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Incident command post exposure to polycyclic aromatic hydrocarbons and particulate matter during a wildfire

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ABSTRACT

Wildland firefighters engaged in fire suppression activities are often exposed to hazardous air pollutants such as polycyclic aromatic hydrocarbons (PAHs) and particulate matter ($PM_{2.5}$) during wildfires with no respiratory protection. Although the most significant exposures to smoke likely occur on the fireline, wildland firefighters may also be exposed at the incident command post (ICP), an area designated for wildfire suppression support operations. Our objective was to characterize exposures of PAHs and $PM_{2.5}$ near an ICP during a wildfire event in California. We collected area air samples for PAHs and $PM_{2.5,}$ during the first 12 days of a wildfire event. PAH area air samples were actively collected in 12-hr shifts (day and night) using XAD4-coated quartz fiber filters and XAD2 sorbent tubes and analyzed for 17 individual PAHs. Hourly area $PM_{2.5}$ concentrations were measured with an Environmental Beta Attenuation Monitor. Most PAH concentrations generally had similar concentrations during the day and night. $PM_{2.5}$ concentrations were higher during the day, due to increased fire activity, than at night. The highest concentrations of the 17 PAHs measured were for naphthalene, phenanthrene, and retene. The location of an ICP may be a critical factor in reducing these potential exposures to firefighters during wildfire events. Additionally, exposures could be reduced by utilizing clean air tents or sleeping trailers with HEPA filtration or setting up smaller camps in less smokey areas closer to the fireline for firefighters. Although measured exposures to PAHs for firefighters from smoke are lower at an ICP, these exposures still contribute to the overall cumulative work exposures.

KEYWORDS

Firefighters; incident command post; PAHs; particulate matter; wildfire

Introduction

Over 4 million hectares burned in 2015, the highest amount of forested land burned during the previous decade. Twenty-seven thousand federal wildland firefighters worked to suppress these wildfires across the United States which included 8,745 wildfire incidents in California.[1,2] Wildland firefighters engaged in suppression operations work long shifts and are exposed to wood smoke on the fireline while wearing no respiratory protection.[3–5] In addition to exposure to smoke on the fireline, wildland firefighters and incident management personnel may be exposed to smoke at the incident command post (ICP) that is situated near the wildfire to support suppression operations. Unlike wildland firefighters who are physically fit, incident personnel at ICPs may include individuals that would be considered a sensitive population with pre-existing health conditions.[6] Incident personnel will spend all day, for up to 21 days, at the ICP providing management decisions and incident support. ICPs provide logistical support to the wildfire suppression operation by providing firefighters and incident personnel sleeping locations (camping), morning and evening meals, and administrative services.

Wildland fire smoke contains hazardous air pollutants such as carbon monoxide, benzene, formaldehyde, fine particulate matter ($PM_{2.5}$), acrolein, and polycyclic aromatic hydrocarbons (PAHs).[5,7] PAHs are hazardous air pollutants formed during incomplete combustion, are detected in food, air, and soil, and exist in the environment in both the gaseous and particulate phases.[8] Commonly, PAHs are most known for their carcinogenic potential, but have also been

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associated with other adverse health outcomes. The International Agency for Research on Cancer has reported benzo[a]pyrene as a human carcinogen, benz[a]anthracene, and dibenzo[a,h]anthracene as probable human carcinogens and benzo[b]fluoranthene, benzo[j]-fluoranthene, benzo[k]fluoranthene, chrysene, and naphthalene as possible human carcinogens.[9–11] Past occupational health studies found that inhalational exposure to PAHs was associated with a higher risk of lung and bladder cancer and dermal exposure was associated with an increased risk of skin cancer.[12,13] Additionally, PAH exposure has been associated with cardiopulmonary mortality and immunotoxicity.[14,15]

Past health studies examining wildfire smoke exposure have generally measured $PM_{2.5.}$ These studies have reported that $PM_{2.5}$ from smoke is associated with adverse respiratory health effects and possibly increased mortality and cardiovascular health effects in the general population.[16–19] Previous studies of wildland firefighters have shown increases in airway responsiveness and decreases in lung function across fire season, and a significant increase in biomarkers of systemic inflammation.[4,20–22] Further, a health risk assessment conducted by Booze et al., calculated elevated risks (hazard index = 3.5) of developing non-cancer health effects from exposure to $PM_{3.5}$.[23]

Due to their location near the wildfire, ICPs can be impacted by smoke by winds transporting smoke or inversions that decrease air mixing and increase smoke concentration.[24,25] This exposure to smoke at ICPs impacts incident personnel and is an additional smoke exposure for wildland firefighters off the fireline. The objective of our study was to characterize and understand the exposures of PAHs and $PM_{2.5}$ near an incident command post during a wildfire event in California.

Methods

Air samples were collected during the Willow Fire which burned 2,307 ha (5,702 ac) in the Sierra National Forest from July 26, 2015–August 5, 2015.[26] The monitoring site was located on the Bass Lake Ranger Station in North Fork, CA; 1.1 km from the Willow Fire ICP (Figure 1). This site was selected for our sampling PAH equipment could be co-located with an existing $PM_{2.5}$ monitor that was deployed earlier in the summer to measure wildfire smoke from an earlier incident. The monitor was located in a storage area of the Ranger Station that was isolated from any adjacent vehicle or dust emissions. Air concentrations were measured for the first 12 days of the Willow Fire. As firefighters work in day/night shifts, we examined differences between day and night exposures by collecting two 12-hr integrated samples. Generally, day samples were collected from 7:01 to 19:00 and night samples were collected from 19:01 to 7:00 the next day.[6]

PAH sample collection and analysis

Gas-phase and particle-bound PAH samples were collected using XAD2 sorbent tube and a 37 mm closed faced cassette (SKC, Inc. Eighty Four, PA) with two quartz fiber filters (PallFlex Tissuquartz) in series impregnated with XAD4.[27] Air was drawn through the cassettes using SKC Airchek 52 sampling pumps (SKC, Inc. Eighty Four, PA) at 1.5–2 L/min and the XAD2 sorbent tubes using PAS-500 Micro Air Samplers (Spectrex, Redwood City, CA) sampled at 0.2 L/min. To verify flow rates, pumps were calibrated before and after each sampling event with representative sampling media using a Defender 510 DryCal (Mesa Labs, Inc. Butler, NJ).

Filters were extracted by sonication in dichloromethane followed by filtration under vacuum. XAD2 sorbent tubes were extracted by shaking in dichloromethane (2 mL) for 1 hr and analyzed for naphthalene (NAP) only. Filters were analyzed for 16 PAHs: acenaphthylene (ACY), acenaphthene (ACE), fluorene (FLU), phenanthrene (PHE), anthracene (ANT), fluoranthene (FLT), pyrene (PYR), retene (RET), benz[a]anthracene (BAA), chrysene (CHR), benzo[b]fluoranthene (BBF), benzo[k]fluoranthene (BKF), benzo[a]pyrene (BAP), indeno[1,2,3-cd]pyrene (ICD), dibenz[a,h]anthracene (DBA), and benzo[ghi]perylene (BGP). All analyses were performed on a gas chromatograph (Hewlett Packard model 6890) equipped with a 30 m (50%-Phenyl) methylpolysiloxane-fused silica capillary column and a 5972 Mass Selective Detector operated in selected ion-monitoring mode. Individual PAH concentrations measured by sorbent tubes and filters were summed for statistical analysis.

For all XAD sorbent tubes and filters, we checked for sample breakthrough, blank corrected the mass, and adjusted for limit of quantitation. We determined breakthrough by inspecting if the amount of each analyte in the back filter was greater than 25% of the total amount of analyte detected in both the front and back filters or section of sorbent tube. Field blanks for filter cassettes and XAD sorbent tubes were collected at each sampling event to examine contamination of sampling media in the field or during transport.

$PM_{2.5}$ monitoring

Hourly $PM_{2.5}$ concentrations were measured with an Environmental Beta Attenuation Monitor

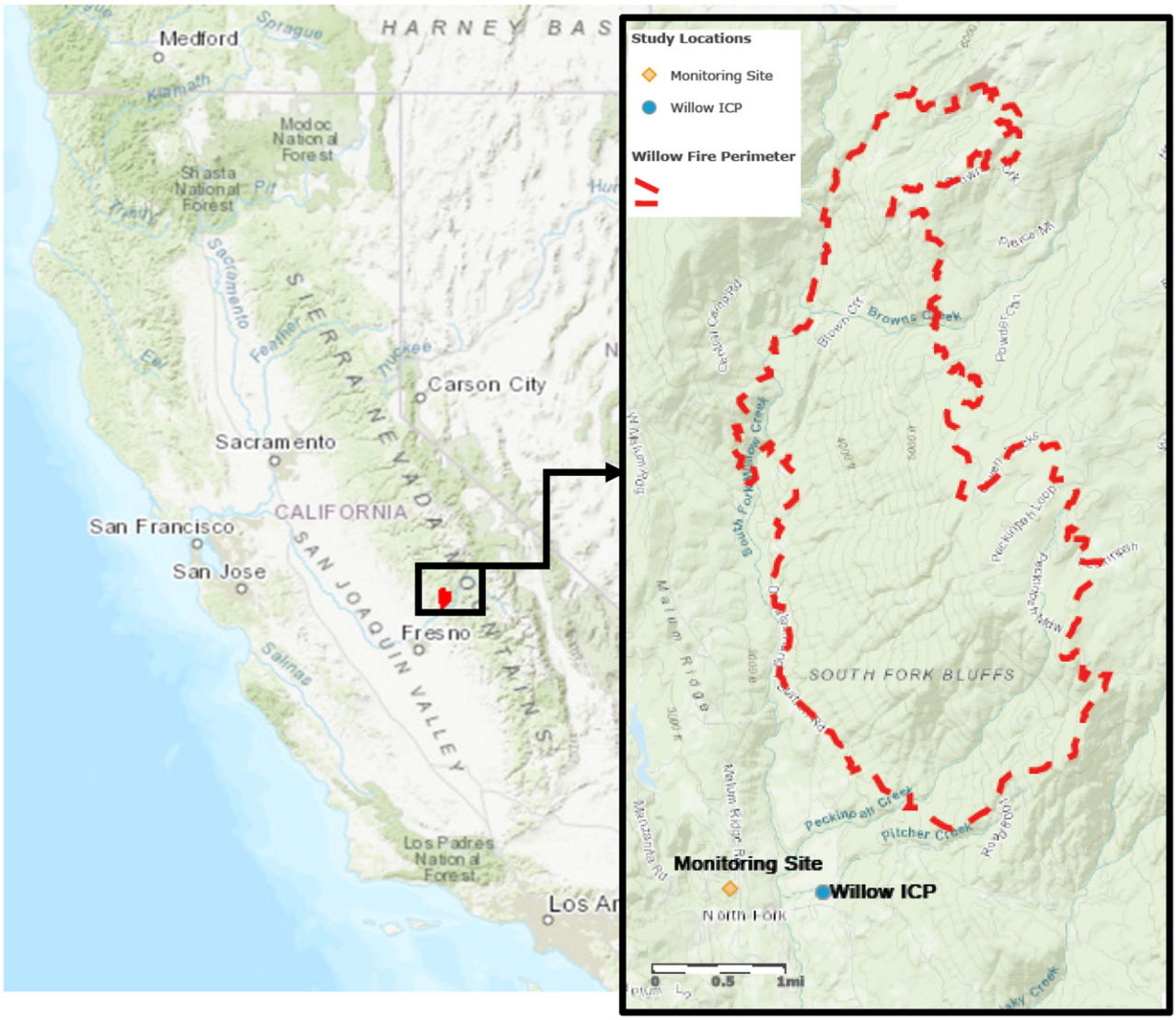


Figure 1. Map of Willow Fire perimeter, ICP, and monitoring site.

(EBAM; Met One Instruments, Inc. Grants Pass, OR) and were used to calculate 12-hr averages that corresponded to the PAH samples, as well as $PM_{2.5}$ daily 24-hr averages. In addition, the 12-hr averages were used to examine day vs. night trends in the concentrations of $PM_{2.5}$. EBAMs are designed for fast and temporary deployment, are accurate within 10% of the true concentration measured, and have detection limits of 6.0 µg m^{-3} and 1.2 µg m^{-3} for hourly and 24-hr measurements, respectively.[28] Measurements were considered abnormal and excluded if the measurement included an alarm code, indicating instrument malfunction, or the flow rate of the monitor was below 15 L/min. Twelve-hour average concentrations were considered valid if they included at least 9 hr of sampling per 12-hr period (75%). In addition, the EBAM collected wind speed and direction and was used to examine how the monitor was impacted by smoke.

Statistical analysis

For each PAH analyte, the mean, standard deviation (SD), minimum, and maximum concentration were determined for day samples, night samples, and 24-hr samples. For comparison across other wildfire exposure studies, we report both the arithmetic mean (AM) and geometric mean (GM). Similarly, summary statistics for were calculated for total PAHs, low/medium/high molecular weight PAHs, and $PM_{2.5}$ for day samples, night samples, and 24-hr samples. Specifically, the daily integrated 12-hr samples were averaged to calculate a 24-hr mean PAH concentration, with the exception of 7/25/2015 when only one 12-hr sample

Table 1. PAH and $PM_{2.5}$ Concentrations at the ICP monitoring site during the Willow Fire from July 26, 2015–August 5, 2015 (12 days).

		Concentration (ng m^{-3})[A]							
		Arithmetic Mean			Geometric Mean				
PAH Analyte (# of rings)	Above LQ (%)	24-hr (SD)	Day (SD)	Night (SD)	24-hr (SD)	Day (SD)	Night (SD)	Min[C]	Max
Naphthalene (2)[B]	100%	467 (579)	589 (755)	344 (302)	284 (3)	327 (3)	253 (2)	80	2515
Acenaphthylene (3)	4%	1 (1)	1 (0)	1 (1)	1 (1)	1 (1)	1 (2)	<1	4
Acenaphthene (3)	52%	3 (4)	3 (6)	2 (1)	2 (2)	2 (3)	2 (2)	<1	19
Fluorene (3)	43%	14 (34)	14 (35)	13 (33)	3 (5)	3 (5)	2 (5)	<1	120
Phenanthrene (3)	74%	10 (15)	9 (10)	11 (19)	5 (3)	5 (3)	5 (3)	<1	70
Anthracene (3)	4%	1 (0)	1 (0)	1 (0)	1 (1)	1 (1)	1 (1)	<1	2
Retene (3)[2]	78%	37 (66)	38 (77)	34 (55)	16 (3)	15 (3)	15 (4)	<3	268
Fluoranthene (4)	57%	2 (3)	2 (2)	3 (3)	2 (2)	2 (2)	2 (2)	<1	13
Pyrene (4)	4%	1 (0)	1 (0)	1 (1)	1 (1)	1 (1)	1 (1)	<1	3
Benz[a]anthracene (4)	4%	3 (2)	1 (1)	1 (1)	3 (1)	3 (2)	3 (1)	<4	2
Chrysene (4)	4%	4 (1)	3 (2)	3 (0)	4 (1)	4 (1)	4 (1)	<4	11
Benzo[b]fluoranthene (5)	9%	3 (2)	4 (0)	4 (1)	3 (1)	3 (2)	3 (1)	<3	8
Benzo[k]fluoranthene (5)	9%	4 (4)	3 (2)	3 (1)	3 (2)	3 (2)	3 (1)	<3	10
Benzo[a]pyrene (5)	13%	3 (2)	4 (6)	3 (1)	3 (1)	3 (2)	3 (1)	<3	23
Indeno(1,2,3-cd)pyrene (6)	4%	2 (2)	3 (3)	3 (0)	1 (2)	1 (2)	1 (1)	<1	12
Dibenz(a,h)anthracene (6)	13%	4 (6)	2 (3)	1 (1)	3 (2)	3 (2)	3 (1)	<3	11
Benzo[ghi]perylene (6)	9%	1 (0)	5 (8)	3 (1)	1 (1)	1 (1)	1 (1)	<1	29
Total PAH (Σ15 PAH)	83%	56 (55)	57 (52)	52 (59)	43 (2)	45 (2)	40 (2)	<29	236
Low Molecular Weight (3)	83%	29 (50)	28 (47)	28 (54)	14 (3)	14 (3)	13 (3)	<8	197
Medium Molecular Weight (4)	57%	10 (4)	10 (3)	10 (4)	10 (1)	10 (1)	10 (1)	<8	22
High Molecular Weight (5-6)	13%	16 (16)	19 (22)	13 (2)	14 (2)	15 (2)	13 (1)	<12	86
$PM_{2.5}$ (μg m^{-3})	100%	29 (50)	58 (31)	13 (5)	14 (2)	49 (2)	12 (1)	7	105

[A]Concentration units for $PM_{2.5}$ is μg m^{-3}
[B]Naphthalene (collected by XAD2 sorbent tube) and retene were not used to calculate Σ15 PAH or Σ3 ring PAH.
[C]Minimum values denoted with a less than sign (<) were below the limit of quantification.

was collected. PAHs were summed to create summary measures by the number of fused benzene rings. We created summary metrics for low molecular weight PAHs (Σ3 ring PAH), medium molecular weight PAHs (Σ4 ring PAH), high molecular weight PAHs (Σ5-6 ring PAH), and a total PAH metric of three through six ring PAHs, composed of 15 PAHs (Σ15 PAH). These summary measures did not include naphthalene, because it is generally regulated separately, or retene, because it is not commonly measured. Pearson product-moment correlation coefficient was calculated to examine the association between NAP, PHE and RET and $PM_{2.5}$. We used SAS (v. 9.4, SAS Institute Inc. Cary, NC) and R (v. 3.1.0 R Foundation, Vienna, Austria) for all data cleaning, calculations, and graphical processing.

Results

As noted above, PAH air samples were collected day and night (approximately 12-hr samples) for the first 12 days of the Willow Fire (N = 23) (Table 1). Blank correction for the 16 PAHs on the filters was generally low with an average blank mass of 2.4 ng (median 0.6, range 0.04–16 ng); FLU had the highest field correction. The blank correction for naphthalene collected by sorbent tubes was on average 18 ng (range 7–28 ng). The limit of quantification (LQ) was determined for each PAH and day of laboratory analysis. The average LQ was 2.6 ng with a range of 1.0–5.3 ng (1.71–7.14 ng m^{-3}). To perform statistical analyses, samples that were less than the LQ were assigned half the LQ. Sample breakthrough was tested in 25% of the samples; no breakthrough was observed, thus, for all XAD4 filters and XAD2 sorbent tubes the front and back sections of each were analyzed together. In general, we detected gas-phase PAHs (2–3 ring PAHs) more frequently than the particle phase PAHs (4–6 rings). NAP was detected in 100% of XAD2 sorbent tube samples. PAHs with 3–4 fused benzene rings varied in detection frequency from 4–78% of XAD4 filters. PAHs with 5 or 6 fused benzene rings were detected in only 4–13% of filter samples collected.

The 24-hr GMs of the highest concentration of PAHs measured were 284 ng m^{-3} and 16 ng m^{-3} for NAP and RET, respectively. All other PAH analytes had GMs that ranged from 1–5 ng m^{-3} and were generally found to be below the LQ. The Σ15 PAH was slightly higher during the day (GM = 45 ng m^{-3}) compared to the night measurements (GM = 40 ng m^{-3}). NAP concentrations were higher during the day (GM = 327 ng m^{-3}) compared to the night (GM = 253 ng m^{-3}). Generally, all other PAH analytes had similar concentrations during the day and night. The 24-hr GMs for Σ3ring PAH and Σ5-6 ring PAH were both 14 ng m^{-3} and contributed equally to the Σ15 PAH GM of 43 ng m^{-3} compared to the GM for Σ4ring PAH (10 ng m^{-3}).

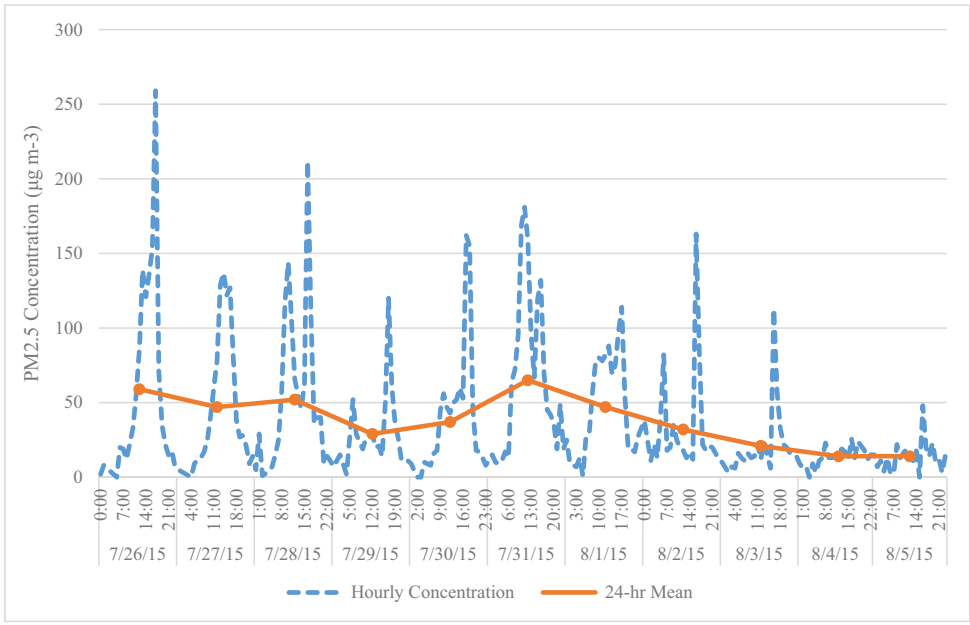


Figure 2. Hourly and 24-hr mean $PM_{2.5}$ Concentrations at North Fork monitoring site during the Willow Fire.

We evaluated hourly data for $PM_{2.5}$, to understand potential trends in concentrations across 24-hr intervals for the 12-day sampling period. In Figure 2, for almost every sampling day, hourly $PM_{2.5}$ concentrations peaked between 14:00 and 17:00, when fire activity was usually high. During evening hours around 18:00, $PM_{2.5}$ concentrations dropped sharply and rose again the following morning as shown in Figure 2. Day/Night 12-hr average $PM_{2.5}$ concentrations measured ranged from 7–105 µg m^{-3} and had a GM of 49 and 12 µg m^{-3} for day and night sampling times, respectively. The 24-hr GM of $PM_{2.5}$ was 14 µg m^{-3}. Pearson product-moment correlation coefficient between NAP and $PM_{2.5}$ ($r = 0.34$), PHE and $PM_{2.5}$ ($r = 0.02$), and RET and $PM_{2.5}$ ($r = 0.13$) did not show a strong correlation between any concentrations of the two air contaminants. Additionally, we evaluated how wind may have impacted the monitor through plotting the mean PM2.5 concentrations in polar coordinates (WD) and wind speed with generalized additive model smoothing (Figure 3). $PM_{2.5}$ concentrations during the day shift were highest at the IC when ground level wind speeds were lower (<1.0 ms^{-1}) and from the direction of the fire (Figure 3A). The day shift experienced the lowest concentration when winds were from the south and higher (0.8–1.8 ms^{-1}) or basically blowing the smoke from the Willow Fire away from the IC. Conversely, the night shift was predominantly impacted when winds were from the west north-west at higher wind speeds (1.0–1.2 ms^{-1}) when smoke would be transported back to the IC (Figure 3B).

Discussion

In this study, we measured concentrations of PAHs and $PM_{2.5}$ from wildfire smoke near an incident command post during the Willow Fire in California. We detected measurable concentrations of 17 individual PAHs, with NAP, RET, and PHE being the highest measured PAHs. $PM_{2.5}$ concentrations were highest during the day (with concentrations peaking between 14:00 and 17:00), when fire activity was high. These results collected near the Willow Fire indicated that firefighters and incident personnel were exposed to PAHs and $PM_{2.5}$ at ICP.

Previously, NIOSH reported NAP (Mean = 3,375 ng m^{-3}), ACE (Mean = 1,042 ng m^{-3}), and FLU (Mean = 594 ng m^{-3}) concentrations at three ICPs (N = 9) during the day at the 1988 Yellowstone Fires.[29] These concentrations are notably higher than the concentrations of NAP, ACE, and FLU we measured (Table 2A). This difference could be due to the numerous fires burning a large amount of acreage at the time of measurement in Yellowstone National Park. In total, 793,880 acres were burned during the 1988 fire season, which included 160,000 acres burning in 1 day during the sampling period.[29] When compared to PAH and $PM_{2.5}$ area concentrations measured during a prescribed fire, an intentional fire used for resource benefit, the Willow Fire ICP had lower total PAH and $PM_{2.5}$ area concentrations from smoke. Robinson et al. reported $PM_{2.5}$ concentrations that ranged from 900–6,980 ng m^{-3} and total PAH concentrations that ranged from 606–3,647 ng m^{-3}.[30] The authors noted that the monitor was well placed

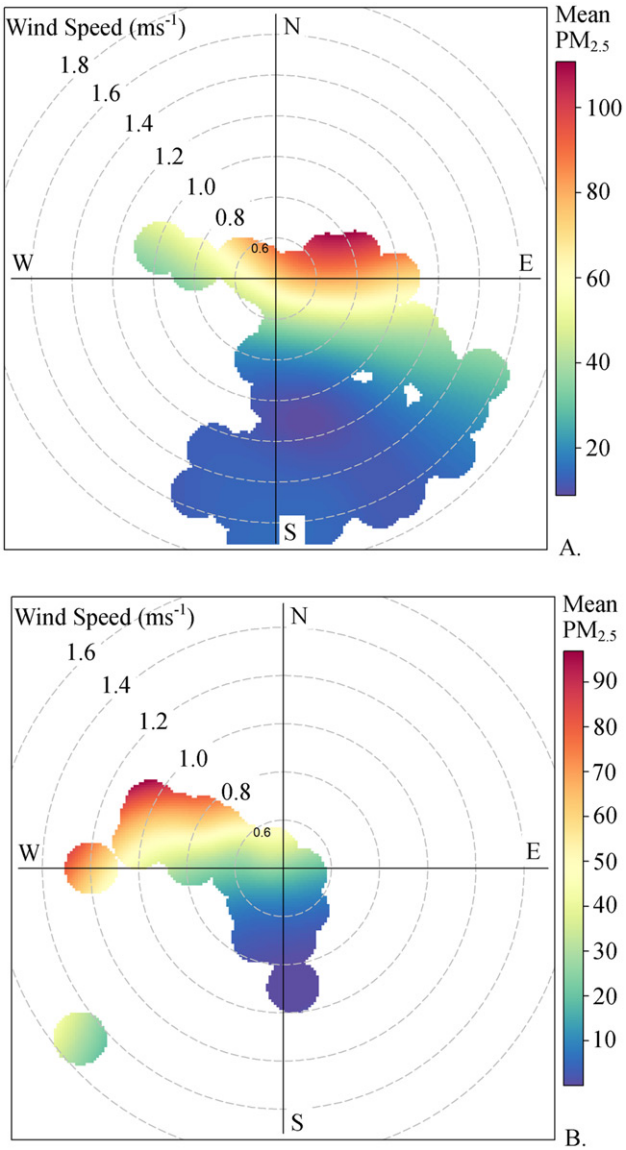


Figure 3. $PM_{2.5}$ mean concentrations during the Willow Fire plotted by wind direction from fire and speed for day shift (A) and night shift (B).

in the plume of smoke with a constant wind direction, which contributed to the high measured concentrations. The National Institute for Occupational Safety and Health has a recommended exposure limit for naphthalene (50 mg m^{-3}) and for coal-tar pitch volatiles (0.1 mg m^{-3}), which includes ANT, BAP, CHR, PHE, and PYR.[31] Although we did not calculate 8-hr time-weighted averages across work shifts, the NIOSH exposure limits are orders of magnitude larger than the concentrations we measured at ICP.

McNamara et al. sampled 1-min time-weighted averages of $PM_{2.5}$ at three ICPs in California, Oregon, and Washington and reported daily means ranging from 9–41 µg m^{-3} and night-time means (11–44 µg m^{-3}) that were higher than daytime means (5-37 µg m^{-3}).[6] The 24-hr mean $PM_{2.5}$ for our study period was 23 µg m^{-3} and is within the range of daily mean concentrations reported by these authors (Table 2B). However, we observed higher 12-hr mean $PM_{2.5}$ concentrations during the daytime (19–105 µg m^{-3}), specifically during 14:00 and 17:00, compared to the nighttime (7–24 µg m^{-3}). The higher nighttime concentrations observed by McNamara et al. could be due to the increased vehicle traffic when firefighters return to camp (vehicle emissions and road dust) and generator use at ICPs. Since our monitor was located slightly away from the ICP and located in a storage area of the Ranger Station, it would not have measured these additional sources of $PM_{2.5}$. The monitor location was on a U.S. Forest Service district compound in an area that was used for outdoor storage and housed a communication tower and was rarely visited by employees or vehicles. In addition, monitoring location, fire activity, and metrological conditions can affect $PM_{2.5}$ concentrations near wildland fires. Further, the 24-hr arithmetic mean $PM_{2.5}$ concentration collected from the 2013 Rim Fire ICP of 121 µg m^{-3} is much higher than the 24-hr arithmetic mean $PM_{2.5}$ concentration at the ICP in our study of 29 µg m^{-3}. This is likely due to the extremely large size of the Rim Fire (257,314 acres) with much higher quantities of biomass burned per day emitting higher quantities of pollutants.[24] Gaughan et al. reported higher average area concentrations of respirable particulate (0.1–10 μm) on the fireline (590 µg m^{-3}) compared to the ICP (GM = 380 µg m^{-3}) at the Red Eagle Fire in 2006.[32] Our results are consistent with other studies that indicate that although firefighters are exposed to higher concentrations of smoke on the fireline, exposure to smoke also occurs during off-fireline time periods.

Not only can the results of our study be representative of exposures to personnel working at the Willow fire incident, but can also be representative of the community nearby as well. The U.S. EPA Air Quality Index (AQI) category, a system used to provide guidance on recommended actions to mitigate exposures to smoke for public health officials, the media, and affected communities.[33] The AQI has six categories that correspond with 24-hr average concentrations of $PM_{2.5}$: Good (0–12 µg/m^3), Moderate (12.1–35.4 µg/m^3), Unhealthy for Sensitive Groups (USG; 35.5–55.4 µg/m^3), Unhealthy (55.5–150.4 µg/m^3), Very Unhealthy (150.5–250.4 µg/m^3), and Hazardous (250.5–500 µg/m^3). During the Willow Fire the 24-hr mean concentration was Moderate for 5 days, USG for 4 days, and Unhealthy for two days (Figure 2). Compared to ambient PAH concentrations in the nearby urban area of Fresno, CA, the concentrations

Table 2. Comparison of current study exposure concentrations with previously reported PAH (Table A) and $PM_{2.5}$ (Table B) concentrations.

Table A. Comparison of PAH concentrations (ng m^{-3}).

Study Name	NAP	ACE	FLU	PHE	PYR	BKP	BAP	DBA
Current study ICP[A]	589	3	14					
Reh et al. (1988)[A]	3,375	1,042	594					
Current study ICP[B]	284	2	3	5	1	3	3	3
Tager et al. (2006)[C]	228		2.9	5.7	0.96	0.25	0.32	0.7

Table B. Comparison of $PM_{2.5}$ concentrations (μg m^{-3}).

Study Name	$PM_{2.5}$ Daily Range	$PM_{2.5}$ Daily Mean	$PM_{2.5}$ Daytime 12-hr Mean	$PM_{2.5}$ Nighttime 12-hr Mean
Current study ICP	7–105	29	19–105	7–24
Robinson et al. (2008)	900–6,980			
McNamera et al. (2012)		9–41	5–37	11–44
Navarro et al. (2016)		121		

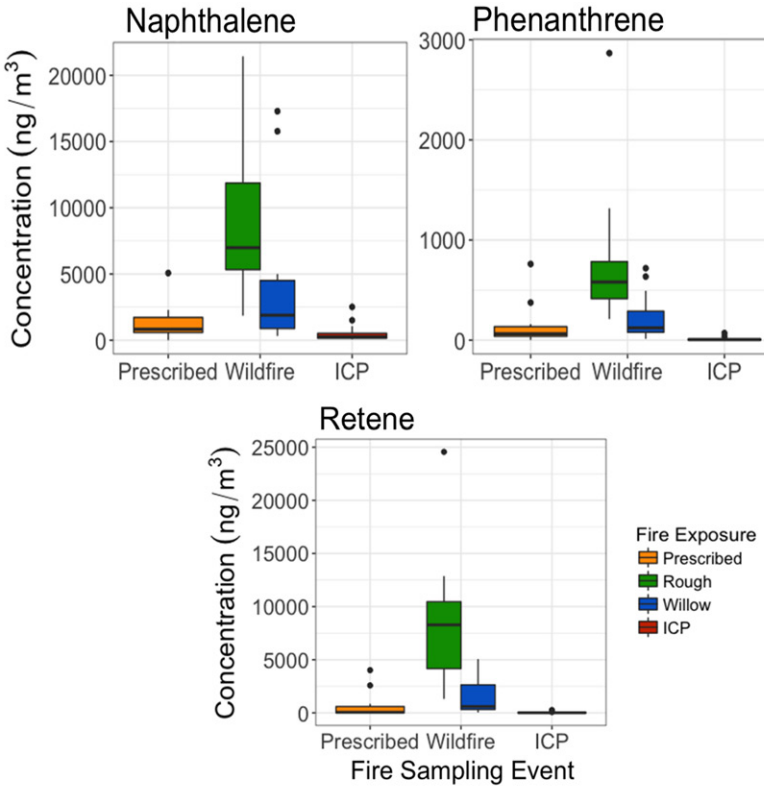
[A]Daytime mean concentrations from current study.
[B]24-hr geometric mean concentrations from current study were used for comparison with Tager et al. 2006.
[C]Tager et al. (2006) reported 24-hr median concentration.

of PAHs near the Willow Fire ICP were generally higher (Table 2A). A previous exposure assessment in central Fresno measured 24-hr PAH concentrations at a regulatory monitoring station from April 2001 to March 2003 with a median concentration for NAP of 228 ng m^{-3}.[34] NAP, PHE, PYR, and FLU had median concentrations that were similar to GMs that we measured in North Fork. However, GM concentrations of the 5-6 ring PAHs near the Willow Fire ICP were significantly higher than median concentrations in Fresno. For example, GM concentrations for DBA, BKF, and BAP were 42, 12, and 7- fold higher, respectively, near the Willow Fire ICP. However, many of the 5–6 ring PAHs measured in this study were below the limit of detection. Although the usual sources of PAHs are different in these urban areas, the measured concentrations demonstrate that PAHs from wildland fires can lead to significantly higher concentrations of PAHs in the ambient environment and affect communities near wildfires.

Figure 4 highlights the distributions of NAP, PHE, and RET, the highest measured PAHs between the ICP area samples reported in this study and personal exposures measured by Navarro et al. 2017 on firefighters working at prescribed fires and wildland fires reported in a previous publication.[35] Across the sampling events, concentrations measured from firefighters at the prescribed fire and samples collected from the ICP in this study samples were similar. Few concentrations measured at prescribed fire sampling events were comparable to concentrations measured at wildland fire sampling events. Samples collected at the Rough Fire had higher concentrations and more varied distributions compared to the Willow Fire or the prescribed fires we monitored or at the ICP. This difference could be due to the Rough Fire's size; compared to the Willow Fire, in which total acres burned were 5,702, the Rough Fire grew from 79,973–85,894 acres during the sampling campaign.[26,36] Another possible explanation for the higher PAH concentrations at the Rough Fire may be temperature inversions, since all firefighter samples collected at the Rough Fire were during the nightshift. Temperature inversions often occur at night near wildland fires which lowers the mixing layer and increases concentrations of air pollutants.[37] The significantly larger area of land burning and nighttime temperature inversions may have contributed to the higher PAH concentrations in the ambient environment at the Rough Fire, which would result in an increase in firefighter PAH exposure.

To reduce potential smoke exposures at ICPs the following recommendations have been suggested by the National Wildfire Coordinating Group Smoke Committee.[37] Remote base camps (spike camps) can be set-up for firefighters near the fireline in areas that are above smoke inversions and upwind from the fire. Additionally, this can reduce the hazards and risks associated with driving longer distances to an ICP. Incident management teams could provide clean air tents for clean airtime during the day or enclosed sleeping trailers (bunks in semi-truck trailers) with filtered air at night. However, the sleeping trailers do not provide ideal sleeping conditions nor are they always utilized by firefighter because they do not fit the culture preferences of firefighters—sleeping as a crew and the physical toughness of sleeping outside. Last, the design and layout of camp should provide adequate distance between sleeping areas and sources of $PM_{2.5}$ (generators and vehicle traffic and exhaust).

Our results add to the literature quantifying exposure to smoke for wildland firefighters, and less studied, the exposure for incident personnel. However, our study has limitations that should be



Prescribed and Wildfire Concentrations are personal firefighter exposure measurements reported in Navarro et al. 2017

Figure 4. Naphthalene, Phenanthrene, and Retene concentrations across prescribed and wildfire firefighters and ICP.

considering when interpreting our findings. Although we were able to sample over many days (12) at the start of the wildfire incident, our results only encompass one wildfire incident. We only sampled in California, in a mixed-conifer forest. As $PM_{2.5}$ concentrations from wildfire have been shown to vary by fire and fuel type, our study is limited by measurements at only one ICP location.[38] When comparing our results to other exposure concentrations, our summary metrics may have been influenced by the large number of non-detectable measurements. Last, the location of our monitoring site was not directly in the ICP, but was located nearby at a Forest Service Ranger Station. For that reason, our measurements may not be completely representative of all air pollutant exposures at an ICP. Additionally, the single monitoring location was not able to provide information on the spatial distribution or other emission sources of $PM_{2.5}$ in the area and limits the representation of our study data. Our monitoring location was chosen so that our PAH sampler could be co-located with an incident deployed $PM_{2.5}$ monitor. However, the concentrations measured at our monitoring site were more likely due to smoke only, and not influenced by other sources of emissions at ICPs such generators, vehicle exhaust and dust due to the isolated location of the monitor at the Ranger Station.

Conclusions

This assessment demonstrates that wildland firefighters and incident personnel are exposed to PAH and $PM_{2.5}$ air contaminants at ICPs while off the fireline. This additional exposure contributes to a higher cumulative work exposure and increases the potential for adverse health effects. For this reason, location and the potential for smoke exposure from the nearby wildfire should be considered when choosing the placement of an ICP. As wildfire activity continues to increase and more people become involved in suppression operations, it is important to characterize such exposures to all workers at wildfires.

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References

[1] **NIFC**: *Wildland Fire Summary and Statistics Annual Report 2015*. National Interagency Coordination Center, 2016.

[2] **Committee on Agriculture Conservation and Forestry Subcommittee**: *The 2015 Fire Season and Long Term Trends*. In Committee on Agriculture Conservation and Forestry Subcommittee, 2015.

[3] **Broyles, G.:** "Wildland Firefighter Smoke Exposure". US Forest Service, 2013.

[4] **Hejl, A.M., O. Adetona, D. Diaz-Sanchez, et al.:** Inflammatory effects of woodsmoke exposure among wildland firefighters working at prescribed burns at the Savannah River site, SC. *J. Occup. Environ. Hyg. 10(4)*:173–180 (2013).

[5] **Naeher, L.P., M. Brauer, M. Lipsett, et al.:** Woodsmoke health effects: A review. *Inhal. Toxicol. 19(1)*:67–106 (2007).

[6] **McNamara, M.L., E.O. Semmens, S. Gaskill, C. Palmer, C.W. Noonan, and T.J. Ward:** Base camp personnel exposure to particulate matter during wildland fire suppression activities. *J. Occup. Environ. Hyg. 9(3)*:149–156 (2012).

[7] **Adetona, O., T.E. Reinhardt, J. Domitrovich, et al.:** Review of the health effects of wildland fire smoke on wildland firefighters and the public. *Inhal. Toxicol. 28(3)*:95–139 (2016).

[8] **ATSDR:** *Toxicological Profile for Polycyclic Aromatic Hydrocarbons*. Atlanta, GA: Agency for Toxic Substances and Disease Registry, 1995.

[9] **IARC:** *Some Traditional Herbal Medicines, Some Mycotoxins, Naphthalene and Styrene, IARC Monographs on the Evaluation of Carcinogenic Risks to Humans*. Lyon, France: International Agency for Research on Cancer, 2002.

[10] **IARC:** *Some Non-Heterocyclic Polycyclic Aromatic Hydrocarbons and Some Related Exposures, IARC Monographs on the Evaluation of Carcinogenic Risks to Humans*. Lyon, France: International Agency for Research on Cancer, 2010.

[11] **IARC:** *IARC Monographs on the Evaluation of Carcinogenic Risks to Humans: Chemical Agents and Related Occupations*, p. 9: International Agency for Research on Cancer, 2012.

[12] **Armstrong, B., E. Hutchinson, J. Unwin, and T. Fletcher:** Lung cancer risk after exposure to polycyclic aromatic hydrocarbons: A review and meta-analysis. *Environ. Health Perspect. 112(9)*:970–978 (2004).

[13] **Boffetta, P., N. Jourenkova, and P. Gustavsson:** Cancer risk from occupational and environmental exposure to polycyclic aromatic hydrocarbons. *Cancer Causes Control: CCC 8(3)*:444–472 (1997).

[14] **Jack, H.D.:** *Immunotoxicology and Immunopharmacology*. Boca Raton, FL: CRC Press, 1994.

[15] **Martin, S., L. Dawidowski, P. Mandalunis, F. Cereceda-Balic, and D.R. Tasat:** Characterization and biological effect of Buenos Aires urban air particles on mice lungs. *Environ. Res. 105(3)*:340–349 (2007).

[16] **Delfino, R.J., S. Brummel, J. Wu, et al.:** The relationship of respiratory and cardiovascular hospital admissions to the southern California wildfires of 2003. *Occup. Environ. Med. 66(3)*:189–197 (2009).

[17] **Henderson, S.B., and F.H. Johnston:** Measures of forest fire smoke exposure and their associations with respiratory health outcomes. *Curr. Opin. Allergy Clin. Immunol. 12(3)*:221–227 (2012).

[18] **Henderson, S.B., M. Brauer, Y.C. Macnab, and S.M. Kennedy:** Three measures of forest fire smoke exposure and their associations with respiratory and cardiovascular health outcomes in a population-based cohort. *Environ. Health Perspect. 119(9)*:1266–1271 (2011).

[19] **Rappold, A.G., S.L. Stone, W.E. Cascio, et al.:** Peat bog wildfire smoke exposure in rural North Carolina is associated with cardiopulmonary emergency department visits assessed through syndromic surveillance. *Environ. Health Perspect. 119(10)*:1415–1420 (2011).

[20] **Adetona, A.M., O. Adetona, R.M. Gogal, D. Diaz-Sanchez, S.L. Rathbun, and L.P. Naeher:** Impact of work task-related acute occupational smoke exposures on select proinflammatory immune parameters in wildland firefighters. *J. Occup. Environ. Med. 59(7)*:679–690 (2017).

[21] **Liu, D., I.B. Tager, J.R. Balmes, and R.J. Harrison:** The effect of smoke inhalation on lung function and airway responsiveness in wildland fire fighters. *Am. Rev. Respir. Dis. 146(6)*:1469–1473 (1992).

[22] **Swiston, J.R., W. Davidson, S. Attridge, G.T. Li, M. Brauer, and S.F. van Eeden:** Wood smoke exposure induces a pulmonary and systemic inflammatory response in firefighters. *Eur. Respir. J. 32(1)*:129–138 (2008).

[23] **Booze, T.F., T.E. Reinhardt, S.J. Quiring, and R.D. Ottmar**: A screening-level assessment of the health risks of chronic smoke exposure for wildland firefighters. *J. Occup. Environ. Hyg. 1(5)*:296–305 (2004).

[24] **Navarro, K.M., R. Cisneros, S.M. O'Neill, D. Schweizer, N.K. Larkin, and J.R. Balmes**: Air-Quality impacts and intake fraction of PM2.5 during the 2013 Rim Megafire. *Environ. Sci. Technol. 50(21)*:11965–11973 (2016).

[25] **Schweizer, D., and R. Cisneros:** Wildland fire management and air quality in the southern Sierra Nevada: Using the Lion Fire as a case study with a multi-year perspective on PM(2.5) impacts and fire policy. *J. Environ. Manage. 144*:265–278 (2014).

[26] **InciWeb the Incident Information System**: Willow Fire (2016). Retrieved from inciweb/mwcg/gpv.

[27] **Noth, E.M., S.K. Hammond, G.S. Biging, and I.B. Tager**: A spatial-temporal regression model to predict daily outdoor residential PAH concentrations in an epidemiologic study in Fresno, CA. *Atmosph. Environ. 45*(14):2394–2403 (2011).

[28] **Met One Industries Inc.**: *E-BAM Particulate Monitor Operation Manual EBAM-9800 Rev*. Grant Pass, OR, 2008.

[29] **Reh, C.M., and S.D. Deitchman**: *NIOSH Health Hazard Evaluation Report* (HETA 88-320-2176). U.S. Department of Interior, National Park Service, Yellowstone National Park, Wyoming. 1992.

[30] **Robinson, M.S., T.R. Anthony, S.R. Littau, et al.:** Occupational PAH exposures during prescribed pile burns. *Ann. Occup. Hyg. 52(6)*:497–508 (2008).

[31] **National Institute for Occupational Safety and Health**: *NIOSH Pocket Guide to Chemical Hazards and Other Databases."* Dept. of Health and Human Services, Center for Disease Control and Prevention, National Institute for Occupational Safety and Health, 2007.

[32] **Gaughan, D.M., C.A. Piacitelli, B.T. Chen, et al.:** Exposures and cross-shift lung function declines in wildland firefighters. *J. Occup. Environ. Hyg. 11*(9):591–603 (2014).

[33] **Office of Environmental Health and Hazard Assessment**: *Wildfire Smoke: A Guide for Public Health Officials Revised July 2008 (with 2012 AQI Values).* California Environmental Protection Agency, 2012.

[34] **Tager, I.B., S.K. Hammond, M. Hjelmroos-Koski, et al.:** *Fresno Asthmatic Children's Environment Study (FACES) Final Report*. Research Division California Air Resources Board, 2006.

[35] **Navarro, K.M., R. Cisneros, E.M. Noth, J.R. Balmes, and S.K. Hammond:** Occupational exposure to polycyclic aromatic hydrocarbon of wildland firefighters at prescribed and wildland fires. *Environ. Sci. Technol. 51*(11):6461–6469 (2017).

[36] **InciWeb the Incident Information System**: *Rough Fire* (2016).

[37] **Hardy, C.C., R.D. Ottmar, J.L. Peterson, J.E. Core, and P. Seamon:** Smoke management guide for prescribed and wildland fire. Boise, ID: National Wildfire Coodination Group, 2001. p. 226.

[38] **US, E.P.A.:** Chapter 13: Miscellaneous Sources, 13.1 Wildfires and Prescribed Burning. In *AP-42: Compilation of Air Emission Factors.* Research Triangle Park, NC, 1995.