitate indirect suppression strategies that allow fires to burn inside large areas buffered by treatments, previously burned areas, or other terrain features that limit fire spread [22] or facilitate suppression efforts [23]. Indeed, leveraging treated areas to support the use of fire is a principal objective of fuel treatment strategies [21], yet little information exists to evaluate its successes or failures.

Recognizing that the successful use of wildland fire is a necessary component of long-term fire risk management, the National Cohesive Wildland Fire Management Strategy identified priority areas where fuel treatments might be used as a precursor to a safer and expanded use of wildland fire [24]. Successfully implementing this aspect of the Cohesive Strategy requires a programmatic and strategic alignment of resources and management objectives from the national to local level; national resources are allocated across agencies and geographical regions where the need to reduce fuel loadings is most critical, and local managers respond by capitalizing on low-risk opportunities to manage unplanned ignitions to achieve long-term fire and land management objectives. Disconnects within this management framework will result in inefficiencies and help to reinforce the current fire and land management paradigm [25]. For example, suppressing wildland fire within a matrix of previously treated areas, especially during moderate weather conditions, forgoes a low-risk opportunity to capture the fuel treatment benefits provided and maintained by wildland fire [22,26]. In turn, managers will be forced to use their limited resources to retreat previously treated areas to maintain low fire hazard rather than expand treatment networks. Quantifying interactions between fuel treatments and subsequent fire at large spatial extents provides managers and policy makers with a means to track their successes and may also reveal where progress towards achieving the goals of the Cohesive Strategy is lacking.

Due to data limitations, previous attempts to characterize fire and fuel treatment interactions in the United States made broad assumptions when estimating the probability that treated areas would burn by unplanned fire [14], most notably the assumption that fire and fuel treatments are randomly located. Findings based on such assumptions may have limited ability to inform contemporary fire and fuels management strategies because the likelihood of fire occurrence and spread is known to exhibit spatial patterning and be highly variable across large landscapes [27,28]. The advent of modern datasets containing spatially referenced fire and fuel treatment data [29,30] enables a more refined assessment of fire and fuel treatment interactions that accounts for fire's natural variability and improves our ability to assess fuel treatment efficacy.

In this study, we used spatially-explicit, standardized datasets of fuel treatments and wildland fires that occurred between 1999 and 2013 on federal lands to summarize the frequency, extent, and geographic variation of recent fire and fuel treatment interactions across the conterminous United States (CONUS). We focused on fire and fuel treatment interactions outside of the wildland–urban interface (WUI), where forest restoration goals are assumed to supersede other potential fuel treatment objectives (i.e., fire-mitigation) [31]. We quantified the percentage of fuel treatments that were encountered by subsequent fire during the study period in terms of ecoregion, treatment size, treatment regime (i.e., number of times treated), and treatment age. Our findings are discussed in the broader context of potential implications for fire and fuel management strategies.

2. Materials and Methods

2.1. Study Area

We evaluated fire and fuel treatment interactions on federal lands in the CONUS. Federal lands were identified from the Protected Areas Database (Version 1.3, United States Geological Survey Gap Analysis Program, USA) [32] (Figure 1). We restricted our analysis to fuel treatments located >2.5 km outside the WUI [3,31] (Figure 2). This distance threshold has been suggested as an appropriate buffer around WUI communities for community wildfire protection zones where fire-mitigation treatments are prioritized [33,34]. The WUI was defined as both the 'interface', where housing is in the vicinity of contiguous vegetation, and the 'intermix', where housing and vegetation intermingle. A spatial data layer of both the interface and intermix was obtained from the SILVIS lab [35] and was developed following federal definitions of the WUI [36]. For clarity, we refer to the WUI and its 2.5 km buffer as $WUI_{2.5}$.

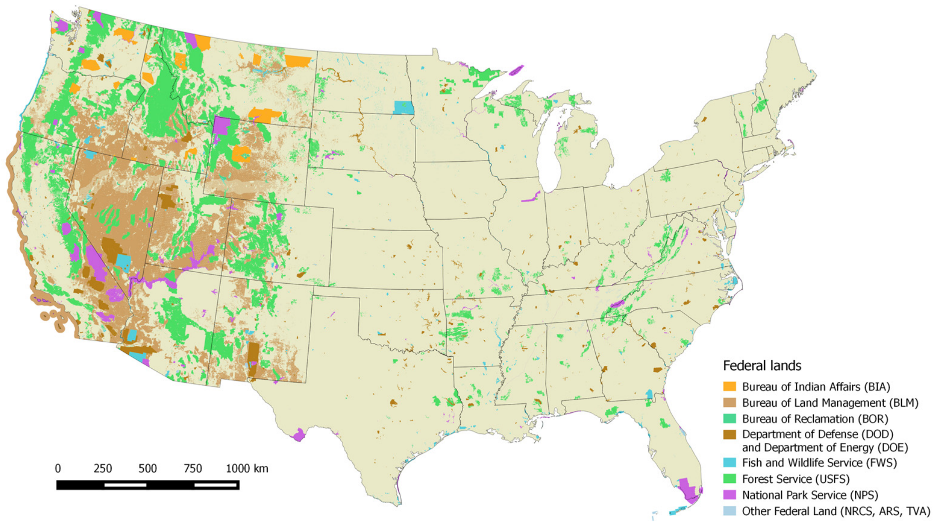

Figure 1. Map of federal lands across the conterminous United States (CONUS).

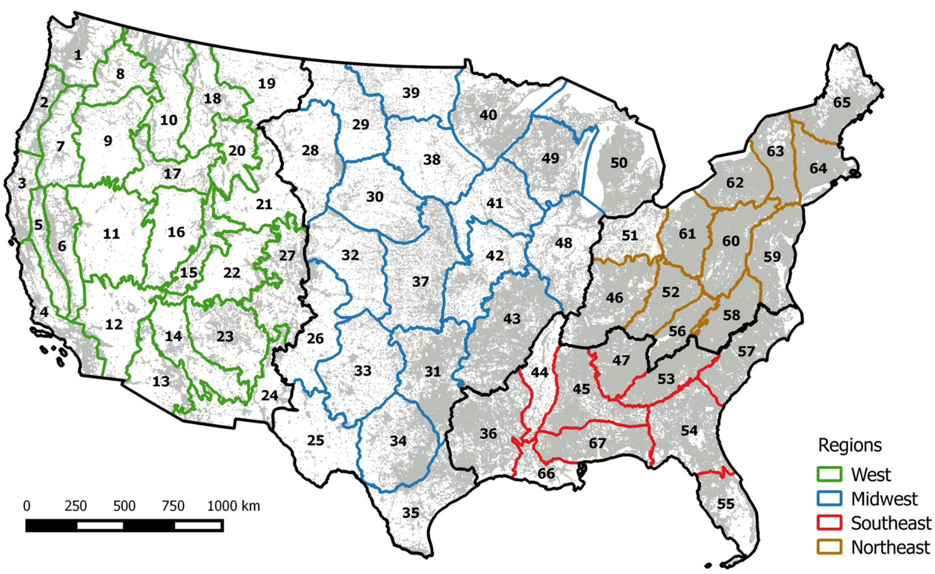

Figure 2. Distribution of wildland–urban interface (WUI) lands including 2.5 km buffer (gray) among regions and ecoregions of the CONUS. See Figure S1 for corresponding ecoregion names.

2.2. Data Background

Our primary datasets were obtained from the LANDFIRE program [29] and the Monitoring Trends in Burn Severity (MTBS) project [30]. The LANDFIRE program produces geospatial datasets (e.g., historical fire regime, existing vegetation type, and recent fuel treatments) to support strategic fire and resource management and planning. The LANDFIRE fuel treatment dataset comprises treatment events that occurred between 1999 and 2012. Each fuel treatment event is a spatial polygon representing a treatment boundary and is attributed by year and type of treatment (Table 1).

Table 1. Description of treatment types from the LANDFIRE Public Events Data Dictionary.

Treatment Type	Description
Clearcut	The cutting of essentially all trees, producing a fully exposed microclimate for the development of a new age class
Harvest	A general term for the cutting, felling, and gathering of forest timber. The term harvest was assigned to events where there was not enough information available to call them one of the two distinct types, clearcut or thinning
Mastication	Means by which vegetation is mechanically "mowed" or "chipped" into small pieces and changed from a vertical to a horizontal arrangement
Other mechanical	Catch all term for a variety of forest and rangeland mechanical activities related to fuels reduction and site preparation including: piling of fuels, chaining, lop and scatter, thinning of fuels, Dixies harrow, etc.
Prescribed fire	Any fire ignited by management actions to meet specific objectives. A written, approved prescribed fire plan must exist, and NEPA requirements (where applicable) must be met prior to ignition.
Thinning	A tree removal practice that reduces tree density and competition between trees in a stand. Thinning concentrates growth on fewer, high-quality trees, provides periodic income, and generally enhances tree vigor

MTBS data are derived from Landsat TM, ETM+, and OLI imagery and include perimeters for fires greater than 200 ha in the eastern US and greater than 405 ha in the western US since 1984. Although these perimeter data are not without error [37], the consistent mapping methodologies and comprehensive coverage reduce potential data bias over time and space relative to other potential data sources; these data have been successfully used to investigate fire frequency, severity, and size over significant geographic and temporal extents [38–40].

Fires labeled by MTBS as 'prescribed' or 'unknown origin' were removed. Prescribed fires from the MTBS dataset that occurred between 1999 and 2012 (n = 4543) were added to the LANDFIRE fuel treatment dataset. Duplicate prescribed fire records between the LANDFIRE and MTBS datasets were subsequently removed.

2.3. Assessing Fuel Treatment Regimes

Many treated areas received several treatments throughout the study period, presumably for treatment maintenance purposes. For example, an area might first be mechanically thinned to reduce vertical and horizontal fuel connectivity, and then treated with prescribed fire the next year to remove residual surface fuels. In such cases of multiple treatments, we identified and delineated all sets of overlapping fuel treatment polygons that constituted a treatment 'unit' and used the most recent treatment type when summarizing interactions between treatments and subsequent fires. In the case where the two most recent treatment types comprised a mechanical treatment (i.e., clearcut, thinning, harvest, mastication, or other mechanical) followed by prescribed fire, we assigned a new treatment type, 'thin-and-burn'. To quantify treatment maintenance and summarize the overall treatment regime for a treatment unit, we recorded the number of original treatment polygons that intersected each treatment unit. Inconsistent digitizing of original treatment boundaries resulted in the creation of many 'sliver' treatment units, so all treatment units less than 415 m^2 were removed (the 1st percentile in the treatment size distribution). A total of 136,107 treatment unit polygons were identified and analyzed.

2.4. Deriving Encounter Rates

All treatment units that occurred on federal land from 1999 to 2012 that were encountered by a subsequent wildland fire between 2000 and 2013 were identified; by definition, treatment units could not be encountered by a fire that occurred in the same year or previous to the treatment. We calculated the encounter rate as the percentage of treatment unit polygons that were intersected by wildland fires and summarized this rate across four variables: ecoregion, treatment size class, treatment regime (i.e., number of times treated), and time-since-treatment. Sixty seven ecoregions were determined from a spatial layer obtained from The Nature Conservancy [41] which is loosely based on Bailey's ecoregion delineation [42].

Calculating encounter rates in terms of treatment age was a two-step process. First, for treatment units encountered by a subsequent fire, we calculated the time-since-treatment as the difference between the years of the fire and treatment. Where multiple treatments occurred within a treatment unit, we used the most recent treatment year before the fire occurred, and when a treatment was encountered by multiple subsequent fires, we used the earliest fire date. Second, we normalized the number of treatments within each time-since-treatment interval to remove the bias introduced by a truncated fire record. For example, only treatment units installed in 1999 were evaluated for the 14 years-since-treatment interval because treatment units installed after 1999 did not have the opportunity to be burned by a fire 14 years later. Conversely, all treatments were evaluated for the one year-since-treatment interval because treatments from each year had the opportunity to be encountered by a fire the next year. We derived encounter rate within each time-since-treatment interval as the number of treatments encountered by a subsequent fire divided by the total number of treatments within each time interval.

3. Results

Our final sample of 3908 unique fire events that occurred between 2000 and 2013 on federal lands in the CONUS burned a total of 18,851,801 ha. Total treated area between 1999 and 2012 was 2,804,850 ha. A total of 9249 of the 136,483 treatment units were encountered by subsequent fire, resulting in an overall encounter rate of 6.8% (Table S1). Of the total treated area, 216,287 ha (7.7%) burned by subsequent fire.

The number of treatments and area treated varied widely among the treatment types (Table 2). Prescribed fire was the most commonly observed fuel treatment fuel treatment type and comprised more area than all other treatment types combined. Thin-and-burn units were more frequent and comprised a larger area compared to clearcut, harvest, or mastication units.

Table 2. Summary statistics for all fuel treatment units. All areal units are in ha.

Treatment Unit Type	Number of Treatment Units	Total Treatment Unit Area	Mean Treatment Unit Size (25th, 75th Percentiles)
Clearcut	2847	29,729	10.44 (1.94, 12.47)
Harvest	7929	92,432	11.66 (1.50, 13.59)
Mastication	2209	38,465	17.41 (0.49, 14.73)
Other mechanical	29,173	473,957	16.25 (0.40, 9.50)
Prescribed fire	47,261	1,631,087	34.51 (0.29, 11.20)
Thin-and-burn	9397	107,311	11.42 (0.72, 12.36)
Thinning	37,667	431,869	11.47 (1.74, 13.13)

Treated area and area burned varied among ecoregions (Figure 3). Treated area was greatest in the Cascade Mountain Range (303,731 ha), Blue Mountain Region of the Columbia Plateau (252,501 ha), and Floridian Coastal Plain (229,163 ha) (Figure 3a). The highest area burned by wildland fires on federal lands occurred in the western United States (Blue Mountain Region of the Columbia Plateau, Snake River Plain, and Northwestern Rocky Mountains ecoregions) (Figure 3b). In the eastern CONUS, area burned was greatest in the Southeastern Coastal Plain ecoregion. Five ecoregions contained zero wildland fires on federal lands during the study period.

Treated area burned tended to exhibit similar spatial patterns to treated area, although some ecoregions of the interior western United States with relatively high treated area had relatively low treated area burned (e.g., Wyoming Basin, Middle Rocky Mountains) (Figure 3c). The encounter rate substantially varied among ecoregions (Figure 3d). The highest encounter rates across the CONUS were observed in the Southern California, Mogollon Rim, and Snake River Plains ecoregions. Encounter rate was less than 5% in 19 of the 25 westernmost ecoregions. During the study period, there were 23 ecoregions with a 0% encounter rate.

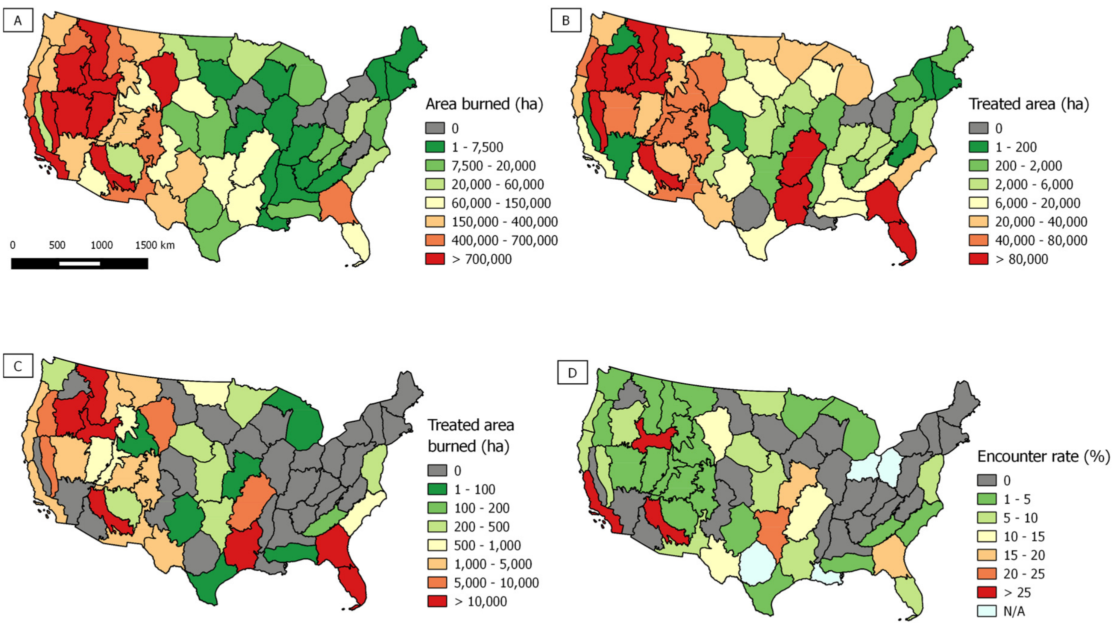

Figure 3. Distribution of (**A**) area burned; (**B**) treated area; (**C**) treated area burned; and (**D**) the encounter rate between fuel treatments and fires on federal lands, summarized for each of 67 ecoregions across the CONUS.

The encounter rate increased with treatment size, especially when treatments were larger than 200 ha (Table 3). However, only 1.4% of all treatment units evaluated were greater than 200 ha. About one-third of all fuel treatment units received at least two treatments during the study period (Table 4). The vast majority of treated area (77.6%) and treated area that was subsequently burned by fire (70.5%), however, was attributable to treatment units that only received one treatment during the study period. Encounter rates between treatments and subsequent fires increased with number of times treated (Table 4).

Encounter rates were highest within one year of the most recent treatment and tended to decline with time since treatment (Figure 4).

Table 3. Summary statistics of frequency, area treated, treated area burned by wildland fire, and encounter rate by treatment unit size class.

Treatment Unit Size Class (ha)	Number of Treatments	Area Treated (ha)	Treated Area Burned (ha)	Encounter Rate (%)
0–5	74,966	99,547	6331	6.8
5–10	21,809	158,899	9718	6.5
10–25	24,156	374,289	21,107	6.2
25–50	8125	281,081	15,543	6.8
50–100	3755	259,466	13,981	7.2
100–200	1753	244,308	11,783	8.1
200–500	1122	352,008	23,844	10.9
500–1000	503	352,731	23,907	15.5
1000–5000	276	498,034	61,382	21.4
>5000	18	184,486	28,690	50.0

Table 4. Summary statistics of frequency, area treated, treated area burned by wildland fire, and encounter rate by treatment regime.

Number of Times Treated	Number of Treatments	Area Treated (ha)	Treated Area Burned (ha)	Encounter Rate (%)
1	85,337	2,178,223	152,405	5.2
2	32,955	461,365	42,889	7.9
3	12,143	126,897	17,985	11.3
4	3992	25,021	2206	13.3
≥5	2056	13,344	802	15.7

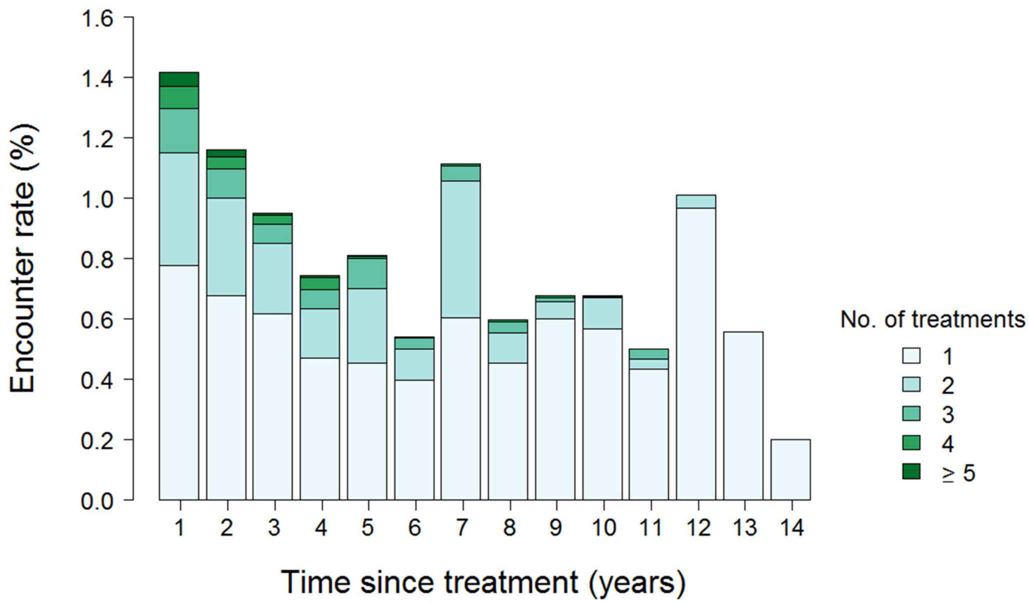

Figure 4. Encounter rate as a function of time since most recent treatment and treatment regime. Number of treatments represents the number of times an area was treated before being encountered by a subsequent fire.

4. Discussion

Characterizing interactions among fuel treatments and wildland fires at broad spatial and temporal scales is an important step to track investments made in fuels reduction programs. Prior efforts have quantified interactions between certain types of fuel treatments and subsequent fire. Rhodes and Baker [14] estimated that between 7.2% and 16.5% of treated areas in ponderosa pine forests of the western United States are encountered by fire within 20 years of treatment assuming random locations of fire and fuel treatments. An empirical study in southeastern Australia found that 22.5% of all prescribed fire patches were subsequently burned by unplanned fire within five years [43]. Our more comprehensive CONUS-wide analysis examined additional fuel treatment types and we observed similar, though somewhat lower encounter rates overall. We found that 6.8% of treatment units created between 1999 and 2012 on federal lands outside of the $WUI_{2.5}$ were encountered by a subsequent fire by 2013.

The Cohesive Strategy identified portions of both the western and southeastern United States as priority areas for active restoration where wildland fire can be more safely used to help achieve long-term land management objectives [24]. In the southeastern United States, treated area was relatively high in four ecoregions (Ouachita Hills, Ozark Highlands, Southeastern Coastal Plain, and Floridian Coastal Plain), and their associated encounter rates were slightly higher than those found in much of the western US (Figure 3). Although western ecoregions contained the highest area burned and treated area during the study period, only six ecoregions experienced encounter rates greater than 5%. Treated area was relatively high across the western CONUS but did not correlate to encounter rates (Spearman's r = 0.12); several western ecoregions had high treated area but a low encounter rate (e.g., Northwestern Rocky Mountains, Cascade Mountain Range, and Blue Mountain Region of the Columbia Plateau). This finding has implications for fuels treatment planning in the western US because simply treating more area may not help to achieve long-term fire and land management goals if wildland fire cannot be safely managed. Strategically placing fuel treatments to create conditions where wildland fire can occur without negative consequences [21] and leveraging low-risk opportunities to manage wildland fire will remain critical factors to successful implementation of the Cohesive Strategy.

Not surprisingly, we found that the encounter rate increased with treatment unit size (Table 4). In addition to being more likely to be encountered, larger fuel treatments can be more effective at moderating fire behavior relative to smaller treatments because they contain more interior area and less edge [7,44,45]. Implementing large fuel reduction treatments in fire-excluded forests on federal lands, however, is challenging due to regulatory and funding constraints [46]. Indeed, our fuel treatment data suggest that 55% of all fuel treatment units on federal lands were less than 5 ha, while only 2.7% of treatment units were greater than 100 ha. These large fuel treatment units (i.e., >100 ha) comprised a significant amount of the total treated area burned; 149,606 ha out of the 216,287 ha (69.2%) of treated area burned occurred within large treatment units. A large portion of this (59,324 ha) occurred inside large treatment units in three ecoregions in southeastern United States where large tracts of federal lands are regularly treated with prescribed fire (i.e., Ouachita Hills, Floridian Coastal Plains, Southeastern Coastal Plain) (Figure S1) [47]. For comparison, 72,447 ha of treated area burned within large treatment units in the ten most treated ecoregions of the western CONUS combined, with over half (37,420 ha) attributable to the Snake River Plain ecoregion alone. Because many of the regulatory, institutional, and social barriers to large scale fuel treatment implementation are likely to remain in place in the near future, alternative solutions to reducing fuel loads across millions of hectares of federal lands, especially in dry forests of the western CONUS, are needed [16].

Fuel treatment longevity is influenced by several factors, including treatment type, vegetation, and fuel decomposition and accumulation rates [10]. Treatment longevity can be extended by applying prescribed or managed fire within the temporal window that fuel treatments remain effective to consume surface fuels and regenerating vegetation that increase fire hazard [48]. In general, treatments have been found to be most effective at moderating fire behavior within the first few years of treatment [49], though in less productive forest types with low fuel accumulation rates, treatments can moderate burn severity for up to 20 years post-treatment [7]. In this study, encounters of fuel treatment units with a subsequent fire occurred most frequently within one year of the most recent treatment (Figure 4). However, nearly half of the treatment units encountered by a fire within one year of treatment had received at least two treatments during our study period. This finding reveals the tradeoff that exists between management of existing treatments to maintain low fire hazard and implementation of additional treatments to reduce fire risk at larger spatial extents [48]. Treatment maintenance is a necessary component of fuel management [2], but maintenance comes at the expense of restoring additional forested lands. One option to extend the longevity of existing treatments is to leverage treated areas during incident management to encourage the use of unplanned fire to maintain and create low fire hazard conditions [17,48]. Wildland fire can rapidly change landscape structure and successional pathways at much larger spatial extents than restoration treatments [18]. Indeed, our data show that the ratio of area burned by wildland fire to treated area exceeds 5:1 for most of the western CONUS ecoregions (Figure S2). The long-term success of fuels management programs depends upon the successful use of fire to achieve land management goals [21], but with only 7.8% of the total treated area in the CONUS burned by a subsequent fire, our results suggest that existing treatments are not being sufficiently exploited to accelerate the pace of forest restoration.

Even though we used the best spatial datasets available to quantify encounters between treatments and subsequent fires, these estimates cannot be used to formally evaluate the success of fuels management at the programmatic level without additional context. Comparing these encounter rates with what might occur under random chance may highlight where in the CONUS they are lower or higher than their expected value. Such an analysis could address whether or not treatments are being strategically placed across large landscapes. Geospatial decision support tools can prioritize treatment locations to establish large, contiguous tracts of land where managed fire can occur without loss of important ecological functions, such as those provided by old growth stands of a fire-resistant species [50]. Implementing such treatment regimens could potentially increase encounter rates and help expedite restoration of forest ecosystems. In addition, risk-based decision support tools are being developed to identify low-risk opportunities for the management of unplanned ignitions [51,52]. Integrating these two approaches could aid local fuel treatment planning efforts by identifying priority areas for active restoration where managed fire can occur without posing an excessive risk to resources, assets, and ecological values.

Although we used the most comprehensive, standardized datasets of fire and fuel treatments available, our analysis was limited by the length and completeness of the data records. While we observed relatively low encounter rates, it's expected they will increase as time goes on, especially if projections of increasing fire activity in North America are accurate [53,54]. Continued efforts to maintain and distribute spatial databases of fire and fuel treatments will aid future investigations of fuel treatment and fire interactions. We focused on treatments and encounter rates occurring outside of the $WUI_{2.5}$ because treatments in these areas are more likely to have had the goal of forest restoration [31]. However, we recognize that these treatments may have included other fire and land management objectives, including WUI protection [31], and may have helped to achieve important land management goals unrelated to forest restoration and independent of being encountered by a wildland fire. Future research can evaluate fire and fuel treatment interactions with respect to treatment objectives when such data become available. MTBS fire perimeters can fail to detect unburned islands and oversimplify complex polygon geometries [55]; these limitations are unlikely to affect the interpretation of our results due to the spatial scale of our analysis and the metrics we summarize. Even though large, recently treated areas can mitigate fire spread [56] and therefore affect future encounter rates, we did not explicitly evaluate fire sizes. This is likely to have a negligible effect on our results because 98.6% of treatments in our dataset were less than 200 ha and the average fire size was 4824 ha. Lastly, the LANDFIRE fuel treatment dataset is by no means a complete record of all treatments implemented on federal lands, and its accuracy is likely to vary among the agencies and groups who contributed their data. Nonetheless, we found it useful in this broad scale analysis as a first approximation of fuel treatment and fire interactions across the CONUS.

5. Conclusions

In this study, we used standardized spatial datasets of fire and fuel treatments to systematically quantify the frequency, extent, and geographic variation of fire and fuel treatment interactions on federal lands across the CONUS. Overall, we found that 6.8% of treatment units between 1999 and 2012 were encountered by a subsequent fire through 2013, with significant geographic variability among ecoregions. Identifying opportunities to jointly reduce fuel loadings on federal lands and safely reintroduce wildland fire will likely remain a priority into the near future. Continued maintenance and distribution of standardized spatial datasets of fire and fuel treatments will allow researchers to monitor interactions among fuel treatments and fires over space and time, hopefully exposing opportunities to improve both fire and fuel treatment planning and management to expedite forest restoration on federal lands.

Supplementary Materials: The following are available online at http://www.mdpi.com/1999-4907/7/10/237/s1, Table S1: Summary statistics of wildland fires, fuel treatments, and their interactions across ecoregions of the CONUS, Figure S1: Distribution of WUI lands including 2.5 km buffer (gray) among regions and ecoregions of the CONUS, Figure S2: Map showing the ratio of area burned to area treated across ecoregions of the CONUS.

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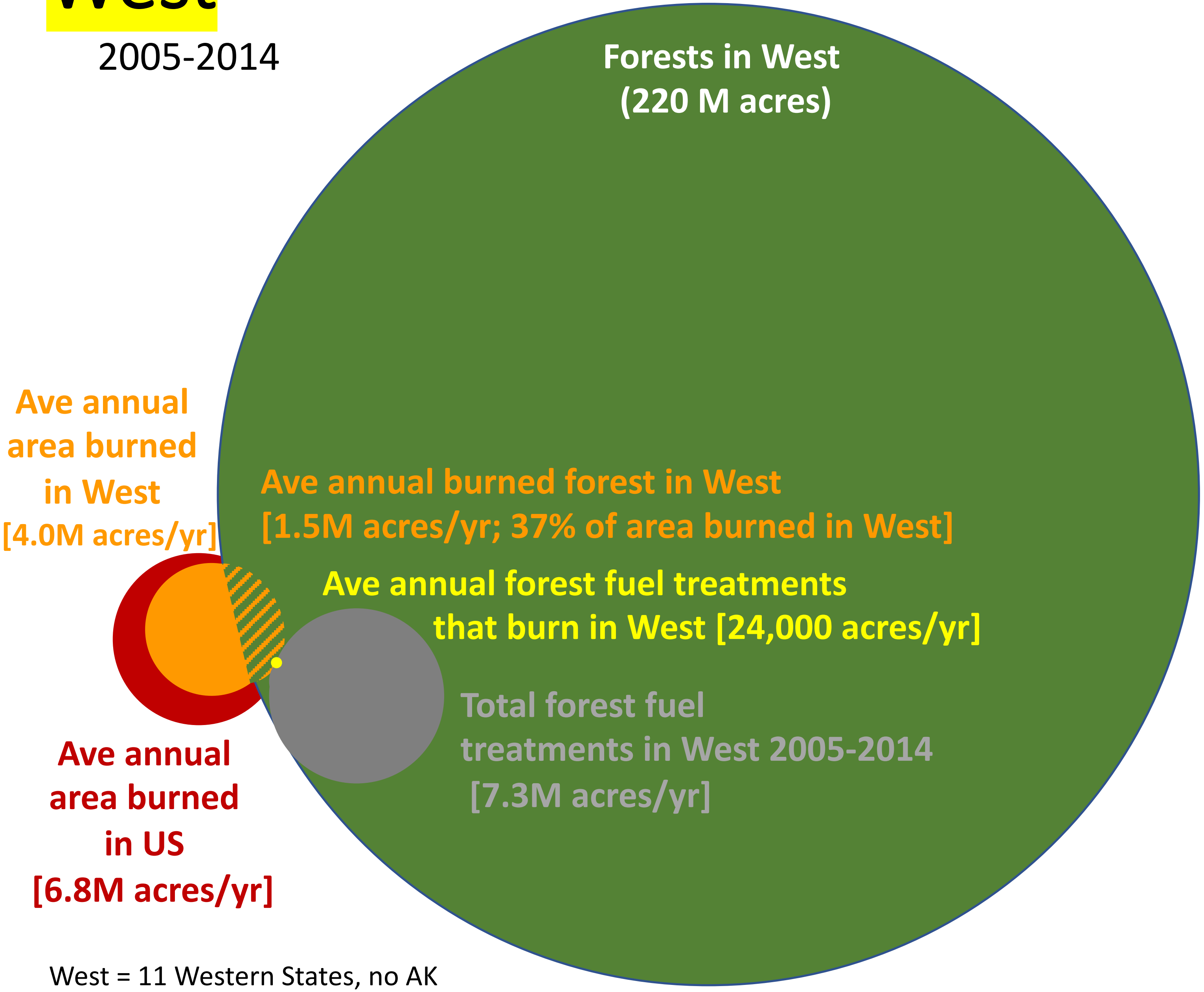
Wildfires RARELY encounter forest fuel treatments in West

2005-2014



West = 11 Western States, no AK

Forests Fuel treatments = thinning and prescribed fire

Treatments only affect wildfire if they burn. <1% treated forest area burns/year.

Only about 1% forests burn/year on average.

Only 40% of area burned in West is forest.

Forest treatments cannot reduce regional area burned due to the scale of flammable ecosystems in the arid West.

We need more strategic approaches to wildfire.

Sources:
-Schoennagel et al. 2017. Proceedings of the National Academy of Sciences. Adapt to wildfire in western North American forests as climate changes.
-Barnett et al. 2016. Forests. Beyond Fuel Treatment Effectiveness: Characterizing Interactions between Fire and Treatments in the US.

Wildfires RARELY encounter forest fuel treatments in West

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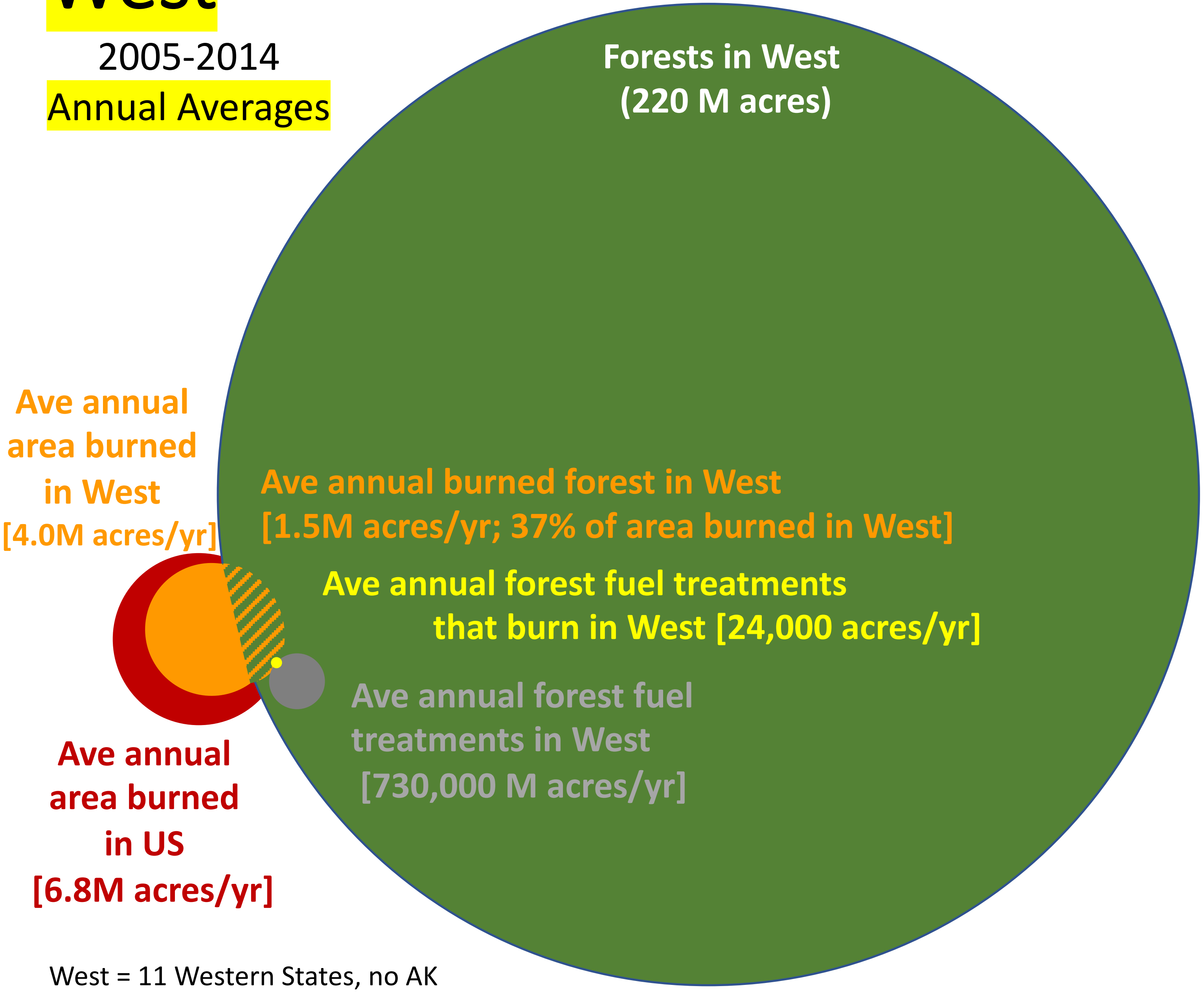
Annual Averages



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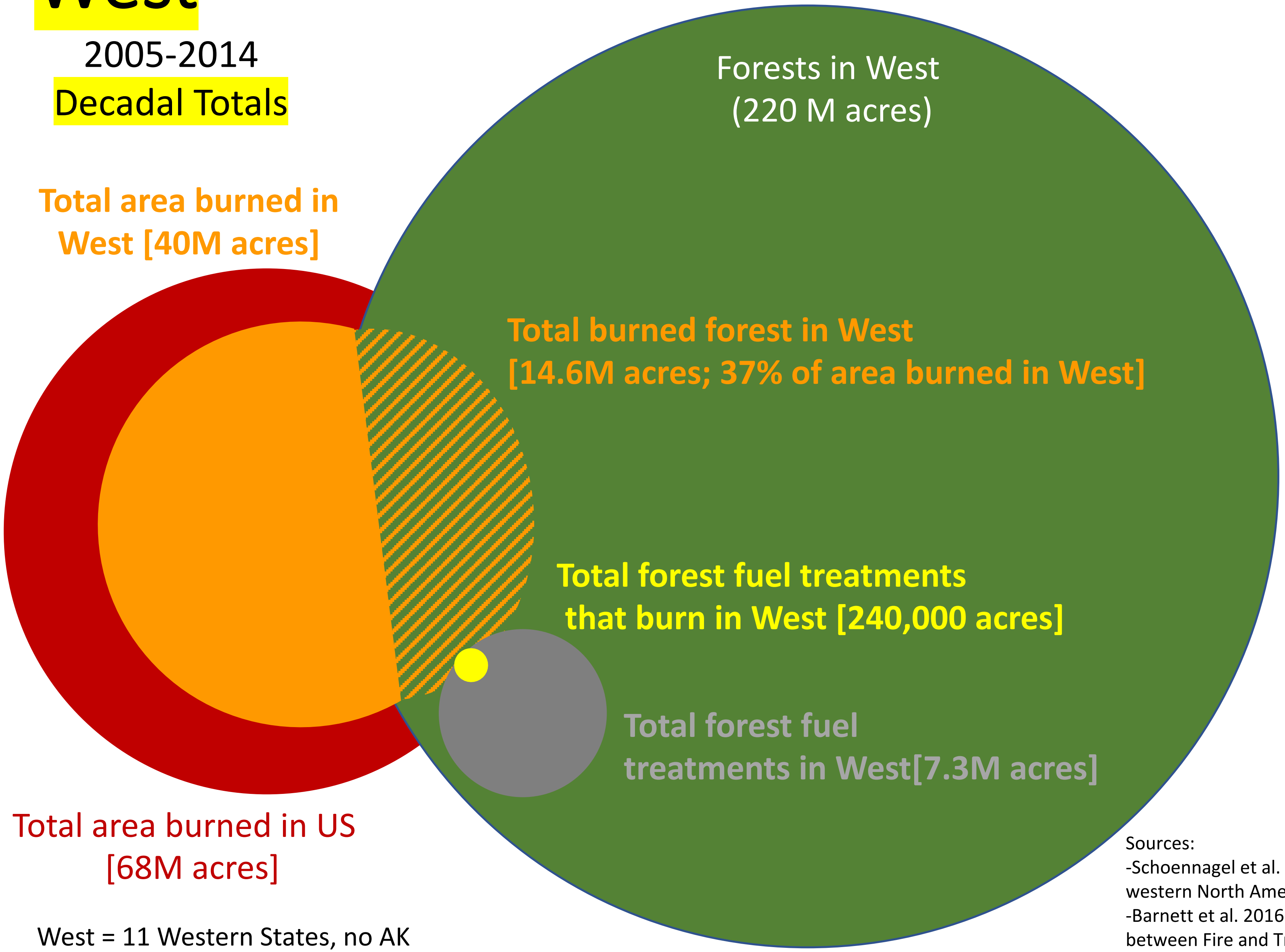
Wildfires RARELY encounter forest fuel treatments in West

2005-2014
Decadal Totals



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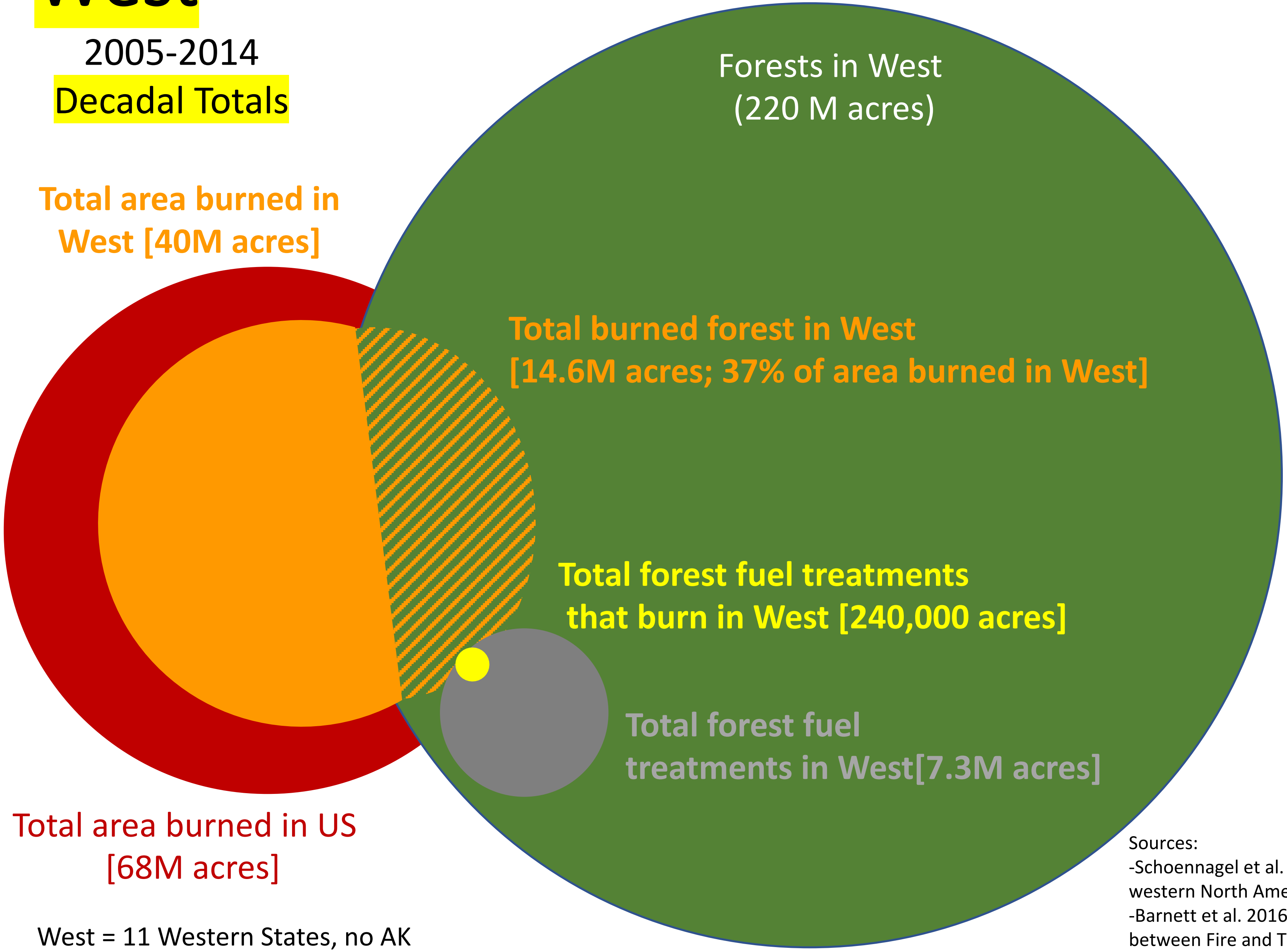

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The majority (~60%) of western wildfires are not in forests.

Schoennagel et al. 2017. Proceedings of the National Academy of Sciences. Adapt to wildfire in western North American forests as climate changes.

Recent Australian wildfires made worse by logging and associated forest management

The recent fires in southern Australia were unprecedented in scale and severity. Much commentary has rightly focused on the role of climate change in exacerbating the risk of fire. Here, we contend that policy makers must recognize that historical and contemporary logging of forests has had profound effects on these fires' severity and frequency.

David B. Lindenmayer, Robert M. Kooyman, Chris Taylor, Michelle Ward and James E. M. Watson

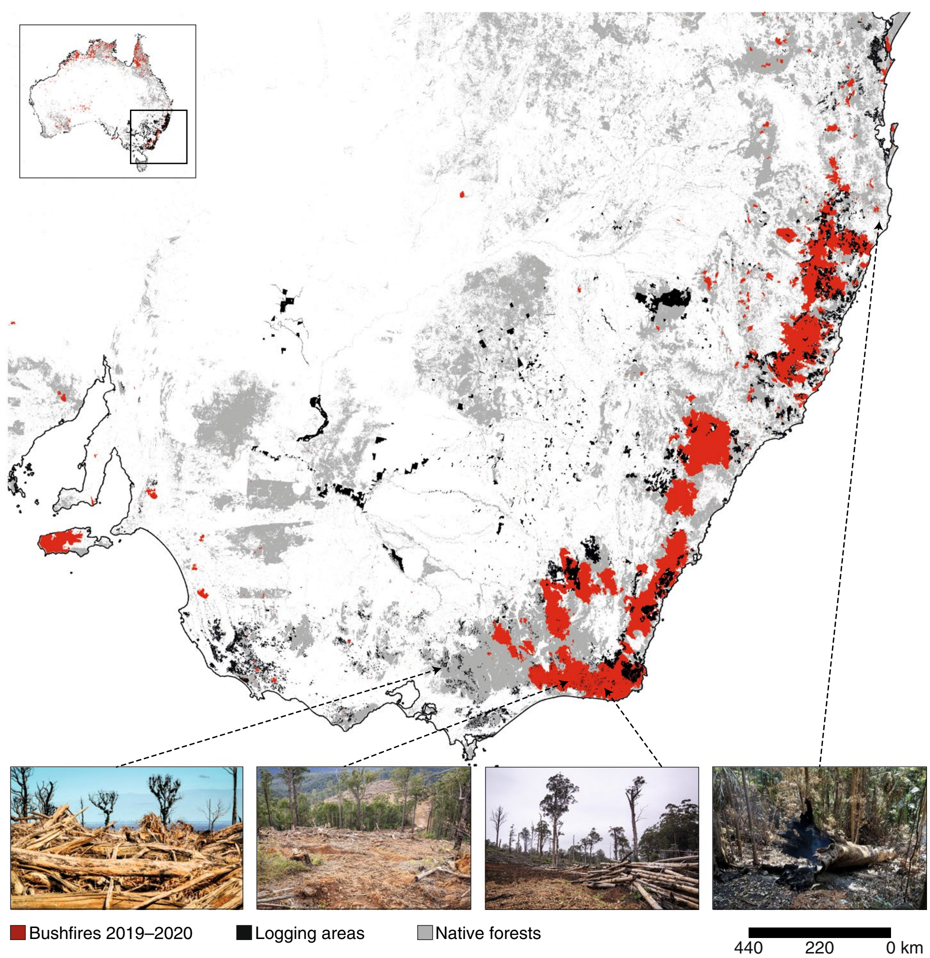

Fig. 1 | Fires within logged areas of native forests. Southeast Australian fires (red) within native forests (grey) and previously logged areas ('logging areas'; black). The first image (left to right) is of the debris remaining after logging in eucalypt forests in central Victoria, the second and third images are of the aftermath of logging in East Gippsland, and the fourth image is of burned Brush Box (*Lophostemon confertus*) within the world heritage Gondwana Rainforest (an ecosystem that has evolved in the complete absence of fire). Logging areas are derived from publicly available data from Forestry Corporation of NSW and VicForests, both of which underestimate the full extent of historic logging. Credit: images 1–3 by C. Taylor; image 4 by R. M. Kooyman

More than 5.8 million hectares of Australia burned between September 2019 and January 2020[1], with several million more hectares burned in subsequent months. Discussions among land managers, politicians, policy makers and scientists have now focused on the origins and behaviour of the wildfires to try to ensure they do not happen again. Not unreasonably, much of this discussion has centred around the role of human-forced climate change[2], and the associated prolonged drought and extreme weather conditions as major drivers of these recent conflagrations. It is clear that discussions about links between climate change and fire are warranted and should galvanize action to halt climate change[3].

However, the contribution of land management, and especially forestry practices, to wildfires has often been neglected in these discussions. This is an oversight given that land management is well within the control of Australians (unlike global action to abate climate change) and that there is an extensive body of science available to decision-makers. Some parts of the forest industry are now calling for increased logging within both the burnt and unburnt forest estates[4]. Here we provide a summary of recent scientific evidence of the impacts of forestry on these fires and discuss strategies to limit future catastrophic conflagrations.

Forest logging and fire

Since European settlement, Australian forests have had a long history of land-use change. While the full extent of forest loss and degradation is unknown, some estimates show that at least 30% of eucalypt open forest and 30% of rainforest have been lost due to logging and agriculture[5]. Most of this loss occurred in the latter half of the nineteenth century. More recently, industry reports show that between 1996 and 2018, 161 million cubic metres of native forest was logged by the forestry industry across Australia[6]. Logging operations have had severe impacts on biodiversity; 181 forest-dependent species listed as threatened with extinction are directly affected by loss of habitat specifically due to logging[7]. However, this figure is an

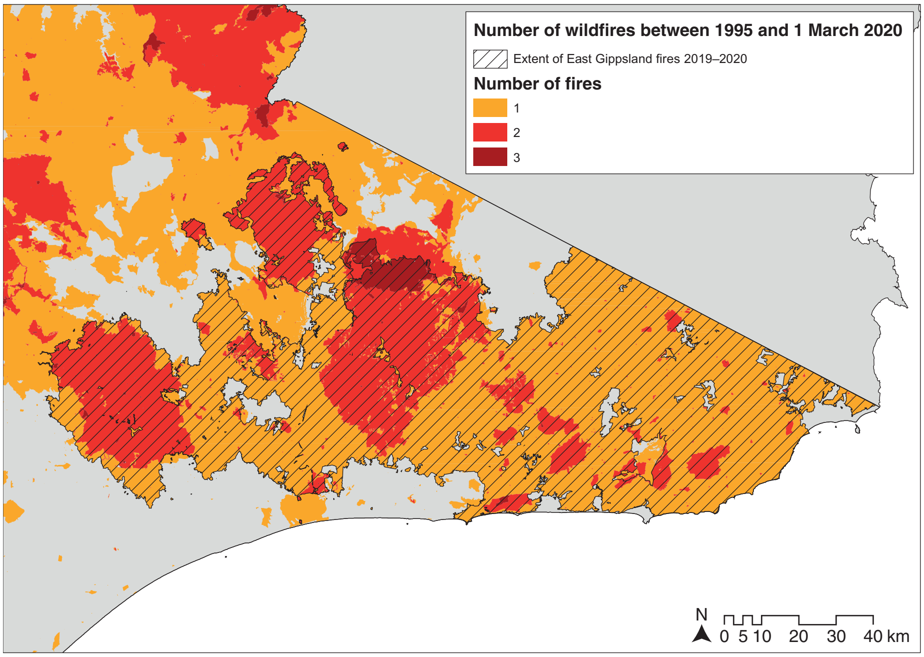

Fig. 2 | Fires within East Gippsland. Analyses of wildfires in East Gippsland, northeastern Victoria between 1995 and 2020 showing that of the ~1 million hectares burnt in the 2019–2020 bushfire season across East Gippsland, ~36% has burnt two or more times since 1995. Credit: map by C. Taylor

underestimate, due to the complexities of listing endangered species in Australia[8]. In addition to the direct impacts of tree felling on species at logging sites, activities associated with production like road construction further fragment already disturbed landscapes — with corresponding negative impacts on biodiversity[9]. For example, in the damp forest ecological vegetation class in the Central Highlands of Victoria, the average distance from logged wood production forests to undisturbed forest is just 71 m relative to 1,700 m in protected areas of the same vegetation type[10]. This difference will be further magnified under plans for continued logging over the coming 5–10 years[10].

Beyond the direct and immediate impacts on biodiversity of disturbance and proximity to disturbed forest, there is compelling evidence that Australia's historical and contemporary logging regimes have made many Australian forests more fire prone and contributed to increased fire severity[11] and flammability[12]. At a site level, logging and other silvicultural treatments leave large amounts of debris (up to 450 tonnes per hectare) (Fig. 1)[13]. This addition of fuel close to ground level increases the severity of subsequent wildfire[11]. Other major logging-generated changes in forest composition and stand architecture, such as the creation of extensive areas of young even-aged stands characterized by densely stocked trees of short stature and a paucity of mesic elements such as tree ferns and rainforest life forms, can influence fire dynamics[11] and patterns of spatial contagion in wildfires[14]. For example, fires spreading from logged areas have burnt into adjacent old growth eucalypts and rainforests dominated by ancient Gondwanan lineages[15]. The former have either never burned since establishment or are subject to extremely rare fires (for example, every 300–500 years), and the latter have never burned, with fire only at the rainforest edges at intervals of ~1,000 years[16].

Extensive areas of logged and regenerated forest have burned repeatedly in the past 25 years (Fig. 2). Of the ~1 million hectares burnt in the 2019–2020 bushfire season across East Gippsland (in northeast Victoria), ~36% had burnt previously at least once since 1995. Current understanding of the ecology of forests such as those dominated by the damp ecological vegetation classes suggests they should burn no more than once every 50–150 years[17]. Repeated fires in these and other ecosystems can lead to tree species failing to resprout[18], seed production and germination failure, and the death of young trees, triggering potential ecosystem collapse[14].

Appropriate land management response post-fire is now needed

It is important that policy makers acknowledge that climate change affects fire weather and is making fires worse across Australia[3]. Policy makers must additionally recognize that land management such as logging operations also has profound effects on fire severity, fire frequency and other key aspects of fire regimes. Efforts to prepare for wildfires therefore require responses not only to climate change but also to historic and current land management.

There are solutions to reduce the risks of further catastrophic fire seasons in the future. First is the removal of logging from areas where it adds considerably to fuel loads and creates forest structures that increase fire severity and risks to human safety. In particular, logging of moist forests must not occur near human settlements. Second, it is essential that landscape-scale impacts of forest fragmentation are reduced; this demands proactive restoration of some previously logged forests to build resilience to future fire events. There is also a need to protect remaining undisturbed or lightly disturbed areas as these are important fire refugia for many species, including arboreal marsupials and birds[19]. In the event of wildfires, land managers must avoid practices such as post-fire ('salvage') logging that can impair recovery and make regenerating forests more prone to further fires[20]. Finally, there is a need to restructure forest industries so that wood production is focused on tree plantations. This is important to maintain employment in the forestry sector and at the same time, limit impacts on the native forest estate, including through a reduction in logging-generated fire proneness in forest ecosystems.

Now is the time for policy makers to recognize and account for the critical values of intact native forests because they are where fire severity is lowest, species persistence during fires is greatest, and rates of recovery after fires are highest[20]. Forests not degraded by logging, together with the biota they support, are more resilient than degraded forests to pre-fire conditions such as higher temperatures and short-term climatic anomalies (for example, droughts)[21]. Intact forests are critical not just in terms of fire resilience, but also in their role in mitigating climate change, maintaining hydrological cycles and other key ecosystem processes, and providing habitat for a wide range of flora and fauna[9]. Australians must therefore work to de-fragment the forest estate through policies that facilitate the expansion of old growth forest, as these actions can help reduce the patterns of extensive spatial contagion of mega-fires. ❐

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Author contributions

D.B.L., R.M.K., C.T., M.W. and J.E.M.W. contributed equally to the preparation of this paper.

Competing interests

The authors declare no competing interests.