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Comparative Assessment of the Impacts of Prescribed Fire Versus Wildfire (CAIF): A Case Study in the Western U.S.

Office of Research and Development
Center for Public Health & Environmental Assessment, Research Triangle Park, NC

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Office of Research and Development
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Research Triangle Park, NC

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ACRONYMNS AND ABBREVIATIONS

Acronym/Abbreviation	Meaning
ABI	Advanced Baseline Imager
AC	air conditioning
AERONET	AErosol RObotic NETwork
AI	aerosol index
AIRACT-4	Air Indicator Report for Public Awareness and Community Tracking
AMI	acute myocardial infarction
AOD	aerosol optical depth
AQI	Air Quality Index
AQMD	Air Quality Management District
AQS	Air Quality System
ARA	Air Resource Advisor
ASDP	AirNow Satellite Data Processor
ASL	above sea level
ASOS	Automated Surface Observing System
avg	average
AWOS	Automated Weather Observing System
B	billions
BC	black carbon
BDA	biological disturbance agent
BenMAP-CE	Benefits Mapping and Analysis Program—Community Edition
BLM	Bureau of Land Management
C+L	Cost plus Loss
CA	California
CAA	Clean Air Act
CAI	climatologically aided interpolation
CAIF	Comparative Assessment of the Impacts of Prescribed Fire Versus Wildfire
CAPS	cavity attenuated phase shift
CARB	California Air Resources Board
CBSA	core-based statistical area
CDC	Centers for Disease Control and Prevention
CFR	Code of Federal Regulations
cfs	cubic feet per second
CH_2O	formaldehyde
CH_4	methane
CHF	congestive heart failure
CI	confidence interval
cm	centimeter(s)
CMAQ	Community Multiscale Air Quality (model)
CO	carbon monoxide
CO_2	carbon dioxide
COPD	chronic obstructive pulmonary disease
C-R	concentration-response (relationship)
CSN	Chemical Speciation Network
CSV	comma-separated value
CV	cross-validation
CVD	cardiovascular disease
D	data available
DBP	disinfectant byproduct
DEM	digital elevation model
DL	distributed lag
DOAS	differential optical absorption spectroscopy
DOI	Department of the Interior
DQF	data quality factor
EC	elemental carbon
ED	emergency department
EDF	Environmental Defense Fund
EDXRF	energy dispersive x-ray fluorescence
EGU	European Geosciences Union
ESA	European Space Agency

ESP	electrostatic precipitator
ET	evapotranspiration
EV	expected value
EVT	existing vegetation type
FCCS	Fuel Characteristic Classification System
FCMAQ	fused CMAQ
FDMS	Filter Dynamic Measurement System
FEM	Federal Equivalent Method
FEMA	Federal Emergency Management Agency
FEPS	Fire Emission Production Simulator
FIA	Forest Inventory and Analysis
FMP	Fire Management Plan
FRG	Fire Regime Group
FRM	Federal Reference Method
FRP	fire radiative power
FS	Forest Service
ft	feet
FY	fiscal year
g	gram
g carbon/m^2	grams of carbon per square meter
GBM	Generalized Boosting Model
GEOS-Chem	Goddard Earth Observing System with a global chemical transport model
GHG	greenhouse gas
GNN	gradient nearest neighbor
GOES	Geostationary Operational Environmental Satellite
GWR	geographically weighted regression
h	hour(s)
H_2O	water
H_2O_2	hydrogen peroxide
ha	hectare
HA	hospital admission
HCN	hydrogen cyanide
HCUP	Healthcare Cost and Utilization Project
HEPA	high-efficiency particulate air (filter)
HF	heart failure
HMS	Hazard Mapping System
HNO_2	nitrous acid
HNO_3	nitric acid
HVAC	heating, ventilation, and air conditioning
HYSPLIT	Hybrid Single Particle Lagrangian Integrated Trajectory
IC	ion chromatography
ICD	International Classification of Disease
IHD	ischemic heart disease
IMPROVE	Interagency Monitoring of Protected Visual Environments
ISI	Influential Scientific Information
IWFAQRP	Interagency Wildland Fire Air Quality Response Program
kg	kilogram(s)
km	kilometer(s)
L/RMP	Land/Resource Management Plan
LED	light-emitting diode
LEMMA	Landscape Ecology, Modeling, Mapping, and Analysis
LF	LANDFIRE Program
LiDAR	Light Detection and Ranging
LLC	Lessons Learned Center
LP	lodgepole
m	meter(s)
M	million
MA	moving average, Massachusetts
MAIAC	Multiangle Implementation of Atmospheric Correction algorithm
max	maximum
MCE	modified combustion efficiency
MCL	lower mixed conifer
MCU	upper mixed conifer
MDA8	maximum daily 8-hour average

MERV	minimum efficiency reporting value
MFI	mean fire intervals
mg	milligram(s)
MI	myocardial infarction
min	minute(s)
$\mu g/m^3$	micrograms per cubic meter
MISR	Multiangle Imaging Spectroradiometer
mL	milliliter(s)
mo	month(s)
MODIS	MODerate resolution Imaging Spectroradiometer
MPLNET	Micro-Pulse LiDAR Network
N_2O	nitrous oxide
NAAQS	National Ambient Air Quality Standards
NAM	North American mesoscale
NASA	National Aeronautics and Space Administration
NCore	National Core (multipollutant monitoring network)
NDIR	nondispersive infrared photometry
NED	national elevation data set
NEI	National Emissions Inventory
NetCDF	Network Common Data Form
NFIRS	National Fire Incident Reporting System
NFPA	National Fire Protection Association
NH_3	ammonia
NICC	National Interagency Coordination Center
NIOSH	National Institute for Occupational Safety and Health
NIST	National Institute of Standards and Technology
NJ	New Jersey
NO	nitric oxide
NO_2	nitrogen dioxide
NOAA	National Oceanic and Atmospheric Administration
NO_X	oxides of nitrogen
NPS	National Park Service
NR	not reported
NV	not available, Nevada
NVC	net value change
NWCG	National Wildfire Coordinating Group
NY	New York
O_3	ozone
OC	organic carbon
OHCA	out-of-hospital cardiac arrest
OMB	Office of Management and Budget
OR	Oregon, odds ratio
ORD	Office of Research and Development
OSHA	Occupational Safety and Health Administration
PAH	polycyclic aromatic hydrocarbon
PAMS	Photochemical Assessment Monitoring Station
PBLH	planetary boundary layer heights
PE	pulmonary embolism
PEV	present expected value
PGN	Pandonia Global Network
PM	particulate matter
PM_{10}	particulate matter with a nominal mean aerodynamic diameter less than or equal to 10 µm.
$PM_{10-2.5}$	particulate matter with a nominal mean aerodynamic diameter greater than 2.5 µm and less than or equal to 10 µm.
$PM_{2.5}$	particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm
$PM_{2.5}$ Tot	monitored $PM_{2.5}$ data
$PM_{2.5}$ TotCMAQ	$PM_{2.5}$ estimated using CMAQ
$PM_{2.5}$ TotCMAQ-M	$PM_{2.5}$ estimated using CMAQ in locations and times with monitoring data

PM_4	particulate matter with an aerodynamic diameter less than or equal to 4 µm (the pollutant size used in the OSHA standards for wildland firefighters)
PP	ponderosa pine
ppb	parts per billion
PRISM	Parameter-elevation Regressions on Independent Slopes Model
PSA	public service announcement
PTSD	post-traumatic stress disorder
PVD	peripheral vascular disease
QA	quality assurance
QA/QC	quality assurance/quality control
QAPP	Quality Assurance Project Plan
r	correlation coefficient
R^2	coefficient of determination
R&D	research and development
RR	relative risk
Rx	prescribed
SABA	short-acting β2 agonists
S.E.	standard error
SD	standard deviation
SERA	Smoke Emissions Reference Application
SFS	Smoke Forecasting System
SHL	significant harm level
SIP	State Implementation Plan
SLAMS	State and Local Air Monitoring Stations
SMOKE	Sparse Matrix Operator Kernel Emissions
SO_2	sulfur dioxide
SOA	secondary organic aerosol
SOP	standard operating procedure
STN	Speciation Trends Network
TC6	Timber Crater 6
TEMPO	Tropospheric Emissions: Monitoring Pollution
TEOM	tapered element oscillating microbalance
TIA	transient ischemic attack
TOR	thermal optical reflectance
TROPOMI	TROPOspheric Monitoring Instrument
U.S.	United States of America
U.S. EPA	U.S. Environmental Protection Agency
UCN	Unified Ceilometer Network
UMBC	University of Maryland, Baltimore County
URI	upper respiratory infection
USDA	U.S. Department of Agriculture
USFS	U.S. Forest Service
USGS	U.S. Geological Survey
UT	Utah
UV	ultraviolet
UV/VIS	ultraviolet-visible
VCAPD	Ventura County Air Pollution Control District
VELMA	Visualizing Ecosystem Land Management Assessments
VIIRS	Visual Infrared Imaging Radiometer Suite
VOC	volatile organic compound
VSL	Value of Statistical Life
WA	Washington
WF	wildland fire
WFEIS	Wildland Fire Emissions Information System
WFLC	Wildland Fire Leadership Council
WONDER	Wide-ranging ONline Data for Epidemiologic Research
WRCC	Western Regional Climate Center
WRF	Weather Research and Forecasting (model)
WRF-Chem	Weather Research and Forecasting Model with Chemistry
WUI	wildland-urban interface
yr	year(s)
ZCTA	ZIP-code tabulation areas

EXECUTIVE SUMMARY

In January 2020, the Wildland Fire Leadership Council (WFLC), an intergovernmental committee formed to support the implementation and coordination of Federal Fire Management Policy and chaired by senior leadership in the U.S. Department of Agriculture (USDA) and Department of the Interior (DOI), requested that the U.S. Environmental Protection Agency (U.S. EPA) lead an assessment to characterize and compare the smoke impacts of wildland fires (i.e., prescribed fire and wildfire) under different fire management strategies, including prescribed fire. In this role, the U.S. EPA, in collaboration with the U.S. Forest Service (USFS), DOI, and the National Institute of Standards and Technology (NIST) conducted an assessment, focusing on the smoke impacts of prescribed fire and wildfire, while also recognizing the direct fire effects (i.e., from the flame itself) of each, as a means to help inform future land management and fire management strategies.

The *Comparative Assessment of the Impacts of Prescribed Fire Versus Wildfire (CAIF): A Case Study in the Western U.S.* consists of two parts that, collectively, provide a qualitative and quantitative assessment of the different effects of wildland fire, with a focus on the air quality and health impacts due to smoke. **Part I: Conceptual Framework, Background, and Context** presents an integrated discussion of topics that are important in comparing the effects of wildland fire, from both smoke and direct fire:

- A conceptual framework and model for evaluating different fire management strategies;
- Background information on different fire regimes, including land management practices and the associated effects (both beneficial and detrimental) of wildland fire;
- A discussion of air quality monitoring as it pertains to prescribed fire and wildfire, including current monitoring capabilities and resources available, to obtain information on air quality measurements and pollutant concentrations;
- A broad overview of the direct fire effects of wildfire with a focus on effects to society (i.e., economic and welfare effects);
- A discussion of the health effects of wildland fire smoke on firefighters and the broader population. This includes a characterization of population-level health effects based on an assessment of the epidemiologic evidence in the U.S. that examined health effects due to wildfire smoke exposures, along with quantitative information on public health measures that could be instituted to reduce individual and population-level exposures to wildfire smoke; and
- Characterization of ecological effects due to wildfire smoke.

Part II: Quantitative Assessment of Smoke Impacts of Wildland Fire in Case Study Areas consists of a quantitative assessment of the air quality and corresponding public health impacts of wildland fire smoke by focusing on two case studies. Part II concludes with an integrated synthesis of the entire assessment with a focus on the results of the case study analyses.

The first case study analysis focuses on a small fire (~3,000 acres), the Timber Crater 6 (TC6) Fire, that occurred in Oregon from July 21−26, 2018. The second case study focuses on a larger fire, the

Rough Fire, which occurred in California from July 31−October 1, 2015 and burned substantially more acres (~150,000 acres) than the TC6 Fire. Both case study fires were selected because they occurred on federal land and were fires managed by USFS and DOI. The TC6 Fire was selected because it had extensive data on land management, fuel treatment, prescribed fire, and wildfire activity. Although the Rough Fire was selected because it represented a larger fire to allow for a scaling up of the modeling approach developed for the TC6 Fire, there was no actual prescribed fire activity close to the fire. Thus, the Rough Fire case study relied on modeled prescribed fire activity that had been conducted by USFS in preparation for prescribed fires that were planned for, but never occurred. For both case studies, hypothetical scenarios assuming different fire management strategies that could have resulted in smaller or larger wildfires were developed based on expert judgment. These hypothetical scenarios allowed for a comparison of the air quality impacts, specifically fine particulate matter (particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm [$PM_{2.5}$]) and ozone, and health impacts due to smoke from the actual case study fires, as well as from prescribed fires in each location using U.S. EPA's Environmental Benefits Mapping and Analysis Program—Community Edition (BenMAP-CE).

The quantitative case study analyses presented herein demonstrate the importance of having refined information on prescribed fire activity to support air quality modeling of wildland fires. Within the area of each case study fire, air quality modeling indicates that the overall air quality impacts of wildland fires are primarily from $PM_{2.5}$. Wildfires, like the TC6 Fire, that occur in more remote locations and are not near large population centers result in relatively small air quality and health impacts compared with larger fires like the Rough Fire. The estimated air quality impacts, as reflected by $PM_{2.5}$ emissions, and the societal economic value of damages from illnesses and deaths due to smoke from each actual fire were:[1]

- TC6 Fire:
    - 1,869 tons of $PM_{2.5}$ emissions
    - $18 million (M; 95% confidence intervals [CI]: $2 M to $47 M)
- Rough Fire:
    - 85,638 tons of $PM_{2.5}$ emissions
    - $3,000 M (95% CI: $260 M to $7,900 M)

The larger size of the Rough Fire and its closer proximity to population centers provided for a more meaningful comparison of the air quality and health impacts from smoke due to different fire management strategies. Initial evidence indicates that a smaller wildfire (i.e., Sheep Complex Fire) adjacent to the Rough Fire that yielded positive resource benefits (i.e., positive ecological benefits) did not substantially reduce the overall fire perimeter of the Rough Fire, and thus minimally reduced the

[1] The difference in economic values between scenarios and case studies reflects the high value placed on reducing the risk of premature death. Even small changes in risk can have economic value because one statistical premature death is valued at $9.5 million.

public health impacts. The addition of a prescribed fire targeted in a specific location to reduce fire spread, in combination with a wildfire that yielded resource benefits, could have dramatically reduced the overall size of the Rough Fire, resulting in an approximate 40% reduction in excess respiratory- and cardiovascular-related emergency department visits and hospital admissions, and premature deaths. The hypothetical scenarios for both case studies demonstrate that prescribed fires targeted for specific locations can help reduce the overall size of a wildfire. Although prescribed fires are timed for days with specific meteorological conditions to reduce population exposures to smoke, analyses show that air quality and public health impacts, while small, are still observed. The estimated air quality impacts, as reflected by $PM_{2.5}$ emissions and the societal economic value of damages of illnesses and deaths attributed to smoke from prescribed fires in each case study were:

- TC6 Fire – Prescribed Fires:
  - 1,071 tons of total $PM_{2.5}$ emissions, ranging from 117 to 565 tons across each prescribed fire
  - $4 M (95% CI: $0 to $9 M)
- Rough Fire – Prescribed Fire:
  - 499 tons of $PM_{2.5}$ emissions
  - $60 M (95% CI: $5 M to $160 M)

It is important to recognize that the confidence intervals surrounding the quantitative estimates of smoke-related health impacts and the corresponding economic values from the case study analyses presented above are reflective of the parametric estimates of uncertainty from epidemiologic studies and the economic valuation literature used by BenMAP-CE. Although not captured within this assessment, additional uncertainties, such as those from the estimation of smoke emissions and the prediction of $PM_{2.5}$ and ozone concentrations through air quality modeling, as well as the timing and location of wildfires themselves, may also contribute additional uncertainty in the estimation of public health impacts. The combination of these different sources of uncertainty may increase or decrease the overall true uncertainty in the quantified impacts depending on correlations between sources of uncertainty.

In addition to estimating air quality and health impacts, preliminary analyses within this assessment demonstrate that campaigns promoting actions and interventions to reduce or mitigate exposure to wildfire smoke can result in public health benefits. These analyses estimate potential reductions in population $PM_{2.5}$ exposures ranging from 14 to 31% depending on the action employed (e.g., use of an air cleaner; running home heating, ventilation, and air conditioning [HVAC] system, etc).

This assessment also details several limitations that should be recognized when interpreting the results of the quantitative analyses presented. Overall, the results are limited to the geographic locations of the case study fires, which have unique land management practices and resulting fire behavior that is specific to the ecosystems of each. In addition, although the results of this assessment demonstrate differences in the air quality and health impacts due to different fire management strategies, this analysis was unable to account for key relationships between prescribed fire and wildfire that should be considered in future analyses because both case studies were retrospective (i.e., based on locations that experienced a wildfire). The analyses also treat prescribed fire activity as occurring at one point in time and does not consider the temporal and spatial patterns of likely fire management strategies that include prescribed fire. Therefore, the analyses do not consider how prescribed fires intersect with wildfire activity, including the probability of a wildfire occurring within the same spatial domain of prescribed fires. Consequently, the comparison of costs and benefits from smoke impacts between prescribed fires and the hypothetical scenarios presented within this assessment is based on case studies where a wildfire occurred and does not account for how the relationship between costs and benefits could differ in instances where wildfires have not yet occurred.

In addition to the limitations surrounding the case study analyses, this assessment also identifies additional limitations in the current scientific understanding of wildland fire smoke. These limitations, which if addressed in the future, could enhance the overall understanding of exposures and health effects to wildland fire smoke, including, but are not limited to, (1) the sparse availability of ground-level air quality monitoring data for wildfire smoke; (2) limited understanding of the health implications of exposures to different durations of wildland fire smoke; (3) limited accounting of prescribed fire activity over space and time; (4) variability in exposure indicators used to represent wildfire smoke exposure across epidemiologic studies; and (5) relative lack of epidemiologic studies specifically examining the health effects of prescribed fire smoke exposure.

Overall, this assessment demonstrates the positive effect that interagency collaborations can have on complex issues at the intersection of land management and environmental public health, such as wildland fire. This initial assessment lays the foundation for future collaborative research and analyses by the partnering agencies to inform future land management and fire management strategies with the goal of reducing the air quality and health impacts due to wildland fire smoke.

CHAPTER 1 INTRODUCTION

1.1 BACKGROUND

The Wildland Fire Leadership Council (WFLC) was established in 2002 by "the Secretaries of Agriculture and the Interior to provide an intergovernmental committee [consisting of federal, state, tribal, county, and municipal government officials] to support the implementation and coordination of Federal Fire Management Policy" (F&R, 2020b). The U.S. Department of Agriculture and the Department of the Interior (DOI) are official members and the cochairs of WFLC. One of the aims of WFLC is to improve communication and coordination with the public, specifically as it pertains to understanding the benefits and tradeoffs of prescribed fire versus wildfire.

At the request of WFLC, in January 2020, the U.S. Environmental Protection Agency (U.S. EPA) agreed to lead an assessment to characterize and compare the smoke impacts[2] of different fire management strategies, including prescribed fire. In this role, U.S. EPA led the development of the *Comparative Assessment of the Impacts of Prescribed Fire Versus Wildfire (CAIF): A Case Study in the Western U.S.* in coordination with the U.S. Forest Service (USFS) and DOI, with contributions from the National Institute of Standards and Technology (NIST). This report would provide a better understanding of the health and environmental impacts of wildland fire (i.e., prescribed fire and wildfire), specifically pertaining to smoke. The interagency approach used to conduct this assessment is critical because USFS and DOI are experts in understanding various aspects of fire (e.g., fire management, fire planning, fire effects and ecology, incident management), NIST is an expert in quantifying the direct and indirect damages due to fire, and U.S. EPA provides expertise in understanding the public health and environmental impacts of fire, especially smoke. This collaborative interdisciplinary effort is essential for characterizing complicated system-level impacts across varying fire management strategies, and to establish the interagency linkages needed to address identified research gaps.

1.2 RATIONALE

Fire, both prescribed and cultural, is used as a land management tool to return nutrients to the soil and remove detritus and excess fuels to reduce wildfire risk (i.e., intensity and severity) and associated effects, and to manage watersheds and the habitats of wildlife, plants, and other organisms. Prior to modern land management, Native Americans were using fire for these same purposes as well as other

[2] Within this assessment, the term "impacts" refers to the main quantitative results, which includes the estimated air pollutant concentrations from the air quality modeling and the number of health events and associated economic values calculated using U.S. EPA's Environmental Benefits Mapping and Analysis Program—Community Edition (BenMAP-CE). The term "effects" is used to denote the other positive and negative consequences of wildland fire.

purposes for millennia (Agee, 1993; Lewis, 1985, 1973). Over time, our relationship with wildland fire, and the smoke that comes from these fires, became more complicated. A confluence of events have all contributed to increasing the likelihood of catastrophic wildfires, including but not limited to, a history of fire suppression that has left a backlog of fuel; a changing climate with warmer temperatures; and humans moving at increasing rates into the line, area, or zone where structures and other human development meet or intermingle with undeveloped wildland or vegetative fuels, referred to as the wildland–urban interface [WUI; F&R (2020a)].

Over the past 30 years, on average approximately five million acres of wildlands in the U.S. have burned annually, with over nine million acres burned in 2020 (Hoover and Hanson, 2021; NIFC, 2018). Although the number of fires has not changed significantly over this period, the size and intensity of the fires have increased due to multiple factors, including higher ambient temperatures, drought, earlier snowmelt due to climate change, the spread of invasive species which increases fuel continuity, and historically high fuel loading [e.g., undergrowth, tree density; Landis et al. (2018)].

Although wildfire can be beneficial, it can also detrimentally affect ecosystems, damage animal and plant habitats, decrease water quality and quantity, and in some instances, create conditions leading to increased overland water flow and flooding. Additionally, with the rapid expansion of the WUI, human development is extending further into fire-prone wildlands resulting in American communities being at increased risk of wildfires and subsequently posing threats to lives, critical infrastructure, and property (Lewis et al., 2018). The direct effects of fire itself are compounded by the equally significant effects of the smoke generated from fires, which can travel transcontinental distances and have substantial adverse effects on public health (U.S. EPA, 2019b). Because the risk that wildfire poses to property and health has increased, especially when a wildfire is severe and catastrophic, the need to address this growing risk has also increased. At the same time, there is a need to recognize that fire has always been part of natural landscape processes and there is a need to maintain its many ecological benefits, such as fuel reduction.

Various fire management strategies have been used over time with the overall goal of reducing the potential for negative effects of wildfire, such as reducing the overall size of a wildfire and the direct effects of the fire itself. These actions, which include prescribed fire and pile burns from thinning activities, have associated risks, specifically degradation in air quality and subsequent health and environmental effects. Prescribed fire is perceived as lower risk compared with wildfire because the timing and area to be burned can be managed to limit smoke impacts (i.e., dispersed both spatially and temporally). Prescribed fires are conducted when meteorological conditions are favorable, smoke production (fuel consumption) is less, atmospheric conditions support adequate smoke dispersion, and wind patterns allow smoke to move away from sensitive areas (e.g., populated areas, hospitals, schools, roadways). Although there is a small chance that land managers can unintentionally lose control of a prescribed fire, prescribed fire is considered low risk. However, there is a risk continuum for wildfire that can change daily based on fire behavior resulting in a dynamic set of management actions. As a result, wildfire management can shift between full suppression efforts and, if conditions allow (e.g., wet fuel, anticipated precipitation), management that may achieve resource benefits. To date, there is limited information that allows for a direct, systematic, and comprehensive comparison of the air quality and associated health impacts of smoke from prescribed fire and wildfire. To ensure the effective use of prescribed fires to reduce the risk of catastrophic wildfire, decision makers need information on the air quality impacts associated with fire management strategies that include prescribed fires compared with strategies that do not.

Numerous research activities have focused on examining the nexus between fire, smoke, and ecological and health impacts. These activities have focused on this complex issue by examining how various conditions (e.g., fuel type, temperature, moisture) influence the subsequent emissions from a fire, how these emissions move over various geographic scales and topographies, the toxicological and ecotoxicological effects from smoke exposure, and both wildland firefighter and population-level health impacts of smoke exposure. Recent studies have also evaluated actions and interventions that can be instituted to reduce the public health impacts during smoke episodes by melding together social science, behavioral science, and health risk communication. While all these activities have led to significant advancements in the science, the overall air quality impacts of different fire management strategies, which consist of different land management practices, including prescribed fire and wildfire, are not well characterized. Thus, these uncertainties complicate the decision-making process in determining the appropriate fire management strategy and land management action to implement at governmental levels ranging from local to federal.

1.3 ANALYSIS APPROACH

The CAIF Report represents a unique opportunity to bring together experts spanning multiple disciplines related to fire science (e.g., air quality, monitoring, modeling, health effects, ecological effects) to conduct an integrated interagency assessment. This report focuses on a novel modeling approach to estimate the air quality impacts, specifically of fine particulate matter (i.e., particulate matter with a nominal mean aerodynamic diameter ≤ 2.5 µm [$PM_{2.5}$]) and ozone, in response to different fire management strategies, and the associated health and economic impacts. To conduct such an analysis, this report will focus on two case study fires, both of which occurred in the western U.S.: (1) Timber Crater 6 (TC6) Fire that occurred from July 21−26, 2018 in Oregon and (2) Rough Fire that occurred from July 31−October 1, 2015 in California. These fires were selected, in part, because they represented interagency fires managed by both USFS and DOI and because both had data available, to a varying degree, on previous land management practices. Because of the difference in the scale of these two fires—the TC6 Fire burned approximately 3,000 acres and the Rough Fire burned approximately 150,000 acres—and the different land management and fire management strategies employed in both locations, there will be slight differences in the resolution of the analyses and the analytical approaches between the fires.

The modeling component of the analysis, which is the main focus of this report, will estimate $PM_{2.5}$ and ozone concentrations for the actual fire and compare those air quality impacts with hypothetical scenarios based on different fire management strategies resulting in smaller or larger fires for each of the case studies. In addition to the hypothetical smaller and larger fires, analyses also examine prescribed fire activity. In the case of the Rough Fire, the perimeter included the footprint of a recent wildfire that burned at lower intensity and yielded positive resource benefits. Resource benefits refers to the positive ecological effects realized from managing a fire. For both case studies, the prescribed fire analyses do not account for the periodic nature of prescribed fires that are conducted over years to decades to keep fuel loads at a level needed for fire suppression opportunities. For the TC6 Fire case study, as a result multiple prescribed fires that occurred over many years in the vicinity of the TC6 Fire were modeled as individual events within the same month when prescribed fire activity was known to have occurred. This contrasts with the Rough Fire case study where the focus was on the modeling of a single prescribed fire event that was planned but did not occur. Prescribed fire activity was treated this way within this assessment for numerous reasons, including current limitations in the ability to account for the timing of retrospective prescribed fire activity and sparseness of available data. Further, the prescribed fires examined within these case studies are not intended to account for the entirety of a spatial area needed to prevent the spread of a larger wildfire in both areas.

For both the TC6 and Rough Fire analyses, to facilitate comparison of impacts across the different fires being examined within the case study areas, the region-wide air quality impacts (i.e., $PM_{2.5}$ and ozone) were compared to a baseline of ambient air pollution with no case study area fire. This approach allows for an estimation of the burden associated with each of the case study fires and a direct comparison of the health impacts and associated economic values, across each fire and hypothetical scenario using U.S. EPA's Environmental Benefits Mapping and Analysis Program—Community Edition [BenMAP-CE; U.S. EPA (2019a)].

The TC6 Fire was selected for the first case study because of the extensive land management data available and the overall small size of the fire, which allowed for the modeling framework used throughout this assessment to be developed and refined. A map of the area around the actual TC6 Fire is depicted in Figure 1-1, with its fire perimeter denoted by the solid red line. For the TC6 Fire case study the hypothetical scenarios developed consist of:

- Scenario 1 (small): defined as the green hatched area inside the TC6 Fire perimeter in Figure 1-1, which is a smaller hypothetical TC6 Fire in a heavily managed area (e.g., most prescribed fire activity), which would equate to a wildfire with less fuel consumption, a smaller fire perimeter, and less daily emissions;
- Scenario 2a (large): defined as the blue dashed line and hatched area outside the TC6 Fire perimeter in Figure 1-1, which is a larger hypothetical TC6 Fire, but not the "worst-case" scenario, due to no land management which would equate to a wildfire with more fuel consumption, a larger fire perimeter, and more daily emissions; and
- Scenario 2b (largest): defined as the brown dashed line and hatched area outside the Scenario 2a fire perimeter in Figure 1-1, which is a much larger, hypothetical "worst-case" modeled scenario

TC6 Fire with no land management (i.e., no prescribed fire) which would equate to a wildfire with the most fuel consumption, largest fire perimeter, and largest daily emissions.

In addition to each of these scenarios, analyses will include an examination of only the prescribed fires that occurred around the TC6 Fire, for a comparison of air quality and health impacts between prescribed fires and the actual wildfire. These prescribed fires were selected based on actual historical prescribed fire activity in this area as a preliminary comparison point to the TC6 Fire and hypothetical scenarios.

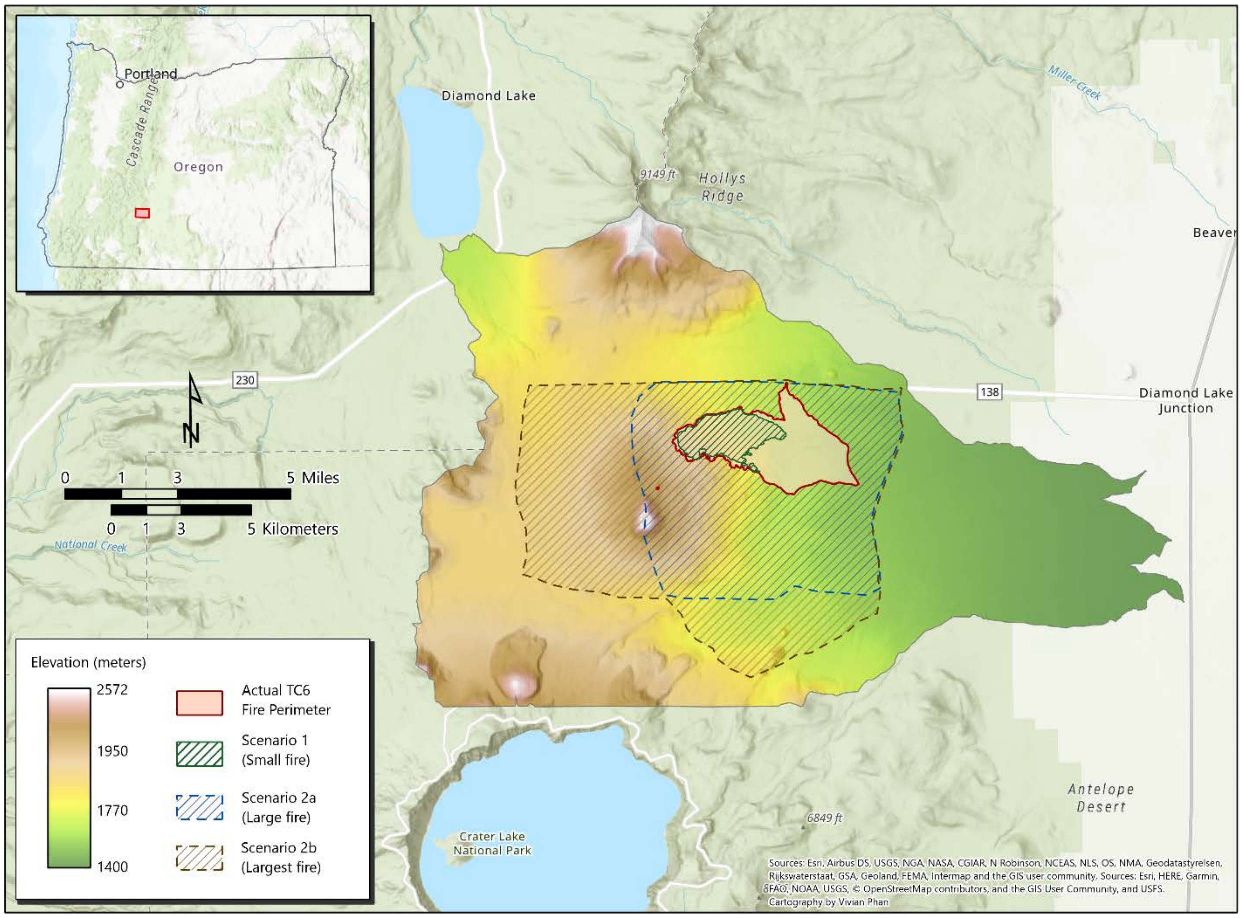

Figure 1-1 Map of fire perimeters of hypothetical scenarios and actual fire for the Timber Crater 6 (TC6) Fire case study.

The Rough Fire was selected for the second case study because its larger size and location provide an opportunity to assess impacts on a larger downwind population and evaluate differences in both air quality and health impacts for different hypothetical fire management strategies versus the TC6 Fire. A map of the area around the Rough Fire area is presented in Figure 1-2, and its fire perimeter is denoted by the solid red line. For the Rough Fire, analyses will encompass the actual fire, which occurred over approximately 2 months, and the impact of multiple hypothetical scenarios, representing different land management practices, on both the spread of the Rough Fire and corresponding air quality impacts. In comparing air quality impacts between the actual Rough Fire and the hypothetical scenarios, air quality impacts are modeled for the entire 2 months of the actual Rough Fire. The model diverges at the point where the Rough Fire would have reached the perimeters of two fires considered within this case study, the Boulder Creek Prescribed Fire (i.e., the red shaded and hatched area, Figure 1-2) and the Sheep Complex Fire (i.e., the blue line and shaded area, Figure 1-2). Within the Rough Fire area there was no previous prescribed fire activity; as a result, this case study models the proposed Boulder Creek Prescribed Fire, which was a prescribed fire that USFS had planned but did not carry out and the Sheep Complex Fire, which is a wildfire that occurred in 2010 due to a lightning strike and as a result of relatively wet fuel conditions resulted in resource benefits. For the Rough Fire case study (USFS, 2021), the hypothetical scenarios developed consist of:

- Scenario 1 (small): defined as the red shaded and outlined area above the black dashed line in Figure 1-2, which examines the combined impact of the Boulder Creek Prescribed Fire and the Sheep Complex Fire on reducing the spread and air quality impacts of the Rough Fire; and
- Scenario 2 (large): defined as the entire red perimeter of the Rough Fire and the blue area of the Sheep Complex Fire in Figure 1-2, which allows for the fire perimeter of the Rough Fire to progress into the area of the Sheep Complex Fire as if both the Boulder Creek Prescribed Fire and Sheep Complex Fire did not occur.

In addition to comparing each hypothetical scenario to the actual Rough Fire, air quality and health impacts will also be compared individually to the Boulder Creek Prescribed Fire and the Sheep Complex Fire.

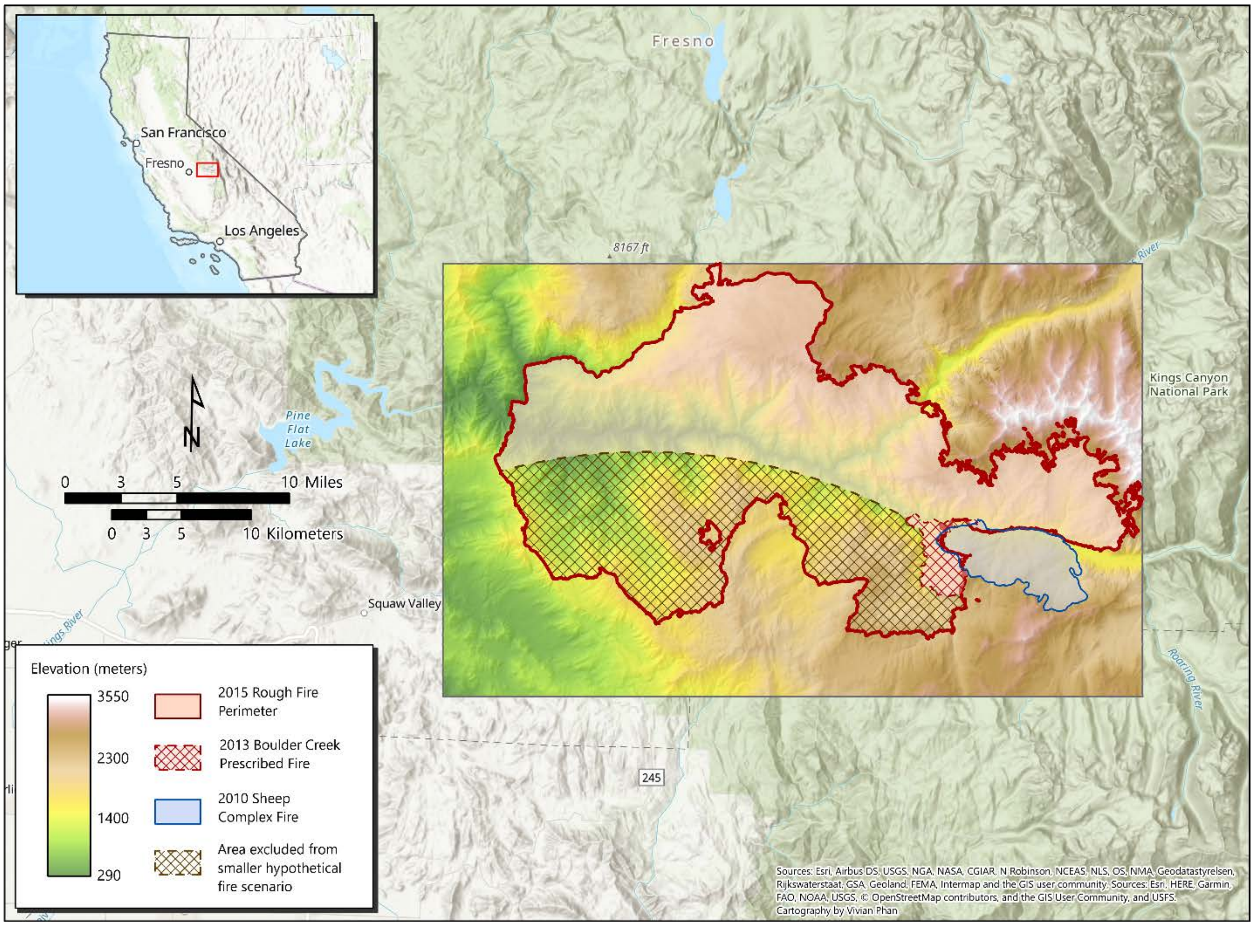

Figure 1-2 Map of fire perimeters for the Rough Fire case study.

While the direct comparison of the air quality impacts of different fire management strategies can inform the benefits and tradeoffs of each, it is also important to recognize that specific actions or interventions could also be taken to minimize public health impacts. However, the likelihood of individuals taking precautionary measures to reduce smoke exposure can vary between wildfire and prescribed fire events depending on the presence and effectiveness of public health messaging as well as the amount of lead time available for messaging to give the public time to act on that messaging. As a result, when evaluating the tradeoffs between wildfire and prescribed fire it is important to also consider the potential public health implications of actions or interventions that could be employed to reduce population exposure to smoke. Therefore, an illustrative example is provided to estimate the potential public health benefits that could be realized in each case study analysis for different actions meant to reduce or mitigate smoke exposure. For the actual TC6 and Rough fires, the USFS deployed Air Resource Advisors (ARAs), who in combination with the respective state and local air quality agencies allowed efforts to be taken to predict smoke impacts and to warn the public of the hazards of smoke and the benefit of minimizing exposure. The examination of smoke exposure reduction actions within this assessment does not reflect a formal analysis of post-fire effectiveness of public health messaging for either the TC6 or Rough fires.

The comparison of air quality impacts and associated health and economic impacts between the different fire management strategies represents the main output of the CAIF Report; however, in order to put the results in the proper context, the report also captures qualitatively, and in some cases quantitatively, other factors that can influence a full accounting of the benefits and damages associated with each fire management strategy. This includes information pertaining to baseline forest conditions, air quality monitoring of fires, direct fire effects on health, damages due to fire and smoke, and ecosystem benefits and damages.

1.4 GOALS OF THIS REPORT

The goal of the CAIF Report is to provide an initial quantitative assessment of the air quality and associated health, and economic impacts attributed to different fire management strategies, including prescribed fire, through an extensive modeling exercise. This quantitative assessment will be supplemented with qualitative discussions to highlight the current state of the science that informs this assessment, and identify deficiencies that if addressed, can further inform analyses of fire management strategies. The collective assessment within this report of the benefits and damages associated with both fire and smoke can contribute to a fuller characterization of the benefits and tradeoffs of different fire management strategies.

This report represents an initial step in the process of conducting assessments to characterize the impacts of different fire management strategies to inform both public health actions to reduce population exposures to wildfire smoke, and future land management decisions. By attempting to more fully account for the impacts of different fire management strategies, tradeoffs can be assessed to ensure the appropriate land management actions are taken to maintain forest health and minimize the public health impacts attributed to wildland fire smoke.

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PART I: CONCEPTUAL FRAMEWORK, BACKGROUND, AND CONTEXT

CHAPTER 2 CONCEPTUAL FRAMEWORK FOR EVALUATING AND COMPARING DIFFERENT FIRE MANAGEMENT STRATEGIES

2.1 INTRODUCTION

Fire is an important element of the natural landscape and is highly influenced by both natural and anthropogenic factors. Fire management decisions are made at multiple governance levels to influence the types of fires that affect different vegetative and human systems. Goals include increasing overall forest and rangeland health and resilience and reducing the potential for the occurrence of uncontrolled and sometimes catastrophic wildfire. Current federal fire policy recognizes the importance of wildland fire (i.e., prescribed fire and wildfire) "as an essential ecological process and natural change agent that will be incorporated into the (land management) planning process [which includes the development of] Fire Management Plans (FMPs), programs, and activities support[ing] Land and Resource Management Plans and their implementation" (Interagency Federal Wildland Fire Policy Review Working Group, 2001). Different fire management strategies before, during, and after a fire can result in different effects on the landscape and adjacent communities, including the smoke that results from wildland fire. Understanding the effects of different fire management strategies, defined as a planned set of activities to achieve resource objectives, can help land and fire managers make informed decisions that reduce adverse effects, both directly from the fire itself as well as from the smoke it produces, while yielding desired ecological and risk management benefits. In this chapter, we describe a conceptual framework for evaluating and comparing different fire management strategies, using a range of metrics to characterize and quantify effects.[3] Fire management strategies are developed to achieve multiple objectives, including promotion of ecological benefits, protection of lives and property, safe and effective responses that minimize risks to firefighting personnel, and reduction in likelihood of severe and catastrophic wildfire. While the focus of this assessment is on the quantification of the air quality and associated health impacts attributed to smoke exposure, it is also important to recognize the broader effects (both positive and negative) of wildland fire when considering different fire management strategies. Therefore, subsequent chapters provide more in-depth discussions of the elements of this framework and its implementation in comparing the effects of different fire management strategies.

[3] Within this assessment, the term "impacts" refers to the main quantitative results, which includes the estimated air pollutant concentrations from the air quality modeling and the number of health events and associated economic values calculated using U.S. Environmental Protection Agency's (U.S. EPA's) Environmental Benefits Mapping and Analysis Program—Community Edition (BenMAP-CE). In this case, all of these impacts are negative, but a broader range of both positive and negative impacts would be included in a more comprehensive assessment. The term "effects" is used to denote the other positive and negative consequences of wildland fire.

The overarching question that guides the evaluation conducted within this framework is “What are the expected effects (both positive and negative) of alternative fire management strategies over both short- (during the event) and long-term (post-event) time horizons?” with an emphasis within this assessment on the smoke impacts. Critical to this question are the ideas of expected positive and negative effects, a recognition that fire needs to be viewed over a management-relevant temporal and spatial frame, and that fire is inevitable and necessary. While some effects can be quantified and monetized broadly (i.e., nationally), and thus used in a more traditional cost-benefit comparison, such analyses can be challenging when examining the effects of individual fires. Many of the effects of wildland fire are not easily quantified or assigned a dollar value. As a result, while this assessment estimates the air quality and the dollar value of health impacts of smoke (one of the larger negative effects of fire) for quantitative comparisons, it also provides additional qualitative discussions of other effects (i.e., positive and negative) of both direct fire and smoke (see Table 2-1 and the appendix).

2.2 EXPECTED VALUE FRAMEWORK

An expected value (EV) framework is used as the conceptual basis for this assessment because of the inherent stochastic nature of fire in the landscape. While in many cases, a wildfire is likely to occur given a sufficient time horizon, both the timing and location of a wildfire event is uncertain compared with prescribed fires which are planned events that occur at specific times of the year and in specific locations. Wildfires can also reburn the same area with very different outcomes because of the changes in fuel characteristics (e.g., types, loads, arrangement, continuity, etc.) while prescribed fire can be used to reduce fuels to maintain low fuel conditions. For example, many of the prescribed fires in the southeastern U.S. are maintenance burns designed to keep fuel loads low and occur on a fairly frequent basis. A range of periodicity between fires has been established for different ecosystems; however, under a changing climate, the previous assumptions on potential risk of wildfires are often challenged. The management of wildland fire can result in a desired outcome (positive effect) or an undesirable outcome (negative effect) or both. Fire management strategies such as prescribed fires can reduce the uncertainty in outcomes from wildfires. When comparing strategies, both stochastic and nonstochastic elements need to be expressed in a way that allows for equivalent comparison. In a typical cost-benefit framework, comparisons between alternatives require a complete accounting for all costs and benefits, both direct and indirect. The conceptual framework used in this assessment aims to provide a full accounting of the overall effects of wildland fire; however, the ability to quantify all elements is limited, and the specific quantitative case studies that are the focus of the assessment are limited to the air quality and health impacts associated with smoke. As such, while this chapter describes the full range of elements of the conceptual model, there is greater emphasis on the elements that will form the basis of and be incorporated into the main component of this assessment: the quantitative comparison of the smoke impacts of wildland fire. Key details of the inputs in this comparative analysis are the air quality modeling of smoke directly from the case study fires and health impact analyses described in Chapter 7 and

Chapter 8, respectively. This focus on smoke impacts is to address a key gap in the overall knowledge base regarding wildland fire management; however, this emphasis is not intended to suggest that the other positive and negative effects of wildland fires and fire management strategies are less important. A full accounting of costs and benefits of those strategies will require further development of models and methods to quantify effects across the full range of domains, including ecological, health, safety, prevention, and risk to highly valued resources and assets.

The expected value of a specific fire management strategy requires knowledge of (1) the effects associated with different fire types (e.g., prescribed fire vs. wildfire), (2) the effects associated with different management techniques (e.g., targeted thinning, prescribed fires), and (3) probabilities of these effects. Two other key concepts are fire ignition probabilities and the management of a wildfire once it has ignited. Ignition probabilities, a key factor in determining risk from wildfires, indicate the chance that a wildfire will occur over a specified time period within a defined spatial domain (Hunter and Robles, 2020). In managing wildfire risk, land managers use an operational risk framework that gives primary consideration to public and firefighter safety. This risk framework is intended to consider the degree to which the extent, intensity (energy output), and severity (effects on ecosystems) of a wildfire can be mitigated once started based on the land management plans; fire history; fuel condition, amount, and configuration; and weather conditions. Both ignition probability and the characteristics of a wildfire can be positively or negatively affected by the fire management strategy.

Within this report, costs of management strategies are defined as the specific economic expenditures associated with implementing specific management actions. For example, the costs associated with a management strategy that includes mechanical thinning would include, but not be limited to, the costs of equipment and labor costs for equipment operators. Costs here do not refer to the outcomes of management actions, but instead these outcomes are referred to as effects, which can be either positive or negative (see Table 2-1). One consequence of a fire management strategy may be reductions in future costs of fire management (Sánchez et al., 2019).

For this conceptual framework, the expected value (EV_i) of effects (positive + negative) for a fire management strategy M_i is specified as:

$$EV_i = \sum_{t=0}^{T} \{PF_{it}|M_i + NF_{it}|M_i + P_t(WF\ ignition|M_i) \times (WF_{it}|M_i)\}$$

Equation 2-1

$$PEV_i = \sum_{t=0}^{T} \frac{\{PF_{it}|M_i + NF_{it}|M_i + P_t(WF\ ignition|M_i) \times (WF_{it}|M_i)\}}{(1+r)^t}$$

Equation 2-2

Where PF_{it} are prescribed fire-related effects in Year t conditional on M_i, NF_{it} are nonfire effects from M_i in Year t, P_t(WF ignition|M_i) is the probability of wildfire ignition in Year t conditional on M_i, and $WF_{it}|M_i$ are wildfire-related effects in Year t conditional on M_i and land management objectives once a wildfire is ignited. T is the time horizon for decision making. T is not a fixed value, but may depend on natural fire cycles, land management decision horizons, or other factors. For the purposes of comparing strategies where air quality and health impacts are expected to occur over multiple years of a management strategy initiated in Year 0, it is appropriate to discount the value of the future impacts. Equation 2-2 provides the present expected value (PEV_i) taking into account discounting of impacts in future years as noted by *r* which is the discounting rate. Effects include all the positive and negative effects associated with a fire management action.[4] These include both the nonfire effects (e.g., ecosystem effects from thinning operations, effects from fire managers accessing remote sites for prescribed fires, etc.) and fire effects (related to burning, smoke, ash, and post-fire damages to water quality). In most applications, *EV* will be expressed in dollars, a unit in which all damages can be theoretically expressed, for comparison with the dollar costs of the management strategy. Essentially, the present expected value is the sum over time of all the effects of the fire management action itself plus the ignition-probability-weighted effects of wildfire, conditional on the management strategy. For fire management strategies that do not include prescribed fire, the first term will be zero.[5]

The net present expected value (*NPEV*) is the *PEV* minus the discounted sum over time of the costs of a management strategy:

$$NPEV_i = PEV_i - \sum_{t=0}^{T} \frac{C_{it}}{(1+r)^t}$$

Equation 2-3

Within this assessment, fire management costs are treated as a known quantity. There is likely to be uncertainty in those fire management costs as well; however, addressing this uncertainty is beyond the scope of the assessment.

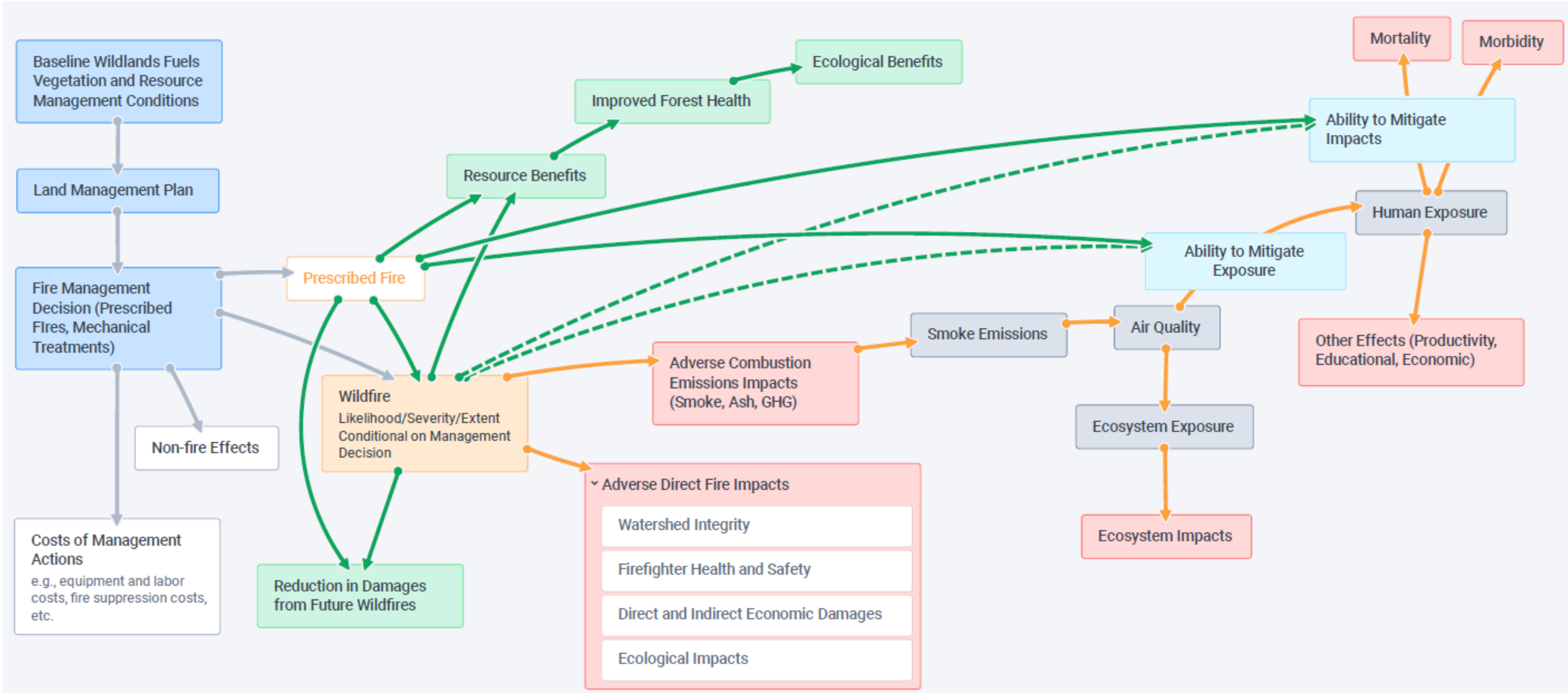
2.3 COMPONENTS OF THE CONCEPTUAL FRAMEWORK

A graphical representation of the overall conceptual framework is presented in Figure 2-1, and a simplified graphic showing just the elements covered in the quantitative case study is presented in Figure 2-2. These figures are meant to serve as an anchor for discussions of elements of the framework. Figure 2-1 presents the key components without formally representing the expected value framing, although it recognizes the conditional nature of wildfire likelihood (probability of ignition), severity, and extent. As discussed above, in a formal expected net present value framework, the probability-weighted value of air quality and health impacts of smoke would need to be calculated each year within a specified

[4] Fire management strategies do not reduce the risks of wildfire to zero, thus effects include the direct consequences of the management action as well as the probability-weighted effects of wildfire.

[5] There may be some nonsmoke or fire-related benefits and damages associated with other fire management approaches such as mechanical thinning. In addition, while there are air pollution emissions impacts from heavy equipment used in fire management, the quantitative case studies in this assessment are focused on the air quality and health impacts of smoke from wildland fires and do not include these additional air quality impacts.

fire management planning horizon and the discounted sum of those impacts would be compared with costs. This simplified conceptual diagram does not attempt to convey the temporal dimensions of the relationship between management decisions and occurrence of wildfires, although it does recognize that one benefit of both prescribed fires and wildfires is the reduction in damages from future wildfires. The following discussions of each element provide a short description and references to the chapters and sections of this report, which provide more detailed qualitative discussions, and where possible, quantification methods and modeling results.



GHG = greenhouse gas.

Note: Forest management inputs are colored dark blue, management decisions and their non-fire-related effects are colored white, resource benefits are colored green, mitigation actions are colored light blue, fires are colored orange, fire damages are colored red, and smoke exposure-related elements are colored gray. The green arrows indicate positive effects, and the orange arrows indicate negative effects.

Figure 2-1 Conceptual framework for evaluating and comparing fire management strategies.

Note: This figure focuses on the elements that are most relevant for the case studies and does not provide detailed information about the wide range of adverse direct fire impacts or actions that can mitigate those impacts.

Figure 2-2 Key conceptual elements of the quantitative case studies.

2.3.1 BASELINE WILDLAND FUELS VEGETATION AND RESOURCE MANAGEMENT CONDITIONS

Baseline vegetation conditions, which are discussed in detail in Chapter 3, influence the probability of a wildfire occurring and the intensity and characteristics of a wildfire, including smoke generation. These wildland fuels vegetation conditions include location, size, density, stand composition, ladder fuels,[6] height to live crown, understory condition, and surface fuel loads. Other vegetation and resource management attributes included in land management plans (see Chapter 3) or that influence the management and outcomes of a fire include distance from the wildland to populated areas (e.g., location in or relative to the wildland–urban interface [WUI]); proximity to Superfund sites, mining sites, and other contaminated sites; distance to watersheds that provide community drinking water; plant and wildlife habitats; infrastructure; and consideration of positive effects from fire (e.g., restoring ecosystems, fuels reduction).

2.3.2 TYPES OF FIRES

There are two types of wildland fire, as designated in statute 40 CFR § 50.1—Definitions (U.S. EPA, 2020a), and by policy, as stated in National Wildfire Coordinating Group (NWCG) Glossary of Wildland Fire (NWCG, 2021). The following two definitions will be used throughout this assessment to remain consistent with their use in air quality regulation and in federal wildland fire management policy.

- Prescribed fire: Also referred to as planned fires, controlled burns, or prescribed burns, 40 CFR § 50.1(m) defines a prescribed fire as “any fire intentionally ignited by management actions in accordance with applicable laws, policies, and regulations to meet specific land or resource management objectives” (U.S. EPA, 2020b).
- Wildfire (natural and human caused): 40 CFR § 50.1(n) defines a wildfire as “…any fire started by an unplanned ignition caused by lightning; volcanoes; other acts of nature; unauthorized activity; or accidental, human-caused actions, or a prescribed fire that has developed into a wildfire. A wildfire that predominantly occurs on wildland is a natural event” (U.S. EPA, 2020c).

Effects are expected to vary based on characteristics such as types of fuels, burn conditions (e.g., temperature, humidity, wind), season, duration, intensity, and location relative to populated areas (which can vary from minute to minute, day to day, and site to site) within each area burned. Fires also vary based on the history of previous fire occurrences, the periodicity and intensity of previous occurrences, and the management and land use history of the area in question. For the purposes of this conceptual framework, the focus is on these two different types of fires (i.e., prescribed fire and wildfire), recognizing that within each category, there will be a high degree of variability based on these characteristics.

[6] Fuel that allows fires in low-growing vegetation to jump to taller vegetation.

Prescribed fires can be declared a wildfire if they are no longer meeting objectives (e.g., escaping boundaries, intensity, smoke management), but this rarely occurs. A 2013 report from the Wildland Fire Lessons Learned Center (LLC) reported that in 2012, only 0.08% of prescribed fires escaped their planned boundaries (LLC, 2013). This includes all escapes on federal, state, tribal, and private lands that were reported in the Wildland Fire LLC Incident Review Database, along with additional agency notifications and media reports that were available.

Wildfires vary widely in their effects depending on location, meteorological conditions during the fire, and the types of forests where they occur. A wildfire may also be deemed "catastrophic" (Wooten) when it results in severe economic, social, and ecological effects (Carey and Schumann, 2003), including a high percentage of dead trees (Wooten). There is a great deal of year-to-year variability in the number of acres burned across the U.S. In recent decades, wildfires have affected an increasing number of acres. Total acreage burned increased from an average of 6.9 million acres burned from 2000−2019 to an average of 3.2 million acres burned from 1980−1999 (NICC, 2019).

On February 13, 2009, the Guidance for Implementation of Federal Wildland Fire Management Policy was issued (FEC, 2009). This guidance allows for consistent implementation of the 1995 Federal Fire Policy and the 2001 update. By policy, management response to a wildfire on federal land is based on objectives established in an applicable Land/Resource Management Plan (L/RMP) and or FMP. Fire management objectives are affected by changes in fuels, weather, topography, varying social and political understanding and involvement of other governmental jurisdictions that may have different missions and objectives. Managers use a decision-support process to guide and document wildfire management decisions. The process includes land management objectives, situational awareness, analysis of hazards and risk, defining of implementation actions, and the fire management decision documentation and rationale.

A full range of fire management strategies can be used to achieve L/RMP and FMP objectives. Wildfire may be managed solely to meet protection objectives, such as protecting valued assets at risk of loss by suppressing the fire in the safest, most effective, and efficient way. The initial response may be as simple as evaluating the location of the fire without further on-the-ground active suppression action in areas where the fire is distant from valued assets that require action to protect or where the risks from exposure for firefighters is higher than the value of the assets that would be protected. Wildfire may be managed concurrently for one or more objectives, and the objectives can change as the fire spreads across the landscape. For example, a wildfire can be managed for suppression to protect points of valued resources while at the same time taking no action when or where resource values are being enhanced.

No matter how a wildfire is being managed, firefighter and public safety is the priority. All fire management activities and decisions must reflect this commitment. A fuller description of how wildfire can be used as a land management tool can be found in Chapter 3.

2.3.3 FIRE MANAGEMENT STRATEGIES

Severity of fires is determined by several factors, some of which can be affected by management practices (e.g., forest structure, fuels, vegetation composition) and other factors that cannot be controlled (e.g., weather, location). Most fire management strategies focus on fuel load reduction, which is a management strategy that involves "manipulation, including combustion, or removal of fuels to reduce the likelihood of ignition and/or to lessen potential damage and resistance to control" (USFS, 2003a). Fuel-reduction strategies aim to reduce the probability of ignition and reduce the intensity and uncontrolled spread of wildfires (Agee and Skinner, 2005). Thus, fuel-reduction strategies directly affect two key parameters in the framework, i.e., the probability of wildfire ignition (P_t(WF ignition|M_i)) and wildfire-related effects ($WF_{it}|M_i$). Two common practices for fuel load reduction include prescribed fires and mechanical treatments.

2.3.3.1 PRESCRIBED FIRES

Prescribed fires, as defined in Section 2.3.2, are a fire management tool that uses planned, controlled fires to reduce fuel loads and achieve the social, economic, and ecological benefits of fires while reducing the potential for catastrophic uncontrolled fires. There are decades of evidence that prescribed fires can reduce surface fuels and fire severity while maintaining or improving forest health (Hunter and Robles, 2020; Kalies and Kent, 2016; USFS, 2003b).

Prescriptions for fire are based on clearly defined objectives, which might include ecological aspects, such as habitat diversity and endangered species recovery, reductions in the risk to highly valued resources and assets including communities, as well as fuel reduction to reduce the potential of high intensity, high severity fires. Prescriptions also take into account environmental and meteorological conditions, fuels, burn area, and planned approaches for suppression once objectives have been met to reduce potential adverse effects, including effects associated with smoke emissions (USFS, 2021; U.S. EPA, 2020d). The effectiveness of prescribed fires in reducing the potential for severe wildfires is dependent on weather patterns and ecosystem characteristics such as types of fuels, as well as the interactions between them [e.g., drought may affect fuel moisture content; Fernandes and Botelho (2003)] and treatment prescription and implementation.

On federal and most state lands, prescribed fire is only used after thorough preplanning (e.g., by creating land management plans, environmental assessments, burn plans, etc.) and coordination with partners. Such planning is only done by highly trained and experienced professionals. Prescribed fires are only implemented when the resource benefit as outlined in the burn plan is met and adequate contingencies are in place or confirmed by managers and agency administrators. Go/no-go checklists are used to determine compliance with policies and the prescribed fire plan parameters (NWCG, 2017).

2.3.3.2 MECHANICAL FUEL REDUCTION

Mechanical treatments to thin trees and remove fuels can be used in conjunction with prescribed fires or be employed in places and times when prescribed fires cannot be used (McIver et al., 2013). Mechanical treatments require equipment, as well as plans for disposal or subsequent use of significant quantities of small trees (Agee and Skinner, 2005; Rummer et al., 2003). Thinning trees can reduce surface fuel loads and also reduce risks of crown fires (fires that spread across tree canopies), which can cause severe damage. There are multiple types of thinning that affect different aspects of forest composition, including low thinning that removes small trees, crown thinning that removes medium size trees, and selection thinning that removes larger, more marketable trees (Agee and Skinner, 2005). How the residual wood from the thinning operations is disposed of can have a substantial impact on surface fuel availability, with pile burning of the unusable tops of trees having the greatest impact on reducing fuel loads. Mastication is another mechanical treatment (with or without thinning) that changes the fuel profile.

There are limited observational data on the degree to which mechanical thinning, alone or in conjunction with prescribed fires changes the probability of ignition. Simulations have shown that removing small trees and ladder fuels can be effective in reducing fire severity, especially when removal is done in conjunction with prescribed fires (Agee and Skinner, 2005).

2.3.3.3 FUEL TREATMENT EFFECTIVENESS

In 2006, the U.S. Department of Agriculture Forest Service and Department of the Interior (DOI) Bureau of Land Management (BLM) initiated a program to evaluate the effectiveness of hazardous fuel treatments (prescribed fire and mechanical) designed to reduce the potential of high intensity, high severity wildfires. When a fuel treatment is tested by wildfire, an evaluation is performed to determine the effectiveness of the treatment in changing the fire behavior (e.g., going from a crown fire to a surface fire) and/or helping manage the wildfire. In 2011, the Forest Service and the DOI land management agencies (Bureau of Indian Affairs, BLM, Fish and Wildlife Service, and National Park Service) made the effectiveness assessment mandatory whenever a wildfire impacted a previously treated area.

Since 2006, almost 14,860 assessments have been completed (IFTDSS, 2021). About 89% of the fuel treatments were effective in changing fire behavior or helping with management of the wildfire or both (IFTDSS, 2021; Hunter and Robles, 2020). In addition, prescribed fire treatments were observed to be the most effective in changing fire behavior and reducing overstory mortality from wildfires. Unfortunately, until recently, the ability to detect all wildfire fuel treatment interactions has been limited due to constraints in reporting systems. This has resulted in a significant under-sampling of fuel treatment effectiveness monitoring, mostly on the smaller fires (less than 1,000 acres), which is critical because the majority of wildfires do not reach this size.

2.3.4 EFFECTS OF WILDLAND FIRES

Prescribed fires and wildfires can have both positive and negative effects, although the magnitude of potential effects differs. The goal of prescribed fires is to reduce the fuel loads which will lower the frequency, intensity, and severity of a wildfire, while providing for safe and effective response to wildfire and protecting highly valued resources and assets. In general, positive effects that occur directly from fire are improvements in landscape/watershed health which yield ecological benefits or ecosystem services. Similar to prescribed fires, wildfires can also reduce fuel loads and result in decreases in frequency, intensity, and severity of subsequent wildfires. Negative effects occur both directly, as a result of the fire itself, or indirectly, through emissions of smoke and ash. The magnitude, scale, and duration of these effects will depend highly on the type of fire, the fuel conditions, the terrain, and the fire weather conditions, as well as the location of the fire relative to the WUI and downwind/downstream populations.

Air quality impacts result from smoke emissions that affect ambient concentrations of numerous pollutants, including ozone and particulate matter, specifically fine particulate matter (particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm [$PM_{2.5}$]; see Chapter 4 and Chapter 7), which has been shown to contribute to a wide variety of adverse health and ecological impacts [see Chapter 6; Holm et al. (2021); Jaffe et al. (2020); Cascio (2018)]. The severe wildfires occurring in the western U.S. over the past few years have caused loss of life and property, and their emissions have reversed trends in air quality improvements in the western states (McClure and Jaffe, 2018). These facts have drawn the attention of the National Academies of Science, Medicine and Engineering (NASEM, 2020) and other medical professional organizations (Kaufman et al., 2020; Rajagopalan et al., 2020; Rice et al., In Press), which are strongly invested in finding solutions to prevent such severe wildfires, while simultaneously mitigating the adverse effect of exposure to smoke.

2.3.4.1 DIRECT FIRE EFFECTS

2.3.4.1.1 BENEFITS TO WILDLAND ECOSYSTEMS

Many wildland ecosystems have adapted to periodic fires. In fact, several conifer tree species such as some pines depend on fire for reproduction, maintaining very low surface fuels, and reducing ladder fuels that cause crown fires, as do many shrubs and most grasses. Other species, such as sequoias, rely on periodic fires to open forest canopies to allow saplings to grow and flourish. Open canopies also support the growth of shade-intolerant plants and reduce the probability of crown fire. Fires also convert brush and dead trees and plants to nutrient-rich ash, which can be beneficial to established trees and provide essential nutrients for new forest growth. These nutrients are also important to support soil microbes, which increase the overall health of wildland ecosystems. Fires and smoke can also remove invasive species not adapted to fires, as well as reduce populations of destructive insects and diseases

(Neary et al., 2005; Brown and Smith, 2000; Smith, 2000). Detailed information on the benefits of wildland fire on wildland ecosystems is provided in Chapter 3.

2.3.4.1.2 BENEFITS TO FIRE MANAGEMENT (POST-EVENT ABILITY TO MITIGATE RISKS OF FUTURE IMPACTS)

As discussed in Section 2.3.3.1 and Section 2.3.3.3, prescribed fires and strategic and safe management of wildfires are designed to reduce the potential for severe fire damages by changing the behavior of a subsequent wildfire and making it easier to manage, or to meet social and ecological objectives. This can result in fewer risks to firefighting personnel during subsequent wildfires, protecting life and property in and around communities, as well as reducing economic damages, ecological damages, and health impacts to populations from fires and poor air quality caused by smoke.

2.3.4.1.3 FIRE DAMAGES

Direct fire damages, described in Chapter 5, include effects to populations in the vicinity of fires, economic damages, and ecological damages. Effects to populations in the vicinity of fires include deaths, injuries, and psychological damages (Thomas et al., 2017). Economic damages include the value of lost property; loss of marketable timber; direct and indirect costs of evacuations, including business interruption; damages to infrastructure, such as downed power lines or damaged roadways; and the value of lost recreational resources due to either safety-related closures or fire damage (Thomas et al., 2017). Ecological damages can occur from changes in vegetation composition; conversion from one vegetation type (e.g., forest) to another (e.g., shrubs); damage to soils, which could lead to flooding and degraded water quality and quantity; loss of habitat and endangered or threatened species; increased susceptibility to insects and diseases; and climate-related damages resulting from releases of greenhouse gases (GHGs) and loss of carbon sequestration potential (Thomas et al., 2017).

2.3.4.2 EFFECTS FROM SMOKE AND ASH

All wildland fires produce smoke and ash. The amount and composition of smoke can vary between the types of fires due to the types of burn conditions and type, loading, and consumption of fuels. Release height and transport of smoke can also vary between types of fires (as well as within types of fires) depending on meteorological conditions and burn conditions. For example, plume rise will depend on the temperature of the fire, and long-range transport of smoke will depend on wind speed and direction, as well as plume rise. The impacts associated with smoke emissions will depend on the emissions density, how far and in which direction the smoke travels, other wildfires and their emission production rates, and on the proximity of fires to downwind populated areas. Chapter 4 and Chapter 5 describe approaches used to monitor and model air quality impacts from wildland fire smoke.

2.3.4.2.1 SMOKE-RELATED EFFECTS

Smoke has immediate impacts directly next to a fire, as well as impacts downwind of a fire because of worsened air quality. There are smoke transport mechanisms that function under flaming and smoldering phases of a fire. These phases are important in determining the types of emissions, how far those emissions will travel, and how they affect safety concerns such as roadway visibility and air quality. Chapter 4 describes the current state of knowledge about smoke contributions to poor air quality based on monitoring, while Chapter 7 examines smoke contributions through the modeling of two case study fires. Chapter 6 describes health (i.e., both on firefighter health and at the population level) and ecological effects associated with smoke and worsened air quality. With respect to firefighters, health effects can be immediate due to extreme heat, burns, asphyxiation, overexertion, or accidents or can be delayed due to smoke-related diseases such as cancers and other chronic conditions like heart disease that may be associated with prolonged and repeated exposures to extreme heat, overexertion, and stress (Domitrovich et al., 2017). Health effects at the population level vary from respiratory symptoms to more severe effects that require a visit to an emergency department or a hospital and could even result in death. While health effects represent negative effects, Chapter 6 also recognizes that smoke can have some positive effects, such as stimulating flowering of some perennial grasses and herbs and contributing to climate cooling.

2.3.4.2.2 ASH-RELATED EFFECTS

Ash from fires, discussed in Chapter 6, can deposit on soils, water, vegetation, and man-made structures and vehicles. Ash deposition can lead to increased nutrient availability in soils, and depending on what types of materials are burned, can also lead to increased levels of metals. Ash deposition can also affect water quality, either directly through ash residues entering water bodies, or through increases in nutrient loadings that result from movement of excess nutrients through soils.

2.3.4.2.3 EFFECTS ON GREENHOUSE GAS (GHG) EMISSIONS

Fires result in the release of several GHGs, both from burning of trees and other woody biomass, as well as from soils. Greenhouse gases released include carbon dioxide (CO_2,) nitrous oxide (N_2O), nitrogen oxides (NO_X), and methane. Emissions are a function of climate, soil properties, and vegetation composition and management practices. Emissions of GHGs occur both during the fire, as well as longer term, due to changes in soil and surface fuel carbon and nitrogen pool sizes, conversions from one vegetation type to another, and changes in soil moisture and temperature associated with canopy removal. There are differences in plume rise and fuels consumed between most wildfires and prescribed fires, resulting in substantially different areas of impact as well as potential entrainment into long-range transport and retention of GHGs in the upper atmosphere [see U.S. EPA (2012)]. A clear benefit of fuel treatments including prescribed fire, is the potential to improve long-term carbon sequestration [see Flanagan et al. (2019); CARB (2015); Wiedinmyer and Hurteau (2010)].

2.3.5 PROGRAMS TO MITIGATE EXPOSURES AND IMPACTS

Prescribed fires occur after extensive planning with one of the objectives being to reduce population exposures to smoke and provide an opportunity to reduce smoke exposures of downwind communities through public health messaging campaigns. As a result, the ability of behavioral actions such as staying indoors or using N95 facemasks when outdoors to mitigate exposures can play a role in reducing the health impacts associated with smoke emissions during prescribed fires. While there is some limited opportunity to encourage these types of behavior during wildfires, prescribed fires provide the opportunity to increase those behaviors in populations that may be at increased risk of smoke-related health effects (i.e., at-risk populations) through preplanned communication and public awareness activities. Likewise, communities can increase readiness for smoke during prescribed fires through public information messaging about nearby burning activity or through messaging campaigns to ensure populations, especially those at increased risk, are taking measures to protect themselves. Consideration of programs that increase awareness of prescribed fire events, including the projected path of smoke plumes, could have a large influence on reducing health impacts. The ability of communities and individuals to engage in behavioral actions to reduce exposures may depend on existing socioeconomic conditions and may be limited by inequities in community and individual capacities to respond to information on burning activities and smoke.

Wildfire smoke also has some opportunities for mitigation of exposures and effects. The implementation of the Interagency Wildland Fire Air Quality Response Program as authorized by PL 116-9 March 12, 2019. Page 617; Section 1114(f), as well as efforts by U.S. Environmental Protection Agency (U.S. EPA), state, tribal, and local air quality regulatory agencies and public health agencies warn the public and at-risk populations of wildfire smoke exposures and ways to mitigate impacts. Through these efforts, the public is becoming more aware of the risks of wildfire smoke exposure and air quality and health impacts. Chapter 6 (Section 6.3) provides a discussion of the various actions and interventions that can be employed by individuals to mitigate or reduce wildland fire smoke exposures.

2.4 IMPLEMENTING THE CONCEPTUAL FRAMEWORK

Wildland fire results in a range of beneficial and detrimental effects, some of which can be quantified, while others are more difficult to quantify. Table 2-1 lists the primary categories of effects associated with wildland fire, both the direct fire effects and those specific to smoke exposure, and highlights those impacts that are the focus of the quantitative analyses that revolve around the case study fires (i.e., Timber Crater 6 [TC6] and Rough fires) examined within this assessment.[7] The nature and magnitude of these effects will depend on the type of fire, vegetation affected, fuels, weather, climate, terrain, and the timescale, but there is potential for these effects in both prescribed fires and wildfires. Effects can occur directly within the fire boundary, adjacent to the fire, or distant from the fire (e.g., impacts of smoke emissions on air quality or degradation of water quality). Additionally, effects can occur within a few days, or over months or years. Effects occurring directly within the fire boundary and adjacent to the fire are referred to as direct fire effects, and effects occurring distant from the fire or occurring after the fire has been extinguished as indirect. Effects can be positive or negative, with positive effects providing some advantage, which could include ecologically restoring ecosystems or mitigating the risk or loss from a wildfire, while negative effects describe detrimental consequences from a fire, which could include damages to public health, property, or infrastructure. The conceptual framework outlined within this chapter described the linkages between the direct fire and smoke effects of wildland fire to lay the foundation for discussions in subsequent chapters that qualitatively and quantitatively evaluate the effects of prescribed fire and wildfire to provide an overall comparison of the benefits and costs associated with different fire management strategies, with a focus on the smoke impacts.

Table 2-1 Primary effects associated with wildland fire: quantified and unquantified for the case study analyses.

	Categories of Expected Effects
Unquantified Effects[a]	***Firefighting (Chapter 6)***
	• Firefighter safety
	• Firefighter injuries/fatalities
	• Firefighter health, both mental and physical
	Economic (Chapter 5)
	• Evacuations
	• Property (e.g., structures)
	• Property (e.g., loss of ecosystem services)
	• Timber and grazing
	• Infrastructure (e.g., powerlines, recreation, others)

[7] Additional categories of effects are provided in the appendix.

Table 2-1 (Continued): Primary effects associated with wildland fire: quantified and unquantified for the case study analyses.

	Categories of Expected Effects
Unquantified Effects[a] (Cont.)	• Municipal watersheds (e.g., reservoirs, industry, agriculture, drinking)
	• Tourism (e.g., recreation, lodging, restaurants, etc.)
	• Aesthetics (e.g., property value, view shed, etc.)
	• Natural and cultural resources
	• Fuel reduction—cost-effective method of treating acres
	• Fuel reduction—treatment opportunities not limited to local markets[b]
	Ecological (Chapter 3, Chapter 5, Chapter 6)
	• Ecological services including game and endangered species
	• Ecosystem health and resiliency
	• Restoration/maintenance of historic natural fire regime
	• Invasive species
	• Climate change (e.g., GHGs, carbon)
	• Redistribution of toxics and nutrients (e.g., mercury, metals, sulfur, nitrogen)
	• Soil and water quality and quantity
	Public Health: Direct Fire (Chapter 5)
	• Injuries
	• Emergency department visits and hospital admissions
	• Premature mortality
	Other: Air Quality (Chapter 5)
	• Property values
	• Productivity (especially for outdoor workers)
	• Educational effects (restrictions in activities, increased absences)

Table 2-1 (Continued): Primary effects associated with wildland fire: quantified and unquantified for the case study analyses.

	Categories of Expected Effects
Quantified Effects[c]	***Air Quality (Chapter 7)***
	• Changes in $PM_{2.5}$ concentrations
	• Changes in ozone concentrations
	Public Health: Air Quality (Chapter 6, Chapter 8)
	• Respiratory- and cardiovascular-related emergency department visits and hospital admissions
	• Premature mortality

GHG = greenhouse gas; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm.

[a]These effects are not quantified for this case study-based analysis that is focused on air quality and corresponding health impacts. Many of these effects have been quantified in the literature for observed fires. Some of the unquantified effects in the case study are not discussed in this assessment.

[b]This fuel-reduction effect reflects the issue that in some locations fuel-reduction options are limited by the lack of local markets for products such as merchantable timber of biomass, resulting in prescribed fire and chipping as the only fuel-reduction options available. The presence of local markets reduces costs and increases the fuel-reduction options available.

[c]Examining these effects represents the primary focus of this assessment.

See Appendix A.2 (Appendix Table A.2-1) for a more detailed version of this table that accounts for whether the effects are positive or negative due to prescribed fire and wildfire.

Fully implementing the conceptual framework detailed within this chapter would require a diverse set of data and models. The ultimate results would be a complete set of quantified, and in some cases monetized impacts, specifically health and ecosystem impacts and corresponding economic values, associated with each selected fire management strategy. Such a comprehensive, fully quantitative assessment, however, would be an enormous undertaking and would furthermore be limited by existing knowledge gaps that make quantification and/or monetization of key effects challenging or impractical at this time. The conceptual framework thus represents an aspirational goal toward which future assessments, informed by targeted research efforts, can build. Therefore, for the purpose of this assessment, quantification is limited to the smoke impacts associated with different fire management strategies, reflecting a comparison of only one (though critical) area of negative effects of wildland fire. Monetization of impacts is useful because it provides a consistent way to aggregate disparate effects. Economic theory and practice typically recommend discounting of benefits and costs that occur in the future to account for societal time preferences [e.g., benefits occurring today are in most cases valued higher than benefits occurring in the future; U.S. EPA (2014)]. Because of uncertainty regarding when wildfires occur relative to when prescribed fires occur, it is challenging to determine the time frames for comparing the two types of fires. For this assessment, we present undiscounted dollar values, which assume that benefits and costs of fire management strategies all occur in the same current year. Comparisons would differ if prescribed fire effects are assumed to occur earlier than wildfire effects. This assessment includes an illustrative calculation of the impact of discounting on the value of wildfire effects for different time delays between prescribed fire and wildfire (Chapter 8). A full accounting of comparisons between strategies would require aggregating all of the monetized benefits and damages for each fire management strategy, and then computing the expected value of damages using Equation 2-2, and differencing the expected values between strategies (e.g., fire management strategy *i* will have benefits compared with fire management strategy *j* if $PEV_i - PEV_j > 0$). However, given the limited availability of data to model many nonhealth impacts, this assessment only aggregates the values of health outcomes (i.e., emergency department visits, hospital admissions, and premature mortality) associated with air quality changes due to smoke.

Net benefits can also be compared between fire management strategies. With a complete set of potential wildland vegetation management strategies, the optimal strategy will be the one with the highest net benefits (net present expected value). Even with an incomplete set, fire management strategy *i* is preferred to management strategy *j* if $NPEV_i > NPEV_j$.

2.5 GUIDE TO THE ASSESSMENT

Each subsequent chapter in the report presents information that is highly relevant to an assessment of the air quality impacts between different fire management strategies. The remaining chapters in Part I provide important background and contextual information that informs, and aids in interpretating the case studies. In addition, these chapters provide an overview and discussion, based on reviews of the relevant literature, of the wildfire effects not quantified as part of the case study analyses. Moving from left to right across the conceptual framework (Figure 2-1), Chapter 3 captures many of these initial components. This includes the baseline forest/ecological conditions of ecosystems similar to the case study areas, provides background information on different fire management decisions, and a history of fire activity, including the implementation of prescribed fire. In addition, the qualitative discussion in Chapter 3 highlights the instances where a wildfire can yield resource benefits, which are quantitatively evaluated in the Rough Fire case study through the modeling of the Sheep Complex Fire in Part II (Chapter 7 and Chapter 8). Chapter 3 also discusses how fire on the landscape can contribute to improved vegetation health and result in ecological benefits.

Chapter 4 summarizes the current national regulatory ambient air quality measurement infrastructure, nonregulatory temporary incident response measurement capabilities, air quality sensor capabilities, and remote sensing products and their utility in estimating the impact of wildland fire smoke on air quality. Although air quality measurement data is not directly used in the quantitative case studies, it is the basis for the development and validation of the deterministic air quality models used to conduct quantitative analyses, such as those developed for this report, as well as wildland fire smoke exposure and health effects research (i.e., epidemiologic studies). The direct fire effects of wildfire (Chapter 5), including societal effects such as economic and ecological and welfare effects, which, while important to consider broadly when making comparisons between different fire management strategies, cannot be quantified at the case study level. Although there are opportunities to mitigate these direct fire effects, they are not accounted for quantitatively within this assessment. The other nonsmoke combustion-related effects of fires, which include GHG emissions (Chapter 3) and ash deposition (Chapter 6), are characterized qualitatively to varying degrees, including the ecological effects of ash deposition. Chapter 6 also provides a qualitative discussion of both health (i.e., firefighter and population level) and ecosystem impacts attributed to smoke exposure, as well as describes the scientific evidence that supports the availability and efficacy of various actions and interventions that can be employed at the individual and community level to mitigate the public health impact of smoke exposure.

Part II is the quantitative modeling component that forms the backbone of this assessment. Chapter 7 describes the smoke emissions and corresponding modeling of air quality impacts, which represent the key inputs to the quantitative analyses. The results of this air quality modeling directly inform both human and ecosystem exposure, although only the resulting human health impacts are quantitatively examined. The current understanding of the health effects of wildland fire smoke exposure (Chapter 6), as well as ambient $PM_{2.5}$ and ozone exposure, are subsequently used within BenMAP-CE to quantify the number of deaths and illnesses attributed to smoke from the different scenarios examined within both case studies, as described in Chapter 8. The overall population $PM_{2.5}$ exposure reductions estimated from these actions and interventions also allow for a limited quantitative assessment of the potential public health implications of promoting such measures. Although these actions and interventions can be instituted for both wildfires and prescribed fires, the planned nature of prescribed fires enhances opportunities for public engagement surrounding prescribed fires and increases the opportunities to inform populations at risk of wildfire smoke-related health effects of actions they can take to protect themselves. Finally, Chapter 9 provides a synthesis of these quantitative findings, discusses study limitations that inform their careful interpretation, and summarizes key knowledge gaps.

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CHAPTER 3 FIRE REGIMES, FIRE EFFECTS, AND A HISTORY OF FUELS AND FIRE MANAGEMENT IN THE WESTERN U.S.

3.1 INTRODUCTION

As described at the end of Chapter 2, each chapter in this report presents information relevant to assessing the effects associated with different fire management strategies. This includes background and contextual information important for informing the development and interpretation of the case study analyses of the Timber Crater 6 (TC6) and Rough fires presented in Part II (Chapter 7 and Chapter 8), as well as a literature-based overview of key wildland fire effects described in the conceptual framework (Figure 2-1) but not quantified as part of those case studies. This chapter covers a number of these contextual and interpretive components, including the baseline forest/ecological conditions of ecosystems similar to the case study areas, background information on different fire management decisions, and a history of fire activity, including the implementation of prescribed fire. In addition, this chapter highlights instances where a wildfire can yield resource benefits, which are quantitatively evaluated in the Rough Fire case study through the modeling of the Sheep Complex Fire, as described later in the report. Finally, this chapter discusses how fire on the landscape can contribute to improved forest health and result in ecological benefits.

3.2 FIRE REGIMES AND ECOLOGICAL CONDITION OF FORESTS

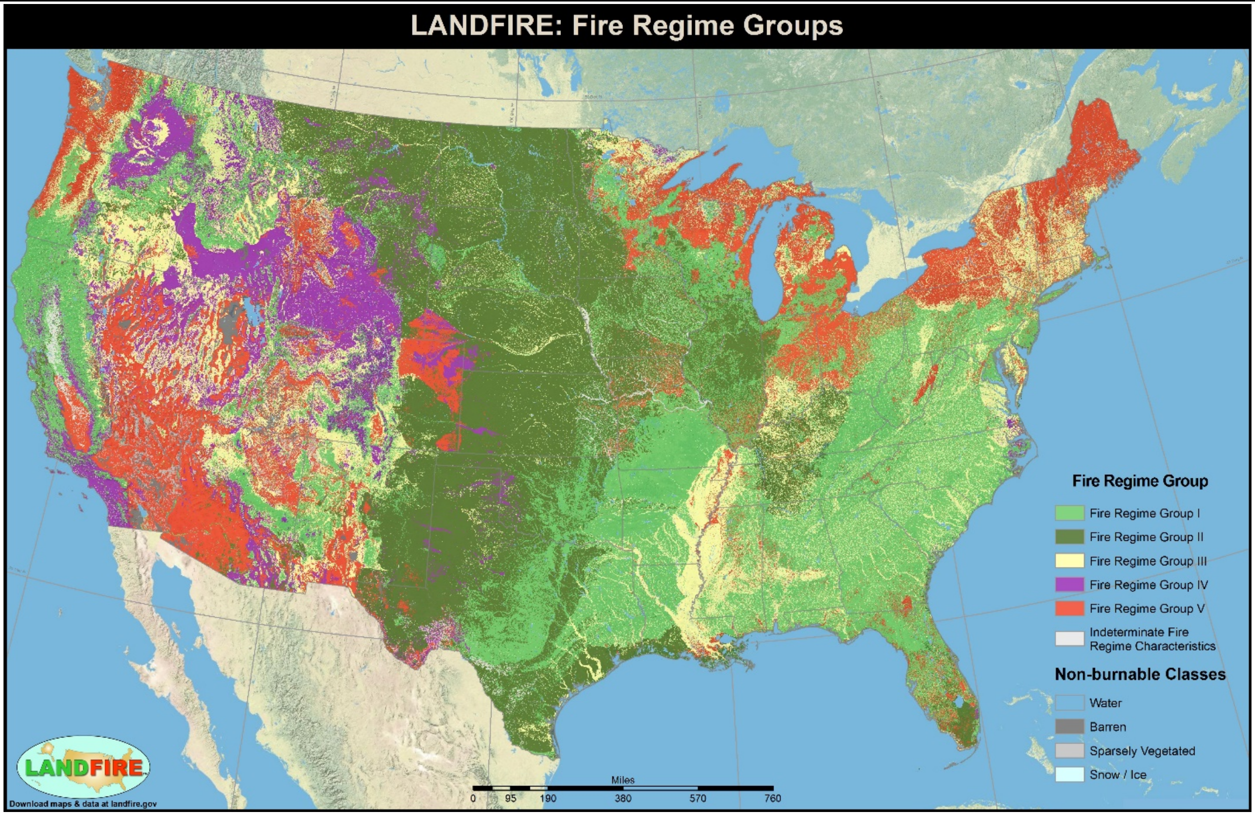
Fire regimes are patterns of fire size, intensity, severity, recurrence, or frequency and the resulting ecological effects that are typical of vegetation assemblages in spatial scales from individual sites to broad regions of the county (Agee, 1993). They are typically based on historical patterns determined by human observation, ecological records, and geological records, depending on the length of available data. They are temporally dynamic, depending on longer term vegetation and climatic distributions, as well as on long periods of human interaction and resource use. Fire regimes have changed with climate over long time periods; they are likely changing now as well, although we are not able to clearly define the changes while they are occurring. They influence forest recovery, succession, structure, and ecosystem functioning (Agee, 1993). Fire regimes are influenced largely by climate, vegetation types, and by topographic and geologic features that either facilitate or restrict fire spread and vary by season and geographic region resulting from regional weather patterns (Taylor and Skinner, 1998; Agee, 1993). However, fire regimes are also strongly influenced by human actions, including those of indigenous people (Carter et al., 2021).

There are numerous classification systems for describing fire regimes, often depending on the context and purpose of the classification system. The most used classifications consider the frequency, severity (or scale of ecological impacts), and spatial scale of wildfire in a natural or quasi-natural condition, although many other variables have been used in classification schemes [Ryan and Opperman (2013); Table 3-1, Figure 3-1]. Fire frequency, or the mean fire return interval, is a measure of how often fire returns, on average, to a specific area. There may be a wide range around this mean, which has important ecological implications for stand development and forest structure (Baker and Ehle, 2001). Landscape fire rotation, often used to characterize fire regimes, refers to the years required for a defined area to experience fire and helps to smooth out variations over space and time to help characterize typical fire regimes (Farris et al., 2010; Romme, 1980; Van Wagner, 1978; Heinselman, 1973).

Table 3-1 Fire regime groups and descriptions.

Group	Frequency (yr)	Severity	Severity Description
I	0−35	Low/mixed	Generally low-severity fires replacing less than 25% of the dominant overstory vegetation; can include mixed-severity fires that replace up to 75% of the overstory
II	0−35	Replacement	High-severity fires replacing greater than 75% of the dominant overstory vegetation
III	35−200	Mixed/low	Generally mixed-severity can also include low-severity fires
IV	35−200	Replacement	High-severity fires
V	200+	Replacement/any severity	Generally replacement-severity; can include any severity type in this frequency range

Source: Hann et al. (2008).



FRG = Fire Regime Group; LF = LANDFIRE.

Note: FRG definitions best approximate the definitions outlined in the Interagency Fire Regime Condition Class Guidebook and refined to create discrete, mutually exclusive criteria appropriate for use with LF's fire frequency and severity data products.

Source: LF (2012).

Figure 3-1 Fire Regime Groups characterizing the presumed historical fire regimes within landscapes based on interactions between vegetation dynamics, fire spread, fire effects, and spatial context.

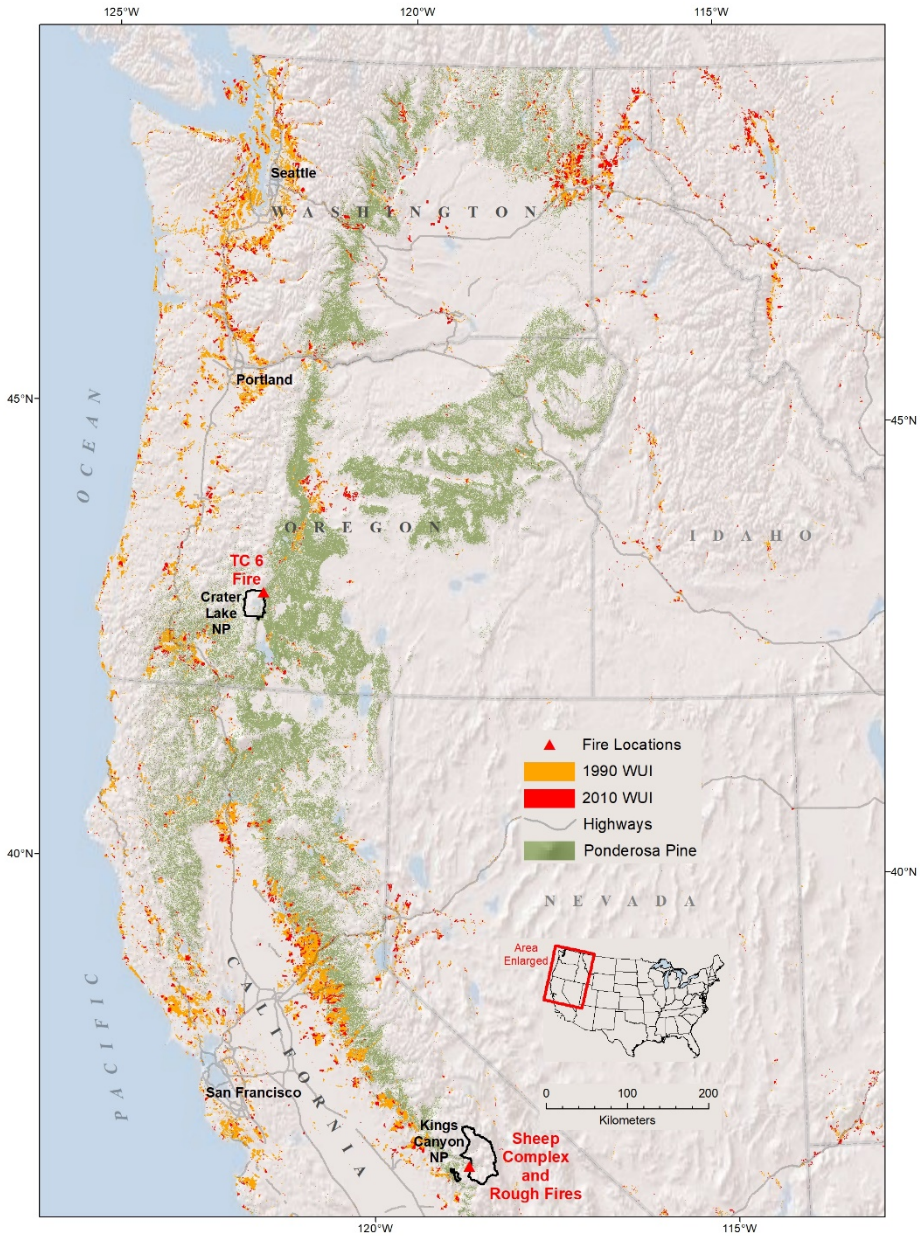
Fire severity, assessed by a mixture of qualitative and quantitative methods, is an estimate or measured assessment of fire effects on soils and vegetation (Table 3-1). Fire intensity, a major factor in severity, is a measure of heat or energy released (kW) per unit length (m) along the fireline and can be estimated by measuring flame length as the flaming front passes a known point (Rothermel and Deeming, 1980). High-intensity fires (e.g., long flame lengths), for example, result in more consumption and charring of surface fuel, increased exposure of soil and alteration of soil properties, and more damage and mortality of trees and other vegetation. Duration of burning at a given site has profound implications for fire severity and smoke production. Although it is somewhat difficult to estimate flame length (a proxy for fireline intensity), estimating actual duration and intensity are more difficult, especially over large areas and when not observed and recorded as fires burn.

3.2.1 HISTORIC FIRE REGIMES IN THE PONDEROSA PINE REGION

This chapter focuses on the characterization of ponderosa pine (*Pinus ponderosa*) ecosystems because they are very well understood and comprise a large portion of the ecosystem within the two case study areas (i.e., Oregon for the TC6 Fire and California for the Rough Fire) that form the basis of the quantitative analyses within this assessment (see Chapter 7 and Chapter 8). At a finer resolution, the case studies do contain some different forest types as well as shrub, grass, and understory vegetation components. However, these areas represent much smaller areas than ponderosa pine and dry mixed conifer forest.

Historically—before widespread European settlement in the late 19th century and establishment of fire suppression as a national policy in the early 20th century—ponderosa pine forests and much of the adjacent dry mixed conifer zone experienced frequent, mixed- to low-intensity fire (Agee, 1993). Periodic fires consumed accumulated fuels, thinned young seedlings and saplings, and consumed shrubs and herbaceous plant material, leaving the large, fire-resistant trees intact. Some individual large trees or small groups of large trees may have been directly killed or stressed by fire and later attacked and killed by bark beetles (Munger, 1917). This fire regime aligns geographically with the current distribution of ponderosa pine, which occupies 76,997 km^2 (14.7% of the land area) in Oregon and Washington, and approximately 94,200 km^2 (11% of the land area) in northern California (Figure 3-2). For the purposes of this assessment, the area occupied by these forests is collectively referred to as the ponderosa pine region.

The continental climate of the ponderosa pine region is semiarid and largely controlled by a rain shadow effect from the Cascade, Coast, and Sierra mountain ranges to the west. Annual summer droughts are a common characteristic as less than 20% of precipitation falls during May−September, based on precipitation data from the Parameter-Elevation Relationships on Independent Slopes Model (PRISM) Climate Group at Oregon State University (Daly et al., 2008). Historically, low-severity surface fires were more frequent and burned over larger areas compared to nondrought years (Hagmann et al., 2019; Johnston, 2017; Heyerdahl et al., 2008; McKenzie et al., 2004). However, drought is usually not the sole or ultimate cause of most tree mortality, but it interacts with pests and diseases, collectively termed biological disturbance agents (BDAs), to influence tree mortality (Kolb et al., 2016). These factors, drought and BDAs, account for much of the tree mortality throughout the region (Hessburg et al., 1994).



TC6 = Timber Crater 6; NP = national park; WUI = wildland–urban interface.

Note: Distribution and expansion of the WUI (Radeloff et al., 2018) in Washington, Oregon, and California has increased from 41,318 to 50,856 km^2 (23%) between 1990 and 2010 and is depicted in orange and red with approximately 8.3% of the WUI in the ponderosa pine region as of 2010. The growth of the WUI in the ponderosa pine region, from 3,072 km^2 in 1990 to 4,211 km^2 in 2010 (37%), highlights how recent fire activity in dry fire-prone forests affects an expanding human population. Locations of the Timber Crater 6 and Rough fires are identified by red triangles.

Source: https://lemma.forestry.oregonstate.edu LEMMA (2020).

Figure 3-2 The ponderosa pine region as defined by the distribution of *Pinus ponderosa* in Oregon, Washington, and northern California (94,000 km^2, 11% of the land area shown) based on 2017 Gradient Nearest Neighbor maps.

3.2.2 HISTORIC FOREST CONDITIONS

Historically, forests in the ponderosa pine region consisted of multiage stands with a structural backbone of large old-growth trees that persisted because of resistance to frequent and extensive fires, severe and prolonged droughts, and BDAs. Douglas fir (*Pseudotsuga menziesii*), grand fir (*Abies grandis*), and white fir (*Abies concolor*) are common associates of ponderosa pine at higher elevations across the region (Safford and Stevens, 2017; Franklin and Dyrness, 1988), while blending at lower elevations to pinyon and juniper woodlands, or savannas of oaks (*Quercus* spp.) and gray pine [*Pinus sabiniana*, Miller et al. (2019)]. Presettlement forests throughout the region were characterized by open, park-like stands of large-diameter trees with a few seedlings and saplings in the understory (Figure 3-3). Stands were typically uneven-aged, with many stands containing a few large individual trees 400 to 600 years old (Youngblood et al., 2004; Arno et al., 1997).

Source: left, BIA photo; right, photo: PA Beedlow.

Figure 3-3 Photos showing the open character of old growth ponderosa pine resulting from high-frequency, low-intensity fire on the Klamath Indian Reservation in south-central Oregon in the 1930s (left) and present-day ponderosa pine forest 10−15 years after natural fire in Ochoco National Forest, central Oregon (right).

3.2.3 FIRE INFLUENCES ON FOREST STRUCTURE AND COMPOSITION

Comparing forest conditions under a frequent low-severity fire regime with infrequent mixed- to high-severity fire illustrates how an open and heterogeneous structure historically resulted in resistant forest conditions over long time periods and across the ponderosa pine region. Patches of high-severity fire historically were small and rare (Heyerdahl et al., 2019; Merschel et al., 2018; Agee, 1993) because fire maintained low surface and canopy fuel loads (Johnston et al., 2016). The result was heterogeneity in horizontal structure at fine (Churchill et al., 2013) and coarse scales (Hessburg et al., 2005). Furthermore, most trees were large and, consequently, fire-resistant (Hagmann et al., 2014, 2013).

3.2.4 ECOSYSTEM RESILIENCE/RESISTANCE TO FIRE

Resilience is the capacity of an ecosystem to recover its essential characteristics following a disturbance, whereas resistance is the property of an ecosystem to remain essentially unchanged when disturbed. Resistance is often thought of as a component of resilience, but the two ecological processes are distinct mechanisms that maintain the essential characteristics of an ecosystem including taxonomic composition, structure, ecosystem function, and process rates (Holling, 1973). Within the ponderosa pine region, open forest structure and fine-scale heterogeneity predominated historically, and this conveyed resistance to fire and other disturbances at fine scales (Koontz et al., 2020), as well as broadly across entire landscapes (Hessburg et al., 2005). However, after years of fire exclusion, in addition to logging and livestock grazing, low intensity surface fires have been excluded in many areas, resulting in dense stands that show both reduced resistance and resilience because of changes in species composition (Busse et al., 2009). Extreme severe fire is now much more likely to occur, reflecting decreased resistance.

3.2.5 CHANGES TO HISTORIC FIRE REGIMES

3.2.5.1 LAND MANAGEMENT PRACTICES

Forest ecosystems in the ponderosa pine region have undergone structural and functional changes in the last 140 years since European settlement (Hessburg and Agee, 2003). Before Euro-American settlement, many dry forests of the western U.S. were maintained by frequent low to moderate severity fires (i.e., cultural fires) often set by indigenous tribes (Hessburg et al., 2005; Agee, 1993). Native American tribes used cultural fires to purposely burn forests and grasslands to promote habitat diversity, environmental stability, predictability, and maintenance of ecotones, but perhaps the most important difference between then and now was the lack of advanced fire suppression technology (Raish et al., 2005). The absence of fire suppression allowed wildfires, both lightening and human caused, to naturally progress across the landscape. Over the last 140 years, forests in the ponderosa pine region have changed immensely and bear little resemblance to their presettlement condition. The original old-growth ponderosa pine forests were once considered an endless resource to early pioneers and settlers, and the vast "yellow pine" forests were used to fuel economic growth and the development of western North America.

Past and current land use activities, along with active fire suppression, eliminated natural surface fires from these forests. Heavy grazing in the late 1800s and early 1900s, as well as active fire suppression after the 1910 fires and other land uses, have disrupted the natural fire regime in these ecosystems. Tree establishment and survival increased in the late 19th and early 20th centuries, resulting in denser forests characterized by increased homogeneity in horizontal structure, increased canopy layering and connectivity, inter-tree competition, and canopy cover. This densification combined with widespread logging of large and old fire-resistant trees (Naficy et al., 2010; Hessburg and Agee, 2003) contributed to mesophication—a shift from drought and fire-resistant shade-intolerant species to shade-tolerant species adapted to competition but not as resistant to drought and fire (Nowacki and Abrams, 2008). Aggressive fire suppression since 1910 ensured that densification and mesophication continued to the present. The forests of today are the cumulative result of tree establishment and growth versus mortality from drought, pests and diseases, fire, and land management [e.g., timber harvesting, thinning, prescribed fire; Merschel et al. (2021)].

3.2.5.2 HABITAT FRAGMENTATION FROM HUMAN POPULATION GROWTH

Wildfires pose the greatest risk to people in the wildland–urban interface (WUI)—the area where houses are in or near wildland vegetation (Radeloff et al., 2005). It is the fastest growing land use type in the conterminous U.S. From 1990 to 2010 new houses in the WUI increased by 41%, from 30.8 to 43.4 million and land area increased 33%, from 581,000 to 770,000 km^2 (Radeloff et al., 2018). A more current study estimates ~49 million residential homes in the WUI, a number that has been increasing by roughly 350,000 houses per year over the last two decades (Burke et al., 2021). In the ponderosa pine region of Oregon, Washington, and California (Figure 3-2) the land area of WUI increased by 37% between 1990 and 2010 to 4,211 km^2. Further, the expansion of the WUI leads to an increase in human-caused fires (BLM & USFS, 2019).

3.2.5.3 INVASIVE SPECIES AND ENCROACHMENT

Invasive species can establish permanency within ponderosa pine landscapes, but less frequently than within other biomes. The conditions required for invasive species to dominate ponderosa pine landscapes is complex. Many site features favor invasive plant suppression such as frequent small to moderate fires, fire resistant trees, rugged terrain, and high elevation (Zouhar et al., 2008). The establishment of invasive species within the ponderosa pine region has been less relative to grasslands and deserts, likely due to "less activity by humans, relatively intact shrub and tree canopies, [and] harsh climates"(Zouhar et al., 2008). Particularly concerning are invasive exotic grasses in that they may significantly affect fire regimes (Kerns et al., 2020). There are published reports of invasive annual grasses in ponderosa pine forests in Oregon and California. Keeley and McGinnis (2007) specifically note cheatgrass (*Bromus tectorum*) invasions were considered problematic in the vicinity of the Rough Fire as far back as the late 1990s. Sites that do contain abundant levels of invasive plants have usually been disturbed first by human activity (Keeley et al., 2003; Moore and Gerlagh, 2001). Moderating fire intensity and targeting areas of high severity for remediation may reduce post-fire invasive plant outbreaks (Symstad et al., 2014). Without the periodic occurrence of fire, because of fire suppression or due to cycles of climate change, the distribution of native species may change [e.g., the encroachment of woody species into areas formerly dominated by grasses, herbs, and shrubs; Miller et al. (2005)].

3.2.5.4 CHANGING CLIMATIC CONDITIONS AND BIOLOGICAL DISTURBANCE AGENTS

Topography, fire weather, and fuels have generally not limited chronic low-severity fire even in relatively cool–moist environments where relatively fire susceptible Douglas fir and true fir (*Abies* spp.) were common prior to fire exclusion (Hagmann et al., 2019; Merschel et al., 2018; Johnston et al., 2016; Heyerdahl et al., 2008). However, in the last 30 to 35 years, the West has seen a steady rise in the intensity of wildfires, as well as area burned, tied to human-caused climate change (Goss et al., 2020). Drought conditions occurred in 15 of 18 years during 2000−2015 as air temperature was increasing at 0.3°C/decade (Abatzoglou and Williams, 2016). The years from 2000−2018 contained the driest 19-year period in western North America since the late 1500s (Williams et al., 2020). Recent drought in western North America was partially a product of natural variability, but its concurrence with anthropogenic warming resulted in intensity and duration on par with the most extreme drought events since 800 CE (Williams et al., 2020). As climate continues to warm in the 21st century, drought and related impacts to forests are projected to increase (Luce et al., 2016).

Increasing drought severity in combination with climate-driven fungal pathogens and insect pests are thought to exacerbate the fire hazard (Allen et al., 2019). A central paradigm of forests with significant impacts from these BDAs is that they pose an increased fire hazard, that fires are more likely to occur, and fire intensity, severity, and ecological impacts are greater in forests that have been impacted (Halofsky et al., 2020; Parker et al., 2006). Both drought- and BDA-induced tree decline and tree mortality change forest structure, including the abundance and architecture of dead wood in the forest. However, research on drought effects, bark beetle-caused mortality, and defoliator influence on forest structure suggest that their interaction with fire is complicated and the relative influence of BDAs remains unclear (Kane et al., 2017).

3.3 LAND MANAGEMENT APPROACHES TO REDUCING FIRE RISKS

Fire is an important tool to improve forest conditions, reduce fuels, and decrease the threat of large, high-severity wildland fires (Vaillant and Reinhardt, 2017). Fire managers have used natural ignitions as a key component in the restoration of historical forest conditions and fuel loadings. The 2009 Policy Guidance (FEC, 2009) provided federal land management agencies and their state partners greater flexibility to use natural ignitions to meet resource objectives through strategies other than full suppression. Though some land managers have increasingly used wildfire to meet resource objectives since the 1970s (Hunter and Robles, 2020; Collins et al., 2007), managers more commonly resort to full suppression strategies as a consequence of current land-management policies and local land use planning (Meyer et al., 2015; Thompson et al., 2013) which favor protection of homes and other human infrastructure, especially around the WUI. However, land management and planning policies are beginning to be revised to be more inclusive of using prescribed fire, mechanical treatments, biological control, and natural ignitions from lightning to meet resource objectives (Young et al., 2020). This section reviews our current understanding of the effectiveness and limitations of these approaches.

3.3.1 FIRE DEFICIT

With increasing growth of the WUI, long-term suppression of wildfires and resulting forest changes, and an era of increasing drought, wildfire has become a profound ecological and social issue in forests formerly dominated by frequent low intensity wildfires (Moritz et al., 2014). In response to decades of fire suppression, resulting in a fire deficit, and increasing periods of drought, wildfires have tended to become both larger and more severe. Compared to the area burned historically, there exists today an enormous fire deficit in the region, especially for low-severity fire. The fire deficit extends to a vast portion of dry forests of the conterminous U.S. (Kolden, 2019). Ponderosa pine ecosystems may present the clearest example of a large-scale vegetation type with a fire deficit. Historically, open forests characterized by larger trees was the most common structural condition in the ponderosa pine region (Hagmann et al., 2014, 2013). However, in some high-elevation and alpine forests, humid temperate forests, and shrublands, there may not be a deficit, and may indeed be a surplus of fire; these areas are beyond the scope of the current assessment and the two case studies. Invasive plant species, for example cheatgrass (*Bromus tectorum*), may accelerate fire spread and increase fire frequencies on large landscapes, and the occurrence of invasive plant species may preclude ground disturbance in land management, including the use of prescribed fire. Tree regeneration and growth in the absence of frequent low-intensity fire in contemporary times has resulted in the loss of open resistant forest structure and composition, sparse woodlands, and nonforest cover (Stevens-Rumann et al., 2018). Wildfires in these denser forests tend to be more severe and have a greater chance of converting forested areas to different vegetation types [e.g., becoming shrublands in drier areas; Moreira et al. (2020); Parks and Abatzoglou (2020)].

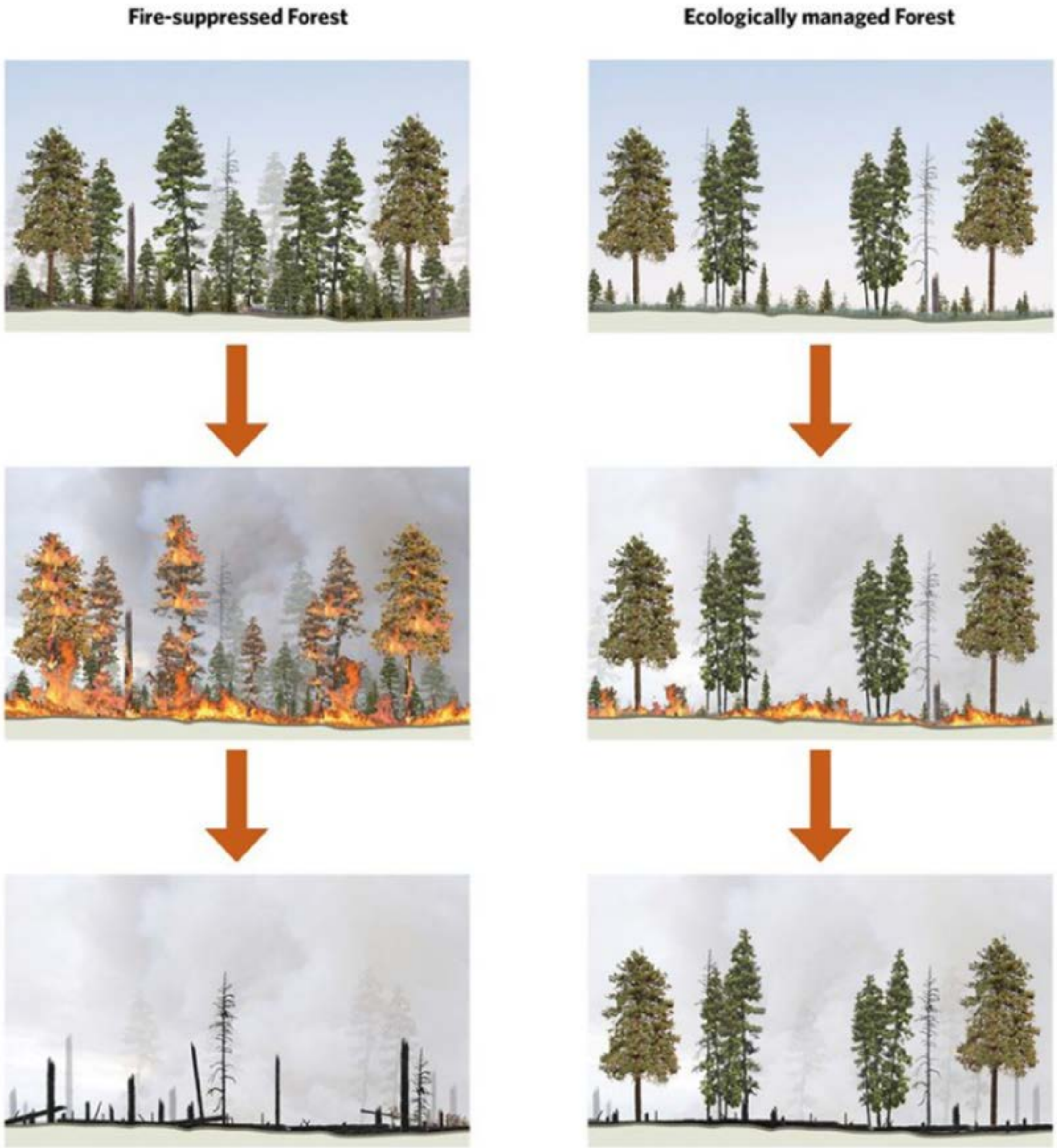
3.3.2 LAND MANAGEMENT ACTIVITIES AFFECT FIRE BEHAVIOR

Forest policy and management practices are slowly changing from predominantly fire suppression to managing fire and associated risks to communities (Thompson et al., 2018; Ingalsbee and Raja, 2015). Fire exclusion over the last century in ponderosa pine forests has allowed for the buildup of surface fuels on the forest floor and shrub cover and tree regeneration to increase. This buildup has created “fuel ladders” where surface fuels are now connected to the overstory canopy by dense understory and midstory saplings and medium-sized trees. As a result, it is easier for surface fires to move up and torch tree crowns and, under the right weather conditions and topographic setting, support active crown-to-crown fire spread.

Removing accumulated surface fuels or targeting the removal of specific brush fuels (such as bitterbrush [*Purshia tridentata*] because of its high energy content) reduces flame lengths, making it more difficult to initiate torching of tree crowns. Also, the higher the base of tree crowns, the more difficult it is for surface flames to torch them. Once a fire begins torching and moving up into the canopy, the rate of spread and density of the crowns determines the likelihood of an actively moving crown fire. Increasing the space between tree crowns reduces the ability for fire to spread from tree crown to tree crown and allows a crown fire to transition back to a surface fire.

Currently, the forest area being managed to reduce density, restore large ponderosa pine trees, and reintroduce low-intensity, frequent fire is very small compared with the forest area experiencing continued densification and succession. Fire is not being adopted into management practices at a scale necessary to affect the fire deficit in the western U.S. and reduce the potential for more wildfire disasters; the area burned by prescribed fire actually decreased in the Pacific Northwest from 1998−2018 (Kolden, 2019).

Land Managers have tools and methods to improve fire resilience and resistance in the ponderosa pine region: these include reducing surface fuels, removing ladder fuels, leaving large fire-resistant trees, and spacing tree crowns (see Chapter 5 for economic considerations). These conditions can be achieved with a variety of methods including prescribed fire, use of naturally ignited wildfire to achieve land management objectives, mechanical thinning, and biological control (Agee and Skinner, 2005). The use of multiple tools to reintroduce fire as a natural process in fire-prone forests has come to be known as ecological forestry (Kelsey, 2019) and involves targeted removal of forest fuels plus implementation of prescribed fire and managed wildfire where it is safe to do so (Figure 3-4). Management to improve fire resilience decreases drought stress, and also increases climate resilience.



Note: Fire-Suppressed Forest (left): Forests become dense with thickets of young trees and shrubs in the understory and are prone to high-severity fires that can kill most of the trees. Ecologically Managed Forest (right): Strategic thinning the understory can reduce overall fuel load so fire can safely be reintroduced to maintain healthy forests. (Kelsey, 2019).

Figure 3-4 Comparison of differences between a fire-suppressed and ecologically managed forest.

3.3.2.1 PRESCRIBED FIRE

Prescribed fire is one of the most widely advocated management practices for meeting land management goals and objectives and has a long and rich tradition rooted in indigenous and local ecological knowledge. The scientific literature has repeatedly reported that prescribed fire is often the most effective means to reduce fuels and wildfire hazard to restore sustainable ecological functioning to fire-adapted ecosystems in the U.S. following a century of fire suppression (Kolden, 2019).

As defined in Chapter 2, a prescribed fire is "any fire intentionally ignited by management actions in accordance with applicable laws, policies, and regulations to meet specific land or resource management objectives" (U.S. EPA, 2020). Prescribed fire is used on the landscape to remove accumulated surface fuels, consume slash generated from thinning activities, kill and thin out encroaching trees in the understory, and rejuvenate herbaceous plants and shrubs (Sackett and Haase, 1998; Walstad et al., 1990; Ffolliott and Thorud, 1977). Prescribed fire also scorches and kills lower branches of trees, which, in the long run, results in lifting the canopy much like pruning, increasing the height from the forest floor to the lower canopy and increasing fire resistance (Figure 3-5).

Source: Photo: PA Beedlow.

Figure 3-5 Prescribed fire in ponderosa pine, Deschutes National Forest.

Periodic burning can prevent the development of fuel ladders and can be used to maintain fire-resilient stands. However, in most forests of the ponderosa pine region, prescribed fire is limited as an initial treatment to reduce fuel loads because of heavy accumulations of surface and ladder fuels. In many cases, other mechanical treatments are needed prior to prescribed fire to reduce fuels to a level that will allow fire to be used without unnecessary damage to the forest.

The ability to control fire while minimizing human exposure to smoke and achieving the desired ecological results are central goals of prescribed burning (Long et al., 2018). On federal and most state lands, prescribed fire is only used after thorough preplanning and only by highly trained and experienced professionals (NWCG, 2017). Go/no-go checklists are used to determine compliance with policies and the prescribed fire plan parameters. Based on these guidelines, prescribed fire is only implemented when weather conditions are favorable, such as under good smoke clearance conditions, moderate temperatures, dry fuel conditions that result in rapid consumption and ventilation, and an incoming cool/moist weather pattern. Further, burning when smoke is not being produced by many wildfires over a large area is favored to reduce the magnitude and duration of smoke exposure. In much of the western U.S, spring or after the start of fall rain provide good opportunities to manage wildfires due to environmental conditions resulting in low-severity, shorter duration fires. In the ponderosa pine region, most prescribed burns are conducted in the spring and late fall because personnel are available and weather conditions are favorable.

3.3.2.2 MECHANICAL TREATMENTS

Prescribed fire as a restoration tool, while often the cheapest to implement, is not practical in many cases owing to limited burning seasons, excessive accumulation of fuel due to fire exclusion, concerns over potential undesirable fire effects, concerns about human exposure to smoke, visibility, and the chance that a fire will escape and cause damage. Mechanical treatments can create a variety of uneven-aged or even-aged stand structures depending on the desired treatment goals (e.g., fuel reduction to meet fire behavior goals), wildlife habitat maintenance requirements (e.g., for endangered species), and restoration of spatial and structural condition (Huggett et al., 2008). They require equipment as well as plans for disposal or use of significant quantities of small trees (Agee and Skinner, 2005).

How the residual wood from the thinning operations is disposed of can have a large impact on surface fuel availability with chipping or burning of the unusable tops of trees having the greatest impact on reducing fuel loads. Mechanical treatments include activities, such as cutting and piling or stacking trees, cutting and piling brush, pruning lower branches of trees, and creating fuel breaks based on treatment objectives. Typically, mechanical treatments are emphasized in the WUI, while both mechanical and fire treatments, alone or in combination, are emphasized in adjacent lands from which wildland fire might spread into the WUI (Barros et al., 2019).

3.3.2.3 BIOLOGICAL AND CHEMICAL CONTROL

Biological control involves the intentional use of domestic animals, insects, nematodes, mites, or pathogenic agents such as bacteria or fungus that can cause diseases in plants to reduce or eliminate vegetation. Biological controls are used mostly to control invasive plants but can be used to control native vegetation for fire management purposes. For instance, cattle may be used for target grazing in defined areas for the creation of fuel breaks on rangelands and in some instances in forested lands.

In addition to natural agents, chemical agents such as herbicides can be used to kill or injure plants to meet land management objectives. Herbicides can be categorized as selective or nonselective. Selective herbicides kill only a specific type of plant, such as broad-leaved plants, while nonselective can kill all plants. Only those herbicides approved for use can be used to manipulate vegetation to meet land management goals and objectives.

3.3.2.4 USE OF WILDFIRE

Remote forest areas, as well as designated wilderness areas and national parks, provide the best opportunities for taking advantage of natural ignitions to reduce fuel loads. Although fire managers may choose to suppress fire inside or outside of wilderness areas, it is also federal policy to use fire “to protect, maintain, and enhance resources and, as nearly as possible, be allowed to function in its natural ecological role” (FEC, 2009). The very definition of wilderness in the Wilderness Act of 1964, as an area “managed so as to preserve its natural conditions and which generally appears to have been affected primarily by the forces of nature” aligns closely with federal fire policy and thus these areas are often excellent locations to achieve this goal. Moreover, wilderness area ignitions are often in steep, rugged terrain too dangerous for firefighters to attack directly or that limit the technologies and equipment that can be deployed.

Agencies permit lightning-caused fires to play a natural ecological role to reduce the risks and consequences of wildfire both within and outside wilderness areas. Fire managers seek to prevent fires from causing damage to nearby communities. In pursuit of that goal, minimum impact suppression techniques are implemented that cause the least alteration of the wilderness resource and the least disturbance to the land surface, air quality, and visitor solitude (USFS, 2007). The initial response to lightning-caused wildfires is suppression if they occur in a landscape without a fire management plan, do not meet certain conditions, or cannot achieve land management objectives.

3.4 FOREST CHARACTERISTICS FOR THE TIMBER CRATER 6 (TC6) AND THE ROUGH FIRE CASE STUDIES

This assessment focuses on a quantitative analysis of the air quality and associated health impacts of smoke (Part II, Chapter 7 and Chapter 8). The information presented in the preceding sections of this chapter informed the application of a suite of models to quantitatively assess air quality and human health impacts of representative wildfires and prescribed fires within the ponderosa pine ecoregion. Two case study fires were chosen for this assessment (see Chapter 1), both of which occurred in the western U.S.: (1) the TC6 Fire that occurred from July 21−26, 2018 in Oregon; and the (2) Rough Fire that occurred from July 31 to October 1, 2015, in California. The TC6 Fire burned approximately 3,000 acres in Crater Lake National Park from July 21 to July 26, 2018 (https://vimeo.com/287892212). The Rough Fire burned in parts of the Sierra National Forest, Sequoia National Forest, and Kings Canyon National Park between July 31 and October 1, 2015 (https://www.nps.gov/seki/learn/nature/rough-fire-interactive-map.htm), burning approximately 150,000 acres. These fires were chosen as case studies because they were on federal land previously subjected to fuel reduction management. Both areas are in dry forests characteristic of the ponderosa pine region. The following sections describe the forest characteristics of the case study areas. Additional details describing the spatial domains, fuel types, fuel loads, burn characteristics, and air quality results associated with each case study fire are provided in Chapter 7.

3.4.1 TIMBER CRATER 6 (TC6): CRATER LAKE NATIONAL PARK/FREMONT-WINEMA NATIONAL FOREST

Crater Lake National Park spans the divide of the Cascade Mountains in central Oregon. Forests in the western part follow an elevational gradient from low elevation Douglas fir forests, to mixed conifer forests dominated by red fir (*Abies magnifica*), to mountain hemlock (*Tsuga mertensiana*)-dominated stands at high elevation (Forrestel et al., 2017). The eastern portions of the park are dominated by ponderosa pine grading into mixed-conifer forests at higher elevations. Forests in which ponderosa pine is a dominant tree principally occur up to 1,675 m elevation (Adamus et al., 2013). Ponderosa pine forests can contain a mixture of ponderosa pine, white fir (*Abies concolor*), and scattered sugar pine (*Pinus lambertiana*) and Douglas fir. Where ponderosa pine shares dominance with these species, the forests can be called mixed conifer. On the east side of the park, lodgepole pine (*Pinus contorta* var. *murrayana*) is a common associate with ponderosa pine, and understory species may include the Great Basin shrubs, such as antelope bitterbrush (*Purshia tridentata*), greenleaf manzanita (*Arctostaphylos patula*), and a greater abundance of native grasses.

The TC6 Fire started with a lightning strike in the northeast portion of the park on July 15, 2018 and within 4 days spread into a nearby section of the Fremont-Winema National Forest (Figure 3-6). The fire spread to property where the U.S. Forest Service had invested in fuel treatments starting in the 1990s. Treatments included mowing and small tree thinning followed by pile burnings and prescribed burning.

The fire had the potential to burn about 81 km^2, but because of the fuel treatments, it was contained to 12.7 km^2 (Delamarter, 2019).

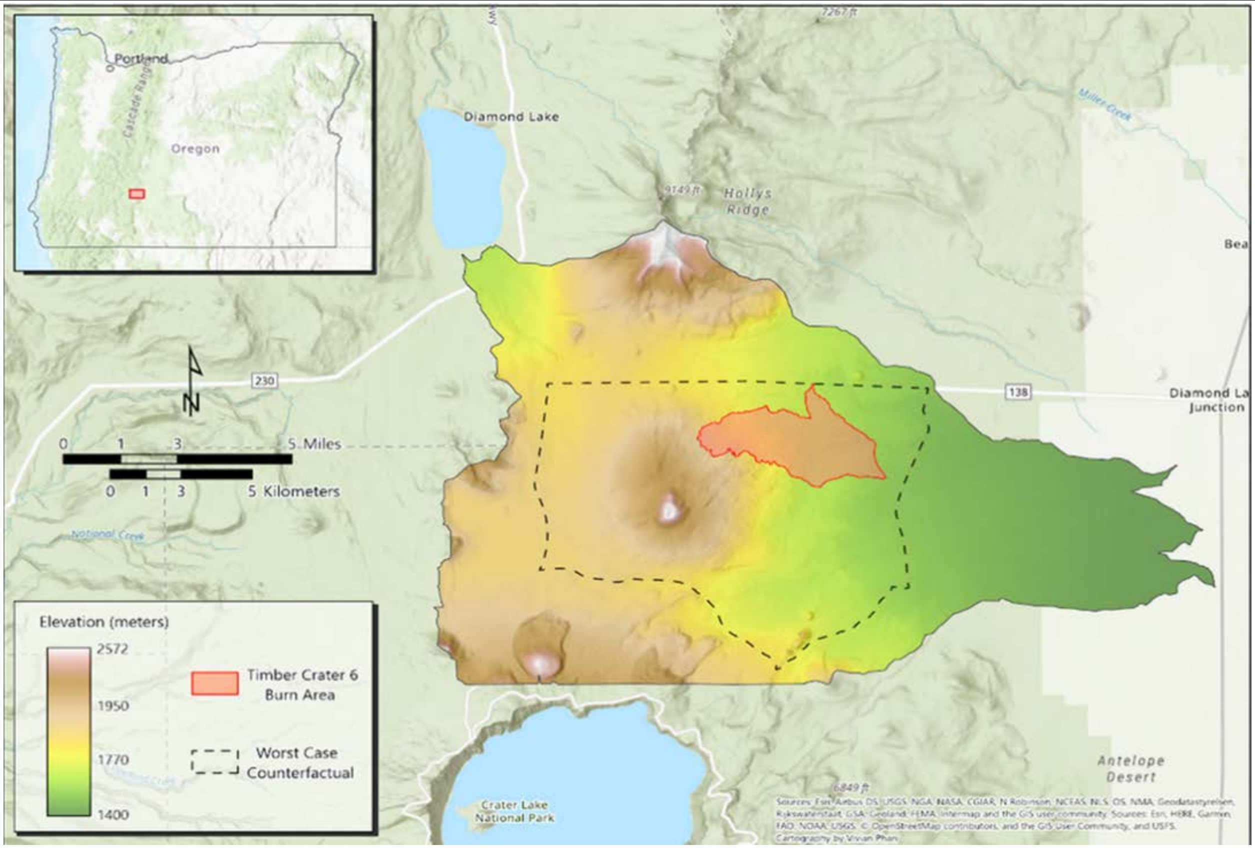


Figure 3-6 **Timber Crater 6 (TC6) Fire, Crater Lake National Park and adjacent Fremont-Winema National Forest.**

3.4.2 ROUGH FIRE: SIERRA AND SEQUOIA NATIONAL FORESTS AND KINGS CANYON NATIONAL PARK

In the Sierra Nevada, especially on the western slope exposed to moisture off the ocean, much of the area where ponderosa pine occurs is considered mixed conifer (Safford and Stevens, 2017), often referred to in California as Yellow Pine Mixed Conifer. The Rough Fire burned a substantial area of the Kings Canyon, one of the deepest canyons in California, in a footprint that spanned an elevational gradient of more than 2,100 m, from ~300 m above sea level (ASL) to just under 2,500 m ASL (Figure 3-7). On the western side of the southern Sierra Nevada Mountain range, which is exposed to storms and prevailing winds coming off the Pacific Ocean, this area of the canyon encompasses a steep precipitation gradient, with a distinct Mediterranean annual pattern allowing for high productivity, but also requiring robust summer drought tolerance. This precipitation gradient and moisture availability pattern in turn drives a diverse range of vegetation assemblages and growth strategies, from grassland and oak woodlands in the lower elevations (~300 to 1,200 m ASL) to highly productive yellow pine (ponderosa pine) mixed conifer (including giant sequoia [*Sequoiadendron giganteum*]) in the mid elevations (1,200 to 2,100 m ASL) to red fir and lodgepole pine typical of boreal forest in the higher elevations (over 2,100 m ASL) of the Rough Fire footprint. Pure ponderosa pine stands occur in the lower to mid-elevations but make up a relatively small fraction (~7%) of the total area burned by the Rough Fire. However, ponderosa pine (and its higher-elevation cousin, the Jeffrey pine) is often an important component of the mixed conifer zone, which comprises a majority (~33%) of the forested area burned by the Rough Fire (Huang et al., 2018; Safford and Stevens, 2017).

Throughout this complex and highly heterogeneous matrix of vegetation types (Figure 3-8), fuels generally increase with elevation from under 2,000 Mg/km^2 in the lower elevation oak woodlands and grasslands to over 18,000 Mg/km^2 in the mixed conifer and upper montane vegetation of the mid-upper elevations. The overall amount of those fuels was also likely enhanced by an unprecedented mortality event, wherein about 30% of the area burned by the 2015 Rough Fire had experienced at least 10% canopy cover loss due to tree mortality prior to the fire. Ponderosa pine and mixed conifer stands in the lower to mid-elevations in particular appeared to have suffered the most uniformly severe mortality (Fettig et al., 2019; Paz-Kagan et al., 2017). Though the proximate cause of this mortality was likely a bark beetle infestation that opportunistically attacked trees weakened by several years of antecedent drought from 2012 through 2015 (Restaino et al., 2019), these lower elevations had also experienced chronically phytotoxic levels of ozone and nitrogen deposition for decades [e.g., Yates et al. (2020); Panek et al. (2013); Peterson et al. (1991)], which likely contributed to their susceptibility to those beetles and the drought (Jones et al., 2004). By the time the Rough Fire burned in 2015, this mortality event was in the "red needle" phase, wherein the tree canopy consisted of dead or dry needles and twigs, which contributed to increased crown fire potential and higher forest fire severity (Stephens et al., 2018; USFS, 2016). Torching potential and ember production were also thought to have occurred in areas affected by tree mortality (Reiner, 2017). In the years after a fire, dead trees not consumed in the fire decay, and the coarser "gray phase" fuels fall to the ground increasing fuel loads potentially contributing to larger scale, "mass fire" events similar the more recent 2020 Creek Fire (Stephens et al., 2018).

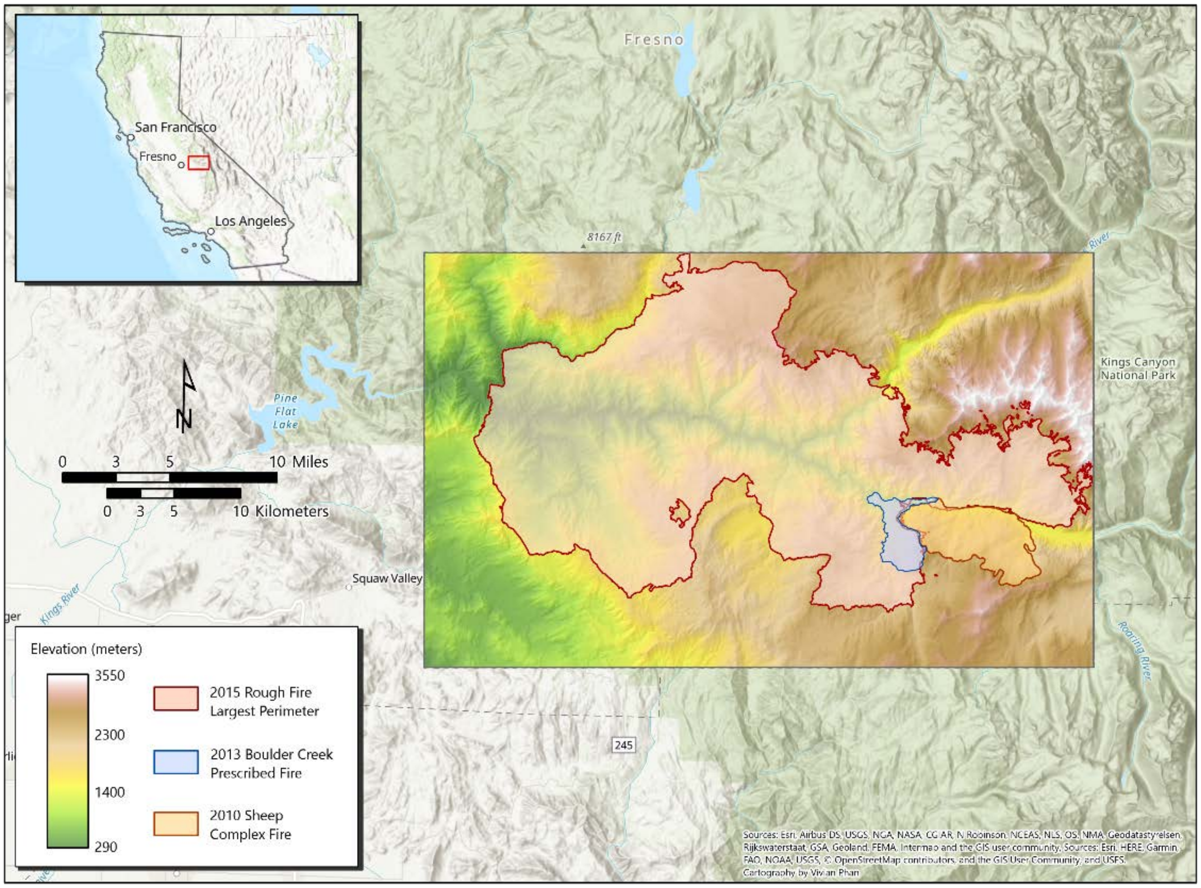
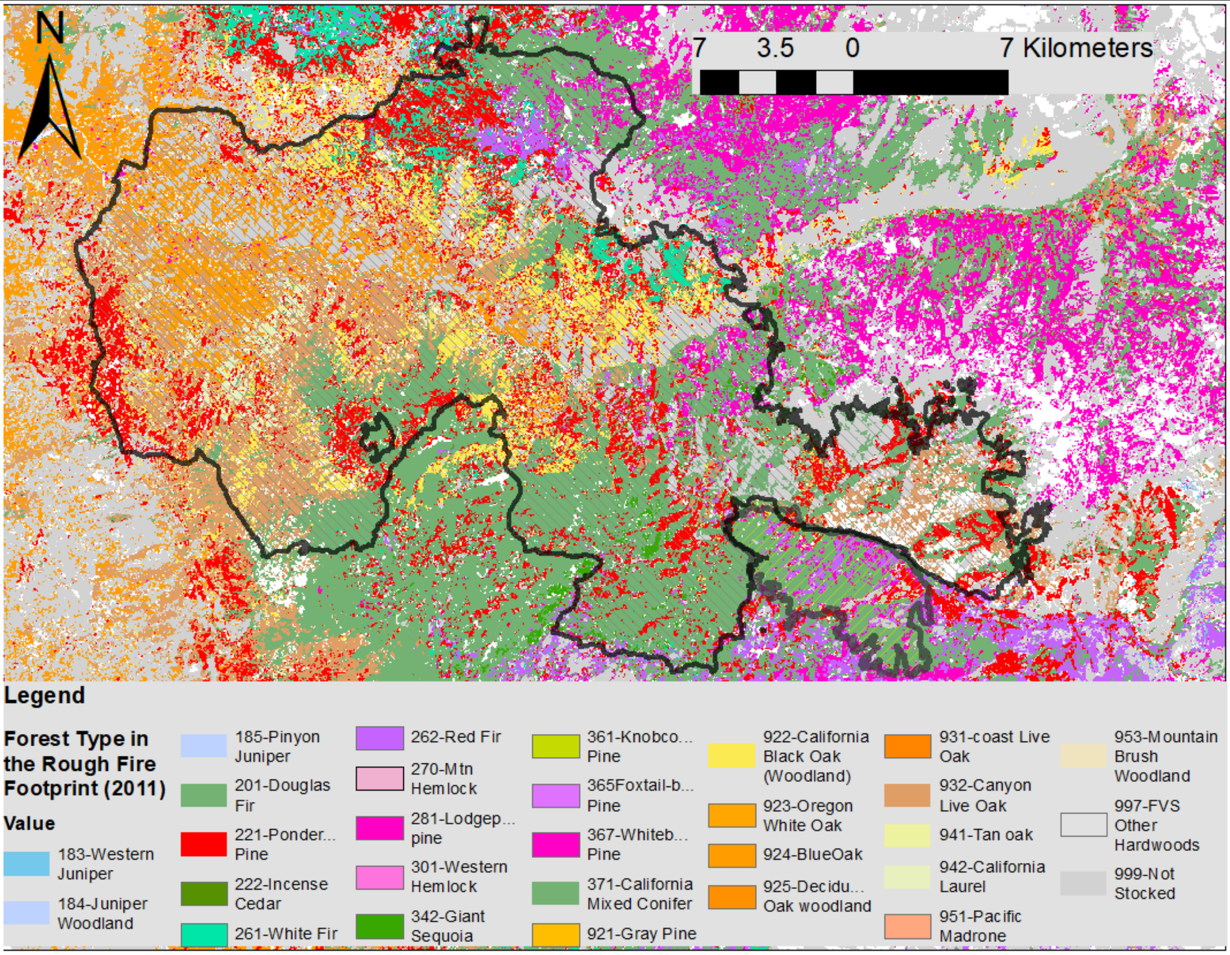


Figure 3-7 Rough Fire: Sierra and Sequoia National Forests and Kings Canyon National Park.



Note: Based on Forest Inventory and Analysis (FIA) and satellite data, see Huang et al. (2018), copyright permission pending.

Figure 3-8 Tree species maps for the area of the Rough Fire.

3.5 CONCLUSIONS

From an ecological perspective, restoration of frequent low-severity fire is essential to restoring sustainable ecosystems in the dry forests of the ponderosa pine region. However, extensive densification and mesophication of these dry ecosystems due to land management practices in the 20th century, followed by an increase in wildland fire frequency and severity, drought, invasive species, pests and diseases, as well as the rapid expansion of the WUI pose serious ecological and socioeconomic challenges to human well-being in the 21st century. Key to living with fire in the ponderosa pine region is an all-lands and all ownerships approach to forest management planning that helps determine where prescribed fire and mechanical treatments are appropriate and should be prioritized, and where fires can be safely managed to achieve the desired resource benefit (Dunn et al., 2020).

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CHAPTER 4 AIR QUALITY MONITORING OF WILDLAND FIRE SMOKE

4.1 INTRODUCTION

Wildland fires (i.e., prescribed fire and wildfire) can produce significant air pollution emissions that can pose health risks to fire crews, first responders, and nearby residents, as well as downwind populations (see Chapter 5, Chapter 6, Chapter 7, Chapter 8). Wildland fire smoke is a complex mixture of thousands of different organic, inorganic, gaseous, and particulate phase compounds (Reisen et al., 2015). The primary constituents of emitted wildland fire smoke that affect air quality are fine particulate matter (PM with a nominal mean aerodynamic diameter less than or equal to 2.5 µm [$PM_{2.5}$]), aerosol black carbon (BC), carbon monoxide (CO), oxides of nitrogen (NO_X), and volatile organic compounds (Urbanski, 2014)]. The secondary photochemical formation of $PM_{2.5}$ and ozone (O_3) from wildland fire emission precursors is also a concern (Liu et al., 2017; Alvarado et al., 2015; Jaffe and Wigder, 2012).

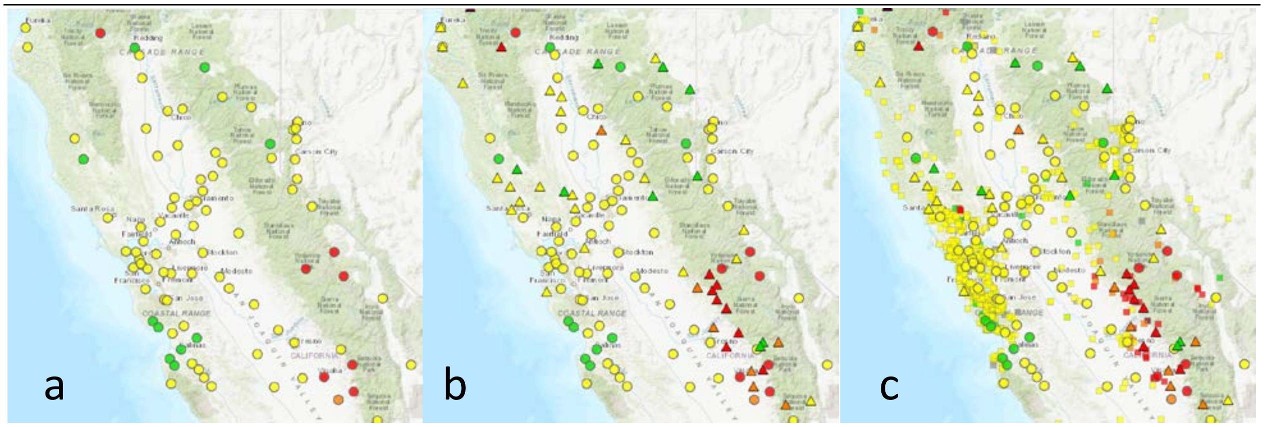
The impact of wildland fire smoke plumes on specific downwind locations is influenced by the behavior and location of the fire, how the emissions are lofted into the atmosphere, and subsequent transport, chemical transformation, and dispersion. The effect of surface-level smoke can be highly spatially/temporally variable, and air quality monitoring sites within affected regions may not adequately represent the very dynamic temporal evolution of smoke beyond its immediate location. Information on general ambient air quality, the effect of wildland fire smoke on current ambient air quality conditions, and air quality forecasts is available to the public through the multiagency AirNow website (AirNow, 2021a), as well as state and local websites. Several western states have websites ("smoke blogs") dedicated to providing the public with information on wildfire smoke (Appendix A.4.1). The material delivered by these smoke blogs varies from state to state with the sites leveraging data from a variety of sources such as smoke and fire observations and forecast products. AirNow also provides modeled forecasts for future air quality and links to numerous resources for understanding air quality during smoke episodes and protecting public health [e.g., U.S. EPA (2019e)]. The accuracy of the reported air quality data and the appropriateness of the associated AirNow public health messaging are a direct function of the underlying measured observational air quality data and spatial interpolation models.

This chapter summarizes current national regulatory ambient air quality measurement infrastructure, nonregulatory temporary incident response measurement capabilities, air quality sensor capabilities, and remote sensing products and their usefulness in estimating the impact of wildland fire smoke on air quality. Limiting exposure is the principal measure available to mitigate human health impacts of smoke, and real-time measurements of air quality are critical to providing actionable guidance to communities for minimizing population exposure. Air quality data from the current discrete federal, state, local, and tribal monitoring programs, remote sensing products, and ad hoc air quality sensor manufacturer's public web portal data are the basis for wildland fire smoke exposure and health assessment research (Chapter 6) and deterministic air quality model development and validation (Chapter 7). This chapter will also describe the current availability of air quality monitoring data, the relative accuracy of different types of monitoring instruments, public availability of measurement data, gaps in smoke monitoring capabilities, and the challenges of ambient smoke monitoring. It will also provide recommendations to improve future ambient monitoring and data curation efforts to better characterize the impact of wildland fire smoke on air quality.

4.2 OBJECTIVES OF AIR QUALITY MONITORING

4.2.1 REGULATORY COMPLIANCE

The Clean Air Act (CAA) requires the U.S. Environmental Protection Agency (U.S. EPA) to protect public health and welfare by promulgating National Ambient Air Quality Standards (NAAQS) for common harmful pollutants. U.S. EPA and its partners at state, local, and tribal monitoring agencies manage several routine regulatory monitoring networks. Each of these ambient air monitoring networks have regulatory requirements and policy objectives that dictate decisions on the location and pollutants measured at each site. The implementation of the network objectives results in monitoring sites that are predominantly concentrated in larger population centers where anthropogenic air pollution sources are concentrated (Figure 4-1, panel a). The relatively high cost of establishing and maintaining regulatory monitoring sites limits their overall numbers and reach into smaller and more remote communities. The Code of Federal Regulations [CFR; (U.S. EPA, 2016)] requires the use of U.S. EPA designated Federal Reference Method (FRM) or Federal Equivalent Method (FEM) instruments for regulatory NAAQS compliance monitoring. However, some flexibility is provided to monitoring agencies in using nonregulatory PM measurements when reporting the U.S. EPA Air Quality Index (AQI) as detailed in 40 CFR Appendix G to Part 58. Although there are efforts by individual state, local, and tribal monitoring agencies, U.S. EPA currently has no national air quality monitoring programs specifically designed to evaluate the impact of wildland fire smoke on air quality. There are no national programs designed to require the establishment of new sites in smoke-prone areas, no grant opportunities to otherwise encourage optional supplemental smoke monitoring, and no program to evaluate the performance of designated FRM/FEM monitoring instruments for smoke. As a result, even though U.S. EPA and its state, local, and tribal partners developed and maintain a relatively advanced set of regulatory air monitoring networks, remote wildland firefighter camps and smaller population centers affected by smoke in most instances lack adequate observational air quality data, and in those instances where regulatory monitors are present, the accuracy of the reported smoke-impacted air pollution data is uncertain (Landis et al., 2018; Long et al., In Press).



AQI = Air Quality Index; CARB = California Air Resources Board; FEM = Federal Equivalent Method; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm; USFS = U.S. Forest Service.

Note: AQI categories defined by colors as depicted in Table 4-1.

Figure 4-1 AirNow Fire and Smoke website display for October 7, 2020 for layers of $PM_{2.5}$ monitors across central California and their associated AQI category for (a) regulatory FEM instruments (circles), (b) with additional CARB and USFS temporary monitors (triangles), and (c) with the addition of PurpleAir sensors (squares).

4.2.2 PUBLIC REPORTING OF AIR QUALITY THROUGH THE AIR QUALITY INDEX (AQI)

The CAA also requires U.S. EPA to establish a uniform AQI for reporting of air quality for CO, nitrogen dioxide (NO_2), O_3, $PM_{2.5}$, particulate matter with a nominal aerodynamic diameter less than or equal to 10 µm (PM_{10}), and sulfur dioxide (SO_2). AQI values (0−500) are calculated individually for each of the five major air pollutants and are based on measured or forecast concentrations. The single AQI value reported on the multiagency AirNow represents the current highest individually calculated pollutant value (NowCast AQI) and is used to communicate how clean or polluted the air is and as a guidance for planning outdoor activities (Table 4-1). The specific colors associated with each AQI level of concern "Good" (green) through "Hazardous" (maroon) was established by U.S. EPA for public communication consistency (U.S. EPA, 2018). During wildland fire smoke events, $PM_{2.5}$ is typically the primary pollutant responsible for the elevated AQI values and the specific suggested intervention strategies to lower population $PM_{2.5}$ exposures to smoke and resulting negative health outcomes.

Table 4-1 Understanding the U.S. Environmental Protection Agency Air Quality Index (AQI): An example for $PM_{2.5}$.

Level of Concern Air Quality Conditions Are:	AQI Color as Symbolized by This Color:	Value of Index When the AQI Is in This Range:	$PM_{2.5}$ (µg/m^3) with a 24-h Concentration of:	PM_{10} (µg/m^3) with a 24-h Concentration of:
Good	Green	0−50	0.0−12.0	0−54
Moderate	Yellow	51−100	12.1−35.4	55−154
Unhealthy for sensitive groups	Orange	101−150	35.5−55.4	155−254
Unhealthy	Red	151−200	55.5−150.4	255−354
Very unhealthy	Purple	201−300	150.5−250.4	355−424
Hazardous	Maroon	301−400 401−500[a]	250.5−350.4 350.5−500.4	425−504 505−604

µm/m^3 = micrograms per cubic meter; AQI = Air Quality Index; h = hour; PM = particulate matter; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm; PM_{10} = particulate matter with a nominal aerodynamic diameter less than or equal to 10 µm; SHL = significant harm level.

[a]An index value of 500 represents the SHL. SHLs are those ambient concentrations of air pollutants that present an imminent and substantial endangerment to public health or welfare, or to the environment, as established in 40 CFR 51.151 (U.S. EPA, 2001). For PM there is only a published SHL for PM_{10}.

State, local, and tribal agencies regularly monitor and report their air quality data to U.S. EPA for the calculation of AQI. However, most monitoring agency networks are designed around urbanized areas known as core-based statistical areas (CBSAs). These networks are typically designed to evaluate the pollution exposure associated with anthropogenic pollution sources under meteorological conditions of pollution maxima as required by the CFR (U.S. EPA, 2015c). Air pollutant monitoring networks such as for $PM_{2.5}$ also include upwind, downwind, and transport sites for each state. State, local, and tribal agencies report all available air monitoring data for calculation of the AQI as well as to understand transport into and out of their monitored jurisdictions. However, a major limitation of many state, local, and tribal agency networks is that the design requirements associated with urbanized areas concentrate sites inside of major population centers. The focus on urbanized areas as well as the large geographical footprint of unmonitored rural areas often results in very limited or no monitors in areas adversely affected by wildland fire smoke. Specifically, a recent U.S. Government Accountability Office report found that 2,120 of the 3,142 counties (67.5%) in the U.S. had no regulatory monitor (GAO, 2020).

AQI values for $PM_{2.5}$ and O_3 presented on the AirNow website, mobile application, or widget that are titled “current air quality” are calculated using the U.S. EPA NowCast algorithms. The full AQI is based on averaging times used for the NAAQS: 24-hour local midnight-to-midnight average for $PM_{2.5}$ and PM_{10}; maximum 8-hour average for CO and O_3; and maximum 1-hour average for NO_2 and SO_2. The

NowCast algorithm is complex but designed not only to approximate the full AQI but also to be more responsive to recent data trends and to be calculable after each new hour's data (U.S. EPA, 2020d). The NowCast algorithms dynamically scale the duration of past hourly monitoring data used to calculate the Current AQI based on the observed temporal trend of ambient concentrations, using longer time averages during stable concentrations and shorter time averages when air quality is changing rapidly. In practice the algorithms often approximate a 3-hour running average. The hourly updated NowCast $PM_{2.5}$ and O_3 AQI values are useful during wildland fire events when downwind ambient air quality can change abruptly but do not necessarily reflect up-to-the-minute current conditions.

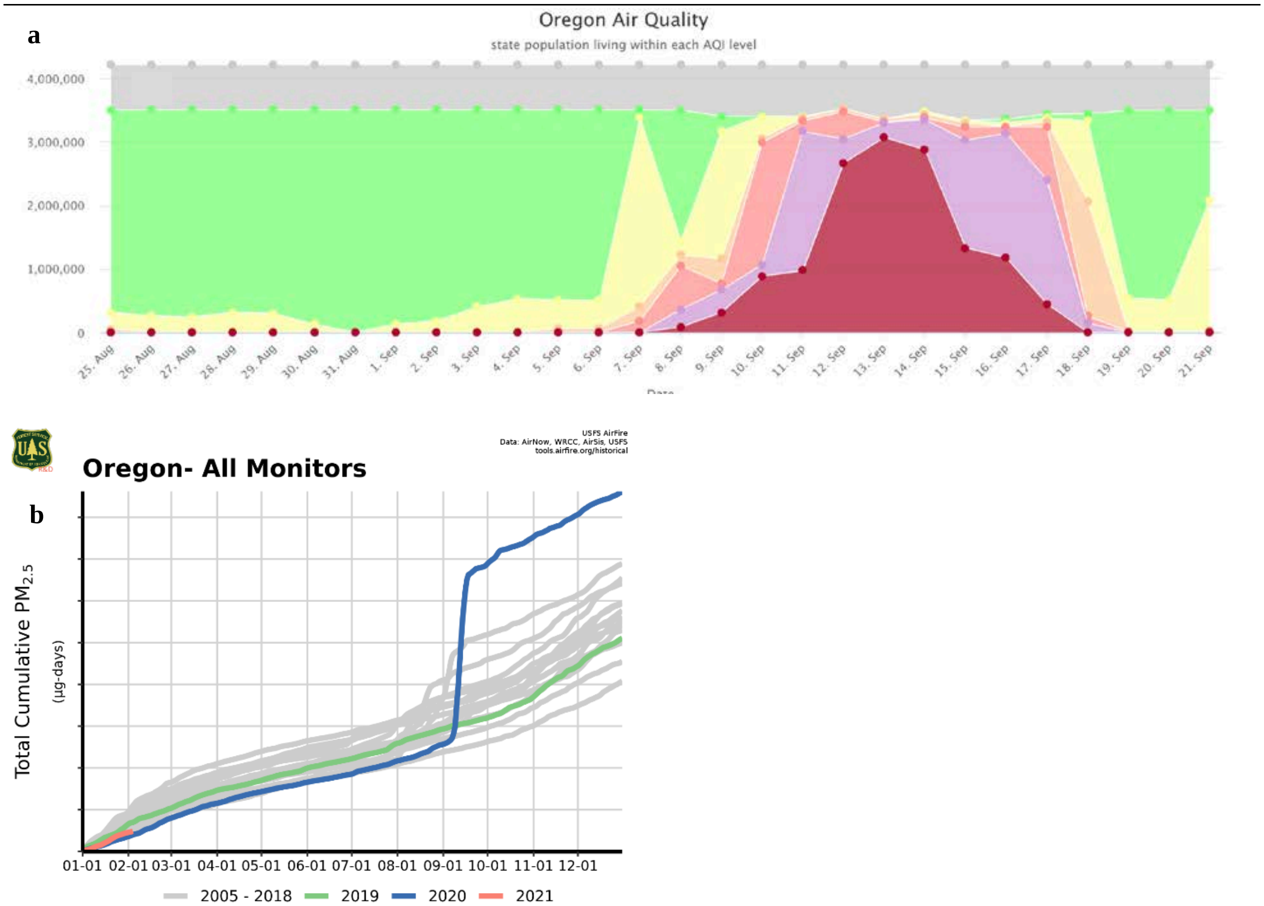
The U.S. EPA and U.S. Forest Service (USFS) have partnered to develop the AirNow Fire and Smoke Map [https://fire.airnow.gov/; AirNow (2021b)] through a pilot project that incorporates temporary monitors (Figure 4-1, panel b) and, beginning in 2020, air quality sensor data (Figure 4-1, panel c), initially from PurpleAir $PM_{2.5}$ measurements (PurpleAir, 2021), to provide spatially improved AQI and associated public health messaging during wildfire season. The associated public health messaging on the site is augmented through direct access to the Interagency Wildland Fire Air Quality Response Program (IWFAQRP) daily smoke outlooks for specific incident-impacted areas. PurpleAir and similar commercially available air quality sensors have just started to be evaluated under high concentration smoke conditions in laboratory and field studies. These evaluations demonstrate the sensors' variable accuracies under different smoke conditions but highlight their potential for providing timely public health messaging during wildland fire smoke events after calibration of reported raw results (Landis et al., 2021; Delp and Singer, 2020; Holder et al., 2020; Mehadi et al., 2019).

4.2.3 ANALYZING AIR QUALITY TRENDS

In addition to directly informing AQI, regulatory network air quality data collected from fixed state, local, and tribal monitoring stations with at least several years of data allow for the characterization of air quality trends and provide context for understanding wildland fire smoke conditions. U.S. EPA maintains an annual air trends report in the form of an interactive web application [e.g., https://gispub.epa.gov/air/trendsreport/2020/; U.S. EPA (2020g)]. The online report features a suite of visualization tools that allow the user to:

- Learn about air pollution and how it can affect our health and environment.
- Compare key air emissions to gross domestic product, vehicle miles traveled, population, and energy consumption back to 1970.
- Review how the number of days with unhealthy air has dropped since 2000 in 35 major U.S. cities.
- Explore how air quality and emissions have changed over time for each of the common air pollutants.
- Review air quality trends where they live.

Information about long-term air quality trends can be useful in determining the extent to which air quality management strategies are helping reduce concentrations of pollutants to the levels specified by the NAAQS. Online resources are also available that present daily trends in air quality during wildland fire events that can be used to estimate daily (Figure 4-2, panel a) and year-to-date (Figure 4-2, panel b) population exposure [https://tools.airfire.org; regional air quality and historical tools; USFS (2021b)]. The AirFire resource can be used to contextualize the current air quality conditions during large wildland fire events and the dramatic impact on population $PM_{2.5}$ exposure like that from a September 2020 wildfire event in the state of Oregon presented in Figure 4-2, panel b.



$PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm; USFS = U.S. Forest Service; WRCC = Western Regional Climate Center.

Colors shown in Figure 4-2, panel a are U.S. EPA AQI categories (Table 4-1) and gray indicates no data.

Source: https://covid.airfire.org/tracking/. IWFAQRP (2021); site uses U.S. EPA regulatory monitor data and an analysis of LANDSCAN population data within 20 km of each monitoring site.

Figure 4-2 Tracking of Air Quality Index (AQI) in Oregon during the 2020 wildfire season (a) and the cumulative annual population exposure to $PM_{2.5}$ in Oregon from 2005—2021 showing the impact of wildland fire events (b).

4.2.4 INFORMING FIRE MANAGEMENT

Smoke from wildland fires can affect the health and safety of fire personnel and the public, interfere with fire suppression operations and transportation, and disrupt local economies (USFS, 2020a). Because of the scale of these smoke-specific effects, such as those seen during the 2020 and 2021 western U.S. wildfire seasons, smoke can become the focus of fire managers, air quality regulators, and public health officials. During many large wildfire incidents, the USFS-led IWFAQRP augments long-term regulatory monitoring networks with temporary nonregulatory air quality monitors dispatched with Air Resource Advisors [ARAs; Figure 4-1, panel b; USFS (2020b, 2020a)] to provide real-time information on air quality to assist local officials and communities in making informed decisions to minimize their exposure to smoke. IWFAQRP uses emergency deployable air quality monitoring equipment, state-of-the-art wildland fire smoke dispersion models, and ARAs for dispatch to ongoing wildfires to develop and publicly disseminate smoke information (USFS, 2020a). ARAs are technical specialists that deploy nationwide to large wildfires to assist with understanding and predicting the effect of smoke on local communities and fire personnel. They work on incident management teams with their public information, fire behavior, and fire weather specialists, as well as coordinate with local emergency response, air regulatory, and public health agencies to provide timely smoke outlooks that address the public health risks and concerns from smoke (USFS, 2020a). ARAs are also a point of contact for the public and commonly present smoke information at public meetings and address smoke-related concerns of local citizens. In addition, USFS regional offices, states, local, and tribal agencies also maintain and deploy nonregulatory samplers for monitoring smoke from wildfires and prescribed burns. However, the cost, technical expertise required, and need for electrical power/data telemetry infrastructure generally limits the number and location of temporary nonregulatory monitors that are deployed.

Smoke from prescribed fires also present significant challenges to land management agencies. Prescribed fire is an important tool for achieving key management objectives such as ecosystem restoration and maintenance and hazardous fuel reduction. Smoke management concerns are among the top impediments to prescribed burning [Melvin (2018, 2015); see Chapter 3]. While nuisance smoke is the most common smoke issue, prescribed fires can subject local communities and sensitive populations to unhealthy levels of $PM_{2.5}$ (Melvin, 2018, 2015). Prescribed fire smoke can also endanger public safety by reducing visibility on roadways leading to serious and/or fatal traffic accidents (Bartolome et al., 2019; Ashley et al., 2015). Additionally, unlike wildfires, prescribed fires are considered a controllable emission source and the resultant smoke can trigger a regulatory violation of NAAQS. However, the 2016 Exceptional Events Rule states that prescribed fire on wildland can be a human-caused event eligible for treatment as an exceptional event, and properly managed prescribed fires are generally less likely than wildfires to cause or contribute to an exceedance or a violation of the NAAQS (U.S. EPA, 2019d). In instances where smoke from a prescribed fire leads to an exceedance or violation of a NAAQS, and all rule criteria are satisfied, air agencies or federal land managers can prepare an exceptional events demonstration and request the event-influenced data to be excluded from the data set used for certain regulatory determinations. To help mitigate these deleterious effects of smoke and obtain observational data to improve smoke management techniques and tools, land management agencies and atmospheric researchers have increasingly begun to deploy temporary smoke monitors as part of operational prescribed burns (Pearson, 2021).

4.2.5 QUANTIFYING THE IMPACT OF WILDLAND FIRES ON AIR QUALITY

One of the key objectives of the U.S. EPA regulatory air monitoring program is quantifying the effect of specific anthropogenic sources on NAAQS pollutant concentrations. However, there are no existing national monitoring programs specifically designed to evaluate the effect of wildfires or prescribed fire programs on air pollutant concentrations even though the U.S. EPA National Emissions Inventory (NEI) has reported that wildland fires contributed a substantial amount to the total national annual CO (30−43%) and $PM_{2.5}$ (32−44%) emissions from 2011−2017 (U.S. EPA, 2021b). However, it remains unclear how emissions of these pollutants from wildland fires translate to overall contributions to annual ambient concentrations. To date, U.S. EPA has not undertaken a national measurement-based integrated assessment to examine the effect of wildland fire emissions on (1) ambient air quality, (2) regulatory NAAQS compliance, or (3) human health. There are numerous local/regional assessments in the scientific literature that document the deleterious changes on ambient air quality, human exposures, and human health outcomes related to specific wildfire events (Stowell et al., 2019; Landis et al., 2018; Reid et al., 2016; Cisneros et al., 2012; Rappold et al., 2011). Incremental progress in the examination of the local/regional effect of wildland fire smoke on ambient air quality, human exposure, and human health effects is being made (Johnson et al., 2020). However, in the absence of a national measurement-based assessment, the full impact of wildfire smoke remains largely unknown on a national scale, particularly at population centers that are often distant from wildfire events.

4.3 AMBIENT AIR QUALITY MONITORING CAPABILITIES

4.3.1 OVERVIEW

The fundamental understanding of wildland fire source emission estimates, the impacts of smoke on air quality, human exposures and health outcomes, and the ability to develop and validate predictive deterministic air quality models, are predicated on accurate measurements of air pollutants in smoke. Although there are no U.S. EPA national air quality monitoring networks focused on wildland fire smoke, there are several discrete federal, state, local, and tribal monitoring programs; remote sensing platforms; and ad hoc air quality sensor networks that provide critical observational air quality data during wildland fire events. This section summarizes current national regulatory ambient air quality measurement infrastructure, nonregulatory temporary incident response measurement capabilities, air quality sensor capabilities, and remote-sensing products and their utility in estimating the impact of wildland fire smoke on air quality.

4.3.2 U.S. EPA ROUTINE REGULATORY MONITORING NETWORKS

U.S. EPA has established NAAQS for the criteria pollutants and maintains multiple national regulatory air pollution monitoring networks as required by the CAA that are set forth in Title 40, Part 50 of the Code of Federal Regulations (U.S. EPA, 2016). Each monitoring network has associated regulatory requirements and policy objectives that dictate decisions on the location of and pollutants measured at each site. In addition to reporting the AQI in large population centers, key monitoring objectives include NAAQS compliance, trend analysis, quantifying specific source impacts, and improving the performance of air quality forecast models. National monitoring of air quality is accomplished through a partnership of U.S. EPA delegated federal, regional, state, city, and tribal stakeholder organizations. U.S. EPA regulatory monitoring is carried out as part of a national network of approximately 4,400 monitoring sites, called the State and Local Air Monitoring Stations (SLAMS). The air quality data obtained from these sites are reported to U.S. EPA's Air Quality System (AQS) database, along with other information, and are used for determining compliance with the NAAQS, assessing effectiveness of State Implementation Plans (SIPs) in addressing NAAQS nonattainment areas; characterizing local, state, and national air quality status and trends; and associating health and environmental damage with air quality levels/concentrations.

To ensure the accuracy, integrity, and uniformity of the SLAMS air quality monitoring data collected, the U.S. EPA has established one or more FRMs for measuring each of the six criteria pollutants. These FRMs are set forth in appendices to 40 CFR Part 50 and specify a measurement technique and other design requirements to be implemented in commercially produced monitoring instruments (U.S. EPA, 2020f, h, i, j, k, 2011a, b, c). These monitoring instruments must also be shown to meet specific performance requirements detailed in the U.S. EPA regulations (U.S. EPA, 2019a), in which case the instrument may be designated by the U.S. EPA as an FRM analyzer. To encourage innovation and development of new air quality monitoring methods, the U.S. EPA has also provided for FEMs. An FEM is not constrained to the specific measurement technique or design requirements of the corresponding FRM. However, an FEM must meet the same or similar performance requirements as specified for the corresponding FRM, and in addition, it must show a high degree of comparability to collocated FRM measurements at one or more field testing sites under typical ambient conditions (U.S. EPA, 2019a). A monitor that is shown to meet all applicable requirements may be designated by the U.S. EPA as an FEM monitor. A current listing of all designated FRMs and FEMs as of December 2020 can be found at https://www.epa.gov/sites/production/files/2019-08/documents/designated_reference_and-equivalent_methods.pdf (U.S. EPA, 2020e).

The siting criteria and regulatory monitoring methodologies are briefly described here and a detailed discussion of specific air pollution networks, criteria air pollutants measured, and measurement methods are provided in Appendix A.4.2 ($PM_{2.5}$ Mass Monitoring), Appendix A.4.3 ($PM_{2.5}$ Speciation Monitoring), and Appendix A.4.4 (Criteria Gas Monitoring). The U.S. EPA $PM_{2.5}$ monitoring program is the largest component of the national monitoring infrastructure and $PM_{2.5}$ monitors are mostly sited in urban areas at the neighborhood scale as defined in 40 CFR Appendix D to Part 58 (U.S. EPA, 2015a, b), where typical $PM_{2.5}$ concentrations are reasonably homogeneous throughout an entire subregion in the absence of wildland fire smoke. There are four main components of the U.S. EPA $PM_{2.5}$ monitoring program: 24-hour integrated filter-based FRM samplers, continuous FEM mass instrument measurements reported as 1-hour concentrations, continuous non-FEM mass instrument measurements reported as 1-hour concentrations, and 24-hour integrated filter-based Chemical Speciation Network (CSN) samplers. Continuous $PM_{2.5}$ FEM and criteria gas FRM/FEM (Appendix A.4.1) real-time data support NAAQS compliance and is integrated with non-FEM continuous $PM_{2.5}$ data to support public AQI communication and air quality smoke forecasting on AirNow (AirNow, 2021a). The top three states using the non-FEM continuous $PM_{2.5}$ instruments for AQI reporting are Washington (n = 47), Oregon (n = 45), and California (n = 43) primarily to communicate changes in air quality from wildland fire smoke.

4.3.3 TEMPORARY/INCIDENT RESPONSE MEASUREMENTS

The IWFAQRP provides significant incident response smoke monitoring capabilities by maintaining a cache of smoke monitoring equipment for nationwide deployment by their ARA personnel. The IWFAQRP smoke monitor cache consists of ~40 Met One Instruments (Grants Pass, OR) E-SAMPLER and E-BAM nonregulatory $PM_{2.5}$ samplers (USFS, 2020a). These $PM_{2.5}$ samplers are available upon request to land management agency administrators and incident management teams for monitoring wildfires and to federal land managers conducting prescribed burns. The monitors are typically used by ARAs supporting wildfire incident management teams. The ARA will often consult with local land managers, air quality regulatory agencies, and public health officials for guidance on positioning temporary smoke monitors. When siting monitors, ARAs attempt to meet the same siting criteria as used for regulatory FRM/FEM monitors like avoiding interferences from other emission sources or physical barriers that may obstruct air flow around the sampler (U.S. EPA, 2012). Deployment of either the E-BAM or E-SAMPLER requires a landline power hookup, but because these monitors are intended to inform communities, this infrastructure requirement is not typically an issue as they are often deployed at fire stations, schools, or other municipal buildings. The E-BAM and E-SAMPLER both upload their measurement data by satellite to a cloud-based data acquisition system where it is reported on an hourly basis. The data must then pass a quality assurance (QA) check before being publicly distributed through the www.airfire.org or the fire.airnow.gov websites (see Section 4.4.3).

Beyond the national smoke monitoring resources offered by IWFAQRP, jurisdictions within federal agencies (e.g., USFS regions), states, and tribal agencies also maintain and deploy PM samplers for monitoring smoke from wildfires and prescribed burns. The PM samplers used include ThermoFisher Scientific (Franklin, MA) DataRAM pDR-1500 and Met One Instruments BAM-1020, E-BAM, and E-SAMPLER systems (USFS, 2020b). Remote telemetry and satellite data transmission are used to gather and present the raw data in “near-real-time” (USFS, 2020b). These data are also collected and integrated into the fire.airnow.gov website (see Section 4.4.3). Several states have programs to monitor smoke from wildland fires. The most extensive program is in California, where the California Air Resources Board (CARB) and local air districts have >100 E-BAM samplers available for deployment to monitor smoke (Pearson, 2021). The CARB program initially targeted wildfires but was expanded in response to state legislation (California SB-1260, 2018) which sought to increase hazardous fuels reduction and included funding for prescribed fire smoke monitoring. Other states with monitoring efforts for wildfire and prescribed fire include Alaska, Arizona, Colorado, Idaho, Montana, Nevada, New Mexico, Oregon, and Washington, as well as some tribes (Appendix A.4.1). Even with the deployment of temporary PM monitors supplied by federal, tribal, state, or local agencies, gaps in air quality observations often persist, especially in lower population foothill and mountain communities. To address monitoring gaps, ARAs began deploying PM sensors (PurpleAir) in 2020 to estimate the air quality in communities that previously would have gone without monitoring. Likewise, states and local agencies are augmenting existing monitoring networks with air quality sensors. In 2018, CARB initiated a pilot program that distributed several hundred PM sensors to augment existing air quality monitoring networks and capabilities (Pearson, 2021).

Historically, smoke monitoring for wildfire response has relied on existing and temporarily deployed stationary monitors to provide air quality information to incident command and state/local public health officials. The relatively small number of local monitors, the dynamic nature of smoke emissions and transport, and the dispersed nature of firefighting personnel and downwind communities make predicting exposures challenging. Mobile monitoring capabilities are another strategic approach to measure and communicate real-time smoke information. Personal monitoring of firefighters, vehicle-mounted instruments, and airborne drones are all viable mobile monitoring platforms. These approaches have been used in focused research studies (Apte et al., 2017; Navarro et al., 2016; Villa et al., 2016) but not for routine operations. Research into both built-for-purpose and commercially available small-form-factor air quality measurement systems have been reported by Landis et al. (2021) and Holder et al. (2020), respectively; and others are working on continuous reading mobile air quality platforms (2B Tech, 2021; Mui et al., 2021; Apte et al., 2017).

Except for the BAM-1020, monitors used by federal, state, local, and tribal agencies for temporary smoke monitoring are not expected to be U.S. EPA designated FEM monitors. Across all agencies the most frequently deployed monitors are E-BAM and E-SAMPLERS. The performance of both samplers in measuring $PM_{2.5}$ in fresh smoke has been evaluated in limited laboratory testing which indicates high correlation ($r^2 > 0.9$) and relatively low bias range for the E-BAM (1−21%) and E-SAMPLERS (8−18%) compared with reference FRM/FEM monitors across concentration ranges of 20−1,700 μg/m^3 (Mehadi et al., 2019; Trent, 2006, 2003). However, it is unclear how the samplers perform across the natural range of smoke properties (chemical composition, size distribution) and how performance may vary over extended periods of sampling in smoke-impacted environments. Additionally, federal interagency smoke monitor inventories also include DataRAM monitors, and laboratory evaluation of these monitors indicates they overestimate $PM_{2.5}$ in smoke by a factor of ~2 (Trent, 2003).

4.3.4 SENSORS

Over the last decade there has been rapid development of miniaturized, user-friendly air quality sensor systems (Karagulian et al., 2019; Malings et al., 2019; Baron and Saffell, 2017; Williams et al., 2015). Significant advancements in internal gas and PM sensor components, compact microprocessors, power supply/management, wireless data telemetry, advanced statistical data fusion/analysis, real-time sensor calibration, and graphical data interfaces hint at the future potential of accurate small form factor integrated air quality sensor systems. This technology is being developed for a variety of potential applications, including human exposure assessment (Morawska et al., 2018), industrial emissions (Thoma et al., 2016), local source impact estimation (Feinberg et al., 2019), and to increase the spatial density of outdoor monitoring networks (Bart et al., 2014; Mead et al., 2013). Some manufacturers of air quality sensor systems have built cloud-data systems and public websites to host measurement data and allow public access (2B Tech, 2021; Kunak, 2021; PurpleAir, 2021). The large number of installed sensors and centralized data hosting capabilities of PurpleAir (Draper, UT) led the U.S. EPA and USFS to launch a pilot project to provide data from air quality sensors calibrated with U.S. EPA's correction equation (Barkjohn et al., 2020) and the derived AQI and NowCast on the AirNow Fire and Smoke Map. The goal of the pilot project was to provide additional AQI ($PM_{2.5}$ only) information during wildfires in those areas not adequately served by regulatory monitoring sites (AirNow, 2021a).

However, the reliability, accuracy, stability, and longevity of many types of air quality sensors under smoke conditions is largely unknown. Routine performance testing of many air quality sensors, to date, has been mostly limited to typical ambient conditions (Collier-Oxandale et al., 2020; Zamora et al., 2019; Feinberg et al., 2018; Jiao et al., 2016; Williams et al., 2015), with more limited assessment of certain technologies at higher ambient concentrations (Johnson et al., 2018; Zheng et al., 2018). These previously published findings have indicated, in some cases, high correlation between collocated sensors and FRM/FEM reference monitors; however, there are also many sensor test results that exhibit measurement artifacts (Hossain et al., 2016; Lin et al., 2015; Spinelle et al., 2015; Mead et al., 2013), inconsistency between identical sensors (Sayahi et al., 2019; Castell et al., 2016; Williams et al., 2015), drift over time (Sayahi et al., 2019; Feinberg et al., 2018; Artursson et al., 2000), sensitivity to environmental conditions [e.g., temperature, relative humidity; Wei et al. (2018); Cross et al. (2017)], and limitations to upper-limit measurement capabilities (Zou et al., 2020; Schweizer et al., 2016). U.S. EPA has endeavored to improve the reliability, consistency, and accuracy of air quality sensor data by regularly engaging academia, industry, nonprofit groups, community-based organizations, and regulatory agencies to develop recommendations, performance targets, and best practices (Duvall et al., 2021a, b; Williams et al., 2019; Clements et al., 2017). U.S. EPA has also created an online Air Sensor Toolbox (U.S. EPA, 2021a) as a clearinghouse for information on the use of air quality sensors. The U.S. EPA Air Sensor Toolbox aims to improve the operation, data collection, and quality assurance of air sensor data by providing resources such as the *Air Sensor Guidebook*, Air Sensor Standard Operating Procedures (SOPs), Air Sensor Performance Targets and Test Protocols, Air Sensor Collocation Instrument Guide, Sensor Evaluation Report, Quality Assurance Handbook and Guidance Documents for Citizen Science

Projects, and air sensor loan opportunities to enable the public to learn about air quality in their communities (U.S. EPA, 2020b).

More recently U.S. EPA partnered with other U.S. federal agencies (Centers for Disease Control and Prevention [CDC], National Aeronautics and Space Administration [NASA], National Park Service [NPS], National Oceanic and Atmospheric Administration [NOAA], USFS) to sponsor the Wildland Fire Sensor Challenge to advance wildland fire air measurement technology to make measuring instruments easier to deploy, suitable to use for high concentration events, durable to withstand difficult field conditions, and have the ability to report high time-resolution data continuously and wirelessly (Landis et al., 2021). The Wildland Fire Sensor Challenge encouraged innovation worldwide to develop sensor prototypes capable of measuring $PM_{2.5}$, CO, carbon dioxide (CO_2), and O_3 during wildfire episodes. The raw $PM_{2.5}$ sensor accuracies of the three winners ranged from ~22−32%, while smoke-specific U.S. EPA regression calibrations improved the accuracies to ~75−83%, demonstrating the potential of these systems for providing reasonable accuracies over conditions that are typical during wildland fire events (Landis et al., 2021). Selected commercially available $PM_{2.5}$ sensors have also been evaluated under smoke conditions with collocated FEM measurements , and the results of these analyses highlight the potential use of such sensors in supporting the development of public health messaging during wildland fire smoke events (Delp and Singer, 2020; Holder et al., 2020; Mehadi et al., 2019). These research studies like the Wildland Fire Sensor Challenge have shown that raw $PM_{2.5}$ sensor data requires post-processing using smoke-specific calibration functions to account for differences in aerosol chemistry, particle size distribution, aerosol density, and optical properties. The range of smoke-specific calibration correction factors are summarized in Appendix Table A.4-2 and show a broad range of responses depending on specific fire conditions and reference instruments used. As more information is gathered on the performance and calibration of air quality sensors in wildland fire smoke, the utility of their reported air quality measurements for informing public health messaging will improve. These sensors present the opportunity to qualitatively improve the assessment of the spatial variability of wildland fire smoke due to their ability to be deployed in large numbers [Figure 4-1, panel c; 2B Tech (2021); Clarity (2021); PurpleAir (2021); Gupta et al. (2018)].

4.3.5 REMOTE SENSING/SATELLITE DATA

Remote sensing is the science of acquiring information about the earth's surface or atmosphere without actually being in contact with it and requires a source of reflected, emitted, or absorbed and re-emitted energy which interacts with the geophysical parameter being measured, such as aerosols (e.g., $PM_{2.5}$) in the atmosphere. As a result, remote sensing allows for the estimation of wildfire smoke in areas of the country that lack other sources of ground-based observational data (Wu et al., 2018; Krstic and Henderson, 2015; Mei et al., 2012; Liu et al., 2009). The two types of remote sensing are referred to as passive and active. Passive remote sensing uses the sun as the energy source, whereby the solar radiation is reflected by the earth's surface or scattered in the atmosphere (for visible wavelengths) or absorbed and then re-emitted from the earth's surfaces (for thermal infrared wavelengths). Because measurements made in the visible wavelengths require reflected solar radiation, they can only be conducted during daylight hours. Active remote sensing techniques require their own energy source, where the emitted radiation is directed at the target of interest and reflected back to the instrument. In active remote sensing, lasers often provide this energy source, and the pulsing energy can provide information on three-dimensional structure of the geophysical variable being measured; this technique is called Light Detection and Ranging (LiDAR).

4.3.5.1 SATELLITE MEASUREMENTS

Satellite observations from both low earth and geostationary orbit have become the source of important sets of measurements for monitoring air pollutant abundances and transport across large spatial scales. Instruments aboard low-earth-orbit satellites most often provide once-a-day observations over the region of interest and with large swaths provide global coverage every day. Geostationary satellites are in fixed-orbit position relative to the earth's surface and are used to observe phenomena that require high temporal-resolution observations, such as severe weather and disasters like wildland fires. Most satellite instruments used to provide information on air quality are passive remote-sensing instruments and span wavelengths in the ultraviolet–visible (UV/VIS) range or the thermal infrared. The visible wavelengths are used to provide true color imagery which can be used to identify smoke from wildland fires in a cloud-free scene. More quantitative measurements involve the measure of backscatter radiances in the UV/VIS or thermal infrared emission through the atmosphere to provide a derived geophysical column measurement dependent on physics-based retrieval algorithms (Martin, 2008). Over the past two decades, satellite column measurements are increasingly being used to provide near-surface information on pollutants such as $PM_{2.5}$, O_3, NO_2, SO_2, CO, and formaldehyde (CH_2O). Polarimetric, multispectral, multidirectional, and active remote-sensing observations provide information on the aerosol amount, size, type, and vertical distributions of column abundances of the geophysical parameter of interest.

One of the major challenges of passive remote measurements from satellites is resolving the vertical distribution of the parameter of interest, and in some cases the sensitivity of the satellite measurement to the lowermost atmosphere, which is the region with substantial variability and the most relevant for gaining an understanding of ambient air quality and subsequent public health impacts (Martin, 2008). Nevertheless, it is well documented that satellite-derived geophysical parameters of column integrated abundances such as aerosol optical depth (AOD) can be used to constrain estimates of near-surface pollution concentration, especially when used in combination with model (chemical transport or statistical)-based predictions, and provide valuable information on the horizontal distribution of the pollutant burdens because of the satellite instrument's synoptic field of view. Smoke plume height characteristics from the Moderate Resolution Imaging Spectroradiometer (MODIS) Multi-Angle Implementation of Atmospheric Correction (MAIAC) algorithm (Lyapustin et al., 2019), which is based on 11-µm absorption of fire-emitted gases in the plume, have shown potential for improving surface $PM_{2.5}$

concentration estimates derived from AOD (Cheeseman et al., 2020). LiDARs aboard satellites have a unique capability of resolving the vertical distribution of aerosols in the atmosphere and can make measurements both day and night (Winker et al., 2010) but have very limited spatial coverage to capture wildland fire plumes (Raffuse et al., 2012). The TROPOspheric Monitoring Instrument (TROPOMI) provides an aerosol height index, which is a developing data product (Griffin et al., 2020), and offers the ability to diagnose the aerosol plume height to help assess if the majority of aerosols seen by satellite are within the boundary layer or being transported aloft.

4.3.5.1.1 CORRECT REFLECTANCE TRUE COLOR IMAGERY—SMOKE PLUME IDENTIFICATION AND TRACKING

Satellite-corrected reflectance from visible wavelengths, also referred to as true color imagery, from both geostationary and low earth (polar) orbiting satellite instruments is one of the basic satellite data products used to identify the spatial extent of smoke plumes and transport from wildland fires. In addition to providing access to true color images from Geostationary Operational Environmental Satellites (GOES), satellite analysts at NOAA develop a daily smoke analysis over the contiguous U.S. and adjacent area of Canada through their Hazard Mapping System (HMS) Fire and Smoke Product website (https://www.ospo.noaa.gov/Products/land/hms.html#maps). The HMS products use multiple data inputs to create a digitized data product displaying the extent of visible smoke and a qualitative classification on the density of the smoke as low, medium, or high based on visible opacity determined by the analysts (NOAA, 2020). In combination with surface observation of pollutants or visibility these imagery-based data products can help identify areas with air quality impacts from wildland fires, but alone, these products provide no information on air quality at the surface.

4.3.5.1.2 SATELLITE (GEOPHYSICAL) COMPOSITION OBSERVATIONS

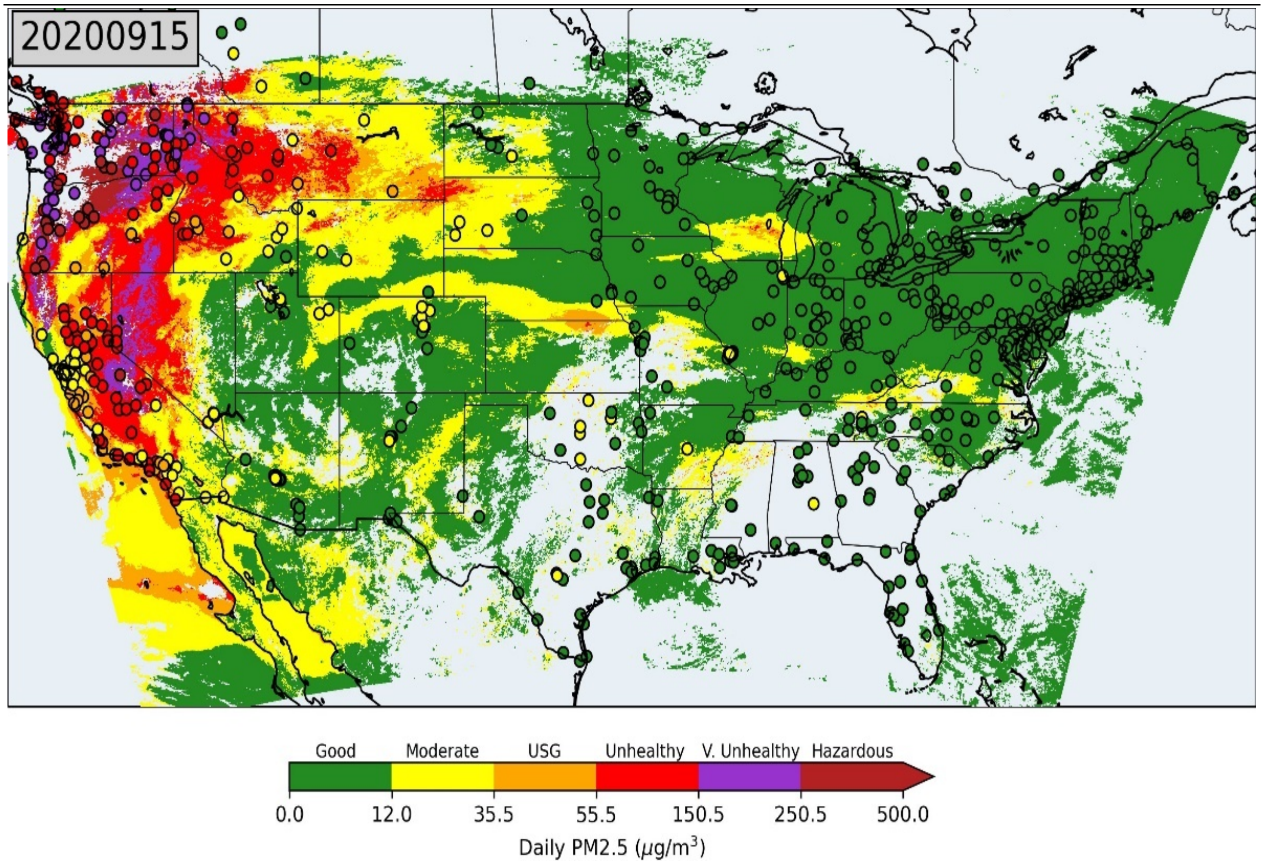
AOD is an integrated measure of extinction through the atmosphere that is the derived geophysical variable from satellite instruments most relevant to $PM_{2.5}$ and PM_{10} mass concentrations. Both operational and research algorithms are used to generate AOD from passive satellite sensors such as MODIS, Multiangle Imaging Spectroradiometer (MISR), Visual Infrared Imaging Radiometer Suite (VIIRS), and the GOES Advanced Baseline Imager (ABI). Deriving near-surface PM concentrations from AOD values is difficult due to uncertainties in the microphysical (intrinsic) properties of the particles, their size distribution, aerosol type, and hygroscopic state, as well as key extrinsic properties, such the vertical profile distribution (Hoff and Christopher, 2009). Few measurements studies have examined the uncertainties associated with the use of AOD measurements to estimate ground-based $PM_{2.5}$. Early results from the NASA DISCOVER-AQ mission over the urban Baltimore region (Crumeyrolle et al., 2014) found that accurate quantification of the aerosol mixed layer height is critical for predicting $PM_{2.5}$ concentrations, with aerosol type variability being of lesser importance. In addition, the results indicate the presence of aerosol layers above the boundary layer introduced significant uncertainties in surface $PM_{2.5}$ concentrations estimates when using a column-integrated AOD measurements, and that active remote-sensing techniques such as LiDARs can provide a characterization of aerosol layers to improve upon the $PM_{2.5}$ estimates. The transport of smoke plumes can often result in stratified aerosol layers, including aerosol layers above the boundary layer, so proper characterization of such aerosol layer structure remains a critical variable in using satellite AOD to predict surface $PM_{2.5}$ concentrations.

Geophysical retrievals of trace gas column abundance from satellite have seen great improvements in spatial resolution over the past several decades with the European Space Agency's (ESA's) Sentinel 5 Precursor TROPOMI launched in 2017 now producing global daily NO_2 observations at a resolution (7 × 3.5 km) consistent with chemical transport modeling used for wildland fire air quality forecasting. In addition to NO_2, TROPOMIs standard trace gas data products include column abundances of CH_2O, SO_2, CO, and methane at varying spatial resolutions (Levelt et al., 2006). The NASA Tropospheric Emissions: Monitoring Pollution (TEMPO) mission, scheduled to launch in mid-2022 into a geostationary orbit, will provide hourly observations of NO_2 and O_3, across the North American continent during daytime (Zoogman et al., 2017).

Because there are significant portions of the U.S. that have no continuous surface monitors, a very active stream of research developed in the early 2000s focused on the use of AOD and trace gas satellite data products to help predict pollutant surface concentrations. Over the past 20 years many research groups have developed a multitude of methods to model surface $PM_{2.5}$ concentrations using AOD from numerous satellites as discussed in a review by Chu et al. (2016) with a primary focus on estimates of annual $PM_{2.5}$ concentrations. Some research groups have continued to improve on the methods over time as inputs in the methods have improved (Hammer et al., 2020), including estimates of PM chemical composition (van Donkelaar et al., 2019), and have made their data sets available for public use (Atmospheric Composition Analysis Group, 2021). Similar research efforts combine chemical transport model and NO_2 column abundances to infer surface concentrations (Cooper et al., 2020). Most of these research efforts focus on predictions of annual means for these pollutants versus daily predictions of surface concentrations. For example, Alvarado et al. (2020) has recently demonstrated the transport and tracking of several trace gases associated with wildland fires in western Canada.

Useful and actionable information for wildland fire pollutants require daily predictions of surface pollutants, such as $PM_{2.5}$, or more temporally resolved information because of the diurnal nature of wildland fire emissions and meteorological transport patterns (Baker et al., 2019). Methods focused on the use of satellite data to aid in daily pollutant surface predictions during wildland fires have been demonstrated on a very limited basis through a case-study approach, and include simple regression analysis (Raffuse et al., 2013), machine learning techniques (Reid et al., 2015), and generalized geographically weighted regression models (Gupta et al., 2018; Gupta and Christopher, 2009), all with moderate success. Resolving the apportionment of AOD impacting the surface concentrations is complicated because of long-range and high-altitude transport of aerosols which often occurs for wildland fire events. Jin et al. (2019) used a geophysical approach to estimate daily surface $PM_{2.5}$ concentrations and conducted a detailed assessment of uncertainties using this approach; estimating uncertainties in the modeled $PM_{2.5}$/AOD led to an error of 1 μg/m^3 in daily $PM_{2.5}$ predictions, and satellite AOD uncertainties produced errors of 8 μg/m^3. However, none of these efforts provided an ongoing source of data and were not associated with surface $PM_{2.5}$ predictions for wildland fires.

The U.S. EPA AirNow Program application called the AirNow Satellite Data Processor [ASDP, Pasch et al. (2013)] integrates AOD from the MODIS instruments (Terra and Aqua) with a focus on improving the accuracy of daily ground-level $PM_{2.5}$ concentrations. The ASDP approach provides $PM_{2.5}$ predictions using climatological scaling factors (van Donkelaar et al., 2019) from GEOS-Chem. The NOAA Aerosol Watch provides access to a variety of relevant GEOS (16 and 17) and VIIRS (S-NPP and NOAA-20) data products, including true color imagery and AOD retrievals that can be overlaid with AirNow $PM_{2.5}$ data to help assess whether the satellite data and surface concentrations are spatially correlated in time and space, which is an indication that the smoke extent observed by the satellite is at or near the surface impacting ground-level air quality. Figure 4-3 is a result of recent efforts by NOAA Aerosol Watch to product an operational daily satellite-derived $PM_{2.5}$ product for September 15, 2020 during the Oregon wildland fires. This approach aggregates VIIRS AOD from two polar orbiting satellites, S-NPP and NOAA-20, and applies a regression algorithm from available surface $PM_{2.5}$ data to produce a daily satellite-derived $PM_{2.5}$ field.



$PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 μm; USG =unhealthy for sensitive groups.

Note: This figure captures spatial extent of poor air quality associated with several large western wildland fire complexes over the western U.S. Closed circles in the plot represent surface monitors of $PM_{2.5}$.

Image Source: Shobha Kondragunta (NOAA/NESIDS).

Figure 4-3 Image of surface Air Quality Index (AQI) for $PM_{2.5}$ from U.S. EPA AirNow overlayed plotted with AQI for $PM_{2.5}$ derived from National Oceanic and Atmosphere Administration (NOAA) aerosol optical depth from Visible Infrared Imaging Radiometer Suite (VIIRS) instruments (Soumi-NPP and NOAA-20 satellites) for September 15, 2020.

4.3.5.2 GROUND-BASED MEASUREMENTS

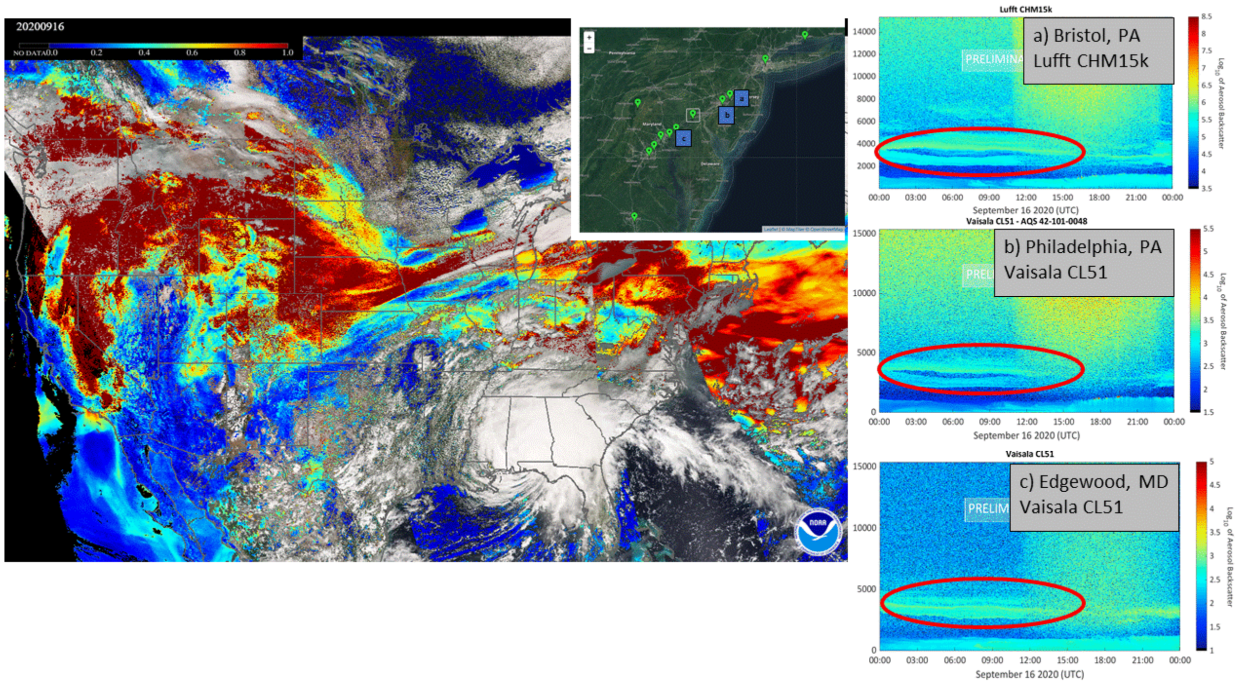
Ground-based remote sensing networks across the U.S. serve a wide range of functions, such as the highly operational surface weather observation stations which contain several remote-sensing instruments in combination with in situ instruments used to provide continuous observations to generate routine weather reports to more research-based networks, such as the NASA Micro-Pulse LiDAR

Network (MPLNET), a small federated network of compact LiDARs designed to measure aerosols and cloud vertical structure and boundary-layer heights. The combination of these networks provides relevant data on surface visibility, the vertical distribution of aerosols, boundary-layer heights, AOD, and total column NAAQS gaseous pollutants.

The combined Automated Surface Observing System (ASOS)/Automated Weather Observing System (AWOS) networks consist of over 1,000 sites across the U.S., with ASOS containing over 900 sites. The primary remote sensing measurement at ASOS/AWOS sites is surface visibility. The visibility measurement uses a forward scatter sensor and detector to measure the attenuation of light by scattering and absorption at the wavelength of 550 nm. The sensor measures a 1-minute average extinction coefficient and reports a 10-minute average. The 550 nm measurement is very sensitive to $PM_{2.5}$ and therefore can be used to understand reduced visibility caused by wildland fire smoke. The ASOS/AWOS sites also used ceilometers for reporting cloud-based heights. Ceilometers are a type of LiDAR, capable of providing vertical profile information on aerosols in the troposphere through their attenuation of backscatter. While NOAA operates a large network of ceilometers as part of ASOS, the instruments are not currently configured to report the aerosol backscatter profiles, which can be used to define aerosol layer heights and derive a mixing layer height/planetary boundary-layer height. Ceilometer technology is being implemented by U.S. EPA Photochemical Assessment Monitoring Station (PAMS) program.

Recent updates to the U.S. EPA PAMS network require the stations to measure and report an hourly mixing-layer height. This measurement requirement was fully implemented in June 2021 and is primarily being satisfied through the installation of ceilometers across the network sites. Although state and local agencies are required to only report an hourly mixing-layer height, a U.S. EPA collaboration with the University of Maryland, Baltimore County (UMBC), NASA, and NOAA is focused on the development of a near-real-time data system to archive and display ceilometer backscatter profiles, aerosol layer heights, and planetary boundary-layer heights (PBLH) from PAMS and non-PAMS ceilometers into the Unified Ceilometer Network [UCN; https://www.ucn-portal.org/; 2021)]. The UCN will use a common algorithm to determine PBLH (Caicedo et al., 2020) and display near-real-time aerosol backscatter vertical profiles, which can be used to track the vertical structure of aerosol plumes, including wildland fire smoke, as the plumes are transported across the U.S. as shown in Figure 4-4.

The NASA MPLNET is a global federated LiDAR network which supports research and the NASA Earth Observing System program (Wielicki et al., 1995). Value-added network data sets are made available to the community via an online repository [http://mplnet.gsfc.nasa.gov; NASA (2021); Campbell et al. (2008)]. The micropulse LiDAR operates at 532 nm in contrast to ceilometers which operate in the 900 nm range or 1,064 nm, which allows the micropulse LiDAR system the benefit of being more sensitive to $PM_{2.5}$.



Note: Aerosol backscatter profiles from ceilometers located at air quality monitoring sites (a) Bristol, PA; (b) Philadelphia, PA; (c) Edgewood, MD show the smoke layer being transported above the boundary layer, with little to no impacts to surface air quality.
Source: Unified Ceilometer Network— https://www.ucn-portal.org/. 2021).

Figure 4-4 Image of western U.S. wildfire smoke transported to the northeastern U.S. as captured in the Visual Infrared Imaging Radiometer Suite (VIIRS) true color image overlayed plotted with VIIRS aerosol optical depth for September 16, 2020.

For over two decades the NASA AErosol RObotic NETwork (AERONET), a federated association of ground-based sun and sky scanning radiometers, has provided high-temporal-resolution measurements of the optical, microphysical, and radiative properties of aerosols. One of the primary data products, columnar AOD, is used as a primary validation resource for satellite validation of AOD. The Angstrom exponent from the measurements can be used to provide an estimate of the dominant aerosol size within the AOD measurement (Giles et al., 2019). Like AERONET, the Pandonia Global Network (PGN) is an emerging federated global network of ground-based spectrometers led by NASA and ESA and was developed to validate trace gas column abundances from satellites such as TROPOMI (Judd et al., 2020; Zhao et al., 2020). The instrument, called Pandora, is a UV/VIS spectrometer and currently provides near-real-time data products of total column O_3 and NO_2, tropospheric NO_2, and a derived NO_2 surface concentration, with tropospheric column CH_2O moving from a research data product to a standard data product in the coming year (Szykman et al., 2019). The number of AERONET and PGN sites across the U.S. can vary on a year-to-year basis as both instruments are often used to support research field campaigns; at the end of 2020, AERONET reported approximately 100 active sites and PGN 14 active sites. The emergence of a ground-based ceilometer network, the UCN; (https://www.ucn-portal.org/), through a collaboration between U.S. EPA, UMBC, NASA, and NOAA, will provide three-dimensional aerosol backscatter profile measurements at numerous sites across the U.S. The ceilometer measurements will allow characterization of the smoke plume heights, including multiple layers, when smoke is transported over the sites.

4.4 AMBIENT AIR QUALITY MONITORING DATA AVAILABILITY AND QUALITY

4.4.1 OVERVIEW

Observational air quality data is used in many facets of wildland fire smoke management from first-responder force protection and public health messaging where real-time data availability is critical, to regulatory NAAQS review and public health research (e.g., epidemiologic studies) where delayed data access is acceptable but rigorous data quality assurance/quality control (QA/QC) review is required. This section discusses observational air quality data availability and relative data quality that is routinely used by wildland fire smoke managers, public health officials, and researchers.

4.4.2 U.S. EPA ROUTINE REGULATORY DATA AVAILABILITY

As described above, near-real-time measurements of $PM_{2.5}$ and O_3 are reported from state, local, and tribal air monitoring agencies to AirNow (Appendix Table A.4-3). The data are then made publicly available through NowCast reporting of the AQI. The raw hourly data for $PM_{2.5}$ and O_3 as well as all other reported real-time air pollution and meteorological parameters are stored and available to the AirNow technical community through the website www.AirNowTech.org. AirNow-Tech is a password-protected website for air quality data management analysis, and decision support. AirNow-Tech is primarily used by the federal, state, tribal, and local air quality organizations that provide data and forecasts to the AirNow system, as well as researchers and other air data users. Automated availability of large amounts of AirNow data can be accomplished by registered users by accessing the AirNow application programming interface. There are important distinctions between the AirNow data system and the AQS database described below. First, to ensure real-time availability of data in AirNow, data are reported as soon as practical after the end of each hour. Therefore, data are available to support forecasting and reporting of the AQI but are not used for regulatory decisions until all QA/QC checks are performed and validation of data is certified by the responsible state/local/tribal agency. Second, data reported to AirNow include many monitoring stations for communities outside the U.S. For example, air monitoring programs for Canadian provinces and cities report their $PM_{2.5}$ and O_3 data to AirNow. However, data from outside the U.S. are usually not reported to the AQS data system described below.

U.S. EPA's long-term repository of data is provided by the AQS. The AQS contains ambient air pollution data collected by state, local, and tribal air pollution monitoring agencies. The data set includes data from both automated methods reported to AirNow, but also from manual methods where data are not available for several weeks to months because of post-sampling laboratory analysis. In addition to pollutant concentrations and meteorological data, AQS contains descriptive information about each monitoring station (including its geographic location and its operator), and data quality assurance/quality control information. Although data are reported to AirNow within minutes after the end of an hour, data are not required to be reported to AQS until 90 days past the end of a calendar quarter. This lag and difference in data reporting allow monitoring agencies the time needed to validate ambient air monitoring data for NAAQS compliance. Data reported to both AQS and AirNow are matched on a routine basis with AQS data, overwriting any reported data to AirNow. This allows monitoring agencies the opportunity to invalidate data in one location while ensuring validation decisions are carried through to both databases. By May 1st of each year, monitoring agencies are required to "certify" (U.S. EPA, 2019c) their criteria pollutant data used for NAAQS compliance determinations so that it is available to use in design value calculations. A user-friendly portal to access reports and data from AQS data is available at: https://www.epa.gov/outdoor-air-quality-data.

4.4.3 TEMPORARY/INCIDENT RESPONSE DATA AVAILABILITY

The temporary $PM_{2.5}$ monitors deployed by federal, state, tribal, and local agencies for incident response typically report hourly data through satellite communications. The AirNow Fire and Smoke Map project, a collaborative effort between IWFAQRP and U.S. EPA collects these data through the AirSis and Western Regional Climate Center (WRCC) data feeds (AirNow, 2021b, c). Following quality assurance checks on different monitor parameters including flow rate, internal humidity, battery levels, as well as whether measurement values are within a acceptable range, the data are made available through the AirNow Fire and Smoke Map. PM data from permanent monitors (Section 4.3.2) obtained through the U.S. EPA AirNow system (Section 4.4.2), as well as sensors (Section 4.4.4), are also included in AirNow Fire and Smoke Map. The system also provides the locations of large fire incidents from the U.S. National Interagency Fire Center's active incident feed and satellite-based active fire detections and smoke plume locations (Section 4.3.5.1.1) from the NOAA Hazard Mapping System. Currently, the system functions as an operational data viewer—data is not available for download and viewing is limited to data <10 days old. Data downloads (<10 days old) for temporary and permanent monitors are available through the USFS AirFire V4.1 smoke monitoring system [https://tools.airfire.org/monitoring; USFS (2021a)], a predecessor to the AirNow Fire and Smoke Map (https://fire.airnow.gov/). Limited historical PM data from some temporary monitors can be accessed through the WRCC [https://wrcc.dri.edu/cgibin/smoke.pl; WRCC (2021)]. No comprehensive archive of temporary $PM_{2.5}$ monitor data is currently available to researchers, land managers, or the public.

The www.airfire.org website provides visualization tools for ARAs to evaluate temporal and spatial smoke trends and how PM concentrations vary between observational surface measurements and smoke prediction model estimates. Temporospatial trends and smoke model performance are important for ARAs to contextualize with current fire conditions and observed smoke production during large wildfire events. Diurnal smoke behavior is particularly important for predicting how the smoke will effect some areas, especially when the smoke dispersion is dominated by terrain-driven winds in foothill and mountain communities. A limitation to the publicly available AirNow data is that it reflects the NowCast concentration, not necessarily the current concentration at any given time, so it could be anywhere from 1 to 3 hours behind in providing the appropriate trend. This is important for public health officials when tracking concentrations, especially when they are trying to provide schools and athletic organizations information on whether outdoor activities are safe or whether air quality is remaining in the unhealthy range.

AirNow does provide another link to the information—this is the primary public-facing sites and resources provided to better understand the trends in air quality. The forecasting reported on AirNow by local air pollution control districts are quite often accurate for a 24-hour period; however, there is a limitation in how the reporting area is determined. Quite often, the reporting area is based on the largest metropolitan area with an air quality forecast. This forecast may be an accurate estimate of smoke if the area has uniform terrain. However, when reporting areas for cities in the foothills or neighborhoods with substantial elevation change, the actual smoke concentration may be substantially different than predicted because of terrain-induced drainage flows. So even when air quality improves in the closest metropolitan area, the smoke may linger and take longer to dissipate in certain areas and may change the 24-hour estimate of the AQI. In foothill communities during terrain-driven wind events, air quality improvements will often be delayed compared with centrally located monitors because of smoke transport behavior following typical diurnal upslope and downslope winds patterns. The delay in AQI improvement has been particularly evident during extended periods of high pressure over fires where smoke continues to hang in the valleys over a period of days and sometimes weeks. Therefore, the smoke reporting for certain areas, especially in the wildland-urban interface (WUI) and in foothill communities provide AQI prediction challenges where actual air quality is not adequality represented by the closest central monitoring site.

4.4.4 SENSOR DATA AVAILABILITY

Commercially available air quality sensors use either local data storage for end-user use only or vendor-specific cloud-based data telemetry, storage, quality assurance review, and graphical presentation of summary monitoring data. Most air quality sensor manufacturers that maintain cloud-based systems do so to provide secure storage and analysis tools for each end user. 2B Tech (2021), Clarity (2021), and

PurpleAir (2021) are examples of manufacturers that do allow the end users to choose whether to keep the monitoring data private or allow for public dissemination of the data through each manufacturer's proprietary map-based web portals as part of the sensor registration process. The Environmental Defense Fund (EDF), OpenAQ, and other nongovernmental organizations have undertaken independent initiatives that advocate for the development of centralized repositories of data collected from ambient air quality sensors that includes data standards and definitions of terms with the vision of making integrated air quality sensor data from all manufacturers publicly available (EDF Air Sensor Workgroup, 2021; OpenAQ, 2021). The value of publicly available sensor data was demonstrated by U.S. EPA and USFS as part of their pilot AirNow Fire and Smoke Map project in 2020 (AirNow, 2021a). The pilot used the data from a single manufacturer (PurpleAir), because of their relatively large number of deployed sensors. The pilot project documented $PM_{2.5}$ sensor performance (Barkjohn et al., 2020) and the public availability of the data. However, there is currently no centralized publicly accessible air quality data repository from ambient sensors that are available for wildland fire incident teams, air quality regulators, researchers, or public health officials to access during wildland fire events.

4.4.5 REMOTE SENSING DATA AVAILABILITY

Data latency and reliable data availability are critical attributes when using satellite data, particularly for air quality uses associated with smoke plume tracking and improved predictions of pollutants distributions during active wildland fires. Operational satellite instruments such as VIIRS, GOES-ABI, and TROPOMI are designed for low data latency and reliable data availability because of the reliance on such instruments to inform weather and air quality forecasts. Such considerations are usually not a high priority for research satellites. However, the direct broadcast *X*-band downlink and near-real-time science data production software International MODIS/AIRS processing package (Strabala et al., 2003) implemented for the MODIS sensors aboard the Terra and Aqua satellites facilitated use of the data for tracking wildland fire plumes to improve $PM_{2.5}$ forecasts (Al-Saadi et al., 2005). The availability and latency for satellite- and ground-based remote sensing data is summarized in Appendix Table A.4-4 and Appendix Table A.4-5, respectively.

4.4.6 MEASUREMENT DATA QUALITY

FRM and FEM methods include instrument design requirements, strict performance specifications, and routine calibration and maintenance requirements. In addition, monitoring requirements (U.S. EPA, 2019b) prescribe routine onsite auditing of instrument performance, rigorous data quality assurance/quality control review of all regulatory measurements, and adherence to siting criteria (e.g., distance from obstructions). Monitoring agencies carry out and perform ambient air monitoring in accordance with the U.S. EPA's requirements and guidance, as well as often needing to meet their own state monitoring needs that may go beyond the minimum federal requirements. As previously stated, air quality data obtained from state, local, and tribal monitoring sites are reported to U.S. EPA's AQS database, along with other information, and are used for determining compliance with the NAAQS; assessing effectiveness of mitigation strategies; characterizing local, state, and national air quality status and trends; and associating public health outcomes with air pollution concentrations/population exposures. Therefore, regulatory measurements are the highest quality air pollution measurements available.

Nonregulatory instruments used for temporary incident response measurements like the ARA deployed E-BAM and E-SAMPLERS are maintained/calibrated to manufacturer specifications by IWFAQRP at their Lakewood, CO facility prior to field deployment each fire season. The USFS conducted tests on these two devices ranging from 30 to 1,700 $\mu g/m^3$ under smoke conditions against a gravimetric, filter-based U.S. EPA FRM sampler (Trent, 2006, 2003). On average, both the E-SAMPLER and the E-BAM samplers overestimated the mass concentration by approximately 13% over the FRM, yet correlation coefficients were very high, over 0.96 and over 0.99, respectively (Trent, 2003). The IWFAQRP instruments are received, installed, and maintained by trained ARA professionals following established program SOPs. The data quality from other state, local, or tribal agency temporary incident response instruments are expected to be of similar quality to IWFAQRP deployments when established training and instrument SOPs are followed.

Raw $PM_{2.5}$ concentration data from air quality sensors is generally considered qualitative during wildland fire smoke events owing to the general lack of smoke-specific performance testing, routine maintenance and calibration procedures, and data QA/QC screening and validation. There is also a recognition that certain sensor systems are better categorized by objective testing organizations, such as U.S. EPA (U.S. EPA, 2020c), the South Coast Air Quality Management District's Air Quality Sensor Performance Evaluation Center (SCAQMD, 2021), and the European Commission Joint Research Center (JRC, 2021), and that sensor networks deployed, characterized relative to FRM/FEM measurements, and maintained by governmental/professional organizations may be of higher quality. Adoption of formal air quality sensor performance targets, calibration, maintenance, data quality review guidelines, and certification requirements that are currently being investigated by U.S. EPA (Duvall et al., 2021a, b) would provide a path forward for ensuring that future air quality sensor data would better serve the observational air quality monitoring requirements of the wildland fire smoke management community.

4.5 CHALLENGES IN AMBIENT SMOKE MONITORING

Wildland fire smoke events can produce extreme near-field air pollutant concentrations that exceed monitoring instrument linear dynamic range and reporting limits, cause analytical interference(s), and generally increase the uncertainty in reported air pollution concentrations. In many areas of the country wildfire smoke is responsible for the highest air pollution concentration values experienced and may dominate the local populations' exposure to air pollution (e.g., $PM_{2.5}$) on an annual basis. Some initial evaluations of UV-photometric FEM O_3 instruments (Landis et al., 2018; Long et al., In Press) and visible spectrum FEM $PM_{2.5}$ instruments (Landis et al., 2021) have documented measurement accuracy degradation under smoke conditions. In addition, wildland fire smoke events present many inherent ambient measurement, quality assurance, data latency, data integration, data availability, and communication challenges for land management, wildland fire smoke management, air quality management, and public health officials including:

- Wildland fire events and downwind smoke impact zones occur disproportionally in areas of the U.S. having diffuse population centers and lacking U.S. EPA regulatory air quality monitoring infrastructure typically used to measure AQI and communicate appropriate public health messages. Complex terrain and unpredictable smoke plume behavior can also complicate accurate determination and spatial interpolation of AQI and the associated public health recommendations for limiting smoke exposure.
- Wildland fire smoke can be highly spatially and temporally variable. Smoke can be confined to topographic areas such as valleys or in specific vertical or meteorological layers (e.g., inversions), meaning that air quality monitors only a few kilometers apart can report dramatically different concentrations. Smoke concentrations can change substantially over short time periods as fire activity and meteorological dispersion changes make it difficult to predict and manage hazardous conditions (e.g., measured average hourly concentration values may not match the experience of smoke even at that location because of subhourly temporal fluctuations).
- Wildland fire smoke can be transported for long distances. Smoke plumes from specific wildfires have been traced across continental or even oceanic/transcontinental scales. Air pollution concentrations (e.g., $PM_{2.5}$) can be significantly elevated thousands of km away without an obvious connection to distant fire events.
- The availability, validity, comparability, and integration of observational air quality measurements during wildland fire events is improving (e.g., sensor data pilot, smoke modeling tools); however, there is a long way to go to enable real-time (low latency), integrated, and publicly available data and modeling tools that are required for effective management activities at local, state, and regional scales.

The air quality monitoring challenges during wildland fire events are inherently linked to the associated limitations in current U.S. EPA regulatory monitoring networks. The objectives of these networks do not include smoke monitoring. The current network designs that prioritize densely populated urban and suburban areas where most anthropogenic air pollution sources are concentrated result in a lack of network site density and spatial/elevation distribution of monitors in more remote areas where wildland fire events are more likely to occur. Issues with data telemetry, latency, and QA/QC review accumulate to create a situation where the effects of wildland fire smoke on air quality are not well captured by existing regulatory networks. Temporarily emplaced monitors, remote sensing, and air quality sensors offer future opportunities to supplement regulatory monitoring infrastructure. However, as discussed above, these observational monitoring technologies have their own issues with accuracy, reliability, availability of measured concentration values, and their inability to quickly emplace and telemeter data to fill the most important gaps in spatial coverage.

4.6 RECOMMENDATIONS

Currently, the fundamental understanding of wildland fire source emissions, the impact of smoke on ambient air quality, the estimation of human exposures, the quantification of adverse health outcomes, and the ability to develop and validate predictive deterministic air quality models are predicated on accurate measurements of criteria air pollutants and their precursors in smoke. This chapter presented and discussed the contemporary sources of ambient air quality monitoring data, the relative accuracy of data sources, the latency and availability of data, and the tools for accessing and analyzing air pollution monitoring data and smoke dispersion modeling in the U.S. Although U.S. EPA's current regulatory monitoring network objectives do not include smoke monitoring, it is evident that recent advances in measurement technologies, cloud computing capabilities, and online data accessibility tools have improved the national capacity to measure, predict, and disseminate public health information on smoke from wildland fire events. However, it is also clear that there are fundamental gaps in the ability to (1) accurately measure air quality impacts from wildland fire smoke over relevant spatial and temporal scales, (2) integrate and archive available observational data streams into common data format standards, and (3) provide timely access to integrated data analysis and visualization tools necessary for smoke management and public health officials to take effective control and abatement actions.

Based on these gaps, enumerated below are several actions that could help address the identified challenges and advance national capabilities for wildland fire smoke monitoring:

- Establishment of a program to evaluate the performance of U.S. EPA designated FRM/FEM regulatory monitors under wildland fire smoke conditions.
- Inclusion of wildland fire smoke monitoring as a routine air quality continuous monitoring objective for areas of the country with recurring wildland fire smoke. Including the addition of continuous air pollution measurements emitted from wildland fires (e.g., BC, CO, CO_2, NO_X).
- Inclusion of national wildland fire smoke monitoring as a routine air quality monitoring objective for integrated filter-based $PM_{2.5}$ samples. Including the addition of one or more well-established tracer species for biomass combustion like levoglucosan (anhydrosugar produced from the combustion of cellulose) into available routine national filter-based monitoring networks (CSN, FRM, IMPROVE) analytical suite to help elucidate the relative impact of wildland fire smoke on already collected filter-based $PM_{2.5}$ samples.
- Establishment of guidelines for evaluating commonly used commercially available nonregulatory instruments and air quality sensors under wildland fire smoke conditions.
- Establishment of data and QA/QC review standards for commonly used commercially available nonregulatory instruments and air quality sensors.
- Development of mobile air quality monitoring capabilities around wildland fire events as an added capability for ARAs working on large incidents particularly in more remote areas with limited existing monitoring infrastructure. Mobile wildland fire smoke measurements would provide public health officials the means to inform the placement of their temporary stationary monitors, evaluate the wildland fire smoke exposure risks across multiple communities, and to provide timely and actionable public safety information.

- Collaborative effort across federal agencies (e.g., U.S. EPA, USFS, NOAA, NASA) to establish common data-sharing agreements for remote-sensing data.
- Development of a publicly available cloud-based data integration and visualization platform for all available regulatory, nonregulatory, air quality sensor, and remote-sensing data streams for wildland fire smoke management and wildland fire smoke impact analysis. AirNow serves some of this capacity now and could be enhanced with the suggested functionalities.

Through these actions it is possible to chart a collaborative interagency path forward in addressing current wildland smoke monitoring challenges, such as unknown accuracy of air pollution monitors in wildland fire smoke; lack of network site density and spatial/elevation distribution of monitors, data telemetry and latency issues; and the availability and comparability of wildland fire smoke-impacted monitoring data products. In addition, the collaborative nature of the proposed actions would allow for the formation of a constructive community of wildland fire smoke practitioners and researchers focused on improving the quality, integration, and availability of air quality monitoring data.

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CHAPTER 5 DIRECT DAMAGES FROM WILDLAND FIRE

5.1 INTRODUCTION

The primary focus of this assessment is a quantitative analysis of the smoke impacts, on both air quality and health, from wildland fires. As detailed in the conceptual framework outlined in Chapter 2, in the process of examining the trade-offs between prescribed fire and wildfire it is also important to consider the potential effects, both positive and negative, of the fire itself. Although it is not possible in this assessment to quantify these effects because location-specific data are limited, the qualitative characterization of these additional effects helps add context to the overall examination of the trade-offs of smoke impacts due to different fire management strategies.

This chapter discusses the direct fire damages (value of economic loss) that are often experienced as a result of wildland fire. As detailed in Chapter 6 and quantitatively examined in Chapter 8, the health effects and overall population impacts of smoke exposure are well characterized. Although there are ecological benefits to fire (see Chapter 3), severe wildfires can adversely affect ecosystems, lead to substantial effects on public welfare, and incur societal costs (Table 5-1). In considering the costs incurred from wildfires, preparedness, mitigation, and suppression efforts are included, along with numerous losses that have substantial effects on society. The following chapter provides a broad discussion of these additional effects often experienced because of wildfires.

5.2 ECONOMIC BURDEN OF WILDFIRE

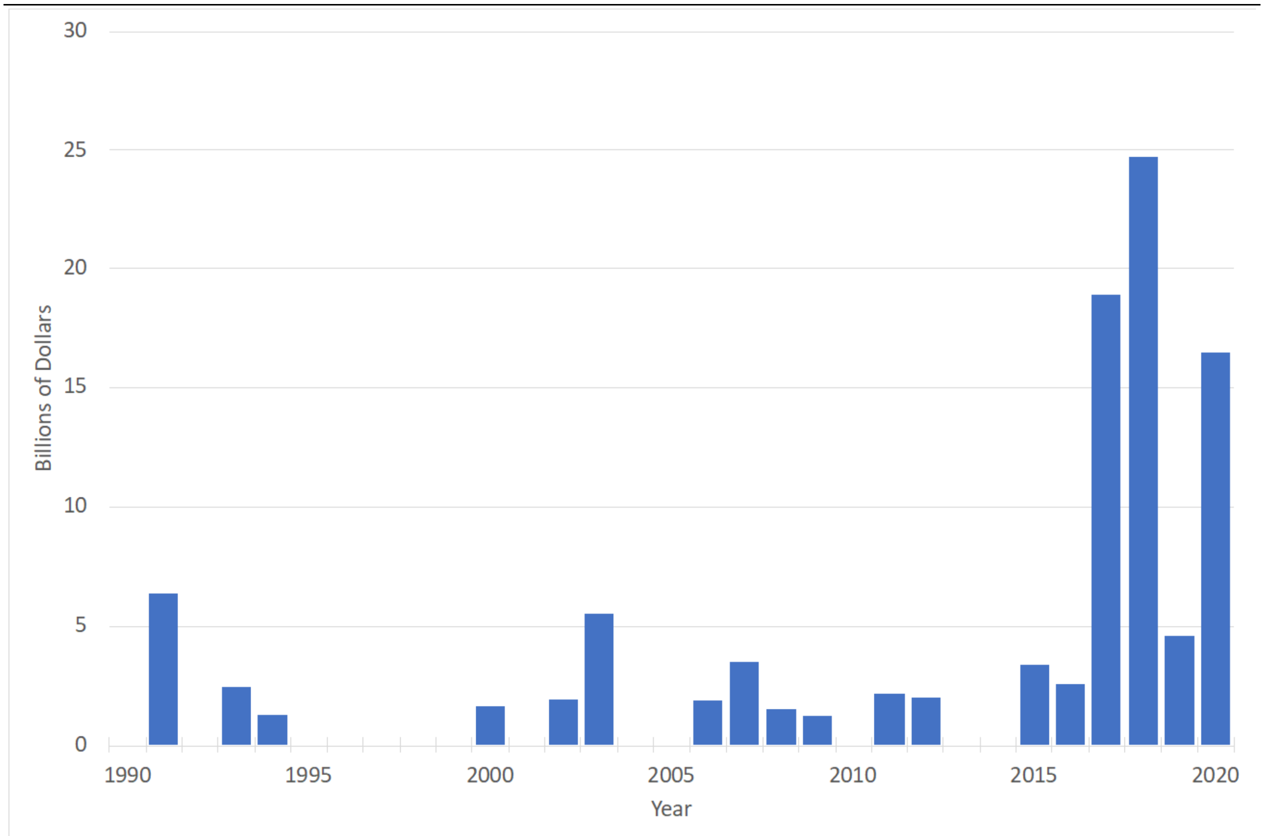
The National Institute of Standards and Technology (NIST) Special Publication 1215 (Thomas et al., 2017) quantified the burden on the U.S. economy from wildfires. The economic burden includes wildfire-induced damages and losses, and also the management costs to suppress and mitigate ignition and fire spread (see Table 5-1). The annualized burden was estimated to be between $71.1 billion to $347.8 billion in 2016 dollars ($77.4 billion to $378.7 billion in 2020 dollars). The estimates were based on literature or data available in early to mid-2017. Not included, for example, were recent catastrophic wildfire incidents. [Note, however, the estimates in Thomas et al. (2017) were significantly larger than the previous estimates found in the NIST Special Publication 1130 Hamins et al. (2012)].

Table 5-1 The economic burden of wildland fires.

Costs	Losses
Prevention	***Direct***
• Education and training	• Deaths and injuries
• Detection	• Psychological effects
• Enforcement	• Structure and infrastructure loss
• Equipment	• Environmental impact
Mitigation	• Habitat and wildlife loss
• Fuels management	• Timber loss
• Insurance	• Agricultural loss
• Disaster assistance	***Indirect***
Suppression	• General economic impacts
• Federal	• Evacuation costs
• State	• Accelerated economic decline
• Municipal (paid)	• Utility and pipeline interruption
• Rural (volunteer)	• Transportation interruption
Cross-cutting	• Government service interruption
• Legal	• Psychological effects (loss of amenities)
• R&D	• Housing market impact
• Building codes and standards	• Loss of ecosystem service
• Regulations	• Increase risk of other hazards
	• Loss of tax base
	• Health effects from fire retardant use

R&D = research and development.

Based on National Oceanic and Atmosphere Administration (NOAA) billion-dollar weather and climate disaster data (Smith, 2020), which include direct losses from insured and uninsured sources, the largest losses from billion-dollar wildfire disasters have all come since 2017 (Figure 5-1; note: there were no billion-dollar events prior to 1991). Since 1980, no year experienced more than a single billion-dollar wildfire disaster (direct losses from a single-event), meaning each year represents a single event in Figure 5-1. Accounting for more than just direct losses, Wang et al. (2020) measured the economic ramifications of the 17 largest wildfires in California during 2018 and estimated their direct, indirect, and health costs. The study authors estimated wildfires to have caused $148.5 billion ($126.1 billion to $192.9 billion, 95% confidence interval) in losses associated with direct capital losses ($27.7 billion), health effects ($32.2 billion), and indirect economic effects [$88.6 billion; Wang et al. (2020)].



Source: Developed from data presented in Smith (2020)

Figure 5-1 Billion-dollar wildfire event losses (1980−2020).

The economic burden from wildfire seems to have been increasing over time. Although the wildfires of the last few years have been particularly devastating, the increasing ability in measurement science to better account for the effects of wildfires can also partly explain the increase in reported costs and losses. In particular, until recently, the economic loss due to human-health effects from wildfire smoke has been underappreciated.

The next section discusses economic issues related to wildfire management, followed by a section on management costs, and then a section covering economic issues related to valuing wildfire net value change (NVC).

5.2.1 ECONOMICS OF WILDFIRE: MANAGEMENT IMPLICATIONS

Economics is a discipline concerned with the allocation of scare resources and the understanding of trade-offs. Central to the economics of wildfire management is the search for the understanding of trade-offs between management inputs (e.g., prevention and suppression) and the consequences of unwanted wildfire ignitions (e.g., life-safety, acres burned, structure loss). The economics of wildfire management is not a new concept. Headley (1916) discussed ideas of suppression effectiveness, efficiency, and waste of effort. Sparhawk (1925) introduced the idea of the “Cost plus Loss” (C+L) model as the management trade-off between prevention and “prefire suppression activities” (e.g., fuels management) expenditures, suppression expenditures, and wildfire losses. A central finding of the C+L model is that prevention and prefire suppression expenditures can be selected to minimize the sum of all costs (i.e., prevention, prefire suppression activities, and suppression spending) plus the resulting wildfire losses to identify the optimal level of management effort. The optimal level corresponds with the C+L minimum, and it can be shown that at the minimum, any other allocation of management resources will result in either (1) an increase in spending that exceeds the expected avoided loss or (2) a reduction in spending that surpasses an increase in expected loss. This concept of the C+L model is depicted in Figure 5-2, where the inputs of prefire suppression activities and suppression are independent inputs, and prefire suppression activities expenditures are held constant (Donovan and Rideout, 2003; Sparhawk, 1925). Suppression costs increase with increases in suppression effort, while the value of corresponding loss decreases. The minimum point of the (suppression) cost plus loss curve reveals the economically optimal level of suppression effort (holding prefire suppression activities constant).

The C+L model has been revised several times [e.g., Gorte (2013); Gorte and Gorte (1979)], with modern depictions acknowledging the potential for positive effects of wildfires, necessitating a change in the term “loss” to “NVC” (Rideout and Omi, 1990; Simard, 1976). Although the graphical depiction of the C+NVC is useful for illustration, it is less useful for identifying the minimum C+NVC when presuppression expenditures are allowed to be unconstrained. Further, because management activities and recent wildfire activity can have lasting effects on the fuels, affecting future wildfire risk (Prestemon et al., 2002), intertemporal optimization is required. Intertemporal optimization introduces additional considerations such as discounting and risk perception, which affect the optimal timing of forest management activities (Mercer et al., 2007; Amacher et al., 2005a, b).

There are two immediate challenges making the optimal levels of intervention difficult to determine. First, an understanding of the functional relationship between wildfire management activities and the resulting NVC is needed. Second, and perhaps more fundamental, is that many of the effects from wildfire are not well known or measured, particularly indirect or cascading effects. However, additional challenges include (1) the costs and losses are not incurred by the same subsets of the population, creating equity concerns and barriers to aligning economic interests and (2) the spatial, temporal, and economic boundaries of the C+L loss model are hard to define. Many of the sections that follow build from work detailed in the NIST Special Publication 1215 (Thomas et al., 2017) and describe categories of the costs and losses associated with wildland fire for the U.S.

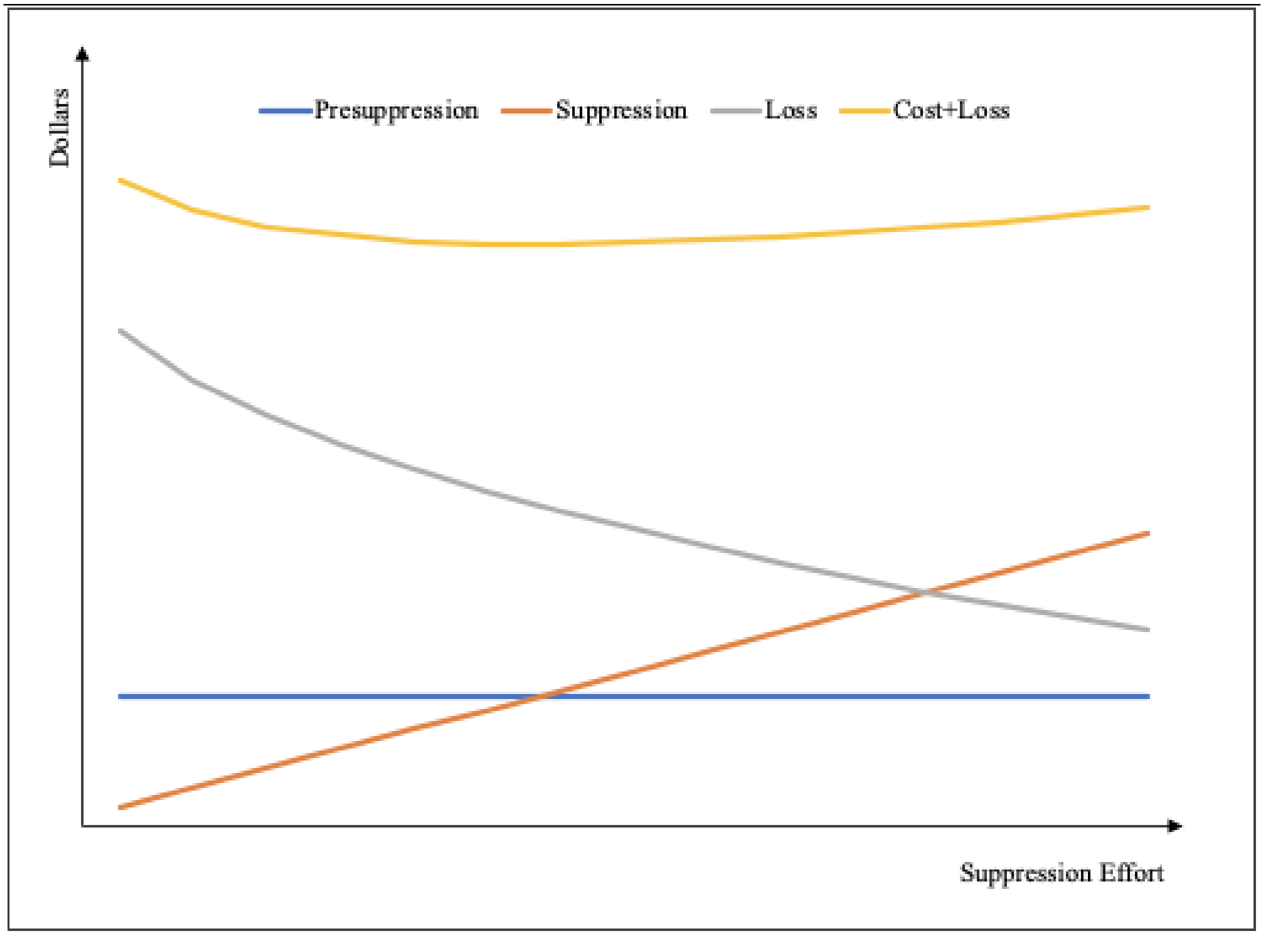


Figure 5-2 Illustrative example of the Cost plus Loss (C+L) Model of wildfire management.

5.2.2 MANAGEMENT COST CATEGORIES

Management cost categories include those expenditures spent on preparing for, mitigating, suppressing, and recovering from wildfires. Presuppression activities include prevention and preparedness. Suppression accounts for firefighter labor, equipment, firefighter training and wellness programs, as well as the monetary equivalence of volunteer time from local, nonpaid fire departments. Post-fire rehabilitation and recovery include efforts to return lands to prefire functionality. The "cross-cutting" cost category includes activities that impact multiple management activities; for example, research and development efforts result in more effective suppression technologies, improved building codes, and fire-resistant building products.

5.2.2.1 PREPAREDNESS AND PREVENTION

At the federal level, prevention and mitigation activities, including wildfire detection and education, are aggregated together in budget line items as "preparedness." Preparedness is considered to be "comprise[d] [of] a range of tasks to ensure readiness for wildfire response, including workforce preparation, equipment and resource management, and wildfire outlook conditions for forecasting" (Hoover, 2020). For Fiscal Year (FY) 2020, preparedness spending was $1.672 billion dollars in total for the U.S. Forest Service (USFS; 80%) and the Department of the Interior [DOI; 20%; Hoover (2020)].

Wildfire prevention activities include awareness efforts to promote fire safety to reduce unintentional wildfire ignitions. Awareness programs, such as public service announcements and media spots, community townhall-style presentations by wildfire prevention specialists, distribution of brochures and flyers containing educational messaging, and community wildfire hazard assessment performed by risk specialists have all been shown to reduce the number of human-caused unintentional wildfire starts and generate positive economic return on investment (Prestemon et al., 2010). For example, Prestemon et al. (2010) estimated that the benefit-cost ratio of prevention to be 35 to 1 on the margin. Abt et al. (2015), who also accounted for law enforcement efforts and intentionally set wildfires, found benefits were 5 to 38 times larger than prevention costs. Prevention efforts have been shown to have differential effects that vary by ignition cause type [e.g., escaped campfire, debris fire; Butry and Prestemon (2019); Abt et al. (2015)], and the timing of activities can be exploited to yield larger economic benefits (Butry et al., 2010b; Butry et al., 2010a) or coupled with other risk reduction activities, such as fuels management (Butry et al., 2010b).

Early wildfire warning and detection systems, including aerial and satellite technologies, can lead to improved firefighting response time, limiting fire growth after ignition or assist in monitoring wildfire progression, and increase suppression effectiveness (Cardil et al., 2019). Satellite-based wildfire detection information has been shown to improve fire commanders' decision making during suppression activities, yielding better firefighting safety and economic outcomes (Herr et al., 2020). Steele and Stier (1998) found that wildfire surveillance from fixed lookouts yielded benefit-cost ratios of 6 to 1 in terms of reduced suppression costs and property losses.

Wildfire risk assessments and related tools can be used to identify occurrences of elevated temporal or spatial (landscape-level) risks, by examining factors such as prior wildfire history, weather, climate, fuel conditions, and socioeconomic factors. Such information can be used to inform decisions on the prepositioning of mitigation and suppression resources (Bayham et al., 2020; Thomas et al., 2011; Prestemon and Butry, 2005). Improved suppression response time can yield economic benefits by reducing burned areas (Cardil et al., 2019).

5.2.2.2 MITIGATION

Mitigation activities are designed to reduce the consequences from wildfire (e.g., area burned, value of economic loss). For wildfires, the primary mitigation approaches are fuels management, insurance, and disaster assistance.

5.2.2.2.1 FUELS MANAGEMENT

Fuels management activities result in the reduction of hazardous fuels in forests. This can be accomplished by a number of methods, including prescribed burning and mechanical and chemical thinning of materials (as discussed in Chapter 3). In FY 2020, the federal government spent $194.0 million on the line item “hazardous fuels/fuels management” on federal lands and the line item “other Forest Service wildfire appropriations,” which also includes fuels management that amounted to $545.3 million (Hoover, 2020). Fuels management spending is not readily available at the state, local, and private levels, nationally.

There is statistical evidence that fuel treatments can impact wildfire behavior (Mercer et al., 2007; Prestemon et al., 2002), resulting in suppression cost savings in excess of treatment costs (Thompson et al., 2017; Taylor et al., 2013; Butry, 2009). However, some research suggests that fuel treatments may lead to increased suppression spending, due to more aggressive suppression strategies as an option in treated landscapes [e.g., see Belval et al. (2019); Loomis et al. (2019); Rideout and Ziesler (2008)]. Research into optimization has shown that with careful planning, fuel treatments can be leveraged to yield larger economics returns, when considering timing (Butry et al., 2010a) or when allowing for the sale of harvested materials after forest thinning (Prestemon et al., 2012). Beyond avoided suppression costs, Huang et al. (2013) identified additional benefits that included fatalities avoided, timber loss avoided, avoided regional economic impacts, rehabilitation costs avoided, and loss of carbon storage avoided. In addition, Houtman et al. (2013) considered the impact of “free” fuel treatments (i.e., wildfire that are allowed to continue to burn to achieve multiple objectives which can include resource benefits) on future suppression costs avoided and found instances of large economic returns. However, policies allowing for more wildfires to burn (wildland fire use), instead of immediate suppression actions, may be more economically favorable with a low or zero discount rate. Furthermore, wildland fire use is controversial and carries inherent risk. Current federal fire management policy, for example, allows for limited wildland fire use (i.e., as long as the managers determine that it would not endanger the public). To increase the amount of wildland fire use, the risk thresholds would need to be relaxed, potentially resulting in more unintended losses of people, structures, and resources [see Houtman (2011)].

Fuel modification also occurs on private land, often as part of a program to create an area around a structure designed to reduce wildfire ignition and spread (i.e., "defensible space"). The major barriers to use of defensible-space programs are related to cost, aesthetics, and privacy (Absher et al., 2013; Kyle et al., 2010; Absher et al., 2009). For some, climate change and risk perceptions have lessened some of the resistance (Wolters et al., 2017), while for others it is a familiarity with the programs and expectations of its effectiveness that have led to acceptance. Stockmann et al. (2010) evaluated the cost-effectiveness of various homeowner risk reduction strategies including fuels management and structure hardening. They found that fuel reduction within 61 m (200 ft) of the house was the most cost-effective. Nevertheless, homeowner actions to reduce wildfire risk are potentially limited by the homeowners' own inaccurate assessment of risk factors [e.g., Champ et al. (2009)].

5.2.2.2.2 INSURANCE

In measuring the U.S. fire problem, the cost of insurance has typically been calculated as the difference between premiums paid in and claims paid out (Hall, 2014), which constitutes overhead costs. These costs would include employees' wages, underwriting expenses, administrative expenses, taxes, real-estate expenses, legal expenses, and cost of capital. There are a number of insurance markets that are exposed to wildland fire, including homeowner's insurance, commercial insurance, automobile insurance (Hall, 2014), health and life insurance, and reinsurance markets. Frequently, wildfire losses are reported as direct, insured losses.

Although insurance could be part of the solution to increased efforts to reduce overall risk to wildland fire on private lands, very few firms offer insurance focused, in particular, on forests (Chen et al., 2014). A leading limiting factor to widespread adoption of such insurance is a lack of actuarial information on wildfire risk at fine spatiotemporal scales. There is additionally a need to develop a better understanding of the approaches for reducing moral hazard and adverse selection in the issuance of policies. As a result, policies tend to be expensive and out of reach of small forestland owners, meaning that an insurance-based incentive structure for reducing overall wildfire risks on private lands remains elusive.

5.2.2.2.3 DISASTER ASSISTANCE

Disaster assistance is financial assistance provided by the federal government following a disaster declaration. Because assistance can be used for things such as temporary housing, lodging expenses, repair, replacement, housing construction, child-care, medical expenses, household items, clean-up, fuel, vehicles, moving expenses, and other necessary expenses determined by the Federal Emergency Management Agency (FEMA), care needs to be taken in tracking the economic burden of wildfires because counting these costs or reimbursements directly and also as disaster assistance may result in double counting.

5.2.2.3 SUPPRESSION

In FY 2020, at the federal level, suppression spending exceeded $1.4 billion dollars, split between the USFS (73%) and the DOI [27%; Hoover (2020)]. These are resources used for firefighting. State suppression expenditures are estimated at $1 to 2 billion per year (Gorte, 2013).

An estimate for local (municipal) fire departments is more difficult to determine. An approximation can be calculated assuming the cost of wildfire prevention and suppression is proportional to the incident volume of fire involving wildland fuels. In 2014, based on Zhuang et al. (2017), it is estimated that career fire department expenditures amounted to $41.9 billion ($46.21 billion in 2020 dollars), and the value of volunteer (rural) fire departments is estimated at $46.9 billion [see “Method 5” used in Zhuang et al. (2017); $51.72 billion]. Based on call volume (27.8 million calls) reported to the National Fire Incident Reporting System (NFIRS) from 2018, fires involving natural vegetation represented 0.8% of all calls (20% of all fire incidents). In combination with fire department expenditures, this information could be used to estimate the amount spent to suppress wildland fires in local jurisdictions.

Gebert et al. (2007) found suppression spending to be impacted by burned area, suppression strategy, and region of the country. Statistical models developed to forecast USFS suppression costs by region of the country show that forecasted suppression spending is influenced by factors such as prior suppression expenditures, sea surface temperatures, and weather [e.g., temperature and precipitation; Gebert and Black (2012); Abt et al. (2009)]. The models found that suppression strategy influences total suppression costs for large wildfires, with direct suppression being the most expensive on a per-acre-burned and per-day basis but leads to smaller wildfire sizes and duration. However, studies have found that overall suppression strategy can be complicated by other factors, which also impact total suppression expenditures. For example, Liang et al. (2008) found that the percentage of private land within the burned area influenced suppression expenditures on large wildfires, while Rossi and Kuusela (2020) indicated that management risk attitudes (risk aversion) affected expenditures.

5.2.2.4 POST-FIRE REHABILITATION AND RECOVERY

Post-fire rehabilitation is funded at the federal level as part of "other activities," and in FY 2020 the other activities amounted to $41.9 million. This accounts for costs associated with landscape-level restoration activities. Also included in this line item are activities related to research and development, construction and maintenance of fire facilities, and forest health management (Hoover, 2020).

5.2.2.5 CROSS-CUTTING COST CATEGORIES

Some costs cut across various organizations and categories. These include legal costs, research, and regulations. Legal costs include the prosecution, defense, and incarceration of fire-setters. In 2019, there were 785,500 prisoners in local prisons [all crimes; Zeng and Minton (2021)]. In 2019, there were 1,430,805 prisoners in federal and state facilities, with 0.9% sentenced for "other" property crimes, which include arson [all types; Carson (2020)]. The Bureau of Prisons (2018) estimated that the average cost of incarceration for a federal inmate in FY 2016 was $36,299.25 ($39,566.18 in 2020 dollars).

Many public and nonprofit organizations are involved in research and development to reduce the costs and losses associated with wildland fires. For federal research and science agencies, some of these costs are included in the $41.9 million "other activities" listed above (Hoover, 2020).

Each state has its own building codes and fire regulations, based on the international model codes. In addition, some consumer products are built for fire safety. Zhuang et al. (2017) estimated in 2014 that fire-safety related costs for building construction were $57.4 billion ($63.30 billion in 2020 dollars) and for consumer products were $54.0 billion ($59.55 billion in 2020 dollars). This includes fire safety from all ignition and risk sources. In a study comparing the construction costs of a typical house with a "wildfire-resistant" house, Quarles and Pohl (2018) found that the costs of components are slightly less expensive for the wildfire-resistant house ($79,230 vs. $81,140). The cost components included the roof, exterior walls, deck, and landscaping. The largest savings were found for the exterior walls, which more than offset increases to the other components.

5.2.3 WILDFIRE LOSS CATEGORIES

Wildfire-induced losses are grouped into two categories: direct and indirect. Direct losses are those that occur as a primary result of wildfire (e.g., structure loss), while indirect losses are those that occur as a secondary, or cascading, result of wildfire (e.g., economic downturn due to business structure loss). Indirect losses are often more difficult to quantify due to latency and many may only be realized years after the wildfire.

5.2.3.1 DIRECT LOSSES

5.2.3.1.1 FATALITIES AND INJURIES

The National Fire Protection Association (NFPA) reported 80 civilian (nonfire-service) fatalities and 700 injuries in 2019 from fire incidents reported as "outside and other fires" (Ahrens and Evarts, 2020). The "outside and other fire" incident type includes wildland, grass, crop, timber, and rubbish fires. The estimates are based on a survey to U.S. fire departments, meaning the fatalities and injuries would tend to include those observed or reported immediately following the fire incident. Long-term health consequences made worse due to fire exposure, but not known until well after the incident, would not be captured. In 2017, there were 10 firefighter deaths associated with wildland suppression activities (USFA, 2018). The Incident Management Situation Report system, which tracks data on wildfires in federal jurisdictions, includes firefighter injuries. From 2003 to 2007, an average of 260 injuries per year were reported (Britton, 2010).

5.2.3.1.2 PSYCHOLOGICAL EFFECTS

Studies from wildfires have found depression, post-traumatic stress disorder (PTSD), and other anxiety disorders to have resulted from exposure to wildfire events. Estimates for civilian rates of PTSD and other anxiety disorders after a disaster range from 30% (Cole, 2011) to 60% (Kuligowski, 2017), with effects sometimes taking years to manifest (Kuligowski, 2017). For first responders, rates of PTSD have been estimated to occur in up to 20% of firefighters and paramedics (Rahman, 2016).

5.2.3.1.3 STRUCTURE AND INFRASTRUCTURE LOSS

The National Interagency Coordination Center (NICC) reported 963 structures lost by wildfire in 2019, under the annual average of 2,593 (NICC, 2019). NICC reported 25,790 structures lost in 2018 (NICC, 2018) and 12,306 structures lost in 2017 (NICC, 2017). NICC does not provide dollar lost estimates.

5.2.3.1.4 ENVIRONMENTAL EFFECTS

Environmental effects can take many forms, including effects on vegetation, soil as well as erosion, watershed including increased sediment deposition, and carbon sequestration. Vegetation loss can create the need to reseed and regrow forest and grasslands. Soil degradation can result in poor soil nutrients and vegetation growth. Both vegetation and soil loss can result in erosion and increase the risk of flooding and debris flow (Ren et al., 2011; Benda et al., 2003). Trees sequester carbon and provide oxygen, but carbon can be released to the atmosphere if trees are burned. Wildfires can decrease water quality by introducing carbon, metals, other containments, and changes to nutrients, which can affect aquatic ecosystems and drinking water (Rhoades et al., 2019b). In addition to increased treatment costs for potable water, poor water quality can impact agricultural and industrial operations (Bladon et al., 2014). Treatment costs include the increased need to remove solids and dissolved organic carbon in water impacted by discharge from burned forests and wildlands (Emelko et al., 2011). However, traditional water quality protection strategies may fail to recognize the effects from wildfire that would result in the need for water treatment (Emelko et al., 2011).

5.2.3.1.5 TIMBER AND AGRICULTURAL LOSS

Wildfires on lands managed for timber and agricultural purpose result in business losses. The 1998 Florida wildfires resulted in pine timber damage of between $300 to $500 million in 1998 dollars ($479 to 798 million), which represented over half of the quantified costs and losses of the wildfire event (Butry et al., 2001). The timber losses were from two effects: (1) value from the physical loss of timber and (2) a price increase, due to scarcity, after all salvageable timber was sold. Prestemon et al. (2006) evaluated salvage harvest scenarios following the 2000 Bitterroot wildfire and found similar (direction of) effects to consumers, owners of damaged stands, and owners of undamaged stands. They demonstrated that the value of timber lost due to wildfire could be more than offset (in general welfare effects) through salvage.

5.2.3.2 INDIRECT LOSSES

5.2.3.2.1 GENERAL ECONOMIC IMPACTS

Wildfires, and disasters in general, can have long lasting impacts on an economy. They can include business interruption (temporary and permanent closures) and effects that disrupt the supply chain. Supply chain disruption can affect businesses and customers far removed from the wildfire threatened areas.

Butry et al. (2001) found the 1998 Florida wildfires impacted the tourism and service sectors. In an analysis of the 2002 Hayman Fire in Colorado, Kent et al. (2003) found the wildfire induced overall employment growth of 0.5%, by creating shifts in the economy resulting in a decline in average wages by 3%. Focusing on employment and wage dynamics, Davis et al. (2014) examined the impact of the 2008 large wildfires in Trinity County, CA. They found that employment in the natural resource sector increased by 30%, while average wages fell by 19%; whereas wage growth was experienced in the other sectors, again demonstrating disparate effects. Borgschulte et al. (In Press) found that wildfire smoke impacts annual labor income and employment in the U.S. and estimates the economic loss to be four times that from mortality ($83 billion in 2020 dollars).

Nielsen-Pincus et al. (2014) explored the economic impacts of large wildfires (fires where suppression exceed $1.0 million) in the western U.S. states by economic sector. For counties with populations under 250,000, they found sectors with employment increases included natural resources and mining; trade, transportation, and utilities; information services; financial services; and federal employment. Sectors that lost employment included construction, manufacturing, professional and business services, education and health services, and leisure and hospitality services. For larger counties, total employment was reduced after a large wildfire by 0.04%.

Loomis et al. (2001) found in a study of visitors to forests in Colorado that hikers and mountain bikers responded with fewer visits in areas with crown fires, but the time since the fire also played a role. Englin et al. (2008) and Englin et al. (2001) found the linkage to recreation demand is time dependent, with recent wildfires correlated with increased visitation and older wildfires linked to fewer, with Englin et al. (2001) also noting a rebound effect with the oldest wildfires. Hesseln et al. (2003) found crown and prescribed fires reduced visitation but consumer surplus differed between hikers (increased) and mountain bikers (decreased) in New Mexico. In Montana, Hesseln et al. (2004) found hikers decreased visitations after a crown fire, but increased visitations after a prescribed fire. They found mountain bikers displayed the opposite pattern.

5.2.3.2.2 EVACUATIONS

Evacuation costs include temporary lodging and travel to and from the impacted area. Kent et al. (2003) found the Hayman Fire in Colorado resulted in other expenditures, which included evacuation, that were estimated to be up to $14 million ($19.5 million in 2020 dollars). In addition to expenditures, McCaffrey et al. (2015) mentioned the nonmonetary expenditures, including the “logistical” and “emotional” toll of fire evacuation.

5.2.3.2.3 LOST NATURAL AMENITIES

National forests provide a stream of values including historic, use and recreational, and existence (value someone places on knowing something exists whether or not they may ever visit or use). Some of these values can be monetized in the form of entrance and use fees. The National Parks were estimated to be worth $92 billion dollars [$100 billion in 2020 dollars; Haefele et al. (2016)].

5.2.3.2.4 HOUSING MARKET

Hedonic analyses that relate home sales prices to nonmarket amenities and other property attributes can detect the values of environmental goods and services not directly traded in markets. Several studies have evaluated the effect of wildfire risk on home sales prices, with the expectation that higher risk lowers sales prices, all else being equal. Loomis (2004) compared housing sale prices before and after the 1996 Buffalo Creek Fire (Colorado) and found a price decline between 13 to 15% of undamaged homes near the wildfire. Kim and Wells (2005), in a study of the greater Flagstaff area (Arizona), found moderate crown canopy closure (40 to 69%) was preferred by home buyers; whereas high crown canopy closure (70% and higher), which posed a higher wildfire risk, was shown to decrease sale prices.

Meldrum et al. (2015) explored whether wildfire risk perceptions of residents of homes in Ouray County, in southwestern Colorado, aligned with professionals' data-based assessments of wildfire risk based on features of the home and property, including whether the property had vegetation nearby. Residents underestimated the risks of wildfire nearby. In many other aspects of the property's features, residents' perceptions were generally not highly correlated with the assessments of the professionals. The implication is that economic motivations to undertake risk-reduction efforts would be lower if risk were more accurately quantified by residents. Donovan et al. (2007) compared housing sales prices before and after homes were rated based on wildfire risk in Colorado Springs, CO. They found that the availability of risk information was correlated with a decrease of a representative home sales value by 13.7%. Champ et al. (2009) explored whether home prices in Colorado Springs, CO were aligned with risks of wildfire. They found that homebuyers prefer risky locations due to their favorable amenities (e.g., topography) but that homebuyers were less cognizant of wildfire risks than objective assessments would identify. Although these homebuyers preferred less fire-prone building materials, they tended to undervalue features of their properties from the perspective of wildfire risk reduction.

Hjerpe et al. (2016), in a study of house prices in four western cities, found that the sales of homes with medium forest density (34 to 66%) within 100 m of a house was associated with lower sales prices; yet, homes with high forest density (67% and greater) within 500 m of a house was associated with higher sales prices. Stetler et al. (2010) estimated home sales prices in Montana and found that distance to the wildfire, time since, size of fire, and whether the home was within sight distance of the wildfire affected home sales, for an average price loss of −13.7% for a home within 5 km of the fire.

Kalhor et al. (2018) evaluated the impact of visible fire scars from the 2000 Cerro Grande Fire (New Mexico) on assessed house values in 2013. They found the impact of the previous damage equated to a 1.7 to 4.4% decline in assessed house value, while measures of future wildfire risk were found to be correlated to an increase in assessed house value by 0.3 to 0.4%. The latter impact was attributed to the crown area likely accounting for the aesthetic value of vegetation.

5.2.3.2.5 LOSS OF ECOSYSTEM SERVICES

Ecosystem services are generally defined as "any positive benefit that wildlife or ecosystems provides to people" (NWF, 2017). Few studies exist on a national scale. Most tend to be regional in scope and not specific to wildfire. For example, Loomis et al. (2000) evaluated the value of better watershed services for a 45-mile section of the Platte River, Desvousges et al. (1983) valued lake preservation, Moore and McCarl (1987) valued the preservation of the Mono Lake ecosystem, and Hanemann et al. (1991) valued increased salmon stock in the San Joaquin River. Such examples provide methods that could be used to value avoided losses to ecosystem services from wildfire mitigation.

Wildfire Fire and Prescribed Fire Effects on Forest Health and Wildlife

Studies in the ponderosa pine ecoregion of California, Oregon, and Washington have shown that fire management based on low-intensity prescribed fire coupled with mechanical thinning can, over time, approximate historical landscape conditions that are much less susceptible to catastrophic fires (Prichard et al., 2017a; Prichard et al., 2017b; Allen et al., 2002). Where it is feasible to use such practices, low-severity fires can promote important wildlife habitat and forest health benefits (Pausas and Keeley, 2019). These ecological benefits include improvements in habitat quality for threatened and endangered species (Pausas and Keeley, 2019); reductions in ground layer and understory "ladder" fuels; reduced losses of forest floor nutrient capital and water holding capacity (Murphy et al., 2006); and increased forest resistance to drought, pests, and diseases, all of which are being exacerbated by climate change (Spies et al., 2019; Vose et al., 2019).

To date, prescribed low-intensity fire and thinning treatments have not been adopted into local, state, and federal forest management practices at a scale necessary to affect the overall fire deficit, and associated fuel load excess, in western forests. The potential effects of ignoring the fire deficit is underscored by the growing body of evidence for the role of climate change in amplifying recent increases in the frequency and intensity of wildland fires (Kolden, 2019; Abatzoglou and Williams, 2016) and consequent effects on ecological benefits associated with low-intensity fire regimes.

Water Resources

Wildfire can both directly and indirectly affect water resources as well. Direct effects can occur via downwind smoke and ash deposition on the surface of water bodies (see Section 6.4), and damage to drinking water infrastructure. Indirectly, fire affects water resources primarily through increased runoff of water and other materials into nearby water bodies. Together, these direct and indirect effects can alter the physical, chemical, and biological characteristics of water resources, and by doing so, impact their end use, such as for recreation, aquatic life, and drinking water.

The direct effects of fire on drinking water infrastructure is an area of rising concern. For example, fires can damage water treatment facilities or water supply lines. In two locations in California (Santa Rosa and Paradise), benzene and other volatile organic compounds (VOCs) were detected in tapwater post-fire, with concentrations of benzene exceeding federal and state drinking water standards (Proctor et al., 2020). This was likely caused by the partial melting of plastic water-supply lines to homes and infiltration of hot gas and other materials when the supply system became depressurized (Proctor et al., 2020). As fires become more frequent, they are increasingly likely to burn into urbanized areas, and direct effects on drinking water infrastructure could become more common.

The indirect effects of fire are more widespread, including the indirect effects on water bodies used as drinking water sources. Fire-prone ecosystems are major sources of the national water supply. Fire effects on forested watersheds are particularly concerning because these watersheds provide much of the drinking water consumed in the lower 48 states (Liu et al., In Press). Most of these watersheds are at high risk from wildfire now or in the near future (Hallema et al., 2018).

Fire can impact the physical supply and timing of water delivery by altering runoff and streamflow. The loss of ground layer vegetation and canopy leaf biomass reduces interception and evapotranspiration, increasing runoff (Stevens, 2013; Seibert et al., 2010). Moreover, on some soil types, intense wildfires can dramatically increase runoff by increasing water repellency of near-surface soil layers, a condition that can persist for years (Certini, 2005). Depending on fire severity, rainfall patterns, and watershed soil and land cover characteristics, post-fire streamflow can increase in the days, months, and years following fire (Niemeyer et al., 2020). Severe fires can also increase the risk of downstream flooding (Stevens, 2013). Additionally, fire can alter the amount and timing of snowmelt. For instance, mountain snowpack beneath charred forests absorbed more solar energy, causing earlier melt and snow disappearance in >11% of forests in the western seasonal snow zone over the past two decades (Gleason et al., 2019). Fire and climate change effects on snowpack can also have a substantial impact on late summer runoff when it is most needed by fish and wildlife (Pausas and Keeley, 2019).

By increasing runoff and flow, fires can also increase erosion and delivery of sediments, ash, and other constituents to downslope ecosystems. The increased sediment loads and land destabilization that can occur post-fire (Ren et al., 2011; Benda et al., 2003) may be characterized by a large influx of suspended solids to headwater streams (Rinne, 1996). Although not always (Cawson et al., 2013), effects can often depend on fire severity, with greater sediment erosion associated with higher severity fires (Benavides-Solorio and MacDonald, 2005). A wide variety of chemical constituents are often mobilized along with the sediments and ash. This includes nutrients and cations, heavy metals, organic compounds, like polycyclic aromatic hydrocarbons (PAHs), and dissolved organic carbon (Smith et al., 2011). Besides direct additions to water resources, fire can indirectly increase disinfection byproducts (DBPs), compounds that form during drinking water treatment when disinfectants (e.g., chlorine, chloramine) react with organic carbon and nitrogen compounds present in higher concentrations post-fire (Bladon et al., 2014). Some DBPs pose health risks, with the potential to cause certain cancers, reproductive issues, and anemia.

Encroachment of wildfire into the wildland-urban interface (WUI) can also release largely unknown types and quantities of anthropogenic contaminants into streams. Combustion of houses, buildings, vehicles, waste sites and other infrastructure present risks from hazardous chemicals, such as benzene and VOCs, as well as heavy metals (Proctor et al., 2020; Uzun et al., 2020). Finally, the use of fire retardants may also increase nutrient and chemical loading to post-fire landscapes.

Beyond physical and chemical changes, fires can also indirectly alter biological assemblages in downstream waters. Fire can increase coarse woody debris in streams (Young, 1994), positively impacting long-term habitat for fish, yet over the shorter term, fish and macroinvertebrate populations typically decline post-fire [e.g., Rinne (1996)]. Concomitantly, burning in riparian areas can increase light levels to streams, and studies have often recorded increases in stream temperatures post-fire [e.g., Dunham et al. (2007)]. This could negatively affect cold-water fish species, like salmonids (Beakes et al., 2014). Combined with the increased light and temperature, an influx of nutrients and sediment can also promote harmful algal blooms and the production of cyanotoxins (Bladon et al., 2014; Smith et al., 2011). These cyanotoxins both contaminate drinking water and negatively affect aquatic life.

While wildfire has been a part of the natural ecology of many ecosystems for millennia, an increase in fire frequency, area burned, and/or severity can have deleterious effects on water resources, altering their physical, chemical, and biological characteristics. In general, the more severe the fire, the more likely downstream waters will be affected, with greater potential for flooding, higher sediment loads, and other effects on water quality. By contrast, lower severity fires could positively effect downstream water users because the effect on water quality may be lower while water supply is temporarily increased. Effects following fire are generally most pronounced in the first few years but may persist for more than a decade in some cases (Rhoades et al., 2019a; Smith et al., 2011). Increased concentrations of nutrients, heavy metals, organic compounds like benzene, and DBPs pose particular risks, along with increased algal blooms and cyanotoxins. Communities will need to be aware—and plan for—the potential for post-fire contamination of water resources, especially following severe fire. The provisioning of safe drinking water from burned watersheds may require additional treatment infrastructure and increased operations and maintenance costs to remediate effects.

5.2.3.2.6 OTHER EFFECTS

Other effects of wildfire include accelerated economic decline, loss of utilities and transportation systems, disruption to government services, interference with military operations (e.g., smoke visibility issues), cascading natural hazard risks (e.g., increase risk of mudslide or growth of invasive species), loss of tax base due to housing and building stock, and health and environmental effects from fire retardants.

Many of these effects are not well defined or monetized. (Focused on California, CCST (2020) provides a discussion on some of these categories and others.)

5.2.4 MAGNITUDES, GAPS, AND UNCERTAINTY

Table 5-2 shows estimated magnitudes of value of the costs and losses and levels of uncertainty in their measurement or ability to measure at a national scale [reproduced from Thomas et al. (2017)]. The estimated magnitudes and uncertainties were based on the values found in the report, and where not available, were estimated using expert judgment of the report authors. The largest cost and loss categories were fuel treatments and defensible space, suppression, economic value of deaths and injuries, evacuation costs, and effects on the housing market. The largest sources of uncertainty tended to be indirect economic effects, insurance, and some of the cross-cutting categories (e.g., building codes and standards, regulations).

Although there is significant literature detailing components of the costs and losses associated with wildland fire, producing an annual national estimate, which could be tracked over time to evaluate management success, is difficult at this time without introducing large sources of uncertainty in the estimates. However, it does appear that the economic burden from wildland fire is increasing over time.

Table 5-2 Magnitude and uncertainty associated with the economic burden of wildfire at the national level.

	Order of Magnitude	Uncertainty
Costs		
Preparedness	$$$$	?
Mitigation		
Fuels management		
Fuel treatments (Rx fire, thinning)	$$$	?
Defensible space/firewise	$$$$	???
Insurance	$$	????
Disaster assistance	$	??
Suppression		
Fire departments (labor, equipment, training)		

Table 5-2 (Continued): Magnitude and uncertainty associated with the economic burden of wildfire at the national level.

	Order of Magnitude	Uncertainty
Federal	$$$$	?
State	$$$$	?
Municipal (professional)	$$$$	???
Rural (volunteer)	$$$$	???
Cross-cutting		
Legal		
Prosecution	$$	??
Incarceration	$$$	??
Civil/liability	$$	????
Science/research and development	$$	???
Building codes and standards	$$	????
Regulations (e.g., zoning)	$$	????
Losses		
Direct		
Deaths and injuries (civilian and firefighter)	$$$$	??
Psychological effects (PTSD)	$$	???
Structure and infrastructure loss	$$$	???
Environmental impact	$$$	????
Habitat and wildlife loss	$$	????
Timber loss	$$$$	???
Agriculture loss	$$$	????
Remediation/cleanup	$$	???
Indirect		
General economic impacts (business interruption, tourism, supply chain)	$$$	????
Evacuation costs	$$$$	???

Table 5-2 (Continued): Magnitude and uncertainty associated with the economic burden of wildfire at the national level.

	Order of Magnitude	Uncertainty
Accelerated economic decline of community	$$$	????
Utility and pipeline interruption (electricity, gas, water, oil)	$$$	????
Transportation interruption (e.g., roads and rail)	$$	????
Government service interruption (including education)	$$	????
Psychological effects (loss of natural amenities)	$$	????
Housing market impact (loss due to fire risk)	$$$$	???
Loss of ecosystem services (e.g., watershed/water service)	$$$	????
Increased risk of other hazards (e.g., mudslide, invasive species)	$$$	????
Decrease in tax base (structure loss or decline in value of structure)	$$$	???
Decrease in government services	$$$	????
Health/environmental effects from use of fire retardants/suppressants	$$$	????

PTSD = post-traumatic stress disorder; Rx = prescribed.

Note: Classification of “order of magnitude”: $ = <millions; $$ = 10s millions; $$$ = 100s millions; $$$$ = billions; “uncertainty”: ? = low; ?? = medium; ??? = high; ???? = unknown.

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CHAPTER 6 HEALTH AND ECOLOGICAL EFFECTS OF WILDLAND FIRE SMOKE EXPOSURE

6.1 INTRODUCTION

Wildland fire (i.e., prescribed fire and wildfire) smoke can have detrimental effects on both human and ecological health, but can also provide ecological benefits (see Section 6.5.1.2) as well as cultural values when used as part of indigenous cultural fires (Raish et al., 2005). Although the health impacts of wildfire smoke exposure can be quantitatively estimated using the Environmental Benefits Mapping and Analysis Program—Community Edition [BenMAP-CE; U.S. EPA (2019a)], it is much more challenging to quantify the potential ecological impacts. This chapter summarizes the health effects associated with wildland fire smoke exposure, both at the population level and more specifically by wildland firefighters. It also characterizes the different actions and interventions that can be employed at a population and individual level to reduce smoke exposure and highlights the ecological effects associated with wildfire smoke. This literature-based overview complements the new, model-based analysis presented in Chapter 7 and Chapter 8 which quantifies certain specific components of this overall picture of wildfire smoke-related health effects, as per the conceptual framework for this report described in Chapter 2.

In assessing the evidence base spanning both human and ecological health, the current understanding of the effects from wildland fire smoke primarily stems from studies examining effects due to exposures to ambient fine particulate matter (particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm [$PM_{2.5}$]), with a growing body of evidence focusing specifically on wildfire smoke, and only a few studies focusing on prescribed fire smoke. Although smoke also contains precursors that can lead to ozone (O_3) formation downwind from a wildland fire (see Chapter 7), fewer studies have examined wildfire-specific health effects associated with ozone. However, extensive evidence demonstrating health effects from ambient ozone exposures indicates the potential for ozone formed from wildfires to result in an additional significant public health burden (U.S. EPA, 2020a). Additionally, although $PM_{2.5}$ is a primary pollutant of concern, fire smoke contains a multitude of gases and other elements harmful to health. As such, during wildfire smoke episodes, $PM_{2.5}$ characterizes the exposure to both smoke and gases, whereas in ambient settings $PM_{2.5}$ generally characterizes anthropogenic combustion pollution.

The characterization of the health effects associated with wildland fire smoke at the population level focuses on U.S.-based epidemiologic studies to aid in informing the selection of concentration-response (C-R) functions for BenMAP-CE estimation (i.e., in both the main analysis and sensitivity analyses) of the potential health impacts of smoke for each case study area (i.e., Timber Crater 6 [TC6] and Rough fires; see Chapter 8). Wildland firefighters represent a subset of the population that continuously experiences smoke exposure. As a result, the health effects experienced within this population are also discussed to provide a complete characterization of the current evidence base regarding the health effects associated with wildland fire smoke.

The extent of prescribed fire and wildfire smoke exposure depends on proximity to the fire and the location (i.e., not everyone is exposed to smoke from fires), duration, and intensity of smoke plumes. Therefore, individuals can plausibly take actions to reduce or mitigate exposure to smoke from prescribed fires or wildfires. In addition to identifying the potential human health effects of smoke exposure, this chapter also evaluates and characterizes the effectiveness of various actions that can be employed at the population and individual level to reduce smoke exposure and subsequently protect public health.

While the characterization of the human health effects of wildland fire smoke is a main focus of this assessment, wildland fire can also have positive and negative ecological effects due to both the fire itself and smoke. Although other chapters of this assessment capture the direct ecological effects of fire, this chapter captures those effects attributed specifically to wildfire smoke.

6.2 WILDFIRE SMOKE EXPOSURE AND HEALTH

Scientific evidence examining the health effects associated with wildfire smoke exposure has grown significantly in recent years. The underpinnings of this evidence are rooted in the decades of research examining the health effects of ambient air pollutants, many of which are components of wildfire smoke. Of these components, particulate matter (PM), specifically $PM_{2.5,}$ is a main component and has been shown to have a significant public health impact, which is demonstrated by the range of health effects associated with $PM_{2.5}$ exposure, including respiratory and cardiovascular effects, as well as mortality (Jaffe et al., 2020; U.S. EPA, 2019b). Recent epidemiologic and experimental studies examining the health effects of wildfire smoke exposure report findings that are generally consistent with the broad body of evidence from studies examining short-term $PM_{2.5}$ exposure (U.S. EPA, 2019c; Black et al., 2017; Reid et al., 2016a). The consistency in results across studies of wildfire smoke exposure and $PM_{2.5}$ are further supported by studies that compared the health effects associations between various sources of $PM_{2.5}$, including wildfire smoke, and ambient $PM_{2.5}$. These studies have not provided evidence indicating differences in the toxicity between different sources, particularly combustion-related sources, of $PM_{2.5}$ and total ambient $PM_{2.5}$ (DeFlorio-Barker et al., 2019; U.S. EPA, 2019b). For example, a recent study examined differences in risk estimates which are naturally higher during wildfire periods due to higher concentrations of $PM_{2.5}$, but this study focused on differences in absolute risk and not toxicity (Aguilera et al., 2021). However, experimental studies have provided some evidence of differential toxicity and mutagenicity due to both the flaming and smoldering of different individual fuel sources, which may be important to consider when examining the trade-offs between prescribed fire and wildfire (Kim et al., 2018).

Most studies that examine the health effects of wildfire smoke exposure at the population level focus broadly on wildfire smoke, without accounting for potential differences in composition of smoke emissions between prescribed fire and wildland fire. A recent epidemiologic study conducted by Prunicki et al. (2019) reported initial evidence of differences in markers of immune function, DNA methylation, and worsened respiratory outcomes in school-aged children in Fresno, CA exposed to wildfire smoke compared to prescribed fire smoke. The difference in effects observed coincided with higher concentrations of air pollutants from wildfires compared to prescribed fires. However, it is unclear what aspects of the difference between prescribed fires and wildfires resulted in the differential health effects (e.g., differences in duration, air pollutant concentrations, fuel types, burn conditions). Additionally, it remains unclear whether there are differences in more overt health effects (e.g., hospital admissions, mortality) between the two fire types.

Overall, wildfire smoke exposure studies report results that are generally consistent with epidemiologic studies of short-term $PM_{2.5}$ exposure, and are also remarkably consistent with each other considering the large degree of variability in both the exposure indicators (e.g., $PM_{2.5}$, wildfire-specific $PM_{2.5}$, smoke day) and exposure assessment methodologies employed. Without evidence supporting a single universal indicator to represent smoke exposure, the variability in the indicator used across studies directly influences the application of results from individual studies in quantitative assessments, such as a risk assessment or a cost-benefit analysis. Furthermore, as noted previously, there is limited evidence regarding the health effects associated with ozone derived from wildland fire smoke even though there is extensive evidence of numerous health effects from studies of ambient ozone exposure (U.S. EPA, 2020a). As a result, this section consists of an evaluation of epidemiologic studies conducted within the U.S., published through May 2021, that could be used, either alone or in combination with studies of ambient $PM_{2.5}$ and ozone, in a quantitative assessment using BenMAP-CE of the potential public health impacts associated with the scenarios developed for the TC6 and Rough fire case study areas identified in earlier chapters. Based on the majority of the wildfire smoke epidemiologic studies focusing on $PM_{2.5}$ and because of the consistency in health effects between studies of short-term $PM_{2.5}$ and wildfire smoke exposure, the epidemiologic studies evaluated within this section consist of those that examined health outcomes where the most recent U.S. EPA *Integrated Science Assessment for Particulate Matter* (U.S. EPA, 2019b) concluded that the evidence indicates either a “causal relationship” or “likely to be causal relationship” (i.e., respiratory and cardiovascular effects, and mortality). The selection of health effects to evaluate is consistent with the criteria used by the U.S. EPA in the process of selecting health effects to quantitatively estimate when conducting BenMAP-CE analyses (U.S. EPA, 2021).

The evaluation of the health effects of wildfire smoke exposure within this assessment is not intended to be an exhaustive review of the evidence. Recent reviews and interagency efforts have extensively characterized the current state of the science with respect to the health effects associated with wildfire smoke exposure (Jaffe et al., 2020; U.S. EPA, 2019c; Reid et al., 2016a). In addition, the evaluation within this assessment does not rely on the numerous animal toxicological studies conducted to date that focused on examining health effects from exposures consisting of wildfire smoke from fuel sources commonly found in the U.S. (e.g., individual tree species) or real-world wildfire smoke.

6.2.1 CHARACTERIZATION OF WILDFIRE SMOKE EXPOSURES

Wildfires are often natural, spontaneous events, and this has complicated the ability of epidemiologic studies to characterize population exposures to wildfire smoke. Thus, studies have used a variety of approaches to estimate wildfire smoke exposure in terms of both the exposure indicator and exposure assessment methodology used (Appendix Table A.6-1). Although the exposure assessment approaches used across studies vary in complexity and in the specificity of the indicator in representing wildfire smoke exposure, epidemiologic studies report generally consistent associations between short-term wildfire smoke exposure and health effects (Section 6.2.2).

6.2.1.1 EXPOSURE INDICATOR

Within epidemiologic studies, the exposure indicator is a quantity meant to represent exposure to an environmental contaminant. For wildfire smoke, which consists of a complex mixture of pollutants, various indicators have been used as a surrogate for wildfire smoke exposure. These indicators vary in specificity and sensitivity with respect to how well they represent exposure to wildfire smoke. Because of the public health implications of exposure to $PM_{2.5}$ and because $PM_{2.5}$ is a main component of wildfire smoke, studies often rely on the use of some form of $PM_{2.5}$ as an exposure indicator. Some epidemiologic studies used monitored or modeled $PM_{2.5}$ concentrations as the exposure indicator (Alman et al., 2016; Reid et al., 2016a; Delfino et al., 2009) while other studies used wildfire or smoke-specific $PM_{2.5}$, which consisted of removing $PM_{2.5}$ derived from other $PM_{2.5}$ sources from the concentrations estimated (Stowell et al., 2019; Gan et al., 2017; Rappold et al., 2012). Additionally, some studies use $PM_{2.5}$ concentrations to estimate a range of $PM_{2.5}$ concentrations from an atmospheric model to develop an exposure indicator based on classifying days as either smoke or nonsmoke days or by assigning each day a level of smoke density (i.e., light, medium, or dense). In these studies, the defining of days by smoke status often depended on using criteria to define specific ranges of $PM_{2.5}$ concentrations that are considered indicative of wildfire smoke exposure (Jones et al., 2020; Wettstein et al., 2018; Liu et al., 2017b; Liu et al., 2017a). The use of a broad exposure indicator, such as smoke days, may be more representative of the multipollutant nature of wildfire smoke. However, to date there has been no indication that any one exposure indicator represents a better surrogate of wildfire smoke exposure than another. Overall, the variability in the exposure indicator used across studies partly reflects the difficulty in examining the health effects of wildfire smoke exposure and the air quality data available, or lack thereof in some instances (see Chapter 4).

6.2.1.2 EXPOSURE ASSESSMENT METHODOLOGY

Estimating wildfire smoke exposure for epidemiologic studies is challenging because wildfire smoke is spatially and temporally dynamic, and areas impacted by wildfire smoke often have few ambient monitoring sites because most air quality monitors reside in urban locations (see Chapter 4). Consequently, epidemiologic studies have resorted to using numerous methods that vary in complexity to assign exposures (Appendix Table A.6-1). In contrast, due to the planned nature of prescribed fires, monitors could be deployed to capture population exposure, but to date have not been widely used in this capacity so the data are not always available (see Chapter 4).

Consistent with many epidemiologic studies of ambient air pollution, a few studies examined relationships between short-term wildfire smoke exposure and health effects using monitored $PM_{2.5}$ concentrations and some approach to assign exposures to a defined spatial extent, whether that be a city or ZIP code (Leibel et al., 2020; Zu et al., 2016). Most epidemiologic studies focusing on wildfire smoke exposure use exposure models that rely on data from multiple sources and are often referred to as hybrid exposure models. These models use both monitoring and modeling data, and in some instances, satellite measurements to take advantage of different types of available data to estimate wildfire or smoke-specific $PM_{2.5}$ concentrations. Incorporating all these data sources into the model allows for the calibration of model predictions with monitored data and a broader spatial extent to be included in epidemiologic studies instead of being limited to only those locations within reasonable proximity to air quality monitors, which are primarily in urban centers. Relatively few of the studies that used hybrid exposure models evaluated model performance, but those studies that did indicate the models performed rather well [see Appendix Table A.6-1; Reid et al. (2019); Stowell et al. (2019); Gan et al. (2017); Reid et al. (2016a)].

A few epidemiologic studies relied on other approaches to estimating wildfire smoke exposure. While often included as a component in the exposure model to estimate wildfire smoke, one study used only satellite measurements (i.e., aerosol optical depth [AOD]) to identify areas that were impacted by a smoke plume (Rappold et al., 2011). The remaining studies used various models that were developed to examine wildfire smoke exposures by estimating either wildfire-specific $PM_{2.5}$ exposure or smoke exposure more broadly. Studies that estimated wildfire-specific $PM_{2.5}$ exposure used the Wildland Fire Emissions Information System (WFEIS) or National Oceanic and Atmospheric Administration's (NOAA's) Smoke Forecasting System (SFS) in combination with the transport and dispersion model HYSPLIT (Hutchinson et al., 2018; Tinling et al., 2016; Rappold et al., 2012), while studies that focused on smoke days used NOAA's Hazard Mapping System (HMS) to characterize exposure based on the density of smoke (Jones et al., 2020; Wettstein et al., 2018). Overall, regardless of the exposure assessment approach used, the results across epidemiologic studies provide evidence supporting a relationship between wildfire smoke exposure and various health effects (see Section 6.2.2).

6.2.1.3 UNCERTAINTIES AND LIMITATIONS IN CHARACTERIZING WILDFIRE SMOKE EXPOSURE

A challenge in estimating wildland fire smoke exposure, as detailed within Chapter 4, is the fact the current ambient monitoring network was not designed with the goal of measuring smoke from wildfires for public health surveillance. As a result, as noted within this section, epidemiologic studies have relied on a variety of approaches to estimate smoke exposure, such as collecting $PM_{2.5}$ concentration data from the ambient monitoring network, predicting concentrations from photochemical transport models or satellite measurements, or using hybrid exposure models that use multiple data sources. In addition to using continuous variables to estimate smoke exposure, some studies use smoke plume data in the form of a dichotomous variable to define smoke exposure. Although results across epidemiologic studies are consistent regardless of the approach used to assign exposure, both in terms of the exposure model and the exposure indicator (see Section 6.2.2), there are inherent uncertainties and limitations across each of the approaches used. The lack of information on how different exposure indicators compare with each other and the fact some exposure indicators are not conducive for quantitative analysis such as in BenMAP-CE, complicates the selection of wildfire-specific epidemiologic studies to be used for estimating the potential public health impacts of smoke.

One of the larger uncertainties in the epidemiologic evidence to date is how well exposures represented by smoke plumes reflect $PM_{2.5}$ concentrations experienced on the ground. However, a recent study by Larsen et al. (2018) examined $PM_{2.5}$ monitoring and smoke plume data and found initial evidence that monitored values on the ground reflect the presence of smoke plumes in the vertical column measured by satellites. In the future, more detailed evaluations of the different approaches that can be used and a characterization of their strengths and weaknesses will aid in further supporting the interpretation of results from epidemiologic studies and their use in quantitative analyses.

6.2.2 HEALTH EFFECTS ASSOCIATED WITH WILDFIRE SMOKE EXPOSURE

In the context of wildfires, most U.S.-based studies focus on short-term or daily exposures (i.e., 24-hour average). Across these studies, the primary pollutant of interest is $PM_{2.5}$, with only one study focusing on ozone (Reid et al., 2019). Studies examining exposure durations shorter than a 24-hour average, often referred to as subdaily exposures, have been limited to epidemiologic and controlled human exposure studies of ambient $PM_{2.5}$ focusing on subclinical measures of heart or lung function and not overt population-level effects, such as hospital admissions or mortality (U.S. EPA, 2019b). Therefore, these studies of subdaily exposures do not directly inform the relationship between shorter duration wildfire smoke exposures and health effects because of the difficulty in linking a change in a subclinical effect to an overt health outcome. As a result, most of the evidence informing the current understanding of health effects associated with wildfire smoke exposure comes from epidemiologic studies primarily focusing on exposures over single-day or multiday lags ranging from 0 to 5 days.

The focus on examining health effects associated with short-term wildfire smoke exposures has resulted in a relative lack of information on (1) the health effects due to repeated wildfire smoke exposures (i.e., over many days, weeks, or months), (2) the long-term health effects of wildfire smoke exposure from a single wildfire event, and (3) health effects due to long-term wildfire exposures over many months and multiple fire seasons. To date, studies have not examined the impact of repeated wildfire smoke exposure on health, but an initial study provides preliminary evidence that a wildfire smoke event with high $PM_{2.5}$ concentrations may detrimentally impact health, specifically lung function, over multiple subsequent years (Orr et al., 2020). The examination of longer term exposures to wildfire smoke has been limited to a recent study indicating increased risk of mortality in hemodialysis patients as cumulative exposures increase up to 30 days (Xi et al., 2020), analyses of subclinical effects (e.g., changes in lung function) in wildland fire fighters over multiple fire seasons (Adetona et al., 2016), and a study examining the potential implications of wildfire smoke exposure on the influenza season (Landguth et al., 2020).

6.2.2.1 RESPIRATORY EFFECTS

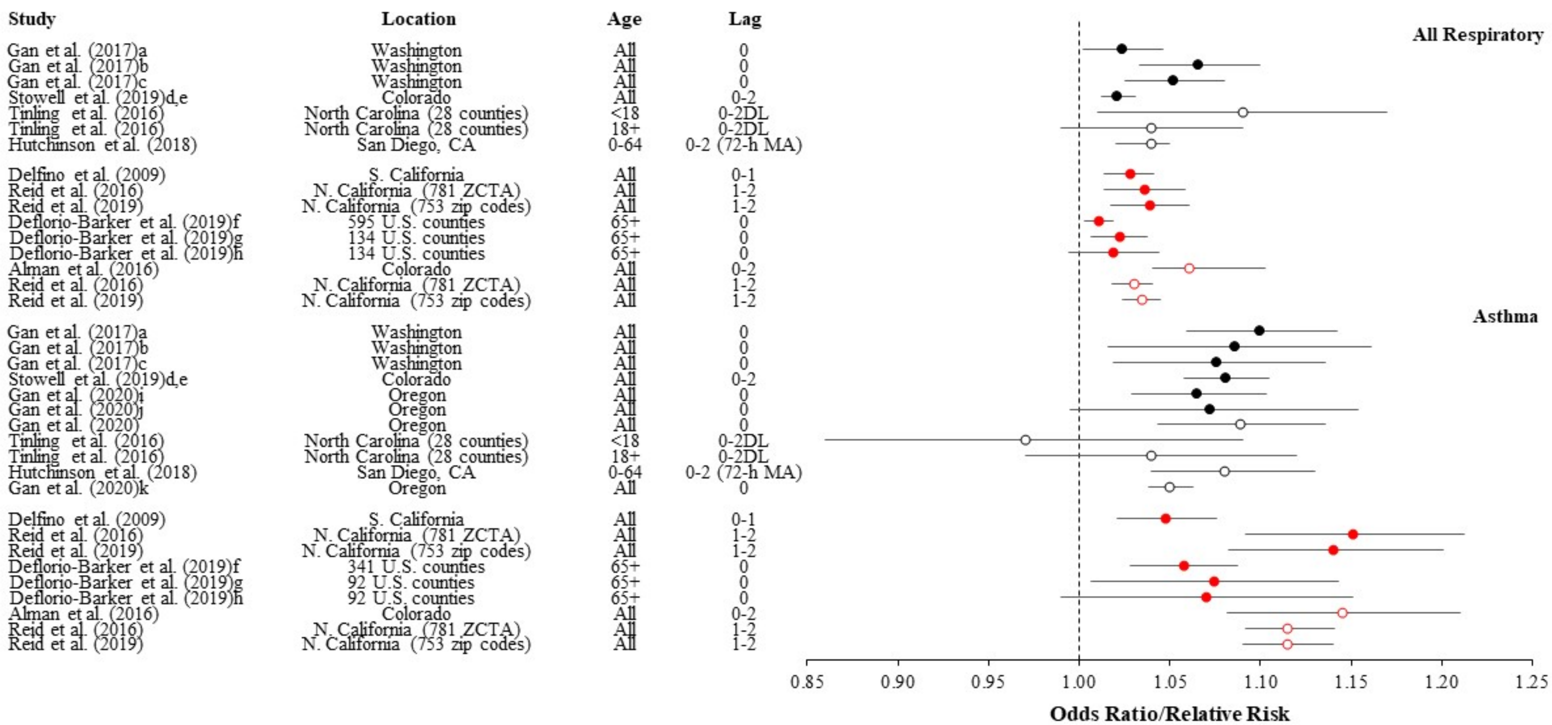
Most studies to date specifically examining the health effects of wildfire smoke exposure focus on respiratory-related outcomes (e.g., emergency department [ED] visits, hospital admissions, and medication use). In addition to the wildfire-specific evidence, there is extensive evidence spanning both experimental and epidemiologic studies focusing on short-term exposures to ambient $PM_{2.5}$ demonstrating a range of respiratory effects, with the strongest evidence supporting relationships with exacerbations of asthma and chronic obstructive pulmonary disease (COPD), as well as respiratory mortality (U.S. EPA, 2019b).

Epidemiologic studies that examined relationships between short-term wildfire smoke exposure and respiratory-related outcomes also provide evidence of positive associations, which are consistent with the results of studies focusing on ambient $PM_{2.5}$. The pattern of associations across studies of wildfire smoke and ambient $PM_{2.5}$ are generally observed within the first few days after exposure [i.e., at lags in the range of 0−2 days; U.S. EPA (2019b); Figure 6-1 and Figure 6-2]. However, there has been limited examination of longer durations of exposure (i.e., exposures over multiple days) for both ambient $PM_{2.5}$ and wildfire exposures and respiratory effects. Initial evidence indicates respiratory effects of larger magnitude due to prolonged exposure (i.e., over a series of days with lags ranging from 0 to 5 days), which is important to consider when examining wildfire smoke exposure that often lasts for many weeks or months (U.S. EPA, 2019b; Rappold et al., 2011).

Across the epidemiologic studies examining respiratory-related outcomes, the most extensive evidence comes from studies examining combinations of respiratory-related diseases (i.e., all

International Classification of Diseases [ICD] codes for the entire range of respiratory diseases or a subset of ICD codes for only a few respiratory diseases, noted as “all respiratory” in Figure 6-1) and asthma. These studies provide consistent evidence of positive associations for both ED visits and hospital admissions when using different exposure indicators, including smoke/wildfire $PM_{2.5}$ or ambient $PM_{2.5}$ (Figure 6-1). Some of the studies evaluated examined whether there was evidence of differential risk across age groups (Appendix Table A.6-1), and although in some instances the magnitude of the association was reported to be larger for a specific age range, the results presented in Figure 6-1, capture the main results of each study.

In addition to the studies that relied on $PM_{2.5}$ to develop the exposure indicator, studies that used alternative exposure indicators or applied different techniques to identify wildfire smoke exposures provide supporting evidence of a relationship between short-term wildfire smoke exposure and respiratory effects. Studies that used the exposure indicator of smoke plume or smoke density reported evidence of consistent positive associations when examining both combinations of respiratory-related diseases and asthma (Wettstein et al., 2018; Rappold et al., 2011). Instead of examining associations with respiratory outcomes, Leibel et al. (2020), in a study conducted in San Diego County, CA, reported evidence of excess ED visits and urgent care visits for combinations of respiratory-related diseases during wildfire periods compared to a control period. Lastly, Liu et al. (2017a) reported a positive association between wildfire $PM_{2.5}$ and respiratory disease hospital admissions when 2 consecutive days of wildfire $PM_{2.5}$ concentrations (i.e., a smoke wave) were greater than 37 $\mu g/m^3$.



DL = distributed lag; CMAQ = Community Multiscale Air Quality; ED = emergency department; GWR = geographically weighted ridge regression; MA = moving average; µg/m³ = micrograms per cubic meter; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm; WRF = Weather Research and Forecasting; ZCTA = ZIP code tabulation area.

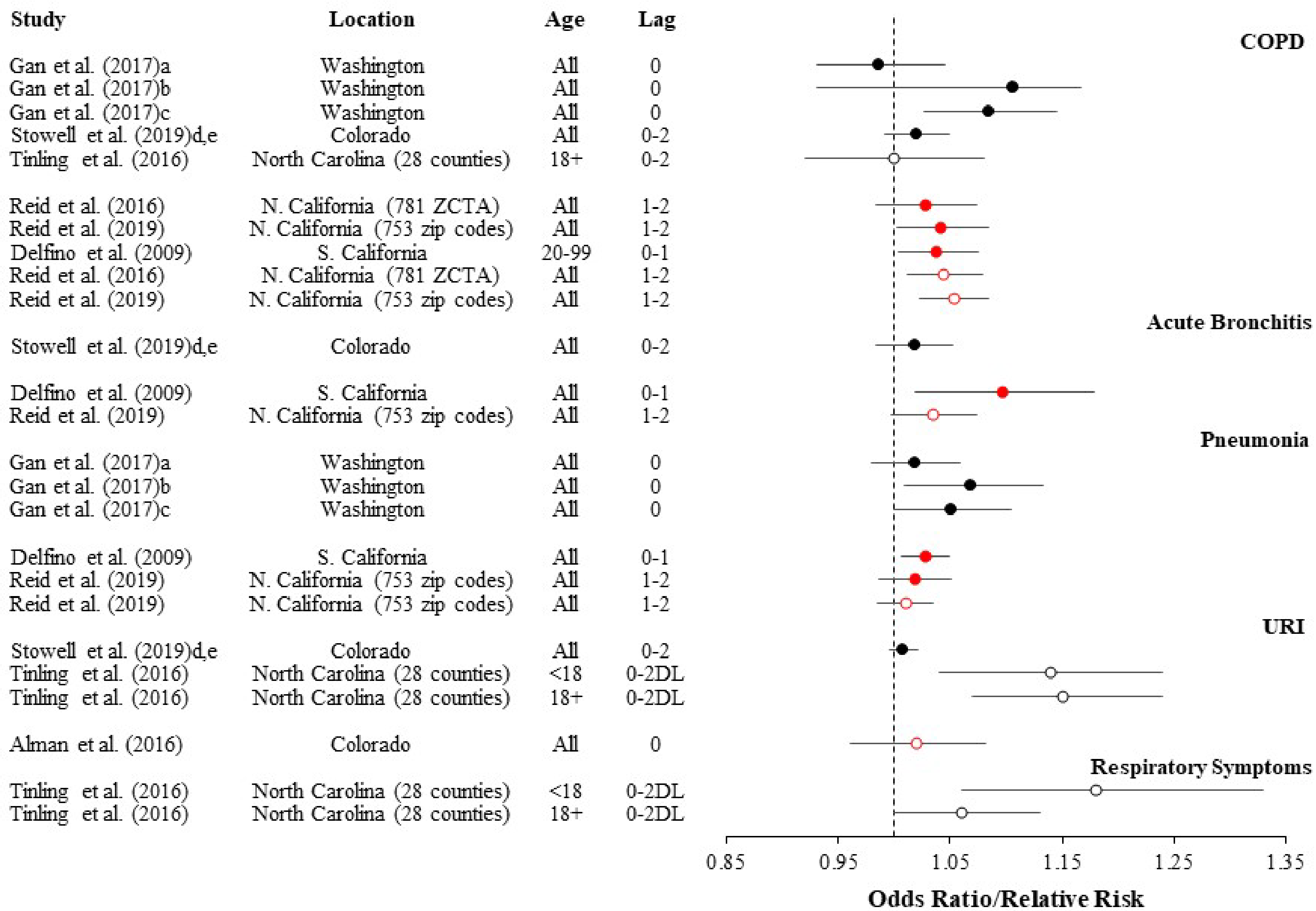
Black circles = studies that used smoke/wildfire $PM_{2.5}$ as the exposure indicator; red circles = studies that used ambient $PM_{2.5}$ measurements as the exposure indicator; solid circles = hospital admissions; open circles = ED visits; odds ratios and relative risks, unless otherwise noted, are for a 10 µg/m³ increase in smoke/wildfire or ambient $PM_{2.5}$ concentrations.

[a]Exposure estimated using WRF-Chem smoke. [b]Exposure estimated from kriging. [c]Exposure estimated using GWR smoke $PM_{2.5}$. [d]Estimate is for a 1 µg/m³ increase in wildfire $PM_{2.5}$. [e]Combination of hospital admissions and ED visits. [f]$PM_{2.5}$ Tot-CMAQ with indicator variable for smoke day. [g]$PM_{2.5}$ Tot-CMAQ-Monitor with indicator variable for smoke day. [h]$PM_{2.5}$ from monitors with indicator variable for smoke day. [i]Outpatient hospital admission. [j]Inpatient hospital admission. [k]Office visit.

Figure 6-1 Odds ratios and relative risks from U.S.-based epidemiologic studies examining the relationship between short-term wildfire smoke exposure and combinations of respiratory-related diseases and asthma emergency department visits and hospital admissions.

Several epidemiologic studies also examined associations between short-term wildfire smoke exposure and other respiratory diseases, including COPD, acute bronchitis, pneumonia, upper respiratory infections (URIs), and respiratory symptoms. Consistent with the studies that examined all respiratory diseases and asthma ED visits and hospital admissions, these studies indicate an increased risk following exposure for a range of respiratory effects (Figure 6-2). Examples include Rappold et al. (2011), which reported positive associations for COPD, pneumonia and acute bronchitis, and URI in North Carolina counties exposed to wildfire smoke estimated by using smoke plume data, as well as Liu et al. (2017b) which reported positive associations for hospital admissions related to COPD and respiratory infection in adults 65 years of age and older exposed to 2 or more consecutive days to wildfire $PM_{2.5}$ concentrations $>37\ \mu g/m^3$.

While the most extensive examination of the health effects associated with wildfire smoke exposure is based on exposure indicators that rely on $PM_{2.5}$, populations can experience exposure to additional pollutants, such as ozone as a result of the mixture of pollutants emitted from wildfires undergoing atmospheric reactions in the presence of sunlight (U.S. EPA, 2019c). There is extensive evidence indicating a relationship between short-term ozone exposure and respiratory effects, including changes in lung function and asthma-related ED visits and hospital admissions (U.S. EPA, 2020a). A recent study by Reid et al. (2019) examined the relationship between ozone produced from wildfire events and respiratory health. The authors reported the strongest evidence of an association with asthma and combinations of respiratory-related ED visits during the fire, but the results across all of the respiratory outcomes examined were attenuated in copollutant models with $PM_{2.5}$ even though the correlation between ozone and $PM_{2.5}$ was low ($r = 0.195$), indicating the complexity in examining health effects associated with both primary pollutants and secondary pollutants from wildfire smoke.



COPD = chronic obstructive pulmonary disease; DL = distributed lag; ED = emergency department; µg/m^3 = micrograms per cubic meter; GWR = geographically weighted ridge regression; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm; URI = upper respiratory infection; WRF = Weather Research and Forecasting; ZCTA = ZIP-code tabulation area.

Black circles = studies that used smoke/wildfire $PM_{2.5}$ as the exposure indicator; red circles = studies that used ambient $PM_{2.5}$ measurements as the exposure indicator; solid circles = hospital admissions; open circles = ED visits; odds ratios and relative risks, unless otherwise noted, are for a 10 µg/m^3 increase in smoke/wildfire or ambient $PM_{2.5}$ concentrations.

[a]Exposure estimated using WRF-Chem smoke.

[b]Exposure estimated from kriging.

[c]Exposure estimated using GWR smoke $PM_{2.5}$.

[d]Estimate is for a 1 µg/m^3 increase in wildfire $PM_{2.5}$.

[e]Combination of hospital admissions and ED visits.

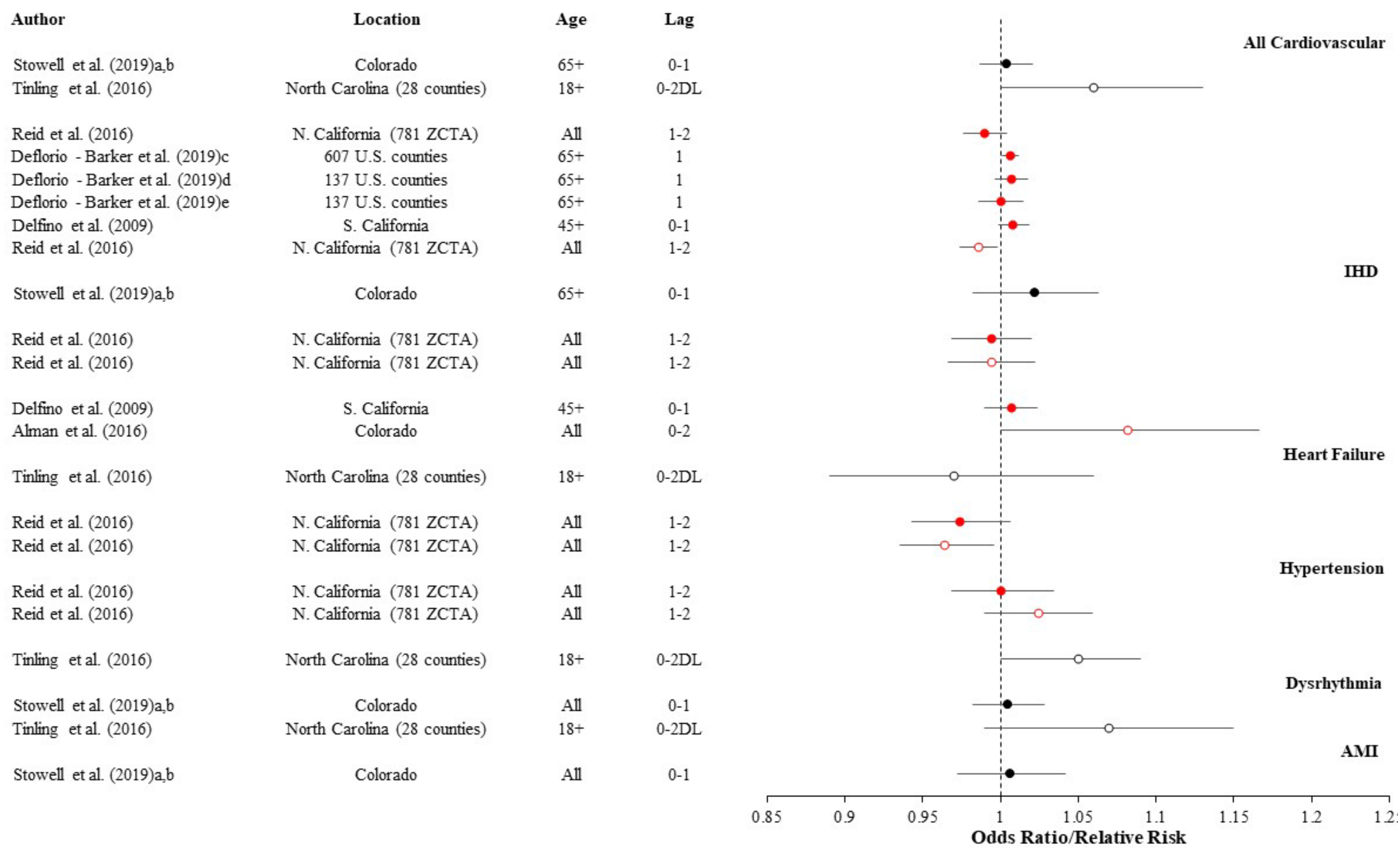
Figure 6-2 Odds ratios and relative risks from U.S.-based epidemiologic studies examining the relationship between short-term wildfire smoke exposure and respiratory-related emergency department visits and hospital admissions.

6.2.2.2 CARDIOVASCULAR EFFECTS

There is extensive experimental and epidemiologic evidence indicating a relationship between short-term $PM_{2.5}$ exposure and cardiovascular effects, particularly for ischemic heart disease (IHD) and heart failure as well as cardiovascular mortality (U.S. EPA, 2019b). While there is a more limited evidence base related to the effects of wildfire smoke exposure on cardiovascular health, compared with respiratory outcomes, these studies report generally positive associations albeit with wide confidence intervals (CIs; Figure 6-3), with the magnitude of associations being relatively consistent to those reported in studies of ambient $PM_{2.5}$ (U.S. EPA, 2019b). However, some studies provide no evidence of an association with cardiovascular outcomes [e.g., (Stowell et al., 2019; Reid et al., 2016b)].

Several studies examining cardiovascular effects used indicators of smoke events to capture the spatial and temporal extent of exposure (Wettstein et al., 2018; Liu et al., 2017a; Rappold et al., 2011). In a study of 561 western U.S. counties, Liu et al. (2017a) did not report any evidence of an association between total cardiovascular-related hospital admissions and smoke wave days (i.e., 2 consecutive days with wildfire $PM_{2.5}$ concentrations >20 µg/m^3) in adults 65 years of age and older. However, in a study of ED visits within 42 North Carolina counties, Rappold et al. (2011) reported an increased risk for combined cardiovascular-related outcomes. When examining, cause-specific cardiovascular outcomes, the authors reported the strongest evidence of an association for heart failure and myocardial infarction. Similarly, in a study of eight California air basins, Wettstein et al. (2018) reported an increased risk of ED visits across combined cardiovascular outcomes at medium ($PM_{2.5}$ concentrations between 10−19 µg/m^3) and dense ($PM_{2.5}$ concentrations >20 µg/m^3) smoke density. The authors observed positive associations in all adults, but associations were larger in magnitude among individuals 65 years of age and older. Additionally, Wettstein et al. (2018) reported a positive association with incidence of stroke among those 65 years and older following smoke exposure. The results of Rappold et al. (2011) and Wettstein et al. (2018) indicate a need for additional exploration of the effect of wildfire smoke exposure on cardiovascular outcomes in older individuals, cause-specific cardiovascular outcomes, and the most appropriate exposure indicator to represent wildfire smoke exposure when focusing on cardiovascular outcomes.

In addition to the outcomes examined through studies of ED visits and hospital admissions, a recent study by Jones et al. (2020) examined out-of-hospital cardiac arrests (OHCAs) attended by emergency medical services. The study was conducted across 14 California counties where daily exposures were classified as light, medium, or high smoke density based on $PM_{2.5}$ estimated from the NOAA Hazard Mapping System. The authors reported positive associations with OHCA at multiple single day lags on heavy smoke density days (i.e., estimated smoke $PM_{2.5}$ concentrations >22 µg/m^3) with the strongest evidence at lag 2 (odds ratio [OR]: 1.70 [95% CI: 1.18, 2.45]). There was no evidence of associations when examining light or medium smoke density days.



AMI = acute myocardial infarction; DL = distributed lag; μg/m^3 = micrograms per cubic meter; GWR = geographically weighted ridge regression; IHD = ischemic heart disease; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 μm; WRF = Weather Research and Forecasting; ZCTA = ZIP code tabulation area.

Black circles = studies that used smoke/wildfire $PM_{2.5}$ as the exposure indicator; red circles = studies that used ambient $PM_{2.5}$ measurements as the exposure indicator; solid circles = hospital admissions; open circles = ED visits; odds ratios and relative risks, unless otherwise noted, are for a 10 μg/m^3 increase in smoke/wildfire or ambient $PM_{2.5}$ concentrations.

[a]Estimate is for a 1 μg/m^3 increase in wildfire $PM_{2.5}$.

[b]Combination of hospital admissions and ED visits.

[c]Exposure estimated using WRF-Chem smoke.

[d]Exposure estimated from kriging.

[e]Exposure estimated using GWR smoke.

Figure 6-3 Odds ratios and relative risks from U.S.-based Epidemiologic studies examining the relationship between short-term wildfire smoke exposure and cardiovascular-related emergency department (ED) visits and hospital admissions.

6.2.2.3 MORTALITY

Across the epidemiologic studies that examine the relationship between short-term wildfire smoke exposure and health effects, to date, only a few U.S.-based studies examine mortality. Although the evidence base for wildfire smoke exposure and mortality from studies conducted in the U.S. is limited to a few studies, there is extensive evidence indicating a relationship between short-term ambient $PM_{2.5}$ exposure and mortality spanning both single and multicity studies conducted in diverse geographic locations, populations with different demographic characteristics, and studies employing different exposure assessment methodologies (U.S. EPA, 2019b).

Doubleday et al. (2020) conducted the most comprehensive assessment of mortality, in a study conducted in Washington state that spanned multiple fire seasons and cause-specific mortality outcomes. Using an exposure indicator that was based on defining smoke days versus nonsmoke days, in a case-crossover analysis the authors reported evidence of a positive association for both respiratory disease (OR: 1.09 [95% CI: 1.00, 1.18], lag 0) and COPD mortality (OR: 1.14 [95% CI: 1.02, 1.26]; lag 0), but no evidence of an association for other mortality outcomes including total (nonaccidental), cardiovascular, and IHD. Unlike Doubleday et al. (2020), which focused on wildfire events and mortality in the same state, Zu et al. (2016) examined daily mortality in Boston, MA and New York, NY in response to long-range transport of $PM_{2.5}$ from wildfires in Quebec, Canada. In time-series analyses comparing the 4-week period of the Quebec wildfires in 2002 with 4-week periods in 2001 and 2003, the authors reported no evidence of increased risk of mortality. Xi et al. (2020) provided evidence to support the results of Doubleday et al. (2020) in a study that examined the relationship between wildfire smoke exposure and mortality among patients managing their end-stage kidney disease with hemodialysis that resided in 253 U.S. counties near at least one of the major wildfires between 2008 and 2012. The authors reported a positive association with all-cause mortality for a 10 $\mu g/m^3$ increase in wildfire-specific $PM_{2.5}$ that was similar in magnitude at both lag 0 (relative risk [RR] = 1.04 [95% CI: 1.01, 1.07]) and for a distributed lag of 0−1 day (RR = 1.05 [95% CI: 1.01, 1.08]), with limited evidence of an association for the cause-specific mortality outcomes examined.

6.2.2.4 UNCERTAINTIES AND LIMITATIONS IN THE HEALTH EFFECTS EVIDENCE

The current state of the science with respect to the health effects of wildland fire smoke exposure stems from the large evidence base demonstrating a range of health effects, including respiratory and cardiovascular effects and mortality, in response to short- and long-term $PM_{2.5}$ exposure. Studies of wildfire smoke exposure report results that are generally consistent with this larger evidence base, but uncertainties remain in gaining a fuller understanding of the health effects of wildland fire smoke. Although this section focuses exclusively on U.S.-based epidemiologic studies, it only represents a fraction of the studies conducted globally, but overall, the results of the U.S.-based studies are consistent with this broader body of evidence.

As noted in Section 6.2.1, there is variability in both the exposure assessment approach and exposure indicator employed across studies, which can complicate the interpretation of results across studies. However, even with this variability, observed health effects are generally consistent across studies, specifically when examining short-term (i.e., daily) smoke exposure. Although there is a general understanding of the health effects associated with short-term smoke exposure, to date, there has been limited investigation and evidence for other exposure durations, including subdaily (i.e., <24-hour average), repeated high exposures over many days, and exposures over multiple fire seasons or years. Additional research focusing on other exposure durations can aid in informing land management strategies, such as prescribed fire; the potential health implications of smoke exposure from single wildfire events, as well as fires over multiple years; and help further enhance public health messaging campaigns. Lastly, although current evidence does not indicate a difference in the health effects between ambient $PM_{2.5}$ exposure and other source-based exposures, such as wildfire smoke (U.S. EPA, 2019b), as wildfires continue to encroach upon the wildland-urban interface (WUI) the complex smoke mixture could change as structures and cars are burned, potentially resulting in different health risks.

6.2.3 SUMMARY

Decades of research on the health effects associated with exposure to ambient air pollution, specifically $PM_{2.5}$ and ozone, provide a strong evidence base for the health effects that could be observed due to short-term (i.e., daily) and long-term (i.e., months to years) exposure to wildland fire smoke. U.S.-based epidemiologic studies, which represent a fraction of the studies conducted globally, examining the health effects associated with short-term wildfire smoke exposure consistently reported evidence of positive associations with respiratory-related health effects, including respiratory and asthma ED visits and hospital admissions, regardless of the exposure indicator used (e.g., wildfire-specific $PM_{2.5}$, smoke density, etc.). Recent studies examining short-term wildfire smoke exposure also provide growing evidence of cardiovascular effects, with more limited evidence for mortality. The same comparison to the ambient air pollution cannot be said about the effects of prescribed fires which are unique in composition, duration, and baseline characteristics of the exposed population. There are few epidemiologic studies that have explicitly examined the health effects associated with exposure to smoke from prescribed fires; therefore, it remains unclear whether there are differential health effects from smoke from prescribed fires compared with wildfires.

Studies of wildfire smoke have not examined the health implications of long-term exposure, such as from wildfires that last multiple months (e.g., the Rough Fire) or over multiple fire seasons, but evidence from studies of long-term $PM_{2.5}$ exposure indicate these types of wildfire events could also result in mortality impacts. Additionally, although there is limited evidence of health effects attributed specifically to ozone produced from wildfire smoke, there is extensive evidence demonstrating health effects from exposure to ambient ozone exposure, indicating potential additional public health impacts from wildfire smoke.

In conclusion, the studies evaluated within this section inform the overall health effects associated with short-term exposures to wildfire smoke. However, there are key study-specific details that complicate the use of data from the evaluated epidemiologic studies to support the main quantitative analyses using BenMAP-CE, presented in Chapter 8. Specifically, complications arise from the different exposure indicators used across studies as well as the geographic locations of some studies. To date, it remains unclear which exposure indicator best represents exposure to wildfire smoke, and the results of studies using some indicators are not conducive to being used to develop a health impact function to estimate public health impacts because they represent a dichotomous exposure (i.e., smoke day or nonsmoke day) versus a continuous exposure (i.e., $PM_{2.5}$ concentrations). This is important to consider because BenMAP-CE estimates the public health impacts associated with changes in air pollutant concentrations, which requires the selection of a concentration-response parameter based on a continuous exposure variable. Additionally, when designing analyses using BenMAP-CE, one of the primary goals is to select concentration-response parameters from epidemiologic studies that are conducted in geographic locations close to, or similar to, the analysis area. In considering the attributes discussed above when identifying studies to be used in the BenMAP-CE analyses within this assessment, of the studies evaluated in this chapter few relied on a continuous variable and were also conducted in a location in close proximity to the case study areas (i.e., TC6 and Rough fires), which precluded their use in the main analyses. The importance in providing criteria around the selection of epidemiologic studies to use in BenMAP-CE type analyses was recently examined in a study by (Cleland et al., 2021). In this study the authors examined the sensitivity of estimated health impacts with respect to different exposure metrics and the concentration-response parameters. Cleland et al. (2021) found that different exposures accounted for the variability in results, but the main sources of the sensitivity in estimated health impacts was the choice of the concentration-response parameter.

Overall, because of these limitations, the main $PM_{2.5}$ analysis presented in Chapter 8 relies on health impact functions derived from epidemiologic studies of ambient $PM_{2.5}$ with sensitivity analyses based on concentration-response parameters from wildfire-specific epidemiologic studies. With respect to ozone, as noted within this section, the overall evidence base of studies examining the health effects of ozone from wildfire events is minimal, resulting in ozone analyses in Chapter 8 relying exclusively on epidemiologic studies of ambient ozone exposure.

6.3 WILDLAND FIREFIGHTER EXPOSURE TO SMOKE DURING PRESCRIBED FIRES AND WILDFIRES

This section is a brief summary of the inhalation health hazards and management implications to wildland firefighters exposed to smoke pollutants at wildfires and prescribed fires. The discussion focuses on exposures to smoke from the combustion of natural fuels (with mention of soil dust) but does not consider smoke exposures from the burning of man-made products encountered by structural and wildland firefighters at WUI fires, or airborne hazards resulting from fires burning across polluted soils.

Like the population as a whole, firefighters experience smoke as both a short-term acute irritation and a long-term chronic health hazard. In the past, firefighters believed smoke was only an inconvenience, irritating the eyes and nose, causing coughing, and occasionally causing nausea and headaches. Many of the exposure limits are established to prevent acute health effects. However, there is evidence there may be serious chronic health effects, and potentially even a reduced life span from long-term exposure to wildland fire smoke (Navarro et al., 2019; Booze et al., 2004).

Because most wildland firefighters are deployed to both wildfires and prescribed fires during their career, an interesting question arises: are wildland firefighters exposed to more smoke, or more hazardous smoke, at wildfires or prescribed fires? This section will address this issue and discuss management implications.

6.3.1 HEALTH HAZARDS OF EXPOSURE TO SMOKE

Wildland fuels are composed of living and dead vegetation, and the burning of this fuel produces smoke. In a complete combustion environment, fuels are consumed by fire and converted mostly to carbon dioxide (CO_2) and water vapor (H_2O) with the release of heat. However, the combustion process in wildland fires is never complete, and incomplete combustion produces dozens of chemicals and hundreds of trace chemicals [Naeher et al. (2007); Reinhardt et al. (2000); Reinhardt and Ottmar (2000); Sandberg and Dost (1990); see Chapter 4 and Chapter 7]. Some of the combustion products may present acute health hazards, others may present chronic health hazards, and some can be both. The main inhalation hazards for wildland firefighters and other personnel at fire camp are carbon monoxide (CO) and respiratory irritants such as PM and several key gases: acrolein, formaldehyde, and to a lesser extent, nitrogen dioxide (NO_2) and sulfur dioxide [SO_2; Navarro et al. (2021); Navarro et al. (2019); McNamara et al. (2012)]. Smoke also includes low concentrations of many other potentially toxic, carcinogenic components such as polycyclic aromatic hydrocarbons (PAHs), and although there is extensive scientific evidence indicating a relationship between long-term exposure to ambient fine particulate matter ($PM_{2.5}$;[8])

[8] $PM_{2.5}$ is the pollutant size most often discussed in context of wildland fire smoke and air quality regulations. PM_4, also known as respirable particulate, is the pollutant size used in the Occupational Health and Safety Administration (OSHA) standards for wildland firefighters.

and lung cancer, the cancer risk of $PM_{2.5}$ derived from wildland fires remains unclear (U.S. EPA, 2019b). Evidence to date indicates a PM occupational exposure limit is likely to be lower than the OSHA standard for respirable nuisance dust (Kim et al., 2018). In addition to PM generated by the fire, wildland firefighters must also be protected against exposure to airborne soil dust, which can result in hazardous exposures to respirable crystalline silica that can contribute to fibrous scarring of the lung resulting is decrease breathing ability.

6.3.2 SMOKE EXPOSURE AT U.S. PRESCRIBED FIRES VERSUS WILDFIRES

A relatively small number of studies have examined acute health effects to firefighters from smoke exposure during prescribed fire and wildfires across individual work shifts and entire fire seasons. Although these studies indicate declines in individual lung function, a general conclusion from these studies is that smoke exposure does not exceed occupational exposure standards most of the time for both fire types (Adetona et al., 2016; Reinhardt et al., 2000; Reinhardt and Ottmar, 2000). However, when there is an exceedance, it is often related to the job assignment and duration of that assignment rather than the type of fire. For example, it has been shown that direct attack, line holding, and extensive mop-up can lead to high smoke exposures (Domitrovich et al., 2017; Reinhardt and Ottmar, 2004; Reinhardt et al., 2000; Reinhardt and Ottmar, 2000). The job assigned to a wildland firefighter and the length of time the individual is carrying out that task will often be the overriding determinant of exceeding occupational exposure limits rather than the fire type.

Most of these studies however, only collected data from individuals during their work shift and did not consider smoke exposures outside their work period, allowing for a potential misinterpretation of the results. Ongoing research by the National Institute for Occupational Safety and Health (NIOSH) is looking into wildland firefighter smoke exposure effects beyond their work shift, considering exposure during their off hours during a work assignment and extending the assessment of health effects to a season and career worth of smoke exposure attributable to wildland fire incidents.

6.3.2.1 DAILY EXPOSURE

Wildland firefighter work shifts average approximately 12 hours with 7 hours on the fire line if assigned to a prescribed fire (Reinhardt et al., 2000). During the work shift, firefighters may be exposed to smoke concentrations that are similar to wildfires, and the exposures will depend on job assignment and duration. However, the firefighter often will return to a clean air environment at their duty station until the next work shift begins, reducing the 24-hour average exposure level. In contrast, firefighters assigned to long-duration project wildfires average 14-hour work shifts with 10 hours on the fire line

(Reinhardt and Ottmar, 2000). The increase in total work shift hours and longer assignments on the fire line increase the duration of exposure to smoke as compared to prescribed fires. An additional concern is the potential for continued exposure after a firefighter returns to a dusty, smoke-filled fire camp following their work shift. Poorly sited fire camps affected by smoke and inversion conditions can increase the 24-hour exposure (Navarro et al., 2021; Navarro et al., 2019; McNamara et al., 2012). For example, if the Air Quality Index (AQI) during off-duty hours exceeds 100 (i.e., orange: unhealthy for sensitive groups) due to PM in the fire camp, this can result in firefighters experiencing continuous exposure to high PM concentrations. As a result, the constant exposure to higher PM concentrations could result in greater long-term health consequences than when the same individual is deployed to a prescribed fire where the duration and concentration of exposure is less. This could have greater long-term health consequences when compared with the same individual deployed to a prescribed fire.

6.3.2.2 CAREER EXPOSURE

Exposure to smoke over the career of an individual will depend on the number and duration of assignments to both wildfire and prescribed fire incidents. Type 1 crews (generally with the most experience, leadership, and availability) will generally be exposed to the most smoke because their primary job is firefighting, and the majority of their work shifts will occur on both wildfires and prescribed fires. Type 2 and Type 3 fire crews (generally with less experience, leadership, and availability than Type 1 crews) are believed to have fewer overall wildfire and prescribed fire assignments resulting in less overall exposure throughout their careers (Navarro et al., 2019). Exposure limits to prevent chronic health effects from career-long exposure patterns have yet to be established for PM exposure from wildfire smoke.

6.3.3 MANAGEMENT IMPLICATIONS

Evidence confirms that wildland firefighters are exposed to a variety of pollutants and respirable crystalline silica at levels that can exceed recommended exposure limits during deployment at both wildfires and prescribed fires. It is common for short-term exposure (usually 15 minutes, where irritation, chronic or irreversible tissue damage does not occur) or maximum exposure limits to be exceeded during brief but intense exposures to smoke at both fire types (Henn et al., 2019). This is often related to job assignment and other associated factors such as the site fuel model, wind orientation (downwind being higher), crew type, relative humidity, type of attack, and wind speed. The resulting acute or short-term effects such as eye or respiratory irritation require management intervention to reduce the exposure. Recent National Wildfire Coordinating Group guidance on smoke exposure during wildland firefighting recommended a reduction in the acceptable exposure limit on a shift-average basis, and this may be adjusted further as ongoing research is completed.

Smoke exposure, whether from a prescribed fire or wildfire, is a health and safety issue for firefighters, requiring fire training classes, and annual refresher courses. A range of literature is available to better understand the potential acute and chronic effects that may result from exceeding smoke exposure limits. The literature also covers how best to manage or limit exposure and inform crew personnel (Sharkey, 1997).

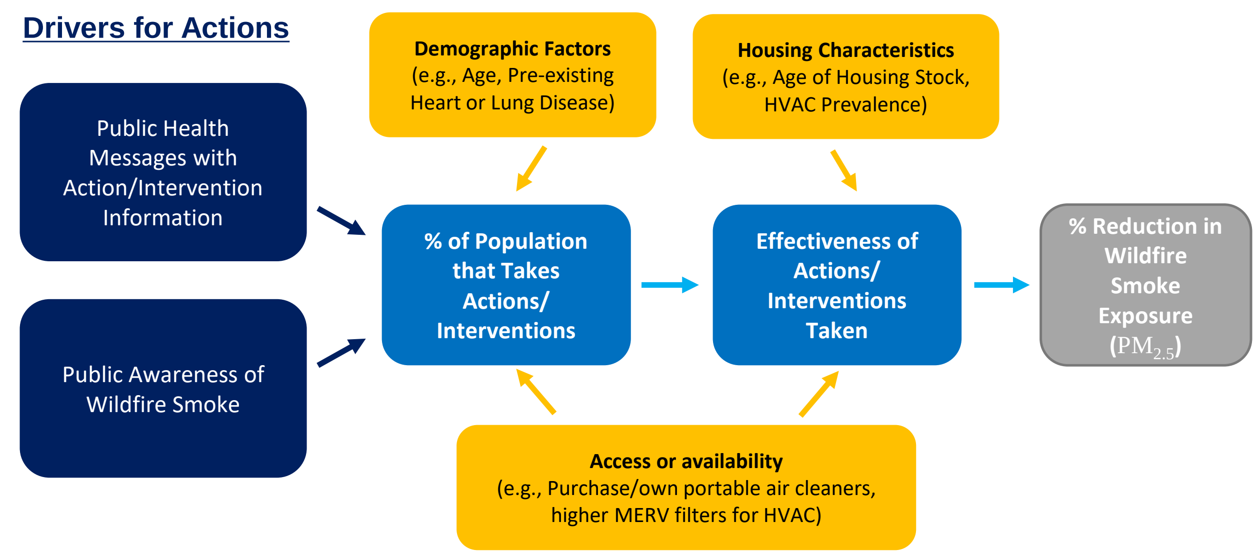
6.4 MITIGATION OF PRESCRIBED FIRE AND WILDFIRE SMOKE EXPOSURE TO REDUCE PUBLIC HEALTH IMPACTS

Characterizing exposure to wildfire smoke is key to examining health effects, and epidemiologic studies have typically used levels of smoke or the concentration of $PM_{2.5}$ in outdoor ambient air as the exposure estimate (Section 6.2). In addition to these studies focusing on relationships between wildfire smoke exposure and health outcomes, several studies have examined individual and community actions that can be taken to reduce or mitigate exposure to smoke during wildfire events. For example, people spend most of their time indoors at home, work, or school (Klepeis et al., 2001), where smoke exposure can be reduced relative to outdoors depending on factors such as building ventilation and use of air filtration (U.S. EPA, 2020c). This section describes a framework for, and the type of data needed to quantify, the potential public health benefits of actions that reduce or mitigate smoke exposure. These actions, also often referred to as interventions, consist of some form of individual behavioral change, such as staying indoors with windows and doors closed or reducing activity levels; the use of personal protective measures, such as a respirator; using a portable air cleaner indoors or the extended use of a heating, ventilation, and air conditioning (HVAC) system equipped with a high efficiency particle filter; or community-level interventions (e.g., providing clean air spaces). While each of these actions can reduce wildfire smoke exposure for an individual by some percentage, the overall fraction of the population taking preventative measures depends on many factors, such as population demographics, access to interventions, whether smoke is visible or can be smelled, and perceived risk of smoke exposure, all of which may also be impacted by public health messaging campaigns. Of these factors, the timing, content, and extent of public health messaging campaigns may represent a major difference in how prescribed fire and wildfire events are managed. However, whether there are differences in the percent of the population taking actions between the fire types has not been assessed and is an important knowledge gap.

The following sections provide an overview of a framework that captures the factors that need to be accounted for to estimate the potential reduction in overall smoke exposure for a population during both prescribed fire and wildfire events. Additionally, these sections evaluate and summarize results from studies that provide data on how often people take action during smoke events and the exposure reduction that occurs from those actions (see Appendix Table A.6-2 for details on study inclusion criteria). The information presented within these sections will be used to provide a crude estimate of the potential reduction in health impacts in the case study areas that could be achieved through specific actions or interventions to reduce smoke exposure (see Section 8.3.3).

6.4.1 FRAMEWORK FOR ESTIMATING THE IMPACT OF ACTIONS TO REDUCE SMOKE EXPOSURE

Estimating potential reductions in wildfire or prescribed fire smoke exposure that a population could experience as a result of actions or interventions is based on a series of events and assumptions (Figure 6-4). The overall exposure reduction for a population will be determined by the likelihood and/or ability to take a particular action combined with the exposure reduction effectiveness of the action. There are multiple factors that influence both of these elements, but two major factors for any action is the awareness of the need (dark blue boxes in Figure 6-4) and the ability (yellow boxes in Figure 6-4) to take exposure reduction actions. Community demographics and socioeconomic characteristics likely affect both factors and may result in the communities most at risk to smoke exposure harder to reach and less able to take exposure reduction actions. There has been no comprehensive analysis of the interplay of these factors, and such analysis is out of the scope of this assessment, but individual studies have examined some aspects of how demographics and socioeconomic status affect the likelihood of taking action and will be discussed below.



HVAC = heating, ventilation, and air conditioning; MERV = minimum efficiency reporting value; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm.

Figure 6-4 Considerations for estimating potential reduction in wildfire smoke exposure through actions and interventions.

Information dissemination, specifically focusing on the potential risks of wildfire smoke exposure and actions a population can take is the initial step that can ultimately dictate whether individuals take actions to reduce exposure. However, public health messaging on its own is not enough if the proper information is not conveyed. The limited assessment of public health messaging campaigns has shown that only 14−46% of wildfire-smoke-related messages disseminated by government and media entities indicate the individual and administrative actions that can be taken to reduce smoke exposure (Van Deventer et al., In Press). Additionally, people may take actions to reduce exposure regardless of public health messaging campaigns as a result of general awareness of the presence of wildfire smoke (Kolbe and Gilchrist, 2009; Künzli et al., 2003). Both public health messaging and general awareness of smoke factor into the percent of the population that takes an action or institutes an intervention to reduce exposure.

Whether or not people take actions to reduce smoke exposure can depend on their knowledge of the potential impact of environmental exposures on their health (Rappold et al., 2019) as well as their personal experiences with smoke, perceptions of risk, and level of self-efficacy (Hano et al., 2020). This is often a reflection of the age or underlying health status of an individual or family member. While not directly factored into an estimation of potential exposure reductions due to various actions, it is important to acknowledge that the population-level response to taking actions could vary within the population based on sociodemographic factors. Additionally, the ability to take actions depends on the accessibility and availability of interventions, such as portable air cleaners and high efficiency HVAC filters. Access and availability may depend on having the financial means to purchase interventions, but also whether programs for providing interventions exist within an area. Even low-cost interventions can have barriers to their use, such as staying indoors with doors and windows closed without air conditioning when smoke and high temperatures co-occur.

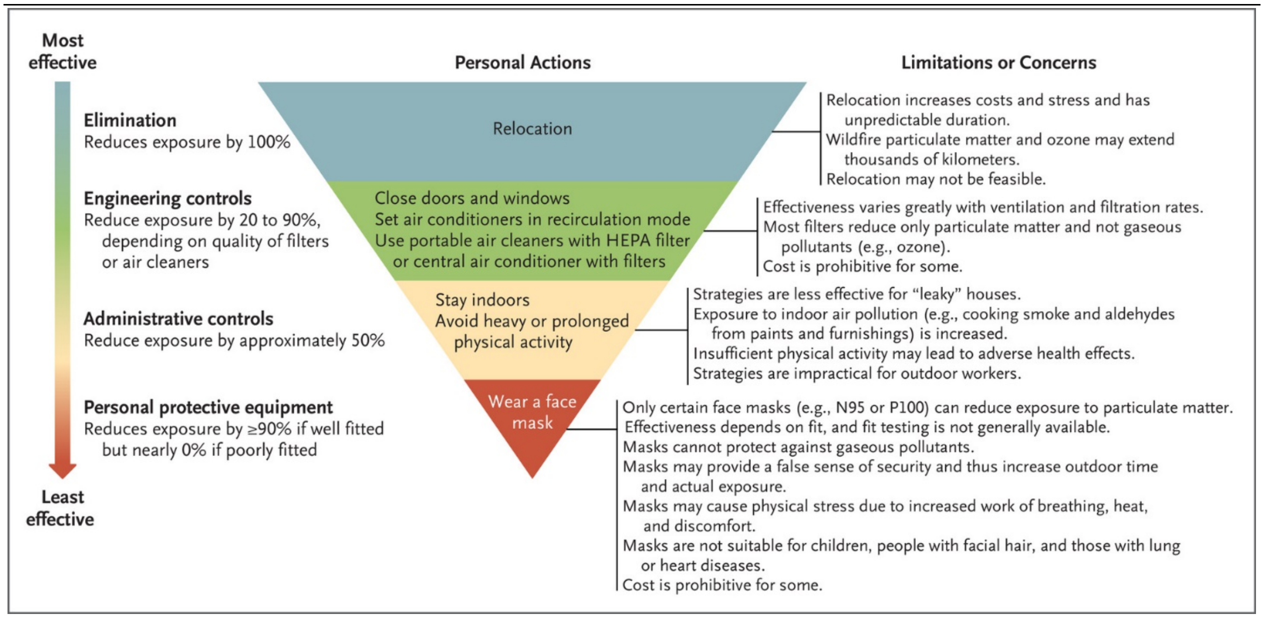
There are numerous actions people can take to reduce exposure to smoke, with a large degree of variability in the efficacy of each (Xu et al., 2020; Laumbach, 2019). The primary focus for several actions is reducing indoor $PM_{2.5}$ concentrations while at home where people spend most of their time. Housing characteristics, such as age of the home and presence and type of HVAC system, influence the infiltration of particles indoors under normal conditions, and also influence the efficacy and availability of these actions for reducing smoke exposure in homes (Davison et al., 2021; Joseph et al., 2020; U.S. EPA, 2020c). Therefore, housing characteristics of the geographic area impacted by smoke is another important factor, and if variability in the housing stock is not accounted for in some way then estimates of exposure reduction could be under- or overestimated.

Embedded within the potential actions and interventions a population within a defined geographic area may take to reduce smoke exposure are a series of operating conditions that can directly influence the overall percent reduction in smoke, specifically $PM_{2.5}$. These conditions, such as how often a building's HVAC system runs, and whether a high-efficiency filter and/or portable air cleaner is used, are important to consider when constructing potential exposure reduction scenarios. Similar to housing characteristics, the distribution of these operating conditions can vary throughout the population being examined, may depend on public awareness of the presence of smoke, and can contribute to over- or underestimation of the overall exposure reduction of a particular action.

At each step of the process of developing scenarios to estimate the influence of actions to reduce smoke exposure, there are decision points that rely on both data from published studies and assumptions regarding the population being examined. Outlining these decision points will allow for a clear articulation of the factors that influence each exposure reduction scenario and the ability to construct scenarios meant to represent the range of exposure reductions that could be experienced.

6.4.2 INDIVIDUAL AND COMMUNITY ACTIONS TO REDUCE SMOKE EXPOSURE

In identifying the overall percent reduction in smoke exposure that can be achieved in response to public health information dissemination, the key factors to consider are the actions that can be taken at both the individual and community level and the effectiveness of those actions in reducing exposure, particularly to $PM_{2.5}$. Recent publications by Xu et al. (2020) and Laumbach (2019) provide overviews of the actions that individuals can take to reduce smoke exposure, which are delineated into four broad categories according to the hierarchy of controls traditionally used for occupational hazards (NIOSH, 2015): elimination, engineering controls, administrative controls, and personal protective equipment. As depicted in Figure 6-5, there are a range of smoke exposure reductions that can be achieved depending on the approach instituted, but each have limitations and concerns that should be considered. This section characterizes the broader body of studies that examined the effectiveness of information dissemination and various exposure reduction actions, which collectively provide evidence that supports the range of smoke exposure reductions that could be achieved if individuals are well informed and take the necessary steps to reduce/mitigate exposure.



HEPA = high-efficiency particulate air.

Source: From Xu et al. (2020) Wildfires, Global Climate Change, and Human Health, Vol. 383, Page 2178. Copyright © (2020). Massachusetts Medical Society. Reprinted with permission from the Massachusetts Medical Society.

Figure 6-5 Summary of individual-level wildfire smoke exposure reduction actions and their effectiveness.

6.4.2.1 FACTORS THAT INFLUENCE TAKING ACTIONS TO REDUCE SMOKE EXPOSURE

Several studies examined how awareness of smoke, whether by direct observation or through public service announcements (PSAs), can translate into a population taking exposure reduction actions. Most of the information available on the effectiveness of PSAs stems from studies conducted in California or in Australia where wildfires impact large population centers and occur on a near yearly basis. Of the available studies, all were conducted in the context of wildfire with no information currently available on the likelihood of actions taken in response to prescribed fire smoke. Studies on prescribed fire have focused on the factors governing the tolerance of smoke and optimal risk communication (Olsen et al., 2017; Blades et al., 2014), rather than exposure reduction actions taken in response to smoke.

Studies of exposure reduction actions are often conducted through retrospective surveys of communities impacted by major wildfires to determine population awareness of smoke, PSAs or other health risk communications, and the resulting actions as a function of the messaging medium (Kolbe and Gilchrist, 2009; Mott et al., 2002), content (Sugerman et al., 2012), and the characteristics of the community (Kolbe and Gilchrist, 2009). These studies have investigated the impact of population demographics (e.g., age, gender, income level, etc.), pre-existing conditions, and experiencing symptoms on the type and extent of exposure reduction action taken.

Across studies that examined PSAs, in most communities the awareness of PSAs was high (74−88% of those surveyed recalled a PSA) with many people (43−98%, Appendix Table A.6-2) taking some exposure reduction action in response to a PSA, but the most effective method of communication varied by community. Television was the most effective communication medium in studies conducted in San Diego [77%; Sugerman et al. (2012)] and Australia [68%; Kolbe and Gilchrist (2009)] while radio was the most effective medium in a rural tribal community in northern California (Mott et al., 2002). However, in this community a wide variety of information sources (e.g., the medical establishment, friends and family, and the workplace) were recalled in greater frequency than television, demonstrating the impact of the community type on the most effective method of risk communication.

When considering the implications of the demographic composition of a population, older adults were less likely to be aware of PSAs, with only 58% of those over 75 years of age aware of the PSA compared with 74% for the entire population (Kolbe and Gilchrist, 2009). Those with pre-existing conditions (81%) were also found to be slightly less likely than those without a pre-existing condition (85%) to be aware of PSAs (Mott et al., 2002). While it is important to be aware of the message, message comprehension is also extremely important when considering whether individuals take the necessary actions to protect themselves. Sugerman et al. (2012) observed that message comprehension was reduced in those that did not speak the primary language or when the message was too technical in nature (e.g., stay inside vs. run HVAC system more often).

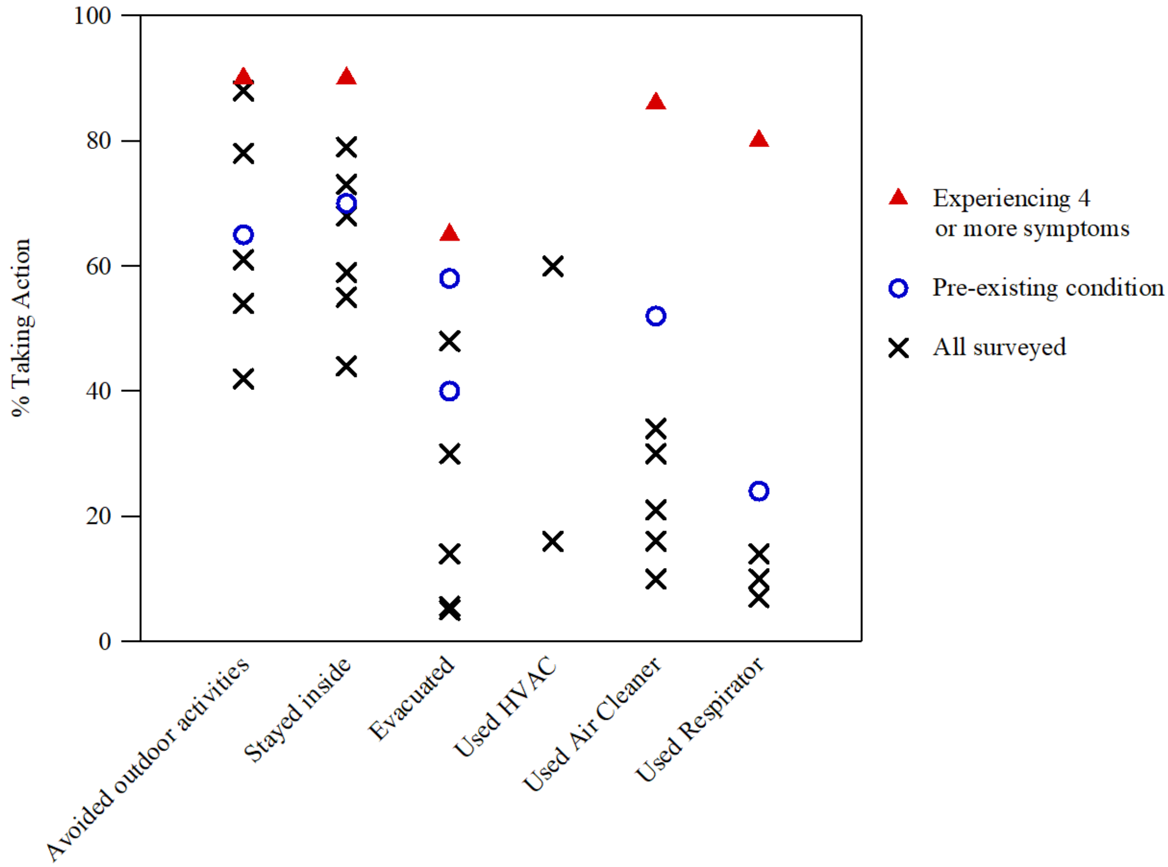
Overall, across studies it was found that most people aware of a PSA took some action to reduce exposure [66−98%; Sugerman et al. (2012); Kolbe and Gilchrist (2009); Mott et al. (2002)]. The awareness of smoke also prompted people to act. For example, in Australia of the 76% of the population that took an exposure reduction action 43% did so because of the PSA and 28% because of the presence of smoke (Kolbe and Gilchrist, 2009). Furthermore, the percentage of people reducing outdoor activities, closing doors and windows, and evacuating was similar between those responding to the smoke and those responding to a PSA (Kolbe and Gilchrist, 2009). The more technical actions were much more likely to be used by those aware of the PSA than those that were not, like using a mask (8.1% aware of PSA vs.

1.3% not aware) or using ceiling fans [10.5% aware of PSA vs. 2.9% not aware; Kolbe and Gilchrist (2009)].

The most commonly used actions were those that are easiest to carry out, including reducing or avoiding outside activity, and staying inside or closing windows and doors (Figure 6-6, Appendix Table A.6-2). On average for the total surveyed population, the least likely actions were using an air cleaner (10−34%) or respirator (7−14%). Sugerman et al. (2012) found that more technical actions (e.g., using home air conditioning, using high-efficiency particulate air [HEPA] air filtration, wearing an N95 mask during ash clean up) were least likely to be done in part due to a poor recall of the PSA and a difficulty understanding the PSA. Accessibility to measures that can reduce exposure, such as an HVAC system, air cleaner or respirators/masks, while not formally characterized in any of the studies evaluated, may significantly impact the probability of an individual taking an exposure reduction action.

As depicted in Figure 6-6, there is a wide distribution in the percentage of each population taking action to reduce or mitigate smoke exposure, which is in large part due to the different populations surveyed. People actively experiencing symptoms from wildfire smoke were much more likely to take an action than the general population. This is most striking for actions that require equipment, like air cleaners or respirators. For example, Rappold et al. (2019) reported that 86% of people with four or more symptoms used an air cleaner versus 24% of the average population (averaged across all studies).

Most studies provided some indication of the smoke concentration and duration in the community by either reporting $PM_{2.5}$ or PM_{10} (particulate matter with a nominal aerodynamic diameter less than or equal to 10 µm) concentrations. For example,Mott et al. (2002)] reported PM_{10} concentrations, and if assuming $PM_{2.5}$ is 85% of PM_{10} concentrations as detailed in Lutes (2014), this equated to 2 days of $PM_{2.5}$ >425 µg/m^3 and 15 days of $PM_{2.5}$ >128 µg/m^3,. However, inconsistent reporting prevents the determination of a clear association between smoke exposure (duration or peak concentration) and the probability of taking a particular action. The level and duration of smoke exposure are likely major determinants in what actions a community will take and are important factors to be considered in future studies.



HVAC = heating, ventilation, and air conditioning.

Note: All surveyed is from the general population indiscriminate of health history or status. Data presented is from Rappold et al. (2019); Richardson et al. (2012); Sugerman et al. (2012); Kolbe and Gilchrist (2009); Mott et al. (2002).

Figure 6-6 Percentage of the population taking a specific exposure reduction action as a function of the characteristics of the surveyed population.

6.4.2.2 EFFECT OF ACTIONS/INTERVENTIONS ON REDUCING $PM_{2.5}$ EXPOSURE CONCENTRATIONS

The effectiveness of various actions or interventions in reducing $PM_{2.5}$ exposure concentrations has been quantified in several studies. However, studies that examined $PM_{2.5}$ exposure reduction actions during wildfire or prescribed fire smoke periods were limited. More studies examined the effect of actions for typical ambient $PM_{2.5}$ conditions. Most relevant studies evaluated the effectiveness of portable air cleaners and more efficient HVAC system filters with residential $PM_{2.5}$ monitoring, with a few additional studies conducting modeling of residential buildings to estimate effectiveness (see Appendix Table A.6-3). Studies of nonresidential building types were limited to a few studies focusing on office buildings, with no other building types (e.g., schools) examined. Studies that examined the effectiveness of masks for reducing exposure to particles in air have primarily been conducted for occupational exposure and other purposes (Allen and Barn, 2020), and not specifically for examining their effect in reducing smoke exposure within the general population.

Reviews by Xu et al. (2020) and Laumbach (2019) compare percent reductions for various actions that could be taken to reduce or mitigate smoke exposure. Elimination of smoke exposure can be achieved by relocation (exposure reduction = 100%), while engineering controls such as closing windows and doors or indoor air filtration can also be effective (20−80% exposure reduction), as are administrative controls such as staying indoors and avoiding outdoor activities (~50% exposure reduction). Additionally, both Xu et al. (2020) and Laumbach (2019) noted that wearing N95 or P100 masks can be 90% effective or more, but only if properly fitted along with other limitations (e.g., not suitable for children). The results reported in Xu et al. (2020) and Laumbach (2019) are generally consistent with the levels of effectiveness for the different actions reported in recent studies.

U.S. EPA (2018) reviewed residential measurement studies that used portable air cleaners and central HVAC system filters to reduce indoor $PM_{2.5}$ exposures overall, not $PM_{2.5}$ specific to wildfire smoke. Portable air cleaners were found to substantially reduce indoor concentrations of PM of both indoor and outdoor origin, often reducing indoor $PM_{2.5}$ concentrations by around 50% on average.

Residential measurement studies that examined portable air cleaner effectiveness in homes during wildfire smoke events (Barn et al., 2008; Henderson et al., 2005) also reported a similar percent reduction in indoor $PM_{2.5}$ concentrations compared with the elevated outdoor $PM_{2.5}$ concentrations during these events. Barn et al. (2016) also reviewed many of the same studies as U.S. EPA (2018) and concluded that portable air cleaners can reduce indoor $PM_{2.5}$ concentrations by 32−88% and recommended their use during fire events.

U.S. EPA (2018) also noted a few residential measurement studies that showed higher efficiency central HVAC system filters such as those rated minimum efficiency reporting value (MERV) 13 or above can reduce indoor $PM_{2.5}$ concentrations. Singer et al. (2017) reported a 90% reduction in $PM_{2.5}$ using HVAC filtration with high efficiency MERV filters in a single test house in California during typical ambient $PM_{2.5}$ concentrations, which was comparable to running a portable air cleaner in the home. However, results from a recent study by Alavy and Siegel (2020) showed actual in-home effectiveness of HVAC filtration for $PM_{2.5}$ was much lower (average ~40%) and varied widely across homes even for filters with the same MERV rating depending on the home. Filter performance was strongly linked to home- and system-specific parameters including ventilation rate and system run time.

Of the studies evaluated, Reisen et al. (2019) is the only available residential measurement study that examined the effectiveness of closing windows and doors during a smoke event. However, the study only included four homes in Australia that experienced smoke due to a prescribed fire. Simple infiltration modeling of the measurements showed that remaining indoors with windows and doors closed reduced exposure to peak $PM_{2.5}$ concentrations by 29 to 76% across the homes and that a tighter house, in terms of reduced ventilation, provided greater protection against particle infiltration.

A comprehensive residential modeling study by Fisk and Chan (2017b) compared central HVAC system filtration and portable air cleaners for six different home type scenarios during a wildfire smoke event in California. The combined effect of continuous HVAC fan use with a high efficiency (MERV 12) filter and continuous portable air cleaner was most effective (62% reduction in $PM_{2.5}$), while continuous portable air cleaner use in homes without forced-air HVAC systems provided 45% reduction in $PM_{2.5}$ concentrations.

Although most of the studies conducted focused on examining the effectiveness of interventions in residential locations, a few studies examined the effectiveness of HVAC systems and filters in office buildings during wildfire smoke events. Stauffer et al. (2020) compared offices with and without portable air cleaners during a wildfire season. They reported 73 and 92% reduction in $PM_{2.5}$ concentrations indoors with portable air cleaner use for daytime and nighttime, respectively. Pantelic et al. (2019) reported a 60% reduction in $PM_{2.5}$ for a mechanically ventilated office building and higher efficiency filters compared with a naturally ventilated building during a wildfire.

Fisk and Chan (2017a) conducted a modeling study comparing improved filtration using filters in residential forced-air systems and/or portable air cleaners for homes and higher efficiency filters in commercial buildings in three U.S. cities (Los Angeles, Houston, Elizabeth, NJ) for ambient $PM_{2.5}$ concentrations. Additional higher efficiency filtration in other buildings only slightly reduced overall $PM_{2.5}$ exposures due to the amount of time spent in these locations compared with at home.

In summary, although limited in number, studies that examined the effectiveness of actions or interventions to reduce $PM_{2.5}$ exposure provide relevant data for considering the potential implications of public health messaging campaigns and the most effective actions to recommend to the public to reduce exposure to wildfire smoke (see Appendix Table A.6-3). Portable air cleaners were shown to reduce residential indoor $PM_{2.5}$ concentrations from ~40−90%, depending on the study and home characteristics. Increasing filtration efficiency in residential forced-air systems and/or running the system more/continuously can also reduce indoor $PM_{2.5}$ concentrations by a similar percent, but data from these studies were more variable between homes and efficiency of the filters. The data also suggest office buildings with high-efficiency filters in HVAC systems or that use portable air cleaners can achieve a similar reduction in indoor $PM_{2.5}$ concentrations (~60−90%) as homes. Lastly, there is limited data to fully assess the effectiveness of only closing windows and doors and staying inside as a means to reduce wildfire smoke exposure.

6.4.3 ESTIMATING THE OVERALL EXPOSURE REDUCTION TO WILDFIRE SMOKE FOR INDIVIDUAL-LEVEL ACTIONS

Although the available data on individual and community actions that can be taken to reduce smoke exposure is currently limited, the data detailed within this section provides information on many of the factors to consider that are depicted in Figure 6-4 for estimating the potential impact of actions/interventions on reducing $PM_{2.5}$ exposure from wildfire smoke. An approximation of the overall percent reduction in $PM_{2.5}$ exposure for a population that could be achieved by individual-level actions can be estimated by combining the data on the likelihood of taking actions in response to smoke with the effectiveness of the various actions (Table 6-1). However, across the studies evaluated there was a wide range of data on both the likelihood and effectiveness of exposure reduction actions (see Appendix Table A.6-2 and Appendix Table A.6-3). Therefore, the values reported in Table 6-1 represent the average with standard deviation (SD) across studies for likelihood and effectiveness of the different actions and interventions.

Table 6-1 Summary of data available for various exposure reduction actions.

Exposure Reduction Action	Likelihood of Taking Action in Response to Wildfire[a] Mean ± SD	Effectiveness of Action[b] Mean ± SD	Average Overall Exposure Reduction[c]
Reduced activity	64.4% ± 18.5	No data	--
Stayed inside	63.0% ± 12.8	49.8% ± 22.8	31.4%
Ran home HVAC system	38.0% ± 31.1[d]	64.0% ± 32.8	24%
Evacuated	20.5% ± 18.4	100%	24%
Used air cleaner	22.2% ± 9.9	63.7% ± 21.0	14%
Used respirator	9.5% ± 3.3	No data[e]	--

HVAC = heating, ventilation, and air conditioning; SD = standard deviation.

[a]From studies in Appendix Table A.6-2 for respondents regardless of health history or status.

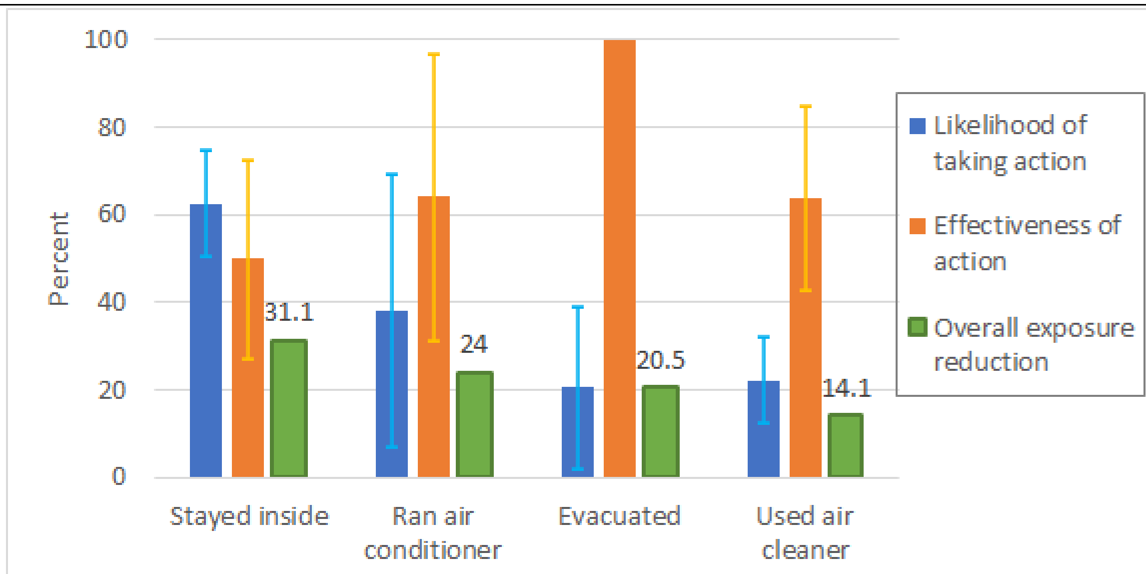
[b]From studies in Appendix Table A.6-3.

[c]Average likelihood of taking the action multiplied by the average effectiveness of the action.

[d]May include the use of other air conditioning systems in addition to HVAC systems.

[e]No data available on the effectiveness of respirators for reducing wildfire smoke exposure.

For each exposure reduction action, the average overall percent exposure reduction, at the population level, was calculated by multiplying the average likelihood of taking the action by the average effectiveness of the action. Although simplistic, this approach provides an initial comparison that shows the more effective actions are generally less likely to be used, resulting in a lower overall exposure reduction (Figure 6-7). For example, the data on portable air cleaner use showed an average ~64% reduction in $PM_{2.5}$ but the likelihood of using them was ~22% on average, resulting in an overall exposure reduction of ~14%. The exposure reduction action with the highest average overall percent reduction was staying inside (~31%), due to the greater likelihood of people taking this action (~63% on average) and its relative effectiveness (~50% on average). It should be noted that combining these two types of study data assumes a reasonable match between the interventions reported in the survey studies of PSA effectiveness and those evaluated in the effectiveness studies, which may not be appropriate in all cases.



$PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm.

Figure 6-7 Comparison of estimated percent overall $PM_{2.5}$ exposure reduction by action.

6.4.4 UNCERTAINTIES AND LIMITATIONS IN ESTIMATING EXPOSURE REDUCTION TO WILDLAND FIRE SMOKE

While it is clear from Figure 6-7 that there are actions that can be taken at the individual level that could substantially reduce overall population exposure to wildfire smoke, there are multiple assumptions and limitations that should be considered in the process of using this information to estimate the potential public health benefit of messaging campaigns. The studies conducted to date examining actions and interventions to reduce wildfire smoke exposure, specifically $PM_{2.5}$, have been conducted over a limited geographic scale, and therefore may not be transferrable across locations. The limited geographic scale of available studies could be accounted for by including location-specific information in an analysis, such as detailed information on the housing stock (e.g., age, type of HVAC, etc.), population demographics, and community characteristics (e.g., urban vs. rural). However, awareness of the need to take action and the ability to reduce smoke exposure in relationship with socioeconomic status may change over time and the studies reviewed here may no longer be applicable. As wildfire smoke impacts become more widespread, communities are increasingly aware of the health impacts from smoke exposure and approaches to reduce exposure (Davison et al., 2021). Recently, lower cost approaches to reduce smoke exposure, like the do-it-yourself box fan and furnace filter, have been promoted by many public health agencies. These lower cost air cleaners, concomitant with air cleaner distribution programs, may have led to increasing air cleaner usage, especially in lower income communities.

Additionally, the average overall exposure reductions presented do not account for the likelihood that taking actions may differ significantly between wildfire and prescribed fire smoke events, due to potential differences in public health messaging campaigns for each fire type (e.g., PSAs in preparation for prescribed fires are not uniform across locations). These potential differences between wildfire and prescribed fire may also include differences in the effectiveness of an action or intervention due to variability in $PM_{2.5}$ concentrations. Specifically, the effectiveness may be reduced at the very high $PM_{2.5}$ concentrations associated with large wildfire events.

Perhaps the greatest difference in potential smoke exposure reductions can be associated with the different level of public awareness of smoke for the two different types of fires. Smoke from prescribed fires may be present for a short duration, as little as several hours, and at lower concentrations that may not be noticeable. Alternatively, wildfires may lead to prolonged high smoke concentrations with noticeable odor and visibility impacts. Wildfires are often reported by the local news media, which may include public service announcements about actions to reduce smoke exposure. Additionally, most major wildfire incidents have an Air Resource Advisor (ARA) that develops and disseminates information on smoke forecasts, air quality, and messaging to address public health concerns. The ARA generates daily smoke reports that are posted online on InciWeb [https://inciweb.nwcg.gov; NIFC (2021)], state smoke blogs, and on fire information boards through impacted communities. Prescribed fires are not as widely publicized and depending on the state or local regulations may be conducted without any notifications or alerts to the surrounding community. Therefore, public awareness of prescribed fires may be very limited, greatly reducing the potential for exposure mitigation actions to be taken.

Another difference is that wildfires and prescribed fires often occur at different times of the year when residential ventilation rates may vary. In the study areas and many parts of the western U.S., wildfires largely occur during July through October (Jaffe et al., 2020; Ryan et al., 2013). Prescribed fires are often done in the late fall or early spring during cooler weather (see Section 3.3.2.1), while pile burns of mechanically thinned biomass are typically done in the winter months. These different seasons of the year for the fire types may have ambient conditions that lead to different behaviors with respect to home ventilation (Marr et al., 2012; Yamamoto et al., 2010). In areas where residential air conditioning systems are not prevalent, wildfires may frequently coincide with time periods when ventilation rates may be highest as windows and doors would be opened to cool the indoor environment. In contrast, in areas where air conditioning systems are prevalent, prescribed fires may coincide with time periods when ventilation rates may be greater due to window and door opening during the more temperate months.

In addition to the lack of data on differences in public awareness, likelihood of taking actions, and smoke exposure between fire types noted above, there are also data gaps that complicate the ability to quantitatively estimate the overall exposure reduction that could be achieved. Within this assessment, a crude approach is taken to estimate the potential public health impact of different actions and interventions to reduce smoke exposure, but it does not account for the fact that in reality a combination of these actions or interventions will be employed across the population (see Section 8.3.3). As depicted in Figure 6-4, a real-world estimation of the overall percent reduction in smoke exposure requires multiple pieces of data, including demographic data, housing characteristic data, and data on access or availability to various actions or interventions. Therefore, each of these pieces of data will vary depending on geographic location, demonstrating that a one-size-fits-all approach is not ideal, but can provide an estimation of the potential public health implications of reducing smoke exposure using different actions or interventions as presented within this assessment. In the future, as more data is collected on how people respond to wildfire smoke and how that differs from prescribed fire smoke, such as through the Smoke Sense application [https://www.epa.gov/air-research/smoke-sense-study-citizen-science-project-using-mobile-app; U.S. EPA (2020b)], it could be possible to more fully account for and quantify the actions taken by individuals affected by smoke through data analysis or exposure modeling, and subsequently assess the potential overall smoke exposure reduction for a population.

6.5 ECOLOGICAL EFFECTS ASSOCIATED WITH WILDFIRE SMOKE AND DEPOSITION OF ASH

Wildfire smoke and the deposition of ash can have wide-ranging effects on plants and animals. For example, pathogenic fungi have been shown to be aerosolized on smoke particulates and transported downwind from wildfires. Forest pests can be stimulated by smoke which serves as an attractant to pyrophilous beetle species that are adapted to reproduce in the downed lumber and freshly burned wood following a fire (Lesk et al., 2017; Hart, 1998; Evans, 1971). In addition to effects on lower trophic levels, smoke effects have also been documented to occur in vertebrates. After the fires of 1988 in Yellowstone National Park, for example, hundreds of large mammals, including elk, moose, mule deer, and bison were found dead: necropsy evidence suggested that smoke inhalation killed nearly all these animals (Singer and Schullery, 1989). Smoke inhalation has also been associated with mortality in raptors as a result of promoting pulmonary fungal infection following smoke exposure (Kinne et al., 2010). While numerous adverse effects from wildfire emissions have been documented, smoke can also have a stimulatory effect on the environment (McLauchlan et al., 2020). The following sections more fully characterize the ecological effects of wildfire smoke and ash deposition.

6.5.1 PARTICULATE MATTER (PM)

Although this section focuses on how smoke affects ecological receptors, it's important to recognize the potential climatological impact of wildfire smoke. Wildland fire is an increasing source of particulate matter emissions (see Chapter 7), specifically $PM_{2.5}$, which have been shown to have a variety of impacts on the environment (Bond and Keane, 2017). Particulate matter generated from wildfires has been shown to affect cloud cover and ice nucleation and interact with solar radiation through absorption and scattering. Specifically, the deposition of the $PM_{2.5}$ component black carbon has been shown to increase soil temperature through absorption of solar radiation (U.S. EPA, 2012). This is noteworthy in Pacific Northwest forests given the increasing rate and quantity of black carbon deposition in today's unprecedented fire regime because increased soil temperature is associated with concurrent decreases in tree growth which is a precursor to tree mortality. In contrast, certain highly reflective PM components in the atmosphere can scatter incoming solar radiation with much of that energy returning to space, resulting in an overall cooling effect on the climate (U.S. EPA, 2019b).

6.5.1.1 TRANSPORT OF BACTERIA AND FUNGI FROM SOIL AND PLANTS THROUGH SMOKE

A relatively recent advancement in fire ecology includes the nascent field of pyroaerobiology, which considers the living component of smoke particles generated from wildfires; specifically, microbes aerosolized and transported on particles by wildland fire (Hu et al., 2020; Kobziar and Thompson, 2020; Kobziar et al., 2018). Elevated concentrations of bacteria and fungi have been documented in smoke from burning of woody materials (Mirskaya and Agranovski, 2020) and coniferous forests (Kobziar et al., 2018) through the collection of microorganisms on passive samplers downwind of fires. The authors showed that microbial counts were significantly elevated above ambient conditions and decreased with distance from the fire's flaming front. It has been hypothesized that these microorganisms could represent an infectious risk to the public (Kobziar and Thompson, 2020) while also serving as an important inoculum for reseeding the soil flora following a fire event.

6.5.1.2 SMOKE-STIMULATED FLOWERING/SEED GERMINATION, SEED RELEASE, AND PLANT PRODUCTIVITY

One of the better-studied aspects of the effects of wildfire smoke on the environment is smoke-stimulated flowering, seed germination, and seed release. In their review of the ecological effects of fire, Bond and Keane (2017) noted that flowering is common among perennial grasses and herbs, some species of which only flower when cued by smoke (Chou et al., 2012). Wildfire smoke also stimulates germination in soil seedbanks for many species adapted to fire-prone forests and shrublands such as those in California (Keeley et al., 2005). In such environments, seedling recruitment from seed banks is one of the primary means of regeneration following a wildfire event. In fire-adapted environments, germination is the result of either heat shock or exposure to combustion products in smoke. However, germination can also be stimulated in some fire-adapted species through direct deposition on the seed or as a result of smoke binding to soil particles and subsequent aqueous or atmospheric transfer to seeds (Keeley et al., 2005). Although dozens of individual chemicals and particulate matter make up wildfire smoke, Keeley and Fotheringham (1997) showed that it is the nitrogen oxides (NO_X) present as trace gases in smoke that are responsible for seed germination.

In addition to stimulating flowering and seed release, a plant's light-use efficiency and productivity are enhanced by smoke from wildfires (Hemes et al., 2020; Strada et al., 2015). It has been shown that increased atmospheric particulate matter following a wildfire redistributes photons throughout multilayer vegetation canopies. This scattering enhances the distribution of light throughout the canopy architecture where incoming solar radiation may have otherwise been limited. This increased light availability in the understory is captured through photosynthesis and translated into increased plant growth.

6.5.2 EFFECTS OF OZONE (O_3) FROM FIRES

Wildfire smoke consists of numerous components (see Chapter 4 and Chapter 7), including volatile organic compounds (VOCs) and NO_X which can increase ozone production downwind following a wildfire event (Jaffe and Wigder, 2012). Because VOCs are ubiquitous in most geographic locations, it is generally thought that NO_X concentrations are the rate-limiting factor to ozone formation. The amount of NO_X produced during a wildfire is a function of the N content of the fuel, which varies by species, age and type of ecosystem, and the intensity of the burn. Higher temperature fires tend to produce more oxidized forms of N than lower intensity burns, and therefore are thought to produce more of the NO_X precursors available for ozone production (Jaffe and Wigder, 2012). Lower intensity burns tend to produce more oxygenated VOCs. The difference in ozone concentrations that can occur depending on fire type is reflected in the different hypothetical scenarios examined in the case studies presented within this assessment (see Chapter 7).

There is overwhelming evidence linking tropospheric ozone with reductions in growth and productivity in both agricultural and natural ecosystems (U.S. EPA, 2020a). Ecological effects of ozone can be observed across multiple scales of biological organization, from cellular to individual organism to the level of communities and ecosystems. Ozone can affect both aboveground and belowground processes leading to changes in productivity, carbon sequestration, biogeochemical cycling, and hydrology.

At the plant level, ozone enters the leaves through stomates, and quickly disassociates in the leaf apoplast into hydrogen peroxide (H_2O_2), organic radicals, and other reactive compounds that damage cellular membranes (Wohlgemuth et al., 2002; Hippeli and Elstner, 1996). Through both direct effects on stomatal regulation (Grulke, 1999) as well as chloroplast degradation, ozone can decrease photosynthesis and metabolism (Matyssek and Innes, 1999). Reductions in photosynthesis and overall carbon assimilation leads to decreased growth, but also can result in a shift in allocation of carbon resources within the plant, particularly to roots. These shifts in carbon allocation can lead to a change in the physiological functioning of the plant, including changes in gene regulation (U.S. EPA, 2020a; Andersen, 2003).

The direct effects of ozone on carbon assimilation and plant growth can subsequently alter the competitiveness of individuals in ecosystems. A reduction in carbon allocation to roots can alter rhizosphere interactions and symbiotic associations, both potentially leading to changes in nutrient uptake (U.S. EPA, 2020a). Changes in nutrient uptake therefore can lead to further reductions in growth, and to a change in the competitive stature of the plant. Because not all species are equally susceptible to ozone, there is often a shift in the competitive structure of ecosystems exposed to ozone, with sensitive species dropping out and ozone-tolerant species becoming more abundant. Through these direct and indirect effects, both ecosystem structure and function can be altered by ozone stress.

Ozone also influences the susceptibility of natural ecosystems to future wildfire. Ozone stress often results in early senescence of leaves, which can increase fuel load in conifer forests such as ponderosa pine that shed older leaf whorls in response to ozone stress (Miller et al., 1982). Ozone-sensitive species are also more susceptible to other stresses, such as insects and pathogens, increasing tree mortality and potentially increasing the fuel load in stressed ecosystems. Because ozone tends to reduce carbon allocation to roots, ozone-stressed plants also can become drought stressed, further increasing their susceptibility to other stresses and to wildfire. This can be a positive feedback loop in that as fire occurs, ozone is produced in smoke, potentially leading to susceptibility in future fire events. In addition to ozone, many studies have documented the release of hazardous organic and inorganic chemicals from combustion of biomass through wildfire, and there is a vast literature on the toxicity of organic chemicals and heavy metals on plants and animals. There are, however, no studies available reporting demonstrable ecological effects from hazardous pollutants released or generated from wildland fire.

Although wildfires are expected to increase ozone levels, the short duration and timing of wildfire events may limit the impact of the additional fire-generated ozone. In the western U.S., most wildfires tend to occur late in the growing season when water is limited and carbon assimilation is lower, thereby reducing uptake and effects. In addition, ozone effects tend to be cumulative, while ozone generated through wildfire events is episodic and short-lived; therefore, the impacts of fire-generated ozone may be limited. Nonetheless, plants are more sensitive to higher ozone concentrations and so any additional exposure is likely to increase the overall impact, particularly in areas where background levels are high.

Overall, more research is needed to evaluate the ecosystem impacts of additional ozone generated through wildfire.

6.5.3 ATMOSPHERIC DEPOSITION OF ASH

The most immediate effect of wildland fire on the land surface is the removal of vegetation and the subsequent deposition of a layer of charcoal or ash (De Sales et al., 2019). Ash is the particulate residue that consists of mineral and charred organic materials formed when carbon fuels are burned (Bodi et al., 2014). Characteristics of ash are affected by the type of fuel burned and intensity of combustion, with low-intensity fires yielding ash of greater organic content and hotter fires resulting in more mineralized material. In forested environments, the mass of ash deposition following a fire can range from 2−9% of woody biomass (Raison, 1979).

Ash that is deposited on the ground is incorporated into soil where vegetation has burned. Given its high mobility; however, ash is also readily transported downwind and downstream where it can influence habitats far removed from areas burned by wildfire. Ash deposition is becoming an increasingly common input into ecosystems, and it can have a dramatic effect on the biogeochemical cycling of nutrients and minerals in forested soils. This section considers the ecological effects of ash on soil chemistry and structure, nutrient flux, microbial activity, and plant growth.

6.5.3.1 SOIL CHEMISTRY AND STRUCTURE

Ash deposition following wildfire can profoundly change soil characteristics. In a study of ion release from burning plant material, Grier and Cole (1971) demonstrated greatly increased concentrations of ions entering the soil, which were adsorbed in the uppermost soil horizons, causing major chemical changes such as the influx of basic ions increasing soil pH. In a study of wildfire sites in California, Ulery et al. (1993) showed that ash deposition raised soil pH by as much as 3 pH units (to pH 10.5) compared with unburned soil. More basic pH’s increase the solubility of soil organic carbon (Andersson et al., 1994) and increases the number of binding sites in soil that can hold cationic micronutrients (Raison, 1979).

The physical deposition of ash can act to increase soil water repellency, preventing the infiltration of meteoric water (Doerr et al., 2000) and decreasing the potential for nutrient leaching. Another consequence of ash deposition on soil structure is an increase in bulk density, which is the soil mass divided by the bulk volume of the sample (g/cm^3). The bulk density of soil increases with ash deposition because soil aggregates collapse and the ash clogs pore spaces, both of which serve to decrease soil porosity and permeability (Verma and Jayakumar, 2012). Factors like increased soil hydrophobicity and soil density that limit the infiltration of meteoric water would help to retain otherwise leached soil nutrients.

6.5.3.2 STIMULATION OF MICROBIOLOGICAL ACTIVITY AND PLANT GROWTH

Although it is widely accepted that fire stimulates microbial activity (Bodi et al., 2014), most research on wildfire's effect considers only soil heating where, in extremely hot fires, sterilization of the upper soil layers can occur (Mataix-Solera et al., 2009). Far fewer studies address the effects of ash deposition on soil microbiota and nutrient processing. Compared to the growth of fungi organisms which occurs at lower soil pH, Jokinen et al. (2006) suggested that the increased soil pH and nutrient and carbon availability from ash deposition stimulated bacterial respiration. Bacteria proliferate more quickly than fungi, and their ability to capitalize on a new carbon pool, such as ash-mobilized organic carbon, would favor bacterial growth, suggesting an inhibitory effect of ash deposition on fungal microflora. Mycorrhizal fungi, however, have a symbiotic relationship with plants that depends on the latter's ability to produce carbohydrates through photosynthesis and share sugars with the fungus. In this relationship, plants receive water and nutrients from the soil by the extensive network of fungal mycelial hyphae. The plant's provision of carbohydrates to mycorrhizae makes this group of fungi competitive with bacteria in an otherwise challenging post-fire environment for fungal organisms.

New tree growth in burned forests is highly dependent upon mycorrhizal symbiosis, and the fungal colonization of burned areas is relatively well documented. In a study of burned pine forests in northern California, Grogan et al. (2000) found that wildfire disturbance resulted in marked changes in mycorrhizal community composition and a significant increase in the relative biomass of mycorrhizal-ascomycetous fungi. Additionally, in an experiment to examine the effects of ectomycorrhizal colonization and fire on the growth of Bishop pine seedlings (*Pinus muricata*) in northern California, Peay et al. (2010) showed that the percent nitrogen in needles was greatest in treatments with an ectomycorrhizal inoculum regardless of whether ash was added to soil. These results underly the critical relationship of pine forests and their dependence on mycorrhizal associations.

Immediately downwind of fires, larger particles of ash are deposited onto vegetation with a concomitant observable soiling of leaves, which can adversely affect photosynthesis and plant growth. Nutrients may be released from combustible fuels after fire and transported as ash by atmospheric deposition to stimulate vegetation growth (Bodi et al., 2014). The amounts of calcium, nitrogen, phosphorous, potassium, magnesium, and sulfur released by burning forest vegetation are elevated in relation to both the total and available quantities of these elements in soils (Raison and McGarity, 1980; Raison, 1979). The addition of nutrients in ash tend to stimulate plant growth although germination may be inhibited by deposited ash, due perhaps to ash's hydrophobicity and osmotic pressure excluding water from the seed, the presence of toxic elements in ash, and/or elevated pH (Bodi et al., 2014).

Nitrogen is among the most important nutrients that can stimulate plant growth. Forested systems rely on cycling the nitrogen locked in dead plant matter into more bioavailable forms. Compared with the biological decay of plant remains, burning rapidly releases nutrients into a plant-available form. Nitrogen from wildfires can represent over 30% of nitrogen deposition in forested systems of the Pacific Northwest

(Koplitz et al., In Press), and growth of the predominant forest tree species (Douglas fir, *Pseudotsuga menziesii*) in the Pacific Northwest is stimulated by nitrogen deposition. However, too much nitrogen can be problematic and lead to nitrogen inputs exceeding the critical nitrogen load in Northwest forests and ultimately decreased tree survival.

6.5.3.3 ASH DEPOSITION AND WATER QUALITY

The aerial transport and deposition of materials in smoke and ash may also affect downwind water quality. Increased runoff of ash, sediments, and chemical constituents following fire appear to be the dominant mechanism by which water quality is affected (see Section 5.2.3.2.5) for further discussion of fire effects on water quality, including potential effects on drinking water). Nevertheless, it is logical to assume that some material could be deposited in wet or dry form onto the surfaces of downwind water bodies, such as streams, lakes, or reservoirs or deposited on unburned terrestrial surfaces and subsequently moved via overland or subsurface flow to water bodies. Post-fire increases in nutrient deposition (Ranalli, 2004; Koplitz et al., In Press) and wind dispersion of ash, nutrients, and sediments (Roehner et al., 2020; Bodi et al., 2014) are suggestive of such a mechanism.

Although studies measuring this phenomenon are limited, several have reported water quality changes potentially linked to aerial transport of materials from fires (Earl and Blinn, 2003; Lathrop, 1994; Spencer and Hauer, 1991). For instance, Earl and Blinn (2003) found higher nutrient concentrations in an unburned watershed in southwestern New Mexico, associated in time with a nearby fire. The authors suggested that aerial transport of nutrients from the fire was likely responsible. Initiating sampling within hours of a fire, Spencer and Hauer (1991) observed spikes in nitrogen and phosphorus in stream water, before returning to background levels within several days to weeks. The authors concluded nitrogen from the smoke diffused into the surface water, while phosphorus leached from ash deposited directly into the water bodies. Nitrogen volatilizes at lower temperatures than phosphorus, likely explaining the differences in method of transport of these two nutrients.

6.5.4 UNCERTAINTIES AND LIMITATIONS IN THE ECOLOGICAL EFFECTS EVIDENCE

There are considerable uncertainties and limitations in understanding the ecological effects of emissions and ash on plants and animals. Ultimately ecosystems have adapted to fire regimes, but an understanding of fire’s immediate ecological effects are limited by a dearth of studies on the indirect ecological effects of fire. The influx of fire-liberated nutrients on terrestrial and aquatic receptors is just beginning to be investigated and the time frame over which fires influence air and water chemistry is an area that warrants further investigation.

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PART II: QUANTITATIVE ASSESSMENT OF SMOKE IMPACTS OF WILDLAND FIRE IN CASE STUDY AREAS

CHAPTER 7 AIR QUALITY MODELING OF CASE STUDY FIRES

7.1 INTRODUCTION

Wildland fires (i.e., prescribed fire and wildfire) directly emit fine particulate matter (particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm [$PM_{2.5}$]), and gaseous pollutants emitted from fires can also form secondary $PM_{2.5}$ and ozone (O_3) in the atmosphere (Prichard et al., 2019; Urbanski, 2014; Hu et al., 2008). Estimating emissions and concentrations of pollutants formed from wildfires is challenging because of variability in fuel consumed, fuel types, fuel moisture, plume dynamics, and complex nonlinear chemistry (Prichard et al., 2019; Jiang et al., 2012). A realistic characterization of O_3 and other secondary pollutant formation in a wildfire plume also depends on an understanding of the plume's surrounding chemical and physical environment—factors that evolve as the plume moves further downwind from the fire.

Prescribed fire is a relatively efficient, cost-effective tool implemented by land managers for a range of uses, including ecosystem maintenance (Kobziar et al., 2015) and wildfire mitigation (Prichard et al., 2010). Although the use of prescribed fire as a land management tool is common in some parts of the contiguous U.S., both the specific land management goals and the response of the landscape to prescribed fire can vary significantly (Ryan et al., 2013). For these reasons, it has been historically difficult to synthesize both the environmental trade-offs between wildfire and prescribed fire, as well as the behavioral influence of prescribed fire on wildfire activity (e.g., changes in intensity, risk of ignition, fire size, etc.).

This chapter presents a novel analysis evaluating air quality trade-offs across multiple fire management strategies for two wildfires: Timber Crater 6 (TC6) Fire in 2018 and Rough Fire in 2015. Chapter 3 contains general details and maps describing these fires. In both cases, detailed alternative burn scenarios were developed with fuel information from multiple sources. Actual and alternative burn scenarios were then simulated with air quality modeling to estimate surface concentrations of $PM_{2.5}$ and O_3. Comparing the air quality impacts across different burn scenarios for each fire case study offers insights into relative air quality impacts from hypothetical land management approaches, although downwind transport and the resulting air quality impacts near population centers can be strongly influenced by locally specific features like terrain and meteorology.

7.1.1 EMISSIONS OF WILDLAND FIRES

The relative amounts and chemical composition of emissions depend on the fuel characteristics, combustion conditions, and meteorological conditions (Urbanski, 2014). Additionally, these factors are interrelated; for example, the combustion intensity is dependent on the meteorological conditions (temperature, relative humidity, and wind conditions) and the fuel characteristics [structure, moisture, and loading; Surawski et al. (2015)]. Meteorology can also modulate combustion conditions, with strong winds increasing the rate and extent of spread and peak heat release rate (i.e., intensity). Emissions from fires are typically calculated using Equation 7-1.

$$\text{Emissions of a given pollutant} = (\text{area burned}) \times (\text{fuel loading}) \times (\text{emission factor})$$

Equation 7-1

In Equation 7-1, fuel loading is the mass of fuel consumed per area and the emission factor is the mass of a particular pollutant per mass of total fuel consumed.

The modified combustion efficiency, defined as MCE = excess carbon dioxide (CO_2)/[excess carbon monoxide (CO) + excess CO_2], is widely used as an indicator of combustion conditions. MCEs greater than 0.9 are generally considered flaming dominated and lower MCEs are smoldering dominated. Grasses and other fine fuels (<1/4″ diameter and large surface-to-volume ratios) tend to burn in the flaming phase. Coarse wood, duff, and organic soils tend to burn in the smoldering phase. Fuel loading, density, and geometry also affect the combustion phase (e.g., densely packed fine fuels will smolder). Many wildland fires burn in landscapes with a variety of fuel types, structures, and moisture content and will, therefore, exhibit both flaming and smoldering conditions simultaneously.

The fuel moisture content is a critical factor that affects combustion conditions. Energy is lost in evaporating the water in the fuel rather than volatilizing fuel components needed to sustain flaming combustion. Fuels with higher moisture content take longer to ignite, may smolder before transitioning to flaming, have shorter flaming and longer smoldering durations, lower peak heat release rate, as well as lower and more variable fuel consumption (Possell and Bell, 2013; Chen et al., 2010). Additionally, the moisture content affects the composition of the emissions. CO, volatile organic compounds (VOCs), ammonia (NH_3), and particulate matter (PM) emission factors increase, while CO_2, nitrogen oxides (NO_X), and elemental carbon (EC) generally decrease with increasing moisture (May et al., 2019; Tihay-Felicelli et al., 2017; Chen et al., 2010). PM emissions are especially sensitive to fuel moisture. PM emission factors can be larger than CO emission factors for some fine fuels (e.g., litter, pine needles, etc.) at high fuel moistures [e.g., above 60% dry basis; Chen et al. (2010)].

Most emission factor compilations group emission factors by ecoregions to aggregate the impacts of fuel chemistry, structure, and to some extent moisture (Prichard et al., 2020; Andreae, 2019; Akagi et al., 2011). The emissions model will then predict how much fuel is consumed (or emitted) during the flaming or smoldering phases. The Smoke Emissions Reference Application (SERA) described in Prichard et al. (2020) is the most extensive compilation of smoke emission factors for North American fires to date. However, knowledge gaps persist for emissions factors for wildland fires as summarized by [Prichard et al. (2020); derived from Figure 1, Table 3], resulting in limited information with respect to:

- Wildfire emission factors: Emission factors are predominantly from laboratory studies (72% of the observations); field data are 85% from prescribed fires and 15% from wildfires.
- Smoldering emission factors: Smoldering emission factors account for 31% of the prescribed fire observations and 50% of wildfire observations, but wildfires have no emission factors for long-term (residual) smoldering conditions.
- Fuels that tend to smolder: Most emission factor observations are from western conifer forests, eastern conifer forests, and shrublands, but there are few observations for duff, coarse woody debris, and peat from these regions.
- PM and VOC speciation: PM composition data is largely limited to black carbon and limited VOC data exists, particularly for field data; most emission factor observations are the major pollutants of CO, CO_2, methane (CH_4), and $PM_{2.5}$, but a range of compounds have over 100 observations (propene, acetylene, methanol, formaldehyde, NH_3, nitric oxide (NO), nitrogen dioxide (NO_2), NO_X, hydrogen cyanide [HCN], sulfur dioxide [SO_2]).

In comparing emissions from wildfires and prescribed fires, the different meteorological conditions, potentially different fuels, and combustion conditions mean that emission factors will be different for each type of fire, even in the same region. For example, Urbanski (2014) compared MCEs for wildfires and prescribed fires in northwestern conifer forests and found an average MCE of 0.883 ± 0.010 for wildfires and 0.935 ± 0.017 for prescribed fires. However, the type of fuel that is consumed may be an important factor because prescribed fires and wildfires in the northern Rocky Mountains both had lower MCEs (~0.87) that were due to a larger fraction of coarse woody debris (Urbanski, 2013). It is also possible that the ignition approach used for prescribed burns may affect emissions, although that relationship has not been well characterized. Although meteorological, fuel, and combustion parameters are factored into emissions estimation, there is still a need to understand whether emissions modeling systems accurately capture the differences between fire types.

7.1.2 USING AIR QUALITY MODELS TO ESTIMATE WILDLAND FIRE $PM_{2.5}$ AND OZONE IMPACTS

Quantifying the contribution of wildland fire to ambient O_3 and $PM_{2.5}$ is important for air quality alerts, air quality mitigation programs, and multiple regulatory programs including National Ambient Air Quality Standards (NAAQS) and Regional Haze. Therefore, it is important to understand how wildland fires affect air quality and regional haze so that anticipated changes in land management (i.e., more prescribed fires) could potentially minimize air quality degradation while still meeting ecological goals as well as potentially reducing the impact of wildfire.

Photochemical grid models can provide information about how air quality would change based on changes in emissions due to different types of land management choices (Hu et al., 2008). The Community Multiscale Air Quality [CMAQ; www.epa.gov/cmaq; U.S. EPA (2020a)] model includes emissions, chemical reactions, and physical processes such as deposition and transport. The CMAQ model has been used to estimate the air quality impact of wildland fires as a collective source group (Kelly et al., 2019) and for specific fires (Baker et al., 2018; Zhou et al., 2018; Baker et al., 2016).

Photochemical grid models provide continuous spatial and temporal estimates of smoke impacts from wildfires. These models are particularly useful in areas not covered by ambient measurements (O'Dell et al., 2019). However, fire behavior and associated smoke characteristics can vary substantially by region (Brey et al., 2018). Accurately representing wildfire smoke in models for different geographic areas is an ongoing effort that will continue to be important as landscapes evolve due to climate change and human development (Ford et al., 2018; Liu et al., 2016; Yue et al., 2013). Detailed case studies, like the two in this report, provide some constraints on the representation of wildfire smoke in models for specific areas, but more work is required to improve these estimates at both regional and global scales (Liu et al., 2020; Garcia-Menendez et al., 2014, 2013).

Previous applications of CMAQ for specific fire plumes show a reasonable representation of local- to continental-scale transport (Kelly et al., 2019; Baker et al., 2016). The modeling system treatment of plume rise and transport works best when there are accurate activity data including fire size and timing (Baker et al., 2018; Zhou et al., 2018). Performance related to $PM_{2.5}$ impacts from wildland fire are mixed and do not seem systematically biased high or low (Baker et al., 2018; Koplitz et al., 2018; Wilkins et al., 2018; Baker et al., 2016). This modeling system tends to overestimate O_3 impacts from wildland fire at the surface (Baker et al., 2018; Baker et al., 2016). Predicting wildland fire impacts on O_3 is challenging because formation can be highly variable in time and space. Fresh nitric oxide emissions at the fire tend to inhibit O_3 formation as chemical destruction reactions outpace production. As the plume moves further downwind, O_3 may be formed at the edges of the plume where sunlight and precursors are abundant. Atmospheric transport processes are also important because O_3 may be formed in smoke plumes but not necessarily mix to the surface.

7.1.3 CASE STUDY: TIMBER CRATER 6 (TC6) FIRE

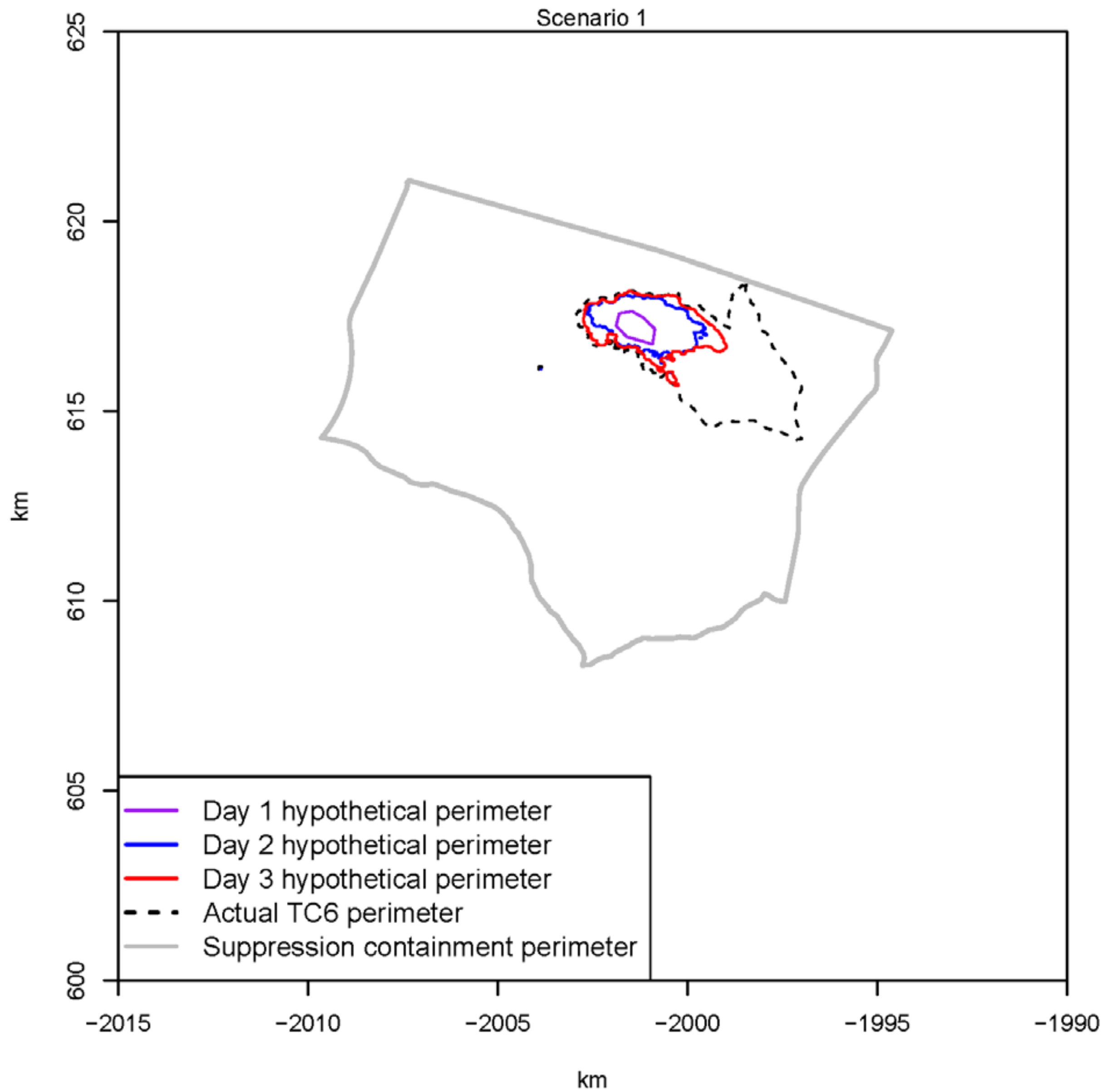
The TC6 Fire burned approximately 3,000 acres in Crater Lake National Park from July 21 to July 26, 2018. The fire covered lands managed by multiple federal agencies. This fire was chosen as a case study for this report because land managers in the area determined that reduced fuel loading from previously managed land slowed fire progression enough to allow for successful suppression (e.g., burning out fire lines). As a result, the TC6 Fire had a smaller total area burned than might have occurred without those suppression efforts.

Three hypothetical scenarios, as detailed in Chapter 1 and reiterated here, were developed to examine the air quality impacts of different fire management strategies compared with the actual TC6 Fire:

- Scenario 1 (small): Defined as the green hatched area inside the TC6 Fire perimeter in Figure 1-1, which is a smaller hypothetical TC6 Fire in a heavily managed area (e.g., most prescribed fire activity). This scenario would equate to a wildfire with less fuel consumption, a smaller fire perimeter, and less daily emissions.
- Scenario 2a (large): Defined as the blue dotted line and hatched area outside the TC6 Fire perimeter in Figure 1-1, which is a larger hypothetical TC6 Fire, but not the "worst-case" scenario with no land management. This scenario would equate to a wildfire with more fuel consumption, a larger fire perimeter, and more daily emissions.
- Scenario 2b (largest): Defined as the brown dotted line and hatched area outside the Scenario 2a fire perimeter in Figure 1-1, which is a much larger, hypothetical "worst-case" modeled scenario TC6 Fire with no land management (i.e., no prescribed fire). This scenario would equate to a wildfire with the most fuel consumption, largest fire perimeter, and largest daily emissions.

As noted above, Scenario 1 assumed a smaller and shorter duration fire than the actual fire, due to less fuel from more intensive land management (Figure 7-1). Scenarios 2a and 2b assumed more fuel in the area due to a lack of past land management. Both Scenarios 2a and 2b are larger than the actual fire and longer in duration. Scenario 2b is the largest fire and extends outward to a contingency perimeter where fire suppression would be aided by roadways and other existing fire breaks. All of these scenarios used fuel data based on a consistent approach that is described in the following sections.

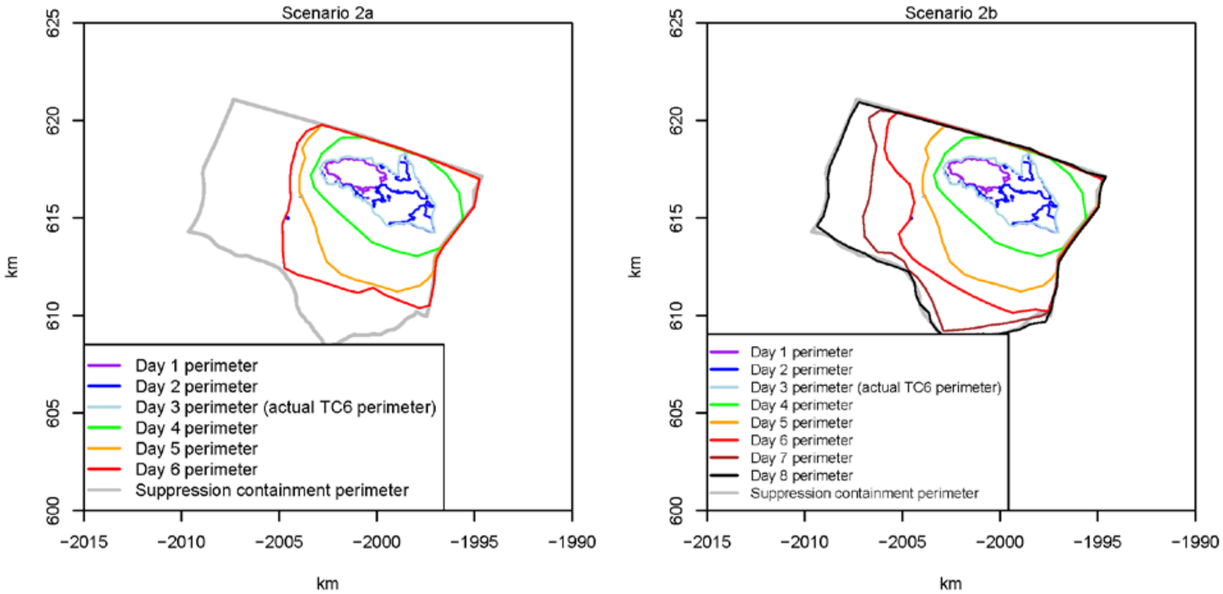
Each of the hypothetical scenarios were based on expert judgment of land managers familiar with Crater Lake National Park, the fuels in the area, meteorology during the TC6 Fire, existing fire breaks (e.g., roadways), and additional suppression techniques that would have been employed if the fire had spread faster than the actual fire. Actual fire perimeters from the TC6 Fire were used for the first 3 days of the larger hypothetical scenarios with Day 3 being the actual final perimeter of TC6. These hypothetical scenarios were not based on fire behavior or fire spread models. Two hypothetical scenarios (2a and 2b) were developed to represent larger fires than the actual fire (Figure 7-2). Both larger hypothetical scenarios (2a and 2b) cover a larger area and extend for more days than the actual fire.



km = kilometer.

Note: The fire perimeter of the Timber Crater 6 Fire is also shown as the dashed line. The solid gray outline shows the fire suppression contingency perimeter which is considered the maximum extent of wildfires in this area. The total area assumed to be burned with Scenario 1 is delineated by the Day 3 perimeter.

Figure 7-1 Daily fire perimeters for the smaller hypothetical Timber Crater 6 (TC6) Fire (Scenario 1).



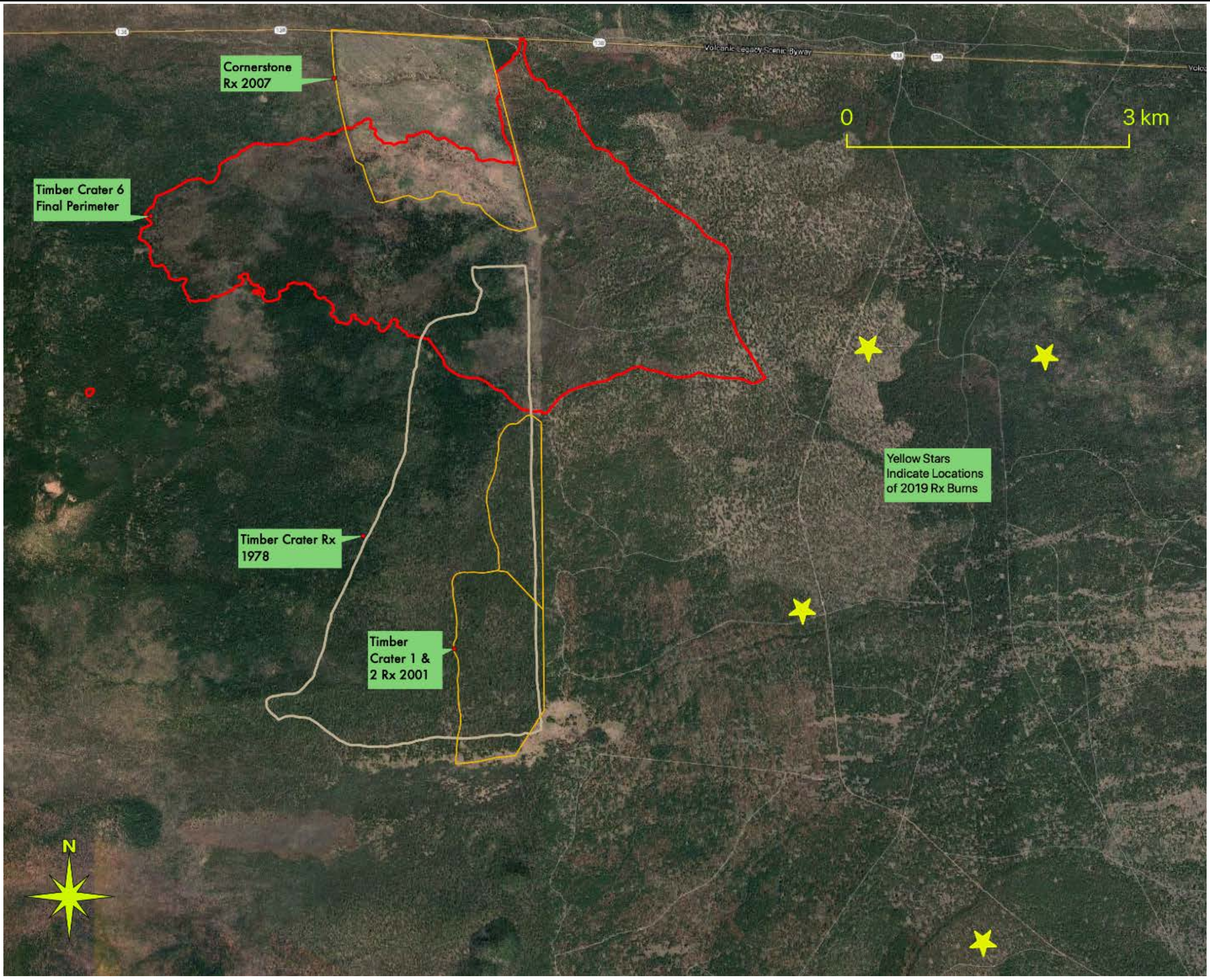
km = kilometer.

Note: The solid gray outline shows the fire suppression contingency perimeter which is considered the maximum extent of wildfires in this area.

Figure 7-2 Daily fire perimeters for the larger hypothetical Timber Crater 6 (TC6) Fires (Scenarios 2a and 2b).

7.1.3.1 PRESCRIBED FIRE NEAR CRATER LAKE NATIONAL PARK

Land management practices in and near Crater Lake National Park include prescribed fire and mechanical thinning. Some of the leftover fuel from mechanical thinning is sold as timber and some is burned in slash piles during the winter. Multiple prescribed burns have been conducted in the area (Figure 7-3), some of which intersect the TC6 Fire perimeter: Cornerstone in 2007 (no specific dates known), Timber Crater 1 and 2 in 2001 (no specific dates known), and Timber Crater 1978 (no specific dates known). More recent prescribed fires (not named) were conducted in this area in September 2019 (13−15 and 26−28). Because the days of the September 2019 prescribed fires were presumed to match criteria for prescribed fire in the region, this time period was used for modeling both actual prescribed fires during that period and provided a basis for modeling other prescribed burn units from previous years. Each prescribed fire (e.g., actual 2019 prescribed fires, Cornerstone, Timber Crater 1 and 2, and Timber Crater 1978) were modeled for these 2019 dates but in separate model simulations so they would not interact with each other.



Rx = prescribed burn.

Figure 7-3 Fire perimeter of the actual Timber Crater 6 (TC6) Fire and multiple prescribed fires.

7.1.4 CASE STUDY: ROUGH FIRE

The Rough Fire burned in parts of the Sierra National Forest, Sequoia National Forest, and Kings Canyon National Park between July 31 and October 1, 2015 [https://www.nps.gov/seki/learn/nature/rough-fire-interactive-map.htm; NPS (2016)]. This wildfire covered approximately 150,000 acres of land managed by multiple federal agencies. The Rough Fire was chosen as a complement to the TC6 Fire because of its much larger size and duration.

Land managers were able to suppress the Rough Fire in several areas where land had been previously managed. One such area was the Sheep Complex Fire in 2010 (~9,000 acres), which resulted in less available fuel and provided a break to stop fire progression. The Sheep Complex Fire in 2010 was a multimonth wildfire that burned at lower intensity and had slow progression related to moist fuels from heavy rains in the area earlier that year.

A prescribed fire (Boulder Creek Unit 1) was originally planned for in 2013 in an area adjacent to the footprint of the 2010 Sheep Complex Fire. Boulder Creek Unit 1 included a 3,200-acre area that was intended to restore fire and reduce fuels and fire behavior for this steep and inaccessible area of the Boulder Creek drainage, through which the Rough Fire subsequently burned. Because this prescribed fire unit was not burned in 2013 as planned, the Boulder Creek Prescribed Fire burn unit was hypothetically burned from September 30 to October 3, 2014 as these days matched meteorological conditions appropriate for a prescribed burn. The Boulder Creek Prescribed Fire plan is available in the appendix to this report and provides more specific details about fuels and approach.

The Sheep Complex Fire and Boulder Creek Prescribed Fire were instrumental in developing two hypothetical scenarios, as detailed in Chapter 1, to examine the air quality impacts of different fire management strategies compared with the actual Rough Fire:

- Scenario 1 (small): Defined as the red shaded and outlined area above the black dotted line in Figure 1-2, which examines the combined impact of the Boulder Creek Prescribed Fire and the Sheep Complex Fire on reducing the spread and air quality impacts of the Rough Fire; and
- Scenario 2 (large): Defined as the entire red perimeter of the Rough Fire and the blue area of the Sheep Complex Fire in Figure 1-2, which allows for the fire perimeter of the Rough Fire to progress into the area of the Sheep Complex Fire as though both the Boulder Creek Prescribed Fire and Sheep Complex Fire did not occur.

As noted above, one hypothetical scenario for the Rough Fire (Scenario 1) consists of a smaller Rough Fire under the assumption that a planned prescribed fire (Boulder Creek Unit 1 Prescribed Fire unit), which did not occur, had occurred prior to the Rough Fire. This planned prescribed fire represents the minimum amount of prescribed fire activity needed to create the suppression anchor that underpins the smaller hypothetical scenario (Scenario 1) because the initial prescription plan for the area called for approximately 5 more years of prescribed fire activity in the area (USFS, 2014). This smaller fire hypothetical scenario assumes that when the Rough Fire got to the area of the Boulder Creek Unit 1 Prescribed Fire, progression downslope toward the Central Valley of California would have stopped. Fire perimeters are shown for the Rough Fire, Sheep Complex Fire, and Boulder Creek Unit 1 Prescribed Fire area in Figure 1-2. Figure 7-4 shows the relationship between these fires and nearby large population centers in the Central Valley of California.

Another hypothetical scenario for the Rough Fire (Scenario 2) was a larger fire that progressed through the area of the Sheep Complex Fire with an assumption that fuels were dry and fuel loading would be similar to the surrounding area as if the Sheep Complex Fire had not happened. The hypothetical larger Rough Fire includes the actual Rough Fire in addition to the area of the Sheep

Complex Fire. The hypothetical wildfire version of the Sheep Complex Fire was based on the original spatial extent of the Sheep Complex Fire. The Sheep Complex Fire activity data was aggregated to the total event/fuelbed/location, then a daily fraction of total acres from the Rough Fire (from September 1 to 10, 2015) to the Sheep Complex Fire aggregated activity data was applied to each of these combined factors. As a result, the Sheep Complex Fire kept the same total area and fuel beds but was temporalized like the Rough Fire activity between September 1 and 10, 2015. This allowed the Sheep Complex area to be burned as part of the Rough Fire at the point the actual Rough Fire progressed to this area and beyond.

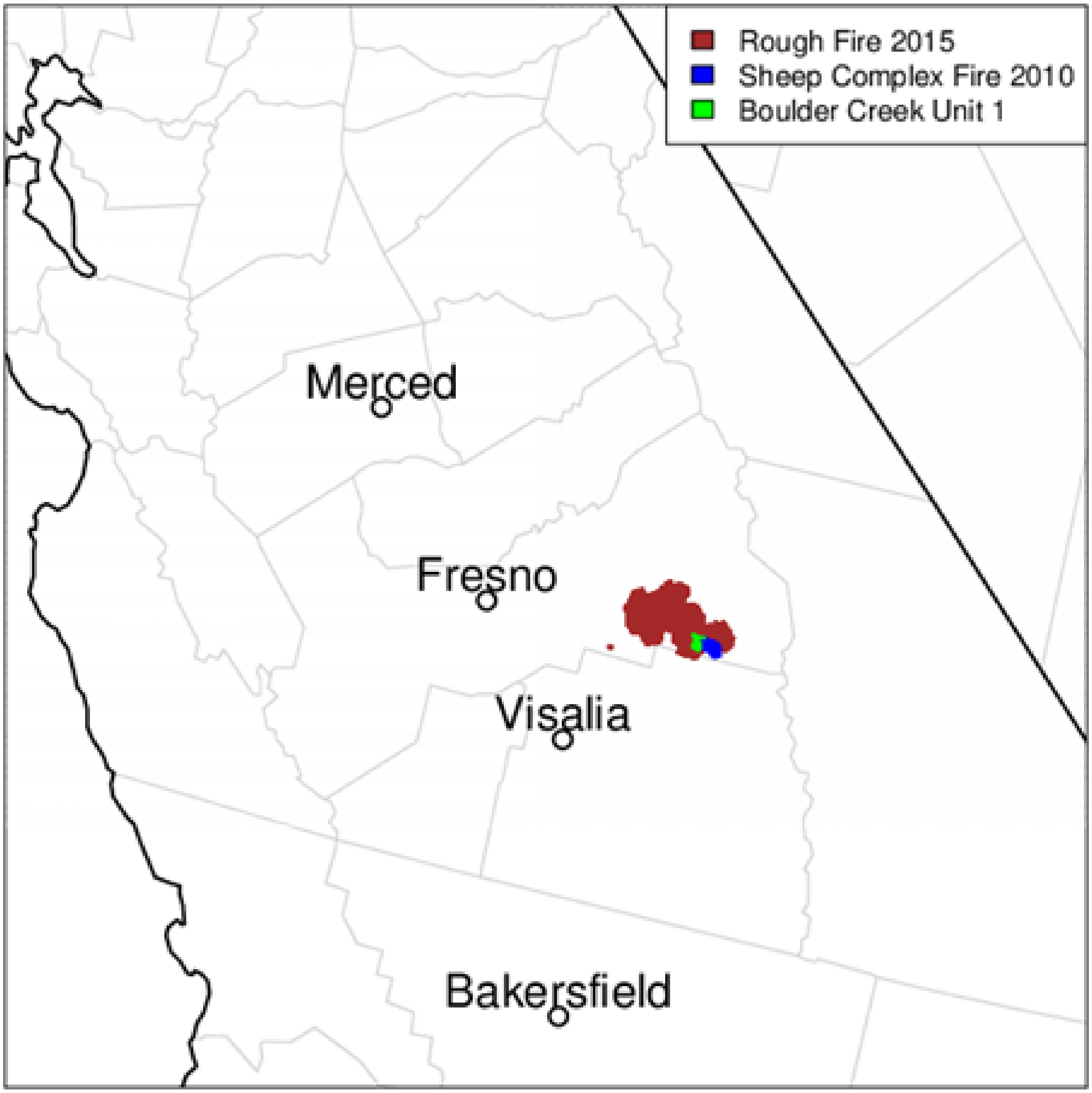
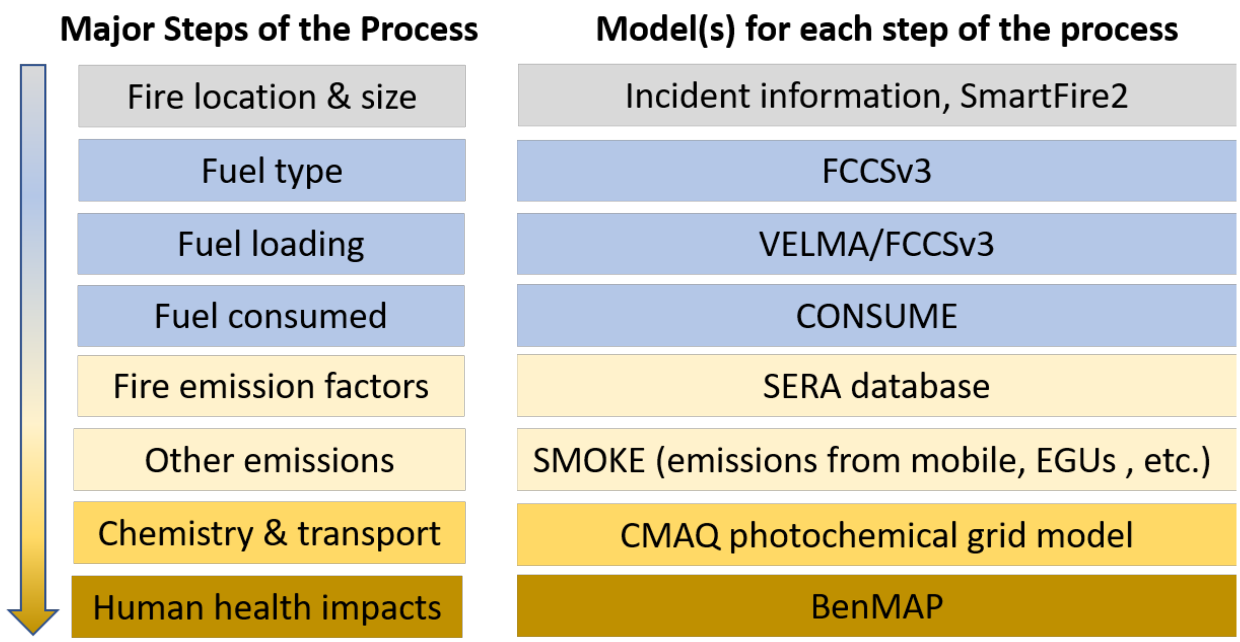


Figure 7-4 **Schematic showing the 2015 Rough Fire, 2010 Sheep Complex Fire, and Boulder Creek Unit 1 Prescribed Fire burn unit in relation to large urban areas in central California.**

7.2 METHODOLOGY

The air quality surfaces for $PM_{2.5}$ and ozone for the TC6 Fire and Rough Fire, each hypothetical scenario, and the prescribed fires, were produced using the modeling framework detailed in Figure 7-5. The figure shows the connectivity and relationships between various tools and models used to develop case study fire emissions. Fire location and timing was based on incident information where available and supplemented with data generated by the Satellite Mapping Automated Reanalysis Tool for Fire Incident Reconciliation Version 2 (SmartFire2; [SF2]) tool (Raffuse et al., 2009). SF2 reconciles data from satellite sensors and ground-based reports to use the strengths of both types of data while avoiding double counting of fires (Larkin et al., 2020; Larkin et al., 2009).



BenMAP = Environmental Benefits Mapping and Analysis Program; CMAQ = Community Multiscale Air Quality; EGU = electricity-generating unit; FCCS = Fuel Characteristic Classification System; SERA = Smoke Emissions Reference Application; SMOKE = Sparse Matrix Operator Kernel Emissions; VELMA = Visualizing Ecosystem Land Management Assessments.

Figure 7-5 Modeling framework used to characterize wildland fire emissions and air quality impacts for case study analyses.

The BlueSky Pipeline (https://github.com/pnwairfire/bluesky) is a version of the BlueSky Framework rearchitected as a pipeable collection of Python-based, stand-alone modules that can be linked or piped together in a series so that the output of one module becomes the input of the next. The BlueSky Pipeline estimates fuel type, fuel loading, fuel consumption, and emissions for each fire. Fuel type is based on the Fuel Characteristic Classification System (FCCS). Fuel loading is based on a combination of FCCS and Visualizing Ecosystem Land Management Assessments (VELMA) model output. Fuel consumption is based on the CONSUME module in the BlueSky Pipeline. BlueSky Pipeline provides daily total emissions of CO, NO_X, SO_2, NH_3, VOCs, and primary $PM_{2.5}$ for each wildfire and prescribed fire. Case study fire emission factors are based on the SERA database (Prichard et al., 2020).

Daily emissions were processed for input to the CMAQ photochemical model using the Sparse Matrix Operator Kernel Emissions [SMOKE, https://www.cmascenter.org/smoke/; CMAS (2020)] emissions model, which also provided emissions of other wildland fires, biogenic, and anthropogenic emissions. The CMAQ model uses emissions generated by the SMOKE model and meteorological data generated by the Weather Research and Forecasting (WRF) model to transport and deposit emissions injected into the model and estimate chemical transformation. The output from the photochemical model was processed for input to U.S. EPA's Environmental Benefits Mapping and Analysis Program—Community Edition [BenMAP-CE; U.S. EPA (2019a)] to estimate the human health impacts related to specific fire scenarios for each case study fire (see Chapter 8). More details about fuels, emissions, and photochemical modeling follow in subsequent sections of this chapter.

7.2.1 FUELS (FUEL CHARACTERISTIC CLASSIFICATION SYSTEM [FCCS])

Fuels for the case study wildfire emissions was based on a combination of FCCS and VELMA, which is discussed in the following section in more detail. This section describes the development of the FCCS component of the fuels for developing case study wildfire emissions.

The FCCS contains a reference library of wildland fuelbeds that can be used for wildland fire planning and smoke management decisions (Ottmar et al., 2007). The FCCS calculator within the Fuel and Fire Tools [https://www.fs.usda.gov/pnw/tools/fuel-and-fire-tools-fft; FERA (2020)] is used to produce a fuel loadings input file for CONSUME v5.0, a fuel consumption module within the BlueSky Pipeline (Prichard et al., 2021).

Although the LANDFIRE system (LF, 2008), contains an FCCS fuelbed layer, it does not include recent small wildfires and prescribed fires. To support emissions trade-offs analyses, we created four separate 30-m FCCS fuelbed raster layers to represent each of the scenarios evaluated in the TC6 case study.

To represent prewildfire fuelbed layers for each of the four scenarios, we assigned base FCCS fuelbeds (Appendix Table A.7-1) based on the 2014 LANDFIRE Existing Vegetation Type (EVT) layer (LF, 2014). We then used an existing Python script developed to update the base fuelbeds to represent canopy and surface fuel changes associated with recent wildfires and prescribed burns within the study area, including the 2010 Phoenix and 2014 Founders Day fires (Appendix Table A.7-2). For the hypothetical TC6 smaller fire (Scenario 1), fuelbeds were assigned to represent a recent prescribed fire over the entire scenario area so that fuel loading would be more like an area post-prescribed fire rather than multiyear fuel buildup. Fuel loading was not similarly modified for the Rough Fire scenarios. A Python script was used to update fuelbeds to recent low-severity prescribed burns immediately post-disturbance (111), recent high-severity wildfires within 0−5 years (132), and older high-severity wildfires within 5−10 years (133).

7.2.2 CHARACTERIZING SURFACE FUEL LOADS FOR USE IN THE BLUESKY PIPELINE

Surface fuel load characterization is an important component of modeling air quality impacts associated with wildfires and prescribed fires. The most commonly used tool for estimating surface fuel loads in the U.S. is the FCCS (Ottmar et al., 2007), which characterizes available fuel loading for various vegetation classification categories across a landscape and includes both vegetation type (e.g., Ponderosa Pine, Red Alder) and fuel load category (e.g., canopy, shrubs, nonwoody).

While FCCS captures the general diversity of available fuels found throughout the U.S., the fuel loadings are assumed to be homogenous within each vegetation type. Studies suggest that FCCS and other vegetation classification-based approaches do not fully characterize the spatial and temporal variability of fuels, site-specific conditions, and the presence of disturbances such as harvests and prescribed fires (Lutes et al., 2009; Brown and See, 1986, 1981). In light of these considerations, an ecohydrological modeling approach was implemented for this assessment to supplement existing FCCS data, specifically to characterize spatial and temporal variations more fully in forest fuel loads arising from site-specific biophysical and disturbance conditions.

The VELMA model is a spatially distributed (grid-based) ecohydrological model that simulates integrated responses of vegetation, soil, and hydrologic components to various inputs of land use, soil, and climate (McKane et al., 2014). It has been widely applied to many terrestrial ecosystem types, including forests, grasslands, agricultural floodplains, and alpine and urban landscapes. Particularly in western U.S. forests and grasslands, VELMA has simulated effects of fire and harvest and subsequent spatial and temporal dynamics of ecosystem recovery (McKane et al., 2020; Yee et al., 2017; Barnhart et al., 2015; Abdelnour et al., 2013; Abdelnour et al., 2011).

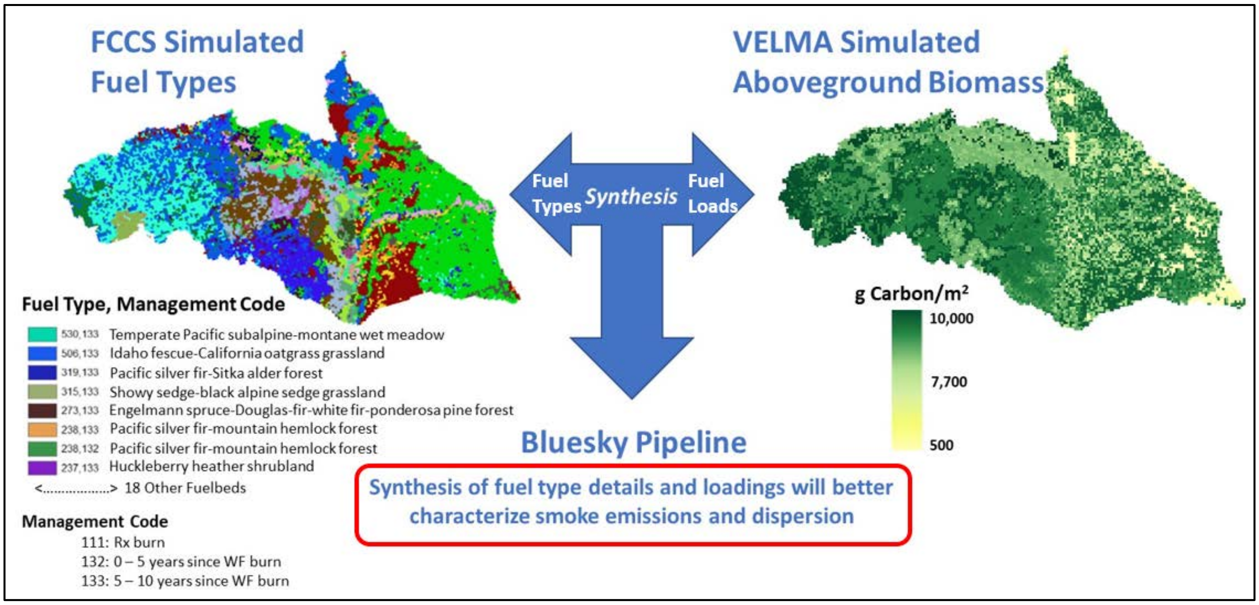
VELMA was used here to simulate aboveground biomass for the two case study fires (i.e., the TC6 Fire in Oregon and the Rough Fire in California). In addition, for the Rough Fire case study, VELMA modeling was conducted for additional fires within the actual Rough Fire vicinity to support the development of hypothetical scenarios. This additional modeling included the areas of the Sheep Complex Fire and the area within the proposed Boulder Creek Prescribed Fire. More detailed information on the actual and hypothetical fuel treatments and boundaries are described in Chapter 3 and in the present chapter. For each case study area, VELMA was spatially initialized using high-resolution (30-m), aboveground total (live and dead) biomass developed for western forest ecosystems (California, Oregon, Washington) by the Landscape Ecology, Modeling, Mapping, and Analysis (LEMMA) project at Oregon State University (LEMMA, 2020; Kennedy et al., 2018; Davis et al., 2015). LEMMA forest biomass map data are developed and updated annually using state-of-the-science, satellite-based, change-detection technology (Landsat) calibrated using the U.S. Forest Service (USFS) Forest Inventory and Analysis (FIA) regional network of forest biomass plot measurements (Bell et al., 2018).

Extensive validation of LEMMA-mapped biomass predictions has previously been performed for the western U.S., including the Deschutes National Forest near the TC6 case study area (Bell et al., 2018). In validation tests, the LEMMA-initialized VELMA TC6 application closely simulated aboveground biomass pools and rates of accumulation published for this dry coniferous forest ecoregion (Smithwick et al., 2002). This is important because VELMA was initialized for the 2018 TC6 Fire based on 2010 LEMMA biomass data, primarily to allow for potential future prefire fuel reduction simulation treatments using VELMA. The LEMMA aboveground live and dead forest biomass data for the Sheep Complex and Rough fires corresponded to the actual 5 years, 2010 and 2015, respectively. See appendix for details.

Following initialization for each case study site, VELMA's LEMMA-based overstory fuel-load estimates were merged with FCCS surface fuel load estimates, specifically to replace FCCS forest overstory fuel load estimates assumed to be homogenous within each vegetation type, rather than on location-specific data (Lutes et al., 2009; Brown and See, 1986, 1981).

Figure 7-6 generally illustrates how VELMA and FCCS data products were merged and fed into the BlueSky Pipeline. The combined VELMA-FCCS fuelbed database for each site was used as an input to the BlueSky Pipeline, specifically to the CONSUME model, to simulate air quality impacts associated with wildfire and prescribed fire simulations. The resulting BlueSky Pipeline input data comprised a raster map of fuelbed classifications and a comma-separated value (CSV) look-up file of fuel loadings for various fuelbed categories (e.g., canopy, shrubs, nonwoody vegetation, woody fuels, litter/lichen/moss, ground fuels) that include merged FCCS and VELMA fuel type and fuel load data, respectively. The combined use of FCCS and VELMA for this purpose plays to the strengths of both models, which together represent the best available science for estimating fine-scale horizontal and vertical distributions of fuelbed types and loadings (Bell et al., 2018; Ottmar et al., 2007).

In summary, whereas FCCS performs well at providing estimates of management-sensitive surface and understory fuel types and loads, VELMA performs well at estimating overstory/canopy fuel loads by virtue of its use of LEMMA initialization and mechanistically modeled live and dead biomass dynamics. Additional details on the methods used to develop LEMMA-initialized VELMA for both case studies and associated VELMA-FCCS fuelbed databases are located in Appendix A.7.



FCCS = Fuel Characteristic Classification System; g Carbon/m^2 = grams of carbon per square meter; Rx = prescribed burn; VELMA = Visualizing Ecosystem Land Management Assessments; WF = wildland fire.

Note: Example shown is for the Timber Crater 6 (TC6) Fire case study in Oregon.

Figure 7-6 Fuel Characteristic Classification System (FCCS) fuel type data and Visualizing Ecosystem Land Management Assessments (VELMA) fuel load data were merged to produce fuelbed inputs for the BlueSky Pipeline.

7.2.3 FUEL CONSUMPTION AND FIRE EMISSIONS (BLUESKY PIPELINE)

The BlueSky Pipeline Version 4.2.14 was used to support this project. For all the fire emission scenarios, the BlueSky Pipeline was used to calculate consumption, emission factors, and emissions using georeferenced area burned input. Generally, as fire data flow through the modules within the BlueSky Pipeline, the modules add to the data without modifying what was already defined. The consumption module used was CONSUME v5.0.2. Fuel loadings were based on either FCCS v3 with LANDFIRE v1.4 fuel beds or FCCS v4 with USFS fuel beds. Emissions were based on University of Washington SERA emission factors for the case study fires and Fire Emission Production Simulator (FEPS) v2 for all other fires.

The BlueSky Pipeline does not have a way to include meteorological variables such as relative humidity or fuel moisture as a dynamic input. Fuel moisture can be specified as fixed values for individual fires or groups of fires through the configuration for different types of fuels. The default wildfire fuel moisture is 50% for 10 hours, 30% for 1,000 hours, 75% for duff, and 16% for litter. The default prescribed fire fuel moisture is 50% for 10 hours, 35% for 1,000 hours, 100% for duff, and 22% for litter. Adding dynamic fuel moisture as an input is a planned update to the system. More work is likely needed to develop confidence in fuel-specific moisture-consumption relationships and how accurate fuel moisture data products are, given the sparsity of available ambient data.

7.2.3.1 TEMPORAL PROFILE FOR TIMBER CRATER 6 (TC6) FIRE

Fire hotspot characterization data from the Geostationary Operational Environmental Satellite (GOES)-16 Advanced Baseline Imager (ABI) were obtained in Network Common Data Form (NetCDF) format from the Amazon Web Service's S3 file system at s3://noaa-goes16/ABI-L2-FDCC/2018/ (GOES-R Algorithm Working Group, 2018). The data set comprises latitude, longitude, fire radiative power (FRP), estimated fire area, fire temperature, and a data quality factor (DQF) for each pixel. The fire data are derived (i.e., not directly measured) products of the GOES-ABI. The algorithms for deriving fire data and data quality are described elsewhere (Schmidt et al., 2013). Data from July 15−29, 2018 were extracted from within a bounding box defined by the points (43.03°N, 122.1°W) and (43.1°N, 121.9°W), centered roughly on the centroid of the final Timber Crater 6 Fire perimeter. Although data are typically available at 5-minute intervals, there are often large temporal discontinuities due to absence of detection because of issues such as low fire power, glare, or obscuration by smoke or clouds. After filtering for validity (DQF = 0), 166 data points were available for the analysis. Analysis was performed using Python 3 code and libraries.

Fire radiative power is proportional to the rate of fuel consumption in wildland fires (Kremens et al., 2012). To derive a characteristic fuel consumption curve, valid FRP values from all days in the data set were binned by hour. A mean value and standard deviation of FRP was calculated for each hour. Valid detections were available for only the hours 11:00 a.m. to 9:00 p.m. PDT each day over the time span of the fire. It is possible that fire radiative powers were too low and/or weather conditions were not favorable for detections outside of that range of times. A Weibull-like curve function (Barnett, 2002) was fitted to the hourly mean FRP values using the "curve_fit" method from the SciPy (Version 1.4.1) Optimize library. To facilitate curve fitting, mean FRP values outside of the available time range were extrapolated using a linear ramp between the end values (Numpy v1.18.5 "pad" function).

The resulting fitted curve gives a realistic profile of diurnal fuel consumption and indicates that, on average, peak FRP, and therefore fuel consumption, occurred around 3:00 p.m. PDT. However, the FRP curves for any given day of the fire varied considerably from the fitted curve, as indicated by the large variations in FRP during the afternoon hours. Importantly, with only 166 data points there is a relative paucity of GOES satellite data for this fire, suggesting that the resulting consumption curve should be used with caution.

The default wildfire temporal profile was used for the Rough Fire. The size and length of the fire made development of day- and location-specific temporal profiles challenging, and the emissions modeling system is not currently well positioned to use information at that specific time and space.

7.2.4 PILE/SLASH BURN EMISSIONS

Typical practices for collecting fuel left over from mechanical thinning operations include collecting the debris into three types of piles: machine landing piles (largest), machine grappling piles, and hand piles (smallest). Each of these practices are common at Crater Lake National Park and surrounding areas. Typical geometry for each was provided by land managers in the region: machine landing pile (50′ × 100′ × 25′), machine grappling pile (15′ × 15′ × 10′), and hand pile (5′ × 5′ × 5′). Although mechanical thinning is a common practice in the area near the TC6 Fire and pile burns are common to eliminate the debris, the scenarios explored as part of this assessment did not include mechanical thinning, so debris piles were not estimated. This deficiency should be considered as part of future comparative assessments.

7.2.5 AIR QUALITY MODELING SYSTEM

The CMAQ v5.3.2 model was applied with aqueous-phase chemistry (Fahey et al., 2017), inorganic thermodynamics (Fountoukis and Nenes, 2007), and gas-phase chemistry based on the Carbon Bond 6 Revision 3 mechanism (Emery et al., 2015). The default option was used where photolysis rates were attenuated in the presence of model-predicted particulate matter (Baker et al., 2016). Secondary organic aerosol (SOA) treatment is a yield-based approach based on precursors, including isoprene, monoterpenes, sesquiterpenes, benzene, toluene, and xylenes. Some of the SOAs become nonvolatile through oligomerization processes (Carlton et al., 2010). Primarily emitted organic aerosol is treated as nonvolatile. The ratio of organic mass to organic carbon is assumed to be 1.7 for primary $PM_{2.5}$ wildland fire emissions (Simon and Bhave, 2012).

The WRF model was used to provide the modeling system meteorological inputs (Skamarock et al., 2008). Both CMAQ and WRF were applied with 35 layers to represent the vertical atmosphere from the surface up to 50 mb. The WRF configuration used here has been evaluated and has shown reasonable performance for winds, temperature, and surface mixing layer height for the Pacific Northwest (Zhou et al., 2018) and California (Baker et al., 2013). WRF was initialized with the 12-km North American mesoscale (NAM) analysis product [https://www.ncdc.noaa.gov/data-access/model-data/model-datasets/north-american-mesoscale-forecast-system-nam; NCEP (2021)]. CMAQ initialization and boundary inflow conditions were extracted from coarser hemispheric CMAQ simulations.

Anthropogenic emissions in the model domain were based on the 2016 National Emission Inventory (U.S. EPA, 2019b) with year-specific data used for electricity-generating units based on continuous emissions monitor data. Biogenic emissions were estimated with the Biogenic Emission Inventory System v3.6.1, which has been shown to perform well for biogenic VOCs in California (Bash et al., 2016). Emissions of wildland fires other than the case studies were based on daily fire location and burn area information using the SmartFire2 system, which is largely based on satellite products and incident information. Location, burn area, and date information is provided to the BlueSky Pipeline to estimate fuel type, fuel moisture, and fuel consumption that is used to estimate daily emissions (Urbanski, 2014) of CO, NO_X, VOC, SO_2, NH_3, and $PM_{2.5}$ based on FEPSv2 emission factors for each non-case-study wildfire and prescribed fire in the model domain (Larkin et al., 2020).

SMOKE is used to apply a fire type-specific diurnal profile and allocates total emissions of NO_X, VOC, and $PM_{2.5}$ to specific model species needed for chemical mechanisms. Speciation profiles are based on those available in the SPECIATE database [https://www.epa.gov/air-emissions-modeling/speciate; U.S. EPA (2020b)]. NO_X emissions were allocated 10% to NO and 90% to NO_2. Speciation profiles for VOC and primarily emitted $PM_{2.5}$ are provided in Appendix Table A.7-3. Daily total emissions were allocated to specific hours of the day based on default profiles for wildfire and prescribed fire (Baker et al., 2020; Baker et al., 2016). Fuel moisture is a global parameter that only varies by fire type (wildfire or prescribed).

Wild and prescribed fire plume rise is based on a modified Briggs approach and calculated in the CMAQ model. This approach has been shown to reasonably replicate the plume top of large wildfires in the western U.S. (Baker et al., 2018; Baker et al., 2016). Also, it has been shown to perform well for smaller fires when realistic parameters such as acres burned were provided as input to the modeling system (Zhou et al., 2018).

7.3 RESULTS—CASE STUDIES

For both the TC6 Fire and Rough Fire case studies total acres burned, $PM_{2.5}$ emissions, fuel, and fuel consumption are shown for the wildfire, alternative hypothetical scenarios, and areas that had been managed in the past in Table 7-1.

Photochemical model predictions of baseline maximum daily 8-hour average (MDA8) O_3 and major components of speciated $PM_{2.5}$ (total carbon, sulfate ion, and nitrate ion) were paired in time and space with measurements from routine surface network monitors. This type of comparison provides information about how well the modeling system is predicting air quality from wildland fire and other sources. A reasonable representation of the chemical environment surrounding fire plumes is important to best capture secondarily formed pollutants like O_3 because wildland fire emit precursors of O_3 (NO_X and VOC) that can react with other sources of pollution to form O_3.

The photochemical modeling system generally compares well with ambient data for the various episodes included in this assessment. Model performance metrics for daily model-observation pairs at routine surface network monitors aggregated over each episode are shown in Appendix Figure A.7-1. Each prediction-observation pair is also shown with scatterplots for each species (Appendix Figure A.7-1 to Appendix Figure A.7-6). Additional model performance information is provided as part of subsequent figures in this section that show episode average surface level modeled $PM_{2.5}$ and MDA8 O_3 compared with measurements made at routine monitors. The modeling system does well at replicating spatial gradients in $PM_{2.5}$ and O_3. It also generally captures synoptic and day-to-day variability in measurements near each of the case study fires. MDA8 O_3 is well predicted (n = 273, 2.3 parts per billion [ppb] mean bias, 4% normalized mean bias, and 14% normalized mean error) and $PM_{2.5}$ organic carbon is slightly underpredicted (n = 46, −1.2 μg/m^3 mean bias, −18% normalized mean bias, and 65% normalized mean error). The performance metrics for these episodes is consistent with the performance shown for this type of modeling system for monitors affected by large wildfires in the western U.S. (Baker et al., 2018; Koplitz et al., 2018; Baker et al., 2016). Very little data exist on episodic model performance for these areas during large wildfire events for performance comparison. However, performance metrics of other studies completed over longer time frames and larger model domains are generally consistent with those estimated for the modeling periods included in this assessment (Kelly et al., 2019; Simon et al., 2012).

Table 7-1 Wildfire and prescribed fires modeled as part of the Timber Crater 6 (TC6) and Rough fire case studies.

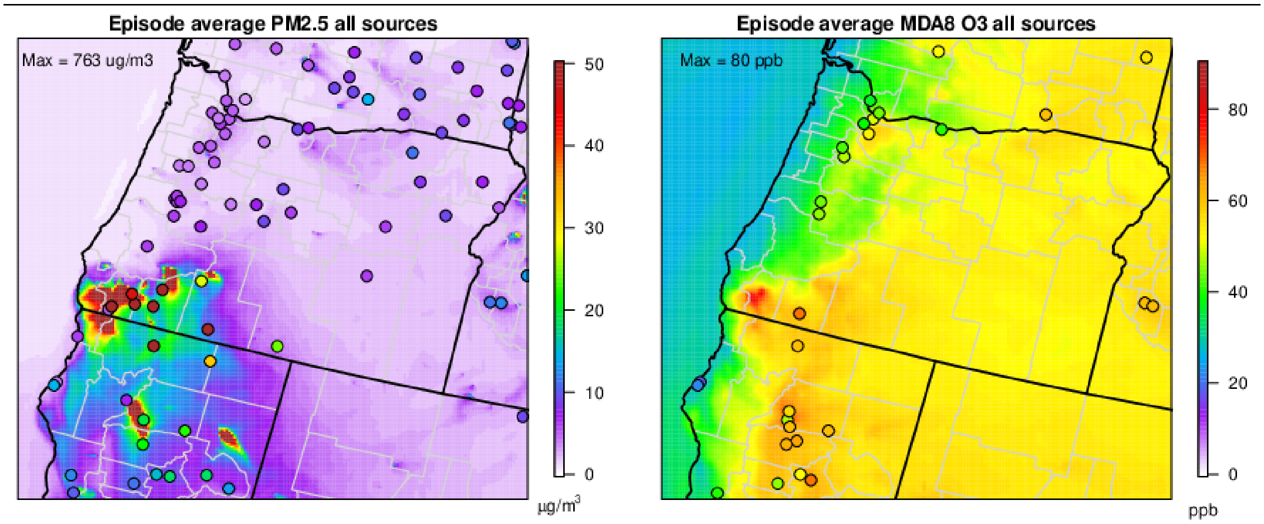
Fire/Burn Unit Name	Type	Modeled Time Period	Acres Burned Acres	Total Fuel Consumption Tons	Total Fuel Tons	$PM_{2.5}$ Emissions Tons
TC6	Wildfire	July 15 to 31, 2018	3,123	213,454	145,985	1,869
TC6 hypothetical smaller fire (1)	Hypothetical wildfire	July 15 to 31, 2018	1,237	37,954	91,419	1,041
TC6 hypothetical larger fire (2a)	Hypothetical wildfire	July 15 to 31, 2018	20,878	468,843	1,249,089	12,794
TC6 hypothetical larger fire (2b)	Hypothetical wildfire	July 15 to 31, 2018	27,373	727,180	1,825,606	20,015
Timber Crater 1978	Prescribed fire undefined date	September 1 to 30, 2019	2,049	26,992	112,362	565
Cornerstone	Prescribed fire undefined date	September 1 to 30, 2019	772	10,671	69,787	232
Timber Crater 1/2	Prescribed fire undefined date	September 1 to 30, 2019	633	7,751	37,649	157
2019 prescribed fires	Prescribed fire	September 1 to 30, 2019	886	6,206	20,955	117
Rough Fire	Wildfire	August 1 to September 30, 2015	145,438	3,284,638	7,128,199	85,638
Rough hypothetical smaller fire (1)	Hypothetical wildfire	August 1 to September 30, 2015	113,349	2,631,258	6,450,696	68,949
Rough hypothetical larger fire (2)	Hypothetical wildfire	August 1 to September 30, 2015	154,354	3,448,094	7,562,392	89,349
Boulder Creek Unit 1	Hypothetical prescribed fire	September 26 to October 7, 2014	3,289	30,163	90,452	499
Sheep Complex Fire	Fire	July 30 to September 30, 2010	8,916	103,037	434,193	2,344

$PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 μm; TC6 = Timber Crater 6.

7.3.1 TIMBER CRATER 6 (TC6) FIRE AIR QUALITY IMPACTS

A domain with 4-km grid cells covering Oregon and northern California were applied for the time period coinciding with the case study fire (July 2018). Initial conditions and boundary inflow were extracted from a CMAQ simulation for a 12-km domain covering the continental U.S. for the entire year of 2018.

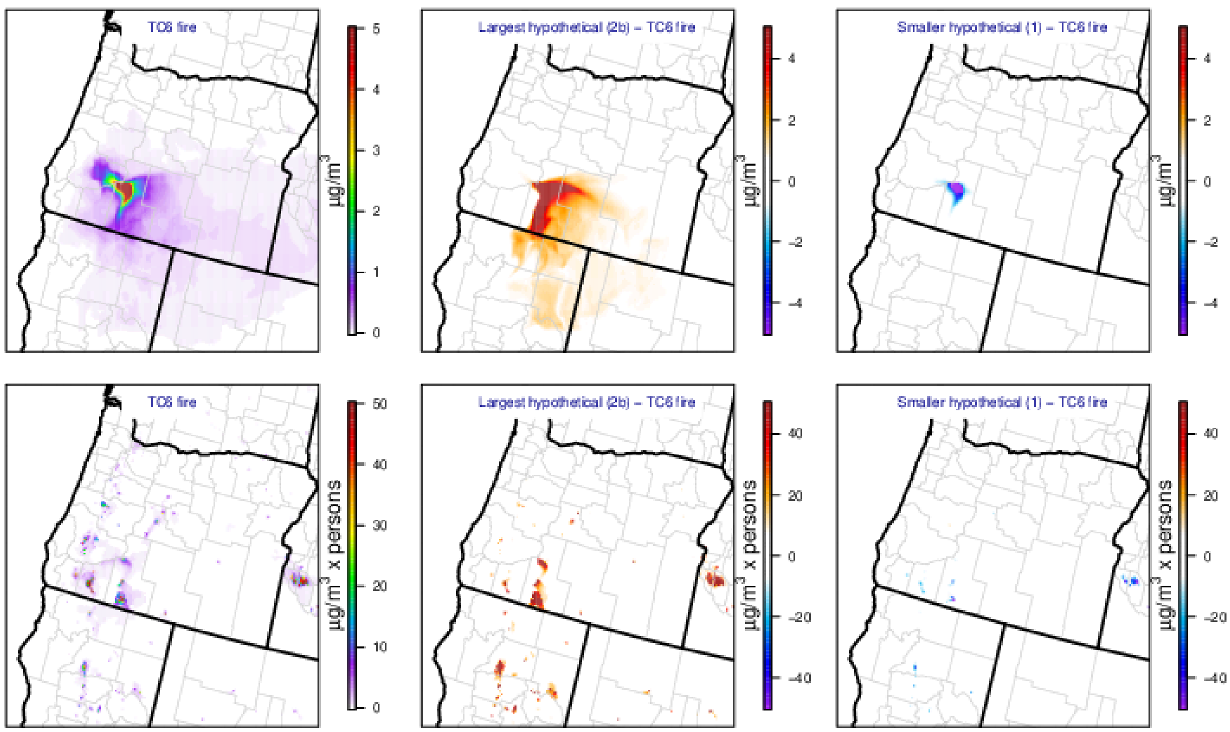
Model-predicted episode average $PM_{2.5}$ and MDA8 O_3 for the 2018 episode compared well with routine surface monitor data (Figure 7-7). Large wildfires in southwestern Oregon and northern California resulted in a strong gradient in $PM_{2.5}$ concentrations across the domain. Enhancements of O_3 from wildfire were less evident because meteorologic conditions during this period were favorable to regional formation. Agreement between model predictions and measurements provides confidence that the actual and hypothetical case study fires are being modeled in a realistic chemical and physical environment.



µg/m³ = micrograms per cubic meter; max = maximum; MDA8 = maximum daily 8-hour average; O_3 = ozone; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm; ppb = parts per billion.

Figure 7-7 Episode average $PM_{2.5}$ and maximum daily 8-hour average (MDA8) ozone (O_3) predicted by the modeling system and measured by routine surface monitors for the 2018 modeling period used for the Timber Crater 6 (TC6) scenarios.

Episode average model predicted $PM_{2.5}$ from the TC6 Fire and hypothetical scenarios are shown in Figure 7-8 (top row). Note that in Figure 7-8 and Figure 7-9 the color scales vary from panel to panel. To assess population exposure to $PM_{2.5}$ produced by the TC6 Fire, model predictions were also multiplied by gridded population to provide an estimate of aggregate population exposure Figure 7-8, bottom row). Figure 7-8 also shows the difference in episode average $PM_{2.5}$ between the largest and smallest hypothetical scenarios and the actual fire scenario. The spatial pattern of differences between the largest hypothetical TC6 Fire scenario (2b) and the TC6 Fire is strongly influenced by days toward the end of the largest hypothetical fire scenario when nighttime winds blew smoke southward toward the Oregon-California border. The spatial extent of impacts from the hypothetical TC6 Fire scenario 2a (not shown) are similar to hypothetical TC6 Fire Scenario 2b, but with a smaller magnitude of change.

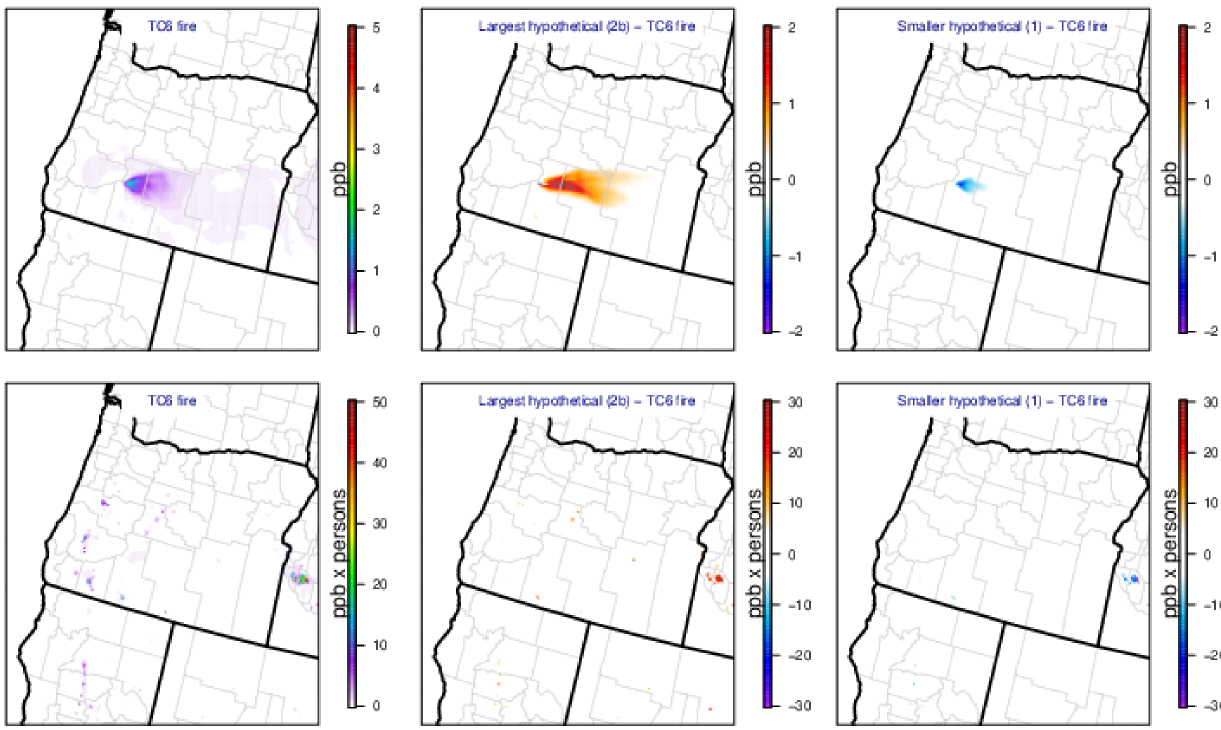


µg/m³ = micrograms per cubic meter; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm; TC6 = Timber Crater 6.

Note: Ambient $PM_{2.5}$ impacts are shown in the top row and aggregate population exposure in the bottom row where $PM_{2.5}$ is multiplied by gridded population.

Figure 7-8 Episode average $PM_{2.5}$ impacts and aggregate population exposure from the actual Timber Crater 6 (TC6) Fire and the difference between the actual fire and largest (2b) and smaller (1) hypothetical scenarios.

The episode average model predicted MDA8 O_3 from the TC6 Fire and hypothetical TC Fire Scenarios 1 and 2b are shown in Figure 7-9 (top row). Model predictions are also multiplied by gridded population to provide an estimate of aggregated population impacts. The spatial pattern of differences between the largest hypothetical TC Fire scenario (2b) and actual TC6 Fire is strongly influenced by daytime winds blowing smoke eastward toward the Oregon-Idaho border. This differs from the spatial extent of $PM_{2.5}$ impacts because the largest $PM_{2.5}$ concentrations are overnight when winds moved air toward the south. Impacts of the daytime wind patterns dominate the spatial extent of O_3 formation because these daytime winds coincide with solar radiation, which is needed for photochemical O_3 production.



ppb = parts per billion; TC6 = Timber Crater 6.

Note: Ambient MDA8 O_3 impacts are shown in the top row and aggregate population exposure in the bottom row where MDA8 O_3 is multiplied by gridded population.

Figure 7-9 Episode average maximum daily 8-hour average (MDA8) ozone (O_3) impacts and aggregate population exposure from the actual Timber Crater 6 (TC6) Fire and the difference between the actual TC6 Fire and largest (2b) and small (1) hypothetical scenarios.

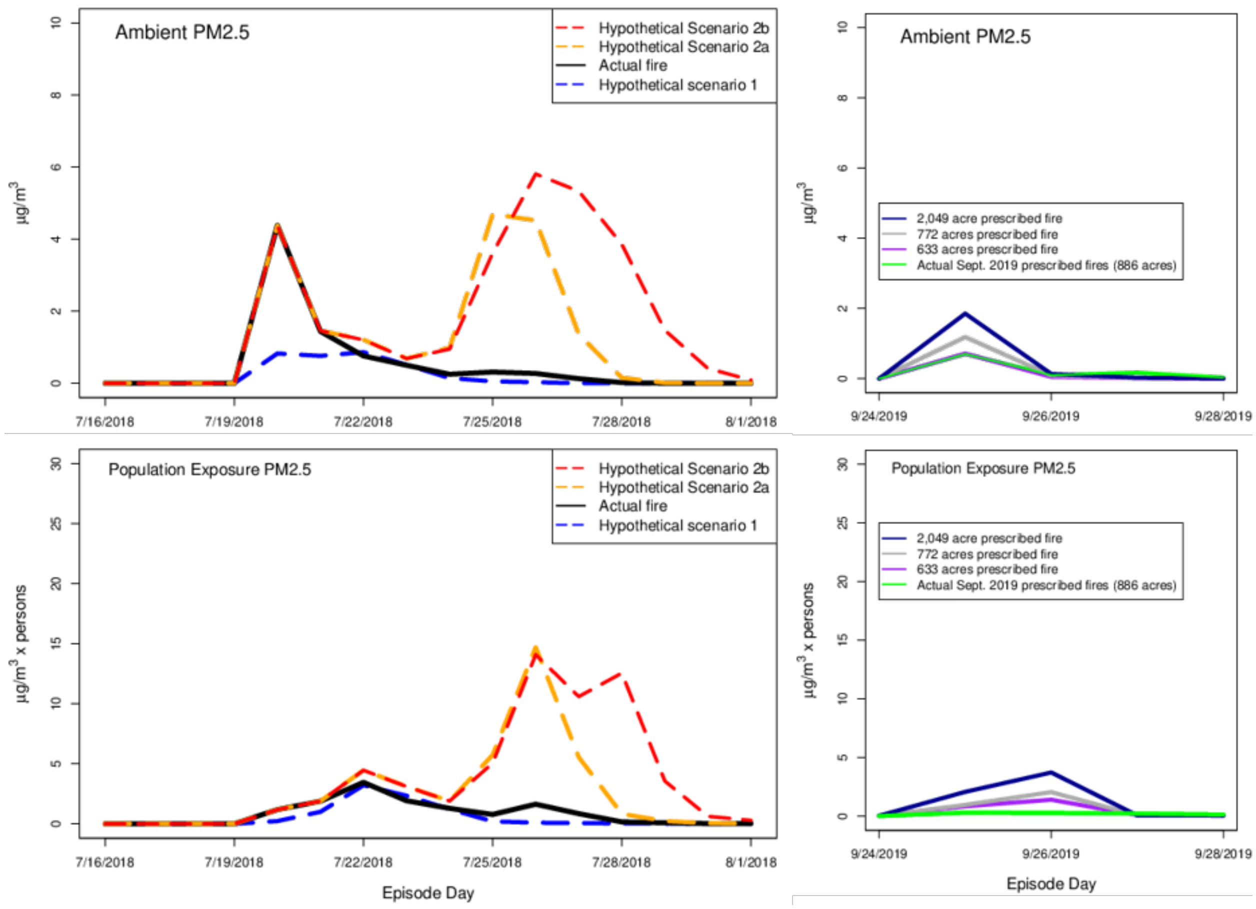
Without considering air quality impacts, based on the TC6 Fire case study and other similar studies, results indicate that land management, such as prescribed fire and mechanical thinning, reduce fuel, which means less fuel is consumed when wildfires happen later. Less fuel available for wildfire consumption in turn means less emissions and lower levels of downwind pollutants. Reduced fuel loading also can lead to smaller fire perimeters, which is represented in Scenario 1 (i.e., the smaller TC6 Fire hypothetical, presented here). This smaller perimeter is based on expert judgment for this hypothetical scenario and is not based on fire behavior or fire spread models. Illustrating the change in air quality related to past land management activity is challenging because spatial and temporal scales of both are quite different. For instance, many prescribed fires may need to be conducted over many years to effectively minimize the rate of spread of wildfire or reduce fuels enough to affect air quality. Further, only a single period of conducive meteorology (September 2019) was used for the prescribed fire impacts, which does not capture the variability possible if other years or time of year were chosen.

Figure 7-10 shows daily domain average $PM_{2.5}$ ambient and aggregate population exposure from the TC6 Fire and hypothetical fire scenarios compared with multiple prescribed fires. All the prescribed fires were modeled in separate simulations with the same days in September 2019 when prescribed fires were happening near Crater Lake National Park. Similar information is shown for MDA8 O_3 in Figure 7-11. The daily average impacts only include grid cell-days where modeled fire impacts exceed a threshold (0.01 $\mu g/m^3$ for $PM_{2.5}$ and 0.01 ppb for MDA8 O_3) so that the average does not include large areas of the model domain with no fire impacts because of wind transport patterns.

Daily aggregate population exposures are notably different than ambient impacts for July 20 when ambient concentrations were high, but winds did not transport smoke to populated areas. The prescribed fires had high ambient impacts but did not affect highly populated areas in this case study. The large estimated population exposures of the biggest hypothetical fires toward the middle and end of the episode are related to larger fire size (e.g., more fuel consumption and emissions) and winds blowing smoke towards populated areas on the additional simulation days.

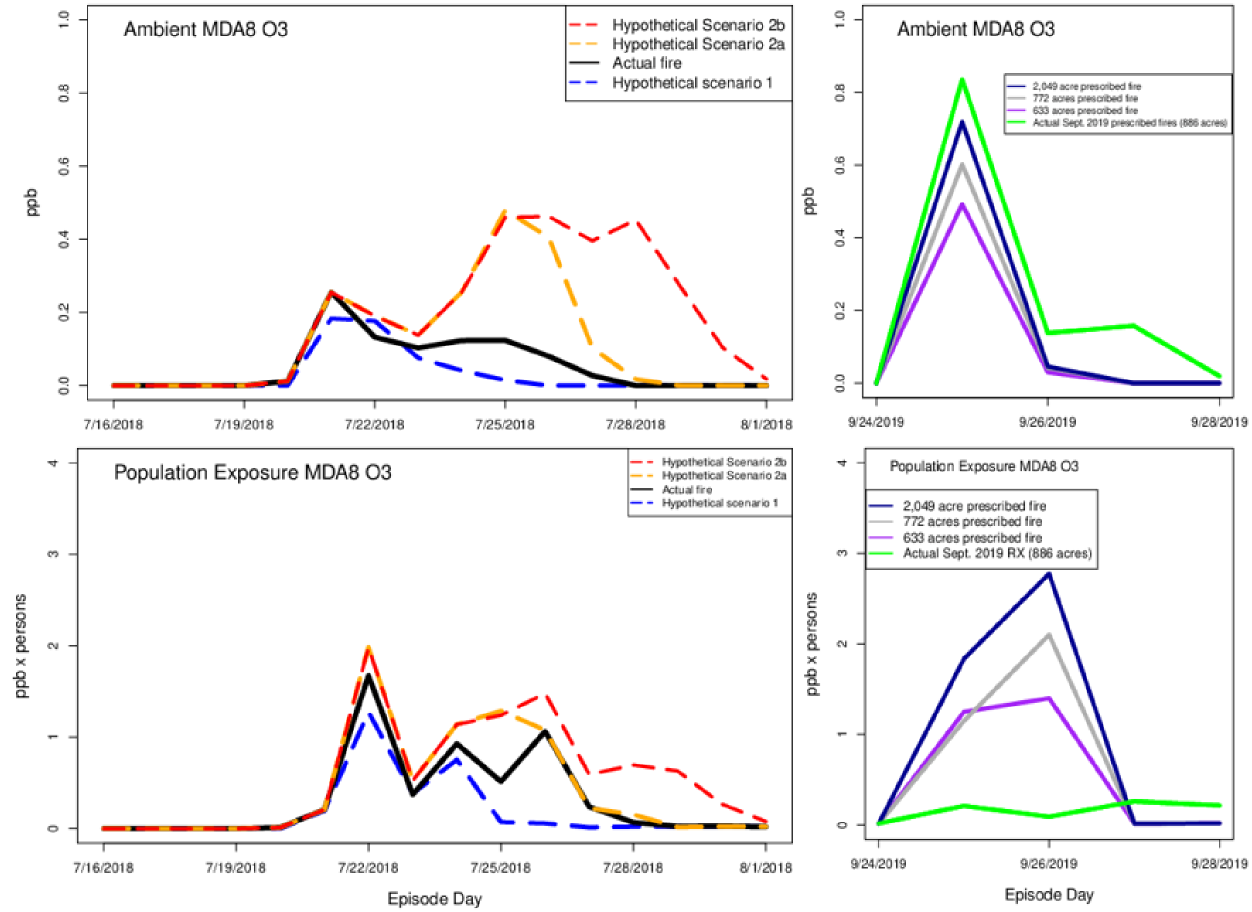
The daily impacts of prescribed fire on $PM_{2.5}$, particularly the estimated population exposures, were typically lower than of wildfire. However, the daily impacts of MDA8 O_3 from prescribed fire were sometimes comparable to or even larger than the wildfire scenarios. This is due to the large amount of fuel burned as part of the hypothetical prescribed fires on a single day compared to the daily amount of fuel consumed by these small (small compared to the Rough Fire for instance) hypothetical wildfire scenarios. Further, the prescribed fire emissions are temporally allocated to daytime hours which means more of the mass is available for photochemical reactions leading to O_3 production compared to wildfire emissions which are spread out over the entire day and night.

Figure 7-12 shows daily average $PM_{2.5}$ impacts of the TC6 Fire. This figure illustrates the day-to-day variability in near-fire and downwind impacts from the TC6 Fire from the 1st day to when the fire was extinguished. Ambient impacts were highest on the 2nd, 3rd, and 4th days. Winds tended to blow air toward the south on the last days of the fire affecting northern California. This wind pattern also helps show how the population impacts were different for the larger hypothetical scenarios because those fires continued to have high emissions on days when winds were affecting more populated areas to the south.



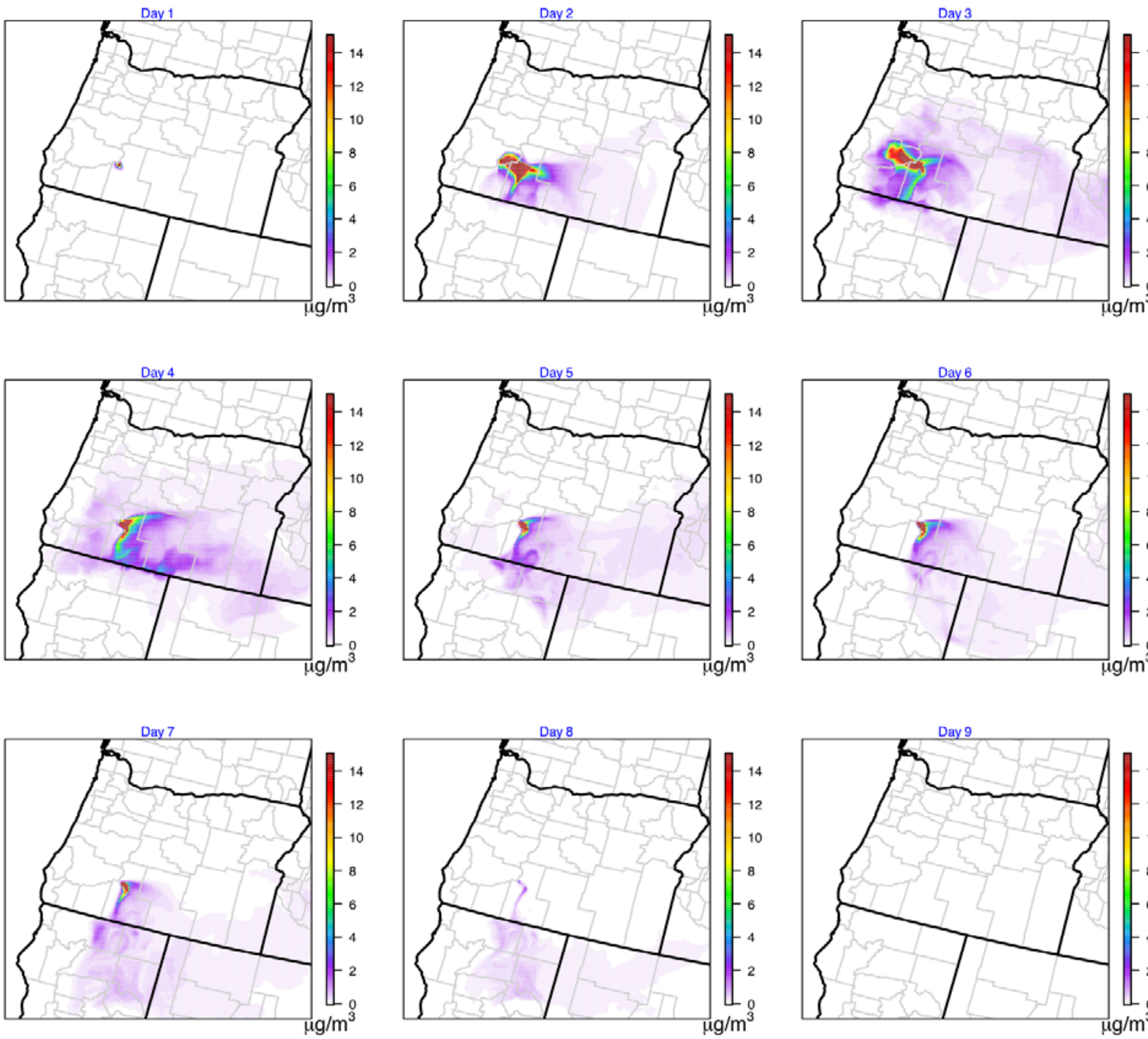
µg/m^3 = micrograms per cubic meter; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm.

Figure 7-10 Daily average $PM_{2.5}$ ambient (top row) impacts and estimates of aggregate population exposure (bottom row) from the actual Timber Crater 6 (TC6) Fire and hypothetical scenarios (left) and each prescribed fire (right).



MDA8 = maximum daily 8-hour average; O_3 = ozone; ppb = parts per billion; Rx = prescribed fire.

Figure 7-11 Maximum daily 8-hour average (MDA8) ozone (O_3) ambient (top row) impacts and estimates of aggregate population exposure (bottom row) from the actual Timber Crater 6 (TC6) Fire and hypothetical scenarios (left) and each prescribed fire (right).



µg/m^3 = micrograms per cubic meter.

Figure 7-12 Daily average $PM_{2.5}$ from the Timber Crater 6 (TC6) fire.

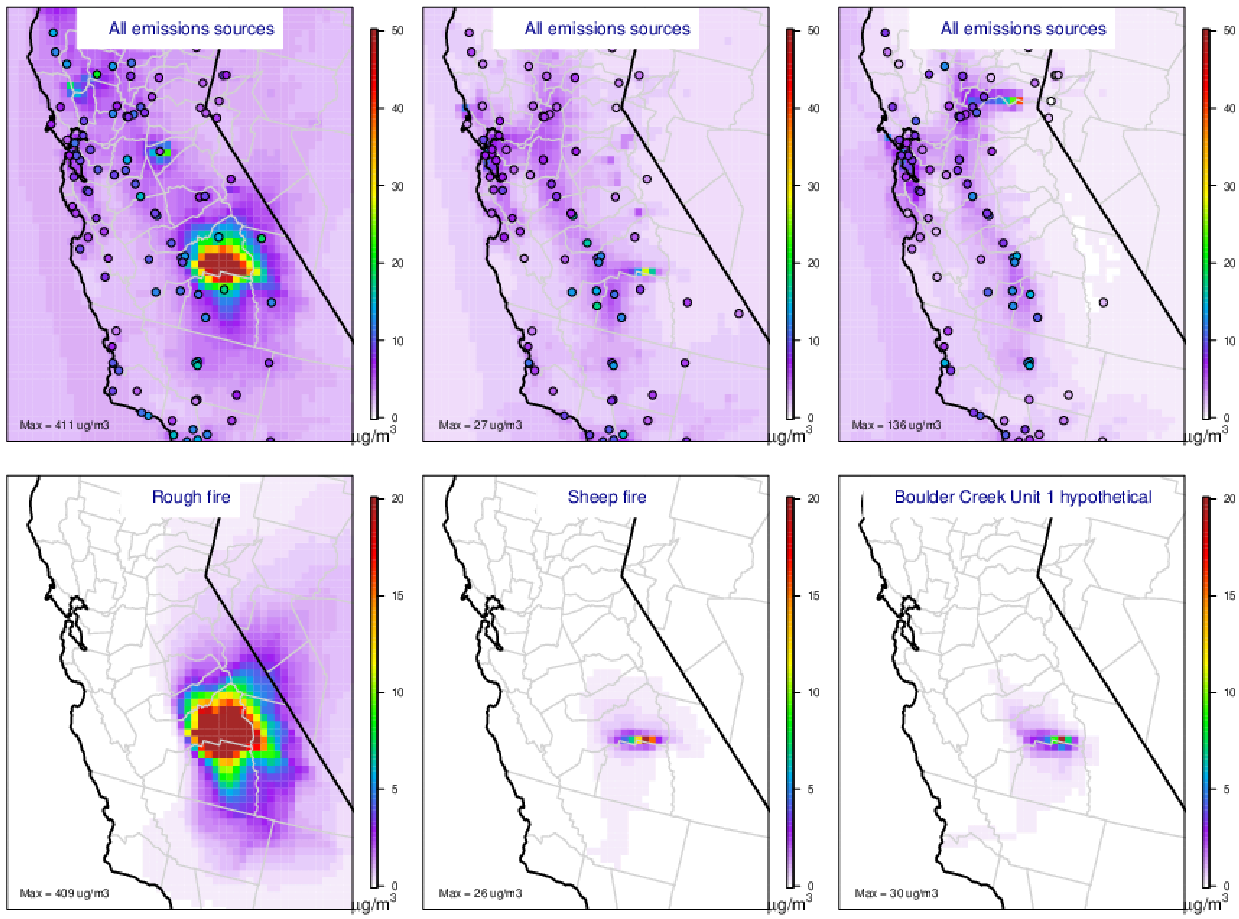
7.3.2 ROUGH FIRE AIR QUALITY IMPACTS

The modeling system was applied for the 2015 Rough Fire, a hypothetical smaller Rough Fire (Scenario 1), a hypothetical larger Rough Fire (Scenario 2), the 2010 Sheep Complex Fire, and a hypothetical prescribed fire (Boulder Creek Unit 1) for a period matching ideal meteorological conditions for prescribed fire in the fall of 2014. The larger Rough Fire hypothetical (Scenario 2) includes the actual Rough Fire in its entirety and also includes the area of the Sheep Complex Fire, which did not burn as part of the actual Rough Fire. The smaller Rough Fire hypothetical scenario (Scenario 1) eliminates sections of the actual Rough Fire that were downslope of an area planned for prescribed fire (Boulder Creek Unit 1) but never happened. This smaller hypothetical fire is based on the idea that if that prescribed fire had happened before the Rough Fire, it would have provided a boundary for fire suppression and stopped progression to areas downslope toward the Central Valley of California.

CMAQ was applied for a 12-km domain covering the continental U.S. Initial conditions and boundary inflow were extracted from a coarser hemispheric scale photochemical model simulation. This coarser grid spacing scale was selected for the larger Rough Fire case study because a larger domain was used in anticipation of impacts much further downwind than the TC6 Fire case study. Model simulations were done for periods coincident with case study fires in 2010 (Sheep Complex), 2014 (hypothetical Boulder Creek Unit 1), and 2015 (actual Rough Fire). Model predicted episode average $PM_{2.5}$ (Figure 7-13) and MDA8 O_3 (Figure 7-14) for each episode compared well with routine surface monitor data. MDA8 O_3 is well predicted (n = 11,510, 0.6 ppb mean bias, 1.2 % normalized mean bias, and 13% normalized mean error) and $PM_{2.5}$ organic carbon is slightly underpredicted (n = 536, −0.57 µg/m^3 mean bias, −19% normalized mean bias, and 60% normalized mean error). This agreement between model predictions and measurements provides confidence that the actual and hypothetical case study fires are being modeled in a realistic chemical and physical environment.

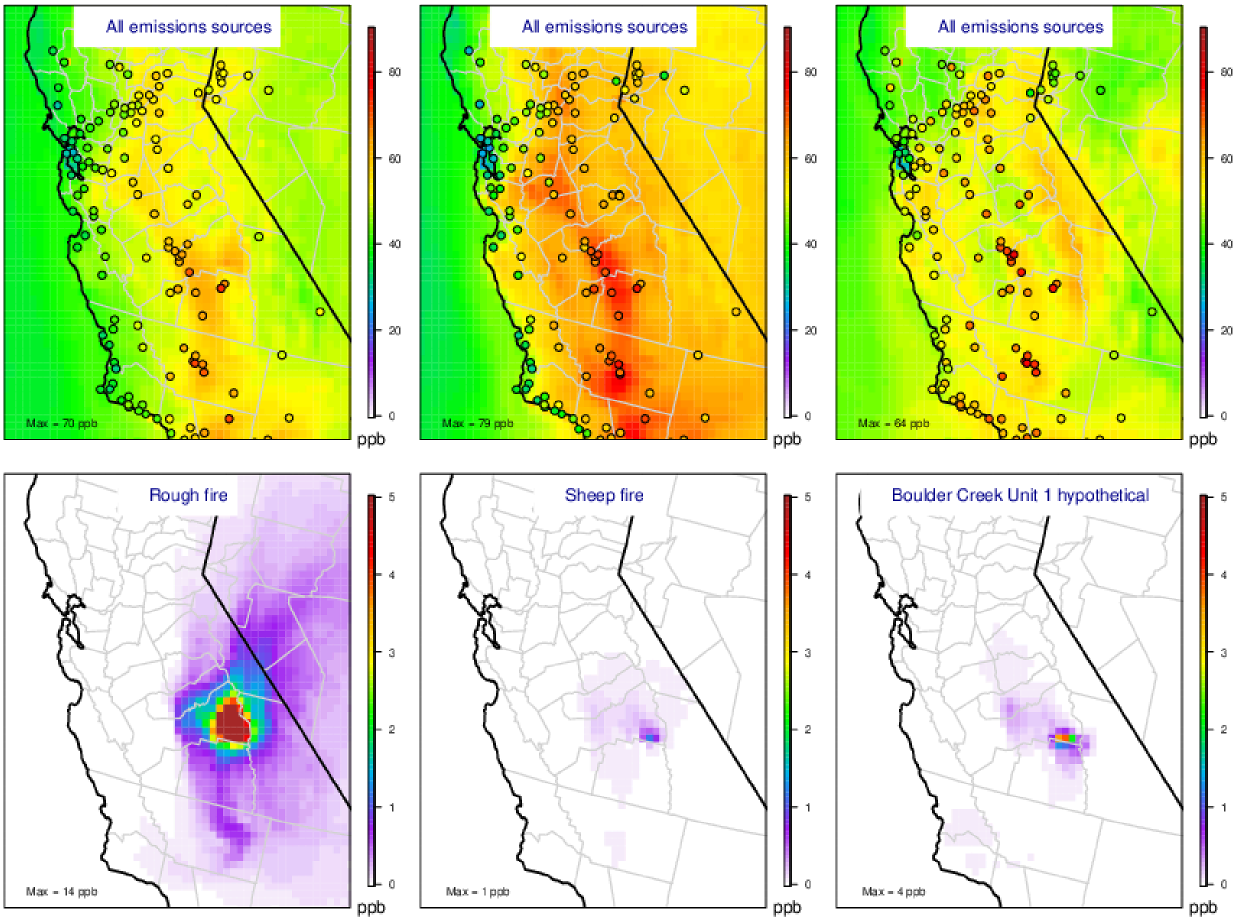
The actual Rough and Sheep Complex fires spanned multiple months. The Rough Fire had much larger downwind impacts which is related to the larger size of that fire in terms of acres burned and fuel consumed. The largest impacts from each of the fires is at the fire location itself with concentrations decreasing as distance from the fire increases. The episode average air quality impacts for the hypothetical Boulder Creek Unit 1 Prescribed Fire are averaged over a much shorter time period (10 days) than the Rough and Sheep Complex fires. This difference should be kept in consideration when comparing these spatial plots.

Each of the fires modeled as part of this case study have some impacts on populated areas in the Central Valley of California and further downwind toward the east. Some of the near-fire impacts on population areas may be overstated because of the 12-km-sized grid cells used for this case study, which may not have captured how complex terrain influenced meteorology and transport. This is particularly important to consider for the hypothetical Boulder Creek Unit 1 Prescribed Fire because the days for this fire were selected based on meteorology that was considered conducive to keeping air in the mountains and minimizing downslope flow to the Central Valley.



$\mu g/m^3$ = micrograms per cubic meter; max = maximum; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm.

Figure 7-13 Episode average $PM_{2.5}$ for the Rough Fire predicted by the modeling system (from all emissions sources) and measured by routine surface monitors (top row) and fire-specific modeled impacts (bottom row).

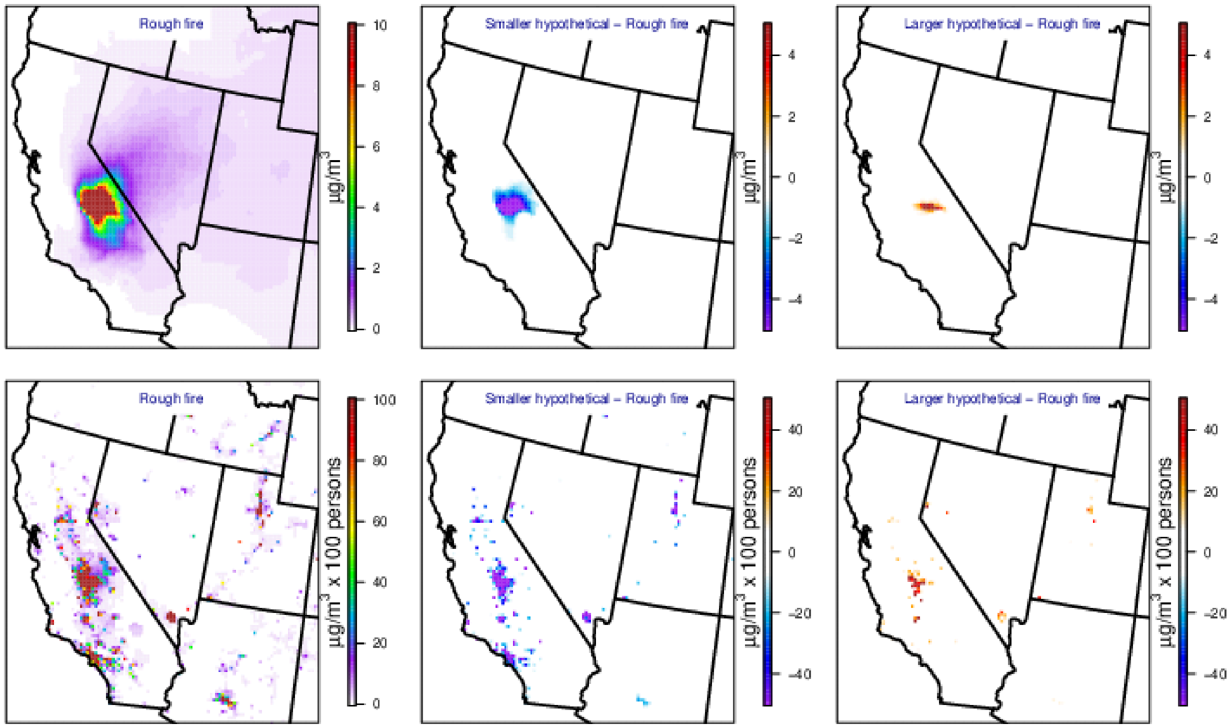


max = maximum; ppb = parts per billion.

Figure 7-14 Episode average maximum daily 8-hour average (MDA8) ozone (O_3) for the Rough Fire predicted by the modeling system (from all emissions sources) and measured by routine surface monitors (top row) and modeled fire impacts (bottom row).

Each of the fires modeled in this case study produce fairly small levels of MDA8 O_3 compared with regional levels measured at surface monitor sites during the same time periods (Figure 7-14). The spatial nature of elevated MDA8 O_3 in California suggests sources other than wildland fire (e.g., anthropogenic, biogenic, lateral boundary inflow) contributed the most to ambient surface level O_3.

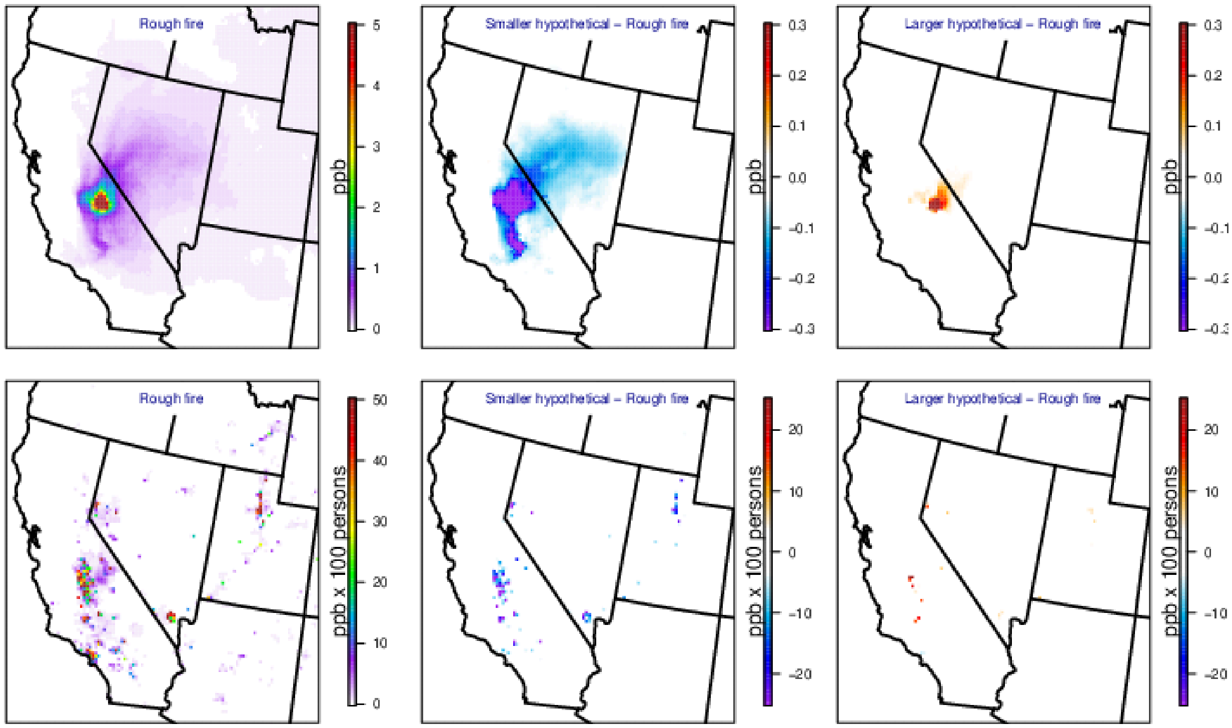
The episode average model predicted $PM_{2.5}$ from the actual Rough Fire is shown in Figure 7-15. Model predictions are also multiplied by gridded population to provide an estimate of aggregated population exposure. Figure 7-15 also shows the difference in episode average $PM_{2.5}$ between the hypothetical scenarios and actual fire scenario. Similar information is presented for MDA8 O_3 in Figure 7-16.



μg/m³ = micrograms per cubic meter; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 μm.

Note: Ambient $PM_{2.5}$ impacts are shown in the top row and aggregate population exposure in the bottom row where estimated $PM_{2.5}$ concentrations are multiplied by gridded population.

Figure 7-15 Episode average $PM_{2.5}$ impacts from the actual Rough Fire and the difference between the actual Rough Fire and smaller (Scenario 1) and larger (Scenario 2) hypothetical scenarios.



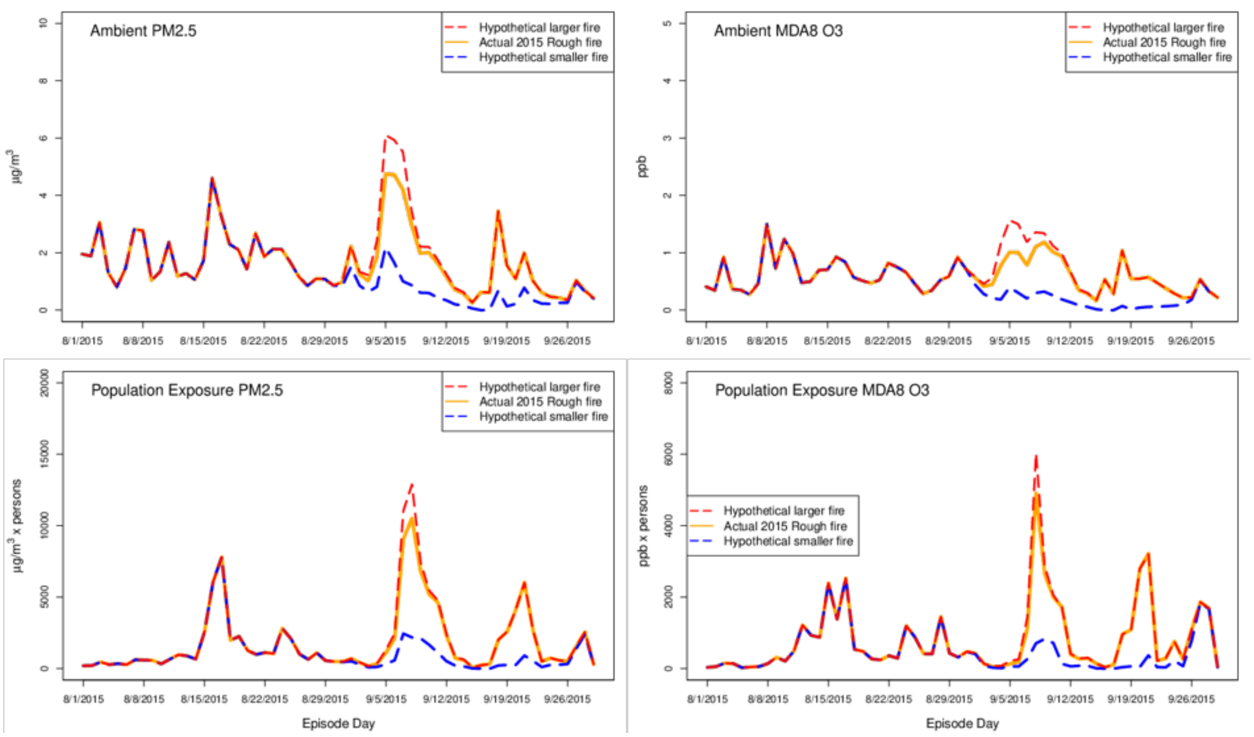
ppb = parts per billion.

Note: MDA8 O_3 impacts are shown in the top row and aggregate population exposure in the bottom row where estimated MDA8 O_3 concentrations are multiplied by gridded population.

Figure 7-16 Episode average maximum daily 8-hour average (MDA8) ozone (O_3) impacts from the actual Rough Fire and the difference between the actual Rough Fire and smaller (Scenario 1) and larger (Scenario 2) hypothetical scenarios.

The ambient impacts of the actual fires and hypothetical wildfire scenarios are highest in California and decrease downwind as air moves smoke into the intermountain west and central plains. When the impacts are multiplied by population, most urban areas in the model domain have nonzero impacts. This shows that very small concentrations of smoke in large population areas can result in aggregated exposure similar to sparsely populated areas near the fire. Rough Fire impacts on regional MDA8 O_3 are highest near the fire with smaller impacts in the Central Valley of California and central Nevada. Population impacts are also notable in large downwind urban areas like Salt Lake City.

Figure 7-17 shows daily domain average $PM_{2.5}$ ambient impacts and aggregate population exposure from the actual and hypothetical Rough Fire scenarios. Similar information is shown for MDA8 O_3 in Figure 7-17.

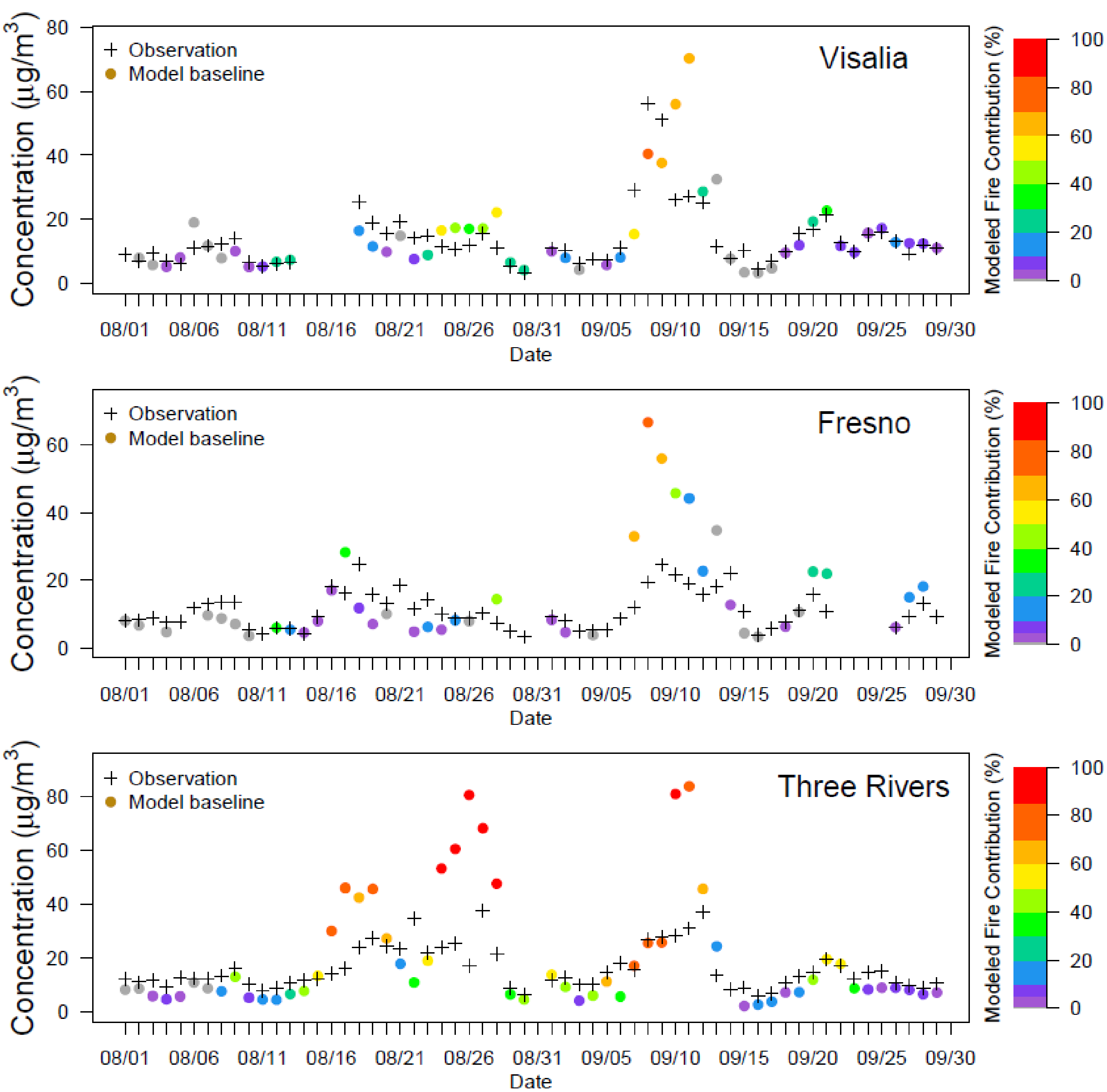


µg/m^3 = micrograms per cubic meter; MDA8 = maximum daily 8-hour average; O_3 = ozone; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm; ppb = parts per billion.

Figure 7-17 Daily average ambient (top row) $PM_{2.5}$ (left) and maximum daily 8-hour average (MDA8) ozone (O_3; right) impacts and aggregate population exposure (bottom row) from the actual Rough Fire and hypothetical scenarios.

Daily average impacts are the same for each scenario during the 1st month of the fire because the emissions are the same. The alternative scenarios diverge from the actual fire at the beginning of September. Aggregate population exposure is greatest when the model predicts impacts in the Central Valley of California for a period in early September and again to a lesser extent in mid-September. Ambient impacts are reduced in the smaller fire hypothetical scenario once the actual fire progresses to the Boulder Creek Unit 1 area and increases in the larger fire hypothetical scenario when the actual fire also includes the area of the Sheep Complex Fire.

Figure 7-18 shows daily $PM_{2.5}$ measurements and model predictions at multiple monitors in the Central Valley of California. These monitors were selected to provide an indication about how well the model captures smoke impacts from the Rough Fire. The model tends to overpredict $PM_{2.5}$ impacts at these monitors when a large contribution from the Rough Fire is predicted. However, there were days at Visalia when the model underpredicted $PM_{2.5}$ impacts toward the beginning of early September. These overpredictions may be related to $PM_{2.5}$ emissions, physical treatment of the plume (evaporation and condensation processes), transport, grid resolution, or some combination of these factors. The large, estimated population exposures of the Rough Fire are most likely overstated during the early September period of high modeled fire impacts in the Central Valley of California.



µg/m³ = micrograms per cubic meter; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm.

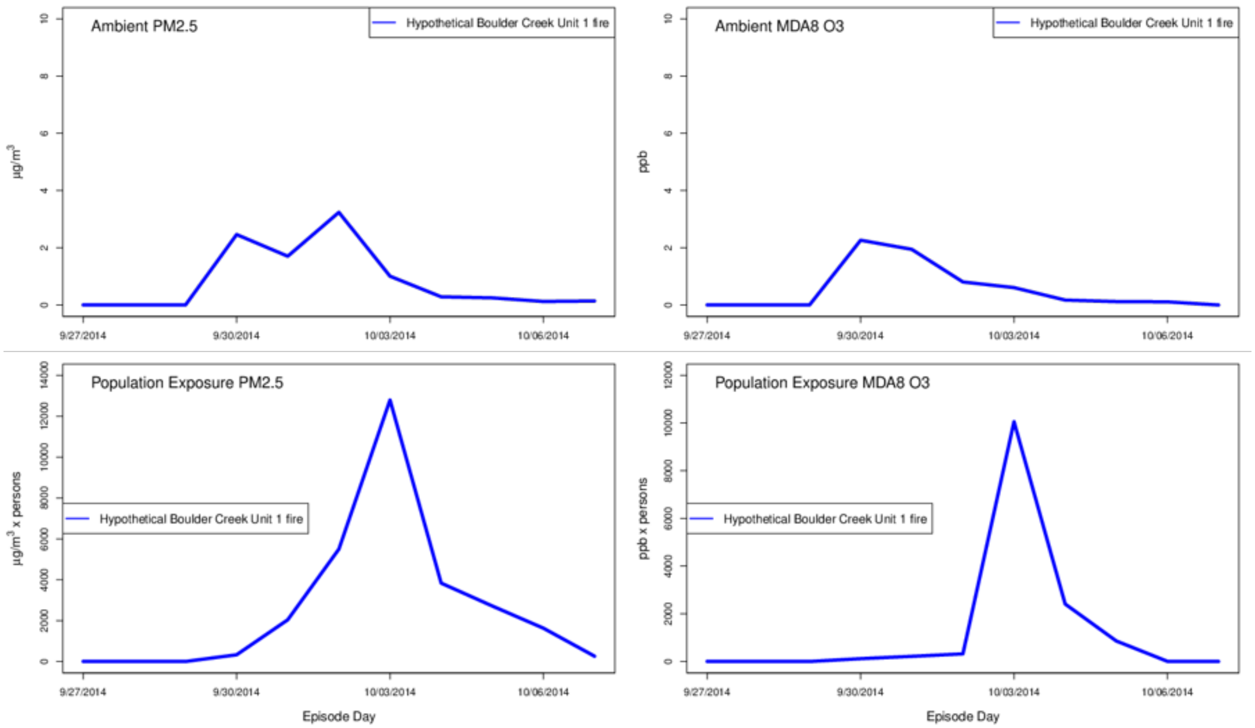
Note: Model predictions are shaded by the percent contribution from the actual Rough Fire.

Figure 7-18 Daily average $PM_{2.5}$ observations and model predictions at monitors in the Central Valley of California for August and September 2015.

Some of the model overprediction at monitors that were affected by smoke may be related to the model resolution not capturing orographically influenced wind flows. Here, organic carbon was treated as nonvolatile in the model. It is also possible that some amount of the primarily emitted organic aerosol might evaporate resulting in smaller downwind surface concentrations. However, recent research suggests generally equivalent aerosol mass after evaporated organics recondense in the smoke plume (Palm et al., 2020). This treatment would result in model predictions closer to measurements as fire impact monitors were often overpredicted (Figure 7-18).

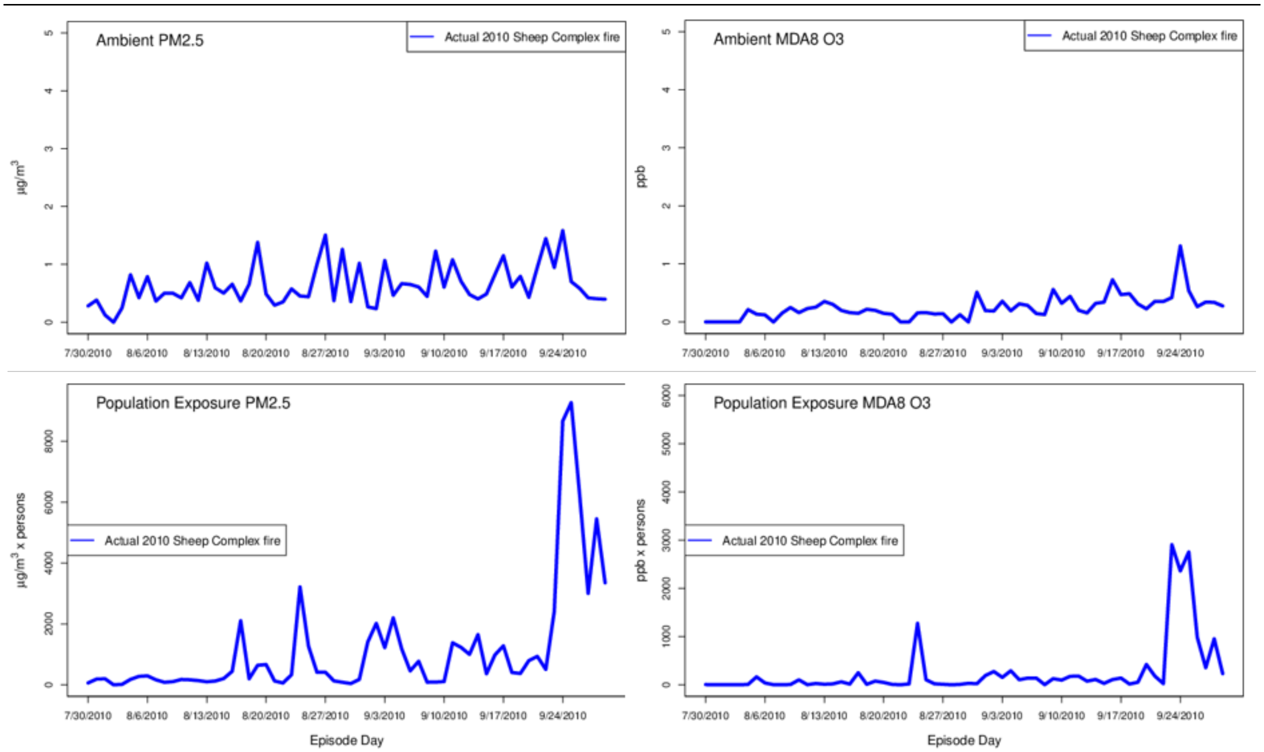
Ambient impacts of the hypothetical Boulder Creek Unit 1 Prescribed Fire (Figure 7-19) are notably smaller on the last 2 days than the first 3 days. Aggregate population exposures are high on 1 day toward the end of the prescribed fire when winds blew smoke toward the Central Valley of California. It is possible that the grid resolution used in this study may exaggerate estimates of population exposure because terrain-influenced meteorology may not be well resolved with 12-km-sized grid cells for this particular fire. The 12-km-sized grid cell resolution was chosen for the Rough-Fire-related scenarios to capture potential continental scale impacts at the expense of capturing near-fire orographic effects. Although daily air quality impacts from the Boulder Creek Unit 1 Prescribed Fire are similar in magnitude to some days of the Rough Fire, the estimates of population exposure are much smaller. These smaller exposures can be attributed to the Boulder Creek Prescribed Fire occurring over a smaller number of days compared to the Rough Fire and the meteorology not being conducive to transporting smoke to large population areas in central California.

Daily air quality impacts of the actual Sheep Complex Fire in 2010 (Figure 7-20) are fairly steady with respect to ambient concentrations and aggregate population exposure. A short period of high PM and O_3 impacts in populated areas was evident at the end of the fire in late September when the model predicted winds transporting smoke to more populated areas of the Central Valley in California. The daily ambient concentrations of the Sheep Complex Fire tend to be lower than the Rough Fire, and aggregate population exposures are much lower than for the Rough Fire. This is attributed to the smaller amount of biomass burned on a given day during the Sheep Complex Fire compared with the Rough Fire.



µg/m[3] = micrograms per cubic meter; MDA8 = maximum daily 8-hour average; O_3 = ozone; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm; ppb = parts per billion.

Figure 7-19 Daily average ambient (top row) $PM_{2.5}$ (left) and maximum daily 8-hour average (MDA8) ozone (O_3; right) impacts and aggregate population exposure (bottom row) from the hypothetical Boulder Creek Unit 1 Prescribed Fire.



µg/m³ = micrograms per cubic meter; MDA8 = maximum daily 8-hour average; O_3 = ozone; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm; ppb = parts per billion.

Figure 7-20 Daily average ambient (top row) $PM_{2.5}$ (left) and maximum daily 8-hour average (MDA8) ozone (O_3; right) concentrations and estimates of aggregate population exposure (bottom row) from the 2010 Sheep Complex Fire.

7.4 LIMITATIONS, IMPLICATIONS, AND RECOMMENDATIONS

Because the air quality impacts of these wildfire and prescribed fire scenarios occur over different time scales, the aggregation of impacts is presented later in this report in the section covering human health effects (see Chapter 8) with a synthesis of the results of the air quality modeling and health impact analyses in Chapter 9. A summary of highlights from the air quality modeling of the case study fires follows:

- Surface fuel load characterization is an important component of modeling air quality impacts associated with wildfires and prescribed fires.
- Outputs from two established fuel load characterization models, FCCS and VELMA, were merged and fed into the BlueSky Pipeline to simulate air quality impacts associated with wildfire and prescribed fire simulations for the TC6 and Rough Fire case studies.

- Whereas FCCS excels at providing estimates of management-sensitive surface and understory fuel types and loads, VELMA excels at characterizing overstory/canopy fuel loads through its use of linked forest inventory and satellite-based (LEMMA) data. The combined use of FCCS and VELMA for this purpose plays to the strengths of both models to better characterize fine-scale horizontal and vertical distributions of fuelbed types and loadings.
- A photochemical grid model was applied to estimate $PM_{2.5}$ and O_3 impacts from an actual wildfire in Oregon and California.
- A photochemical model was also used to estimate how $PM_{2.5}$ and O_3 impacts change for hypothetical smaller and larger realizations of the actual fires.

In considering the assumptions and approach used in the air quality modeling for the case studies presented in this report, it is necessary to also consider the limitations of these analyses to ensure the results are interpreted in the proper context. The prescribed fire impacts presented here represent a small subset of meteorological conditions, fuel loadings, and timing choices and may not be reflective of potential impacts on air quality in other areas or under different conditions. Further, the wildfire impacts shown here will vary based on different types of meteorological patterns influencing transport of smoke and formation of O_3 in the plume. The expected impact of the Boulder Creek Unit 1 Prescribed Fire on the progression of the Rough Fire is considered a "best-case scenario" and would likely require additional land management to reduce fuels in the region to a level needed to stop the progression of the Rough Fire further downslope as hypothesized here.

Other regions of the U.S. with a long history of prescribed fire such as the southeast U.S. and central plains (Kansas) provide some additional context about when choices made about prescribed fire scale can positively and negatively affect compliance with air quality standards and population exposure. For example, despite widespread prescribed fire activity in the southeastern U.S., there are currently no areas in the Southeast that are not in compliance with the PM or O_3 National Ambient Air Quality Standard (NAAQS). This widespread regional compliance with existing NAAQS across the Southeast suggests that carefully chosen timing of prescribed fire coupled with anthropogenic control programs can provide an opportunity for meeting land management goals without compromising public health. However, when prescribed burning activity is concentrated into a small window of time, which is typical, for example, of fires in the Flint Hills region of central Kansas, the enormous amount of fuel being burned on a few days has led to downwind monitors with O_3 and $PM_{2.5}$ sometimes exceeding the level of the NAAQS (Baker et al., 2019).

One challenge related to scale is understanding how the case study information provided in this report would translate to larger fires (size, duration) or larger regions where many fires would be on the landscape. The case studies within this assessment are somewhat limited in considering trade-offs over time because land management techniques would be conducted over multiple years to meet historical fire return interval goals while these case studies are episodic. Further, information about how many acres/total fuel need to be burned, in addition to the time interval between burns, is necessary to place the information here into a broader context of land management and air quality impacts. Additionally, future studies should attempt to include emissions related to fire suppression activity and model near-fire impacts using a horizontal grid resolution that would best capture complex terrain impacts on wind patterns.

Although the interactions between prescribed burns and wildfire characteristics is an active area of research (Hunter and Robles, 2020), more information is needed to understand and apply these dynamics quantitatively in air quality models, especially at the regional and national scales. The lack of a generalizable, mechanistic understanding of the influence of prescribed burning and other land treatments on wildfire activity (and consequently on air pollution due to wildfires) remains a major source of uncertainty when projecting future changes in fire-related air quality impacts, especially in areas where prescribed burning is a common practice.

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CHAPTER 8 ESTIMATED PUBLIC HEALTH IMPACTS OF SMOKE FROM CASE STUDY FIRES

8.1 INTRODUCTION

A main goal of this assessment is to provide a quantitative comparison of the estimated health impacts and associated economic values attributed to smoke from wildland fire (i.e., wildfire and prescribed fire) under different fire management strategies by focusing on two case study fires: the Timber Crater 6 (TC6) Fire and the Rough Fire. Chapter 6 of this assessment described in detail the health effects of wildfire smoke while Chapter 7 defined the air quality impacts of each case study fire and defined hypothetical scenarios meant to reflect different fire management strategies. Collectively, these two chapters provide key inputs to the process of quantitatively estimating the health impacts of wildland fire smoke. This chapter uses information presented in previous chapters to conduct analyses using U.S. Environmental Protection Agency's (U.S. EPA's) Environmental Benefits Mapping and Analysis Program—Community Edition (BenMAP-CE). The results of these analyses provides additional insight on the overall public health impacts of wildland fire smoke and shows how impacts can vary depending on the fire management strategy employed. The approach used within this assessment builds upon those found elsewhere in the literature that have also used the BenMAP-CE tool (Fann et al., 2018; Sacks et al., 2018).

8.2 BENEFITS MAPPING AND ANALYSIS PROGRAM—COMMUNITY EDITION (BENMAP–CE) ANALYSIS

BenMAP-CE quantifies the number and economic value of air pollution-related premature deaths and illnesses (Sacks et al., 2018). The program draws upon a library of preinstalled and user-imported input parameters (Table 8-1) to systematize the procedure for calculating the estimated health impact and then valuing the resulting counts of adverse effects. The sections below describe the steps to configuring and running BenMAP-CE to estimate the number, and corresponding economic impact, of wildland fire-related particulate matter (PM) with a nominal mean aerodynamic diameter less than or equal to 2.5 µm ($PM_{2.5}$) and ozone-attributable effects.

Table 8-1 Key data inputs for Benefits Mapping and Analysis Program—Community Edition (BenMAP–CE) used to estimate health impacts for the case studies.

Data Input	Source
Air quality data	Modeled $PM_{2.5}$ and ozone concentrations from each case study[a]
Population counts	U.S. census data allocated to air quality model grid cells, stratified by race, sex, age, and ethnicity and projected to the Year 2021
Risk coefficients	Concentration-response relationships from U.S-based air pollution epidemiologic studies examining $PM_{2.5}$, ozone, and wildfire-specific $PM_{2.5}$[b]
Baseline rates of death and disease	Centers for Disease Control and Prevention provided death rates, and the Healthcare Cost and Utilization Project (HCUP) provided hospital visit rates for all other areas

$PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm.

[a]For more information see Chapter 5.

[b]For more information on epidemiologic studies examining wildfire-specific $PM_{2.5}$ see Chapter 6.

8.2.1 HEALTH IMPACT FUNCTION

This analysis estimates the number of wildfire and prescribed fire-attributable premature deaths and illnesses associated with the TC6 Fire and Rough Fire case studies using a health impact function. Equation 8-1 details the approach for calculating $PM_{2.5}$-attributable premature deaths. The approach for quantifying PM-attributable morbidity impacts and ozone-related mortality and morbidity impacts is identical except for the ages for which the function is calculated, as detailed below. Counts of $PM_{2.5}$-attributable total deaths (y_{ij}) are calculated for period i (i = 2021) among individuals of all ages (0−99) (a) in each county j (j = 1,…,J where J is the total number of counties) as:

$$y_{ij} = \sum a \; y_{ija}$$

$$y_{ija} = mo_{ija} \times \left(e^{\beta \; cij-1}\right) \times P_{ija,}$$

Equation 8-1

where mo_{ija} is the daily baseline all-cause mortality rate for individuals aged a = 0−99 in county j in Year i stratified in 10-year age groups, β is the risk coefficient for all-cause mortality for adults associated with $PM_{2.5}$ exposure, C_{ij} is annual mean $PM_{2.5}$ concentration in county j in Year i, and P_{ija} is the number of residents aged a = 0−99 in county j in Year i stratified into 5-year age groups. When calculating impacts, the program assigns the 10-year stratified death rate to the corresponding 5-year stratified population bin. The health impact function used to calculate all other impacts is identical to Equation 8-1, except for the effect coefficient. The program performs a Monte Carlo analysis by randomly sampling 5,000 times from a distribution constructed from the standard error reported for each study; the resulting distribution is then used to report 95% confidence intervals.

The function above is calculated using BenMAP-CE (v1.5.5.1), a tool that contains the baseline incidence rates, population counts, and health impact functions needed to quantify counts of $PM_{2.5}$- and ozone-attributable deaths and respiratory hospital admissions (U.S. EPA, 2019; Sacks et al., 2018). This approach to quantifying air pollution health impacts, and the adverse effects of wildland fires in particular, has been used within the peer-reviewed literature (Fann et al., 2019; Fann et al., 2018; Berman et al., 2012). The following sections describe the specification of each input parameter within BenMAP-CE for the purposes of the analyses conducted within this assessment.

8.2.2 AIR QUALITY MODELING

The emissions inputs and photochemical modeling simulations performed to predict the $PM_{2.5}$ and ozone concentrations attributable to each case study fire, prescribed fire activity in each location, and defined hypothetical scenarios are detailed in Chapter 5. As noted in Chapter 1, for each hypothetical scenario, wildfire-specific air quality impacts (the delta used to estimate the change in health impacts) is calculated using a baseline of no case study fire to estimate the burden attributed to the actual fire, prescribed fires, and hypothetical scenarios for each case study. BenMAP-CE used the model-predicted daily mean $PM_{2.5}$ and model-predicted daily 8-hour max ozone concentration to quantify health impacts for the following actual fire and hypothetical scenarios for each case study:

<u>TC6 Case Study</u>

- Actual TC6 Fire.
- Scenario 1 (small): A smaller hypothetical TC6 Fire in a heavily managed area (e.g., most prescribed fire activity). This scenario would equate to a wildfire with less fuel consumption, a smaller fire perimeter, and less daily emissions.
- Scenario 2a (large): A larger hypothetical TC6 Fire, but not the “worst-case” scenario, with no land management. This scenario would equate to a wildfire with more fuel consumption, a larger fire perimeter, and more daily emissions.
- Scenario 2b (largest): A much larger, hypothetical “worst-case” modeled scenario TC6 Fire with no land management (i.e., no prescribed fire). This scenario would equate to a wildfire with the most fuel consumption, largest fire perimeter, and largest daily emissions.
- Prescribed fires: Three prescribed fires that occurred in the past and one prescribed fire that occurred in 2019, all modeled to occur on the same days in September 2019 that fit prescription conditions.

<u>Rough Fire Case Study</u>

- Actual Rough Fire.

- Scenario 1 (small): A small hypothetical Rough Fire that examines the combined impact of the proposed Boulder Creek Prescribed Fire and the Sheep Complex Fire on reducing the spread and air quality impacts of the Rough Fire.
- Scenario 2 (large): A large hypothetical Rough Fire that allows for the fire perimeter of the Rough Fire to progress into the area of the Sheep Complex Fire as though both the Boulder Creek Prescribed Fire and Sheep Complex Fire did not occur.
- Boulder Creek Prescribed Fire: A proposed prescribed fire that was planned for, but did not occur in the fall of 2013.
- Sheep Complex Fire: A wildfire that occurred in 2010 from a lightning strike and because of wet fuel conditions was effectively managed to achieve the same objectives as a prescribed fire.

8.2.3 EFFECT COEFFICIENTS

This analysis quantifies an array of adverse health effects attributable to $PM_{2.5}$ and ozone exposures, including premature death and morbidity. For the main analysis, the chosen studies examine the health effects associated with ambient exposures to $PM_{2.5}$ and ozone and have been used in recent U.S. EPA benefits analyses as detailed in Section 6.2.3. U.S. EPA recently published a technical support document that provides a detailed description of the Agency's systematic evaluation of the epidemiologic literature and the concentration-response (C-R) relationships used to develop health impact functions (U.S. EPA, 2021). In summary for $PM_{2.5}$, analyses focus on the following outcomes: short-term $PM_{2.5}$ exposure and mortality, all ages (Zanobetti and Schwartz, 2009); long-term $PM_{2.5}$ exposure and mortality, ages 30−99 years (Turner et al., 2016); respiratory-related emergency department (ED) visits, all ages (Krall et al., 2016); cardiovascular-related ED visits, all ages (Ostro et al., 2016); respiratory-related hospital admissions, ages 0−18 years (Ostro et al., 2009); and cardiovascular-related hospital admissions, ages 65 years and over (Bell et al., 2015). For ozone, analyses focus on short-term ozone exposure and respiratory mortality, all ages (Katsouyanni et al., 2009); long-term ozone exposure and respiratory mortality, ages 30−99 years (Turner et al., 2016); respiratory-related ED visits, all ages (Barry et al., 2019); and respiratory-related hospital admissions, ages 65 years and over (Katsouyanni et al., 2009).

The analysis quantifies the same morbidity impacts for each case study scenario. However, because the length of the actual TC6 and Rough fires varied, the analysis quantifies mortality impacts differently for each case study. Because the TC6 Fire only lasted a few days, mortality impacts are quantified using a concentration-response parameter from an epidemiologic study of short-term (i.e., day-to-day) changes in $PM_{2.5}$. By contrast, the Rough Fire lasted multiple months, and thus the impacts are more similar to those observed in a long-term exposure mortality study. For this reason, we quantify mortality impacts using a long-term $PM_{2.5}$ exposure function. Mortality impacts due to short-term $PM_{2.5}$ exposure are not quantified in the Rough Fire case study analyses to prevent the double counting of mortality impacts.

Whereas the main analyses rely on health impact functions derived from epidemiologic studies of ambient $PM_{2.5}$ exposures, the sensitivity analysis examined whether estimated health impacts differed when using health impact functions derived from epidemiologic studies that specifically examined wildfire smoke exposure (i.e., wildfire-specific $PM_{2.5}$). In the sensitivity analysis, only respiratory and cardiovascular outcomes are quantified because among the epidemiologic studies evaluated in Chapter 6 (see Section 6.2.2), only these studies used an exposure indicator of wildfire $PM_{2.5}$ and were suitable for use within BenMAP-CE (i.e., were conducted in locations similar to the case studies and represented health outcomes with available incidence data). Of the available respiratory-related, ED-visits studies that used wildfire $PM_{2.5}$ as the exposure indicator, none examined all respiratory-related ED visits; as a result, the sensitivity analysis quantified asthma ED visits using a risk coefficient from a study conducted by Reid et al. (2019) in northern California. With respect to hospital admissions, respiratory-related hospital admissions were quantified using a risk coefficient from a study conducted by Gan et al. (2017) in Washington state, and cardiovascular-related hospital admissions were quantified using a risk coefficient from a study focusing on a wildfire event in southern California conducted by Delfino et al. (2009).

8.2.4 BASELINE INCIDENCE AND PREVALENCE DATA

The epidemiologic studies noted above report estimates of risk (i.e., effect coefficients or β coefficients) that are expressed as being relative to a baseline rate. In this analysis, these effect coefficients were used to quantify cases of ED visits, hospital admissions and premature deaths, and thus baseline rates of all-cause mortality, ED visits, and hospital admissions were used in estimating these health impacts. County-level, age-stratified, all-cause death rates were obtained from the Centers for Disease Control Wide-ranging ONline Data for Epidemiologic Research (WONDER) database (CDC, 2016) for the Year 2010, while ED visit and hospital visit rates were obtained from the Healthcare Cost and Utilization Project (HCUP), which consists of a mixture of county, state, and regional rates.

8.2.5 ASSIGNING $PM_{2.5}$ CONCENTRATIONS TO THE POPULATION

Changes in population-level exposure are quantified by assigning the predicted $PM_{2.5}$ concentrations to the U.S. census-reported population in each 4-km by 4-km model grid cell for the TC6 Fire case study and 12-km by 12-km grid cell in the Rough Fire case study (see Chapter 5 for a detailed description of the air quality modeling simulations). As a first step, the PopGrid population preprocessing tool was used to assign U.S. census-reported population counts at the census block level to each air quality model grid cell. These population counts were stratified by age, sex, race, and ethnicity. The census-reported population counts for the Year 2010 were used and then counts were projected to the Year 2020 using forecast population from Woods & Poole (2016).

To calculate wildland fire $PM_{2.5}$ concentrations, concentrations were weighted to the size of the population exposed to wildland fire $PM_{2.5}$ concentrations for all counties combined (C_i) in Year *i* as

$$C_i = \frac{\sum_j^i C_{ij} \times P_{ij}}{P_i}$$

Equation 8-2

where C_{ij} is the wildfire-attributable annual mean $PM_{2.5}$ concentration in county *j* in Year *i*, P_{ij} is the population in county *j* in Year *i*, and P_i is the total population over all counties combined in Year *i*.

8.2.6 ECONOMIC ANALYSIS

The value of avoided premature deaths was estimated using a Value of Statistical Life (VSL) recommended by the U.S. EPA's *Guidelines for Preparing Economic Analyses* (U.S. EPA, 2014). A VSL is an estimate of society's willingness to pay to reduce the risk of premature death. Following U.S. EPA guidelines, this value was indexed to the inflation and income year of the analysis and does not vary by age. Using a 2015 inflation year and assuming 2020 income levels, a VSL of $9.5 million (M) was used. Avoided PM-attributable deaths are assumed to occur over a 20-year period and are sometimes presented as values discounted over this time span using a discount rate of 3 or 7%. As compared to the value of undiscounted PM benefits, the discounted PM benefits would be approximately 9 to 17% lower.

To value changes in respiratory hospital admissions, a cost of illness estimate was used, which is consistent with the approached used by the U.S. EPA in its Regulatory Impact Analysis for the $PM_{2.5}$ National Ambient Air Quality Standards (U.S. EPA, 2013). This value of $36,000 reflects the direct medical costs associated with the hospital visit as well as lost earnings. Following this same approach, we estimate the value of cardiovascular hospital admissions to fall between $41,000 and $42,000 depending on the age of onset. Finally, we quantify the value of emergency department visits using a simple average of two cost-of-illness values reported by Smith et al. (1997) and Stanford et al. (1999), which produces a value of $430. Although not presented here, it would be possible to calculate a present value of these impacts over a multiyear time horizon. For example, the value of the sum of mortality and morbidity impacts would decline by approximately a quarter by Year 10.

8.3 RESULTS FROM CASE STUDY FIRE ANALYSES

The sections present the estimated health impacts and corresponding economic values from the BenMAP-CE analyses for each of the actual fires, hypothetical scenarios, prescribed fires, and wildfire that yielded positive resource benefits for each case study. The main results presented in Section 8.3.1 are based on risk coefficients from epidemiologic studies used by U.S. EPA in previous benefits analyses as noted above; while Section 8.3.2 presents results from the sensitivity analyses using risk coefficients from studies examining wildfire-specific $PM_{2.5}$ and alternative epidemiologic studies examining ambient ozone exposure. Lastly, building off the discussion presented in Chapter 6 (see Section 6.3), Section 8.3.3 estimates the potential reduction in health impacts presented that could be achieved by implementing various actions or interventions to reduce or mitigate wildland fire smoke exposure.

8.3.1 MAIN RESULTS

The estimated number and value of wildfire-related health impacts varies across the scenarios and the pollutant assessed. $PM_{2.5}$-attributable effects (Table 8-2) are consistently larger than those quantified for ozone (Table 8-3). The estimated number of premature deaths, ED visits, and hospital admissions are larger for the Rough Fire scenarios than for the TC6 Fire scenarios; this can be attributed to differences in the magnitude of the fires, the duration of each fire, and the population density around each fire. For the TC6 Fire scenarios, fractional counts of air pollution-attributable effects are presented to illustrate the small, but meaningful, differences in impacts among the scenarios.

The dollar value of fires for the TC6 Fire case study is as large as $100 M while the value of the Rough Fire case study is as large as $3 billion (Table 8-4). These values represent the sum of the medical costs and productivity losses associated with the ED visits and hospital admissions and the value of air pollution-attributable deaths. This latter value is quantified using a VSL; it is not the value of any individual life.

Table 8-2 Estimated counts of $PM_{2.5}$ premature deaths and illnesses (95% confidence interval).

Case Study	Scenario	ED Visits: Respiratory	ED Visits: Cardiovascular	Hospital Admissions: Respiratory	Hospital Admissions: Cardiovascular	Mortality: Short Term	Mortality: Long Term
Timber Crater 6 (TC6)	Actual fire	0.2 (0.0 to 0.4)	0.1 (−0.0 to 0.2)	0.0 (0.0 to 0.0)	0.0 (0.0 to 0.1)	0.04 (0.01 to 0.08)	---
	Scenario 1 (small)	0.1 (0.0 to 0.2)	0.1 (−0.0 to 0.1)	0.0 (0.0 to 0.0)	0.0 (0.0 to 0.0)	0.03 (0.01 to 0.5)	---
	Scenario 2a (large)	0.8 (0.2 to 1.6)	0.4 (−0.1 to 0.9)	0.1 (0.0 to 0.1)	0.2 (0.1 to 0.2)	0.16 (0.01 to 0.32)	---
	Scenario 2b (largest)	1.2 (0.2 to 2.5)	0.6 (−0.2 to 1.3)	0.1 (0.1 to 0.2)	0.3 (0.2 to 0.3)	0.25 (0.01 to 0.49)	---
	Prescribed fires	0.04 (0.01 to 0.08)	0.02 (−0.01 to 0.05)	0.00 (0.00 to 0.01)	0.01 (0.01 to 0.01)	0.01 (0.001 to 0.02)	---
Rough Fire	Actual fire	47.3 (9.3 to 98.5)	19.7 (−7.6 to 46.0)	6.9 (3.0 to 10.7)	8.6 (6.2 to 10.9)	---	80.0 (53.6 to 105.4)
	Scenario 1 (small)	28.2 (5.5 to 58.7)	11.8 (−4.6 to 27.6)	4.2 (1.8 to 6.5)	5.0 (3.6 to 6.3)	---	48.1 (32.2 to 63.4)
	Scenario 2 (large)	49.8 (9.8 to 103.7)	20.7 (−8.0 to 48.4)	7.3 (3.2 to 11.2)	9.1 (6.6 to 11.5)	---	84.3 (56.5 to 111.1)
	Sheep Complex Fire	6.6 (1.3 to 13.7)	2.7 (−1.0 to 6.2)	0.9 (0.4 to 1.4)	0.9 (0.7 to 1.2)	---	10.1 (6.7 to 13.3)
	Boulder Creek Prescribed Fire	1.1 (0.2 to 2.4)	0.5 (−0.2 to 1.1)	0.2 (0.1 to 0.3)	0.2 (0.2 to 0.3)	---	1.9 (1.3 to 2.5)

ED = emergency department; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 μm; TC6 = Timber Crater 6.

Table 8-3 Estimated counts of ozone (O_3) premature deaths and illnesses (95% confidence interval).

Case Study	Scenario	Respiratory ED Visits	Respiratory Hospital Admissions	Mortality	
				Short Term	Long Term
Timber Crater 6 (TC6)	Actual fire	0.06 (0.02 to 0.1)	0.0 (−0.0 to 0.0)	0.0 (−0.0 to 0.0)	---
	Scenario 1 (small)	0.03 (0.01 to 0.06)	0.0 (0.0 to 0.0)	0.0 (0.0 to 0.0)	---
	Scenario 2a (large)	0.10 (0.03 to 0.2)	0.0 (−0.0 to 0.0)	0.0 (−0.0 to 0.0)	---
	Scenario 2b (largest)	0.15 (0.04 to 0.3)	0.0 (−0.0 to 0.0)	0.0 (−0.0 to 0.0)	---
	Prescribed fires	0.01 (0.0 to 0.02)	0.0 (0.0 to 0.0)	0.0 (0.0 to 0.0)	---
Rough Fire	Actual fire	4.6 (1.3 to 9.6)	0.2 (−0.05 to 0.4)	---	2.0 (1.4 to 2.6)
	Scenario 1 (small)	1.7 (0.5 to 3.6)	0.07 (−0.02 to 0.2)	---	0.9 (0.6 to 1.2)
	Scenario 2 (large)	2.0 (0.05 to 4)	0.06 (−0.02 to 0.1)	---	0.6 (0.4 to 0.8)
	Sheep Complex Fire	0.8 (0.2 to 1.6)	0.03 (−0.01 to −0.6)	---	0.3 (0.2 to 0.4)
	Boulder Creek Prescribed Fire	0.0 (0.0 to 0.0)	0.0 (0.0 to 0.0)	---	0.0 (0.0 to 0.0)

ED = emergency department; TC6 = Timber Crater 6.

Table 8-4 Estimated value of $PM_{2.5}$ and ozone-related premature deaths and illnesses (95% confidence interval; millions of 2015 dollars).

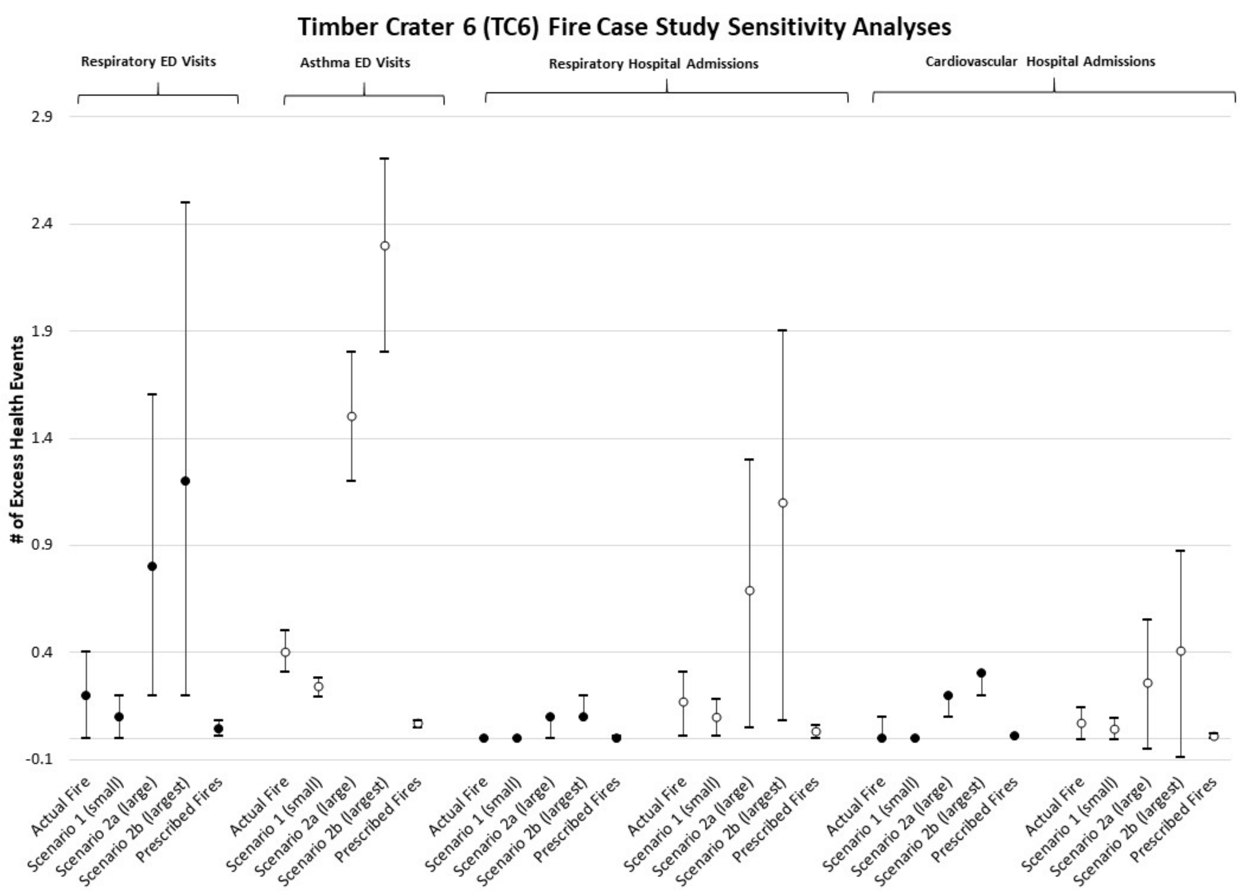
Case Study	Scenario	Sum of Value of Morbidity Impacts and Value of: Short-Term Exposure Mortality ($)	Sum of Value of Morbidity Impacts and Value of: Long-Term Exposure Mortality ($)
Timber Crater 6 (TC6)	Actual fire	18 (2 to 47)	---
	Scenario 1 (small)	10 (1 to 26)	---
	Scenario 2a (large)	66 (6 to 170)	---
	Scenario 2b (largest)	100 (9 to 270)	---
	Prescribed fires	4 (0 to 9)	---
Rough Fire	Actual fire	---	3,000 (260 to 7,900)
	Scenario 1 (small)	---	1,800 (160 to 4,700)
	Scenario 2 (large)	---	3,100 (270 to 8,300)
	Sheep Complex Fire	---	350 (20 to 960)
	Boulder Creek Prescribed Fire	---	60 (5 to 160)

$PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 μm; TC6 = Timber Crater 6.

8.3.2 SENSITIVITY ANALYSES

As noted above, the results presented within this section include estimates derived from health impact functions based on risk coefficients from epidemiologic studies that examined exposures to wildfire-specific $PM_{2.5}$ as a comparison to results from health impact functions based on ambient $PM_{2.5}$ exposures. Compared with the main analysis results, using the wildfire-specific $PM_{2.5}$ functions resulted in an increase in the estimated impacts for each case study (TC6: Figure 8-1; Rough Fire: Figure 8-2). This difference in estimated health impacts between studies examining ambient and wildfire-specific

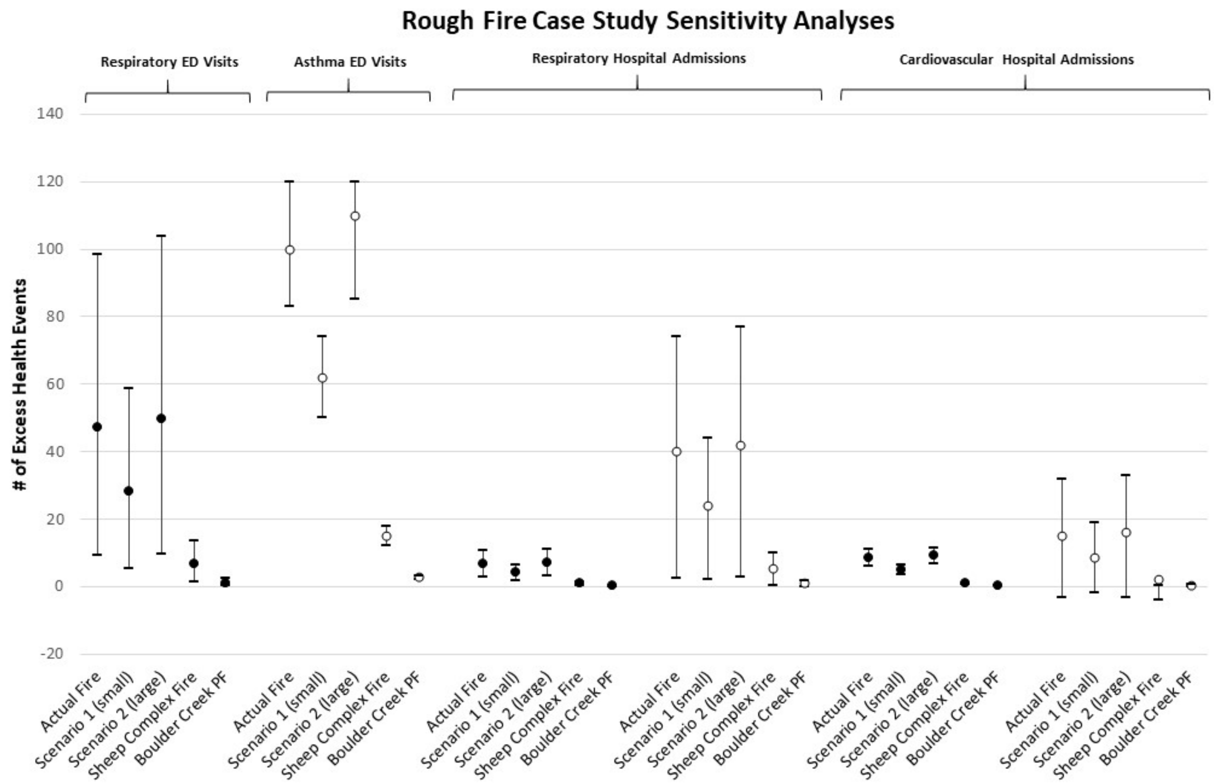
$PM_{2.5}$ exposures could be attributed to a steeper C-R relationship at the higher short-term $PM_{2.5}$ concentrations experienced during wildfire events or the behavior of individuals exposed to $PM_{2.5}$ during a wildfire event. However, additional research focused on examining the C-R relationship for wildfire smoke exposure is required to fully grasp the differences between the main analysis and sensitivity analysis results. The corresponding economic values from the sensitivity analyses are presented in Table 8-5, but these values are not directly comparable to the main analysis because the sensitivity analyses did not estimate premature deaths as noted in Section 8.2.3.



ED = emergency department; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm; TC6 = Timber Crater 6.

Black circles denote results from the main analysis. Open circles denote results from the sensitivity analysis. Lines denote the 95% confidence intervals for each estimate.

Figure 8-1 Estimated number of excess health events from sensitivity analyses using health impact functions based on ambient $PM_{2.5}$ exposures versus wildfire-specific $PM_{2.5}$ exposures for the Timber Crater 6 (TC6) Fire case study.



ED = emergency department; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 μm.

Black circles denote results from the main analysis. Open circles denote results from the sensitivity analysis. Lines denote the 95% confidence intervals for each estimate.

Figure 8-2 Estimated number of excess health events from sensitivity analyses using health impact functions based on ambient $PM_{2.5}$ exposures versus wildfire-specific $PM_{2.5}$ exposures for the Rough Fire case study.

Table 8-5 Estimated value of wildfire-specific $PM_{2.5}$ illnesses (95% confidence interval; 2015 dollars) from sensitivity analyses.

Case Study	Scenario	Sum of Value of Morbidity Impacts ($)
Timber Crater 6 (TC6)	Actual fire	8,600 (−76 to 17,000)
	Scenario 1 (small)	5,100 (−59 to 10,000)
	Scenario 2a (large)	35,000 (−220 to 69,000)
	Scenario 2b (largest)	54,000 (−500 to 110,000)
	Prescribed fires	2,000 (−14 to 3,900)
Rough Fire	Rough Fire (actual)	2,100,000 (−6,600 to 4,000,000)
	Rough Fire (Scenario 1)	1,200,000 (−1,400 to 2,400,000)
	Rough Fire (Scenario 2)	2,200,000 (−7,800 to 4,200,000)
	Sheep Complex Fire	280,000 (−37,000 to 550,000)
	Boulder Creek Prescribed Fire	58,000 (−1,300 to 130,000)

$PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm; TC6 = Timber Crater 6.

8.3.3 $PM_{2.5}$ EXPOSURE REDUCTION SENSITIVITY ANALYSIS

In assessing the health impacts and associated economic values attributed to smoke exposure from the actual fires in each of the case study areas as well as the hypothetical scenarios, the underlying assumption is that the population is exposed to the ambient $PM_{2.5}$ and ozone concentrations estimated through the air quality modeling process for each case study (see Chapter 5). However, as detailed in Chapter 6, it is possible to provide information to the public regarding actions that can be taken to reduce or mitigate smoke exposure from wildfires or prescribed fires, which could ultimately reduce the overall public health impact of smoke.

Using the average overall exposure reduction that could be achieved due to various exposure reduction actions, presented in Table 6-1, an illustrative example of the potential reduction in public health impacts that could be achieved are estimated for both case study fires, the corresponding hypothetical scenarios, and the prescribed fires (either actual or hypothetical) conducted in each location. The estimated overall reduction in total health impacts in Table 8-6 and Table 8-7 assume a linear relationship between population exposure concentrations and estimated health impacts such that the percent reduction in $PM_{2.5}$ exposure corresponds to an equivalent percent reduction in health impacts based on the main analysis results that used ambient $PM_{2.5}$ concentration-response functions. Future analyses would ideally use wildfire-specific functions, although such use would be complicated because the results of epidemiologic studies may be affected by the study population likely employing some unknown amount of actions to reduce its exposure. Also as noted in Section 6.3.3, the reduction in health impacts presented in Table 8-6 and Table 8-7 correspond to an average overall exposure reduction based on data from available studies and accounts for both the magnitude of the intervention and the likelihood that this intervention is employed. The exposure reductions presented do not account for differences in communication efforts between wildfires and prescribed fires or that different concentrations may affect the likelihood of taking action as well as factors specific to the case study areas (e.g., population demographics and housing stock) that can influence the corresponding exposure reduction for these actions. Additionally, estimating the reduction in potential public health impacts due to smoke exposure for each actual fire does not reflect a formal analysis of post-fire effectiveness of public health messaging by Air Resource Advisors (ARAs) deployed by the U.S. Forest Service, in combination with respective state and local air quality agencies, for either the TC6 or Rough fires. Instead, the analysis represents an estimation of the potential implications of exposure reduction actions on reducing the overall public health impact of smoke.

Table 8-6 Overall reduction in the estimated counts of adverse events attributed to $PM_{2.5}$ from wildfire smoke for the Timber Crater 6 (TC6) Fire case study.

Exposure Reduction Action (Overall Exposure Reduction; %)	Actual Fire	Hypothetical Scenarios 1 (Small)	2a (Large)	2b (Largest)	Prescribed Fires
Total health impacts[a]	0.34	0.23	1.66	2.45	0.08
Stayed inside (31.1)	−0.11	–	−0.52	−0.76	−0.02
Ran home HVAC system (24)	−0.08	−0.06	−0.40	−0.59	−0.02
Evacuated (20.5)	−0.07	−0.05	−0.34	−0.50	−0.02
Used air cleaner (14.1)	−0.05	–	−0.23	−0.35	−0.01

ED = emergency department; HVAC = heating, ventilation, and air conditioning; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 μm.

[a]Total number of health impacts represents the sum of ED visits, hospital admissions, and mortality detailed in Table 8-2; negative values in the table represent the estimated overall reduction in total impacts.

Corresponding 95% confidence intervals are not presented, because these results represent an illustrative example.

Table 8-7 Overall reduction in the estimated counts of adverse events attributed to $PM_{2.5}$ from wildfire smoke for the Rough Fire case study.

Exposure Reduction Action (Overall Exposure Reduction; %)	Actual Fire	Hypothetical Scenarios 1 (Small)	2 (Large)	Sheep Complex Fire	Boulder Creek Prescribed Fire
Total health impacts[a]	162.5	97.3	171.5	21.2	3.9
Stayed inside (31.1)	−50.5	−30.2	−53.3	−6.6	−1.2
Ran home HVAC system (24)	−39.0	−23.4	−41.2	−5.1	−0.94
Evacuated (20.5)	−33.3	−19.9	−35.2	−4.3	−0.80
Used air cleaner (14.1)	−22.9	−13.7	−24.2	−3.0	−0.55

ED = emergency department; HVAC = heating, ventilation, and air conditioning; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 μm.

[a]Total number of health impacts represents the sum of ED visits, hospital admissions, and mortality detailed in Table 8-2; negative values in the table represent the estimated overall reduction in total impacts.

Corresponding 95% confidence intervals are not presented, because these results represent an illustrative example.

8.4 SUMMARY

The analyses presented within this chapter estimate the potential public health impacts and associated economic values due to smoke exposure, focusing specifically on $PM_{2.5}$ and ozone, from wildland fire within the case study areas of the TC6 and Rough fires. Analyses for both case studies, which build off the assessment of the air quality impacts of each actual fire, hypothetical scenarios, and prescribed fires presented in Chapter 5, demonstrate that health impacts are dominated by exposure to $PM_{2.5}$ from wildland fire smoke.

The results of the case study analyses indicate that proximity to population centers and atmospheric conditions (e.g., wind patterns) influence the magnitude of health impacts due to smoke. Building off the air quality modeling analyses presented in Chapter 5 that depict differences in both $PM_{2.5}$ concentrations and population exposures, the corresponding BenMAP-CE analyses indicate that fire management strategies targeted to reduce the spread and overall size of wildfires, as depicted in the smaller hypothetical fires, can result in substantial differences in the health impacts and corresponding economic values when compared with the actual fires. Even though prescribed fires in both case study areas, and wildfires managed for resource benefits (i.e., Sheep Complex Fire), are shown to contribute to an estimated reduction in health impacts from wildfire smoke, these fires are not without risk and have their own health impacts, albeit smaller.

Sensitivity analyses that explore potential differences in estimated health impacts between health impact functions derived from epidemiologic studies of ambient $PM_{2.5}$ and wildfire-specific $PM_{2.5}$ provide evidence of potentially larger estimated impacts when using wildfire-specific $PM_{2.5}$ health impact functions. Additional analyses that provide an illustrative example of the potential implications of actions or interventions to reduce and mitigate wildland fire smoke exposure demonstrate the potential public health benefits of messaging campaigns to the public. However, for both sensitivity analyses, additional research is warranted to more fully assess the implications of using ambient and wildfire-specific $PM_{2.5}$ health impact functions and to provide a more representative estimation of the potential public health benefits of actions or interventions to reduce wildfire smoke exposure.

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CHAPTER 9 INTEGRATED SYNTHESIS

9.1 INTRODUCTION

The focus of this chapter is to summarize and synthesize the information presented in the previous chapters that directly informs the quantitative analyses of the air quality impacts and corresponding health impacts of smoke from wildland fire (i.e., wildfire and prescribed fire) under different fire management strategies. The chapter also provides ancillary information that allows for the overall results of the analyses to be put into the proper context.[9] Overall, this assessment demonstrates the successful application of a novel modeling approach to quantitatively estimate the differences in air quality and health impacts based on different fire management strategies for two case study fires.

In theory, an assessment of the air quality impacts and the corresponding human health impacts of prescribed fire compared with wildfire may seem relatively straightforward. However, such an assessment is layered with complexities in both the development of analyses and the interpretation of results because of numerous factors including spatial and temporal differences between prescribed fire and wildfire along with the overall management objectives of each (i.e., suppression objectives or resource objectives), which are dynamic and can change daily, or even hourly, depending on various factors (e.g., fire behavior, as detailed in Chapter 2 and Chapter 3). Although the analyses conducted in this report represent an incremental advancement in the overall understanding of the health implications of smoke from wildland fire on surrounding populations, the results are based on a modeling approach that required various assumptions and decisions based on expert judgment, particularly with respect to fire spread in the design of hypothetical scenarios for each case study.

The preceding chapters of this report are organized around characterizing the components that are important when examining the air quality impacts and corresponding health impacts of smoke from wildland fire under different fire management strategies. In estimating differences between the air quality impacts of prescribed fire and wildfire, this assessment takes a holistic approach of identifying all of the factors and effects (both positive and negative) that should be accounted for when examining different fire management strategies as reflected in the conceptual framework (Chapter 2; Figure 2-1, Figure 9-4 in this chapter). Part I, which includes Chapter 2 along with Chapter 3−Chapter 6, describes the current state of the science with respect to implementing this framework, with the goal of employing the best available science and data to estimate the air quality and health impacts due to smoke in Part II (i.e., Chapter 7 and Chapter 8). A fuller accounting of benefits and costs of fire management strategies, which is not the focus

[9] Within this assessment, the term “impacts” refers to the main quantitative results, which includes the estimated air pollutant concentrations from the air quality modeling and the number of health events and associated economic values calculated using U.S. Environmental Protection Agency’s (U.S. EPA’s) Environmental Benefits Mapping and Analysis Program—Community Edition (BenMAP-CE). The term “effects” is used to denote the other positive and negative consequences of wildland fire.

of this assessment, would quantitatively address the remaining components of the conceptual framework, including management costs, direct fire effects, and ecological effects.

Although the results of this assessment are informative in addressing the larger question of whether there are differences in the public health impacts of wildland fire smoke for different fire management strategies, it is also important to ensure the results are interpreted appropriately and to recognize that the effects characterized represent only a portion of the broader societal, human health, and ecological effects of wildland fire events. Therefore, subsequent sections of this chapter provide an overview of the results of the analyses; broadly assess the limitations and uncertainties surrounding the examination of the air quality and corresponding public health impacts of prescribed fire and wildfire; identify the limitations and gaps in knowledge and data that informed the implementation of the conceptual framework (Figure 9-4); highlight key insights from the case study analyses; and outline additional areas of research that could further characterize the impacts of smoke from wildland fire.

9.2 OVERVIEW OF RESULTS

The overall goals of the case study analyses are twofold: (1) develop a modeling framework to examine the air quality and health impacts of smoke from wildland fire under different fire management strategies and (2) demonstrate the application of the modeling framework for wildfires that encompass different spatial and temporal scales. Because the analyses conducted within this assessment focus on wildfires of different spatial extent that occurred in two different geographic locations (Oregon and California), the results are specific to the locations of the two case study fires and the land management practices used before either fire occurred. Therefore, the results of these analyses cannot be extrapolated to other geographic locations without considering the differences in land management practices (including history) and environmental variables (e.g., geography, vegetation, fire regime, climate, and weather).

For both case studies, the air quality modeling and subsequent health impact analyses using U.S. Environmental Protection Agency's (U.S. EPA's) Environmental Benefits Mapping and Analysis Program—Community Edition (BenMAP-CE) show that air quality impacts due to wildland fire smoke are dominated by changes in fine particulate matter (particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm [$PM_{2.5}$]) concentrations (see Section 7.3). Ozone is formed downwind of a smoke plume as a result of many of its precursors being emitted in wildland fire smoke (see Chapter 4 and Chapter 7). The magnitude of population-level health impacts depends on the intersection of smoke plumes that have elevated $PM_{2.5}$ and ozone concentrations over time with population density. Fires producing smoke plumes or elevated ozone concentrations downwind of a smoke plume that do not intersect with high population areas or last only a few days are less likely to have substantial health impacts as fires affecting larger populations for longer periods. This concept of duration of fire multiplied by population density in the area affected by the smoke plume is the main driver of the difference in results between the Timber Crater 6 (TC6) Fire and Rough Fire case studies, discussed in more detail below.

Both case study fires were selected because they occurred on federal land and were managed by multiple federal agencies. Additionally, the TC6 Fire was selected because it had extensive data on the land management practices employed, including prescribed fire activity within the area. This, in combination with the small size of the fire, allowed for a finer resolution analysis (i.e., at the 4-km scale). In comparison, the Rough Fire was selected to provide an examination of a larger fire, in terms of duration and size, but there was no actual prescribed fire activity in the area. However, with the Sheep Complex Fire yielding positive resource benefits, and detailed information available on the proposed Boulder Creek Prescribed Fire, it was possible to develop hypothetical scenarios for the Rough Fire case study that were consistent with those developed for the TC6 Fire case study (i.e., a smaller and larger fire based on different land management strategies).

9.2.1 TIMBER CRATER 6 (TC6) FIRE CASE STUDY

The analysis of the TC6 Fire case study focused on estimating the air quality and health impacts due to the actual TC6 Fire, as well as hypothetical TC6 Fire scenarios based on assumptions surrounding fire spread and fuel availability that were rooted in the detailed land management data for the area (see Section 7.1.3), resulting in the following hypothetical scenarios:

- Scenario 1 (small): A smaller hypothetical TC6 Fire in a heavily managed area (i.e., most prescribed fire activity). This scenario would equate to a wildfire with less fuel consumption, a smaller fire perimeter, and less daily emissions.
- Scenario 2a (large): A larger hypothetical TC6 Fire, but not the “worst-case” scenario with no land management. This scenario would equate to a wildfire with more fuel consumption, a larger fire perimeter, and more daily emissions.
- Scenario 2b (largest): A much larger, hypothetical “worst-case” modeled scenario TC6 Fire with no land management (i.e., no prescribed fire). This scenario would equate to a wildfire with the most fuel consumption, largest fire perimeter, and largest daily emissions.

Even with the detailed land management data available, in devising the hypothetical scenarios for this case study, expert judgment was used to determine the daily fire perimeters and the overall burn perimeter for each scenario, which was influenced by the prescribed fire history within the area.

One of the main differences between the two case studies is the availability of data on prescribed fire activity around the TC6 Fire. Although there was information on prescribed fire activity within the vicinity of the TC6 Fire that could have affected the spread of the fire, these fires occurred over many years, with one dating back to 1978 (see Section 7.1.4). Thus, to compare the prescribed fire smoke impacts with the actual TC6 Fire and hypothetical scenarios, all prescribed fire activity was modeled for the same month and year (i.e., September 2019). This approach was used because there were detailed data on the days in September 2019 that fit prescription requirements and for which a prescribed fire occurred. However, this strategy does not consider the rate of prescribed fire activity and ignores the episodic nature of prescribed fires compared with wildfires, which is one of the overarching challenges of an analysis devised to compare the air quality and health impacts of prescribed fire with wildfire (see Section 9.3.1).

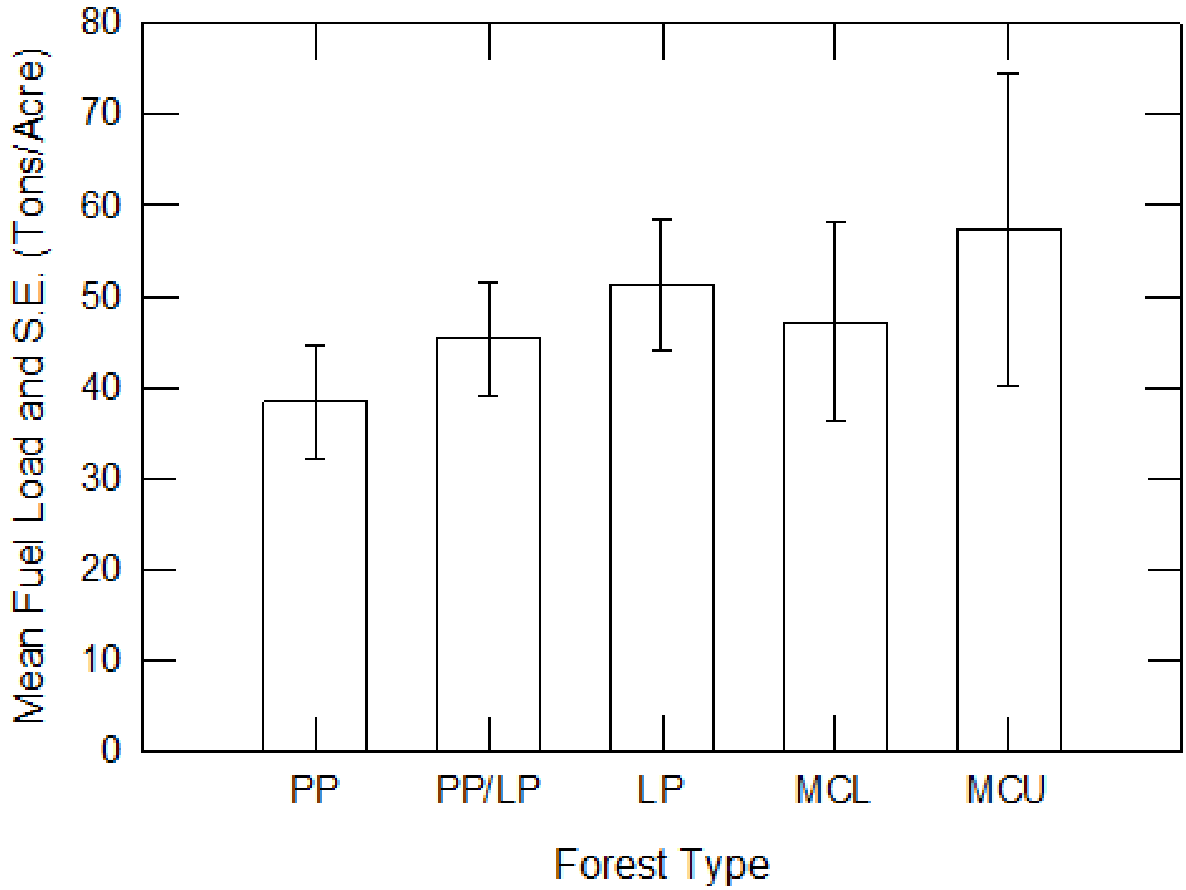
The air quality modeling estimates indicate that there are clear differences in the air quality impacts between the actual TC6 Fire and each of the hypothetical scenarios, with the larger fire hypothetical scenarios (Scenarios 2a and 2b) resulting in higher concentrations (specifically of $PM_{2.5}$) for a longer duration. This estimated difference is also consistent when comparing the actual TC6 Fire and hypothetical scenarios with the air quality impacts from the prescribed fires. The difference in the modeled air quality impacts between the prescribed fires and the actual TC6 Fire and hypothetical scenarios can be attributed to the short duration of each prescribed fire combined with the fact that these fires were scheduled on days that met specific criteria aimed at minimizing population exposure (e.g., meteorology conducive for ventilation and dilution of pollutants).

Although there are differences in air quality impacts across each of the scenarios examined, within the vicinity of the TC6 Fire population density is relatively small. As a result, the examination of aggregate population exposures which combines the influence of daily weather patterns, fire duration, and population proximity to fires, shows that the overall potential public health impacts due to smoke exposure would be smaller for a small fire, such as the TC6 Fire, compared to a larger fire (see Section 7.3.1; Figure 7-10 and Figure 7-11). This inference from the air quality modeling is reflected in the BenMAP-CE analysis for the actual TC6 Fire and the hypothetical scenarios where the overall estimated health impacts and corresponding economic values are small (see Table 8-2, Table 8-3, and Table 8-4). From a health impact perspective, the overall incidence of excess health events is <1 for most health outcomes for $PM_{2.5}$, and for all health outcomes for ozone across each fire type. However, when examining the economic value of these mortality and morbidity outcomes there is a more notable difference between the actual TC6 Fire (~$18 million [M]), prescribed fires (~$4 mil), and each hypothetical scenario (ranging from ~$10 for Scenario 1 to ~$100 M for Scenario 2b). This difference in estimated economic values reflects the high value placed on reductions in the risk of premature death. Even small changes in risk can have economic value because one statistical premature death is valued at $9.5 M.

Although the small smoke-related health impacts from the TC6 Fire can be attributed to the small population density within the case study area, the land management activities employed over time were instrumental in reducing the fuel available, the overall fire perimeter, and ultimately the air quality impacts. Untreated forests within the TC6 Fire case study area are characterized by high fuel loads (live and dead) that pose a significant challenge to fire managers. Combined with hot, dry summers and few natural barriers to fire spread, these spatially contiguous fuel loads create conditions ripe for large fire growth. Baseline surface fuel loads (dead and down biomass) in untreated stands vary along a productivity gradient, ranging from an average of 38 tons per acre in pure ponderosa pine to 46 tons per acre in mixed ponderosa pine/lodgepole pine forests (the most widespread type), and up to 56 tons per acre in the more productive upper elevation mixed conifer forest types (see Figure 9-1). Standing tree densities (live and snags) averaged 881 to 2,899 trees per hectare across the same gradient. These conditions were typical of that encountered during the rapid initial growth of the TC6 Fire where no fuel treatments had occurred. The extensive fuel treatment network employed in other parts of the area prevented these conditions from occurring across the entire TC6 Fire footprint.

This high contemporary fuel loading within the TC6 Fire case study area is an artifact of more than a century of ubiquitous fire exclusion (i.e., eliminating fires from the landscape through fire suppression) in the region beginning in the late 1800s. Prior to about 1890, fires were frequent across this landscape and resulted in limited broad-scale tree density and surface fuel accumulation. Hagmann et al. (2019) recently conducted a detailed fire history study in an area 30 km east of the TC6 Fire consisting of nearly identical terrain and forest composition. They found that years in which fires burned >20,000 hectares occurred every 9.5 years on average for the period 1700 to 1918, and that only 7 of the years had a fire that burned >40,000 hectares. These large, predominantly low-intensity fires were associated with drought years; more frequent but smaller fires occurred in interim periods. At a finer scale, fire frequency and intensity varied along gradients of productivity and surface fuel continuity ranging from 7 to 25 years. Most high-severity burning was restricted to pockets of dense lodgepole pine. This pattern resulted in relatively small patch mosaics of denser stands within a matrix of open, low-density stands. This same gradient occurs in the TC6 Fire study area and is consistent with a description of fire occurrence and fire severity mosaics by Agee (1981).

The cessation of frequent low-intensity fires across the greater Pumice Plateau Ecoregion in central Oregon began in the late 1800s but varied locally (Omernik and Griffith, 2014). Initially, extensive fire exclusion occurred indirectly as a result of changing land uses (such as heavy grazing and logging and development) and the displacement of Native American populations. Direct fire suppression activities continued with the onset of formal fire suppression activities in the early 1900s. Heyerdahl et al. (2014) and Merschel et al. (2018) documented sharp declines in fire occurrence in similar forests approximately 100 km north in the 1880s. Hagmann et al. (2019) also documented declines beginning in the late 1800s adjacent to the TC6 Fire area. There was a single large fire in 1918, but even this fire did not affect the TC6 Fire footprint.



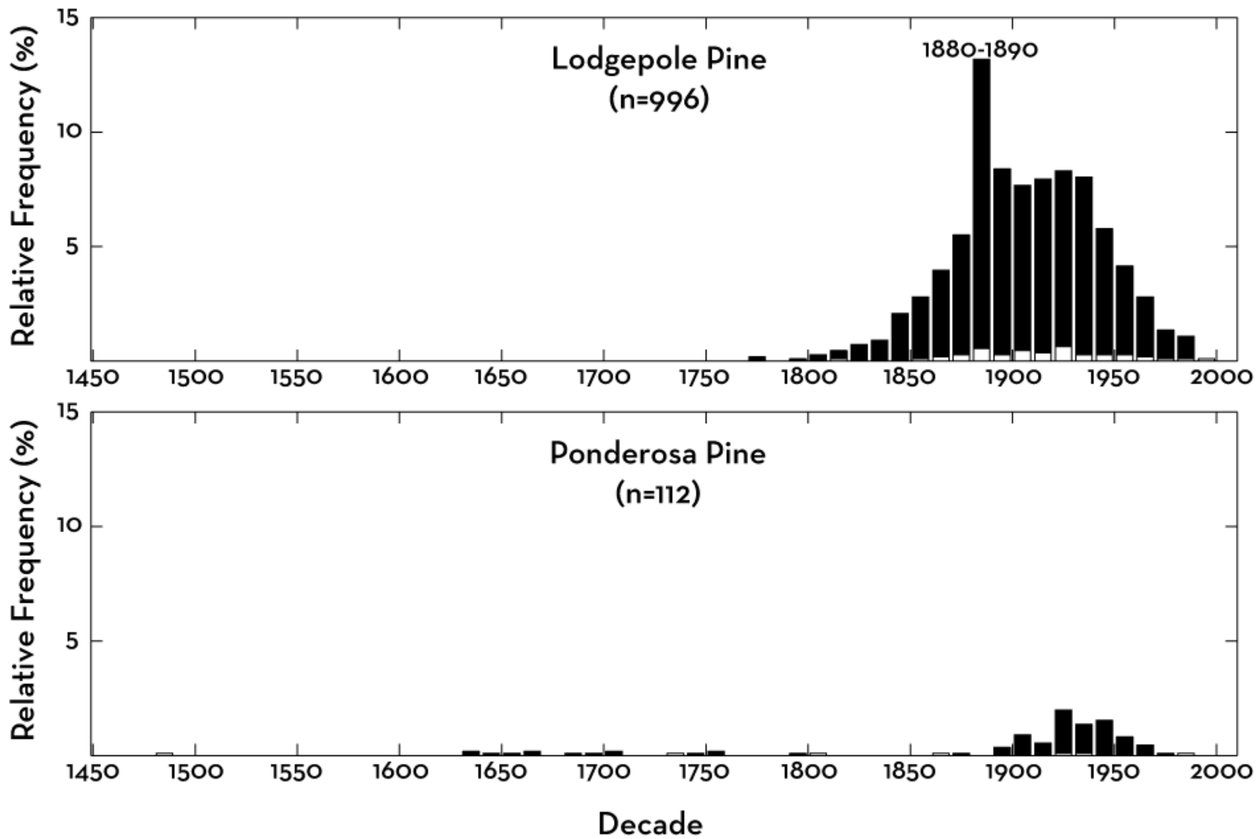
LP = lodgepole pine; MCL = lower mixed conifer; MCU = upper mixed conifer; PP = ponderosa pine; PP/LP = mixed ponderosa pine/lodgepole pine; S.E. = standard error.

Note: Forest types from left to right are: PP (n = 4 plots), PP/LP (n = 5), LP (n = 9), MCL (n = 13), MCU (n = 8).

Source: National Park Service Long-Term Monitoring Plots (Farris, 2017).

Figure 9-1 Surface fuel loading in untreated forests in the Timber Crater 6 (TC6) Fire study area in Crater Lake National Park.

The results of Heyerdahl et al. (2014), Merschel et al. (2018), and Hagmann et al. (2019) are consistent with local tree demography and recruitment data from Crater Lake National Park. In a 200-hectare study area burned by the TC6 Fire (along the U.S. Forest Service [USFS]/National Park Service [NPS] boundary), Kipfmueller (2014) documented a major pulse in tree recruitment in the 1880s, with high levels of recruitment continuing through the 1950s in the absence of fire (Figure 9-2). This recruitment pulse likely reflects a widespread fire exclusion signature and is consistent with the onset of major recruitment pulses elsewhere in Crater Lake National Park (Forrestel et al., 2017). High contemporary fuel loads in the study area are a direct legacy of this broad fire-exclusion recruitment cohort. In the absence of 20th century fires, this cohort created high tree densities across the formerly more heterogenous mosaic of productivity gradients. Many of these trees were converted to surface fuels following density-dependent thinning and periodic insect outbreaks in recent years. Moreover, continuous vertical fuel continuity from extensive ladder fuels create high crown fire initiation risk.



ha = hectare.

Note: White bars represent samples in which the pith date had to be estimated due to rot or other problems.

Source: Kipfmueller (2014).

Figure 9-2 Decadal-scale representation of the age structure of lodegpole pine (LP) and ponderosa pine (PP) aggregated for a 200-ha study area within the Timber Crater 6 (TC6) Fire perimeter.

Despite the high potential for major fires in the area, fire suppression was largely successful in the TC6 landscape because most ignitions were kept small (some of which were aided by the fire management strategies employed within the area). For example, in 2011 alone, there were seven lightning fires suppressed within 4 km of the TC6 Fire ignition. Thus, the only substantial burned acreage and reduction in fuel loads on the NPS side resulted from management-ignited prescribed burning and a lightning fire that yielded positive resource benefits (these efforts have been focused along the park boundary to facilitate future management of more lightning fires). On the USFS side, a combination of prescribed burns and mechanical fuel treatments have reduced fuel loads. Prescribed burning has reduced surface fuel loads by an average of 20% in low-productivity ponderosa pine to an average of 69% in lodgepole pine forests. Corresponding tree densities have been reduced by an average of 25% in ponderosa pine to 78% for lodgepole pine. The duration of treatments varies across a productivity gradient, but typically reach 75% of prefire levels within 15 years on productive sites. The fire management challenges and effects of fire suppression, fuel loading, and potential fire behavior in the TC6 Fire area are similar to other coniferous forest types in the western U.S.

9.2.2 ROUGH FIRE CASE STUDY

The Rough Fire was selected because it represented a much larger fire than the TC6 Fire in terms of both area burned (i.e., ~150,000 acres) and duration (i.e., lasting ~2 months), which directly influenced the amount of smoke produced and the potential for a larger aggregate population exposure. However, compared with the TC6 Fire, there were less data available regarding previous land management practices within the vicinity of the Rough Fire to inform the development of hypothetical scenarios. Consequently, the hypothetical scenarios devised for the Rough Fire are not based on the same type of land management strategies employed in the TC6 Fire case study. Specifically, for the Rough Fire, there was a reliance on a wildfire that burned at lower intensity and yielded positive resource benefits (i.e., Sheep Complex Fire) and a proposed prescribed fire that did not occur as planned (i.e., Boulder Creek Prescribed Fire). However, the use of the proposed Boulder Creek Prescribed Fire and the Sheep Complex Fire achieved the same function of being useful for devising Rough Fire hypothetical scenarios indicative of a smaller and larger Rough Fire, respectively, due to different land management strategies. The hypothetical scenarios for the Rough Fire consisted of the following:

- Scenario 1 (small): A small hypothetical Rough Fire that examines the combined impact of the Boulder Creek Prescribed Fire and the Sheep Complex Fire on reducing the spread and air quality impacts of the Rough Fire.
- Scenario 2 (large): A large hypothetical Rough Fire that allows for the fire perimeter of the Rough Fire to progress into the area of the Sheep Complex Fire as though both the Boulder Creek Prescribed Fire and Sheep Complex Fire did not occur.

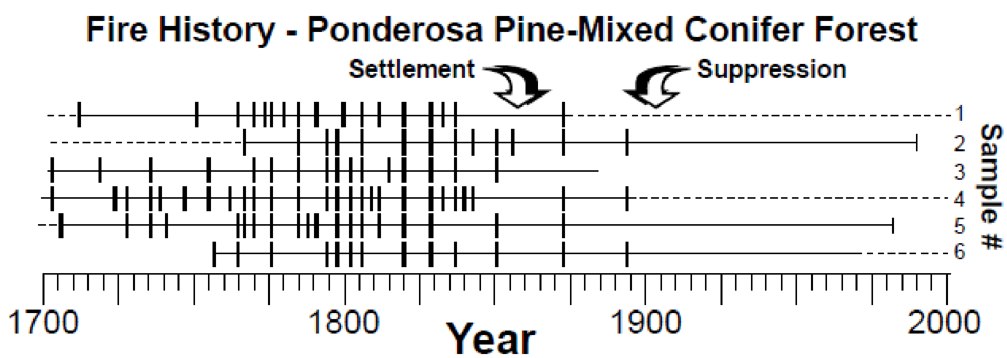
As with the TC6 Fire case study, when examining air quality impacts for the actual Rough Fire and each hypothetical scenario, overall aggregate population exposures are greatest for $PM_{2.5}$ (Figure 7-15 and Figure 7-16) even though ozone concentrations in this case study affect a larger geographic area. This difference can be attributed to ozone being produced only through secondary atmospheric reactions downwind from smoke events, whereas, $PM_{2.5}$ is directly emitted by fires, which represents the predominate downwind exposure. However, $PM_{2.5}$ can also be produced through secondary atmospheric reactions. For both $PM_{2.5}$ and ozone a similar temporal pattern of concentrations is observed between the actual Rough Fire and hypothetical scenarios until later weeks in the duration of each fire, when there was a substantial reduction in concentrations for Scenario 1 (small fire, Figure 7-17). Although there was not an actual prescribed fire in the vicinity of the Rough Fire, air quality analyses of the proposed Boulder Creek Prescribed Fire (Figure 7-19) and the Sheep Complex Fire (Figure 7-20) exhibit a shorter duration and smaller exposure to $PM_{2.5}$, respectively, compared with the actual Rough Fire and each hypothetical scenario.

The differences in the public health impacts between the actual fire, hypothetical scenarios, a prescribed fire (i.e., Boulder Creek Prescribed Fire), and a wildfire that yielded positive resource benefits (i.e., Sheep Complex Fire) are depicted in Table 8-2. The estimated health impacts of the actual Rough Fire, which reflect the occurrence of the Sheep Complex Fire, are relatively similar to hypothetical Scenario 2 (large fire), which assumes the Sheep Complex Fire did not occur. The corresponding economic value of the actual Rough Fire was estimated at ~$3,000 M and ~$3,100 M for Scenario 2 (large fire). The similarity between the actual Rough Fire and Scenario 2 (large fire) can be attributed to the Sheep Complex Fire not substantially affecting the overall spread and fire perimeter of the actual Rough Fire. However, the results of Scenario 1 (smaller fire) demonstrate the potential benefit that could occur, specifically the reduction in fire spread and perimeter by strategically planning the location of a prescribed fire. The modeling of the Boulder Creek Prescribed Fire shows that had that fire occurred on the outskirts of the Sheep Complex Fire perimeter, it could have prevented the spread of the Rough Fire and reduce air quality impacts, resulting in an approximate 40% reduction in health impacts (i.e., the combined number of premature deaths and illnesses) and in a smaller economic value (~$1,800 M) compared with the actual Rough Fire and Scenario 2. However, both the Sheep Complex Fire and the Boulder Creek Prescribed Fire scenarios did have detrimental effects on both air quality and health, equating to an estimated economic value of ~$350 M and ~$60 M, respectively, which is smaller than the estimated economic values for the actual Rough Fire and each hypothetical scenario.

In addition to the air quality and health impacts observed between the different hypothetical scenarios of the Rough Fire case study, it is also important to consider the affect of different land management strategies on the forest ecology around the case study area. Beyond the analogous examples in other parts of the U.S., the particular fire ecology and history of the dry forests of the Sierra Nevada Mountain offer more context for analyzing the Rough Fire area. The Sierra Nevada Mountain forests illuminate how the results of the Rough Fire case study might be used to further understand how to minimize air quality impacts from wildfire smoke, both in this area and in other dry forest regions. There is substantial fuel available in these large, highly productive, west-facing Sierra Nevada drainages that can be released into the air all at once, as witnessed during the Rough Fire, Rim Fire, and any number of megafires (i.e., fires with >100,000 acres burned).

Fire-adapted forest stands are characterized not only by having lower fuel loads, but fuels that are "packaged" into fire adapted-clumps (also known as resilient forest structure) with gaps in between those clumps, resulting in fire that burns more slowly across the land, rather than all at once. Therefore, it is important to consider the spatial configuration of forest stands as well as the amount of fuel available. At larger landscape scales, a mosaic of frequent, smaller, slower growing fires can contribute to reducing the number of megafires. Historically and prehistorically there is overwhelming evidence that the forests of the Sierra Nevada, including the area where the Rough Fire burned, experienced frequent burning. This was noted by Swetnam et al. (2000) and Caprio and Swetnam (1995) in studies of historic fire occurrence along elevation gradients using fire-scar data in the vicinity of the Rough Fire. In general, fire frequency decreased with increasing elevation. For the period 1700 to 1900 mean fire intervals (MFI) was found to range from approximately 4 to 5 years in ponderosa pine stands at the lowest elevations (1,510 m) to approximately 12 years in mixed conifer stands at higher elevations (2,180 m).

In addition to fire frequency, local fire-scar chronologies indicate that most fire years before the 20th century were characterized by relatively small, spatially clustered fire events that were even smaller than the Sheep Complex Fire and that over time there has been a dramatic decline in frequent, widespread fires at most sites (Figure 9-3). Widespread fire events also occurred periodically during severe droughts, as indicated by synchronous scarring across multiple sample sites (Swetnam et al., 2009). Although there is uncertainty about the size or extent of these large fire years, a major difference versus contemporary large fires is that they consisted predominantly of low-severity burning (Mallek et al., 2013).



Note: Reconstruction of past fire occurrence (tic marks) from fire-scarred trees at six sites in the mixed conifer zone from 1700−2000.

Source: Sequoia & Kings Canyon National Parks (2005), copyright permission pending.

Figure 9-3 **Decline in fire frequency in mixed conifer forest (from nearby Sequoia and Kings Canyon National Parks) starting around 1860.**

Impacts to air quality from fires that occurred before 1900 would have likely been similar in intensity, duration, and spatial extent to impacts from the modeled Boulder Creek Prescribed and/or Sheep Complex fires, rather than the Rough Fire. This is because such fires spread more slowly and the fuels over the area in which they burned were substantially less than those currently observed in areas where fuels have accumulated after 100 years of fire suppression (Stephens et al., 2018). Additionally, these smaller, frequent fires created a landscape-scale mosaic of fire footprints wherein fires were limited in their size by the footprints (and the removal of fuel within those footprints) of previous recent fires (Collins et al., 2009).

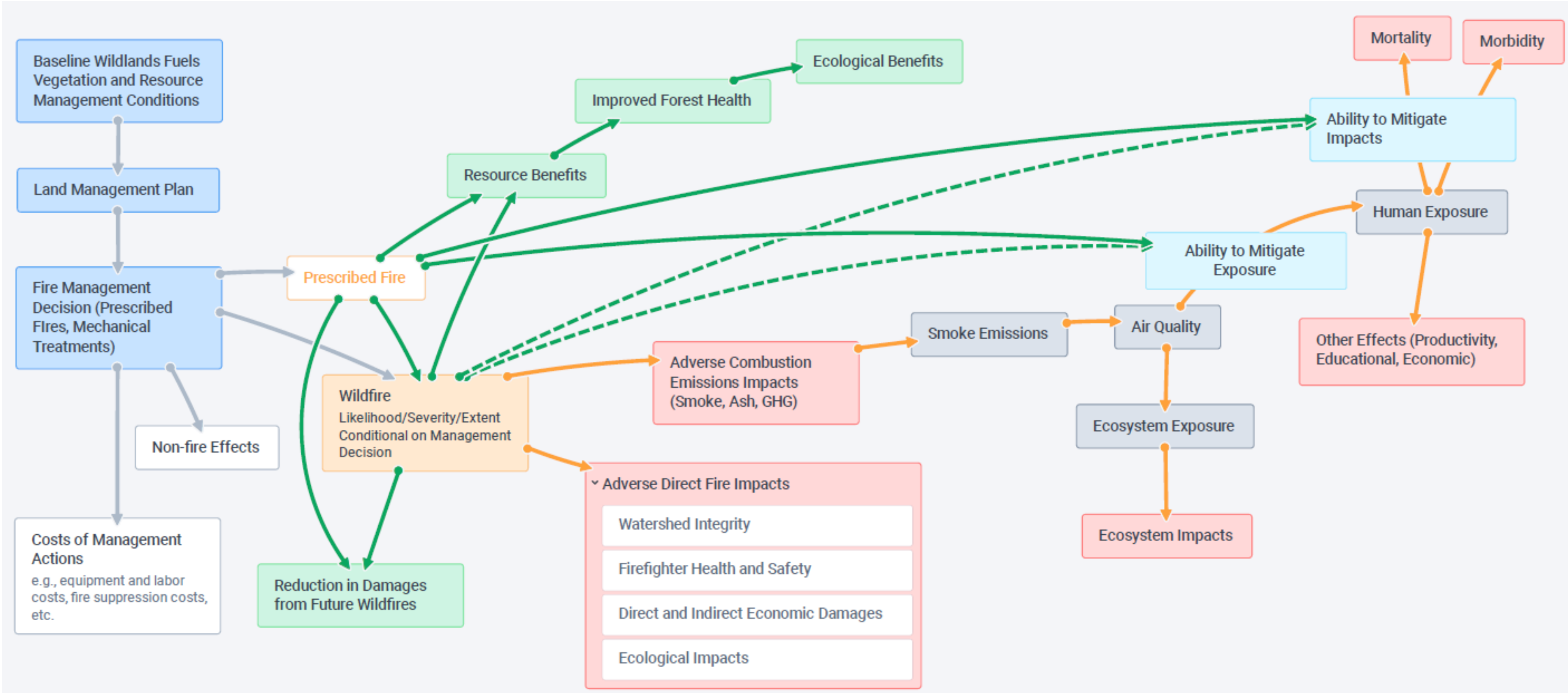
The hiatus from regular fire for the past 100 years has left substantial accumulated fuel on Sierra Nevada forested landscapes, and as a result, frequent, small, regular prescribed fires are not feasible, leading to a high potential for megafires (Stephens et al., 2018; Liu et al., 2016). The Sheep Complex Fire, compared with these megafires like the Rough Fire, was quite small, but the cool, wet conditions under which it was managed limited its ability to spread despite that fuel loading. Resulting air quality and public health impacts were limited directly during the burning of Sheep Complex Fire in 2010, but also contributed to some reductions in impacts from the Rough Fire because the Rough Fire ran into its footprint in 2015. This illustrates the principle that even limited and opportunistic reintroduction of fire to a landscape can reduce the overall footprint of future fires, resulting in quantifiable air quality and public health benefits.

So far, at least in the Rough Fire case study, the results appear to qualitatively corroborate previous case study analyses [e.g., Long et al. (2018); Schweizer and Cisneros (2014); Cisneros et al. (2012)] showing that daily emissions from prescribed fires and fires that yielded positive resource benefits are much lower than those during the Rim fire, which, like the Rough Fire, was ultimately contained through the combination of reduced fuels and fire behavior in previous fire footprints (Long et al., 2018). A limitation of the Rough Fire analysis is the regional-scale resolution (12-km-sized grid cells) of the air quality modeling. This spatial resolution may not fully capture pollutant dispersion in areas with complex terrain, such as the area of the Rough Fire, Sheep Complex Fire, and Boulder Creek Prescribed Fire. When the model does not capture complex meteorology, it is possible emissions from a fire could be unrealistically dispersed over a larger area than would happen in reality and result in an overestimation of air quality and health impacts downwind of the fire and underestimate impacts at the fire itself. Implications for this analysis depend on the degree of over- or underestimation of air quality and health impacts in highly populated areas of the Central Valley of California. Future work using higher resolution modeling (e.g., 2-km resolution), and including a robust comparison of model predictions of $PM_{2.5}$ to observed $PM_{2.5}$ could provide a more refined assessment of the magnitude of trade-offs between the Rough Fire scenarios presented within this assessment.

In summary, in dry forest ecosystems, such as in the area of the Rough Fire, these landscapes will experience some combination of prescribed fire and wildfire. The methodology for assessing public health trade-offs of different fire management strategies developed in this assessment, if deployed on a broader scale, landscape-level analysis, could inform development of management strategies that incorporate protection of regional air quality and public health. In the future, the degree to which the mix of prescribed fire and wildfire for resource objectives can be applied on these landscapes will likely determine whether the effects of future large-scale fires and the corresponding smoke produced can be limited.

9.3 LIMITATIONS IN EXAMINING DIFFERENCES BETWEEN PRESCRIBED FIRE AND WILDFIRE IMPACTS

Throughout this assessment, each chapter characterized the various components of the conceptual framework presented in Chapter 2 (Figure 2-1, and also presented below Figure 9-4) to varying degrees, with some presenting a qualitative characterization of the state of the science, and others providing a quantitative analysis specific to the case study areas. In identifying limitations in the analyses, it is first necessary to review the information presented within each chapter and note which components of the conceptual framework could be addressed broadly and which more specifically within each of the case study analyses (Section 9.3.1). This approach then allows for a discussion of the overarching limitations of the analysis (Section 9.3.2) followed by a discussion of current gaps in the scientific literature that were identified within this assessment (Section 9.3.3). Because the frequency of wildfires continues to grow, along with the frequency of prescribed fire as a land management strategy, considering these limitations and data gaps can aid in further refining the types of analyses conducted within this report and in advancing the overall understanding of the effects of wildland fires.



GHG = greenhouse gas.

Note: This is the same figure presented in Chapter 2, Figure 2-1. Forest management inputs are colored dark blue, management decisions and their nonfire related effects are colored white, resource benefits are colored green, mitigation actions are colored light blue, fires are colored orange, fire damages are colored red, and smoke exposure related elements are colored gray. The green arrows indicate positive effects, and the orange arrows indicate negative effects.

Figure 9-4 Conceptual framework for evaluating and comparing fire management strategies.

9.3.1 IMPLEMENTING THE CONCEPTUAL FRAMEWORK

The ability to implement the conceptual framework, originally outlined in Chapter 2, and the degree to which quantitative information specific to the case study areas is available represents a key aspect of the quantitative estimation of air quality impacts associated with different fire management strategies. Each chapter presents information that is highly relevant to an assessment of the air quality impacts between different fire management strategies; however, this information is often not specific to the case study areas and requires some extrapolation.

Within this assessment, qualitative discussions are presented for multiple components of the conceptual framework because quantitative information specific to the case study areas is lacking. Moving from left to right across the conceptual framework (Figure 2-1, and also Figure 9-4), Chapter 3 captures many of these initial components. This includes the baseline forest/ecological conditions of ecosystems similar to the case study areas, provides background information on different fire management decisions, and a history of fire activity, including the implementation of prescribed fire. In addition, the qualitative discussion in Chapter 3 highlights the instances in which a wildfire can yield resource benefits, which are quantitatively evaluated in the Rough Fire case study through the modeling of the Sheep Complex Fire (Chapter 7 and Chapter 8), and discusses how fire on the landscape can contribute to improved forest health and result in ecological benefits.

The direct fire effects of wildfire (Chapter 5), including effects on society, such as economic and ecological and welfare effects, while important to consider broadly when making comparisons among different fire management strategies cannot be quantified at the case study level. Although there are opportunities to mitigate these direct fire effects, they are not accounted for in this assessment. The adverse combustion emissions impacts, which include greenhouse gas (GHG) emissions (Chapter 3) and ash deposition (Chapter 6), are characterized qualitatively to varying degrees, including the ecological effects of ash deposition.

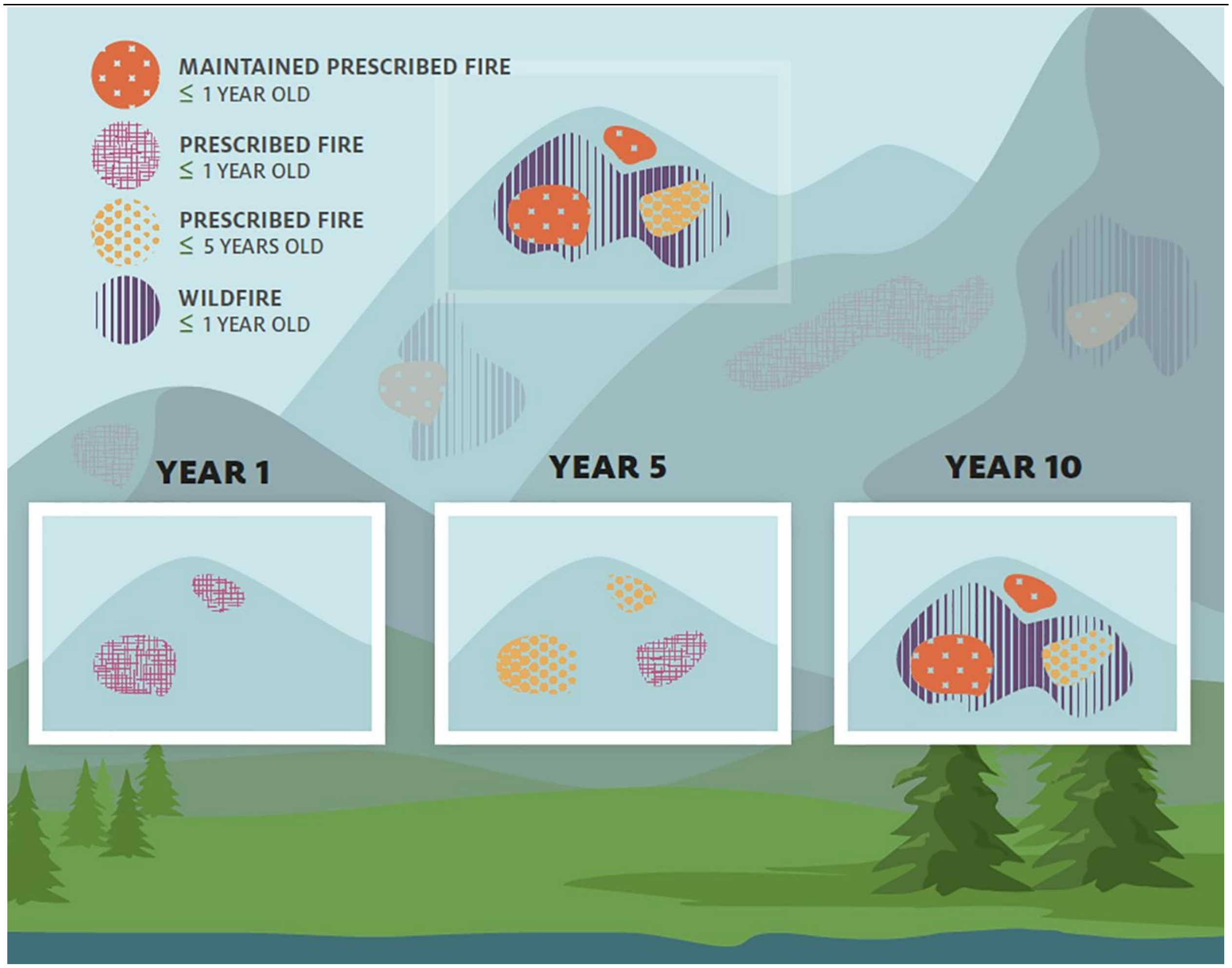
The smoke emissions and corresponding modeling of air quality impacts (Chapter 7) represent the key inputs to the quantitative analyses that form the backbone of this assessment. The results of the air quality modeling directly inform both human and ecosystem exposure with only the resulting human health impacts being quantitatively examined. However, this assessment also provides a qualitative discussion of both health and ecosystem effects attributed to smoke exposure (Chapter 6). The current understanding of the health effects of wildland fire smoke exposure, as well as ambient $PM_{2.5}$ and ozone exposure, are subsequently used within BenMAP-CE to quantify the number of deaths and illnesses attributed to smoke from the different scenarios examined within both case studies.

Additionally, scientific evidence supports the availability and efficacy of various actions and interventions that can be employed at the individual and community level to mitigate the public health impact of smoke exposure (Chapter 6). The overall population $PM_{2.5}$ exposure reductions estimated from these actions and interventions allows for a limited quantitative assessment of the potential public health implications of promoting such measures Chapter 8). Although these actions and interventions can be instituted for both wildfires and prescribed fires, the planned nature of prescribed fires enhances opportunities for public engagement surrounding prescribed fires, and increases the opportunities to inform populations at risk of wildfire smoke-related health effects of actions they can take to protect themselves. In addition to the quantitative and qualitative discussions that directly support components of the conceptual framework, this assessment also presents an overview of the current state of air quality monitoring for wildland fire smoke (Chapter 4). Although the discussion of air quality monitoring does not represent a defined component of the conceptual framework, it is a topic worthwhile to consider in the process of interpreting both the air quality modeling output and epidemiologic studies examining the health effects of smoke, which are the key inputs to the estimation of health impacts.

9.3.2 OVERARCHING LIMITATIONS

As detailed in Chapter 2, and noted in the previous section, the overall conceptual framework for conducting this assessment identifies numerous factors to consider in examining trade-offs between different fire management strategies, including prescribed fire, and the resulting effects, both positive and negative. While many of these factors are characterized in this assessment, there are spatial and temporal dimensions of fire management strategies that are not addressed. In addition, this assessment does not assess the effect of fire management strategies on the probability of wildfire occurrence (i.e., ignition probability), which is potentially a key factor in assessing differences in the cumulative effects of those strategies as depicted in Equation 2-1 in Chapter 2. As recently discussed in Hunter and Robles (2020), the comparison of positive and negative effects of prescribed fire and wildfire is not a static comparison, but one that should be conducted by considering the spatial and temporal aspect of prescribed fires and their interaction with the likelihood, severity, and magnitude of wildfire over a specific time horizon.

In comparison to wildfires, which occur at one uncertain point in time but can vary in length from a few days to months, prescribed fires occur at planned times episodically over many years. Prescribed fires are conducted to achieve a resource benefit (see Chapter 3), with one of the overarching assumptions being that the prescribed fire will contribute to reducing the effect (e.g., size and severity) of a future wildfire. However, to achieve this desired outcome requires a series of prescribed fires over time that provide a patchwork of areas with less fuel, not an individual fire on its own, to minimize the risk of a severe, catastrophic wildfire occurring within the vicinity of the prescribed fires (see Figure 9-5).



Source: Reprinted from Forest Ecology and Management, Vol 475, Hunter and Robles (2020), Tamm review: The effects of prescribed fire on wildfire regimes and impacts: A framework for comparison, Pages No. 118435, Copyright 2020, with permission from Elsevier.

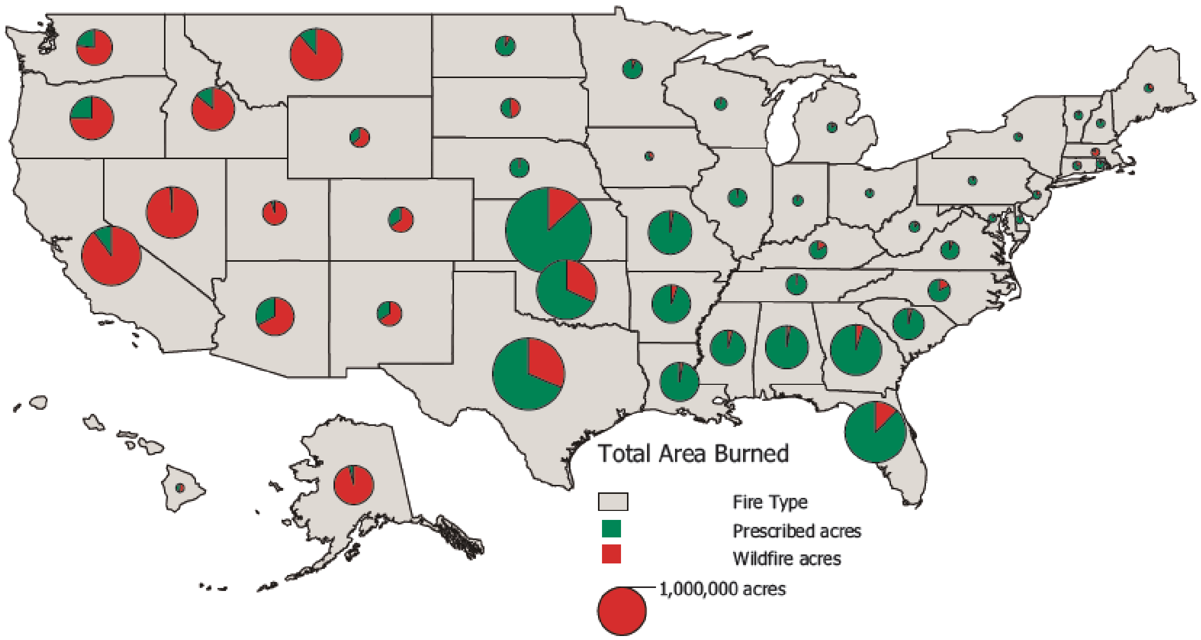
Figure 9-5 Conceptual diagram presented by Hunter and Robles (2020) for assessing the effects of prescribed fire compared to wildfire.

Fully accounting for the trade-offs of smoke impacts between prescribed fire and a wildfire requires an understanding of the intersection of prescribed fire activity (both the total number of prescribed fires and the frequency of prescribed fires) with a wildfire. Although over a long enough time period the probability that a specific location will experience a wildfire can be substantial, yet there is still uncertainty as to when that fire would occur and how severe it would be. Although prescribed fires may reduce both the ignition probability and severity of a future wildfire, they also produce smoke. Therefore, smoke is being produced with the intent of reducing smoke in the future from a wildfire that may, or may not, occur in a location affected by a prescribed fire. Focusing the analyses conducted within this assessment around two previous wildfires and the land management strategies associated with each did not allow for the consideration of ignition probabilities along with the total number and frequency of prescribed fires required to minimize the effects of a wildfire. Instead, these case studies address hypothetical scenarios by asking how the effects of fires that did occur might have differed under different types of fire management strategies. The information provided by these case studies is informative in assessing the benefits of different fire management strategies given the occurrence of fire, but does not address the uncertainty in the time horizon for fire in the landscape, nor the cumulative effects of smoke on health due to a series of prescribed fire activities.

Although for the TC6 Fire case study, there was some information on prescribed fire activity over time, the time window over which these fires occurred complicated the ability to conduct a direct comparison of smoke impacts between prescribed fire and wildfire. As a result, for the TC6 Fire case study, it is assumed that all prescribed fire activity, and subsequent smoke exposures, occurred at one point in time (i.e., September 2019). For the Rough Fire case study, the examination of prescribed fire activity is purely hypothetical because there was no actual prescribed fire activity in the vicinity of the fire. However, by modeling the proposed Boulder Creek Prescribed Fire as if it actually occurred does provide some indication of the potential affect of a prescribed fire on reducing the size of the actual Rough Fire. Therefore, for both case studies, exposure to prescribed fire smoke is being treated as a static event and not the episodic event it is in actuality.

The treatment of prescribed fires as events occurring at one point in time within this assessment, out of both analytical convenience and sparseness of available data, also has ramifications from a health perspective. The removal of the spatial and temporal pattern of prescribed fire activity does not allow for the analyses conducted to consider that the location of prescribed fires varies on a year-to-year basis. By excluding this variability in prescribed fire activity, it is not possible to account for the corresponding spatial and temporal variability in population exposures to smoke that would occur, which could potentially result in a different pattern of health impacts.

In addition to recognizing the spatial and temporal aspects of prescribed fires and wildfires, it is imperative to highlight the vastly different landscapes, in terms of both ecosystem composition (e.g., forests vs. prairie) and the percent contribution of prescribed fire to total wildland fire activity across the U.S. (Figure 9-6). The regional variability in the number of acres burned by prescribed fire and wildfire nationally, specifically in areas with a higher percentage of prescribed fires such as the Southeast, is an additional important consideration when examining air quality impacts associated with different fire management strategies. These regional differences can be attributed to different environmental factors in the Southeast comparted to the West which corresponds to different fire regimes and landscape fire rotations. The greater use of prescribed fire in the Southeast leads to questions on potential air quality impacts as well as potential wildfire levels if prescribed fire use were at greater or lower levels. The variability in the composition of fire activity nationally demonstrates why the results of the case study analyses are not easily transferrable to other parts of the country, especially to areas where the number of acres burned is dominated by prescribed fires. Lastly, as noted earlier in this section, the relationship between prescribed fires and wildfire ignition probabilities are unknown in the case study areas and it is unclear how this relationship varies nationally, particularly in locations dominated by prescribed fires.



Source: Baker et al. (2020), copyright permission pending.

Figure 9-6 Acres burned by wildfire (red) and prescribed fire (green) in the U.S. in 2017.

9.3.3 IDENTIFIED DATA GAPS AND UNCERTAINTIES

In the process of developing the preceding chapters of this assessment, as well as the development of the main modeling framework for the air quality and health impact analyses, gaps were identified in the current scientific understanding of wildland fire smoke. Future efforts to collect data and conduct studies to fill in these gaps could aid in future assessments and allow for a more extensive quantitative estimation of impacts and trade-offs between prescribed fire and wildfire.

A main overarching data gap that filters into multiple aspects of this assessment, but does not represent a key component of the conceptual framework, is the availability of ground-level air quality monitoring data for wildfire smoke. The challenges associated with monitoring wildfire smoke (see Section 4.5), and the resulting paucity of monitoring data, represents an important data gap because air quality monitoring data is instrumental in assessing health effects through epidemiologic studies, as well as in air quality modeling to validate model predictions.

Even without a dense monitoring network to more fully capture the temporal and spatial patterns of population-level exposures to wildfire smoke, epidemiologic studies have still been able to use available air quality data (e.g., satellite, modeling, etc.) to assess the health effects of wildfire smoke. While these studies have been extremely informative and valuable to build upon the broad understanding of the health effects of ambient exposures to $PM_{2.5}$ and ozone, uncertainties remain with respect to both exposure assessment as well as a broader understanding of the health implications of exposures to different durations of wildfire smoke (e.g., repeated peak exposures over many days, exposures over multiple fire seasons) and prescribed fire smoke. Additionally, as reflected in the sensitivity analysis conducted in Chapter 8 (Section 8.3.2), epidemiologic studies that more fully capture wildfire smoke exposure can help inform the concentration-response (C-R) relationship to better understand if there are differences compared to the C-R relationship for ambient $PM_{2.5}$ exposures that should be considered when examining the public health impacts of smoke based on different fire management strategies. However, different exposure indicators are currently used across studies of wildfire smoke, and it remains unclear which exposure indicator best represents wildfire smoke exposure. In addition, better understanding of the differences in the composition of smoke resulting from different burn conditions (e.g., fuel characteristics, moisture levels, and the health effects associated with different smoke composition) can help improve the ability to differentiate between fire management strategies with and without prescribed fire, and also strategies for designing prescribed fire programs to minimize negative health impacts.

Within this report, the economic analysis of smoke impacts between the different fire management strategies for each case study area focuses on mortality and morbidity due to wildland fire smoke exposure. While these economic costs can be substantial depending on the size of the fire (Section 8.3), they only represent a portion of the total economic costs associated with wildland fire smoke exposure. Currently, there is limited information on the other potential effects of smoke (e.g., on the labor market, recreation and exercise, etc.). A better characterization of these other effects would allow for a fuller accounting of the total economic costs associated with wildland fire smoke.

In considering the approach used for the air quality modeling, the assumptions that factored into the methods employed recognize the same overarching limitations discussed in Section 9.3.2 (see Section 7.4). As noted earlier within this chapter, expert judgment was relied upon heavily in defining the hypothetical scenarios for each of the case studies. In addition, in the modeling of prescribed fires for both case studies, all prescribed fire activity over many years was modeled for 1 month in the instance of the TC6 Fire case study or there was no prescribed fire activity in the case of the Rough Fire case study, resulting on the reliance of a proposed prescribed fire that never occurred. Results of analyses, like those conducted here, could more fully capture the differences between different land management and fire management strategies through data that can capture the temporal and spatial scale of prescribed fire activity. Although a fuller accounting of prescribed fire activity over time and space is a key data gap, it also remains unclear how prescribed fire activity could affect the size and duration of a wildfire. The relationship between prescribed fire activity and its influence on wildfire size and duration, especially for larger fires (e.g., Rough Fire) represents a key area that requires additional exploration and prevents extrapolation of results from these case studies to other parts of the U.S.

In addition to the data gaps identified within this section, there are numerous ancillary issues associated with wildfires that are not addressed, but this does not diminish their importance. For example, it is recognized that wildfires can lead to the resuspension of legacy pollutants, such as asbestos, lead, and mercury. These pollutants have been shown to cause a range of health effects, but it remains unclear how much wildfires contribute to population-level exposures to these pollutants. Additionally, over time the wildland-urban interface (WUI) has expanded rapidly in many parts of the U.S. (Radeloff et al., 2018). This expansion has resulted in substantial portions of the population now residing in locations considered high-fire-risk areas. The growth of the WUI not only increases the risk of fire ignitions, but also of direct fire effects. Although Chapter 5 broadly captures direct fire effects, including those associated with the burning of structures that could be experienced within the WUI, currently available information is insufficient for providing location-specific estimates of the costs of wildfire. Lastly, as human development extends further into fire-prone wildlands, it can lead to a change in the composition of smoke as homes and structures are burned and the likelihood of more people being exposed to wildfire smoke.

9.4 KEY INSIGHTS FROM CASE STUDY ANALYSES

This assessment, and the accompanying quantitative analyses, represent an incremental advancement in the understanding of the air quality and health impacts of wildland fires under different fire management strategies. As a reminder, the results of the analyses conducted within this assessment are specific to the case study areas and are not intended to represent the air quality and health impacts that would be observed in other locations around the U.S. The case studies were chosen to illustrate the type and nature of air quality and health impacts associated with different fire management strategies. Additionally, in examining the air quality and health impacts due to wildland fire, the analyses are retrospective and represent locations that experienced a wildfire, and therefore, do not (1) account for the temporal and spatial variability of prescribed fires occurring over many years that happens in reality or would happen in an ideal situation to minimize the risk of catastrophic wildfire and (2) incorporate an estimate of uncertainty to account for the probability that a wildfire may not occur in a location where there was prescribed fire activity. The case study analyses conducted within this assessment support the following observations:

- To provide a reasonable estimation of air quality and health impacts from wildland fire, location-specific information on fuels is needed to support air quality modeling.

- The case study analyses show that the smoke impacts of wildland fire are complex both spatially and temporally, but do not account for the possibility of multiple fires, either wildfire or prescribed fire, occurring concurrently or in sequence across very broad landscapes or multiple geographic areas, including internationally.
- In the case study areas, predicted concentrations of $PM_{2.5}$ from the modeled prescribed fires are smaller in magnitude and shorter in duration than wildfires, and the estimated aggregate population exposure for prescribed fires is smaller than for each hypothetical scenario and the actual fires in both case studies.
    - Smaller estimated aggregate population $PM_{2.5}$ exposures for prescribed fires can be attributed to the small spatial extent of each prescribed fire and the meteorological characteristics of the days in which the prescribed fires occurred.
    - Although prescribed fires are timed for days with specific meteorological conditions to minimize population exposures to smoke, air quality and public health impacts are still observable.
- Well-designed prescribed fires targeted for specific locations may be able to reduce air quality and health impacts of subsequent wildfires. For example, in the Rough Fire case study:
    - A smaller wildfire that yielded positive resource benefits, the Sheep Complex Fire, previously occurred adjacent to the Rough Fire location.
    - Although there was no prescribed fire activity in the vicinity of the Rough Fire, a prescribed fire was planned for but not carried out, the Boulder Creek Prescribed Fire, which would have been adjacent to the Sheep Complex Fire.
    - Modeling showed that if the proposed Boulder Creek Prescribed Fire had occurred adjacent to the Sheep Complex Fire it could have reduced the overall footprint of the Rough Fire, resulting in an approximate 40% reduction in estimated health impacts.
- The case studies were retrospective, i.e., they were based on locations where there were documented wildfires that employed some previous fire management strategies. Thus, case study results reflect that a wildfire occurred in both locations and do not account for the fact a wildfire may, or may not, occur in a location that would be affected by a prescribed fire.
- Smoke impacts on health (i.e., cardiovascular and respiratory-related emergency department visits and mortality) are driven primarily by exposure to $PM_{2.5,}$ but exposure to both $PM_{2.5}$ and ozone from smoke are dependent upon population proximity to wildland fire events and meteorology (e.g., wind speed and direction).
- Within the case study areas, ozone produced from wildland fires is shown to have fewer impacts on air quality and public health, providing additional support to the current public health focus for wildland fires being on reducing exposures to $PM_{2.5}$.
- Wildfires, such as the TC6 Fire that are short in duration, small in size, and not near large downwind population centers can still result in public health impacts, albeit substantially smaller than for larger wildfires such as the Rough Fire.
- Communicating the benefits of reducing wildland fire smoke exposure through individual actions and interventions (e.g., evacuation, air cleaners, filters for heating, ventilation, and air conditioning [HVAC] systems) that decrease $PM_{2.5}$ exposures can contribute to decreasing the public health impacts due to wildland fire smoke if these exposure reduction actions are more widely used.

9.5 FUTURE DIRECTIONS

The analyses conducted within this assessment lay the foundation for future research efforts to examine the air quality and corresponding public health impacts of smoke from wildland fire under different fire management strategies. Although the results of the quantitative analyses provide initial evidence of differences in smoke impacts between prescribed fire and wildfire, additional research efforts that attempt to address the following issues will further enhance the applicability of future analyses examining the trade-offs between different fire management strategies:

- Identification and development of methods to account for the temporal (i.e., frequency) and spatial component of prescribed fires and their relationship with wildfires. This would allow for a better understanding of how to capture the health effects of repeated exposure to smoke from prescribed fires over many years and how that compares to the health effects experienced during singular wildfire events.
- Improved characterization of the relationship between prescribed fire and wildfire on the landscape. This would include analyses that examine specific spatial domains with prescribed fires and the number of those locations that also experienced a wildfire, along with identifying whether prescribed fires were able to reduce characteristics of the wildfire (e.g., size, intensity, duration, etc.). This advancement would then allow for a greater understanding of the costs and benefits of different fire management strategies with and without wildfire.
- Analyses that characterize the role of topography and meteorology, in combination with the frequency of prescribed fires within a spatial domain, on the potential for population centers to experience smoke impacts from wildland fires.
- Characterization of how air quality impacts differ between prescribed fire and wildfire in different parts of the U.S., specifically in locations where prescribed fire is the dominant wildland fire activity, to gain a better understanding of the ability to extrapolate results across geographic locations.
- A centralized repository to capture prescribed fire data to enhance future assessments using more recent data. Such a repository would include, but not be limited to, information on location, timing (dates and approximate start and end time), actual acres burned, fuel type and loading information, and any air quality monitoring data collected.

In addition to these broad areas that require additional research to support future analyses, there are overarching uncertainties and limitations identified in previous chapters that if addressed could further enhance our understanding of the overall impacts of wildland fire smoke. These areas of additional research include enhanced air quality monitoring capabilities for wildfire smoke, better characterization of wildland fire smoke exposures for health studies, additional understanding of the health effects of wildfire smoke over many seasons, and a fuller accounting for the role of public health actions and interventions in reducing or mitigating wildland fire smoke exposure. Future research initiatives and science advancements that attempt to address the current deficiencies related to fire science noted within this section, would allow for a fuller characterization of the air quality and health impacts due to different fire management strategies.

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APPENDIX A

A.1. Supplemental Information for Chapter 1

No supplemental information.

A.2. Supplemental Information for Chapter 2

Appendix Table A.2-1 represents a more detailed version of Table 2-1 that attempts to characterize whether the effects associated with wildland fire are negative or positive.

Table A.2-1 Positive and negative effects associated with Wildland Fire.[a]

Categories	Prescribed Fire		Wildfire	
	During the Event	Post-Event[b]	During the Event	Post-Event
Firefighting				
Firefighter safety	−	+	−	+ and/or −
Firefighter injuries/fatalities	−	+	−	+ and/or −
Firefighter health, both mental and physical (mental and physical)	−	+	−	+ and/or −
Economic				
Evacuations	NA	+	−	+ and/or −
Property (e.g., structures)	NA	+	−	+ and/or −
Property (e.g., loss of ecosystem services)	+ and/or −	+	−	+ and/or −
Timber and grazing	+ and/or −	+	−	+ and/or −
Infrastructure (e.g., powerlines, recreation, others)	NA	+	−	+ and/or −
Municipal watersheds (e.g., reservoirs, industry, agriculture, drinking)	+	+	−	+ and/or −
Tourism (e.g., recreation, lodging, restaurants, etc.)	+ and/or −	+	−	+ and/or −
Aesthetics (e.g., property value, view shed, etc,)	+ and/or −	+	−	+ and/or −

Table A.2-1 (Continued): Positive and negative impacts associated with wildland fire.[a]

Categories	Prescribed Fire		Wildfire	
	During the Event	Post-Event[b]	During the Event	Post-Event
Natural and cultural resources	+ and/or −	+	−	+ and/or −
Fuel reduction—cost effective method of treating acres	NA	+	+ and/or −	+ and/or −
Fuel reduction—treatment opportunities not limited to markets	NA	+	+ and/or −	+ and/or −
Ecological				
Ecological services including game and endangered species	NA	+	+ and/or −	+ and/or −
Ecosystem health and resiliency	NA	+	+ and/or −	+ and/or −
Restoration/maintenance of historic natural fire regime	NA	+	+ and/or −	+ and/or −
Invasive species	+ and/or − or NA	+	+ and/or −	+ and/or −
Climate change (e.g., GHG, carbon)	+ and/or −	+ and/or −	+ and/or −	+ and/or −
Redistribution of toxics and nutrients (e.g., mercury, metals, sulfur, nitrogen)	−(?)	−(?)	−(?)	−(?)
Soil and water quality and quantity	+ and/or −	+ and/or −	+ and/or −	+ and/or −
Public Health: Direct Fire				
Injuries	NA	+	−	+ and/or −
Hospitalizations	NA	+	−	+ and/or −
Premature mortality	NA	+	−	+ and/or −
Public Health: Air Quality				
Hospitalizations and emergency department visits	−	+	−	+ and/or −
Premature mortality	−	+	−	+ and/or −
Nonfatal heart attacks/cerebrovascular events	−	+	−	+ and/or −
Asthma effects	−	+	−	+ and/or −
Other respiratory and illness effects	−	+	−	+ and/or −

Table A.2-1 (Continued): Positive and negative impacts associated with wildland fire.[a]

Categories	Prescribed Fire		Wildfire	
	During the Event	Post-Event[b]	During the Event	Post-Event
Loss of work and school days	−	+	−	+ and/or −

GHG = greenhouse gas; NA = not available.

Note: Positive (+): providing some advantage (e.g., restoring ecosystems, mitigating the risk or loss from a wildfire, etc.). Negative (−): negative consequences from a fire (e.g., property or infrastructure damage or loss). For many of the categories with an NA for prescribed fires, the impact will not be applicable as long as the prescribed fire remains consistent with the management objectives. In the rare cases where prescribed fires are no longer meeting their objectives, they can be reclassified as wildfires and will in those cases have the potential for additional negative impacts.

[a]Signs on the impact categories are based on literature discussed throughout this report as well as expert judgments from the report authors.

[b]Post-event includes impacts expected to occur as a result of reductions in the risk of more severe and damaging wildfires. For example, reduced risk of severe wildfires reduces risks to firefighters and reduces risks of poor air quality and related health effects. Thus, a positive sign on the post-fire effects of prescribed fires on health categories does not indicate the fire itself improves health, but rather that the reduction in risk of severe wildfires improves future public health.

A.3. Supplemental Information for Chapter 3

No supplemental information.

A.4. Supplemental Information for Chapter 4

Table A.4-1 Criteria gas pollutant Federal Reference Methods (FRMs) and most widely employed Federal Equivalent Methods (FEMs) used in U.S. EPA regulatory monitoring.

Pollutant Method	Operating Principle	FRM Regulatory Citation	Notes
CO			
Automated FRM	NDIR	40 CFR Part 50 Appendix C (U.S. EPA, 2020a)	---
Automated FEM	Mercury replacement UV photometry	---	Only existing CO FEM.
O_3			
Automated FRM	Chemiluminescence	40 CFR Part 50 Appendix D (U.S. EPA, 2011b)	Employs chemiluminescence reaction between ozone and ethylene of NO. Ethylene chemiluminescence FRM instruments are no longer commercially available. NO chemiluminescence method was promulgated as a new FRM in 2015. NO chemiluminescence FRM instruments are available commercially.
Automated FEM	UV Photometry	---	Severe smoke interference resulting in overestimation of ozone concentrations (Long et al., In Press).
Automated FEM	Open-path DOAS	---	Employs open monitoring path length from 20−1,000 m.
NO_2			
Automated FRM	Chemiluminescence	40 CFR Part 50 Appendix F (U.S. EPA, 2011a)	Employs the catalytic conversion of NO_2 to NO with subsequent chemiluminescence detection of the reaction between NO and O_3. Known interference by higher oxides of nitrogen (e.g., HNO_3, HNO_2, particulate nitrate).

Table A.4-1 (Continued): Criteria gas pollutant Federal Reference Methods (FRMs) and most widely employed Federal Equivalent Methods (FEMs) used in U.S. EPA regulatory monitoring.

Pollutant Method	Operating Principle	FRM Regulatory Citation	Notes
Automated FEM	Chemiluminescence	---	Employs the photolytic conversion of NO_2 to NO with subsequent chemiluminescence detection of the reaction between NO and O_3. Considered more specific for NO_2 than the FRM and is a candidate for future FRM consideration.
Automated FEM	Spectroscopic	---	Employs methods such as CAPS spectrometry.
Automated FEM	Open-path DOAS	---	Employs open monitoring path length from 50−1,000 m.
SO_2			
Automated FRM	UV fluorescence	40 CFR Part 50 Appendix A-1 (U.S. EPA, 2011c)	Previously an FEM, promulgated as a new FRM in 2010.
Manual FRM	Pararosaniline method	40 CFR Part 50 Appendix A-2 (U.S. EPA, 2020b)	Manual wet chemical method not used at present time.
Automated FEM	UV fluorescence	---	Promulgated as a new FRM in 2010.
Automated FEM	Open-path DOAS	---	Employs open monitoring path length from 20−1,000 m.

CAPS = cavity attenuated phase shift; CFR = Code of Federal Regulations; CO = carbon monoxide; DOAS = differential optical absorption spectroscopy; FEM = Federal Equivalent Method; FRM = Federal Reference Method; HNO_2 = nitrous acid; HNO_3 = nitric acid; NDIR = nondispersive infrared photometry; NO = nitric oxide; NO_2 = nitrogen dioxide; O_3 = ozone; SO_2 = sulfur dioxide; UV = ultraviolet.

Table A.4-2 Summary of low-cost sensors evaluated in biomass smoke.

Vendor	Model	Study Type	Max $PM_{2.5}$	Reference	Reference Regression Slope	Citation
Aeroqual	AQY 1	Field	~300 (μg/m³)	FEM/non-FEM	0.54−2.18	Holder et al. (2020)
eLichens	IAQPS	Field	~150 (μg/m³)	Multiple FEMs	~0.45−0.80	Delp and Singer (2020)
PurpleAir	PA-II-SD	Field	~300 (μg/m³)	FEM/non-FEM	0.93−1.61	Holder et al. (2020)
PurpleAir	PA-II	Field	33 (μg/m³)‡	FEM	0.43	Mehadi et al. (2019)
PurpleAir	PA-II	Field	~150 (μg/m³)	Multiple FEMs	0.39−0.54	Delp and Singer (2020)
Sensit	RAMP	Field	~300 (μg/m³)	FEM/non-FEM	0.77−1.48	Holder et al. (2020)
Sensit	RAMP	Chamber	~1,800 (μg/m³)	FRM	1.35−2.43	Landis et al. (2021)
Thingy	Thingy AQ	Chamber	~1,800 (μg/m³)	FRM	2.14−4.95	Landis et al. (2021)
Wicked Device	Air Quality Egg	Field	~150 (μg/m³)	Multiple FEMs	~0.32−0.65	Delp and Singer (2020)

μg/m³ = micrograms per cubic meter; FEM = Federal Equivalent Method; FRM = Federal Reference Method; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 μm.

‡ Daily average concentration.

Table A.4-3 Summary of routine $PM_{2.5}$ measurement methods and data availability.

Methods	$PM_{2.5}$ FRMs	CSN and IMPROVE	$PM_{2.5}$ Continuous FEMs	Other $PM_{2.5}$ Continuous Methods	Sensor Networks
Method Specifications					
Manual or automated	Manual	Manual	Automated continuous	Automated continuous	Automated continuous
Measurement principle(s)	Gravimetric in laboratory	Ion chromotography, x-ray fluorescence, Thermal Optical Reflectance all in laboratory	Key ones include: β attenuation (BAM), TEOM, and LED broadband spectroscopy	Key ones include: β attenuation, Nephelometers, and TEOMs	Optical PM sensors
Method or manufacturer-reported concentration range	0−200 μg/m^3; however, in AQS, there are a few values in the Hazardous AQI category	0−200 μg/m^3	BAM-Range: 0−1,000 μg/m^3 standard; up to 10,000 μg/m^3; T640-Range: 0.1−10,000 μg/m^3		PurpleAir with U.S. EPA-ORD correction equation (Barkjohn et al., 2020) 0−250 μg/m^3 range (>250 μg/m^3 may underestimate true $PM_{2.5}$)
Manufacturer-reported data resolution	0.1 μg/m^3	0.1 μg/m^3	M1 BAM: 1 μg/m^3 TEOM and T640: 0.1 μg/m^3		0.1 μg/m^3
Data Attributes of Each Method					
Data availability (typical)	~1−3 mo after sample collection	~3−6 mo after sample collection	Hourly data are usually posted to AIRNow within several minutes past the end of the hour		Near real time on PurpleAir website Hourly update on AIRNow fire and smoke map

Table A.4-3 (Continued): Summary of routine $PM_{2.5}$ measurement methods and data availability.

Methods	$PM_{2.5}$ FRMs	CSN and IMPROVE	$PM_{2.5}$ Continuous FEMs	Other $PM_{2.5}$ Continuous Methods	Sensor Networks
Data interval available	24-h midnight to midnight local standard time. Some sites operate daily, others every 3rd or 6th day; some QA samplers every 12th day	24-h midnight to midnight local standard time. Most sites operate every 3rd day; some CSN sites every 6th day	Hourly data is collected and reported by AIRNow; some methods have subhourly data available (T640 has 1-min data available—smoothed in rolling 10-min averages)		Subhourly; data layer on AIRNow fire page is hourly
Where are data available?	AQS— https://www.epa.gov/aqs/obtaining-aqs-data	AQS and UC Davis website— https://airquality.ucdavis.edu/csn https://airquality.ucdavis.edu/improve	AQS, AIRNow, AIRNowTech, and many state and local websites— https://www.airnow.gov/ http://airnowtech.org/ (credentials required)		PurpleAir website, AIRNow fire and smoke page— https://fire.airnow.gov/ https://www.purpleair.com
Highest concentrations reported with this method to AQS (2010−2019).	There are seven cases in the “Hazardous AQI category” all in AK, CA, or OR. The highest reported concentration was 411.7 µg/m^3.	There are no cases in Hazardous AQI category. There are 13 cases in the “very unhealthy” AQI category and 8 by the IMPROVE method; high = 210.2 µg/m^3 all in CA and MT; 1 by a SASS (CSN) at 206.7 µg/m^3 in IL; and four cases listed as a generic filter-based method, high = 230 µg/m^3 all in CA and NV.	Six cases reported in the Hazardous AQI category. All with a BAM in CA, MT, or WA. High = 557.1 µg/m^3.	In the Hazardous AQI category, there are 21 cases with a Correlated Nephelometer all in OR or WA, high reported = 570.3 µg/m^3; 34 cases with a BAM all reported in AK, CA, ID, or MT, high = 642.0 µg/m^3; 1 case with a TEOM at 252.0 µg/m^3 in ID.	NA

Table A.4-3 (Continued): Summary of routine $PM_{2.5}$ measurement methods and data availability.

Methods	$PM_{2.5}$ FRMs	CSN and IMPROVE	$PM_{2.5}$ Continuous FEMs	Other $PM_{2.5}$ Continuous Methods	Sensor Networks
Network Attributes					
U.S. Stations Reporting to AQS (2020)	538	CSN = 145 IMPROVE = 156	660	290	NA
Key network Design features	Most sites are population-orientated locations in CBSAs. Each state should have a background and transport site	CSN includes STN, NCore, and supplemental sites (most in CBSAs.) IMPROVE supports Regional Haze Program with most sites in Class 1 areas and national parks. Some IMPROVE protocol sites are operated in lieu of CSN.	Same as FRM	Same as FRM. In WA and OR, nephelometers are often used to supplement AQI reporting, communicates where NAAQS comparable data are not required; however, smoke impacts may be of concern.	Sites may exist anywhere users report via Internet to PurpleAir site. Users self-describe if ambient air or inside. Note: only sites described as ambient air are used in fire and smoke map layer.

μg/m^3 = micrograms per cubic meter; AQI = Air Quality Index; AQS = Air Quality System; BAM = Beta attenuation monitoring; CBSA = core-based statistical area; CSN = Chemical Speciation Network; FEM = Federal Equivalent Method; FRM = Federal Reference Method; h = hour; IMPROVE = Interagency Monitoring of Protected Visual Environments; LED = light-emitting diode; min = minute; mo = month; NA = not applicable; NAAQS = National Ambient Air Quality Standards; NCore = National Core Network; ORD = Office of Research and Development; PM = particulate matter; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 μm; QA = quality assurance; STN = Speciation Trends Network; TEOM = Tapered Element Oscillating Microbalance.

Table A.4-4 Overview of wildland fire relevant imagery/composition satellite data products.

Satellite Product	Instrument	System Content: NOAA Aerosol Watch	NOAA JSTAR Mapper	NOAA Hazard Mapping System	NASA LANCE/World View	U.S. EPA AIRNow Tech	U.S. EPA Remote Sensing Information Gateway
Corrected Reflectance True Color	GOES-ABI	I		I, D			
	VIIRS	I	I	I, D	I, D		I, D
	MODIS				I, D	I	I, D
Digitized Smoke Analysis	ABI + VIIRS			I, D			
Aerosol Optical Depth	ABI	I					
	VIIRS	I	I		I, D		I, D
	MODIS				I, D		I, D
Aerosol Detection (smoke/dust)	ABI	I				I	
	VIIRS	I	I		I, D		

Table A.4-4 (Continued): Overview of wildland fire relevant imagery/composition satellite data products.

		System Content					
Satellite Product	**Instrument**	**NOAA Aerosol Watch**	**NOAA JSTAR Mapper**	**NOAA Hazard Mapping System**	**NASA LANCE/World View**	**U.S. EPA AIRNow Tech**	**U.S. EPA Remote Sensing Information Gateway**
Fire Characterization/Hot Spots/Active Fires	ABI	I			I, D		I, D
	VIIRS	I	I		I, D		I, D
AI	TROPOMI		I		I, D		I, D
CO							
NO_2							
Satellite predicted $PM_{2.5}$		I, D				I, D (ASDP)	
Surface concentration measurements from AIRNow or AQS ($PM_{2.5}$, O_3, NO_2, SO_2)	AirNow	I (h $PM_{2.5}$ only)				I,D	I,D
	AQS					I,D	I,D

ABI = Advanced Baseline Imager; AI = aerosol index; AQS = Air Quality System; ASDP = AirNow Satellite Data Processor; CO = carbon monoxide; D = data available; EOS = Earth Observing System; GOES = Geostationary Operational Environmental Satellite; h = hourly; I = image available; LANCE = Land, Atmosphere Near real-time Capability for EOS; MODIS = Moderate Resolution Imaging Spectroradiometer; NASA = National Aeronautics and Space Administration; NO_2 = nitrogen dioxide; NOAA = National Oceanic and Atmospheric Administration; O_3 = ozone; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 μm, SO_2 = sulfur dioxide; TROPOMI = TROPOspheric Monitoring Instrument; VIIRS = Visual Infrared Imaging Radiometer Suite.

Table A.4-5 Ground-based remote sensing networks vertical (profile and total column data).

Network and/or Instrument	Lead Organization	Total Number of Sites in U.S.	Date Latency	Initiated Measurement	Relevant Constituent/ Properties	URL For Information On Measurements/Data
ASOS	NOAA	900			Surface visibility	https://www.aviationweather.gov/metar?gis=off
Photochemical Assessment Monitoring Stations	U.S. EPA	~40		2021	Backscatter aerosol profiles (15 km), PBLH, aerosol layer identification	https://www.ucn-portal.org/
MPLNET	NASA (federated)	35		2000	Aerosols and cloud layer heights	http://mplnet.gsfc.nasa.gov/
AERONET	NASA (federated)	~100		1998	Aerosol spectral optical depths, aerosol size distributions, and precipitable water	http://aeronet.gsfc.nasa.gov/index.html
Pandonia Global Network	NASA-ESA	14			Total Column O_3, NO_2, tropospheric column NO_2, CH_2O, and surface NO_2	https://www.pandonia-global-network.org/

AERONET = AErosol RObotic NETwork; ASOS = Automated Surface Observing System; CH_2O = formaldehyde; ESA = European Space Agency; km = kilometer(s); LiDAR = Light Detection and Ranging; MPLNET = Micro-Pulse LiDAR Network; NASA = National Aeronautics and Space Administration; NOAA = National Oceanic and Atmospheric Administration; NO_2 = nitrogen dioxide; O_3 = ozone; PBLH = planetary boundary layer heights.

A.4.1. Example State and Local Sponsored Smoke Blogs

Information on general ambient air quality, the impact of wildland fire smoke on current ambient air quality conditions, and air quality forecasts are available to the public through the multiagency AIRNow website as well as state and local websites. Several western states maintain websites (“smoke blogs”) dedicated to providing the public with information on wildfire smoke impacts (examples listed below). The material delivered by these smoke blogs varies from state to state with the sites compiling smoke and fire observations and forecast products from a variety of sources (e.g., AIRNow, dedicated state/local monitors). Below are some example state and local websites and smoke blogs that provide air quality information to the public and are a resource during wildfire events with the landing page title in parentheses.

- Alaska
  (Wildfire Smoke—Particulate Matter Information)
  https://dec.alaska.gov/air/air-monitoring/wildfire-smoke-info/
- Arizona
  (Wildfire Support)
  http://www.azdeq.gov/node/2913
- California
  Butte County Air Quality Management District (AQMD, Wildfires and Air Quality)
  https://bcaqmd.org/resources-education/wildfires/
- North Coast Unified Air Quality Management District
  http://www.ncuaqmd.org/index.php?page=wildfire
- Santa Barbara Pollution Control District, California (Today’s Air Quality and Forecasts)
  https://www.ourair.org/todays-air-quality/
- South Coast Air Quality Management District, California (South Coast AQMD)
  http://www.aqmd.gov/
- Ventura County Air Pollution Control District (VCAPD)
  http://www.vcapcd.org/
- Idaho
  (Air Quality Index [AQI])
  https://www.deq.idaho.gov/air-quality/air-quality-index/
- Idaho Smoke Information
  http://idsmoke.blogspot.com/
- Montana
  (Wildfire Smoke Update)
  https://svc.mt.gov/deq/todaysair/smokemostrecentupdate.aspx
- Montana Wildfire Smoke
  https://www.montanawildfiresmoke.org/

- Nevada
(Northern Sierra Air Quality Management District)
https://myairdistrict.com/

- New Mexico
(Wildfire and Prescribed Fire Smoke Resources)
https://www.env.nm.gov/air-quality/fire-smoke-links/

- North Carolina
(Air Quality)
https://deq.nc.gov/about/divisions/air-quality

- Oregon
(Oregon Smoke Information)
http://oregonsmoke.blogspot.com/

- South Carolina
(Wildfires—Protect Yourself)
https://scdhec.gov/disaster-preparedness/wildfires-protect-yourself

- Washington
(Washington Smoke Information)
https://wasmoke.blogspot.com/

A.4.2. U.S. EPA $PM_{2.5}$ Mass Monitoring

The particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm ($PM_{2.5}$) monitoring program is one of the major ambient air monitoring programs operated across the country. For most urban locations, $PM_{2.5}$ monitors are sited at the neighborhood scale as defined in 40 Code of Federal Regulations (CFR) Appendix D to Part 58 (U.S. EPA, 2015), where $PM_{2.5}$ concentrations are reasonably homogeneous throughout an entire urban subregion. In each CBSA with a monitoring requirement, at least one $PM_{2.5}$ monitoring station representing area-wide air quality is to be sited in an area of expected maximum concentration.

There are three main components of the $PM_{2.5}$ monitoring program: 24-hour integrated filter-based Federal Reference Method (FRM) samplers, continuous Federal Equivalent Method (FEM) mass instrument measurements reported as 1-hour concentrations, and 24-hour integrated filter-based Chemical Speciation Network (CSN) samplers. The FRM data are primarily used for determining National Ambient Air Quality Standards (NAAQS) compliance, but also serve other important purposes such as developing trends and evaluating the field performance FEM continuous mass instruments. Continuous FEM instrument data are also used for determining NAAQS compliance and their real-time data support public AQI communication and air quality forecasting on AIRNow. FRMs have been available since the $PM_{2.5}$ monitoring network began operation in January of 1999, and $PM_{2.5}$ continuous FEMs became commercially available in 2008. Many state and local agencies are transitioning their regulatory $PM_{2.5}$ monitoring networks to continuous FEMs. However, even if a monitoring agency chooses to run $PM_{2.5}$ continuous FEMs at all their stations, some FRMs are still required. For example, FRMs are required under quality assurance (QA) requirements and at U.S. Environmental Protection Agency (U.S. EPA) National Core Network (NCore) stations [Appendices A and D to 40 CFR Part 58; U.S. EPA (2019, 2015)].

The CSN and related Interagency Monitoring of Protected Visual Environments (IMPROVE) network is used to provide chemical composition of the aerosol, which serves several objectives. The CSN program is managed by U.S. EPA with field operations conducted by state and local agencies and national contract laboratories responsible for shipping, handling, and analysis of samples. The IMPROVE is operated by the Department of the Interior (DOI) under the direction of a multiagency federal/state steering committee. The IMPROVE monitoring program supports the national goal of reducing haze to near natural levels in national parks and wilderness areas.

In 2020 there were 538 FRM filter-based samplers included in the U.S. EPA $PM_{2.5}$ network that provide 24-hour $PM_{2.5}$ mass concentration data. Of these operating FRMs, 68 are providing daily $PM_{2.5}$ data, 340 every 3rd day, 119 every 6th day, and 11 every 12th day. As of 2020, there are 950 continuous $PM_{2.5}$ mass monitors that provide hourly data on a near real-time basis reporting across the country. A total of 660 of the $PM_{2.5}$ continuous monitors are FEMs and therefore used both for comparison with the NAAQS and to report the AQI. Another 290 monitors not approved as FEMs are operated primarily to report the AQI. These legacy $PM_{2.5}$ continuous monitors were largely purchased prior to the availability of designated $PM_{2.5}$ continuous FEM instruments. The most widely used $PM_{2.5}$ continuous monitor not designated as an FEM is the Radiance Research (Seattle, WA) Model M903 nephelometer (locally correlated to an FRM).

The first designated automated $PM_{2.5}$ FEM instrument was the Met One Instruments (Grants Pass, OR) Model BAM 1020 (14C β attenuation radiometric method) in 2008. The BAM 1020 and more recently approved BAM 1022 account for approximately 50% and the Teledyne API (San Diego, CA) Model T640/T640x account for approximately 30% of the nationally operating automated $PM_{2.5}$ FEMs. The U.S. EPA has approved a total of 11 $PM_{2.5}$ automated methods as FEMs including beta attenuation from multiple instrument manufacturers; optical methods such as the GRIMM Aerosol Technik (Ainring, Germany) Model 180 and the Teledyne API Model T640/T640x; and methods employing the Thermo Environmental (Franklin, MA) Model 1405 Tapered Element Oscillating Microbalance (TEOM) with a Filter Dynamic Measurement System (FDMS).

A.4.3. U.S. EPA $PM_{2.5}$ Speciation Monitoring

Particulate matter (PM) is the generic term for a broad class of chemically and physically diverse substances that exist as liquid and/or solid particles over a wide range of sizes. Particles originate from a variety of anthropogenic stationary and mobile sources, as well as from natural sources like wildfires. Particles may be emitted directly or formed in the atmosphere by photochemical transformations of gaseous precursors such as sulfur dioxide (SO_2), nitrogen oxides (NO_X), ammonia (NH_3), and volatile organic compounds (VOCs). The chemical and physical properties of $PM_{2.5}$ vary greatly with time, region, meteorology, and source category. U.S. EPA implemented the CSN to investigate the chemical components of $PM_{2.5}$ at selected locations across the country. This information is commonly used to support $PM_{2.5}$ source apportionment/receptor modeling and mass reconstruction efforts that assist in developing State Implementation Plans (SIP) and can provide valuable information on relative toxicity. The CSN sample filters are analyzed for 33 trace elements using energy dispersive x-ray fluorescence [EDXRF; Watson et al. (1999); Jaklevic et al. (1981)], water soluble major ions (e.g., ammonium, potassium, nitrate, sulfate) using ion chromatography [IC; U.S. EPA (1999)], and elemental carbon (EC)/organic carbon (OC) using thermal optical reflectance [TOR; Chow et al. (1993); Huntzicker et al. (1982)]. The addition of one or more well-established tracer species for biomass combustion like levoglucosan [an anhydro sugar produced from the combustion of cellulose; Sullivan et al. (2014); Sullivan et al. (2011b); Sullivan et al. (2011a)] to the analytical suite of existing filter-based monitoring networks (CSN, FRM, IMPROVE) would be invaluable to elucidate the relative impact of wildland fire smoke on measured $PM_{2.5}$ (Landis et al., 2018).

In 2020 the CSN continued routine long-term $PM_{2.5}$ measurements at 145 predominately urban locations. The major network components of the CSN include the Speciation Trends Network (STN), NCore stations, and supplemental speciation sites. STN sites are intended to be long-term locations where chemical section measurements are taken. NCore is a multipollutant network measuring $PM_{2.5}$ mass, criteria gases, and basic meteorology; it has been in formal operation since January 1, 2011. Particle measurements made at NCore include $PM_{2.5}$ filter-based mass, which is largely the FRM, except in some rural locations that use the IMPROVE program $PM_{2.5}$ mass filter-based measurement. $PM_{2.5}$ speciation using either the CSN program or IMPROVE program; and coarse particulate matter ($PM_{10-2.5;}$ particulate matter with a nominal mean aerodynamic diameter less than or equal to 10 μm and greater than a nominal 2.5 μm) mass using an FRM, FEM, or IMPROVE samplers for some of the rural locations. As of 2020, the NCore network includes a total of 78 stations of which 63 are in urban or suburban areas designed to provide representative population exposure and another 15 rural stations designed to provide regional background and transport information. The NCore network is deployed in all 50 states, District of Columbia, and Puerto Rico with at least one station in each state and two or more stations in larger population states (California, Florida, Illinois, Michigan, New York, North Carolina, Ohio, Pennsylvania, and Texas). Both the STN and NCore networks, which together comprise 75 locations with CSN measurements, are intended to remain in operation indefinitely. The CSN measurements at STN and NCore stations operate on a 1-in-3-day sampling schedule. Another approximately 67 CSN stations, known as supplemental sites, are intended to be temporary locations used to support SIP development and other local or regional monitoring objectives. Supplemental CSN stations typically operate on a 1-in-6-day sampling schedule.

Specific chemical components of $PM_{2.5}$ are also measured through the IMPROVE monitoring program, which supports regional haze characterization and tracks changes in visibility in Class I areas (e.g., large national parks) as well as many other rural and some urban areas. As of 2020, the IMPROVE network includes 110 base network monitoring locations and additional 46 locations operated as IMPROVE protocol sites where a state, local, or tribal monitoring agency has requested participation in the program. These IMPROVE protocol sites operate the same way as the IMPROVE program, but they may serve several monitoring objectives (e.g., SIP development) and are not explicitly tied to the Regional Haze Program. Samplers at IMPROVE stations operate on a 1-in-3-day sampling schedule. Together, the CSN and IMPROVE data provide chemical species information for $PM_{2.5}$ that are critical for use in health and epidemiologic studies to help inform reviews of the primary PM NAAQS and can be used to better understand visibility by calculating light extinction using the IMPROVE algorithm to support reviews of the secondary PM NAAQS.

A.4.4. U.S. EPA Criteria Gas Monitoring

Routine monitoring for criteria gases is performed at State and Local Air Monitoring Stations (SLAMS) using designated FRMs and FEMs. Appendix Table A.4-1 provides information on the FRMs and most widely deployed FEMs for the carbon monoxide (CO), ozone (O_3), nitrogen dioxide (NO_2), and SO_2 criteria gases. The current FRM for measuring concentrations of CO in ambient air is based on nondispersive infrared photometry (NDIR) and is detailed in 40 CFR Part 50 Appendix C (U.S. EPA, 2020a). To date, only one FEM for CO has been designated, and it is based on mercury replacement-ultraviolet (UV) photometry. For O_3, the current FRM is based on the chemiluminescent reaction between O_3 and ethylene or nitric oxide (NO) and is detailed in 40 CFR Part 50 Appendix D (U.S. EPA, 2011b). Currently, FRM instruments based on ethylene chemiluminescence are not available commercially; for this reason, an updated FRM that includes NO chemiluminescence was promulgated in 2015. The most widely used O_3 FEM is based on UV photometry. This method, however, has been shown to have severe interferences in smoke and may result in significant overestimation of O_3 concentrations in smoke-impacted areas (Long et al., In Press). The measurement principle for the NO_2 FRM detailed in 40 CFR Part 50 Appendix F (U.S. EPA, 2011a) consists of the catalytic conversion of NO_2 to NO followed by subsequent detection of the chemiluminescence reaction of NO with O_3. In addition to converting NO_2 to NO prior to detection, this method also converts high oxides of nitrogen (e.g., nitric acid [HNO_3], nitrous acid [HNO_2], particulate nitrate) to NO resulting in a potential overestimation of NO_2 concentrations. FEMs for NO_2 involve direct spectroscopic measurement of NO_2 and the replacement of the catalytic converter with a more specific photolytic converter prior to detection in the chemiluminescence method. Currently, there are two FRMs for measuring concentrations of SO_2 in ambient air. The newer automated FRM, promulgated in 2010, is based on UV fluorescence and is detailed in 40 CFR Part 50 Appendix A-1 (U.S. EPA, 2011c). Prior to promulgation as an FRM, the UV fluorescence method was the most widely used FEM. The second SO_2 FRM is based on the manual wet-chemical pararosaniline method and detailed in 40 CFR Part 50 Appendix A-2 (U.S. EPA, 2020b). Currently, this method is not employed in the routine monitoring of SO_2. For O_3, NO_2, and SO_2, automated open-path FEMs also exist based on differential optical absorption spectroscopy (DOAS). These methods employ long measurement path lengths extending up to 1,000 m.

A.5. Supplemental Information for Chapter 5

No supplemental information.

A.6. Supplemental Information for Chapter 6

A.6.1. Supplemental Information for Section 6.2

Table A.6-1 Study-specific details from U.S.-based epidemiologic studies examining associations between wildfire smoke exposure and respiratory and cardiovascular-related health effects and mortality.

Study; Location; Fire Date	Health Outcomes (Ages)	Exposure Indicator Avg Time	Types of Air Quality Data Used	Exposure Assessment Methodology
ED Visits and Hospital Admissions; Medication Use				
Alman et al. (2016); Colorado; 2012 wildfires (6/5/2012−7/6/2012)	ED visits: asthma and wheeze, URI, pneumonia, bronchitis, COPD, respiratory disease, AMI, IHD, dysrhythmia, CHF, ischemic stroke, PVD, CVD (all; 0−18; 19−64; 65+)	$PM_{2.5}$ (24-h avg; 1-h max)	Modeled	WRF-Chem used to estimate $PM_{2.5}$ concentrations at 12 × 12 km grid cells. Addresses for each patient geocoded and assigned $PM_{2.5}$ concentration from respective grid cell. Model evaluation: Model absolute bias (i.e., average difference between model and monitored $PM_{2.5}$ concentrations), 13 µg/m^3 for 6 monitoring stations around Denver Metro Area, 13 µg/m^3 for 2 stations northeast of Denver, and 19 µg/m^3 for the station east of Denver.

Table A.6-1 (Continued): Study-specific details from U.S.-based epidemiologic studies examining associations between wildfire smoke exposure and respiratory and cardiovascular-related health effects and mortality.

Study; Location; Fire Date	Health Outcomes (Ages)	Exposure Indicator Avg Time	Types of Air Quality Data Used	Exposure Assessment Methodology
DeFlorio-Barker et al. (2019); 692 U.S. counties within 200 km of 123 large fires; >10,000 acres burned (2008−2010)	HA: respiratory; asthma, bronchitis, and wheezing; all CVD (65+)	$PM_{2.5}$ TotCMAQ; $PM_{2.5}$ Tot; $PM_{2.5}$ TotCMAQ-M (24-h avg)	Monitored Modeled	(1) Ambient $PM_{2.5}$ from monitoring stations (>4,000), resulting in countywide averages available for 178 of 692 counties; (2) $PM_{2.5}$ estimated using CMAQ. CMAQ estimated $PM_{2.5}$ at 12 × 12 km grid cells—estimated $PM_{2.5}$ with all emissions ($PM_{2.5}$ TotCMAQ) and without wildfire (No Fire CMAQ [NFCMAQ]). CMAQ data used to calculate area-weighted $PM_{2.5}$ estimates for each county. Difference between CMAQ estimates represented fire-specific $PM_{2.5}$ concentrations ($PM_{2.5}$ FCMAQ). Smoke day = $PM_{2.5}$ FCMAQ > 5 µg/m^3.
Delfino et al. (2009); Southern California; 2003 wildfires (total: 10/1/2003−11/15/2003; prefire: 10/1−10/2; fire: 10/21−10/30; post-fire: 10/31−11/15)	HA: all respiratory, asthma, acute bronchitis, COPD, pneumonia, all CVD, IHD, CHF, dysrhythmia, cerebrovascular and stroke (all; 0−4; 5−19; 20−64; 65−99)	$PM_{2.5}$ (24-h avg)	Monitored	Combination of monitoring data, continuous hourly PM data at colocated or closely located sites, and light extinction from visibility data. Meteorological conditions and smoke data from MODIS at 250-m resolution. For smoke periods, created polygons from smoke-covered areas and measured or estimated $PM_{2.5}$ concentrations from predictive models to assign exposures at ZIP-code centroid.

Table A.6-1 (Continued): Study-specific details from U.S.-based epidemiologic studies examining associations between wildfire smoke exposure and respiratory and cardiovascular-related health effects and mortality.

Study; Location; Fire Date	Health Outcomes (Ages)	Exposure Indicator Avg Time	Types of Air Quality Data Used	Exposure Assessment Methodology
Gan et al. (2017); Washington; 2012 wildfires (7/1/2012−10/31/2012)	HA: all respiratory, asthma, COPD, pneumonia, acute bronchitis, CVD, arrhythmia, cerebrovascular disease, HF, IHD, MI (all; <15; 15−65; 65+)	Smoke $PM_{2.5}$ (24-h avg)	Modeled Satellite	(1) WRF-Chem: Estimated daily $PM_{2.5}$ at 15 × 15 km grid cell, ran additional simulations with biomass burning emissions turned off to estimate nonwildfire smoke $PM_{2.5}$. Model evaluation: slope = 0.67, R^2 = 0.25 (2) Kriging in situ surface monitors: interpolated monitoring data (212 monitors) to 15 × 15 km grid cells. Model evaluation: slope = 0.70, R^2 = 0.69 (3) GWR: estimated $PM_{2.5}$ concentrations at 15 × 15 km grid cells by combining kriged, AOD, and WRF-Chem estimates. Model evaluation: slope = 0.78; R^2 = 0.66. To distinguish wildfire $PM_{2.5}$ for WRF-Chem subtracted out nonsmoke $PM_{2.5}$ produced by WRF-Chem. For kriging and GWR methods, estimated background $PM_{2.5}$ using NOAAs HMS to identify days when wildfire smoke not near a monitor. Smoke plumes in HMS accompanied by estimated $PM_{2.5}$ concentrations from atmospheric models. Calculated median $PM_{2.5}$ concentration for each monitor on nonfire days; these concentrations were interpolated by kriging for each grid cell. These nonfire $PM_{2.5}$ concentrations were subtracted from $PM_{2.5}$ concentrations for each method to estimate $PM_{2.5}$ attributed to smoke.
Gan et al. (2020); Oregon; Douglas Complex Fire Big Windy Complex Fire (5/1/2013−9/30/2013)	HA: asthma Medication use: SABAs pharmacy refills (all; <15; 15−65; 65+)	Smoke $PM_{2.5}$ (24-h avg)	Modeled Satellite	Similar method as Gan et al. (2017), focusing only on the GWR method. Estimated $PM_{2.5}$ concentrations at 15 × 15 km grid cells by combining kriged, AOD, and WRF-Chem estimates. Monitors used in the analysis consisted of both FRM and FEM monitors.

Table A.6-1 (Continued): Study-specific details from U.S.-based epidemiologic studies examining associations between wildfire smoke exposure and respiratory and cardiovascular-related health effects and mortality.

Study; Location; Fire Date	Health Outcomes (Ages)	Exposure Indicator Avg Time	Types of Air Quality Data Used	Exposure Assessment Methodology
Hutchinson et al. (2018); San Diego, CA; 2007 wildfires (9/1/2007−11/29/2007)	ED visits: respiratory index, asthma (0−64)	Wildfire $PM_{2.5}$ (24-h avg)	Modeled	Wildfire emissions from WFEIS were used in HYSPLIT to estimate wildfire $PM_{2.5}$ concentrations at 0.01° grid on an hourly basis. 24-h avg concentrations calculated at the ZIP-code level.
Leibel et al. (2020); San Diego County, CA; Lilac Fire (2011−2017; fire: 12/6/2017−12/17/2017)	ED and urgent care visits: all respiratory (0−19)	$PM_{2.5}$ (24-h avg)	Monitored	24-h avg $PM_{2.5}$ concentrations from 10 fixed site monitors. $PM_{2.5}$ concentrations interpolated using inverse distance interpolation model using stations within 12 miles from each population-weighted ZIP-code centroid, concentrations than averaged and assigned to each ZIP code. Monitors closest to each centroid were given greater weight (weighted using squared inverse distance).
Liu et al. (2017a); 561 western U.S. counties; Wildfire season (May−October, 2004−2009)	HA: all respiratory, all CVD (65+)	Wildfire $PM_{2.5}$; smoke wave day vs. nonsmoke wave day	Monitored Modeled	GEOS-Chem predictions of “all-source $PM_{2.5}$” and “no-fire $PM_{2.5}$” to ~50 × 75 km grid cell. Ground-based or aircraft measurements used to validate model results. Area weighted averaging used to convert gridded predictions to county-level averages. GEOS-Chem predictions biased low during extreme events so model calibrated using county-average monitoring data. Smoke wave defined as 2+ consecutive days of wildfire $PM_{2.5}$ > 20 µg/m^3 (98th percentile, sensitivity analyses focusing on 23 µg/m^3 [98.5 percentile], 28 µg/m^3 [99th percentile], and 37 µg/m^3 [99.5 percentile]).

Table A.6-1 (Continued): Study-specific details from U.S.-based epidemiologic studies examining associations between wildfire smoke exposure and respiratory and cardiovascular-related health effects and mortality.

Study; Location; Fire Date	Health Outcomes (Ages)	Exposure Indicator Avg Time	Types of Air Quality Data Used	Exposure Assessment Methodology
Liu et al. (2017b); 561 western U.S. counties; Wildfire season (May−October, 2004−2009)	HA: respiratory (COPD and respiratory tract infections) (65−75; 75−84; 85+)	Wildfire $PM_{2.5}$; smoke wave day vs. nonsmoke wave day	Monitored Modeled	GEOS-Chem predictions of “all-source $PM_{2.5}$” and “no-fire $PM_{2.5}$” to ~50 × 75 km grid cell. Ground-based or aircraft measurements used to validate model results. Area weighted averaging used to convert gridded predictions to county-level averages. GEOS-Chem predictions biased low during extreme events so model calibrated using county-average monitoring data. Smoke wave defined as 2+ consecutive days of wildfire $PM_{2.5}$ > 37 μg/m^3 (99.5%).
Rappold et al. (2011); 42 North Carolina counties Peet Fire in Pocosin Lakes National Wildlife Refuge; (6/1/2008−7/14/2008)	ED visits: all respiratory, COPD, pneumonia and acute bronchitis, URIs, all CVD, MI, HF, dysrhythmia, respiratory/other chest pain symptoms (all; <65; 65+)	Smoke plume	Satellite	Half hour, AOD at 4 × 4 km averaged over daytime hours to assign county-level exposure. AOD ≥ 1.25 classified as high-density plume. Counties where at least 25% of geographic area of county exceeded AOD threshold were categorized as high-exposure window. Counties with smoke exposure on at least 2 days classified as exposed (18 counties); 23 referent counties (15 exposed 1 day; 8 <1 day).
Rappold et al. (2012); 40 North Carolina counties Peet Fire in Pocosin Lakes National Wildlife Refuge; (6/1/2008−7/14/2008)	ED visits: asthma, CHF (>18; >44)	Wildfire $PM_{2.5}$ (24-h avg)	Modeled Satellite	$PM_{2.5}$ concentrations obtained from NOAA SFS. $PM_{2.5}$ concentrations based on smoke dispersion simulations from HYSPLIT, which relies on satellite information of wildfire location. Hourly $PM_{2.5}$ concentrations at 0.15 × 0.15° (~13.5 km) estimated at lowest 100-m surface area averaged to generate 24-h avg concentrations. Daily averages for each county calculated over county boundaries using Monte Carlo approximation. HYSPLIT data not available for 6/4, underestimating concentrations on that day.

Table A.6-1 (Continued): Study-specific details from U.S.-based epidemiologic studies examining associations between wildfire smoke exposure and respiratory and cardiovascular-related health effects and mortality.

Study; Location; Fire Date	Health Outcomes (Ages)	Exposure Indicator Avg Time	Types of Air Quality Data Used	Exposure Assessment Methodology
Reid et al. (2016); Northern California, 781 ZCTA (air basins: Sacramento Valley, San Francisco Bay Area, Mountain Counties, Lake County, North Central Coast, northern part of San Joaquin Valley) Thousands of wildfires from lightning strikes June 20−21, located in Trinity Alps, Sierra Nevada and Big Sur (prefire: 5/6/2008−6/19/2008; fire: 6/20/2008−7/31/2008; post-fire: 8/1/2008−9/15/2008)	ED visits and HAs: all respiratory, asthma, COPD, pneumonia, all CVD, IHD, CHF, dysrhythmias, hypertension, cerebrovascular disease (all; <20; 65+)	$PM_{2.5}$ (24-h avg)	Monitored Modeled Satellite	Data-adaptive machine learning employing 10-fold CV. Used data from 112 monitoring stations as dependent variable and predictor variables included AOD from GEOS, WRF-Chem model output, various meteorological variables, Julian date, weekend, land use types within 1 km, *X* and *Y* coordinates, elevation, and traffic counts. Used GBM with six most predictive variables for the main model. Estimated exposures at population-weighted centroid of 781 ZCTA. Model evaluation: CV-R^2 = 0.78, CV-RMSE = 1.46 μg/m^3

Table A.6-1 (Continued): Study-specific details from U.S.-based epidemiologic studies examining associations between wildfire smoke exposure and respiratory and cardiovascular-related health effects and mortality.

Study; Location; Fire Date	Health Outcomes (Ages)	Exposure Indicator Avg Time	Types of Air Quality Data Used	Exposure Assessment Methodology
Reid et al. (2019); Northern California, 753 ZIP codes (air basins: Sacramento Valley, San Francisco Bay Area, Mountain Counties, Lake County, North Central Coast, northern part of San Joaquin Valley); Thousands of wildfires from lightning strikes June 20−21, located in Trinity Alps, Sierra Nevada and Big Sur (5/6/2008−9/26/2008)	ED visits: all respiratory, asthma, COPD, pneumonia, acute bronchitis, acute respiratory infections (all)	$PM_{2.5}$ (24-h avg) O_3 (8-h max)	Monitored Modeled Satellite	Used exposure model detailed in Reid et al. (2016). Data-adaptive machine learning employing 10-fold CV. Used data from 112 monitoring stations as dependent variable and predictor variables included AOD from GEOS, WRF-Chem model output, various meteorological variables, Julian date, weekend, land use types within 1 km, *X* and *Y* coordinates, elevation and traffic counts. Used GBM with six most predictive variables for the main model. Estimated exposures at each ZIP-code centroid. Model evaluation: for $PM_{2.5}$, CV-R^2 = 0.78, CV-RMSE = 1.46 µg/m^3. For O_3, CV-R^2 = 0.83
Stowell et al. (2019); Colorado; Wildfire season (April−September, 2011−2014)	ED visits and HAs: all respiratory, asthma, COPD, URIs. bronchitis, IHD, AMI, CHF, dysrhythmia, peripherial/cerebrovasular disease, all CVD (all; 0−18; 19−64; 65+)	Smoke $PM_{2.5}$ (24-h avg)	Monitored Modeled Satellite	Two model approach where data combined from AOD from MAIAC, model simulations from CMAQ, and ground-based $PM_{2.5}$ measurements. Model 1, used random forest modeling to incorporate AOD data, smoke mask, meteorological fields, and land-use variables. Second model used statistical downscaling to calibrate CMAQ $PM_{2.5}$ predictions. Exposure data at 1 × 1 km grid cell. To estimate wildfire smoke $PM_{2.5}$, CMAQ scenarios with and without smoke and dust particles. Difference between scenarios divided by total $PM_{2.5}$ to obtain smoke fraction which was multiplied by total satellite-based $PM_{2.5}$ to obtain smoke $PM_{2.5}$ concentrations. Model evaluation: for CMAQ predictions, CV-R^2 = 0.81; RMSE = 1.85 µg/m. Random forest model improved R^2 from 0.65 to 0.92.

Table A.6-1 (Continued): Study-specific details from U.S.-based epidemiologic studies examining associations between wildfire smoke exposure and respiratory and cardiovascular-related health effects and mortality.

Study; Location; Fire Date	Health Outcomes (Ages)	Exposure Indicator Avg Time	Types of Air Quality Data Used	Exposure Assessment Methodology
Tinling et al. (2016); 28 North Carolina counties with at least one 24-h avg smoke $PM_{2.5}$ concentration > 20 µg/m^3; Pains Bay Fire (5/5/2011−6/19/2011)	ED visits: respiratory/other chest symptoms, all respiratory, asthma, COPD, URI, all CVD, dysrhythmia, HF, hypertension (all; <18; 18−64; 65+)	Wildfire $PM_{2.5}$ (24-h avg)	Modeled	County-level daily wildfire $PM_{2.5}$ estimated from modeled predictions from NOAA SFS.
Wettstein et al. (2018); Eight California air basins (Great Basin Valleys, Lake County, Lake Tahoe, Mountain Counties, North Coast, Northeast Plateau, Sacramento Valley, San Joaquin Valley); 2015 wildfire season (May−September, 2015)	ED visits: all CV, hypertension, IHD, MI, dysrhythmia, HF, PE, all cerebrovascular, ischemic stroke, TIA, all respiratory (19+; 45−64; 65+)	Smoke density	Modeled	Smoke plume data from NOAA HMS, assigning daily maximum density to each ZIP code based on estimated $PM_{2.5}$ concentration data where concentrations within the range of 0−10 µg/m^3 defined as light, 10.5−21.5 µg/m^3 defined as medium, and 22+ µg/m^3 defined as dense.
Out-of-Hospital Events				
Jones et al. (2020); 14 California counties; Wildfires ≥50,000 acres burned or ≥50 days long (May−October, 2015−2017)	OHCA (19+)	Smoke day	Modeled	NOAA HMS used to detect plumes using visual range of satellite images and assigned estimated smoke $PM_{2.5}$ density: light (0−10 µg/m^3); medium (10.5−21.5 µg/m^3); and heavy (>22 µg/m^3). Used geospatial intersect function to assign smoke data at the census block group and then aggregated to census tract; maximum smoke density used to define exposure.

Table A.6-1 (Continued): Study-specific details from U.S.-based epidemiologic studies examining associations between wildfire smoke exposure and respiratory and cardiovascular-related health effects and mortality.

Study; Location; Fire Date	Health Outcomes (Ages)	Exposure Indicator Avg Time	Types of Air Quality Data Used	Exposure Assessment Methodology
Mortality				
Doubleday et al. (2020); Washington; Wildfire season (June−September, 2006−2017)	Total (nonaccidental), cardiovascular, IHD, respiratory, asthma, COPD, pneumonia, cerebrovascular (all)	Smoke day vs. nonsmoke day	Monitored Modeled	4 × 4 km grid cells from AIRACT-4, each grid cell assigned to 1 of 3 AQ monitors closest to each grid cell out of 75 monitors in Washington. Grid cells matched to nearby monitors based on agreement between interpolated and monitored $PM_{2.5}$. Each grid cell then assigned the daily $PM_{2.5}$ monitor concentration. Smoke day defined as days with $PM_{2.5}$ monitor concentrations > 20.4 µg/m^3, with additional criteria if $PM_{2.5}$ concentrations between 9 and 20.4 µg/m^3: (1: 2 of 3 days > 9 µg/m^3; 2: 1 day > 15 µg/m^3; 3: for urban areas at least 50% monitors > 9 µg/m^3).
Magzamen et al. (2021); Front Range Urban Corridor, CO; (May−October, 2010−2015)	Mortality: all respiratory, asthma, COPD, all CVD, HF, cardiac arrest, IHD, MI, cerebrovascular disease (<15, 15−65, 65+)	Wildfire $PM_{2.5}$ (24-h avg)	Monitored Satellite	Ambient $PM_{2.5}$ measurements obtained from 49 U.S. EPA AQS monitors and kriged across 15 km^2 grids. Presence of smoke plumes identified using HMS. Daily concentrations of wildfire $PM_{2.5}$ estimated by subtracting seasonal-median $PM_{2.5}$ concentrations on nonsmoke days for each grid cell where a smoke plume was detected. The approach to estimating wildfire $PM_{2.5}$ is detailed in O'Dell et al. (2019).
Xi et al. (2020); 253 U.S. counties; (2008−2012)	All-cause, cardiac, vascular, infection, other (50+)	Wildfire $PM_{2.5}$ (24-h avg)	Modeled	Ambient $PM_{2.5}$ concentrations were predicted at 12 × 12 km grid cells using CMAQ with and without wildland fire emissions. The difference between the with and without wildland fire emissions represented wildfire-specific $PM_{2.5}$. Hourly concentrations were averaged to calculate a daily county-level 24-h avg $PM_{2.5}$ concentration.

Table A.6-1 (Continued): Study-specific details from U.S.-based epidemiologic studies examining associations between wildfire smoke exposure and respiratory and cardiovascular-related health effects and mortality.

Study; Location; Fire Date	Health Outcomes (Ages)	Exposure Indicator Avg Time	Types of Air Quality Data Used	Exposure Assessment Methodology
Zu et al. (2016); New York, NY; Boston, MA; July 2002 Quebec wildfires (July 2001−2003)	Total (nonaccidental) (all)	$PM_{2.5}$ (24-h avg)	Monitored	Daily average $PM_{2.5}$ concentrations across all monitors in Boston and each borough in New York.

μg/m^3 = micrograms per cubic meter; AIRACT-4 = Air Indicator Report for Public Awareness and Community Tracking; AMI = acute myocardial infarction; AOD = aerosol optical depth; AQ = air quality; AQS = air quality system; avg = average; CHF = congestive heart failure; CMAQ = Community Multiscale Air Quality; COPD = chronic obstructive pulmonary disease; CV = cross-validation; CVD = cardiovascular disease; ED = emergency department; FCMAQ = fused CMAQ; FEM = Federal Equivalent Method; FRM = Federal Reference Method; GBM = Generalized Boosting Model; GEOS-Chem = Goddard Earth Observing System with a global chemical transport model; GWR = geographically weighted regression; h = hour; HA = hospital admission; HF = heart failure; HMS = Hazard Mapping System; HYSPLIT = Hybrid Single-Particle Lagrangian Integrated Trajectories; IHD = ischemic heart disease; km = kilometer; m = meter; MAIAC = Multiangle Implementation of Atmospheric Correction algorithm; max = maximum; MI = myocardial infarction; MODIS = Moderate Resolution Imaging Spectroradiometer; NOAA = National Oceanic and Atmospheric Administration; O_3 = ozone; OHCA = out-of-hospital cardiac arrest; PE = pulmonary embolism; PM = particulate matter; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 μm; $PM_{2.5}$ Tot = monitored $PM_{2.5}$ data; $PM_{2.5}$ TotCMAQ = $PM_{2.5}$ estimated using CMAQ; $PM_{2.5}$ TotCMAQ-M = $PM_{2.5}$ estimated using CMAQ in locations and times with monitoring data; PVD = peripheral vascular disease; RMSE = root-mean-squared error; SABA = short-acting β2 agonist; SFS = Smoke Forecasting System; TIA = transient ischemic attack; URI = upper respiratory tract infection; WFEIS = Wildland Fire Emissions Information System; WRF-Chem = Weather Research and Forecasting Model with Chemistry; ZCTA = ZIP-code tabulation areas.

A.6.2. Supplemental Information for Section 6.3

A literature review was conducted to identify published studies that provide data on individual and community actions to reduce wildfire smoke exposure. The literature review was limited to studies published from 2005 to May 2020 with keywords that included wildfire/prescribed fire and smoke, $PM_{2.5}$, and exposure, along with terms for actions/interventions (e.g., air filtration). Although several hundred published studies were identified with the search terms, after reviewing the titles and abstracts only 243 publications were determined to be relevant to wildfire or prescribed fire smoke exposure. Of those, 26 specifically addressed some aspect of smoke exposure mitigation and were included in the discussion within Section 6.3 of Chapter 6.

To be most informative in assessing the potential implications of public health messaging campaigns that attempt to reduce/mitigate population exposure to wildfire smoke around the case study areas, studies were limited to those conducted in the U.S. and Canada, with a few exceptions. Only three publications were identified that surveyed the likelihood of taking action to reduce wildfire smoke exposure in North America, so the literature review was expanded to include studies that were published before 2005 and from other parts of the world. Two additional studies were included, one published in 2002 conducted in North America and one conducted in Australia. The only published study with data on the effectiveness of staying indoors with windows and doors closed was conducted in Australia and also was included. An additional study that was not identified in the literature review but discovered in peer review met the search criteria and was also included.

Table A.6-2 Likelihood of taking actions to reduce wildfire smoke exposure reported in recent studies.

Study	Exposure Reduction Action	Percent Population Taking the Action	Population Characteristics	Outdoor PM Concentration. µg/m^3
Behavioral Changes—Avoid Outdoor Activity				
Rappold et al. (2019)	Avoided outdoor activity	61	Smoke Sense application users with no reported health history and no symptoms	NR 1
		6	Smoke Sense application users reported health history	
		90	Smoke Sense application users experiencing four or more symptoms	
Jones et al. (2016)	Avoided outdoor recreation/exercise	42	Residents in Albuquerque 3 yr after the Wallow Fire in 2011 in southeastern Arizona	Maximum $PM_{2.5}$ (hourly) = 70.5
Richardson et al. (2012)	Avoided outdoor recreation	78	Residents of five cities within the vicinity of Station Fire in southern California	Maximum $PM_{2.5}$ (daily) = 82.9 (hourly) = 223
Sugerman et al. (2012)	Did not play sports outside	88	Residents of San Diego County during the 2007 San Diego fires	$PM_{2.5}$ 2 (daily) >128 for 10 days Maximum $PM_{2.5}$ (daily) = 803.1 Mean $PM_{2.5}$ (daily) = 89 (Hutchinson et al., 2018)

Table A.6-2 (Continued): Likelihood of taking actions to reduce wildfire smoke exposure reported in recent studies.

Study	Exposure Reduction Action	Percent Population Taking the Action	Population Characteristics	Outdoor PM Concentration. μg/m³
Kolbe and Gilchrist (2009)	Reduced outdoor activities	54	Residents of Albury, New South Wales, Australia during 2003 bush fires	$PM_{2.5}$ 2 (daily) >128 for 9 days Maximum $PM_{2.5}$ 2 (daily) = 597
Behavioral Changes—Stayed Inside/Closed Doors and Windows				
Rappold et al. (2019)	Stayed indoors	68	Smoke Sense application users with no reported health history and no symptoms	NR 1
		70	Smoke Sense application users reported health history	
		90	Smoke Sense application users experiencing four or more symptoms	
Jones et al. (2016)	Stayed indoors	55	Residents in Albuquerque 3 yr after the Wallow Fire in 2011 in southeastern Arizona	Maximum $PM_{2.5}$ (hourly) = 70.5
Richardson et al. (2012)	Stayed inside	73	Residents of five cities within the vicinity of Station Fire in southern California	Maximum $PM_{2.5}$ (daily) = 82.9 (hourly) = 223
Sugerman et al. (2012)	Stayed inside	59	Residents of San Diego County during the 2007 San Diego fires	$PM_{2.5}$ 2 (daily) >128 for 10 days Maximum $PM_{2.5}$ (daily) = 803.1 Mean $PM_{2.5}$ (daily) = 89 (Hutchinson et al., 2018)
	Kept windows closed	76		

Table A.6-2 (Continued): Likelihood of taking actions to reduce wildfire smoke exposure reported in recent studies.

Study	Exposure Reduction Action	Percent Population Taking the Action	Population Characteristics	Outdoor PM Concentration. μg/m^3
Kolbe and Gilchrist (2009)	Closed windows and doors	44	Residents of Albury, New South Wales, Australia during 2003 bush fires	$PM_{2.5}$ 2 (daily) >128 for 9 days Maximum $PM_{2.5}$ 2 (daily) = 597
Mott et al. (2002)	Stayed inside	79	Residents of Hoopa, CA during 1999 wildfire that were aware of PSAs on smoke impacts	$PM_{2.5}$ 2 (daily) >128 for 15 days $PM_{2.5}$ 2 (daily) >425 for 2 days
Behavioral Changes—Evacuated				
Rappold et al. (2019)	Left area	30	Smoke Sense application users with no reported health history and no symptoms	NR 1
		40	Smoke Sense application users reported health history	
		65	Smoke Sense application users experiencing four or more symptoms	
Jones et al. (2016)	Evacuated	5	Residents in Albuquerque 3 yr after the Wallow Fire in 2011 in southeastern Arizona	Maximum $PM_{2.5}$ (hourly) = 70.5
Richardson et al. (2012)	Evacuated	5.6	Residents of five cities within the vicinity of Station Fire in southern California	Maximum $PM_{2.5}$ (daily) = 82.9 (hourly) = 223
Kolbe and Gilchrist (2009)	Travelled out of area	14	Residents of Albury, New South Wales, Australia during 2003 bush fires	$PM_{2.5}$ 2 (daily) > 128 for 9 days Maximum $PM_{2.5}$ 2 (daily) = 597
		12	Residents of Albury, New South Wales Australia during 2003 bush fires who saw, heard, or read smoke advisory	

Table A.6-2 (Continued): Likelihood of taking actions to reduce wildfire smoke exposure reported in recent studies.

Study	Exposure Reduction Action	Percent Population Taking the Action	Population Characteristics	Outdoor PM Concentration. µg/m^3
Mott et al. (2002)	Evacuated area during smoke	48	Residents of Hoopa, CA during 1999 wildfire	$PM_{2.5}$ 2 (daily) >128 for 15 days
		35	Residents of Hoopa, CA during 1999 wildfire that were aware of public service announcements on smoke impacts	$PM_{2.5}$ 2 (daily) >425 for 2 days
		44	Residents of Hoopa, CA during 1999 wildfire without a preexisting condition	
		58	Residents of Hoopa, CA during 1999 wildfire with a preexisting condition	
Exposure Reduction—Ran HVAC system				
Richardson et al. (2012)	Ran air conditioner more than usual	60	Residents of five cities within the vicinity of Station Fire in southern California	Maximum $PM_{2.5}$ (daily) = 82.9 (hourly) = 223
Sugerman et al. (2012)	Used home air conditioner	16	Residents of San Diego County during the 2007 San Diego fires	$PM_{2.5}$ 2 (daily) >128 for 10 days Maximum $PM_{2.5}$ (daily) = 803.1 Mean $PM_{2.5}$ (daily) = 89 (Hutchinson et al., 2018)

Table A.6-2 (Continued): Likelihood of taking actions to reduce wildfire smoke exposure reported in recent studies.

Study	Exposure Reduction Action	Percent Population Taking the Action	Population Characteristics	Outdoor PM Concentration. µg/m^3
Exposure Reduction—Used Air Cleaner				
Rappold et al. (2019)	Ran an air cleaner	30	Smoke Sense application users with no reported health history and no symptoms	NR 1
		52	Smoke Sense application users reported health history	
		86	Smoke Sense application users experiencing four or more symptoms	
Jones et al. (2016)	Used air filter/cleaner	16	Residents in Albuquerque 3 yr after the Wallow Fire in 2011 in southeastern Arizona	Maximum $PM_{2.5}$ (hourly) = 70.5
Richardson et al. (2012)	Used an air cleaner	21	Residents of five cities within the vicinity of Station Fire in southern California	Maximum $PM_{2.5}$ (daily) = 82.9 (hourly) = 223
Sugerman et al. (2012)	Used HEPA cleaner	10	Residents of San Diego County during the 2007 San Diego fires	$PM_{2.5}$ 2 (daily) >128 for 10 days Maximum $PM_{2.5}$ (daily) = 803.1 Mean $PM_{2.5}$ (daily) = 89 (Hutchinson et al., 2018)
Mott et al. (2002)	Used HEPA cleaner	34%	Residents of Hoopa, CA during 1999 wildfire	$PM_{2.5}$ 2 (daily) >128 for 15 days $PM_{2.5}$ 2 (daily) >425 for 2 days
		26%	Residents of Hoopa, CA during 1999 wildfire without a preexisting condition	
		52%	Residents of Hoopa, CA during 1999 with a preexisting condition	

Table A.6-2 (Continued): Likelihood of taking actions to reduce wildfire smoke exposure reported in recent studies.

Study	Exposure Reduction Action	Percent Population Taking the Action	Population Characteristics	Outdoor PM Concentration. μg/m³
Exposure Reduction—Used Respirator/Mask				
Rappold et al. (2019)	Wore a respirator	14	Smoke Sense application users with no reported health history and no symptoms	NR 1
		24	Smoke Sense application users reported health history	
		80	Smoke Sense application users experiencing four or more symptoms	
Jones et al. (2016)	Covered face with mask	7	Residents in Albuquerque 3 yr after the Wallow Fire in 2011 in southeastern Arizona	Maximum $PM_{2.5}$ (hourly) = 70.5
Richardson et al. (2012)	Wore a mask	7	Residents of five cities within the vicinity of Station Fire in southern California	Maximum $PM_{2.5}$ (daily) = 82.9 (hourly) = 223
Mott et al. (2002)	Wore an N95 mask	10	Residents of Hoopa, CA during 1999 wildfire	$PM_{2.5}$ 2 (daily) >128 for 15 days $PM_{2.5}$ 2 (daily) >425 for 2 days
Symptom Mitigation—Took Medicine				
Kolbe and Gilchrist (2009)	Increased regular medication	1.6	Residents of Albury, New South Wales, Australia during 2003 bush fires	$PM_{2.5}$ 2 (daily) >128 for 9 days Maximum $PM_{2.5}$ 2 (daily) = 597
		2.3	Residents of Albury, New South Wales, Australia during 2003 bush fires who saw, heard, or read smoke advisory	
Richardson et al. (2012)	Took medicine	13	Residents of five cities within the vicinity of Station Fire in southern California	Maximum $PM_{2.5}$ (daily) = 82.9 (hourly) = 223

Table A.6-2 (Continued): Likelihood of taking actions to reduce wildfire smoke exposure reported in recent studies.

Study	Exposure Reduction Action	Percent Population Taking the Action	Population Characteristics	Outdoor PM Concentration. μg/m^3
Messaging Effectiveness				
Mott et al. (2002)	Took exposure reduction action due to PSA	66	Residents of Hoopa, CA during 1999 wildfire	$PM_{2.5}$ 2 (daily) >128 for 15 days $PM_{2.5}$ 2 (daily) >425 for 2 days
Kolbe and Gilchrist (2009)	Changed behavior due to messaging	43	Residents of Albury, New South Wales, Australia during 2003 bush fires	$PM_{2.5}$ 2 (daily) >128 for 9 days Maximum $PM_{2.5}$ 2 (daily) = 597
Sugerman et al. (2012)	Took at least one action from messaging	98	Residents of San Diego County during the 2007 San Diego fires	$PM_{2.5}$ 2 (daily) > 128 for 10 days Maximum $PM_{2.5}$ (daily) = 803.1 Mean $PM_{2.5}$ (daily) = 89 (Hutchinson et al., 2018)
	Took all actions from messaging	27		

μg/m^3 = micrograms per cubic meter; HEPA = high-efficiency particulate air; HVAC = heating, ventilation, and air conditioning; NR = $PM_{2.5}$ concentrations not reported; PM = particulate matter; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 μm; PM_{10} = particulate matter with a nominal mean aerodynamic diameter less than or equal to 10 μm; PSA = public service announcement; yr = year.

Note: $PM_{2.5}$ calculated assuming 85% of PM_{10} concentration (Lutes, 2014).

Table A.6-3 Percent reduction in $PM_{2.5}$ concentrations associated with actions/interventions reported in recent studies.

Study	Intervention	Percent $PM_{2.5}$ Reduction	Description of Comparison	Outdoor $PM_{2.5}$ Concentration. µg/m^3
Residential Measurement Studies				
U.S. EPA (2018) Table 4 [from Park et al. (2017)]	Portable air cleaner with HEPA filter	43[a]	Eight homes in California with HEPA filters with activated carbon; eight homes without; 12-week intervention	
U.S. EPA (2018) Table 5 [from Allen et al. (2011)]	Portable air cleaner with HEPA filter	60	25 homes in British Columbia with HEPA filters during half of study period and without HEPA filters during rest; 1-week intervention	11.2 (mean)
U.S. EPA (2018) Table 5 [from Weichenthal et al. (2013)]	Portable air cleaner with electrostatic precipitator	61	20 homes in Manitoba, Canada with Filtrete electrostatic filters during half of study period and without filters during rest; 1-week intervention	42.5
U.S. EPA (2018) Table 5 [from Kajbafzadeh et al. (2015)]	Portable air cleaner with HEPA filter	40	20 woodsmoke impacted homes in Vancouver with HEPA filters during half of study period and without HEPA filters during rest; 1-week intervention	5.0 HEPA off 3.9 HEPA on
Barn et al. (2008) Table 2	Portable air cleaner with HEPA filter	57.7	26 homes in British Columbia during forest fire (summer) or wood smoke (winter); 1 day each with and without filter	3−91 (S) <4−189 (W)
Henderson et al. (2005) Figure 7	ESP air cleaners	63−88	Eight homes in Colorado during wildfire or prescribed fire; paired homes with and without air cleaners	6−38 (outside during fire)
Singer et al. (2017) Table 2	HVAC with MERV13 at return (E)	88−93	Single test house in California; reference system = HVAC MERV4 at return had 65−75% reduction	6−16 (S) 8−31 (F/W)
	HVAC continuous with MERV16 at supply (C)	96−97		

Table A.6-3 (Continued): Percent reduction in $PM_{2.5}$ concentrations associated with actions/interventions reported in recent studies.

Study	Intervention	Percent $PM_{2.5}$ Reduction	Description of Comparison	Outdoor $PM_{2.5}$ Concentration. µg/m^3
	Portable air cleaner with HEPA	90−94		
Alavy and Siegel (2020) Figure 3	HVAC with MERV8, MERV11, MERV14	16 MERV8 36 MERV8E 45 MERV11E 41 MERV14E	21 residences in Toronto; in situ effectiveness compared to system off or no filter	
Reisen et al. (2019) Table 2	Window/door open	12[b]	Home: ~98 yr old, 8 windows, 4 doors; air conditioner (H10)	335.8 (h max)
	Windows open	56.7[b]	Home: 8 yr old, 16 windows, 4 doors; air conditioner (H11)	386.5 (h max)
	Windows/door open	38.5[b]	Home: 28 yr old, 4 windows, 2 doors; air conditioner (H12)	56.1 (h max)
	Closed	48.5[b]	Home: ~30 yr old, 8 windows, 3 doors; air conditioner (H16)	56.0 (h max)
	Windows open 20−60% of time during wood smoke event	67.5−75.7[b]	Home: ~23 yr old, 14 windows, 4 doors; air conditioner (H21)	

Table A.6-3 (Continued): Percent reduction in $PM_{2.5}$ concentrations associated with actions/interventions reported in recent studies.

Study	Intervention	Percent $PM_{2.5}$ Reduction	Description of Comparison	Outdoor $PM_{2.5}$ Concentration. $\mu g/m^3$
Residential Modeling Studies				
Fisk and Chan (2017b) Table 5	HVAC fan (continuous), low efficiency filter (i1)	24	Comparator: home with intermittent operating HVAC system with typical low-efficiency particle filter (home B1 mean = 29.2 $\mu g/m^3$)	56.9
	HVAC fan (continuous), high efficiency filter (i2)	47		
	HVAC fan (intermittent), high efficiency filter (i3)	11		
	HVAC fan (continuous), low efficiency filter, continuous portable air cleaner (i4)	51		
	HVAC fan (continuous), high efficiency filter, continuous portable air cleaner (i5)	62		
	No forced air system, continuous portable air cleaner (i6)	45	Comparator: home with no HVAC, may have moderate window AC (home B2 mean = 31.9 $\mu g/m^3$)	
Office Building Measurement Studies				
Stauffer et al. (2020) Table 4	Portable air cleaner	73 (day) 92 (night)	Offices with and without portable air cleaners during day and night during wildfire season	17.5
Pantelic et al. (2019) Figure 5 and text page 10	HVAC system with filters	60[c]	Office building with HVAC system with filters (MERV8, gas-phase filter, and MERV13) compared with an office building with natural ventilation system	70 (4th St) 53 (Wurster)

Table A.6-3 (Continued): Percent reduction in $PM_{2.5}$ concentrations associated with actions/interventions reported in recent studies.

Study	Intervention	Percent $PM_{2.5}$ Reduction	Description of Comparison	Outdoor $PM_{2.5}$ Concentration. $\mu g/m^3$
Modeling Studies Residential and Other Buildings				
Fisk and Chan (2017a) Table S8 and S9.	Home HVAC MERV6 running 30% of time (i1a, i1b)	2−4	Comparison: Home: HVAC MERV 6 operating 15−20% of the time, no HEPA portable air cleaner Other buildings: MERV8	11.4 (LA) 10.0 (NJ) 10.4 (TX)
	Home HVAC MERV6 running 30−40% of time, HEPA portable air cleaner (i5a, i5b)	27−31		
	Home: HEPA portable air cleaner (i4)	26−30		
	Homes: HVAC MERV6 running 15−20% of time, HEPA portable air cleaner Other buildings: MERV13 (i8)	7−9	Comparison: Home: HVAC MERV 6 operating 15−20% of the time, no HEPA portable air cleaner Other buildings: MERV8	

$\mu g/m^3$ = micrograms per cubic meter; AC = air conditioning; ESP = electrostatic precipitator; F = fall; h = hour; HEPA = high-efficiency particulate air; HVAC = heating, ventilation, and air conditioning; I/O = indoor/outdoor; max = maximum; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm; S = summer; W = winter.

[a]Based on average $PM_{2.5}$ concentration difference between groups.

[b]Based on maximum hourly $PM_{2.5}$.

[c]Calculated as percent difference in median I/O ratios.

A.7. Supplemental Information for Chapter 7

A.7.1. Supplemental Tables for Chapter 7

Table A.7-1 Crosswalk between LANDFIRE existing vegetation types (LANDFIRE, 2014 Existing Vegetation Type) within the four scenario areas and an assigned Fuel Characteristic Classification System (FCCS) fuelbed. Fuelbed descriptions for each of the base fuelbeds can be found within the Fuel and Fire Tools (https://www.fs.usda.gov/pnw/tools/fuel-and-fire-tools-fft).

EVT ID	EVT Name	FCCS ID	Fuelbed Name
11	Ba Open Water	0	Barren
31	Bab Barren	0	Barren
2001	Sps Inter-Mountain Basins Sparsely Vegetated Systems	0	Barren
2002	Sps Mediterranean California Sparsely Vegetated Systems	0	Barren
2003	Sps North Pacific Sparsely Vegetated Systems	0	Barren
2006	Sps Rocky Mountain Alpine/Montane Sparsely Vegetated Systems	0	Barren
2011	Tr Rocky Mountain Aspen Forest and Woodland	42	Quaking aspen/Engelmann spruce forest
2027	Tr Mediterranean California Dry-Mesic Mixed Conifer Forest and Woodland	37	Ponderosa pine-Jeffrey pine forest
2028	Tr Mediterranean California Mesic Mixed Conifer Forest and Woodland	214	Giant sequoia-white fir-sugar pine forest
2030	Tr Mediterranean California Lower Montane Conifer Forest and Woodland	16	Jeffrey pine-ponderosa pine-Douglas fir-CA black oak forest

Table A.7-1 (Continued): Crosswalk between LANDFIRE existing vegetation types (LANDFIRE, 2014 Existing Vegetation Type) within the four scenario areas and an assigned Fuel Characteristic Classification System (FCCS) fuelbed. Fuelbed descriptions for each of the base fuelbeds can be found within the Fuel and Fire Tools (https://www.fs.usda.gov/pnw/tools/fuel-and-fire-tools-fft).

EVT ID	EVT Name	FCCS ID	Fuelbed Name
2032	Tr Mediterranean California Red Fir Forest	17	Red Fir Forest
2033	Tr Mediterranean California Subalpine Woodland	12	Red fir-mountain hemlock-lodgepole pine-western white pine forest
2037	Tr North Pacific Maritime Dry-Mesic Douglas Fir-Western Hemlock Forest	8	Western hemlock-Douglas fir-western red cedar/vine maple forest
2041	Tr North Pacific Mountain Hemlock Forest	238	Pacific silver fir-mountain hemlock forest
2042	Tr North Pacific Mesic Western Hemlock-Silver Fir Forest	238	Pacific silver fir-mountain hemlock forest
2043	Tr Mediterranean California Mixed Evergreen Forest	37	Ponderosa pine-Jeffrey pine forest
2044	Tr Northern California Mesic Subalpine Woodland	12	Red fir-mountain hemlock-lodgepole pine-western white pine forest
2045	Tr Northern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest	52	Douglas fir-Pacific ponderosa pine/oceanspray forest
2053	Tr Northern Rocky Mountain Ponderosa Pine Woodland and Savanna	53	Pacific ponderosa pine forest
2056	Tr Rocky Mountain Subalpine Mesic-Wet Spruce-Fir Forest and Woodland	59	Subalpine fir-Engelmann spruce-Douglas fir-lodgepole pine forest
2058	Tr Sierra Nevada Subalpine Lodgepole Pine Forest and Woodland	12	Red fir-mountain hemlock-lodgepole pine-western white pine forest
2068	Sh North Pacific Dry and Mesic Alpine Dwarf-Shrubland or Fell-Field or Meadow	319	Pacific silver fir-Sitka alder forest
2080	Sh Inter-Mountain Basins Big Sagebrush Shrubland	233	Sagebrush shrubland
2083	Sh North Pacific Avalanche Chute Shrubland	319	Pacific silver fir-Sitka alder forest

Table A.7-1 (Continued): Crosswalk between LANDFIRE existing vegetation types (LANDFIRE, 2014 Existing Vegetation Type) within the four scenario areas and an assigned Fuel Characteristic Classification System (FCCS) fuelbed. Fuelbed descriptions for each of the base fuelbeds can be found within the Fuel and Fire Tools (https://www.fs.usda.gov/pnw/tools/fuel-and-fire-tools-fft).

EVT ID	EVT Name	FCCS ID	Fuelbed Name
2084	Sh North Pacific Montane Shrubland	237	Huckleberry heather shrubland
2098	Sh California Montane Woodland and Chaparral	44	Scrub oak chaparral shrubland
2106	Sh Northern Rocky Mountain Montane-Foothill Deciduous Shrubland	331	Sitka alder-salmonberry shrubland
2125	Sh Inter-Mountain Basins Big Sagebrush Steppe	233	Sagebrush shrubland
2138	He North Pacific Montane Grassland	315	Showy sedge-black alpine sedge grassland
2139	He Northern Rocky Mountain Lower Montane-Foothill-Valley Grassland	506	Idaho fescue-California oatgrass grassland
2145	He Rocky Mountain Subalpine-Montane Mesic Meadow	530	Temperate Pacific subalpine-montane wet meadow
2152	Tr California Montane Riparian Systems	319	Pacific silver fir-Sitka alder forest
2154	Tr Inter-Mountain Basins Montane Riparian Systems	319	Pacific silver fir-Sitka alder forest
2167	Tr Rocky Mountain Poor-Site Lodgepole Pine Forest	22	Mature lodgepole pine forest
2171	He North Pacific Alpine and Subalpine Dry Grassland	315	Showy sedge-black alpine sedge grassland
2172	Tr Sierran-Intermontane Desert Western White Pine-White Fir Woodland	273	Engelmann spruce-Douglas fir-white fir-ponderosa pine forest
2173	Tr North Pacific Wooded Volcanic Flowage	28	Ponderosa pine savanna
2174	Tr North Pacific Dry-Mesic Silver Fir-Western Hemlock-Douglas Fir Forest	8	Western hemlock-Douglas fir-western red cedar/vine maple forest
2181	He Introduced Upland Vegetation-Annual Grassland	57	Wheatgrass-cheatgrass grassland
2182	He Introduced Upland Vegetation-Perennial Grassland and Forbland	57	Wheatgrass-cheatgrass grassland

Table A.7-1 (Continued): Crosswalk between LANDFIRE existing vegetation types (LANDFIRE, 2014 Existing Vegetation Type) within the four scenario areas and an assigned Fuel Characteristic Classification System (FCCS) fuelbed. Fuelbed descriptions for each of the base fuelbeds can be found within the Fuel and Fire Tools (https://www.fs.usda.gov/pnw/tools/fuel-and-fire-tools-fft).

EVT ID	EVT Name	FCCS ID	Fuelbed Name
2902	Bau Developed-Low Intensity	0	Barren
2905	Bau Developed-Roads	0	Barren
2914	Dtc Urban Evergreen Forest	22	Mature lodgepole pine forest
2916	Dgr Urban Herbaceous	66	Bluebunch wheatgrass-bluegrass grassland
2917	Dsh Urban Shrubland	401	Holly-privet shrubland
2926	Dsh Developed Ruderal Shrubland	401	Holly-privet shrubland

EVT = existing vegetation type; FCCS = Fuel Characteristic Classification System.

Table A.7-2 Disturbance update rules for past prescribed burns and wildfires.

FCCS ID	Fuelbed Name	Recent Low-Severity Prescribed Burn	Past Wildfire 0−5 yr	Past Wildfire 5−10 yr
8	Western hemlock-Douglas fir-western red cedar/vine maple forest	8_111	8_132	8_133
12	Red fir-mountain hemlock-lodgepole pine-western white pine forest	12_111	12_132	12_133
16	Jeffrey pine-ponderosa pine-Douglas fir-CA black oak forest	16_111	16_132	16_133
17	Red Fir Forest	17_111	17_132	17_133
22	Mature lodgepole pine forest	22_111	22_132	22_133
28	Ponderosa pine savanna	28_111	28_132	28_133
37	Ponderosa pine-Jeffrey pine forest	37_111	37_132	37_133
42	Quaking aspen/Engelmann spruce forest	42_111	42_132	42_133
44	Scrub oak chaparral shrubland	44_111	44_132	44_133
52	Douglas fir-Pacific ponderosa pine/oceanspray forest	52_111	52_132	52_133
53	Pacific ponderosa pine forest	53_111	53_132	53_133
57	Wheatgrass-cheatgrass grassland	57_111	57_132	57_133
59	Subalpine fir-Engelmann spruce-Douglas fir-lodgepole pine forest	59_111	59_132	59_133
66	Bluebunch wheatgrass-bluegrass grassland	66_111	66_132	66_133
214	Giant sequoia-white fir-sugar pine forest	214_111	214_132	214_133
233	Sagebrush shrubland	233_111	233_132	233_133
237	Huckleberry heather shrubland	237_111	237_132	237_133
238	Pacific silver fir-mountain hemlock forest	238_111	238_132	238_133
273	Engelmann spruce-Douglas fir-white fir-ponderosa pine forest	273_111	273_132	273_133
315	Showy sedge-black alpine sedge grassland	315_111	315_132	315_133
319	Pacific silver fir-Sitka alder forest	319_111	319_132	319_133

Table A.7-2 (Continued): Disturbance update rules for past prescribed burns and wildfires.

FCCS ID	Fuelbed Name	Recent Low-Severity Prescribed Burn	Past Wildfire 0−5 yr	Past Wildfire 5−10 yr
331	Sitka alder-salmonberry shrubland	331_111	331_132	331_133
401	Holly-privet shrubland	401_111	401_132	401_133
506	Idaho fescue-California oatgrass grassland	506_111	506_132	506_133
530	Temperate Pacific subalpine-montane wet meadow	530_111	530_132	530_133

FCCS = Fuel Characteristic Classification System; yr = year.

Table A.7-3 Speciation profiles used for converting volatile organic compound (VOC) and $PM_{2.5}$ to model species.

Prescribed Fires			
Profile ID	Pollutant	CB6 group	Mass Fraction
95423	TOG	ALD2_PRIMARY	0.0223
95423	TOG	FORM_PRIMARY	0.0445
95423	TOG	SOAALK	0.009503
95423	TOG	ACET	0.0115
95423	TOG	ALD2	0.0223
95423	TOG	ALDX	0.036
95423	TOG	BENZ	0.005976
95423	TOG	CH4	0.0968
95423	TOG	ETH	0.0275
95423	TOG	ETHA	0.0132
95423	TOG	ETHY	0.006216
95423	TOG	ETOH	0.004761
95423	TOG	FORM	0.0445
95423	TOG	IOLE	0.0107
95423	TOG	ISOP	0.001913
95423	TOG	KET	0.005659
95423	TOG	MEOH	0.0501
95423	TOG	NAPH	0.006475
95423	TOG	NVOL	0.004562
95423	TOG	OLE	0.0553
95423	TOG	PAR	0.3296
95423	TOG	PRPA	0.004821
95423	TOG	TERP	0.0129
95423	TOG	TOL	0.0476
95423	TOG	UNR	0.1636
95423	TOG	XYLMN	0.038
Prescribed Fires: VOC->TOG factor = 1.14341685			

Wild Fires			
Profile ID	Pollutant	CB6 group	Mass Fraction
95424	TOG	ACET	0.0115
95424	TOG	ALD2	0.0224
95424	TOG	ALDX	0.0353
95424	TOG	BENZ	0.006012
95424	TOG	CH4	0.1095
95424	TOG	ETH	0.0273
95424	TOG	ETHA	0.0161
95424	TOG	ETHY	0.005622
95424	TOG	ETOH	0.004785
95424	TOG	FORM	0.0336
95424	TOG	IOLE	0.0108
95424	TOG	ISOP	0.001929
95424	TOG	KET	0.005694
95424	TOG	MEOH	0.0308
95424	TOG	NAPH	0.006505
95424	TOG	NVOL	0.004606
95424	TOG	OLE	0.0507
95424	TOG	PAR	0.343
95424	TOG	PRPA	0.007611
95424	TOG	TERP	0.013
95424	TOG	TOL	0.0492
95424	TOG	UNR	0.1647
95424	TOG	XYLMN	0.0393
Wildfires: VOC->TOG factor = 1.16417442			

Wild and Prescribed Fires			
Profile ID	Pollutant	CMAQ specie	Mass Fraction
3766AE6	PM2_5	PNO3	2.810E-04
3766AE6	PM2_5	POC	4.688E-01
3766AE6	PM2_5	PSI	6.200E-04
3766AE6	PM2_5	PNA	1.220E-04
3766AE6	PM2_5	PSO4	1.332E-03
3766AE6	PM2_5	PTI	1.500E-05
3766AE6	PM2_5	PNH4	1.105E-03
3766AE6	PM2_5	PEC	3.227E-02
3766AE6	PM2_5	PK	1.203E-03
3766AE6	PM2_5	PNCOM	3.281E-01
3766AE6	PM2_5	PAL	1.540E-04
3766AE6	PM2_5	PCA	3.693E-03
3766AE6	PM2_5	PCL	2.070E-03
3766AE6	PM2_5	PFE	1.800E-04
3766AE6	PM2_5	PMG	1.790E-04
3766AE6	PM2_5	PMN	5.000E-06
3766AE6	PM2_5	PMOTHR	1.599E-01

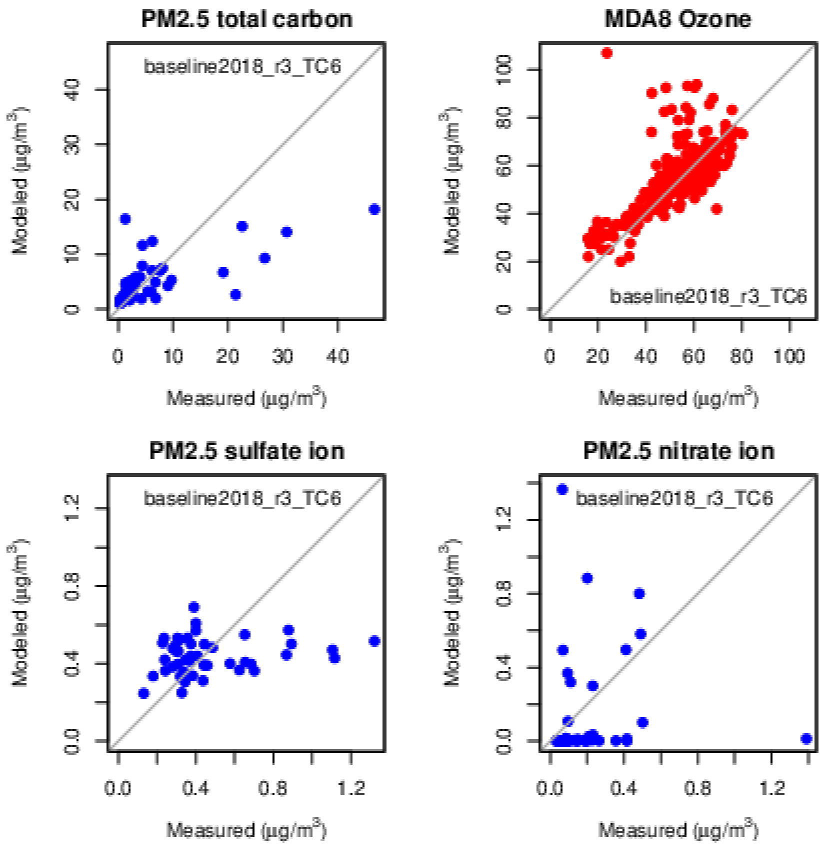
CMAQ = Community Multiscale Air Quality; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 μm; TOG = total organic gases.

Table A.7-4 Model performance metrics estimated for ozone and major speciated components of $PM_{2.5}$. Performance metrics include mean bias, mean error, normalized mean bias, normalized mean error, and correlation coefficient.

Modeling Period	Specie	Data subset	N	Mean Bias	Mean Error	Normalized Mean Bias (%)	Normalized Mean Error (%)	r^2
July 2018	MDA8 ozone	None (all data)	273	2.24	7.39	4.32	14.26	0.52
	MDA8 ozone	Modeled MDA8 O3 > 60 ppb	79	7.97	11.64	12.75	18.62	0.07
	MDA8 ozone	Observed MDA8 O3 > 60 ppb	89	-3.62	7.27	-5.42	10.90	0.16
	PM2.5 nitrate ion	None (all data)	46	-0.07	0.22	-34.82	110.21	0.01
	PM2.5 sulfate ion	None (all data)	46	-0.03	0.19	-5.96	41.28	0.04
	PM2.5 total carbon	None (all data)	46	-1.20	4.29	-18.34	65.85	0.43
Sep 2019	MDA8 ozone	None (all data)	533	3.79	5.58	10.89	16.03	0.57
	MDA8 ozone	Modeled MDA8 O3 > 60 ppb						
	MDA8 ozone	Observed MDA8 O3 > 60 ppb						
	PM2.5 nitrate ion	None (all data)	82	-0.03	0.10	-25.63	84.24	0.30
	PM2.5 sulfate ion	None (all data)	82	0.23	0.26	73.66	84.20	0.15
	PM2.5 total carbon	None (all data)	80	0.87	1.11	89.62	114.36	0.12
Feb/Mar 2019	MDA8 ozone	None (all data)	576	6.03	7.11	15.65	18.44	0.16
	MDA8 ozone	Modeled MDA8 O3 > 60 ppb						
	MDA8 ozone	Observed MDA8 O3 > 60 ppb						
	PM2.5 nitrate ion	None (all data)	163	-0.14	0.15	-80.44	85.33	0.14
	PM2.5 sulfate ion	None (all data)	167	0.30	0.31	164.85	166.27	0.63
	PM2.5 total carbon	None (all data)	169	0.15	0.36	32.18	78.73	0.30
Aug/Sep 2015	MDA8 ozone	None (all data)	11,510	0.64	6.53	1.29	13.08	0.66
	MDA8 ozone	Modeled MDA8 O3 > 60 ppb	2,266	1.26	8.41	1.88	12.54	0.22
	MDA8 ozone	Observed MDA8 O3 > 60 ppb	2,660	-6.47	8.76	-9.24	12.50	0.31
	PM2.5 nitrate ion	None (all data)	720	-0.39	0.47	-70.56	83.73	0.20
	PM2.5 sulfate ion	None (all data)	722	-0.01	0.33	-0.92	44.37	0.25
	PM2.5 total carbon	None (all data)	536	-0.57	1.82	-18.92	59.91	0.37
Aug/Sep 2010	MDA8 ozone	None (all data)	11,764	7.33	10.09	14.31	19.69	0.59
	MDA8 ozone	Modeled MDA8 O3 > 60 ppb	5,373	9.89	12.25	15.78	19.53	0.22
	MDA8 ozone	Observed MDA8 O3 > 60 ppb	3,582	2.76	8.80	3.91	12.47	0.27
	PM2.5 nitrate ion	None (all data)	540	-0.16	0.24	-65.37	96.29	0.02
	PM2.5 sulfate ion	None (all data)	541	-0.01	0.24	-2.31	42.02	0.16
	PM2.5 total carbon	None (all data)	549	1.34	1.83	88.98	121.65	0.05
Oct 2014	MDA8 ozone	None (all data)	1,308	-2.46	7.68	-4.40	13.73	0.52
	MDA8 ozone	Modeled MDA8 O3 > 60 ppb	268	-2.99	7.89	-4.31	11.38	0.32
	MDA8 ozone	Observed MDA8 O3 > 60 ppb	503	-10.15	10.87	-14.54	15.57	0.30
	PM2.5 nitrate ion	None (all data)	77	-0.21	0.32	-50.54	75.59	0.43
	PM2.5 sulfate ion	None (all data)	77	-0.12	0.21	-23.80	42.91	0.26
	PM2.5 total carbon	None (all data)	71	0.77	1.03	66.70	89.49	0.75

MDA8 = daily average maximum daily 8-hour; O_3 = ozone; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm; ppb = parts per billion.

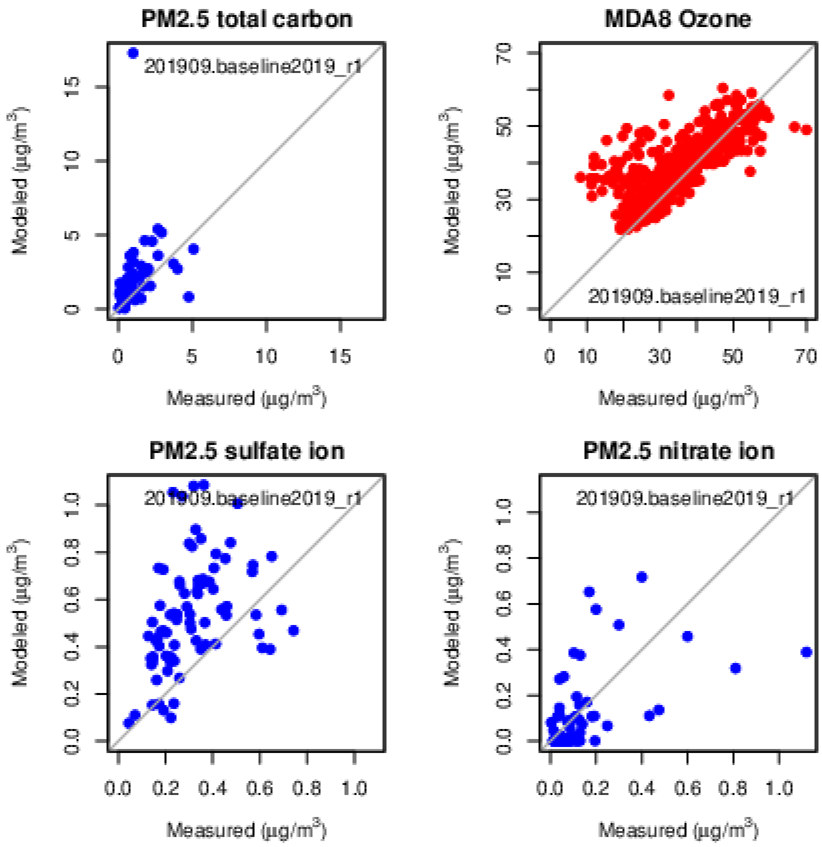
Metrics are aggregated over all monitors in the model domain for each modeling period.



µg/m^3 = micrograms per cubic meter; MDA8 = maximum daily 8-hour average; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm; TC6 = Timber Crater 6.

Model prediction-observation pairs represent monitor locations in the study area region during the 2018 modeling period used to support the Timber Crater 6 scenarios.

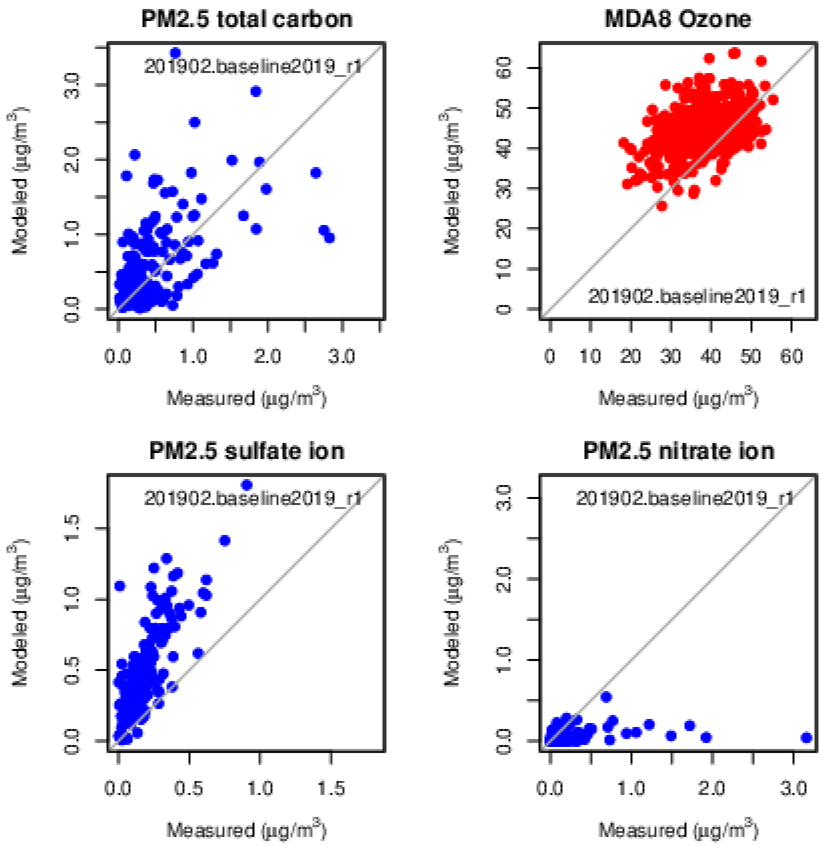
Figure A.7-1 Daily average maximum daily 8-hour average (MDA8) ozone and speciated components of $PM_{2.5}$, including total carbon, sulfate ion, and nitrate ion model predictions paired with routine surface monitor data in space and time.



µg/m^3 = micrograms per cubic meter; MDA8 = maximum daily 8-hour average; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm.

Model prediction-observation pairs represent monitor locations in the study area region during the 2019 fall modeling period used to support the Timber Crater 6 prescribed fire scenarios.

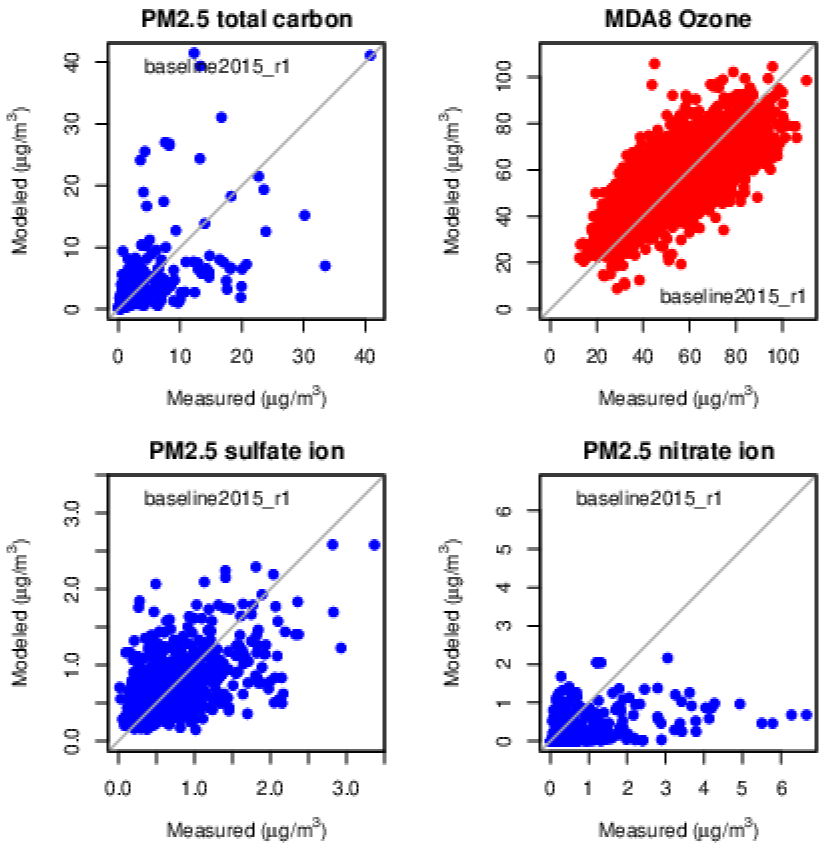
Figure A.7-2 Daily average maximum daily 8-hour average (MDA8) ozone and speciated components of $PM_{2.5}$, including total carbon, sulfate ion, and nitrate ion model predictions paired with routine surface monitor data in space and time.



µg/m^3 = micrograms per cubic meter; MDA8 = maximum daily 8-hour average; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm.

Model prediction-observation pairs represent monitor locations in the study area region during the 2019 winter modeling period used to support the hypothetical slash/pile burn scenarios.

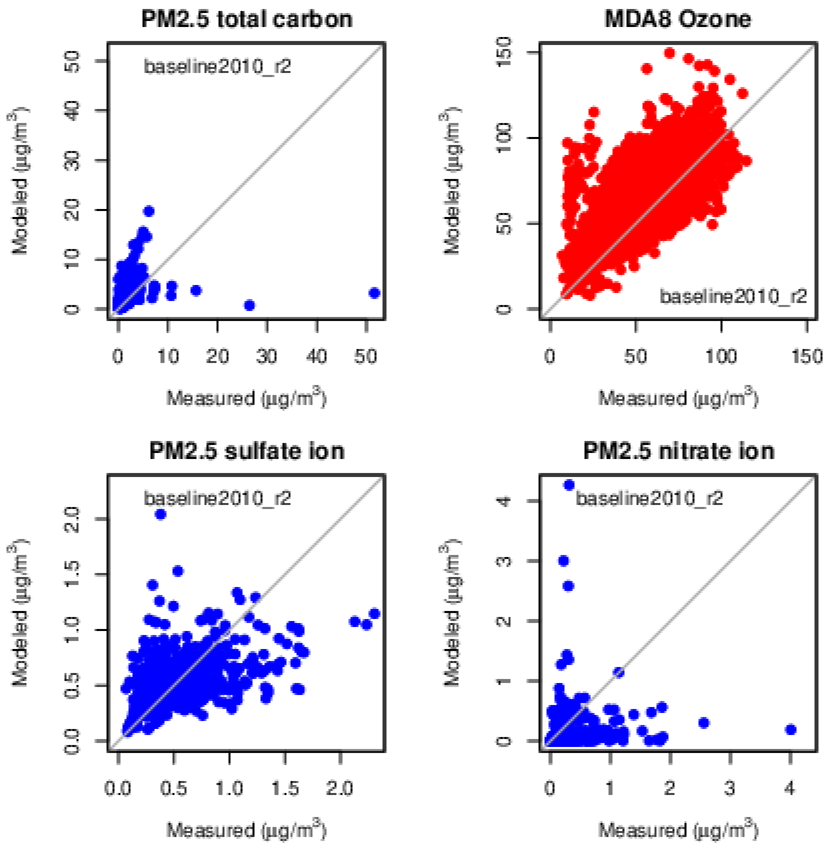
Figure A.7-3 Daily average maximum daily 8-hour average (MDA8) ozone and speciated components of $PM_{2.5}$, including total carbon, sulfate ion, and nitrate ion model predictions paired with routine surface monitor data in space and time.



µg/m^3 = micrograms per cubic meter; MDA8 = maximum daily 8-hour average; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm.

Model prediction-observation pairs represent monitor locations in the study area region during the 2015 modeling period used to support the Rough Fire scenarios.

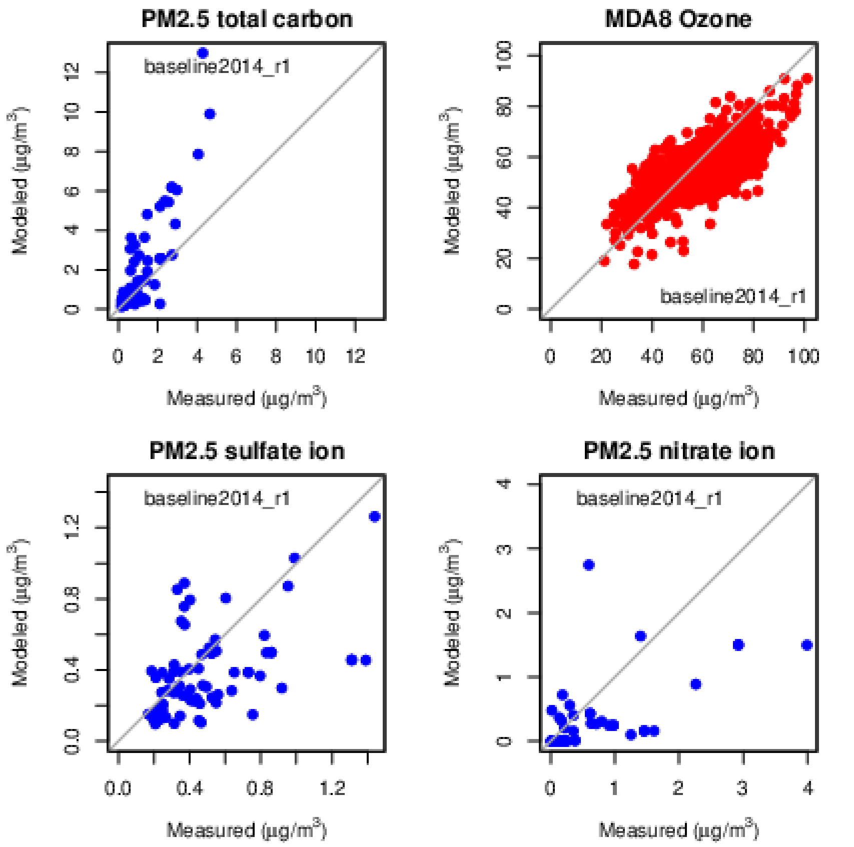
Figure A.7-4 Daily average maximum daily 8-hour average (MDA8) ozone and speciated components of $PM_{2.5}$, including total carbon, sulfate ion, and nitrate ion model predictions paired with routine surface monitor data in space and time.



µg/m³ = micrograms per cubic meter; MDA8 = maximum daily 8-hour average; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm.

Model prediction-observation pairs represent monitor locations in the study area region during the 2010 modeling period used to support the Sheep Complex Fire scenario.

Figure A.7-5 Daily average maximum daily 8-hour average (MDA8) ozone and speciated components of $PM_{2.5}$, including total carbon, sulfate ion, and nitrate ion model predictions paired with routine surface monitor data in space and time.



µg/m³ = micrograms per cubic meter; MDA8 = maximum daily 8-hour average; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 µm.

Model prediction-observation pairs represent monitor locations in the study area region during the 2014 modeling period used to support the hypothetical Boulder Creek Unit 1 Prescribed Fire scenario.

Figure A.7-6 Daily average maximum daily 8-hour average (MDA8) ozone and speciated components of $PM_{2.5}$, including total carbon, sulfate ion, and nitrate ion model predictions paired with routine surface monitor data in space and time.

A.7.2. Supplemental Materials for Section 7.2.2: Surface Fuel Loads

A.7.2.1. Introduction

Supplementary materials included here for Section 7.2.2 provide additional details on methods used to develop Landscape Ecology, Modeling, Mapping, and Analysis (LEMMA)-initialized Visualizing Ecosystem Land Management Assessments (VELMA) applications and associated VELMA-Fuel

Characteristic Classification System (FCCS) fuelbed databases for the Timber Crater 6 (TC6), Rough, and Sheep Complex case study applications.

Extensive technical and quality assurance documentation is referenced in U.S. EPA's ScienceHub data repository (https://catalog.data.gov/dataset/epa-sciencehub).

A.7.2.2. Quality Assurance Project Plan

U.S. EPA has established quality assurance requirements that must be followed within U.S. EPA and by extramural contractors for all work performed that involves environmental data collection, use, or reporting, including modeling-related activities. Consistent with these requirements, all work performed and reported herein using U.S. EPA's VELMA model follow the VELMA Modeling Quality Assurance Project Plan [QAPP; McKane (2020)].

The VELMA Modeling QAPP describes quality assurance practices relevant to all VELMA applications, such as those described in this report. These practices concern issues of data quality, calibration, validation, propagation of error, and other considerations outlined in the Table of Contents (Appendix Figure A.7-7).

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VELMA = Visualizing Ecosystem Land Management Assessments.

Figure A.7-7 Quality assurance topics addressed in the Visualizing Ecosystem Land Management Assessments (VELMA) Modeling Quality Assurance Project Plan (McKane, 2020).

The QAPP also provides the U.S. EPA computer server secure location containing the VELMA applications developed for this project. This information includes VELMA model input and output files used for model calibration and validation, references, and other documentation supporting these activities.

A.7.2.3. Methods

A.7.2.3.1. Characterizing Surface Fuel Load Estimates Using the Fuel Characteristics Classification System (FCCS)

The FCCS is a consistent, scientifically based framework that provides a catalogue of fuelbeds across the U.S. that coincide with various cover types, including grasslands, shrublands, woodlands, and forests (Ottmar et al., 2007). In FCCS, a fuelbed is defined as a relatively homogeneous landscape unit that represents a unique combustion environment. Each fuelbed is separated into categories and subcategories that depict the loading available for fuel and vary depending on the landscape unit being represented (Appendix Figure A.7-8).

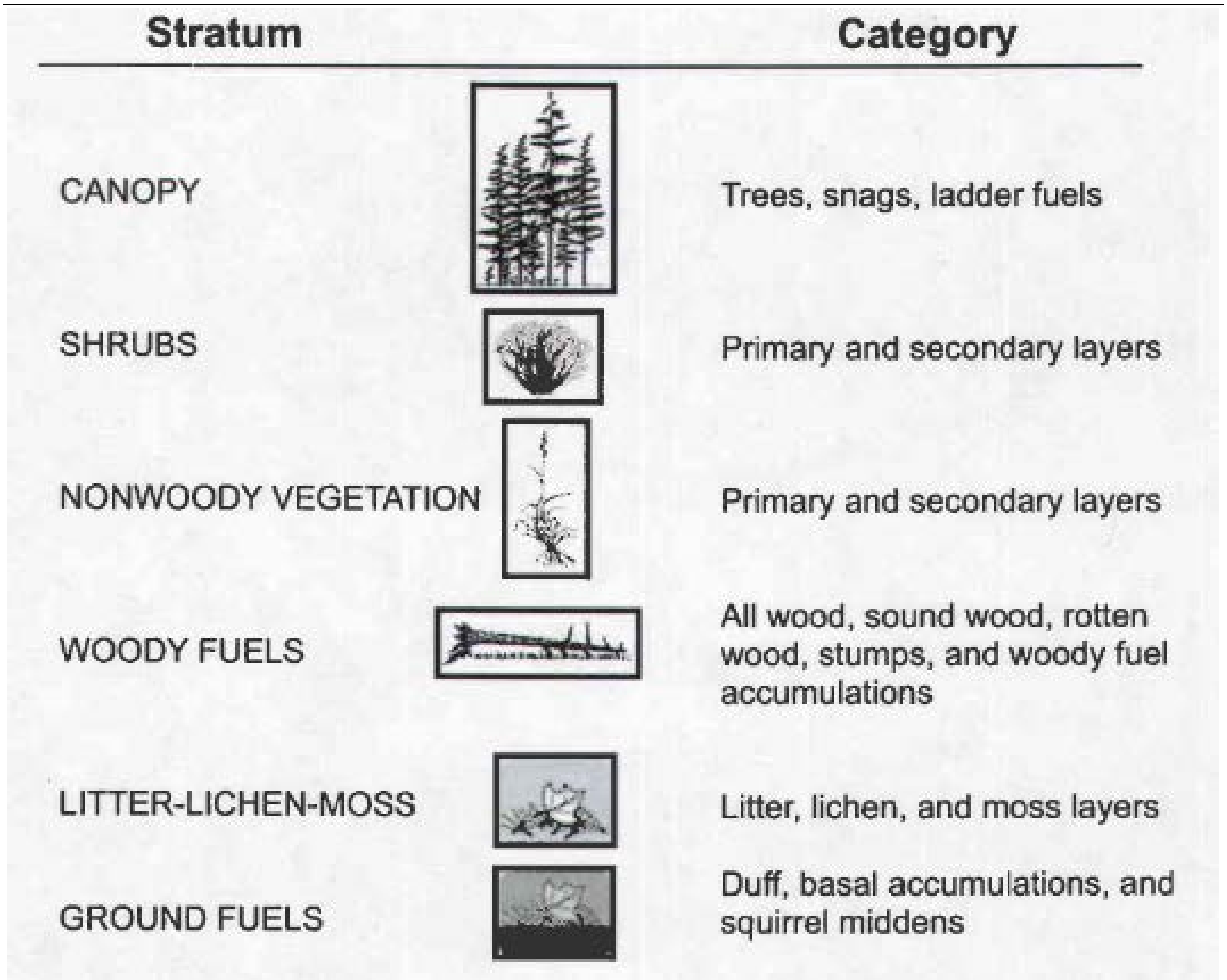


Figure A.7-8 Fuelbed strata and categories included in the Fuel Characteristic Classification System [FCCS; Ottmar et al. (2007)].

The fuel load values (U.S. tons C/acre) for each fuelbed category are derived from scientific literature, fuel databases, and expert knowledge. Further information can be found in Ottmar et al. (2007). For the purposes of this investigation, the FCCS fuelbeds were matched to case study regional boundaries using existing vegetation type layers obtained from LANDFIRE (http://landfire.cr.usgs.gov/viewer/). The resulting FCCS data then consisted of a raster file that described unique identification codes representing various fuelbed types, as well as a look-up table that provided fuelbed loading values (in tons/acre) for each of the fuelbed categories and subcategories.

Although FCCS captures the general diversity of available fuels found throughout the U.S., the fuel loadings are summarized across all plots within a particular vegetation classification category. Studies suggest that the accuracy of FCCS and similar vegetation-based approaches are limited because of the high spatial and temporal variability of fuels, site-specific conditions, and the presence of disturbances including harvests, prescribed fires, and other disturbances (Lutes et al., 2009; Brown and See, 1986, 1981).

A.7.2.4. Improving Fuel Characteristic Classification System (FCCS) Surface Fuel Load Estimates Using Visualizing Ecosystem Land Management Assessments (VELMA), a Spatially Explicit, Process-Based Ecohydrological Model

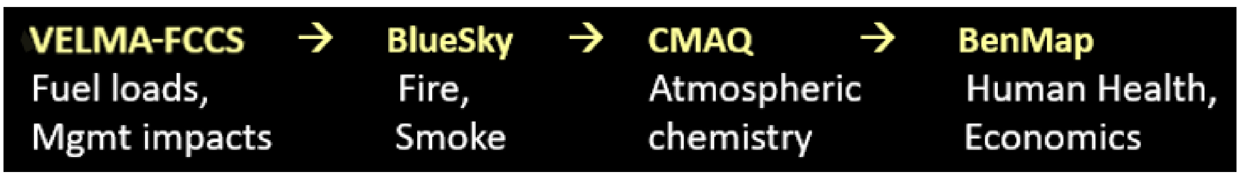
Because of the limitations listed above, a model-based approach was explored to supplement existing FCCS data to more accurately characterize surface fuel loads that could then be used to simulate air quality impacts and the effects of prescribed fire for the various real and hypothetical case studies described in Chapter 7 of this report.

A.7.2.4.1. Overview of Visualizing Environmental Land Management Assessments (VELMA)

The VELMA model is a spatially distributed (grid-based) ecohydrological model that simulates integrated daily responses of vegetation, soil, and hydrologic components to changes in climate, land use, and land cover. VELMA does this through its linkage of a land surface hydrology model with a terrestrial biogeochemistry model. The hydrology model simulates water infiltration and redistribution, evapotranspiration (ET), and surface and subsurface runoff. The biogeochemistry model simulates plant growth and mortality, formation and turnover of detritus and soil organic matter, and associated cycling of carbon and nutrients. The interaction of hydrological and biogeochemical processes in the model constrains changes in ecosystem structure and function in response to various environmental changes, including management. VELMA simulates land management activities in a spatially and temporally explicit manner; these activities include harvest, prescribed fire, and wildfire, among other potential treatments (McKane et al., 2014). VELMA has been applied in many terrestrial ecosystem types, including forests, grasslands, agricultural lands, floodplains, and alpine and urban landscapes (Barnhart et al., 2021; Hoghooghi et al., 2018; McKane et al., 2016; Barnhart et al., 2015; Abdelnour et al., 2013; Abdelnour et al., 2011). Particularly in forests and rangelands, it has been used to simulate the effects of fire and harvest on ecosystem structure and function and subsequent recovery, including impacts on ecosystem services vital to human health and well-being (McKane et al., 2018; Yee et al., 2017).

As noted above, a main advantage of using VELMA to supplement FCCS surface fuel load estimates are that FCCS data varies by fuelbed but each fuelbed does not vary spatially or temporally. This means that two cells with the same fuelbed classification will give the exact same surface fuel load estimations, regardless of their location in the watershed. Conversely, VELMA can be initialized using spatially distributed aboveground biomass or forest age data that are location and condition specific to a defining year. During a simulation, live and dead biomass pools within any watershed pixel can change daily based as a function of water availability, temperature, soil type, and landscape position, as well as any management actions (e.g., clearcutting, thinning, fire) that the user has specified. Therefore, VELMA can capture spatial variations in live and dead biomass pools attributable to spatially and temporally varying conditions within the landscape. For example, VELMA's forest harvest and forest burn tools make it possible to simulate reductions in live and dead fuel loads and subsequent rates of recovery.

As discussed in Section 7.2.2, our goal in combining FCCS and VELMA fuelbed information is to improve the accuracy of spatial and temporal surface fuel load estimates and, therefore, the accuracy of the BlueSky and Community Multiscale Air Quality (CMAQ) air quality models and, ultimately, the accuracy of Benefits Mapping and Analysis Program (BenMAP) and associated tools used to assess air quality impacts on human health at local and regional scales (Appendix Figure A.7-9).



BenMAP = Benefits Mapping and Analysis Program; CMAQ = Community Multiscale Air Quality; FCCS = Fuel Characteristic Classification System; VELMA = Visualizing Ecosystem Land Management Assessments.

Figure A.7-9 Generalized model-to-model workflow for this study.

A.7.2.4.1.1. Visualizing Ecosystem Land Management Assessments (VELMA) Inputs and Initialization

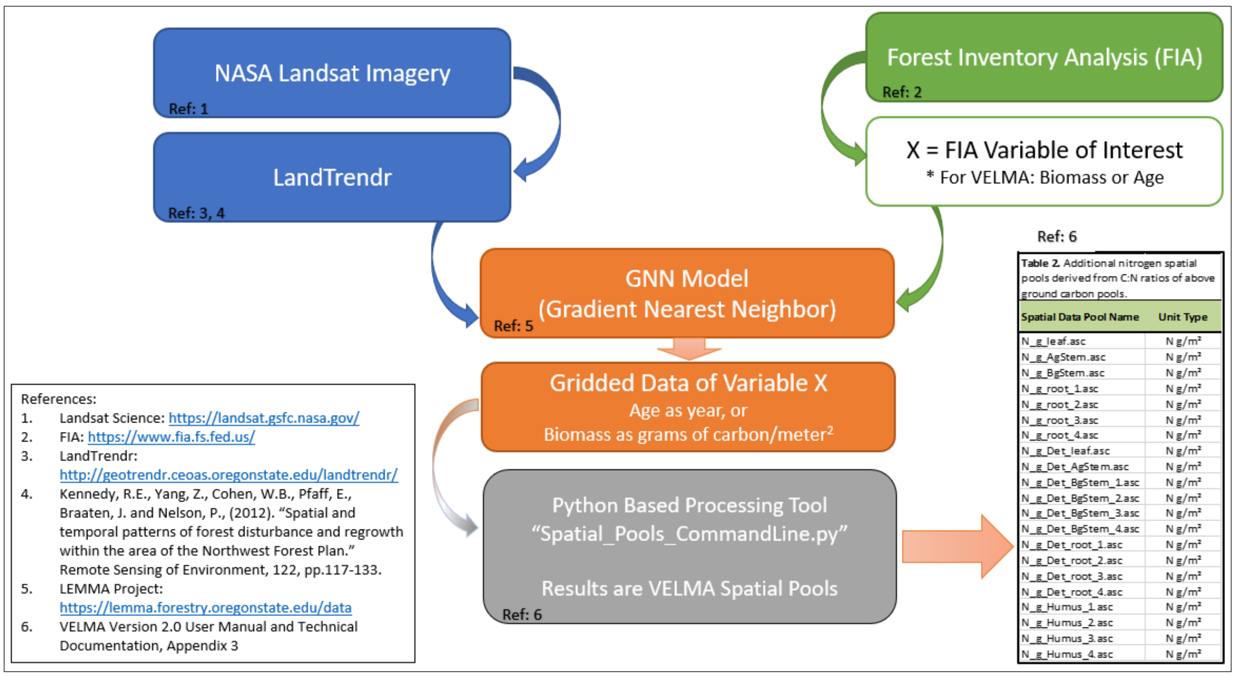
Model inputs and simulation methods varied depending on the case study being implemented—Timber Crater 6, Rough, or Sheep Complex. In this section we summarize the full range of methods and discuss in subsequent sections how specific steps were implemented for each case study. These steps include:

1. Acquire satellite-based LEMMA data to develop a spatial (30-m) description of total aboveground forest biomass and stand age for a specified landscape and year (Appendix Figure A.7-10).
2. Use Step 1 LEMMA data to generate spatial carbon and nitrogen pools for VELMA's 13 plant and soil state variables, per U.S. EPA VELMA documentation, *How To Create VELMA Spatial Chemistry Pools.docx* (McKane et al., 2014). This procedure resulted in carbon and nitrogen pool look-up tables for stand ages ranging from 0 to 400 years old. See Appendix Figure A.7-11 for

an example illustrating age-related (successional) changes in aboveground stem biomass.

3. Initialize VELMA using Step 2 spatial plant and soil carbon and nitrogen pool data. Initialization also requires the additional environmental spatial data described in Appendix Table A.7-5.

4. Use the fully initialized VELMA model (Step 3) to conduct specified actual and hypothetical fire treatments for case study locations. Note: depending on a case study's end goals of combining FCCS and VELMA fuelbed information, Steps 3 and 4 may not be necessary.



C = carbon; FIA = Forest Inventory Analysis; g/m^2 = grams per square meter; GNN = gradient nearest neighbor; LEMMA = Landscape Ecology, Modeling, Mapping, and Analysis; N = nitrogen; NASA = National Aeronautics and Space Administration; VELMA = Visualizing Ecosystem Land Management Assessments.

Figure A.7-10 Procedures for acquiring Landscape Ecology, Modeling, Mapping, and Analysis (LEMMA; LandTrendr/gradient nearest neighbor [GNN]) high-resolution (30-m) satellite data used to initialize Visualizing Ecosystem Land Management Assessments (VELMA) for the case studies described in this report.

Table A.7-5 Spatial data type, source, and years used to initialize Visualizing Ecosystem Land Management Assessments (VELMA) for case study simulations.

VELMA Data Type	Source	Year
Timber Crater 6 Setup		
Weather drivers	PRISM: precipitation and mean air temperature https://prism.oregonstate.edu/explorer/	2010 through 2019
Elevation	USDA Data Gateway DEM: https://datagateway.nrcs.usda.gov/GDGOrder.aspx	2019
Age	LEMMA: https://lemma.forestry.oregonstate.edu/data	2010
Biomass	LEMMA[a] https://lemma.forestry.oregonstate.edu/data	2010
Coverage	Uniform[b]	NA
Soils	Uniform (TC6 per Remillard (1999))	NA
Rough and Sheep Complex Setups		
Age	LEMMA: https://lemma.forestry.oregonstate.edu/data	2012
Biomass	LEMMA[a]: https://lemma.forestry.oregonstate.edu/data.	2012
Coverage	Uniform[b]	NA

DEM = digital elevation model; FCCS = Fuel Characteristic Classification System; LEMMA = Landscape Ecology, Modeling, Mapping, and Analysis; NA = not applicable; PRISM = Parameter-elevation Regressions on Independent Slopes Model; TC6 = Timber Crater 6; USDA = U.S. Department of Agriculture; VELMA = Visualizing Ecosystem Land Management Assessments.

[a]LEMMA aboveground biomass undergoes a unit conversion and is then processed through VELMA's preprocessing tool "Spatial_Pools_Py3_CommandLine.py" script.

[b]FCCS coverage for nonforested cells was included during the combining of the FCCS and VELMA fuelbed information step.

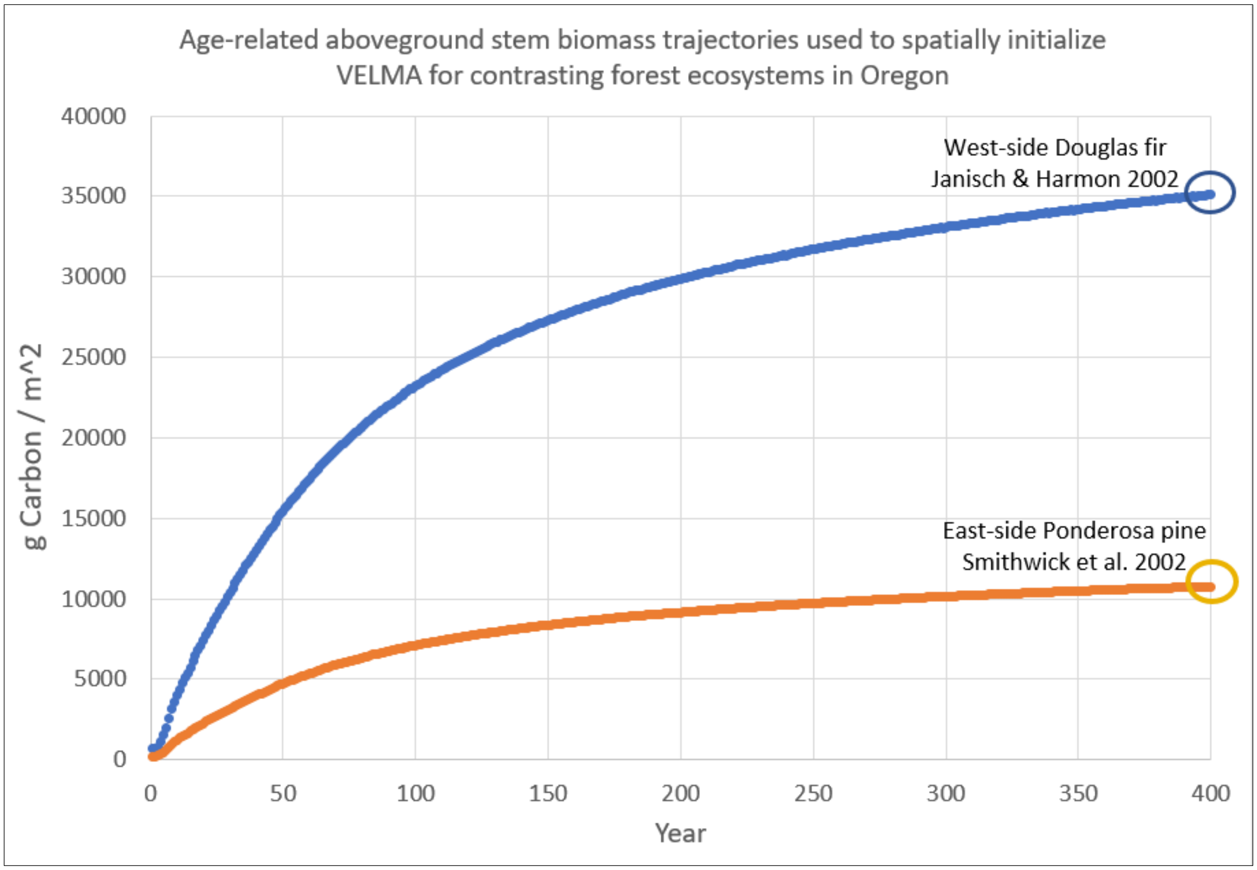
Timber Crater 6 case study. This study site was set up using Steps 1 through 4 so that potential alternate scenarios could be completed prior to the actual forest fire event date. Model initialization occurs for the Year 2010 to leave open the possibility of simulating prefire land management actions prior to the actual August 2018 fire. Simulations carried out to date were restricted to landscape conditions.

Rough and Sheep Complex case studies. These case study sites were developed only up to Step 1, above, then jumped directly to the step of combining FCCS and VELMA fuelbed information, described in Appendix Section A.7.2.4.1.5. In this case, due to fortuitous data timing, the VELMA biomass data was acquired from the time-zero LEMMA biomass and age data initialization and represented the forest state for the actual scenario. If future work requires alternate scenarios of land management actions within these sites, LEMMA biomass and age data initialization should occur for years preceding the actual fires to allow VELMA to be initialized and set up to simulate prefire fuelbed treatments.

LEMMA data capture the effects of fine-scale annual changes in aboveground forest biomass associated with fire, harvest, road construction and other disturbances that have occurred since 1990 across California, Oregon, and Washington. LEMMA data quality is keyed to U.S. Forest Service (USFS) Forest Inventory and Analysis (FIA) survey data, along with extensive local- and regional-scale validation against independent Light Detection and Ranging (LiDAR)-based forest survey methods (Bell et al., 2018).

In practice, age-related biomass trajectories (Appendix Figure A.7-11) take the form of look-up tables, developed using the LEMMA-based procedure described for Steps 1 and 2 in this section for initializing spatial (30-m grid) carbon and nitrogen pools for VELMA's 13 plant and soil state variables across a landscape. See Appendix Figure A.7-15 for a 3-D visualization of spatial variability in aboveground live forest biomass for a LEMMA-initialized landscape for the TC6 case study. Appendix Figure A.7-16 is a histogram showing the number of 30-m pixels represented in Appendix Figure A.7-15 across the full range of aboveground biomass values for this case study domain (Appendix Figure A.7-14).

Note that age-related biomass trajectories, such as the example in Appendix Figure A.7-11, are used for the sole purpose of spatially initializing time-zero plant and soil carbon and nitrogen pools for landscapes simulated using VELMA. Simulated trajectories from Day 1 forward are a function of environmental forcing variables, such as climate, nutrient availability, and disturbances. For example, simulation of a heavily irrigated and fertilized ponderosa pine forest could potentially follow a steeper trajectory than that shown for ponderosa pine (orange line) in Appendix Figure A.7-11.

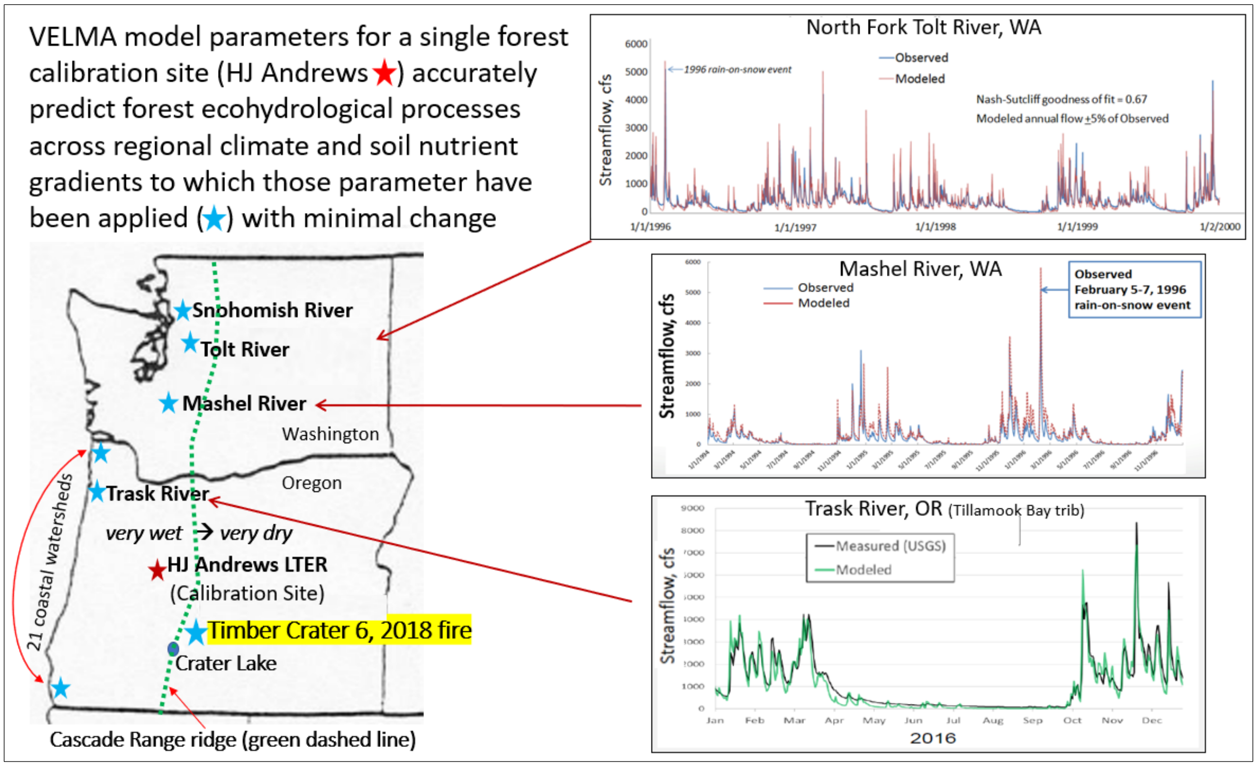


g carbon/m^2 = grams of carbon per square meter; VELMA = Visualizing Ecosystem Land Management Assessments.

Figure A.7-11 Age-related changes (successional trajectories) in aboveground stem biomass for Douglas fir and ponderosa pine growing in western and eastern Oregon, respectively.

A.7.2.4.1.2. Model Calibration and Performance

Prior to this study, Pacific Northwest VELMA applications focused on productive, high biomass Douglas fir/western hemlock forest ecosystems growing on the moist west side of the Cascade Range in Oregon and Washington (annual precipitation range ~2,000−3,500 mm). For those applications a single set of VELMA model parameters, calibrated for the HJ Andrews Experimental Forest (McKane et al., 2014; Abdelnour et al., 2013; Abdelnour et al., 2011), has accurately simulated hydrological and biogeochemical responses across dozens of watersheds in western Oregon and Washington, after accounting for location-specific climate and soil nutrient status (Appendix Figure A.7-12).



cfs = cubic feet per second; LTER = Long Term Ecological Research; TC6 = Timber Crater 6; VELMA = Visualizing Ecosystem Land Management Assessments.

The location of the TC6 study site on the drier east side of the Cascade Range is shown for reference. Figure updated from McKane et al. (2018).

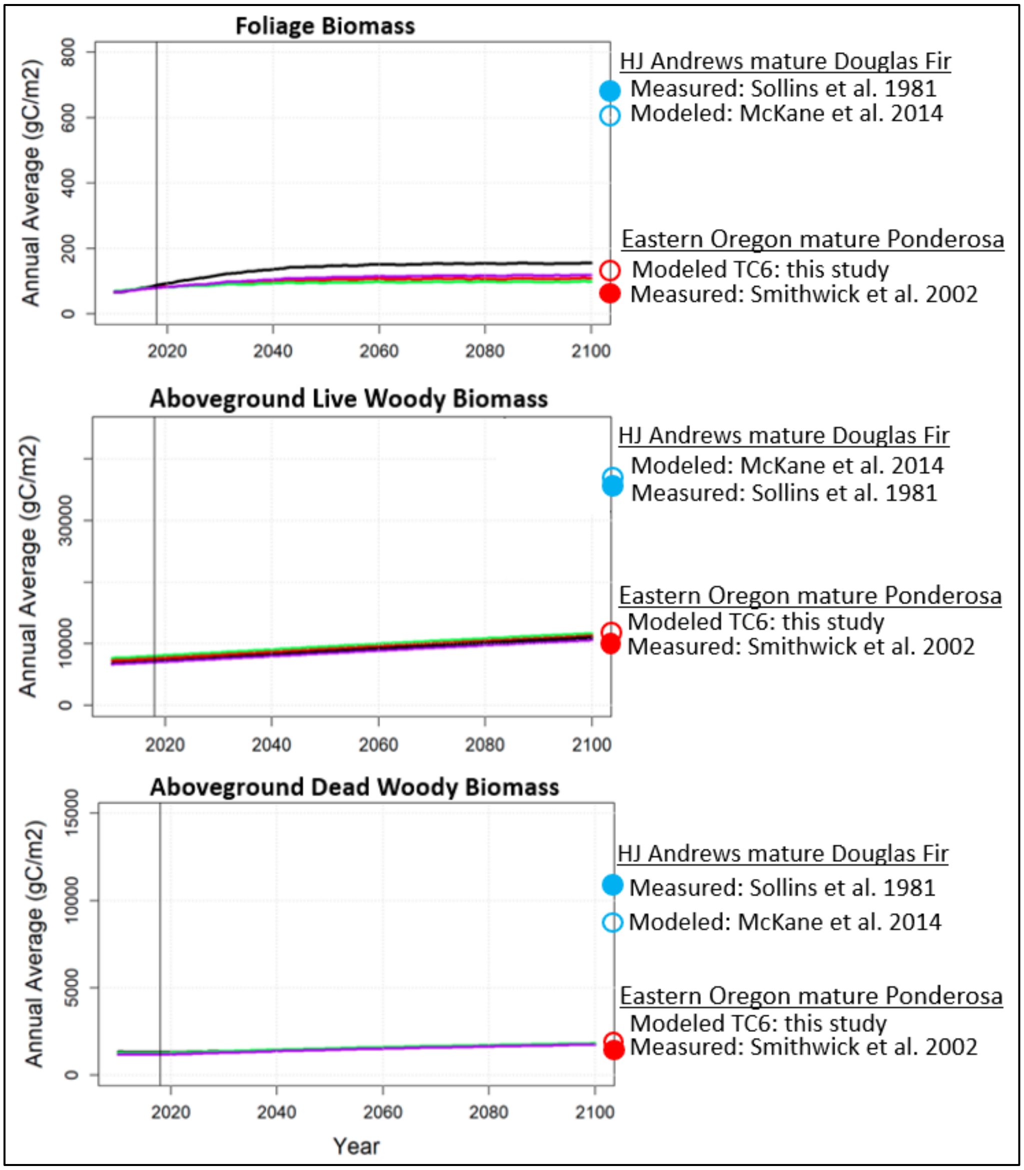
Figure A.7-12 Locations of various coniferous forest sites in western Oregon and Washington for which Visualizing Ecosystem Land Management Assessments (VELMA) have been successfully applied regionally on the basis of a single, broadly applicable set of model parameters developed for the HJ Andrews Experimental Forest.

To explore whether the same west-side HJ Andrews VELMA calibration parameters could be successfully applied to the much drier and nutrient-poor east-side TC6 study site (Appendix Figure A.7-12), we used the procedures outlined in Appendix Section A.7.2.4.1.1 to initialize the HJ Andrews calibration for TC6, replacing LEMMA-based HJ Andrews Douglas fir forest biomass values that are several times higher than east-side coniferous forest values, including those at TC6 (Appendix Figure A.7-11).

No other changes were made except to (1) drive the TC6-initialized HJ Andrews calibration with local TC6 daily climate drivers (Appendix Table A.7-5); and (2) replace HJ Andrews soil carbon and nitrogen values with those for TC6 (Remillard, 1999). Regarding (1), average annual precipitation is about 500 mm at TC6, about 25% as much as the HJ Andrews site receives (Smithwick et al., 2002).

Regarding (2), deep volcanic Mazama ash soils in the vicinity of TC6/Crater Lake contain about 1/4 as much soil nitrogen as HJ Andrews sandy loam soils (Remillard, 1999).

We ran the LEMMA-initialized TC6 VELMA from 2010 to 2100 to examine initial amounts and long-term successional trajectories of live and dead forest biomass pools relevant to fuel load assessments developed for this study (Appendix Figure A.7-13). Although no U.S. Forest Service Forest Inventory and Analysis plots are located within the TC6 study area, published data describing observed biomass for mature ponderosa pine forests at the U.S. Forest Service Pringle Falls Experimental Forest are available to assess model performance.



g C/m^2 = grams of carbon per square meter.

Also shown are modeled and observed biomass data for the HJ Andrews mature Douglas fir/western hemlock forest in western Oregon for which VELMA hydrological and biogeochemical parameters were calibrated (McKane et al., 2014; Abdelnour et al., 2013; Abdelnour et al., 2011) and only recently applied without changes to the TC6 site. See text for details.

Figure A.7-13 Visualizing Ecosystem Land Management Assessments (VELMA) simulated biomass trajectories (2010 to 2100) for the Timber Crater 6 (TC6) case study site versus observed biomass for a mature eastern Oregon ponderosa pine forest [Pringle Falls Experimental Forest reference stand PF29; Smithwick et al. (2002)].

Modeled TC6 biomass trajectories from 2010 to 2100 for stand-level foliage, live aboveground woody biomass, and dead aboveground woody biomass are in good agreement with long-term observed targets for mature ponderosa pine near Pringle Falls, OR. The TC6 and Pringle Falls forest sites are located on the same nutrient-poor Mazama ash soil type, formed about 7,700 years ago when Mt. Mazama erupted, leading to the formation of Crater Lake.

Also shown in Appendix Figure A.7-13 are modeled and observed biomass data for the HJ Andrews mature Douglas fir/western hemlock forest site (Watershed 10) for which VELMA hydrological and biogeochemical parameters were calibrated and applied to TC6. Taken together with the TC6 ponderosa pine results, Appendix Figure A.7-13 indicates that the limited availabilities of water and nutrients in eastern Oregon strongly constrain biomass growth and accumulation compared with conditions at the HJ Andrews site in western Oregon.

These results are encouraging for future VELMA applications, suggesting that it will be possible to use a single, broadly applicable set of VELMA parameters to closely approximate biomass and fuel load dynamics across large landscapes that include steep, complex gradients of climate, soil, vegetation, and disturbance histories. The availability of publicly accessible spatial and temporal databases for all of these variables—with LEMMA annual 30-m forest biomass estimates going back to 1990—make such VELMA applications possible for essentially any forested site in California, Oregon, and Washington.

VELMA case study applications for the TC6, Rough, and Sheep Complex fires are discussed in the following sections.

A.7.2.4.1.3. Case Study 1: Timber Crater 6 (TC6) Fire

VELMA simulations were conducted for the landscape surrounding the Timber Crater 6 Fire that occurred in south-central Oregon, near Crater Lake National Park, from July 21−26, 2018 (Appendix Figure A.7-14). The TC6 actual fire burned ~3,100 acres of forest cover dominated by mixed-age ponderosa pine and red fir. VELMA was used to simulate biomass/fuel loads for two main boundaries, including the actual TC6 burn area (red area in Appendix Figure A.7-14) and the worst-case hypothetical scenario (dotted line in Appendix Figure A.7-14).

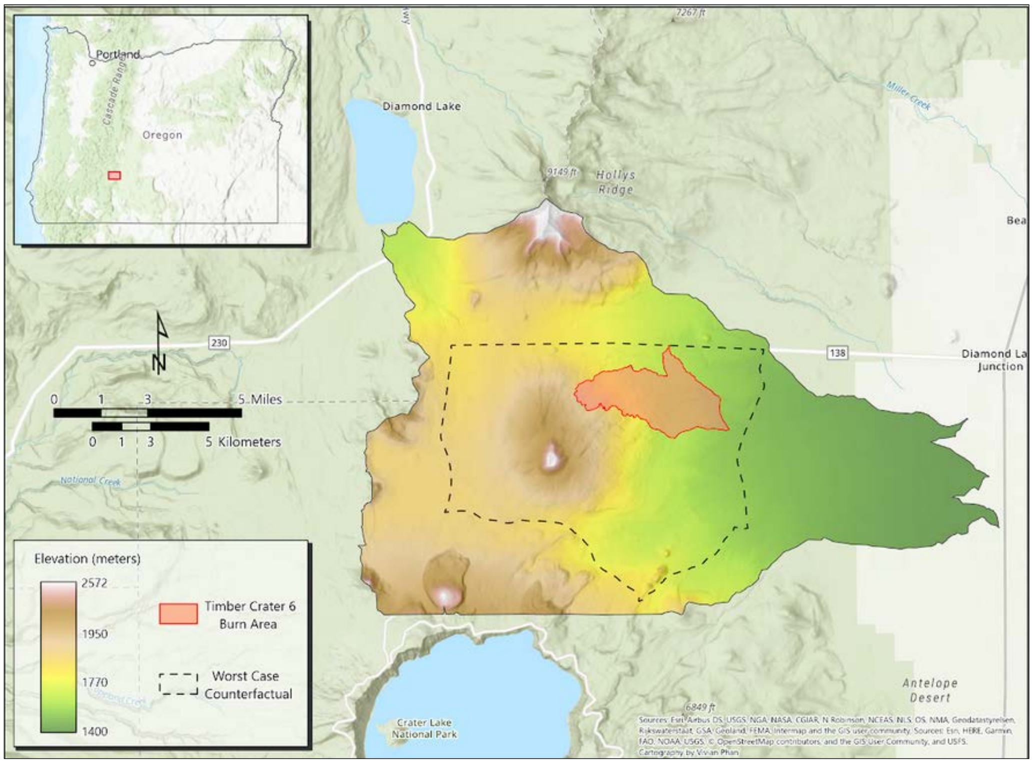


Figure A.7-14 Study location of the actual Timber Crater 6 (TC6) Fire (red shaded area) and the maximum extent of hypothetical fire treatments, for which surface fuel load estimates were made by harmonizing products from both the Fuel Characteristic Classification System (FCCS) and the Visualizing Environmental Land Management Assessments (VELMA) model.

As described in Appendix Section A.7.2.4.1.2, initial (time zero) aboveground total (live and dead) biomass estimates for the TC6 region were obtained from gradient nearest neighbor (GNN) forest biomass and species maps for 2010 from the LEMMA project at Oregon State University (Kennedy et al., 2018; Davis et al., 2015).

The total simulation area was divided into four separate areas because of the large spatial extent and because VELMA is a watershed model that depends on hydrologically created boundaries. Each of the four areas were simulated separately by VELMA and the results were subsequently stitched together to encapsulate the full fire boundary area. Each simulation began in 2009 to stabilize all pools prior to initialization. Spatially distributed biomass quantities from LEMMA were then incorporated on 2010 Julian Day 1. Each simulation was then conducted until 2020, but the relevant surface fuel loads for 2018 Julian Day 201 (July 20, 2018), which represent the day prior to the start of the TC6 Fire, were used for subsequent analysis. Gridded inputs of elevation, land use/land cover, and soils were collected and rescaled to match the 30-m resolution of the FCCS/LANDFIRE vegetation cover data it was intended to supplement.

The study site digital elevation model (DEM) was clipped from the national elevation data set (NED) acquired from the U.S. Geological Survey (USGS) and rescaled from a 1/3-arc-second resolution to 30 m. The 30-m DEM was flat-processed using the JPDEM-Dredge processing tool (McKane et al., 2014; Pan et al., 2012). JPDEM was also used to derive the stream network based on existing elevation changes.

A single forest ecosystem calibration of VELMA was applied to TC6 that has been found to be broadly applicable to Pacific Northwest coniferous forest types, including the ponderosa pine ecoregion of eastern Oregon. Model initialization and validation details are described in Appendix Section A.7.2.4.1.2.

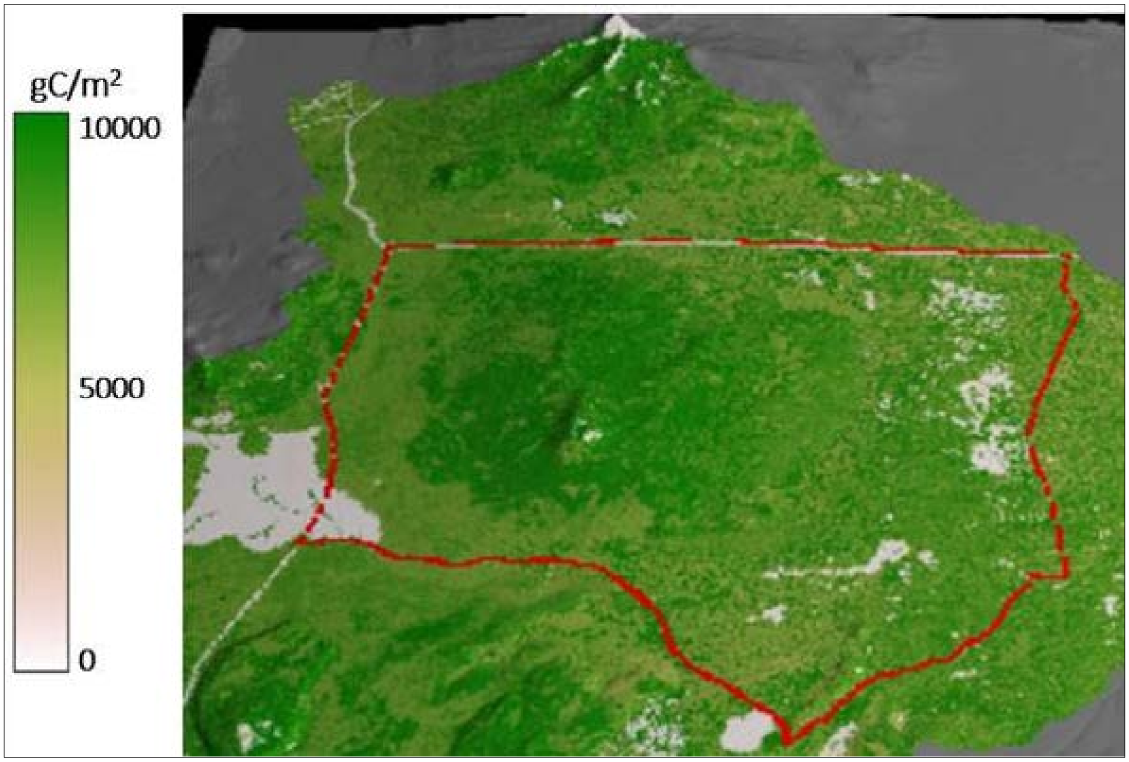
Daily precipitation and temperature drivers were obtained from Oregon State University's Parameter-elevation Regressions on Independent Slopes Model (PRISM) Climate Group for 2010−2020 and consist of climatologically aided interpolation (CAI) values that use both long-term (30-year) averaging and radar measurements as inputs. For more information, see Daly et al. (2008) and https://prism.oregonstate.edu/explorer/. No stream flow data were available for hydrologic validation for this particular region. Nonetheless, VELMA's ability to model hydrologic processes with minimal calibration has been shown to be regionally robust (Appendix Figure A.7-12).

VELMA's simulation outputs include a suite of environmental parameters that can be used to model and better understand spatial and temporal variability in ecosystem properties that result from differences in climate, wildfire, management, and other disturbances. Responses modeled include changes in live and dead aboveground and belowground biomass components, stream flow, stream temperature, stream nutrients and contaminants, and others.

For this case study, VELMA was used to simulate aboveground live and dead biomass pools corresponding to fuel loadings for forest overstory trees (excluding near-surface fuels such as downed coarse woody debris, shrubs, etc. that are not easily detected using Landsat-based satellite technology such as LEMMA). These fuel categories were simulated at 30-m resolution and a daily time step. These spatial and temporal resolutions can be aggregated to lower resolutions using spatial and temporal averaging techniques.

Specifically, for the Timber Crater 6 application, VELMA's simulated aboveground live biomass pools, including stem and leaf components, were exported as 30-m raster data sets. These represent the aboveground live stem and leaf material across the TC6 region that are available on the day prior to the actual TC6 Fire, that is, July 20, 2018. Model performance tests shown in Appendix Figure A.7-13 demonstrate VELMA's capabilities for accurately simulating aboveground biomass pools relevant to fuel load estimation purposes. Appendix Figure A.7-15 shows VELMA's aboveground biomass simulations for the worst-case hypothetical boundary associated with the TC6 Fire.

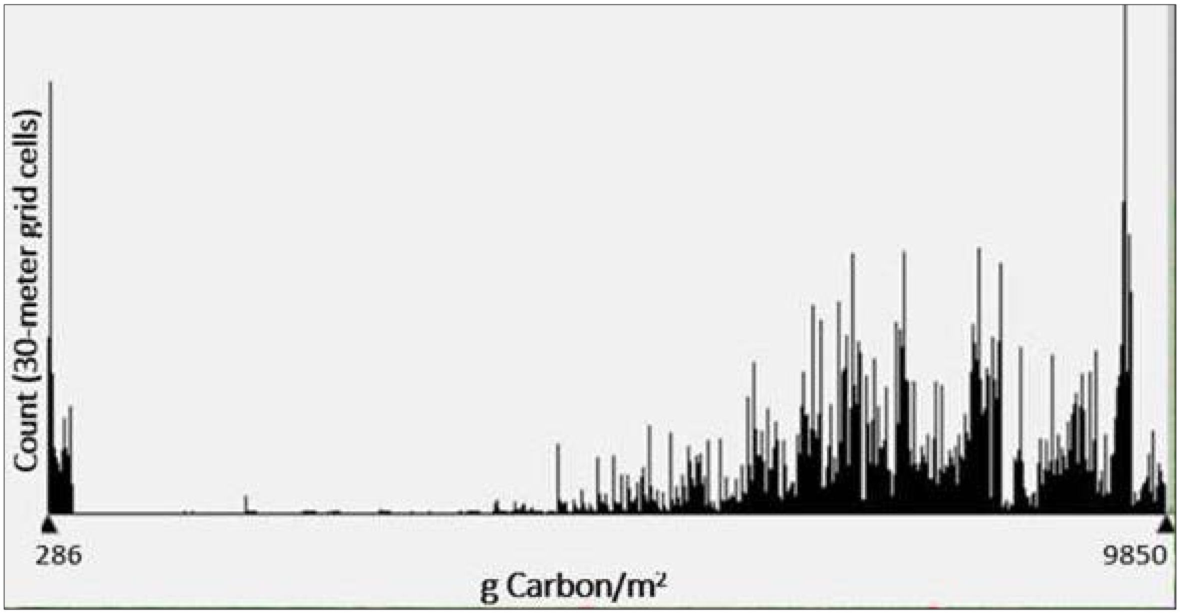
Appendix Figure A.7-16 is a histogram of aboveground stem values, which accounted for the majority of the total aboveground live biomass.



g C/m^2 = grams of carbon per square meter; FIA = Forest Inventory and Analysis; USFS = U.S. Forest Service.

The red line is the simulation boundary for hypothetical TC6 worst-case BlueSky Pipeline modeling scenarios. Spatial variations in VELMA modeled aboveground biomass (g C/m^2) range from near zero (white shading) to a maximum of ~10,000 g C/m^2 (dark green), which corresponds to regional total biomass maxima for ponderosa/lodgepole pine-dominated forests measured on permanent plots maintained by the FIA network (USFS reference) and by the Pringle Falls Research Natural Area (Smithwick et al., 2002).

Figure A.7-15 30-m resolution Visualizing Ecosystem Land Management Assessments (VELMA) aboveground live forest biomass results for the Timber Crater 6 (TC6) case study area for the day before the beginning of the actual TC6 Fire on July 20, 2018 (see also Appendix Figure A.7-14).



g carbon/m^2 = grams of carbon per square meter.

Vertical bars describe the number of 30-m grid cells for the range of biomass values shown on the *y*-axis. See Appendix Figure A.7-15 for worst-case scenario boundary.

Figure A.7-16 Histogram of aboveground stem biomass simulated by Visualizing Ecosystem Land Management Assessments (VELMA) in the worst-case hypothetical scenario associated with the Timber Crater 6 (TC6) Fire (simulation day: July 20, 2018).

Note that a regional maximum observed aboveground biomass of approximately 10,000 g C/m^2 has been reported by Smithwick et al. (2002) at the nearby Pringle Falls Research Natural Area. Data for this old-growth ponderosa pine forest was used to validate VELMA-simulated biomass in this study, as described in Appendix Section A.7.2.4.1.2. Appendix Figure A.7-15 shows that this maximum biomass estimate corresponds well with the western portion of the TC6 boundary, which is older and less disturbed. In fuel-load terms, this is equivalent to 44.6 U.S. tons C/acre or 89.2 U.S. tons dry wt./acre.

These VELMA simulations were used to supplement the FCCS surface fuel load estimations for the TC6 region. The process by which the FCCS and VELMA data products were combined and exported to the BlueSky Pipeline suite of air quality models are described in Appendix Section A.7.2.4.1.5.

A.7.2.4.1.4. Case Study 2: Rough, Sheep Complex, and Boulder Creek Fires

The second case study focused on the 2015 Rough Fire in the Sierra National Forest in California and consisted of a total of 151,000 burned acres (Appendix Figure A.7-17). In late August of that year, the fire expanded eastward, encountering areas partially burned in two earlier, less intense fires—the 2010 Sheep Complex wildfire and the 2013 Boulder Creek Prescribed Fire. These earlier fires mostly reduced surface fuels, likely preventing the speed and severity of the rapidly advancing Rough Fire in 2015, at least in those particular areas and points to the east (Appendix Figure A.7-17). A National Park Service interactive story map of the Rough Fire clearly illustrates these Rough Fire dynamics (https://www.nps.gov/seki/learn/nature/rough-fire-interactive-map.htm).

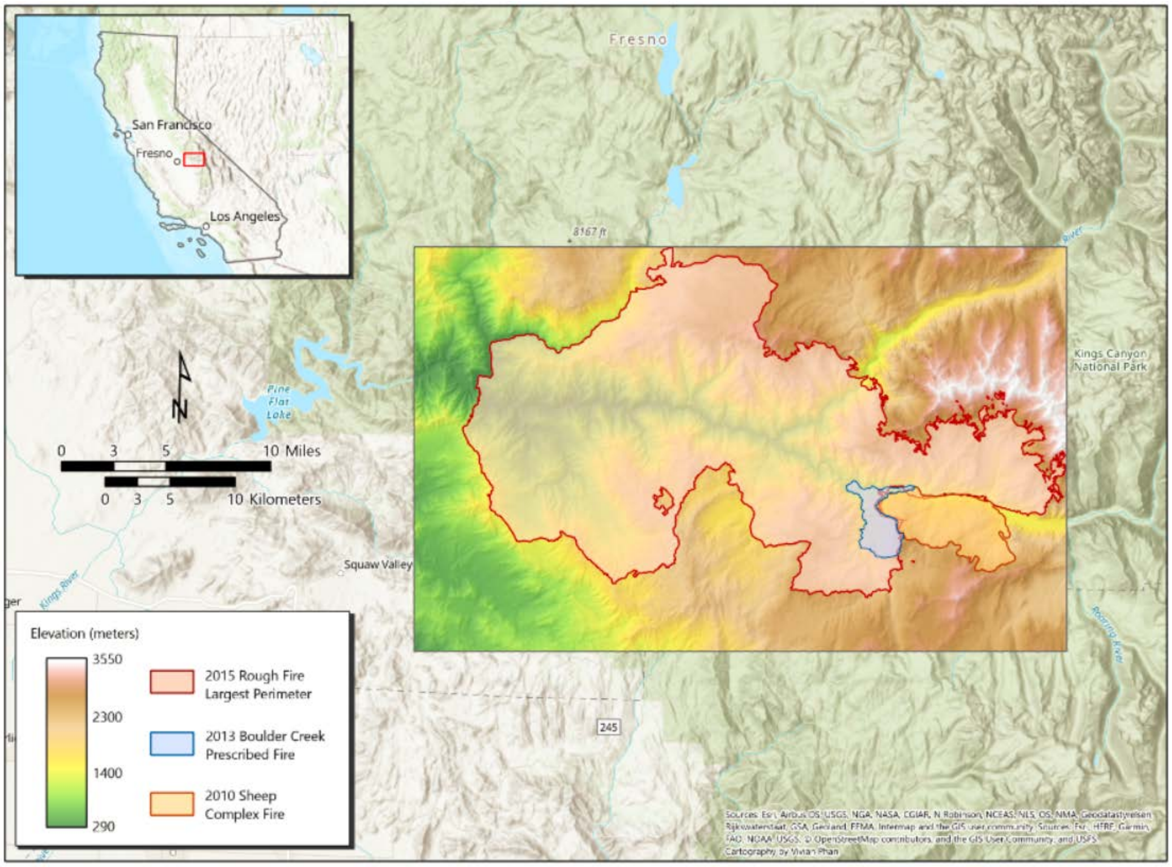


Figure A.7-17 Study location of the 2015 Rough Fire, the 2010 Sheep Complex Fire, and the 2013 Boulder Creek Prescribed Fire.

The fuelbed characterization objectives of this case study were to (1) use LEMMA and VELMA-based methods to augment and improve accuracies of existing FCCS surface fuel load estimates within the Rough, Sheep Complex, and Boulder Creek fire boundaries and (2) provide the combined VELMA-FCCS fuelbed data to the BlueSky Pipeline CONSUME fire simulator.

As with the TC6 case study, the objectives of this case study were accomplished by obtaining aboveground forest biomass estimates from 30-m from 30-m, satellite-derived forest biomass and species maps from the LEMMA project at Oregon State University (Kennedy et al., 2018; Davis et al., 2015; Kennedy et al., 2012).

LEMMA data for 2012 were obtained for the extent of the Rough Fire boundary, whereas LEMMA data for 2010 and 2013 were obtained for the Sheep Complex and Boulder Creek boundaries, respectively.

As described for the TC6 case study (Appendix Section A.7.2.4.1.3), these LEMMA data were processed through the VELMA Spatial_Pools_Py3_CommandLine.py Python tool that converted aboveground biomass from a single layer into VELMA's 13 plant and soil carbon and nitrogen pools, which include forest leaf biomass and aboveground stem wood (boles, branches, twigs) fuelbed categories.

The fuelbed data for aboveground stem and leaf biomass derived from this VELMA/LEMMA method were directly merged with FCCS fuelbed categories, skipping the multiyear VELMA biomass spin-up method applied to TC6. For TC6 there was a significant multiyear gap between the TC6 Fire year (2018) and the closest year of available LEMMA data (2010), which necessitated an 8-year VELMA "spin-up" to account for growth and decay of live and dead biomass/fuelbeds during that time. Because there was a closer overall match between fire years and corresponding LEMMA data years for the Rough, Sheep Complex and Boulder Creek fires, it was not necessary to implement the VELMA spin-up step.

A.7.2.4.1.5. Process for Combining Fuel Characteristic Classification System (FCCS) and Visualizing Ecosystem Land Management Assessments (VELMA) Surface Fuel Load Estimations for All Case Studies

A depiction of the process used to conjoin the FCCS and VELMA data for all case studies is shown in Figure 7-6 from Chapter 7 of this report. The process alters the original landscape units from FCCS to include new fuelbed categories that incorporate different VELMA-simulated aboveground biomass values. The CONSUME model within the BlueSky Pipeline is currently set up to accept inputs using a standard FCCS data format; therefore, VELMA's spatial raster data were processed and incorporated into the current FCCS data format to form a harmonized data product featuring spatially variable surface fuel loads.

VELMA's heterogeneous spatial maps of aboveground live stem and leaf biomass simulations were processed into categories, then spatially merged with the FCCS classes. These tasks were carried out in ArcGIS Pro and described below within the ESRI tool framework, though this data processing routine could be performed in most GIS software.

First, VELMA biomass data were reclassified into discrete bins based on their value using the "*Reclassify*" tool. The live aboveground stem and leaf biomass outputs were reclassified into 11 classes, as shown in Appendix Table A.7-6.

Table A.7-6 Discrete bin classifications used for Visualizing Ecosystem Land Management Assessments (VELMA) and Landscape Ecology, Modeling, Mapping, and Analysis (LEMMA) aboveground biomass values for each of the case studies.

	Timber Crater 6		Rough and Sheep Complex	
Bin Numbers	**Stem**	**Leaf**	**Stem**	**Leaf**
1	0−1,000	0−80	0−3,000	0−60
2	1,000−2,000	80−100	3,000−6,000	60−120
3	2,000−3,000	100−120	6,000−9,000	120−180
4	3,000−4,000	120−140	9,000−12,000	180−240
5	4,000−5,000	140−160	12,000−15,000	240−300
6	5,000−6,000	160−180	15,000−18,000	300−360
7	6,000−7,000	180−200	18,000−21,000	360−420
8	7,000−8,000	200−220	21,000−24,000	420−480
9	8,000−9,000	220−240	24,000−27,000	480−540
10	9,000−10,000	240−260	27,000−30,000	540−600
11	10,000−11,000	260−280	30,000−33,000	600−660

g C/m^2 = grams of carbon per square meter.
Note: The average value in each bin range was used as the actual value in the raster (g C/m^2).

Once the VELMA data were reclassified into discrete bins based on their values, the FCCS fuelbed identification raster was joined with the fuelbed loading look-up table that provided loadings for each of the fuelbed categories using "*Add Join.*" Then, both the VELMA outputs and the FCCS data were joined together using "*Intersect* (Analysis)" after first converting to polygons using "*Raster to Polygon* (Conversion)." The resulting output provides a combined polygon file with the attribute table containing both sets of data in a spatially merged representation. The combined VELMA + FCCS polygon layer was then converted back to a raster using the "*Polygon to Raster* (Conversion)" and exported as a final raster layer. The tabular data was saved as an Excel file (.xlsx) using "*Table to Excel.*"

Although the raster file was now ready to be sent to CONSUME and the BlueSky Pipeline, a number of processing steps were needed to adjust the exported attribute table so that VELMA information replaced FCCS data for particular fuelbed categories and that the table followed the appropriate format. At this step, care was taken to ensure that the units supplied by VELMA were correctly converted to those used in FCCS. In particular, VELMA simulates aboveground biomass values as g C/m^2, whereas FCCS uses U.S. tons/acre and assumes dry weight biomass. Therefore, we conducted the conversion using the relationship 1 g C/m^2 = 0.0044609 U.S. tons/acre. That value was derived from 1 g = 1.10231×10^{-6} U.S. ton and 1 m^2 = 0.000247105 acre. Alternatively, one can specify that 1 U.S. ton = 907,185 g and 1 acre = 4,046.86 m^2. These tons of carbon were then converted to tons of dry weight biomass by assuming that 0.5 g carbon are present in 1 g of dry weight biomass.

In addition, an R software (R Core Team, 2019) processing script was used to convert VELMA's total aboveground biomass estimates (live stem and leaf) to the appropriate quantity to replace fuel load defaults in FCCS.

Note that only forested fuelbeds were replaced using VELMA's simulated data, whereas all grassland and savanna fuelbeds continued to use the standard FCCS inputs. Parameters and equations from Jenkins et al. (2003) were used to derive component ratios for tree crowns for both hardwood and softwood species, and these ratios were multiplied by the VELMA's total aboveground biomass for each of the forested fuelbed classifications to replace the default FCCS "overstory_loading" category. The "midstory_loading" and "understory_loading" categories were set to zero during replacement to avoid double counting. All remaining fuelbed categories (e.g., snags, shrubs, litter, duff) continued to use FCCS default values.

The final outputs of combining FCCS and VELMA data to provide surface fuel loads to the CONSUME model in the BlueSky Pipeline consisted of two data products. The first was a new FCCS + VELMA raster file that included new unique fuelbed identification numbers. These fuelbeds incorporate both FCCS cover type specifications and VELMA's discrete biomass bins. The second data product was a revised fuel loading look-up table that is used to specify loadings for each of the fuelbed categories given in the raster. Results

As mentioned in the previous methods section, the final outputs that combine FCCS and VELMA data were used as inputs to the CONSUME model in the BlueSky Pipeline. Note that although FCCS provides a number of fuel load categories for surface fuel loads (see Appendix Figure A.7-8), VELMA is used only to modify the crown loading estimates for forested cover types. An in-depth comparison of the resulting fuel loading changes is shown for each of the case studies in the following sections.

A.7.2.4.2. Case Study 1: Timber Crater 6 (TC6) Fire

A comparison of the VELMA and FCCS crown loading estimates for all FCCS forested fuelbeds that represent greater than 1% of the total TC6 actual fire boundary are shown in Appendix Table A.7-7. Note that the VELMA values shown in the table are averages across all cells that have the FCCS fuelbed name. VELMA's simulations tend to be generally higher than those from FCCS, where, for example, VELMA predicts 15.28 U.S. tons/acre of overstory crown loading and FCCS predicts 9.51 U.S. tons/acre. An exception is the Red Fir Forest, for which FCCS estimates a loading of 24.97 U.S. tons/acre and VELMA estimates an average value of 14.86 U.S. tons/acre.

Also, it is apparent that VELMA's simulated values are much greater than FCCS values for fuelbeds characterized by prior disturbance—that is, fuelbeds denoted with "WF 5−10 YR." FCCS data were obtained from 2012 so these disturbance categories represent disturbances that occurred between 2002 and 2007 and may therefore underestimate the actual biomass present during the TC6 Fire in 2018.

Table A.7-7 Comparison of Timber Crater 6 (TC6) study domain crown loading estimates between Fuel Characteristic Classification System (FCCS) and Visualizing Ecosystem Land Management Assessments (VELMA) for all FCCS forested fuelbeds that represent greater than 1% of the total boundary area percentage.

FCCS Fuelbed Name	FCCS*	VELMA*	Area (%)
Pacific Ponderosa Pine Forest	9.51	15.28	26
Red Fir Forest	24.97	14.86	15
Red Fir-Mountain Hemlock-Lodgepole Pine-Western White Pine Forest	16.21	16.11	9
Giant Sequoia-White Fir-Sugar Pine Forest	9.36	14.43	5
WF 5−10 yr: Red Fir Forest	4.37	14.98	4
Pacific Silver Fir-Mountain Hemlock Forest	9.38	16.00	3
WF 5−10 yr: Giant Sequoia-White Fir-Sugar Pine Forest	0.00	14.59	3
Mature Lodgepole Pine Forest	4.13	19.07	3
WF 5−10 yr: Pacific Ponderosa Pine Forest	1.48	16.54	3
Pacific Silver Fir-Sitka Alder Forest	2.33	17.94	2
Ponderosa Pine-Jeffrey Pine Forest	8.62	14.71	1

FCCS = fuel characteristic classification system; VELMA = visualizing ecosystem land management assessments; wf = wildland fire; yr = year.

Note: the VELMA values represent the crown fuel loads estimated from VELMA'S aboveground biomass simulations and (Jenkins et al., 2003) tree component ratios, while the FCCS values are the sum of the "overstory_loading," "midstory_loading," and "understory_loading" fuel load categories. All units are provided in U.S. tons/acre dry weight biomass.

A.7.2.4.3. Case Study 2: Sheep Complex and Rough Fires

A comparison of the FCCS and LEMMA crown loading estimates for all FCCS forested fuelbeds that represent greater than 1% of the total Sheep Complex actual fire boundary are shown in Appendix Table A.7-8.

Note that LEMMA data were only produced for a subset of the total number of fuelbeds because of lack of data of the component ratios available from Jenkins et al. (2003) that coincide with the FCCS fuelbed names depicted in the table. When available, these ratios were multiplied by LEMMA's total aboveground biomass for each of the forested fuelbed classifications to replace the default FCCS category, as described previously. As with the TC6 case study, the VELMA/LEMMA crown loading values are lower than the default FCCS values for Red Fir Forest fuelbed type (17.69 vs. 24.97 U.S. tons/acre, respectively), whereas they match well for the ponderosa (8.61 vs. and 8.62 U.S. tons/acre)-Jeffrey pine (8.13 vs. 8.33 U.S. tons/acre) mixes and are greater than the FCCS defaults for the mature lodgepole pine forest type (18.25 vs. 4.13 U.S. tons/acre).

For the Rough Fire boundary, a comparison of the VELMA and FCCS crown loading estimates for all FCCS forested fuelbeds that represent greater than 1% of the fire boundary are shown in Appendix Table A.7-9.

Table A.7-8 Comparison of Sheep Complex study domain crown loading estimates between Fuel Characteristic Classification System (FCCS) and Visualizing Ecosystem Land Management Assessments (VELMA)/Landscape Ecology, Modeling, Mapping, and Analysis (LEMMA) for all FCCS forested fuelbeds that represent greater than 1% of the total Sheep Complex boundary area percentage.

FCCS Fuelbed Name	FCCS	VELMA/LEMMA	Area (%)
Red Fir Forest	24.97	17.69	18
California Black Oak Woodland	19.63		14
Douglas Fir-Sugar Pine-Tanoak Forest	19.30		12
Ponderosa Pine-Jeffrey Pine Forest	8.62	8.61	3
Douglas Fir-White Fir Forest	20.94		3
Jeffrey Pine-Red Fir-White Fir/Greenleaf-Snowbrush Forest	14.38		3
Jeffrey Pine-Ponderosa Pine-Douglas Fir-California Black Oak Forest	8.33	8.13	2
Mature Lodgepole Pine Forest	4.13	18.25	2
Douglas Fir/*Ceanothus* Forest	3.75		2
Subalpine Fir-Lodgepole Pine-Whitebark Pine-Engelmann Spruce Forest	9.55		1

FCCS = fuel characteristic classification system; LEMMA = landscape ecology, modeling, mapping, and analysis; VELMA = visualizing ecosystem land management assessments.

Note: that tree component ratios used by (Jenkins et al., 2003) were unavailable for some fccs fuelbed cover types, and therefore crown loading values could not be computed and are shown as blanks.

*the lemma values represent the crown fuel loads estimated from lemma's aboveground biomass estimates and (Jenkins et al., 2003) tree component ratios, while the fccs values are the sum of the "overstory_loading," "midstory_loading," and "understory_loading" fuel load categories. all units are provided in u.s. tons/acre dry weight biomass.

Table A.7-9 Comparison of Rough study domain crown loading estimates between Fuel Characteristic Classification System (FCCS) and Visualizing Ecosystem Land Management Assessments (VELMA)/Landscape Ecology, Modeling, Mapping, and Analysis (LEMMA) for all FCCS forested fuelbeds that represent greater than 1% of the total Rough Fire area percentage.

Fuelbed Name	FCCS	VELMA/LEMMA	Area (%)
California Black Oak Woodland	19.63		18
California Live Oak-Blue Oak Woodland	1.21		17
Douglas Fir-Sugar Pine-Tanoak Forest	19.30		17
Red Fir Forest	24.97	18.68	15
Jeffrey Pine-Ponderosa Pine-Douglas Fir-California Black Oak Forest	8.33	13.18	4
Jeffrey Pine-Red Fir-White Fir/Greenleaf-Snowbrush Forest	14.38		3
Douglas Fir-White Fir Forest	20.94		2
Ponderosa Pine-Jeffrey Pine Forest	8.62	13.59	2
Subalpine Fir-Lodgepole Pine-Whitebark Pine-Engelmann Spruce Forest	9.55		2
Mature Lodgepole Pine Forest	4.13	19.43	2
Douglas Fir/*Ceanothus* Forest	3.75		1
Black Cottonwood-Douglas Fir-Quaking Aspen Forest	28.68		1

FCCS = Fuel Characteristic Classification System; LEMMA = Landscape Ecology, Modeling, Mapping, and Analysis; VELMA = Visualizing Ecosystem Land Management Assessments.

LEMMA data were only updated for some cover types due to data availability for converting total aboveground biomass estimates to crown loadings using equations from **(**Jenkins et al., 2003**)**.

The LEMMA values represent the crown fuel loads estimated from LEMMA's aboveground biomass estimates and (Jenkins et al., 2003**)** tree component ratios, while the FCCS values are the sum of the "overstory_loading," "midstory_loading," and "understory_loading" fuel load categories. All units are provided in U.S. tons/acre dry weight biomass.

As expected, based on the previous case studies, VELMA estimates lower crown loading values compared with the default FCCS values for Red Fir Forest. However, the remainder of comparisons show that VELMA/LEMMA estimate higher crown loading values compared with the FCCS defaults. Further validation is needed to confirm the canopy estimations from VELMA/LEMMA and their comparison with the original estimates performed by FCCS. Also, note that the values in Appendix Table A.7-7, Appendix Table A.7-8, and Appendix Table A.7-9 represent spatial averages of VELMA data for given FCCS cover types to simplify direct comparison. The combined FCCS/VELMA data products sent to the BlueSky Pipeline, however, include spatially distributed crown loading estimates that are not fully reflected in the previous tables.

A.7.2.5. Conclusions

The use of vegetation-based fuel load classification systems can be extremely helpful for air quality modelers to simulate the air quality impacts of historical or projected wildfires. However, these classification systems are inherently crafted to represent a wide variety of fuel loads across the entire U.S. and therefore do not always capture the fine spatial and temporal heterogeneity associated with landscape-level fuel load changes or disturbance patterns. In this study, we used a spatially distributed ecohydrological landscape model (VELMA) to simulate aboveground live biomass and supplement existing fuel load characterization data for the Timber Crater 6, Rough, Sheep Complex, and Boulder Creek fire boundaries. VELMA was initialized using LEMMA data that provided spatially distributed estimates of live aboveground biomass corresponding to the regions of each of the case studies.

As shown in Appendix Figure A.7-13 and Appendix Figure A.7-15, VELMA fuel load estimates compare well with measured data describing upper limits of aboveground biomass for ponderosa pine stands in eastern Oregon (Smithwick et al., 2002). In addition, VELMA crown loading estimates for forested fuelbeds were compared with FCCS default values. Although there are differences between VELMA and default FCCS estimates for forest crowns and other fuelbeds, further assessments of these estimates based on observed data would be beneficial to examine the validity of surface fuel loads within the regional domain of this study.

Discussions with project partners and others familiar with the case study sites have so far turned up no available georeferenced forest biomass data for assessing the accuracy of model-based estimates for the case study sites. For example, there exists high-quality biomass data for Forest Inventory and Analysis plots within the Rough Fire boundary, but precise coordinates for these plots are inaccessible for security reasons.

Those challenges notwithstanding, the ability of VELMA to accurately simulate ecosystem responses across western and eastern Oregon using a single set of model equations and parameter values provides the strongest possible test of a process-based modeling framework (Appendix Section A.7.2.4.1.2). In essence, VELMA behaves similarly, though imperfectly, to real ecosystems with regard to changes in structure and function in response to environmental changes, whether in situ or across landscape gradients.

These results are encouraging for future VELMA applications, suggesting that it will be possible to closely approximate biomass and fuel load dynamics across large landscapes that include steep, complex gradients of climate, soil, vegetation, and disturbance histories. The availability of publicly accessible spatial and temporal databases for all of these variables—including LEMMA annual 30-m forest biomass estimates going back to 1990—make such VELMA applications possible for essentially any forested site within some western states most hard hit by recent wildfires like California, Oregon, and Washington.

Finally, VELMA is already capable of simulating real and hypothetical land management practices and other disturbances (harvests, wild and prescribed fires, extreme climate events, etc.) at multiple spatial scales. Therefore, future research could incorporate simulations of alternative prescribed burning and mechanical thinning practices to explore local and regional impacts on fuel loads and consequent air quality impacts. Additionally, because VELMA is designed to simulate ecohydrological processes, it can also be used to assess effects of wild and prescribed fires on water quality and quantity, thereby providing an opportunity for integrated air and water quality impact assessments on human health.

A.7.3. Proposed Boulder Creek Prescribed Fire Burn Plan

PRESCRIBED FIRE PLAN

ADMINISTRATIVE UNIT(S): **Sequoia N.F. – Hume Lake R.D.**

PRESCRIBED FIRE NAME: **Boulder Creek Unit 3A1, 3A2 Prescribed Burn**

PREPARED BY: ______________________ **DATE:** ____________
Paul Leusch, RXB2

TECHNICAL REVIEW BY: ______________________ **DATE:** ____________
Brent Skaggs, FFMO

COMPLEXITY RATING: **MODERATE**

APPROVED BY: ______________________ **DATE:** ____________
Teresa Benson, Hume Lake District Ranger

NEPA DOCUMENTATION APPROVED BY & DATE:
Boulder Creek Fuels Restoration Project,
Approved by Sarah LaPlante
4/22/2013

This Prescribed Fire Burn Plan (RXBP) meets direction and guidelines as required by *FSM 5140 Fire Use, Amendment No. 5100-2008-01,and* the *Interagency Standards for Fire and Fire Aviation Operations*, and the *Interagency Prescribed Fire Planning and Implementation Procedures Reference Guide (November,2013)*.

An approved RXBP constitutes the authority to burn. This authority is delegated by the Agency Administrator to the Prescribe Fire Burn Boss and is documented in the RXBP. No one has the authority to burn without an approved RXBP or in a manner not in compliance with the approved RXBP. Actions taken in compliance with the approved RXBP will be fully supported. Personnel will be held accountable for actions taken that are not in compliance with elements of the approved RXBP.

= Must be signed/completed prior to implementation of RXBP.

= Must be completed during implementation of RXBP.

ELEMENT 2: AGENCY ADMINISTRATOR PRE-IGNITION APPROVAL CHECKLIST

Instructions: The Agency Administrator's Pre-Ignition Approval is the intermediate planning review process (i.e. between the Prescribed Fire Complexity Rating System Guide and Go/No-Go Checklist) that should be completed before a prescribed fire can be implemented. The Agency Administrator's Pre-Ignition Approval evaluates whether compliance requirements, Prescribed Fire Plan elements, and internal and external notifications have been or will be completed and expresses the Agency Administrator's intent to implement the Prescribed Fire Plan. If ignition of the prescribed fire is not initiated prior to expiration date determined by the Agency Administrator, a new approval will be required.

YES	NO	KEY ELEMENT QUESTIONS
X		Is the Prescribed Fire Plan up to date? *Hints: amendments, seasonality.*
X		Will all compliance requirements be completed? *Hints: cultural, threatened and endangered species, smoke management, NEPA.*
X		Is risk management in place and the residual risk acceptable? *Hints: Prescribed Fire Complexity Rating Guide completed with rational and mitigation measures identified and documented?*
X		Will all elements of the Prescribed Fire Plan be met? *Hints: Preparation work, mitigation, weather, organization, prescription, contingency resources*
X		Will all internal and external notifications and media releases be completed? *Hints: Preparedness level restrictions*
X		Will key agency staff be fully briefed and understand prescribed fire implementation?
	X	Are there any other extenuating circumstances that would preclude the successful implementation of the plan?
X		Have you determined if and when you are to be notified that contingency actions are being taken? Will this be communicated to the Burn Boss?
		Other:

Recommended by: ______________________________ Date: ____________
FMO/Prescribed Fire Burn Boss

Approved by: ______________________________ Date: ____________
Agency Administrator

Approval expires (date): ______________________________

ELEMENT 2: PRESCRIBED FIRE GO/NO-GO CHECKLIST

	YES	NO
A. Has the burn unit experienced unusual drought conditions or contain above normal fuel loadings which were not considered in the prescription development? If NO proceed with checklist., if YES go to item B.	**xx**	
B. If YES have appropriate changes been made to the Ignition and Holding plan and the Mop Up and Patrol Plans? If YES proceed with checklist below, if NO STOP.	**xx**	

YES	NO	QUESTIONS
		Are ALL fire prescription elements met?
		Are ALL smoke management specifications met?
		Has ALL required current and projected fire weather forecast been obtained and are they favorable?
		Are ALL planned operations personnel and equipment on-site, available, and operational?
		Has the availability of ALL contingency resources been checked, and are they available?
		Have ALL personnel been briefed on the project objectives, their assignment, safety hazards, escape routes, and safety zones?
		Have all the pre-burn considerations identified in the Prescribed Fire Plan been completed or addressed?
		Have ALL the required notifications been made?
		Are ALL permits and clearances obtained?
		In your opinion, can the burn be carried out according to the Prescribed Fire Plan and will it meet the planned objective?

If all the questions were answered "YES" proceed with a test fire.
Document the current conditions, location, and results

Burn Boss

Date

Current Conditions, Location, and Results Present During Go/No-Go Checklist

Location	
Time	
Current Conditions	Dry______ Wet______ RH_______ Wind Direction________ Speed________ Gusts________
Results	

ELEMENT 3 COMPLEXITY ANALYSIS SUMMARY

The Prescribed Fire Complexity Rating was completed utilizing the Prescribed Fire Complexity Rating System Guide (NFES 2474), January, 2004 (or current version).

The purpose of the complexity rating process is to:
1) Assign a complexity rating of *High*, *Moderate*, or *Low* to the prescribed fire.
2) Provide management and implementation personnel a relative ranking as to the overall complexity of the prescribed fire.
3) Provide a process that can be used to identify RXBP elements or characteristics that may pose special problems or concerns.
4) Provide a process that identifies mitigation activities needed to reduce the risk/hazard to the implementation personnel and public as well as mitigating potential resource damage.

The Summary Complexity Rating Rationale will clearly justify the summary rating for prescribed fire organization and Prescribed Fire Burn Boss level. Risks from the Complexity Analysis that are rated *High* and cannot be mitigated are identified with a discussion of the risks associated in the Summary Complexity Rating Rationale. The Prescribed Fire Burn Boss will ensure that the Complexity Analysis is signed by the Prescribed Fire Plan Preparer and the Agency Administrator and attached as an appendix to the RXBP.

The definitions for *High*, *Moderate* and *Low* ratings for *Risk*, *Potential Consequences* and *Technical Difficulty* of each element in the Complexity Analysis are described in the Prescribed Fire Complexity Rating System Guide (NFES 2474), January, 2004. A general summary of overall complexity ratings for a prescribed fire are as follows:

HIGH: These prescribed fires are defined as those where prescribed burning occurs under particularly challenging conditions and/or constraints. This classification includes prescribed fires where the difficulty of achieving resource management objectives is *High*, or where the consequences of project failure may be *High*. Prescribed fire projects involving aerial ignition devices are rated *High* due to the management requirements that surround aircraft use and FSM 5142.2 direction. Helitorch and plastic sphere dispenser (PSD) burn projects are included here. A

Prescribed Fire Manager, Type 1 (RXM1), and/or a Prescribed Fire Burn Boss, Type 1 (RXB1) will implement a *High Complexity* prescribed fire.

MODERATE: This classification includes prescribed fires where the difficulty of achieving resource management objectives is not particularly high or complicated, and where the consequences of project failure are less serious and can be mitigated. A Prescribed Fire Manager, Type 2 (RXM2), and/or a Prescribed Fire Burn Boss, Type 2 (RXB2) will be in command of implementing a *Moderate Complexity* prescribed fire.

LOW: These prescribed fires are defined as those where few constraints, other than the normal prescription parameters exist. This classification includes prescribed fires where achieving resource management objectives is routine and probable consequences of project failure is low.

PRESCRIBED FIRE NAME: Boulder Creek Unit 3A1, 3A2 Prescribed Burn			
ELEMENT	**RISK**	**POTENTIAL CONSEQUENCE**	**TECHNICAL DIFFICULTY**
1. Potential for escape	L	M	L
2. The number and dependence of activities	L	L	L
3. Off-site Values	M	L	L
4. On-Site Values	L	L	M
5. Fire Behavior	M	M	L
6. Management organization	L	L	L
7. Public and political interest	M	L	L
8. Fire Treatment objectives	L	L	L
9. Constraints	M	M	M
10. Safety	L	L	L
11. Ignition procedures/ methods	L	L	L
12. Interagency coordination	L	L	L
13. Project logistics	L	L	L
14. Smoke management	M	M	M

COMPLEXITY RATING SUMMARY	
	OVERALL RATING
RISK	Moderate
POTENTIAL CONSEQUENCES	Low
TECHNICAL DIFFICULTY	Low
SUMMARY COMPLEXITY DETERMINATION	Moderate

RATIONALE: The Boulder Creek Unit 3A1, 3A2 Prescribed Burn is rated as a ***Moderate*** complexity prescribed fire. The achievement of project objectives will require cooperation and communication among the management organization. This teamwork will allow the organization to properly identify the complexities involved (fuel loading, depth, and continuity) and select the prescription parameters that provide the best opportunity for successful completion of the burn. The Risk category scores an overall rating of moderate, the Potential Consequences has a rating of low, and the Technical Difficulty scores an overall rating of low. The Summary Complexity Determination was rated as a moderate. This rating was assigned based on Fire behavior having a moderate risk and moderate potential consequences, Constraints and Smoke Management having moderate risk, moderate potential consequences, and moderate technical difficulty. Based on the overall complexity, an RXB2 is recommended.

ELEMENT 4: DESCRIPTION OF PRESCRIBED FIRE AREA

A. Physical Description

1. Location: The Boulder Creek Unit 3A1, 3A2 Prescribed Burn is on the Sequoia National Forest, Hume Lake Ranger District in Fresno County. Unit 2, the northern unit in the plan is approximately 1.36 miles to the northwest of Kennedy Meadow. Unit 1, the southern unit in the plan is approximately 0.76 miles northwest of Kennedy Meadow. This area is part of the Tornado Creek drainage. The legal description is Township 13 south, Range 29 east, Sections 17, and 20.

To access both units on the Boulder Creek Unit 3A1, 3A2 Prescribed Burn from the Hume Lake District Office take State Hwy. 180 east to NM528 (General's Highway) and head south towards Sequoia National Park, continue on NM528 (General's Highway) to Quail Flat. At Quail Flat take the 14S02 (Burton Road) and head east to the 13S26 (Tornado Meadow Road). Both units are located approximately 2.68 miles from the junction of the 14S02 (Burton Road) and 13S26 (Tornado Meadow Road). The units will be located above the 13S26 (Tornado Meadow Road). See project map in Section 2 of the burn plan folder.

2. Size:

Unit Name	Location	Aspect	Elevation	Drainage	Acres
Unit 3A1	N 36° 46.634 W 118° 50.697 T 13S, R29E, Sec 20.	NW	Top: 7720' Bottom: 7360'	Tornado Creek	81 ac.
Unit 3A2	N 36° 47.011 W 118° 51.156 T 13S, R29E, Sec 17, 20.	SW	Top: 7520' Bottom: 7200'	Tornado Creek	88 ac.
TOTAL PROJECT ACRES					169 ac.

3. Topography:

4. **Unit 3A1**

Topography stays the same across the project area. Elevations range from 7720' at the top of the unit and along the ridgeline to 7360' at the bottom of the unit and along the 13S26 road. The unit is moderately steep and accessible in many locations. The unit is predominately a northwest facing aspect.

Unit 3A2

Topography stays the same across the project area. Elevations range from 7520' at the top of the unit along the ridgeline to 7200' at the bottom of the unit along the 13S26 road. The unit is moderately steep and accessible in many locations. The unit is predominately a southwest facing aspect.

5. Project Boundary:

Unit 3A1

Constructed handline progressing down a spur ridge off of the main ridge from the 13S26A to the 13S26 forms the northern boundary along the left side of the unit. A portion 13S26A and 13S26 form the western boundary at the top of the unit. 13S26 forms the southern boundary at the top of the unit and wraps around to form the western boundary at the bottom of the unit.

Unit 3A2
Constructed handline along the main ridge forms the northeast boundary at the top of the unit. A second handline runs from the ridgeline directly to the 13S26 road and forms the northwest boundary on the left side of the unit. A third handline that runs down a spur ridge to the 13S26 road forms the southeast boundary on the right side of the unit. The 13S26 road forms the southwest boundary at the bottom of the unit.

B. Vegetation/Fuels Description:

Unit 3A1
The unit consists of Mixed Conifer Forest with 50% Fir and 50% Red Fir trees. This unit consists of primarily fuel models TL4 with scattered patches of TU5 accounting for the rest of the unit.

Unit 3A2
The unit consists of Mixed Conifer Forest with 90% Fir and 10% Giant Sequoia trees. This unit consists of primarily fuel models TU5 with scattered patches of TL6 accounting for the rest of the unit.

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Boulder Creek Unit 1, 2 Fuel Loading			
	Fuel Model TU5	Fuel Model TL4	Fuel Model TL6
1 Hour Fuels (0"- ¼")	4.00	0.50	2.40
10 Hour Fuels (¼"-1")	4.00	1.50	1.20
100 Hour Fuels (1"-3")	3.00	4.20	1.20
Live Herbaceous: tons/acre	0.00	0.00	0.00
Live Woody: tons/acre	3.00	0.00	0.00
Total Fuel Load: tons/acre	14.00	6.20	4.80
Avg. Fuel Bed Depth	1.00	0.40	0.30

1. **On-site fuels data**

Vegetation Types:

The project area is comprised of mixed conifer fuel types consisting primarily of Fir overstory with patches of Red Fir, and Giant Sequoia intermixed. These stands are very open with good spacing of the mature trees. In open areas the understory consists of manzanita. The manzanita remains uniform in the open areas and is only broken up by rock outcroppings and sandy soil. The vegetation type and fuel loading stay consistent throughout both units.

Fuel Models:

Timber-Understory Fuel Type Models (TU): The primary carrier of fire in the TU fuel models is forest litter in combination with herbaceous or shrub fuels. TU1 and TU3 contain live herbaceous load and are dynamic, meaning that their live herbaceous fuel load is allocated between live and dead as a function of live herbaceous moisture content. The effect of live herbaceous moisture content on spread rate and intensity is strong and depends on the relative amount of grass and shrub load in the fuel model (Scott and Burgan, 2005).

- **TU5 (165) Very High Load, Dry Climate Timber-Shrub:** The primary carrier of fire in TU5 is heavy forest litter with a shrub or small tree understory. Spread rate is moderate; flame length moderate. Extinction moisture content is high at 25% (Scott and Burgan, 2005).

Timber Litter Fuel Type Models (TL): The primary carrier of fire in the TL fuel models is dead and down woody fuel. Live fuel, if present, has little effect on fire behavior (Scott and Burgan, 2005).

- **TL4 (184) Small downed logs:** The primary carrier of fire in TL4 is moderate load of fine litter and coarse fuels. Includes small diameter downed logs. Spread rate is low; flame length low. Extinction moisture content is high at 25% (Scott and Burgan, 2005).

- **TL6 (186) Moderate Load Broadleaf Litter:** The primary carrier of fire in TL6 is moderate load broadleaf litter, less compact than TL2. Spread rate is moderate; flame length low. Extinction moisture content is high at 25% (Scott and Burgan, 2005).

Fuel arrangement and continuity:

The project area is comprised of a mixed conifer fuel type. Primarily Fir overstory with small patches of Red Fir, and Giant Sequoia intermixed. These stands are open with good spacing of the mature trees. Ground fuels are heavy forest litter with a Manzanita understory. The Manzanita continues to be uniform in the open areas and is broken up by rock outcroppings and sandy soil. Fuel continuity is consistent across the in the Manzanita with a moderate load of leaf litter.

Adjacent fuels data

The vegetation to the southwest of the project area is similar to that within the unit. The vegetation to the northwest has some slight changes. The Fir overstory changes to a Giant Sequoia overstory as you enter the boundary of the Horseshoe Bend Grove. The Manzanita understory and fuel continuity have no change and stay the same.

Timber-Understory Fuel Type Models (TU): The primary carrier of fire in the TU fuel models is forest litter in combination with herbaceous or shrub fuels. TU1 and TU3 contain live herbaceous load and are dynamic, meaning that their live herbaceous fuel load is allocated between live and dead as a function of live herbaceous moisture content. The effect of live herbaceous moisture content on spread rate and intensity is strong and depends on the relative amount of grass and shrub load in the fuel model.

- **TU5 (165) Very High Load, Dry Climate Timber-Shrub:** The primary carrier of fire in TU5 is heavy forest litter with a shrub or small tree understory. Spread rate is moderate; flame length moderate. Extinction moisture content is high at 25%. (Scott and Burgan, 2005).

Timber Litter Fuel Type Models (TL): The primary carrier of fire in the TL fuel models is dead and down woody fuel. Live fuel, if present, has little effect on fire behavior.

- **TL4 (184) Small downed logs:** The primary carrier of fire in TL4 is moderate load of fine litter and coarse fuels. Includes small diameter downed logs. Spread rate is low; flame length low. Extinction moisture content is high at 25%. (Scott and Burgan, 2005).

- **TL6 (186) Moderate Load Broadleaf Litter:** The primary carrier of fire in TL6 is moderate load broadleaf litter, less compact than TL2. Spread rate is moderate; flame length low. Extinction moisture content is high at 25%. (Scott and Burgan, 2005).

C. Description of Unique Features:

Unit 3A1

Plantations exist throughout 60-70% of the unit.

Unit 3A2

Plantations exist throughout 5-10% of the unit. A portion of the Horseshoe Bend Giant Sequoia Grove exists in 10% of the unit.

ELEMENT 5: GOALS AND OBJECTIVES

A. Goals/Objectives:

1. **Resource objectives:**

Desired Conditions for portions of the Boulder project area come from multiple sources:

Vegetation desired condition, including Sequoias, is (2012 Monument Plan p. 22):

1. Forested stands in the Mediterranean climate of the Monument are subject to frequent weather cycles. Years of cooler, wetter weather are often followed by years of hotter, drier weather. The desired condition of a forested stand subject to these extremes is diversity in composition (species, size, age class, distribution) and spatial distribution that are expected to be more resilient to climate changes over time.

Wildlife Habitat desired condition is (2012 Monument Plan p. 24):

1. Lands in the Monument continue to provide a diverse range of habitats that support viable populations of associated vertebrate species, with special emphasis on riparian areas, montane meadows, and late successional forest.... Old forest habitat is in suitable quality, quantity, and distribution to support viable populations of late successional dependent species, including Pacific fishers, American martens, California spotted owls, northern goshawks, and great gray owls. The configuration of habitat in the Monument provides connectivity and heterogeneity

Fire and Fuels desired condition is (2012 Monument Plan p. 24):

1. Fire occurs in its characteristic pattern and resumes its ecological role. Frequent fire maintains lower, manageable levels of flammable materials in most areas, especially in the surface and understory layers. There is a vegetation mosaic of age classes, tree sizes, and species composition, and a low risk for uncharacteristic large, catastrophic fires. The objects of interest are protected; sustainable environmental, social, and economic benefits (such as those associated with tourism) are maintained; and the carbon sequestered in large trees is stabilized.

Air Quality objective: (2012 Monument Plan p. 52):

1. As part of managing prescribed fire and wildfire, develop actions with local air pollution control districts that minimize public exposure to atmospheric pollutants.

2. **Prescribed fire objectives:**

Range of acceptable results

1. Maintain surface fuels in a mosaic that varies between 10-20 tons per acre of dead and down woody material.

- Reduce ground cover 50% to 80%
- Reduce/consume 1 to 10 hour fuels by 80%
- Reduce 100-1000 hour fuels by 20-60%.

2. Restore fuel conditions such that an average live crown base tree height of 20 feet and average flame lengths of 6 feet or lower can be maintained should a wildfire occur under 90th percentile fire weather conditions.

3. Maintain canopy cover to range from 40 to 70 percent in the dominant and co-dominant trees. Create openings that support giant sequoia regeneration in 1 to 10 percent of the grove area (in Evans Complex applies to the areas in which giant sequoias naturally occur).

4. Reduce the stocking and basal areas of shade-tolerant species like white fir and incense cedar to provide more growing space over time for young giant sequoia trees.

Tree mortality in California mixed conifer & oaks:

	Acceptable Range	Desired
1-10"	up to 40%	up to 20 %
11-30"	< 20%	up to 5%
30" +	<10%	up to 5%

Tree mortality in Giant Sequoia:

	Acceptable Range	Desired
1-10"	up to 20%	up to 15 %
11-30"	< 10%	< 5%
30" +	<5%	0%

ELEMENT 6: FUNDING:

A. Cost:

WFPR13				
Unit	**Prep**	**Burning**	**Monitoring**	**Total Cost**
1	$2,588.88	$10,568.00	$674.24	$13,801.12
2	$2,588.88	$10,568.00	$674.24	$13,801.12
Grand Total				**$27,602.24**

WFHF13				
Unit	**Prep**	**Burning**	**Monitoring**	**Total Cost**
1	0.00	$3,917.52	0.00	$3,917.52
2	0.00	$3,917.52	0.00	$3,917.52
Grand Total				**$7,835.04**

- **Total Project Cost: $35,437.28**
- **Total Acres: 169 ac.**
- **Cost per acre: $209.69**

B. Funding source: WFPR13 and WFHF13

ELEMENT 7: PRESCRIPTION

A. Environmental Prescription:

The environmental conditions were determined to give the Prescribed Fire Burn Boss more flexibility to meet objectives. The prescription is not intended to be met while environmental factors (temp, RH, midflame windspeeds and fuel moistures) are all at the upper or lower extremes of the prescription window; instead, a wide prescription gives the Prescribed Fire Burn Boss more opportunity to meet objectives. Examples of this include choosing to burn the units during moderate temperatures when the RH is low, but midflame windspeeds are higher, or when the temperature is high, windspeeds are low and fuel moistures are moderate to high.

The Prescribed Fire Burn Boss is responsible for reviewing the prescription and determining staffing requirements needed based on climatic factors, fuel conditions and potential fire behavior. Prior to ignition, a comparison of individual and collective prescription elements will be made to the current weather conditions, local weather forecasts and any other important factors (such as seasonal effects like drought). The same level of authority required for plan approval will be attained to grant changes to prescription parameters.

If the prescription parameters above are exceeded during ignition operations, conditions will be documented and evaluated by the Prescribed Fire Burn Boss, If conditions are such that continued ignitions will exceed the Range of Acceptable Results, holding actions will be implemented until conditions at the unit will allow for a return to the Range of Acceptable Results.

Fuel and Weather Prescription Guidelines

Fuel Model TU5, TL4, and TL6		
Environmental Variables	**Treatment Window Range**	**Optimum Window**
Temperature	40 to 90 degrees	65 Degrees
Relative Humidity	20 to 60%	35%
Midflame Windspeed	0 to 20 mph	10 mph
Live Fuel Moisture	50 to > 100%	85%
Fuel Moisture: 1hr	4 to >12	8
10 hr	5 to > 13	9
100 hr	6 to > 20	13

B. Fire Behavior Prescription: This information can be used as a guide to the potential range of fire behavior from a free-burning fire, and for contingency planning. The "preferred conditions" under the **Fuel and Weather Prescription Guidelines** were entered into the **BEHAVE** fire modeling system to generate the tables below.

Desired fire behavior Prescription
The desired rate of spread is .4 to 5 chains per hour.
The desired flame lengths are 1-4 ft. and may be attained by backing fire throughout unit.
The desired scorch height is < 10 ft.

TU5			
Predicted Fire Behavior	**Hot Head/ Backing**	**Optimum Head/ Backing**	**Cool Head/ Backing**
ROS (Ch/Hr)	38.0/0.6	20.1/0.4	1.8/0.4
Flame Length (ft.)	15.0/2.2	10.3/1.7	3.2/1.7
Effective Windspeed (mph)	12.6/0.0	10.6/0.0	1.1/0.0
Scorch Height (ft.)	149/2	54/1	11/4
Probability of Ignition (%)	76	39	18

TL 4			
Predicted Fire Behavior	**Hot Head/ Backing**	**Optimum Head/ Backing**	**Cool Head/ Backing**
ROS (Ch/Hr)	8.2/0.1	4.7/0.1	0.5/0.1
Flame Length (ft.)	2.5/0.4	1.7/0.3	0.6/0.3
Effective Windspeed (mph)	11.2/0.0	9.3/0.0	1.3/0.0
Scorch Height (ft.)	2/0	1/0	1/0
Probability of Ignition (%)	76	39	18

TL 6			
Predicted Fire Behavior	**Hot Head/ Backing**	**Optimum Head/ Backing**	**Cool Head/ Backing**
ROS (Ch/Hr)	26.1/0.4	14.5/0.3	1.1/0.2
Flame Length (ft.)	5.6/0.8	3.9/0.6	1.1/0.5
Effective Windspeed (mph)	12.3/0.0	10.4/0.0	1.5/0.0
Scorch height (ft.)	17/0	6/0	2/1
Probability of ignition (%)	76	39	18

Note: Fire behavior modeling has been completed for fuel model TU5, TL4, TL6, keeping in mind that the surrounding areas are similar. The areas with fuel model TU5 have minimal timber intermixed so scorch heights and flame lengths shouldn't affect many trees. BEHAVE plus runs are attached to the burn plan appendices.

Adjacent Fuels – Expected Fire Behavior – Worst Case Scenario

Fuel Model TU5, TL4, and TL6	
Environmental Variables	**Extreme Weather Conditions**
Temperature	96 Degrees
Relative Humidity	15%
Midflame Windspeed	10 mph
Live Fuel Moisture	60%
Fuel Moisture: 1hr	1
10 hr	2
100 hr	5

Predicted Fire Behavior	TU5 Head/Backing	TL4 Head/Backing	TL6 Head/Backing
ROS (Ch/Hr)	39.9/0.7	11.9/0.2	32.1/0.6
Flame Length (ft.)	16.3/2.5	3.2/0.5	6.9/1.1
Effective Windspeed (mph)	11.8/0.0	11.4/0.0	11.0/0.0
Scorch Height (ft.)	219/3	7/0	39/0
Probability of Ignition (%)	100	100	100

Note: Fire behavior modeling has been completed for fuel model TU5, TL4, TL6, and TL7 to show what fire behavior would likely occur during a worst case scenario in fuels adjacent to the project area. BEHAVE plus runs are attached to the burn plan appendices.

When prescription parameters are exceeded it is the Prescribed Fire Burn Boss's responsibility to document:

1. Specific prescription parameters that were exceeded;
2. Time the parameter was exceeded;
3. What management actions were taken in response to the change of conditions; and
4. When conditions fell back within prescription parameters. Exceeded prescription parameters will be documented by the Prescribed Fire Burn Boss in the Table below:

Prescription Parameter Exceeded	Time Exceeded	Action Taken	Time Back within Prescription
Temperature: (40 to 80			
RH: (20 to 60)			
Midflame Windspeed: (0 to 10)			
1hr: (4 to 12%)			
10hr: (5 to 13%)			
100 hr: (20+%)			
1,000hr: (20+ %)			

ELEMENT 8: SCHEDULING

A. Ignition Time Frames/Season(s):

Some of the project area has moderate fuels; they may require wetter conditions to meet objectives. These conditions are expected to be met during the spring or fall. Fire management personnel will determine fuel moistures and evaluate the best burning window to attain a range of acceptable results.

B. Projected Duration:

1-5 days will be required to complete ignition/ holding/ mop-up operations.

C. Constraints:

Units will be within the environmental prescription.
Authorization will be obtained from the San Joaquin Valley Unified Air Pollution Control District.
Road signs must be in place to warn the public of prescribed fire operations in the area.

ELEMENT 9: PRE-BURN CONSIDERATIONS

A. Considerations:

1. On Site:

Document data collected; temperature, wind speed, wind direction, 10-hour fuel moisture (fuel stick or use of an electronic probe), and relative humidity.

The general weather forecast will be monitored 1-3 days prior to ignition. During the ignition stage the weather will be taken at least once every 1 hours or at intervals set by the Burn Boss.

Prior to the prescribed burn, the Prescribed Fire Burn Boss will assess and mitigate for potential hazards. No hazards were present on the unit at the time of the original reconnaissance. Hazards will be communicated to all personnel during the pre-burn briefing and as they are identified during and after the burn.

Any incomplete or improvement to handlines will be determined by the Prescribed Fire Burn Boss and will be completed at the appropriate time by personnel available at the unit.

Prescribed fire signs will be placed at the following locations:

- The intersection of Hwy 180 and 13S42 (Hume Road).
- The intersection of (NM528) General's Highway and Quail Flat.
- The intersection of 14S02 (Burton Road) and 13S26 (Tornado Meadow Road).

2. Off Site

Central California Interagency Coordination Center (CCICC) will be provided a copy of the approved Prescribed Fire Plan prior to implementation.

Notify the following:

1. Adjacent land owners and individuals on contact list
2. Local media
3. CCICC and Hume Lake Ranger District front desk

Smoke approval from San Joaquin Valley Unified Air Pollution Control District will be attained prior to ignition.

Heli-spots, along with GPS coordinates will be identified prior to burning operations.

B. Method and Frequency for Obtaining Weather and Smoke Management Forecast(s): Seasonal weather trends and effects of these trends on the burn units will be considered. Weather forecasts will be monitored to determine possible burn windows. A spot weather forecast will be requested and obtained prior to ignition. The Prescribed Fire Burn Boss and firing boss will discuss the spot weather forecast and determine the potential effects on the prescribed fire. Weather information will be shared with all prescribed burn personnel. Weather observations and forecasts will be filed in the burn plan folder.

Prior to ignition: The general forecast weather will be monitored 1-3 days prior to the ignition. Weather observations will be taken on site, and a spot weather forecast will be obtained. An early observation and spot forecast will also be obtained on the day(s) of ignition. Weather observations may be referenced by using the Cedar Grove RAWS weather station (44719) located 7 air miles to the northwest of the project area. The forecast should be favorable for the next 3 days after ignition is completed. A cold front with wet weather predicted to be coming into the area, even with wind in front of it, is considered favorable; ignition must be within prescription parameters.

During ignition: A Spot Weather Forecast from the National Weather Service is required prior to ignition for each day active ignition is occurring on the burn and days the fire is actively spreading. Projected weather beyond the ignition operation and need from additional spot weather forecasts should be taken into account in order to minimize the risk of a later escape. Local weather phenomena and considerations include evening down slope winds and early morning inversions.

Weather for today, tonight and tomorrow should be requested for spot forecasts. Request a spot at: http://spot.nws.noaa.gov/cgibin/spot/spotmon?site=hnx.

C. Notifications:

It is the Prescribed Fire Burn Boss's responsibility to make a reasonable effort to notify adjacent agencies, land owners, impacted publics, etc. Attempts and/or actual notifications will be documented with date and method by the Prescribed Fire Burn Boss.

Prior to, or on the day of ignition, the Prescribed Fire Burn Boss will notify the following of the intent to burn (ECC, the District Ranger, Forest PIO, etc may make contacts for the Prescribed Fire Burn Boss).

Name	Number	Contact Type/Date	When
Warning Signs on Affected FS Roads and Trails	NA		1 Week Prior
Warning/Closure Signs on Affected FS Roads and Trails	NA		3 Days Prior
Press Release	NA		1 Week Prior
CCICC	(559) 782-3120		Same Day
SJVAPCD	(559) 488-8950		Same Day
Front Desk	(559) 338-2251		Same Day
Sequoia and Kings Canyon National Parks	(559) 565-3195		Same Day
Hume Lake Christian Camp	(559) 335-2000		Same Day
Bearskin Diabetic Camp	(559) 335-2403		Same Day
Kings Canyon Lodge	(559) 335-2407		Same Day

ELEMENT 10: BRIEFING

Briefing Checklist:

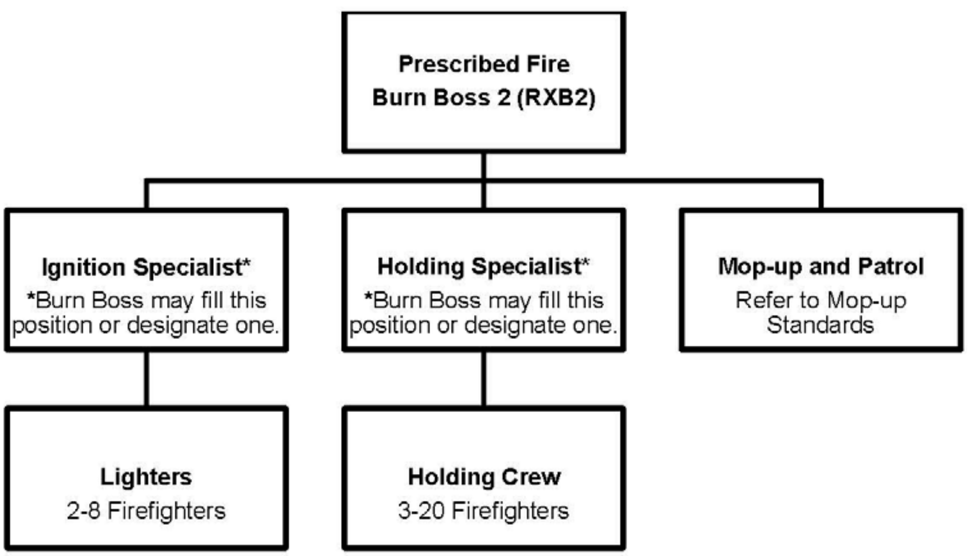
- ☐ Burn Organization
- ☐ Burn Objectives
- ☐ Description of Burn Area
- ☐ Expected Weather & Fire Behavior
- ☐ Communications

- [ ] Ignition plan
- [ ] Holding Plan
- [ ] Contingency Plan
- [ ] Wildfire Conversion
- [ ] Safety

ELEMENT 11: ORGANIZATION AND EQUIPMENT

A. Positions:

A Prescribed Fire Burn Boss will be on site to oversee prescribed fire operations. If conditions are favorable and personnel are available several units may be ignited simultaneously. In this case either the Prescribed Fire Burn Boss or a firing boss will be located at each unit. The Prescribed Fire Plan will identify the minimum organization needed to accomplish an individual burn. No less than the organization described in this Prescribed Fire Plan shall be used to execute the burn. Personnel assigned to positions in the prescribed fire organization will meet all qualifications for their position identified in FSM 5140. This project organization chart will be completed for each position identified prior to implementation, using qualified and available personnel.



Qualified Personnel	Firing and Containment	Mop-up	Patrol
Burn Boss	1	1	0
Firing Boss	0-1	0-1	0-1
Holding Crew	3-20	3-10	1-10

Lighting Crew	2-8	0-2	0

B. Equipment/ Supplies:

Equipment and Supplies/Unit	Firing and Containment	Mop-up	Patrol
T3 Engine	1-2	1-2	0-1
Porta-tank	1-3	0-3	0
Ignition Devices	4-12	0-1	N/A
Hand-tools	3-20	3-20	2-10

ELEMENT 12: COMMUNICATION

A. Radio Frequencies

1. Command Frequency(s): SQF Channel 4
   Rx. Freq. 168.7750 Tx. Freq. 170.6000 Tone 1, 2, or 12

2. Tactical Frequency(s): NIFC Tac 2
   Rx. Freq.168.2000 Tx. Freq. 168.2000

3. Air Operations Frequency(s): R5 Air to Ground 5
   Rx. Freq. 167.4750 Tx. Freq. 167.4750

A. Telephone Numbers:

CCICC Dispatch:	559-781-5780
Hume Lake Ranger District (Dunlap office):	559-338-2251
National Park Service:	559-565-3195

ELEMENT 13: PUBLIC AND PERSONNEL SAFETY, MEDICAL

A. Safety Hazards:

Firefighter

- Travel to and from the project area… Mitigated by reviewing JHA's, fatigue management, signing, and use of emergency lights.
- Exposure to smoke… Mitigated by rotating personnel out of smoke and review JHAs.
- Exposure to burning snags… Mitigated by limiting exposure time, felling of snags along unit boundaries, and JHA review.
- Footing on uneven terrain… Mitigated by fatigue management, Identifying dangerous areas in briefing, and review of JHAs.
- Burning, rolling material and/or rocks… Mitigated by limiting exposure time, review good communications, and Review JHAs.

Public

- Project personnel traffic in the area… Mitigated by site recon, media releases, signing, and use of emergency lights

- Possible exposure to smoke… Mitigated by media releases, contacting near-by home owners, use of wind direction, and use good environmental conditions and burn windows

B. Measures Taken to Reduce the Hazards:

- Signs will be posted in the Boulder Creek Project area indicating burn activities ahead. Signs will be in place prior to any ignitions on the burn.
- A safety briefing will be conducted prior to ignition, mop-up (if needed) and patrol operations. A thorough review of the Job Hazard Analysis will be conducted and any specific hazards within the burn will be identified prior to the day of the burn and mitigation actions will be implemented (i.e.: hazardous snags fallen, adequate escape routes provided, etc.).
- All personnel exposed to smoke conditions will limit such exposures to a minimum as identified during the briefing by either staying out of smoke or personnel rotations.
- Safety briefings will be conducted each day of ignition. All personnel who are within the active burn area are required to wear personal protective equipment.
- The Burn Boss will be notified of any non-operational personnel visiting the project area. Qualified burn personnel may be required to accompany non-operational personnel that have reason to be in the project area (i.e.: Agency administrators, other agency personnel, etc.).

C. Emergency Medical Procedures:

All emergency medical procedures will be implemented by following the chain of command. The Prescribed Fire Burn Boss will be responsible for overseeing the implementation of emergency medical procedures in response to an emergency. The Prescribed Fire Burn Boss will also be responsible for the management of the prescribed fire as well. An injury that requires medical attention will become the priority operation. If an injury occurs, ignition may need to be postponed until the emergency situation has been resolved.

An emergency medical procedure will result if and when an injury occurs that is severe enough to require medical attention beyond the medical attention that is available on site. Everyone on the prescribed burn should be current with their First Aid and CPR trainings as per FS policy. Usually, a First Responder and/or EMT will be present as well. The Prescribed Fire Burn Boss will identify any First Responders/EMTs on the day of the burn and add the contacts to the organization chart.

D. Emergency Evacuation Methods:

Life threatening emergencies will be dealt with through life flight, and all non-life threatening emergencies will use local ambulances for transportation.

Prior to ignition the Prescribed Fire Burn Boss may establish a Lat and Long for an emergency helispot that is closer to the unit.

Medical Facility	Location	LAT/LONG	COMMENTS
Sierra-Kings District Hospital	372 W. Cypress Ave Reedley, CA 559-638-8155	36° 50.19/ 119.45.10	Type 3 Pad/ Grnd Ambulance/No Burn Unit
Community Regional Medical Center	2823 Fresno St. Fresno, CA 93721 559-459-6000	36° 44.581/119.47.107	24 Hour Facility: Level I Trauma and Burn Center
Clovis Community	2755 Herndon Ave. Clovis, CA 93611 (559) 324-4000	36° 50.315 119° 39.544'	1 Helipad/lighted – ground level location

FIELD MEDICAL EVACUATION PLAN
Sequoia National Forest

Project Name:	Boulder Creek	**Forest:**	Sequoia	**District:**	Hume Lake RD
Date:	02/28/2013	**Incident Number:**		**Plan Prepared By:**	Paul Leusch

Qualified First Responders or the most senior qualified medical provider will provide patient assessment and first aid. Evacuation of serious injuries will be coordinated with the Central California Incident Communications Center (CCICC). Minor injuries will be treated, and transported by vehicle to a medical facility as necessary. **Patient Advocacy – The following District Employees will be notified as quickly as possible so that a person or arrangements will be made to have an employee meet the patient at the hospital or medical facility: John Exline – District Ranger, Irma Contreras–Administrative Liaison, or the District Staff Officer who that employee works under. Phone number for these employees is the District Office (559) 338-2251. After hours, coordinate through CCICC.**

Contact							
Contact:		Sequoia Dispatch (CCICC)		**Phone Number:**		559-781-5780	
Frequency	**Rx:**	168.7750	**Tx:**	170.6000	**Tone:**	1, 2, or 12	
Alternate Contact:		Sierra Dispatch		**Phone Number:**		559-291-1877	
Injury Information							
Nature of Injury: Avoid using names							
Number to Transport:				**Estimated Weights:**			
Project Location							
Legal:		**Latitude:**	N	**Longitude:**	W		
Narrative: including major landmarks or cross roads							
Hazards: To ground or aviation resources		**Weather Conditions:** Wind speed and direction, visibility, temperature					
Closest Helispot Location (see attached list for District locations)							
Legal:		**Latitude:**		**Longitude:**			
Narrative: including major landmarks or cross roads							

Medical Facility				
Nearest Facility: (See attached list for nearest facility)			**Phone Number:**	
Travel Time:			**Address:**	
Directions:				
24-Hour Facility: Level I Trauma and Burn center		Community Regional Medical Center	**Phone Number:**	559-459-6000
Travel Time:		Grnd-2+ hrs depending on location, Air-15 to 25 min depending on location	**Address:**	2823 Fresno Street Fresno, CA. 93721
Directions:	Take CA-180 W Take exit #59A/Lemoore/Paso Robles onto CA-41 S Take exit #127B/Divisadero St/Tulare St Turn Right on E Divisadero St Bear Left on Fresno St Arrive at 2823 Fresno St, Fresno, CA 93721 Helipad/Lighted Location on roof **Lat:** N 36° 44.581' **Long:** W 119° 47.107'			

HUME LAKE RANGER DISTRICT
EMERGENCY MEDIVAC LOCATIONS

LANDING SITE	LEGAL DESCRIPTION	COORDINATES	SPECIAL HAZARDS	HELISPOT OR HELIPORT	MAXIMUM TYPE ALLOWED
Hume Lake District Office	T 13S, R 26E, NE ¼ Sec. 32	N 36° 45.429' W 119° 09.911'	Power Lines.	Helispot – Emergency Only	Type 3
Pinehurst Work Center	T 14S, R 27E, NE ¼ Sec. 22	N 36° 41.733' W 119° 01.152'	Power Lines.	Heliport	Type 2
McKenzie	T 13S, R 27E, SE ¼ of Sec. 32	N 36° 44.868' W 119° 03.304'	General Public and Roadside Traffic.	Heliport	Type 1
Big Meadows	T 14S, R 29E, SW ¼ Sec. 9	N 36° 43.098' W 118° 50.132'	General Public and Roadside Traffic.	Helispot	Type 2
Hume Lake (Lakeside)	T 13S, R 28E, SE ¼ Sec. 15	N 36° 47.258' W 118° 54.693'	General Public, Structures, and Power Lines.	Helispot – Emergency Only	Type 3
Hume Lake Christian Camp (Baseball Field)	T 13S, R 28E, S ½ Sec. 15	N 36° 47.314' W 118° 55.205'	General Public, Structures, and Power Lines.	Helispot – Emergency Only	Type 3
Yucca Point	T 13S, R 28E, NW ¼ Sec. 2	N 36°49.681' W 118° 54.283'		Helispot	Type 2
Bailey Bridge	T 12S, R 26E, SW ¼ Sec. 22	N 36° 52.242' W 119° 07.861'	General Public and Roadside Traffic.	Helispot	Type 3
Eshom Point	T 15S, R 28E, SW ¼ Sec. 4	N 36° 38.638' W 118° 56.796'	Power Lines.	Helispot	Type 3
Pierce Pond	T 15S, R 28E, NE ¼ Sec. 4	N 36° 39.065' W 118° 56.406'	General Public.	Helispot	Type 3
Quail Flat	T 14S, R 28E, NW ¼ Sec. 11	N 36° 43.333' W 118° 54.564'	General Public and Roadside Traffic.	Helispot	Type 2
Montecito-Sequoia Lodge	T 14S, R 29E, NW ¼ Sec. 19	N 36° 41.789' W 118° 52.348'	General Public and Roadside Traffic.	Helispot	Type 3
Convict Flat	T 13S, R 29E, SW ¼ Sec. 4	N 36° 49.055' W 118° 49.958'	General Public.	Helispot	Type 2
Trimmer Work Center (SNF)	T 12S, R 24E, NW ¼ Sec. 12	N 36° 54.171' W 119° 18.31'	Structures and Power Lines.	Heliport	Type 1

*ALL COORDINATES ARE IN DEGREES DECIMAL MINUTES FORMAT AND IN MAP DATUM NAD 83.

Medical Facilities
Closest to the Hume Lake Ranger District

Kaweah Delta District Hospital | 1 Helipad – roof location | ***Trauma Center Available***
400 West Mineral King
Visalia, CA
(559) 624-2000

Latitude: N 36° 19.682'
Longitude: W 119° 17.68'

1. Go West on CA-180/E Kings Canyon Rd
2. Turn left at CA-63/S Hills Valley Rd
3. Turn left at Avenue 460/CA-63
4. Turn right at Rd 128/CA-63
   Continue to follow CA-63
5. Slight right at CA-63/N Locust St
   Destination will be on the right

Saint Agnes Medical Center | 1 Helipad/lighted – roof location
1303 E. Herndon
Fresno, CA
(559) 450-7827

Latitude: N 36° 50.151'
Longitude: W 119° 45.899'

1. Head West on CA-180/E Kings Canyon Rd
2. Turn right at CA-180/S Clovis Ave
3. Take the ramp onto CA-180 W
4. Take exit 59B to merge onto CA-41N
5. Take exit 134 for Herndon Ave
6. Turn right at E Herndon Ave
   Destination will be on the left

Clovis Community Hospital | 1 Helipad/lighted – ground level location
2755 Herndon Ave.
Clovis, CA 93611
(559) 324-4000

Latitude: N 36° 50.315'
Longitude: W 119° 39.544'

1. Head West on CA-180/E Kings Canyon Rd
2. Turn right at N Temperance Ave
3. Slight left at Temperance Ave
4. Slight right to stay on Temperance Ave
5. Turn right at Coventry Ave
6. Turn right at Herndon Ave

Sierra-Kings District Hospital
372 W. Cypress Avenue
Reedley, CA 93654
(559) 638-8155

1. Head West on CA-180/E Kings Canyon Rd
2. Turn left at S Reed Ave
3. Turn left at W Parlier Ave
4. Turn right toward W Cypress Ave

Community Regional Medical Center 3 Helipads/lighted – roof location ***Burn and Trauma Center Available***
2823 Fresno Street
Fresno, CA
(559) 459-6000

Latitude: N 36° 44.581'
Longitude: W 119° 47.107'

1. Head West on CA-180/E Kings Canyon Rd
2. Merge onto CA-41 S
3. Take the Divisadero St.
4. Turn slight Left onto Fresno St.

ELEMENT 14 TEST FIRE

A. Planned location: Test fire will be conducted in fuels representative of the unit to determine if fire behavior objectives can be met. If test fire results are outside of the fire behavior parameters, the test fire will be mopped up 100% and the burn will be terminated.

B. Test Fire Documentation:

The Prescribed Fire Burn Boss will document the On-Site Pre-burn Weather Conditions, Test Fire Conditions and Results and the Observed Fire Behavior Conditions and Results in the table below:

Weather Conditions					
Date					
Cloud Cover %					
Temperature					
Relative Humidity					
Fine Dead Fuel Moisture					
Wind Speed					

Test Fire Results					
Flame Length					
Rate of Spread					
Smoke Dispersion					
Other	**The following is the process for notifying CCICC on test fire:** **1. Notify CCICC of starting test fire in burn area.** **2. Notify CCICC whether you are proceeding with ignition or not.**				

ELEMENT 15: IGNITION PLAN

A. Firing Methods:

Broadcast Burn

Backing fire method with strips varying from 4 – 10ft. depending upon weather, topography, and fire behavior. Strip width may also be adjusted depending on the fuel loading, clumps of trees, and the number of leave trees in certain areas. The desired results are flame lengths from 1-3 ft. The first strip will be lit at the top of the unit and a progression of strips will then be made to the bottom. The objective is to allow fire to back down through the unit.

Specific firing techniques, patterns, and sequences may be adjusted by the Prescribed Fire Burn Boss and/or the firing boss during ignition.

NOTE: In order to retain as many of the leave trees as possible, the Prescribe Fire Burn Boss and ignition personnel will use appropriate lighting techniques to keep intensities low.

B. Devices:

Drip torches, Fusees and/ or other hand firing devices.

Backing fire method will be used with strips varying from 4 – 10ft. depending upon weather, topography, and fire behavior. Strip width may also be adjusted depending on the fuel loading, location of leave trees and the number of leave trees in certain areas. The desired results are flame lengths from 1 - 3ft.

C. Sequences:

A strip will be lit at the top of the unit and allowed to back through the unit. As needed a progression of strips will then be made to the bottom.

D. Patterns:

Patterns used will be at the discretion of the Prescribed Fire Burn Boss and/or the firing boss. The patterns are based on the fuels, weather and topography present and the unit objectives.

E. Ignition Staffing:

The appropriate organization chart will be completed by the Prescribed Fire Burn Boss prior to implementation and will reflect the ignition staffing organization required, based upon the ignition method of the unit.

ELEMENT 16: HOLDING PLAN

A. General Procedures for Holding:

The Prescribed Fire Burn Boss will be responsible for reviewing specific prescriptions, weather forecasts and climatic factors, fuel conditions and potential fire behavior and available staffing. These factors will be considered when determining the holding personnel needed to maintain the prescribed fire within prescription. Firing, holding, patrol and mop up procedures as required will be identified (Region Directives ref. FSM 5142.4 for mop-up standards definitions and determination). If actions needed to keep the fire within project area exceed the predetermined definition of holding actions, suppression action will be taken.

The Prescribed Fire Burn Boss will identify a holding boss and holding crewmembers prior to ignition. A minimum of one-Type 3 engine with three personnel will be required. Lighters may be transitioned to holding crew during the burn as needed. Reassignment of personnel will be relayed to all personnel on the burn unit.

Water sources are identified as an on-site fold-a-tank and Hume Lake and/or Lakeshore Fire Station turnaround time for engines from Hume Lake will be approximately 1 hour. Suggested area for engine staging is at the intersection of the 13S26 and 13S26A at the top of Unit 3A1 or along the 13S26 at the bottom of both Units 3A1 and 3A2.

After completion of ignition, all forces will transition to holding crew. All control lines will be patrolled and any spot fires will be identified and prioritized by holding boss and Prescribed Fire Burn Boss. Units will be patrolled following ignition to identify problems and prevent escape. Mop-up forces will be assigned to meet appropriate mop-up category of R5 Guidelines.

B. Critical Holding Points and Actions: See Complexity Analysis Elements 1 through 7 and 11.

Mop-up & Patrol Procedures:

Mop-Up: The Holding Specialist (if filled) will initiate mop-up procedures to the extent necessary to put the burn unit into patrol status. The Burn Boss under a hot prescription will require the burn perimeter to be scanned with a hand-held device to enable mop-up crews to detect and extinguish burning materials.

Patrol: Once the burn is put into patrol status, it will be patrolled as outlined in this section, until the unit is declared out. During the patrol period, any hot spots that have the potential to spot outside the line will either be extinguished or monitored until the threat has abated. The Patrol Personnel will immediately notify the Burn Boss or Prescribed Fire Manager of any spot fires, or other threats of escape, and take action to contain and secure the threat. The Burn Boss or the Prescribed Fire

Manager will make the determination of what, if any, additional resources will be sent to assist. In the event an adverse weather condition or forecast occurs prior to the fire being declared out, the Burn Boss will place the fire back into mop-up status. Mop-up and daily patrols will continue throughout the period of adverse weather.

Matrix Chart:

1. Probability of Ignition (PI) is a factor of the receptiveness of the receiving fuel bed to new ignitions from firebrands.
   PI: 10-49 Low potential for new ignitions
   50-69 Moderate potential for new ignitions
   70+ High potential for new ignitions
2. Wind Speed* determines the horizontal force driving firebrands across control lines outside the burn unit. Three wind speed levels are used in the matrix below:
   WS: 0-12 mph. Minimal effect on holding control lines.
   13-24 mph. Significant effect on holding control lines.
   25+ mph. Adverse effect on holding control lines.
   * Nine years of weather records from the Park Ridge RAWS show the following wind frequencies.
   WS: 0-1.3 mph, classified as calm, 34.4% chance of occurrence.
   WS: 1.3 – 4 mph, 28.0% chance of occurrence. Minimal effect on holding control lines.
   WS: 4 – 8 mph, 33.8 % chance of occurrence,
   WS: 8+ mph, < 3.8 % chance of occurrence.

These two factors, PI and WS with general and or spot weather forecasts, will be used to determine mop-up/patrol standards from the matrix below.

PI *6	20' WS *6	Mop-up distance *1,2	Patrol Frequency *3	Fire may be Un-staffed *4	Available Resources *5
10-49	0-12	Burn Boss	Burn Boss	Yes	Burn Boss
	13-24	Burn Boss	Burn Boss	Yes	Burn Boss
	25+	Burn Boss	1 patrol/day	No	5 Firefighters
50-69	0-12	Burn Boss	Burn Boss	Yes	Burn Boss
	13-24	Burn Boss	1 patrol/day	No	5 Firefighters
	25+	Burn Boss	2 patrols/day	No	8 Firefighters
70+	0-12	Burn Boss	1 patrol/day	Yes	5 Firefighters
	13-24	Burn Boss	2 patrols/day	No	8 Firefighters
	25+	Burn Boss	Continuous	No	18 Firefighters

*Notes:

1. Burn boss to dictate required actions.
2. The declaration of regional contingency levels of III or higher by the Forest/Region will require 100% mop-up.
3. Patrol frequency is defined as the number of times in a 24-hour period that the entire control line will be walked.
4. Fire may be un-staffed is defined as under the specified PI and WS listed above with the prescribed burn plan objectives continuing to be met, the burn remaining within the burn unit, and smoke production remaining within parameters, the burn boss may leave the fire unattended. General or spot forecasts will be reviewed daily and weather observations from the closest RAWS will be monitored every 48 hours at a minimum while the burn is un-staffed and until the prescribed fire is declared out.
5. Available resources are the numbers of firefighters established in the contingency plan. This is in addition to the patrol needs.
6. When in patrol/mop-up status, a spot forecast is recommended when the PI is 70+ with predictions of high winds.

C. Minimum Organization or Capabilities Needed: See Element 11, and Complexity Analysis #6, #11 and Section B above Matrix Chart (At the request of the Holding Boss the lighters could be used as contingency resources).

ELEMENT 17: CONTINGENCY PLAN

The Prescribed Fire Burn Boss will initiate the contingency plan in the event that the prescribed fire is not meeting, exceeds, or threatens to exceed:
1.) Project or unit boundary
2.) Objectives
3.) Prescription parameters
4.) Minimum implementation organization
5.) Smoke impacts
6.) Other Prescribed Fire Plan elements

A. Trigger Points:
When on-site holding forces do not contain fire outside of the sale boundary within the current days burning period or, when the Prescribed Fire Burn Boss feels that the on-site holding forces need additional support. Separate contingency plans will not be necessary for the different units within this RXBP, or for different types of ignitions, or for different phases of the burn implementation.

B. Actions Needed:
If the on-site holding forces need additional support, or when the sale area boundary has been breached, the Prescribed Fire Burn Boss will utilize the identified available contingency resources to continue the holding actions already in progress and/or to hold the critical holding points identified by the Prescribed Fire Burn Boss.

C. Additional Resources and Maximum Response Time(s):
The Prescribed Fire Burn Boss will verify and document availability of identified contingency resources and their response time on day of implementation. If contingency resources availability falls below plan levels, actions must be taken to secure operations until identified contingency resources are replaced. The same contingency resource can be identified for multiple prescribed fire projects. However, once a contingency resource is committed to a specific wildland fire action (wildfire, wildland fire use or prescribed fire), it can no longer be considered a contingency resource for another prescribed fire project and a suitable replacement contingency resource must be identified or the ignition will be halted. The Agency Administrator will determine if and when they are to be notified that contingency actions are being taken. If the contingency actions are successful at bringing the project back within the scope of the RXBP, the project may continue. If contingency actions are not successful, and fire cannot be contained within the second burning period, the contingency plan will be initiated.

As part of the contingency planning efforts, adequate suppression resources will be available. These resources may or may not be present at the burn site, but need to be "on call" and available for the specific burn. *For these units, a minimum of 3 fully qualified fire fighters and 1 Type 3 Engine will be available on 2 hour call, and an additional 3 fire fighters will be available within 24 hours of ignition to serve as contingency resources.* Additional crews, engines and other resources may be requested by the Prescribed Fire Burn Boss during contingency actions. The additional contingency resource needs were based on local knowledge of the Hume Lake RD fire staff.

The following table will be completed by the Prescribe Fire Burn Boss prior to ignition and will be updated if additional contingency resources are ordered. The minimum required holding and contingency resources as stated above, will be confirmed in the table below. The remaining resources listed in the table are resources potentially available to the Prescribed Fire Burn Boss, the corresponding type and availability is listed. The Prescribed Fire Burn Boss will document: 1.) The date that the required contingency resources were confirmed; 2.) The date/time contingency resources were ordered; 3.) and the date/time the contingency resources arrived on scene.

Contingency Resource Documentation

Resource Type	Call Sign/ Contact	Availability	Location	Contingency Resource Status
3 Firefighters **(Required)**		2 Hour Call		Avail. Confirmed: Ordered: On Scene:
Type 3 Engine **(Required)**		2 Hour Call		Avail. Confirmed: Ordered: On Scene:
3 Firefighters **(Required)**		Within 24 hrs		Avail. Confirmed: Ordered: On Scene:
Patrol (s) **(Required)**		Within 24 hrs		Ordered: On Scene:
Additional Crew/Engine				Ordered: On Scene:
Type 3 Helicopter		Within 1 hr		Ordered: On Scene:
Type 3 Helicopter		Within 2 hrs		Ordered: On Scene:
Type 3 Air Tanker		Within 2 hrs		Ordered: On Scene:
Type 3 Air Tanker		Within 2 hrs		Ordered: On Scene:

ELEMENT 18: WILDFIRE CONVERSION

The Prescribed Fire Burn Boss will declare a wildfire. A prescribed fire will be declared a wildfire when the assigned Prescribed Fire Burn Boss determines that one or more of the following conditions or events has occurred or is likely to occur, and if these conditions cannot be mitigated within the next burning period by implementing the contingency actions in the RXBP by on-site holding forces and listed contingency resources staged during this operational period:

a. The prescribed fire leaves the planned unit boundary.
b. The fire behavior exceeds limits described in the RXBP and/or the fire is threatening to leave the planned unit boundary.
c. The fire effects are unacceptable.
d. Smoke production must be reduced because of adverse air quality impacts.
e. Local and/or geographic area fire activity escalates and resources committed as contingency or holding forces are needed for re-assignment to other incidents.

After a wildland fire declaration, an escaped prescribed fire cannot be returned to prescribed fire status. However, the full range of suppression options may be used. A WFDSS analysis will define appropriate future management actions.

A. Wildfire Declared By:

The Prescribed Fire Burn Boss has the authority to declare a wildfire. The District FMO and Prescribed Fire Burn Boss will assist the Agency Administrator with information concerning the conversion of a prescribed fire to a wildland fire and begin the WFDSS process. If the Burn Boss determines there are not adequate resources on scene or responding to contain the incident and determines it cannot be mitigated within the next burning period (24 Hrs.), the Burn Boss will use this process:

1. Notify the CCICC of the current status of the burn project and potential situation.
2. Contact will be made by CCICC or the burn boss, to the District FMO, Forest FMO and District Ranger or acting's. That the determination has been made that the prescribed burn is to be declared a wildland fire.
3. At this point, the project will transition to a wildland fire, and will be treated as such. The standard operating procedures for suppression of wildland fires will be initiated.
4. An IC will be delegated to manage the project area, unit and the escaped fire as one incident to the level of management required by completing a complexity analysis.
5. The District Ranger or delegate will begin the WFDSS process.
6. The Forest will notify to Regional Office within 24 Hours.

B. IC Assignment:

The District Duty Officer will assign an appropriately typed IC to an escaped prescribed fire. During the initial stages of the escape fire and during formal wildland fire declaration, the Prescribed Fire Burn Boss assigned to the prescribed fire will most likely assume command as the IC. As the escape fire transitions to extended attack, IC designation may or may not change. The Agency Administrator and the District Duty Officer will determine what type of IC will assume command of the extended attack fire. The assigned IC will be determined based on the outcome of the WFDSS analysis, work-rest guidelines of the current IC, fire complexity rating, etc.

C. Notifications:

When the Prescribed Fire Burn Boss has determined that one or more of the preceding conditions or events has occurred or is likely to occur, they will notify CCICC Dispatch.

Current District Ranger or acting:
(559) 338-2251 ext. 310

District Fire Management Officer or acting:
Neil Metcalf (559) 338-2251 Ext. 320, Cell (559) 310-0456

Forest Fire Management Officer or acting:
Brent Skaggs (559) 784-1500 Ext. 1120, Cell (559) 280-1744

The Forest will notify to the Regional Office within 24 Hours.

D. Extended Attack Actions and Opportunities to Aid in Fire Suppression:

The area approximately five miles to the east of the project area burned in the 2010 Sheep Fire This area will not support extreme fire behavior at this time. Suppression actions may be anchored from this area should the need arise. Water is available throughout the local area for aerial resources and will provide a short turnaround time if required. It is not considered to be an issue due to the local westerly wind flow and downhill slopes located directly west of the project, the 13S05 and 13S09 roads surround the project to the north and west and provide good areas anchor from to contain any escapes to the west.

If an escape prescribed fire is converted to a wildland fire, a suppression response would occur due to Forest Plan Management Area direction for the project area. After a wildland fire declaration, an escaped prescribed fire cannot be returned to prescribed fire status. However, the full range of suppression options may be used. A WFDSS analysis will define appropriate future management actions.

ELEMENT 19: SMOKE MANAGEMENT AND AIR QUALITY

A. Compliance:

The Hume Lake Ranger District is located within the San Joaquin Valley Air Pollution Control District. The morning of the burn, the Prescribed Fire Burn Boss will ensure that proposed burns have been approved.

The CCICC and District office will log smoke related calls and immediately inform the Burn Boss. The Burn Boss will notify SJUAPCD of any potential smoke related problems. Once the burn boss declares the burn is out of prescription in regards to smoke dispersion objectives he/she has the authority to use whatever reasonable means are necessary to bring the prescribed burn into compliance.

The daily 1300 conference call phone number is (888) 858-2144 passcode 9857932#

B. Permits to be Obtained:

No special permits need to be obtained. State Implementation Plans (SIPs) and/or State or local regulations do not require modeling outputs and mitigation strategies and techniques to reduce the impacts of smoke production. An identified person from the district's fire management office, such as the Prescribed Fire Burn Boss, will obtain authorization through the SJVAPCD.

C. Smoke Sensitive Areas/Receptors:

Class I Areas, including some wilderness areas, parks and wildlife refuges are given the highest protection under the Clean Air Act. Class 1 Areas in the Region include the Sequoia and Kings Canyon National Parks and Yosemite National Park. Impact Zones are areas that the Air shed Group has identified as smoke sensitive and/or having existing air quality problems. The closest Impact Zones are Fresno and Kings Canyon National Park.

Class I Areas and the Impact Zones should not be affected by the prescribed fires because burning will be accomplished when good smoke dispersal is predicted. The Boulder Creek 3A1, 3A2 Units are in close proximity to smoke sensitive areas however, the small acreage size of the unit should not produce a large accumulation of smoke.

D. Impacted Areas:

Surrounding areas could be affected by smoke due to night time inversions that could possibly drift down the Kings River drainage; however, the small acreage size of the units should not produce enough smoke to be a health or visibility hazard. Further impacted areas may include the Monarch Wilderness, and Kings Canyon National Park which are in close proximity. The other adjacent areas may include Cedar Grove, Kings Canyon Lodge, and Hume Lake.

E. Mitigation Strategies and Techniques to Reduce Smoke Impacts:

To reduce the impact of smoke, burning will occur on days with good smoke dispersal and appropriate wind direction. This should mitigate any impacts of smoke to surrounding or sensitive areas. The timing of the prescribed burns would also be coordinated with the California Air Resources Board and the San Joaquin Valley Air Pollution Control District in compliance with Title 17, the Smoke Management Program and the Monument Plan. These requirements and the two additional mitigation measures would reduce the potential direct and indirect impacts to air quality from smoke and particulates entering the airshed.

The Giant Sequoia National Monument Management Plan (USDA 2012) contains several standards and guidelines to maintain or improve air quality, of which the following are applicable to the Boulder Project:

- Minimize resource and air quality effects from air pollutants generated by management activities through use of the following control measures:
    - Follow dust abatement procedures.
    - Conduct an air quality analysis for all projects that may impair air quality to determine effects, mitigations, and/or controls.

• Conduct prescribed burning activities in accordance with air pollution control district regulations and with proper prescriptions to assure good smoke management.
• Notify the public before burning.
• Minimize smoke emissions by following best available control measures (BACMs). Avoid burning on high visitor days. Notify the public before burning.
• Coordinate and cooperate with other agencies and the public to manage air quality. Conduct prescribed burns when conditions for smoke dispersal are favorable, especially away from sensitive or class I areas. Use smoke modeling tools to predict smoke dispersion.

ELEMENT 20: MONITORING

A. Fuels Information Required and Procedures:
Fuel sticks will be placed on the burn site in a representative fuels location at least seven days prior to planned ignition and will be measured for fuel moistures. The 10 hour fuel moisture is to be monitored and documented during ignition.

B. Weather Monitoring (Forecasted and Observed) Required and Procedures:
Weather observations should be measured and recorded on an hourly basis during the ignition phase of prescribed fire operations.

The Burn Boss will monitor the general weather forecast daily until the project or unit is declared out. A request for a spot forecast during ignition stage is required.

The Prescribed Fire Burn Boss will document the on-site weather conditions, and observed fire behavior conditions and comments/results in the table below.

Date/Time/Unit	Wet	Dry	RH	Wind Direction	Wind Speed	Wind Gusts	Flame Length	Comments/Results

C. Fire Behavior Monitoring Required and Procedures:
Visual monitoring will be performed on the day of the burn by a designated fire behavior observer. Observations will be documented.

D. Monitoring Required To Ensure That Prescribed Fire Plan Objectives Are Met:
Post burn monitoring will consist of photos or ocular observations as to fuel consumption, fire severity, and mortality. Comparisons will be made with the pre-burn observations and documentation.

E. Smoke Dispersal Monitoring Required and Procedures:
The Prescribed Fire Burn Boss will monitor the observed smoke dispersal during implementation of the burn. Buck Rock Lookout provides an excellent vantage point to observe operations in Boulder Creek. This location will be staffed throughout operations. Additionally two cameras are also located on the lookout that can provide real time online monitoring capabilities. Any additional smoke monitoring will be completed as deemed necessary by the any restrictions and/or advisories recommended by the SJVAPCD.

ELEMENT 21: POST-BURN ACTIVITIES

Post-burn Activities that must be completed:
Document burn day conditions, fire behavior, smoke dispersal, and fire effects.

A. Attainment of Goals and Objectives:

	Met	Not-Met
1. Reduce 1 and 10 hour fuels by greater than 50%	( )	( )
2. Maintenance 40 – 80% ground cover	( )	( )
3. Reduce duff 50% or greater	( )	( )
4. Reduce 1 and 10 hour fuels by greater than 50%	( )	( )

B. Narrative for Objectives "Not Met"

APPENDICES

A. Maps: Vicinity and Project

B. Technical Review Checklist

C. Complexity Analysis

D. Job Hazard Analysis

E. Fire Behavior Modeling Documentation or Empirical Documentation (unless it is included in the fire behavior narrative in Element 7: Prescription)

A: MAPS

1. Vicinity Map:

2. Project Map:

B. TECHNICAL REVIEWER CHECKLIST

PRESCRIBED FIRE PLAN ELEMENTS:	S /U	COMMENTS
1. Signature page		
2. GO/NO-GO Checklists		
3. Complexity Analysis Summary		
4. Description of the Prescribed Fire Area		
5. Goals and Objectives		
6. Funding		
7. Prescription		
8. Scheduling		
9. Pre-burn Considerations		
10. Briefing		
11. Organization and Equipment		
12. Communication		
13. Public and Personnel Safety, Medical		
14. Test Fire		
15. Ignition Plan		
16. Holding Plan		
17. Contingency Plan		
18. Wildfire Conversion		
19. Smoke Management and Air Quality		
20. Monitoring		
21. Post-burn Activities		
Appendix A: Maps		
Appendix B: Complexity Analysis		
Appendix C: JHA		
Appendix D: Fire Prediction Modeling Runs		
Other		

S = Satisfactory U = Unsatisfactory

Recommended for Approval: **Not Recommended for Approval:**

Technical Reviewer ______ Qualification and currency (Y/N) ______ Date ______

□ **Approval is recommended subject to the completion of all requirements listed in the comments section, or on the Prescribed Fire Plan.**

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C: COMPLEXITY ANALYSIS

Project Name **<u>Boulder Creek Unit 3A1, 3A2 Prescribed Burn</u>**

Complexity elements:

1. Potential for Escape

Risk	**Rationale**
Preliminary Rating: ***Low*** Moderate High	Holding actions will be required at the single resource level. Firing is by strip or backing, the number of holders will be relatively small. However due to the probability of ignition being in the mid-seventies this will be a moderate. Residual burning may last up to 3 days with moderate potential for escapes.
Final Rating: ***Low*** Moderate High	Same.
Potential Consequences	**Rationale**
Preliminary Rating: *Low* ***Moderate*** High	There will not be multiple firing operations and the holding operations will be easy to coordinate with one overhead position. Due to the remoteness of the project, no residences are expected to be involved. The project is located directly adjacent to the Horseshoe Bend Giant Sequoia Grove. Giant Sequoia Groves are considered to be national treasures to special interest groups and some social and political concerns from an escape could be expected.
Final Rating: Low ***Moderate*** High	Same.
Technical Difficulty	**Rationale**
Preliminary Rating: ***Low*** Moderate High	Holding operations will be supervised at the single resource boss level and the unit is relatively easy to access for the holding resources. Roads surround the entire project area minimizing holding issues. Normal weather conditions for the area should easily be within the prescription. All key personnel will be from the local area.
Final Rating: ***Low*** Moderate High	Same.

2. The Number and Dependency of Activities

Risk	Rationale
Preliminary Rating: ***Low*** Moderate High	All the project activities are relatively independent from one another. This is accurate in the mop-up and patrol stages as well.
Final Rating: ***Low*** Moderate High	Same.
Potential Consequences	**Rationale**
Preliminary Rating: ***Low*** Moderate High	Project activities have minimal coordination and should not influence any additional risk of an escape, or create a safety issue.
Final Rating: ***Low*** Moderate High	Same.
Technical Difficulty	**Rationale**
Preliminary Rating: ***Low*** Moderate High	Communications and coordination will not experience any problems with the low complexity and relative independence of operations.
Final Rating: ***Low*** Moderate High	Same.

3. Off-Site Values

Risk	Rationale
Preliminary Rating: Low ***Moderate*** High	The Horseshoe Bend Grove is located directly adjacent to the unit and is a limited area of high value a moderate risk occurs due to this. Risk should be mitigated by firing techniques and ignition during prescription of low to moderate fire behavior.
Final Rating: Low ***Moderate*** High	Same.

Potential Consequences	Rationale
Preliminary Rating: Low ***Moderate*** High	In the event of an escape some negative impacts could occur. The vegetation has high recovery rates over a moderate period of time. Horseshoe Bend Giant Sequoia Grove could be affected in an escape.
Final Rating: ***Low*** Moderate High	Mitigation measures are firing techniques like strip firing or backing fires, and prescriptions emphasizing low to moderate intensity burns. Specialists' inputs are considered and mitigations followed.
Technical Difficulty	**Rationale**
Preliminary Rating: ***Low*** Moderate High	Protection of off-site values will be accomplished by having on-site personnel and contingency personnel available in the event of an escape. Mitigation measures are the same as potential consequences and risk.
Final Rating: ***Low*** Moderate High	Same.

4. On-Site Values

Risk	Rationale
Preliminary Rating: ***Low*** Moderate High	There are few on-site values at risk. The project area does include plantations. These plantations can be protected by adjusting firing patterns and techniques to lower the fire intensity to protect the plantations.
Final Rating: ***Low*** Moderate High	Same.
Potential Consequences	**Rationale**
Preliminary Rating: Low ***Moderate*** High	Implementation problems in terms of an escape or higher than anticipated fire behavior could result in some tree mortality, and the above mentioned use of prescriptions emphasizing low intensity burning and strip or backing fire will be used to mitigate these problems.
Final Rating: ***Low*** Moderate High	Protection of on-site values will be accomplished by having on-site personnel and contingency personnel available in the event of an escape or the potential of resource damage.

Technical Difficulty	Rationale
Preliminary Rating: Low ***Moderate*** High	Some brush thinning along roads and line construction will be needed prior to burning in order to maintain control of the unit perimeters. The backing fire tactics that will be utilized in the firing operations will assist in containment as well.
Final Rating: Low ***Moderate*** High	Same.

5. Fire Behavior

Risk	Rationale
Preliminary Rating: Low ***Moderate*** High	One fuel model has been identified as having a moderate spread rate; and moderate flame length, Other fuel models may be present but in small percentages. A heavy forest litter layer with a shrub or small tree understory covers most of the unit. Some moderate are present within and adjacent to the units. This presents moderate fire behavior. Spotting is expected to be short range.
Final Rating: Low ***Moderate*** High	Same.
Potential Consequences	**Rationale**
Preliminary Rating: Low ***Moderate*** High	Fire behavior outside the unit would largely be similar to within the project area.
Final Rating: Low ***Moderate*** High	Same.
Technical Difficulty	**Rationale**
Preliminary Rating: ***Low*** Moderate High	Standard fire precautions are adequate to ensure personnel safety. Receiving spot weather forecasts, employing good firing techniques, and utilizing low fire intensity should ensure personnel safety. At least one additional barrier for containment exist, a road on the lee side of the ridge.
Final Rating: ***Low*** Moderate High	Same.

6. Management Organization

Risk	Rationale
Preliminary Rating: ***Low*** Moderate High	Two levels of supervision will be utilized. The burn boss, firing boss, and holding boss will be staffed, and then the lighters and holders will have adequate supervision. All personnel will hold the proper red card rating for the position they hold.
Final Rating: ***Low*** Moderate High	Same.
Potential Consequences	**Rationale**
Preliminary Rating: ***Low*** Moderate High	Communication and supervision levels required are minimal and are within the 5-7 supervision to worker ratio covering span of control and should present little to no problems.
Final Rating: ***Low*** Moderate High	Same.
Technical Difficulty	**Rationale**
Preliminary Rating: ***Low*** Moderate High	The district has several personnel available to fill all pertinent positions with outside personnel being utilized when district shortages exist due to training, fire assignments, etc. The area also has personnel from other local agencies: the Sierra National Forest, Sequoia and Kings Canyon National Park, and Cal Fire. These personnel are familiar with local factors and can be used to fill pertinent positions if needed or as contingency resources.
Final Rating: ***Low*** Moderate High	Same.

7. Public and Political Interest

Risk	Rationale
Preliminary Rating: Low ***Moderate*** High	The prescribed fire would be visible to the communities of Dunlap, Miramonte, Pinehurst, Badger, Grant Grove and Hume Lake. Smoke from the project is of moderate political interest, the district lies within an air basin that is in severe attainment problem for several pollutants. Small burn windows exist due to this problem. An escape would also cause additional interest if it threatened the adjacent Horseshoe Bend Giant Sequoia Grove.
Final Rating: Low ***Moderate*** High	Same.
Potential Consequences	**Rationale**
Preliminary Rating: ***Low*** Moderate High	Unexpected or adverse events would attract some public attention due to the proximity of the burn to the Horseshoe Bend Giant Sequoia Grove. News releases and local briefings would be required and would be handled by the Public Affairs Officer. Public Affairs Officer involvement would be minimal and at most would require a press release issued and/or phone call to the local media agencies.
Final Rating: ***Low*** Moderate High	Same.
Technical Difficulty	**Rationale**
Preliminary Rating: ***Low*** Moderate High	Requires no special fire information function beyond the normal pre-burn notification to affected publics, roadside signing will also be utilized. Public Affairs Officer involvement would be minimal and at most would require a press release issued and phone call to the local media agencies.
Final Rating: ***Low*** Moderate High	Same.

8. Fire Treatment Objectives

Risk	Rationale
Preliminary Rating: ***Low*** Moderate High	To achieve objectives low to moderate fire intensity will be employed, which does limit the size of the burn windows that exists. Monitoring will take place and include photo points with basic burn and post burn information recorded.
Final Rating: ***Low*** Moderate High	Same.
Potential Consequences	**Rationale**
Preliminary Rating: ***Low*** Moderate High	This unit is small portion of the Boulder Creek Fuels Restoration Project that will be completed in future years. Because of the small size, other management activities in the Boulder Creek Fuels Restoration Project are not directly dependent on the completion of this project. There will be many opportunities throughout the year to meet objectives.
Final Rating: ***Low*** Moderate High	Same.
Technical Difficulty	**Rationale**
Preliminary Rating: ***Low*** Moderate High	Limitations existing for this project include small burn windows to meet smoke management concerns, and maintaining a low or moderate intensity for protecting the residual stand and preventing an escape. Pre-burn monitoring is needed to determine when the unit is in prescription. During-burn monitoring is necessary to determine if the prescribed fire objectives are being met.
Final Rating: ***Low*** Moderate High	Same.

9. Constraints

Risk	Rationale
Preliminary Rating: Low ***Moderate*** High	The constraints that exist include smoke management limitations and limited windows to meet those concerns. Steep slopes and some road issues will limit access to heavy equipment and engine use. Fire behavior must be kept to low or moderate fire intensity to protect the residual stand and to prevent possible escapes.
Final Rating: Low ***Moderate*** High	Same.
Potential Consequences	**Rationale**
Preliminary Rating: Low ***Moderate*** High	Some windows may be unavailable due to air quality issues and are the same as covered under risk.
Final Rating: Low ***Moderate*** High	Same.
Technical Difficulty	**Rationale**
Preliminary Rating: Low ***Moderate*** High	Constraints moderately increase the difficulty of the project. Due to existing constraints the time needed to complete the project may need to be increased slightly.
Final Rating: Low ***Moderate*** High	Same.

10. Safety

Risk	Rationale
Preliminary Rating: ***Low*** Moderate High	Significant safety issues have been identified. Detailed briefings will be utilized. Mitigation strategies such as falling snags prior to project initiation, and the use of low and moderate intensity fire will be implemented. Safety will be further enhanced by firing activities that include a backing or strip firing method. Activities are such that multiple activities will not be occurring at the same time.
Final Rating: ***Low*** Moderate High	Same.

Potential Consequences	**Rationale**
Preliminary Rating: ***Low*** Moderate High	Minimal potential for serious accidents to firefighters and the public exist. All safety mitigations will be enforced from maintaining proper work rest guidelines, to following the 10 standard firefighting orders and the 18 watchout situations.
Final Rating: ***Low*** Moderate High	Same.
Technical Difficulty	**Rationale**
Preliminary Rating: ***Low*** Moderate High	Safety concerns will be mitigated through LCES. A briefing will be held before each operational period and cover the JHA that has been completed. The use of signage will increase the public's awareness to potentially hazardous situations that will be present during the implementation phase of the project.
Final Rating: ***Low*** Moderate High	Same.

11. Ignition Procedures/Methods

Risk	**Rationale**
Preliminary Rating: ***Low*** Moderate High	Backing and Strip firing will be employed to reduce mortality within the unit and to help prevent an escape. More than one burner may be utilized with a drip torch being the primary ignition source but managed by an ignition specialist. The entire project area is readily visible to the Ignition Specialist/Burn Boss.
Final Rating: ***Low*** Moderate High	Same.

Potential Consequences	Rationale
Preliminary Rating: ***Low*** Moderate High	Firing methods will be by hand but multiple burners may be utilized that will need to be coordinated by a firing boss. The firing boss will need to coordinate with the holding boss. In the event of spots or a slopover, firing will cease until said problem is corrected. Opportunities for remedial actions or corrections are available in the event of problems.
Final Rating: ***Low*** Moderate High	Same.
Technical Difficulty	**Rationale**
Preliminary Rating: ***Low*** Moderate High	Ignition patterns will be designed to minimize mortality loss and maximize our ability to prevent an escape. Multiple burners may be employed and even split into squads on occasion. Most units will see the burning start from the top and worked towards the bottom in a strip firing manner. Different types of ignition devices may be used to achieve objectives. The ignition pattern will require direct control of the lighters to achieve project objectives and manage safety concerns.
Final Rating: ***Low*** Moderate High	Same.

12. Interagency Coordination

Risk	Rationale
Preliminary Rating: ***Low*** Moderate High	The project involves land on just the Hume Lake Ranger District of the Sequoia National Forest. Only one piece of private land exists within a mile of the project area and would not be directly affected by an escape.
Final Rating: ***Low*** Moderate High	Same.

Potential Consequences	Rationale
Preliminary Rating: ***Low*** Moderate High	The main coordination issue is with the air quality district and receiving a daily authorization to burn. Otherwise, the project can be completed as planned.
Final Rating: ***Low*** Moderate High	Same.
Technical Difficulty	**Rationale**
Preliminary Rating: ***Low*** Moderate High	No special issues exist. If needed, interagency resources may be readily available from Sequoia-Kings Canyon National Park with few or no restrictions on their use.
Final Rating: ***Low*** Moderate High	Same.

13. Project Logistics

Risk	Rationale
Preliminary Rating: ***Low*** Moderate High	Adequate burn mix will need to be on hand and the district has a burn trailer that can meet those needs. No other specialized equipment or communications needs have been identified. Project duration is 2 days or less.
Final Rating: ***Low*** Moderate High	Same.
Potential Consequences	**Rationale**
Preliminary Rating: ***Low*** Moderate High	No problems related to logistical problems exist that add to control concerns exist. Completion of the project should be routine.
Final Rating: ***Low*** Moderate High	Same.

Technical Difficulty	Rationale
Preliminary Rating: ***Low*** Moderate High	No logistical support problems are anticipated, supervisors can support their own needs. Supplies and personnel are readily available and easy to obtain.
Final Rating: ***Low*** Moderate High	Same.

14. Smoke Management

Risk	Rationale
Preliminary Rating: Low ***Moderate*** High	The district is located in an air shed that is in severe attainment zone with EPA for certain pollutants including ozone and PM 2.5. So coordination with the air quality district is important. If normal local wind patterns change some smoke may drift to Hume Lake, but most of the smoke will flow to the east towards Sequoia-Kings Canyon National Park.
Final Rating: Low ***Moderate*** High	Same.
Potential Consequences	**Rationale**
Preliminary Rating: Low ***Moderate*** High	Some down canyon flow will occur and push some smoke into the Kings River drainage. The Kings River Drainage is a major drainage that leads to the highly populated San Joaquin Valley. Given normal local wind patterns this does not present an issue unless some unforeseen wind event were to occur. Smoke exposure to firefighters and the public are expected to be minimal.
Final Rating: Low ***Moderate*** High	Same.

Technical Difficulty	Rationale
Preliminary Rating: Low ***Moderate*** High	Authorization from the air quality district will be attained before ignition is started; the wind direction will be validated by the burn boss after ignition is initiated with the approved direction being every direction but an easterly flow.
Final Rating: Low ***Moderate*** High	Same.

COMPLEXITY RATING SUMMARY

RISK	OVERALL RATING	MODERATE
POTENTIAL CONSEQUENCES	OVERALL RATING	LOW
TECHNICAL DIFFICULTY	OVERALL RATING	LOW
SUMMARY COMPLEXITY RATING		MODERATE

RATIONALE: The Boulder Creek Unit 3A1, 3A2 Prescribed Burn is rated as a ***Moderate*** complexity prescribed fire. The achievement of project objectives will require cooperation and communication among the management organization. This teamwork will allow the organization to properly identify the complexities involved (fuel loading, depth, and continuity) and select the prescription parameters that provide the best opportunity for successful completion of the burn. The Risk category scores an overall rating of moderate, the Potential Consequences has a rating of low, and the Technical Difficulty scores an overall rating of low. The Summary Complexity Determination was rated as a moderate. This rating was assigned based on Fire behavior having a moderate risk and moderate potential consequences, Constraints and Smoke Management having moderate risk, moderate potential consequences, and moderate technical difficulty. Based on the overall complexity, an RXB2 is recommended.

Prepared by: ______________________________ Date: ________________

Approved by: ______________________________ Date: ________________

D. JOB HAZARD ANALYSIS

FS-6700-7 (2/98)

U.S. Department of Agriculture Forest Service	1. WORK PROJECT/ACTIVITY Prescribed Fire	2. LOCATION Various locations on Hume Lake Ranger District	3. UNIT 051351
JOB HAZARD ANALYSIS (JHA) References-FSH 6709.11 and -12 (Instructions on Reverse)	4. NAME OF ANALYST Paul Leusch	5. JOB TITLE District Fuels Officer	6. DATE PREPARED 2/12/2013

7. TASKS/PROCEDURES	8. HAZARDS	9. ABATEMENT ACTIONS Engineering Controls * Substitution * Administrative Controls * PPE
*Travel to, from and on Project	Motor Vehicle accident. Slippery road surfaces. Soft Shoulders Narrow roadways Weather Smoke Darkness Other road Users Backing	Perform peruse inspections on equipment. Observe the "Circle of Safety" rule. All FS employees who operate Government vehicles shall hold a valid state driver's license with proper endorsements for the size and class being driven and a FS issued identification card indicating the type of vehicle or equipment the operator is authorized to operate. (FSM 7134.1). Use seat belts. Drivers must attend a FS or National Safety Council defensive driving course at least every 3 years. Identify road conditions during briefings. Post road guards if needed. Mark hazards. Use headlights. Scout roads and identify turnouts before ignition of project. Maintain radio communications. Provide road system map for project. Use backers and chock vehicle's tire. Have vehicles facing out. Know and observe all state and local traffic regulations.
*Qualifications For assigned Position	Lack of Experience	Employees recruited for burn assignments shall meet age,health and physical requirements established for regular firefighting duties. (5109.16) Also meet Prescribed Burn qualifications.
*Briefing / Tailgate Safety & Health Sessions	Lack of Communications	Provide Briefings and Tailgate Safety Sessions. Document briefings and sessions. Clarify firing order, organization responsibilities, communications, hazards, weather and expected fire behavior.
Protective Clothing and Equipment	Injuries Falls Burns	Wear approved hard hat with chin strap, safety glasses, flame resistant fabric pants and shirts NPFA 1977 compliant. keep sleeves rolled down. Avoid undergarments and socks made of, polyester, nylon or acrylic. Wear leather, lace type, boots with skid resistant soles, and tops at least 8" high. Carrying drinking water and fire shelter. Wear OSHA approved firefighting gloves. Wear hearing protection when working around equipment where noise level exceeds 85 dba. Wear additional protective equipment as dictated by local conditions and exposure to special equipment.
*Lighters	Injuries, Falls, Snags Bees Snakes Smoke Rolling material	Always have an escape route. Maintain LCES. Follow the Standard Fire Orders and Watch Out Situations. Maintain communications with other lighters and Firing Boss. Hand Held radios shall be provided to all lighters. Lighters shall be trained in the use of Drip Torches. Do not fill drip torches near ignition sources. Do not spill burn mix on clothing. Be alert to foreign objects dumped in burn pile.
*Fuel Mixing	Burns Spills Fuel saturated clothing and boots Improper labeling Explosive	Transport fuel in approved, labeled containers secured in vehicle beds. Park and secure vehicles hauling flammables / combustibles in a separate, predetermined, safe area. No smoking within 25 feet of mixing and filling area. Do not fill or mix in pick ups bed with bed liners. Avoid use of cellular phones in and around fill or mixing area. Avoid fuel contact with bare hands, clothing and boots. Provide pour spouts. Follow fuel mixture ratio in the Health and Safety Code Handbook 25.14c sub-part 2.

*Holding / Mop Up / Patrol Crew	Smoke, Burns, Falls, Lifting Injuries Bees Snakes Posion Oak Snags Rolling Material Heat Stress Dehydration Eye Injuries CO Posioning	Wear PPE's listed above. Protective clothing and equipment shall be the same as required for firefighting. LCES, Follow Standard Fire Orders and Watch Out Situations. Receive briefing from Holding and Mop Up Boss. Identify and mark hazards in work area. Use warning lights and provide traffic control on roadways during smoky and nights operations. Maintaining a high level of aerobic fitness is one of the best ways to protect yourself against heat stress. Drink lots of fluids before, during and after work. Periodically rotate crews from work sites with high levels of smoke to areas of less smoke or smoke free areas. Set a reasonable work pace and allow adequate rest breaks while on the project. Crews shall follow all guidelines in the NWCG Fireline Handbook 3 Chapter 1 Firefighting Safety (Rev. 3/04). Maintain communications with the CCICC Frequencies in Burn Plan ELEMENT 12: COMMUNICATION. Monitor personnel for symptoms and behavior associated with CO exposure and take appropriate action when necessary.
Hand Tools Pitch Forks	Puncture Wounds	Ensure that tools remain in safe condition through periodic inspection and repair. Monitor employee performance periodically to ensure proper methods are used. Handles must be free of splinters, splits and cracks. Pitch forks not in use on the project should be stored standing with forks in ground.
Workplace	Injury or Threat of violence	Violence occurs at different levels of intensity, and usually increases overtime. In order to prevent violence from escalating, employees and supervisors need to pay attention to the work environment, recognize the signs of possible violence early, and take all necessary actions to reduce the risk to life and property. Violent people may come from inside or outside your organization. Call CCICC for law enforcement if needed.
*Emergency Evacuation Procedures (EEP)	Illness/Injuries	On site FS engines or Patrols shall have BLS equipment to initiate basic life support until EMS arrives. Notify CCICC request medical response from responsible medical first responders. Provide type of injury, location, access and number of patients. Follow Fresno and/or Tulare County EMS protocol. Identify EMT's and available medical equipment on project during briefing / tailgate safety session. Notify supervisor of injury. Complete necessary paperwork.

10. LINE OFFICER SIGNATURE	11. TITLE	12. DATE
	District Ranger	

Previous edition is obsolete (over)

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JHA Instructions (References-FSH 6709.11 and .12)

The JHA shall identify the location of the work project or activity, the name of employee(s) writing the JHA, the date(s) of development, and the name of the appropriate line officer approving it. The supervisor acknowledges that employees have read and understand the contents, have received the required training, and are qualified to perform the work project or activity.

Blocks 1, 2, 3, 4, 5, and 6: Self-explanatory.

Block 7: Identify all tasks and procedures associated with the work project or activity that have potential to cause injury or illness to personnel and damage to property or material. Include emergency evacuation procedures (EEP).

Block 8: Identify all known or suspect hazards associated with each respective task/procedure listed in block 7. For example:

a. Research past accidents/incidents
b. Research the Health and Safety Code, FSH 6709.11 or other appropriate literature.
c. Discuss the work project/activity with participants
d. Observe the work project/activity
e. A combination of the above

Block 9: Identify appropriate actions to reduce or eliminate the hazards identified in block 8. Abatement measures listed below are in the order of the preferred abatement method:

a. Engineering Controls (the most desirable method of abatement). For example, ergonomically designed tools, equipment, and furniture.
b. Substitution. For example, switching to high flash point, non-toxic solvents.
c. Administrative Controls. For example, limiting exposure by reducing the work schedule; establishing appropriate procedures and practices.
d. PPE (least desirable method of abatement). For example, using hearing protection when working with or close to portable machines (chain saws, rock drills portable water pumps)
e. A combination of the above.

Block 10: The JHA must be reviewed and approved by a line officer. Attach a copy of the JHA as justification for purchase orders when procuring PPE.

Blocks 11 and 12: Self-explanatory.

Emergency Evacuation Instructions (Reference FSH 6709.11)

Work supervisors and crew members are responsible for developing and discussing field emergency evacuation procedures (EEP) and alternatives in the event a person(s) becomes seriously ill or injured at the worksite.

Be prepared to provide the following information:

a. Nature of the accident or injury (avoid using victim's name).
b. Type of assistance needed, if any (ground, air, or water evacuation)
c. Location of accident or injury, best access route into the worksite (road name/number), identifiable ground/air landmarks.
d. Radio frequency(s).
e. Contact person.
f. Local hazards to ground vehicles or aviation.
g. Weather conditions (wind speed & direction, visibility, temp).
h. Topography.
i. Number of person(s) to be transported
j. Estimated weight of passengers for air/water evacuation.

The items listed above serve only as guidelines for the development of emergency evacuation procedures.

JHA and Emergency Evacuation Procedures Acknowledgment

We, the undersigned work leader and crew members, acknowledge participation in the development of this JHA (as applicable) and accompanying emergency evacuation procedures. We have thoroughly discussed and understand the provisions of each of these documents:

SIGNATURE	DATE	SIGNATURE	DATE
Work Leader			

E. FIRE BEHAVIOR MODELING DOCUMENTATION OR EMPIRICAL DOCUMENTATION

A.8. Supplemental Information for Chapter 8

Table A.8-1 Corresponding table of estimated wildfire-$PM_{2.5}$ illnesses (95% confidence interval) from sensitivity analyses presented in Figure 8-1 and Figure 8-2.

Case Study	Scenario	Asthma ED Visits	Hospital Admissions	
			Respiratory	Cardiovascular
Timber Crater 6 (TC6)	Actual fire	0.4 (0.31 to 0.5)	0.17 (0.01 to 0.31)	0.07 (−0.01 to 0.14)
	Scenario 1 (small)	0.24 (0.19 to 0.28)	0.1 (0.01 to 0.18)	0.04 (−0.01 to 0.09)
	Scenario 2a (large)	1.5 (1.2 to 1.8)	0.69 (0.05 to 1.3)	0.26 (−0.05 to 0.55)
	Scenario 2b (largest)	2.3 (1.8 to 2.7)	1.1 (0.08 to 1.9)	0.41 (−0.09 to 0.87)
	Prescribed fires	0.07 (0.05 to 0.08)	0.03 (0 to 0.06)	0.01 (0 to 0.02)
Rough Fire	Rough Fire (actual)	100 (83 to 120)	40 (2.3 to 74)	15 (−3.2 to 32)
	Rough Fire (Scenario 1)	62 (50 to 74)	24 (2 to 44)	8.6 (−1.85 to 19)
	Rough Fire (Scenario 2)	110 (85 to 120)	42 (3 to 77)	16 (−3.4 to 33)
	Sheep Complex Fire	15 (12 to 18)	5.4 (0.4 to 10)	2 (0.4 to 4)
	Boulder Creek Fire (proposed prescribed fire)	2.7 (2.1 to 3.2)	0.9 (0.06 to 1.7)	0.3 (−0.007 to 0.7)

ED = emergency department; $PM_{2.5}$ = particulate matter with a nominal mean aerodynamic diameter less than or equal to 2.5 μm; TC6 = Timber Crater 6.

A.9. Supplemental Information for Chapter 9

No supplemental information.

A.10. Quality Assurance

A.10.1. Quality Assurance Summary

The use of QA and peer review helps ensure that the U.S. EPA conducts high-quality science assessments that can be used to help policymakers, industry, and the public make informed decisions. Quality assurance activities performed by the U.S. EPA ensure that environmental data are of sufficient quantity and quality to support the Agency's intended use. The work within this report was conducted under the Agency's quality assurance program for environmental information. The report *Comparative Assessment of the Impacts of Prescribed Fire Versus Wildfire (CAIF): A Case Study in the Western U.S.* is classified as Influential Scientific Information (ISI), which is defined by the Office of Management and Budget (OMB) as a scientific assessment that is novel, controversial, or precedent-setting, or has significant interagency interest (Bolton, 2004). OMB requires an ISI to be peer reviewed before dissemination. To meet this requirement, the U.S. EPA had an independent peer review conducted by Westat, Inc. Peer-review comments provided by Westat, Inc. were considered in the development of the CAIF Report.

Agency-wide, the U.S. EPA Quality System provides the framework for planning, implementing, documenting, and assessing work performed by the Agency, and for carrying out required quality assurance and quality control (QA/QC) activities. Additionally, the Quality System covers the implementation of the U.S. EPA Information Quality Guidelines (U.S. EPA, 2002). This report follows all Agency guidelines to ensure a high-quality document.

Within the U.S. EPA, Quality Management Plans (QMPs) and QAPPs are developed to ensure that all Agency research meets a high standard for quality. U.S. EPA has developed a QMP, *Comparative Assessment of the Impacts of Prescribed Fire Versus Wildfire (CAIF): A Case Study in the Western U.S.* (with QA Track ID: L-HEEAD-003289-QP-1-7) specific to the research conducted for this report. In addition, the EPA developed three QAPPs to describe the technical approach and associated QA/QC procedures for the different research used to develop this report. The QAPP, *Comparative Assessment of the Impacts of Prescribed Fire Versus Wildfire (CAIF): A Case Study in the Western U.S*. (with QA Track ID: L-HEEAD-0032689-QP-1-4) details the technical approach and QA/QC procedures for the secondary data analysis contributing to Chapter 3, Chapter 4, Chapter 5, and Chapter 6 of this report. The technical approaches and QA/QC procedures used for the CMAQ and BenMAP models research of Chapter 7 and

Chapter 8 of this report are identified in the QAPP Wildland Fire Leadership Council Fire Benefits Project (with QA Track ID: OAR-OAPS-HEID-0033256-QP-1-0) and its amendment, Amendment 1 to Quality Assurance Project Plan (QAPP) for Wildland Fire Leadership Council Fire Benefits Project (with QA Track ID: OAR-OAPS-HEID-0033256-QP-1-1). The technical approaches and QA/QC procedures used for the VELMA model research also included in Chapter 7 of this report are identified in the QAPP VELMA Modeling (with QA Track ID: L-PESD-0030840-QP-1-2). All QA objectives and measurement criteria detailed in the QAPPs have been employed in developing this report. U.S. EPA QA staff are responsible for the review and approval of all quality-related documentation. Because this is an ISI, U.S. EPA QA staff performed two separate Technical System Audits on the CAIF Report in February 2021 and July 2021. These audits verified that the appropriate QA/QC procedures and reviews were adequately performed and documented.

A.10.2. Peer-Review Summary

The CAIF Report underwent an external letter peer review by Westat, Inc. from April 19, 2021 through May 6, 2021. The peer-review report will be available on the EPA Peer-Review Agenda website.

A.11. References

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