Heat and smoke from wildfires influence water temperature and dissolved oxygen levels in headwater streams

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Abstract: Wildfire has become increasingly common and severe in forested landscapes across the western United States. Stream and air temperatures within these landscapes are influenced not only by direct heating during the fire but by shading from heavy smoke. In addition, dissolved oxygen (DO) can be affected by increased ash inputs from fire or reduced primary production under lower light conditions. Because collecting data as the event occurs is challenging, most research exploring fire effects on streams has focused on responses months to years after the event as opposed to during and immediately after the fire. We sought to understand how physical stream parameters change as wildfire burns near streams, how stream temperature responses vary through a stream reach, how parameters respond to shading from pervasive smoke during and shortly after the fire, and how fire severity correlates with observed changes. In this study, we report measurements of light, air temperature, stream temperature, and DO across eleven 2nd- to 4th-order streams in the western Cascade Mountains of Oregon, USA, 1 wk before, during, and 1 wk after an extensive wildfire in 2020. Burning near streams resulted in a brief increase in daily maximum stream temperature of 4.5°C at the most severely burned site but small increases or decreases at less affected sites. Longitudinal replication revealed that temperature responses can be patchy at relatively small scales (~50 m). DO daily minima decreased by 1.3 to 16.9% saturation on the day of the fire, with the magnitude of effect corresponding to burn severity. Across all 11 sites, riparian and watershed estimates of Rapid Assessment of Vegetation Condition after Wildfire and soil burn severity were better correlated with stream temperature responses to fire than percentage of watershed burned. Smoke effects were pervasive, both spatially and temporally, resulting in decreased light, stream temperature maxima, and diurnal variation in DO. Our results suggest that acute changes to physical stream parameters caused minimal harm to aquatic biota at our sites, and the effects of smoke on physical stream parameters will likely impose larger immediate effects on headwater streams than heating from the fire itself.
Key words: wildfire, smoke, headwater stream, stream temperature, dissolved oxygen, light, riparian burning

Wildfire has become an increasingly common and severe landscape-scale disturbance as climate change extends fire seasons and promotes dry fuel conditions (Abatzoglou and Williams 2016, Alizadeh et al. 2021, Burke et al. 2021). Riparian areas may serve as fire refugia with lower severity impacts because they occur in valleys and are often wetter than uplands (Rupasinghe and Chow-Fraser 2021); however, under dry conditions, accumulation of fuels in riparian

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areas can create corridors for fire movement and cause high burn severity near streams (Pettit and Naiman 2007). The few studies that have explored acute wildfire effects (i.e., during the period of active burning) on stream temperature have found variable responses, with distinct increases in some cases but only muted effects in others. For example, in central Montana, USA, temperatures in a 3rd-order headwater stream reached 17.8°C as a fire burned directly over the stream (7.8°C warmer than a nearby, unburned control reach; Hitt 2003); however, in another study in western Montana, researchers found no acute fire effects in 3rd- to 4th-order streams (Mahlum et al. 2011). In southeastern Oregon, USA, Schultz et al. (2017) observed an increase of 4°C when wildfire was actively burning, whereas in Yellowstone National Park, researchers found little response to fire, with temperatures remaining <12°C during the wildfire, which they attributed to heavy groundwater influence (Albin 1979, Minshall et al. 1997). Variable responses to fire are due not only to differences in groundwater inputs but are also a result of differences in fire severity, slope, and stream size (Pettit and Naiman 2007, Halofsky and Hibbs 2008). In addition, studies evaluating stream temperature during a fire seldom include spatial replication within the reach, potentially over- or underestimating effects because fires can burn inconsistently in the riparian zone (Bêche et al. 2005).

Adequate dissolved oxygen (DO), which is critical for much of the aquatic life in streams, is also subject to potential acute (i.e., during an event) and immediate (e.g., occurring hours to weeks after an event) impacts from wildfire disturbances. During or following wildfire, depletion of DO concentrations can occur because of increased contributions of organic matter that fuel microbial metabolism (Reid et al. 2006, Marañón-Jiménez et al. 2013) or because of large inputs of ash and black C that create chemically reducing conditions with potential to affect chemical oxygen demand (Dahm et al. 2015). In a stream network in Australia, substantial declines in DO levels were associated with a sediment pulse triggered by a large rain event a few days after a wildfire (Lyon and O'Connor 2008). Similar DO declines in a stream network in New Mexico, USA, were attributed to the absence of gross primary production because of increased suspended sediment that reduced light fluxes to the benthos and increased organic matter that boosted microbial respiration (Reale et al. 2015). Importantly, these studies demonstrated that the input of sediment resulting from rain events soon after a fire can confound differentiation of acute effects of fire alone from short-term effects associated with sediment pulses.

Another important immediate impact of wildfire is smoke, which can have a more extensive spatial and temporal presence than the fire itself. In recent decades, wildfire activity has contributed to a greater duration of smoky days across the entire United States (Burke et al. 2021). In contrast with temperature increases from fire itself, smoke that accompanies large fires is expected to cool streams and reduce the daily range in stream temperature by reducing shortwave radiation (David et al. 2018). Furthermore, smoke could affect stream DO levels through controls on aquatic primary production that are synced to the availability of photosynthetically active radiation (PAR). These metabolic changes could persist for several weeks if severe smoke shades streams before the fire reaches a site and lingers for days or weeks after the fire has passed. Therefore, in addition to the expected highly localized effects of heat from fire on stream temperature, the influence of smoke in shading streams may have a broad temporal and spatial extent, affecting streams within and outside of burn perimeters.

Most research exploring the effects of fire on streams focuses on long-term responses measured years after the event, often describing increases in runoff, sediment loading, and stream light availability (Gresswell 1999, Bixby et al. 2015, Cooper et al. 2015, Klose et al. 2015). Only a handful of studies have quantified the acute or immediate impacts of fire on forested streams, with most focused only on stream temperature responses and with limited spatial replication (Hitt 2003, Mahlum et al. 2011, Beakes et al. 2014, Schultz et al. 2017); however, Sherson et al. (2015) highlight the importance of continuous water quality monitoring to fill in the gaps between grab sampling for biogeochemical metrics. Capturing the acute and immediate effects of wildfire on streams while the disturbance is occurring is difficult because it requires the fortuitous placement of data loggers prior to a fire, the capacity for loggers to persist through the fire, and an ability to retrieve functional loggers.

In this study, we documented the responses of light, air temperature, stream temperature, and DO across a series of small streams in the western Cascade Mountains of Oregon 1 wk before, during, and 1 wk after the Holiday Farm Fire, a large wildfire that occurred in western Oregon in September 2020. We utilized a gradient of 11 streams in unburned to completely burned 2nd- to 4th-order watersheds within or just outside the fire perimeter to evaluate 1) the acute effects of wildfire on stream temperature, air temperature, and DO; 2) the spatial heterogeneity of stream temperature responses through a reach; 3) the immediate effects of smoke on stream temperature, air temperature, DO, and light; and 4) whether and to what degree burn severity metrics correspond to stream responses. In addition, to avoid possible confounding of acute fire effects and rainfall effects on DO, we not only present DO responses after the 1st rain events but identify the effects of fire on DO independent of sediment pulses during and within the 1st wk after the fire.

METHODS

Originating on US Highway 126 near Blue River, Oregon on 7 September 2020, the Holiday Farm Fire burned 701 km^2 of a mix of private and public forestland over the course of 52 d. Extreme fuel aridity combined with rare eastern, downslope winds on Labor Day contributed to the rapid spread of multiple large wildfires that, in total, burned ~11% (4000 km^2) of the Oregon Cascade Range within a 3-d period (7–9 September 2020) (Abatzoglou et al. 2021). Not only did the fires burn forests adjacent to surface waters, but pervasive smoke from these concurrent fires also altered light and temperature regimes throughout the region. The wildfire burned 9 of eleven 2nd- to 4th-order streams that were part of 2 separate pre-existing forest management projects coordinated by the National Council for Air and Stream Improvement Inc, and Weyerhaeuser Company. The fortuitous placement of sampling locations allowed for a study design to assess fire impacts with a natural gradient of fire presence and intensity, as well as a broader analysis of wildfire smoke as a fairly consistent treatment across 11 headwater streams in the same region.

We used timeseries data from stream temperature, air temperature, PAR, and DO loggers recording before, during, and after the fire at a mix of burned and unburned sites to examine the acute effects of fire and smoke on these physical stream parameters, as well as lingering effects during the following weeks. Burn severity at the sites was patchy, but the 11 sites were categorized into burned riparian sites in which both watershed and riparian areas burned; ($n = 6$), burned watershed sites without adjacent riparian burn ($n = 3$), and unburned sites ($n = 2$). We calculated summary statistics for the stream parameters to describe the magnitude of change during and after the fire, and we performed t-tests on pre- and post-fire summary statistics for the smoke effect where sample size permitted. Sites experienced a range in burn severities; therefore, we used correlation analysis to relate burn severity metrics with fire and smoke effects across the 4 physical stream parameters.

Study sites

The 11 study streams (Table 1) are in the foothills of the Cascade Range and drain into the McKenzie River, the Calapooia River, and Fall Creek basins (Fig. 1). Within the McKenzie River basin, 2 streams are in the Ennis Creek subcatchment (EN1, EN2), 2 are in the Quartz Creek subcatchment (QZ1 and QZ2), 3 are in the Mill Creek subcatchment (ML1, ML2, ML3), and 1 is in the Gate Creek subcatchment (WG). The 2 Calapooia River tributaries (CA1, CA2) feed into the mainstem. One tributary (LF) flows into Little Fall Creek. All streams, except WG, are on private land managed for commercial timber production of Douglas fir (*Pseudotsuga menziesii*). Study reaches are bordered by riparian stands aged between 44 and ~200 y old, with managed timber stands upslope varying between ~5 and 60 y old. Riparian zones are dominated by Douglas fir mixed with red alder (*Alnus rubra*), bigleaf maple (*Acer macrophylum*), salmonberry (*Rubus spectabilis*), vine maple (*Acer circinatum*), western hemlock (*Tsuga heterophylla*), and western red cedar (*Thuja plicata*). These streams have relatively high gradients, cobble substrates, bankfull widths from 3.2 to 4.4 m, and watershed areas that range from 2.4 to 14.4 km^2 (Table 1). All streams have Coastal Cutthroat Trout (*Oncorhynchus clarkii clarkii*) and Coastal Giant Salamanders (*Dicamptodon tenebrosus*). Three sites (QZ1, EN1, EN2) contain sculpin (*Cottus* spp.), and 2 sites (CA1, LF) contain Steelhead Trout (*Oncorhynchus mykiss*). This region of Oregon is a temperate forest system with a Mediterranean climate characterized by cold, wet winters (4°C mean air temperature), when most of the ~64 cm/y of precipitation falls, and hot, dry summers (18°C mean air temperature) (means calculated for 2018–2020 from Trout Creek RAWS data available at https://raws.dri.edu/wraws/orF.html). Elevation at the study sites ranges from 400 to 600 m a. s. l. Study watersheds are dominated by andesite and basalt or basalt and breccia originating from the Oligocene to Miocene epochs (United States Geological Survey Geologic Map of Oregon https://mrdata.usgs.gov/geology/state/state.php?state=OR). A nearby gauge in an undammed McKenzie River tributary indicated discharge was at summer low-flow levels and water temperatures were just beginning to cool

Table 1. Stream characteristics for the 11 study sites in the western Cascade Range, Oregon, USA.

Category	Site	Subbasin	Watershed area (km^2)	Bankfull width (m)	Stream order	Elevation (m a.s.l.)	Reach slope (mm/m)
Unburned	CA1	Calapooia	6.2	4.4	4	496	0.75
	QZ1	Quartz	5.5	4.0	3	536	0.71
	ML3	Mill	2.4	3.4	3	494	1.11
Burned watershed	LF	Little Fall	14.4	8.7	3	539	0.05
	QZ2	Quartz	3.8	3.9	3	536	0.39
	CA2	Calapooia	4.4	4.1	3	603	0.62
	ML2	Mill	3.3	3.8	2	420	0.60
Burned riparian	ML1	Mill	3.1	4.4	3	519	3.35
	EN1	Ennis	3.9	3.6	3	470	0.86
	WG	Gate	13.5	6.4	4	410	0.19
	EN2	Ennis	3.4	3.3	3	465	0.44

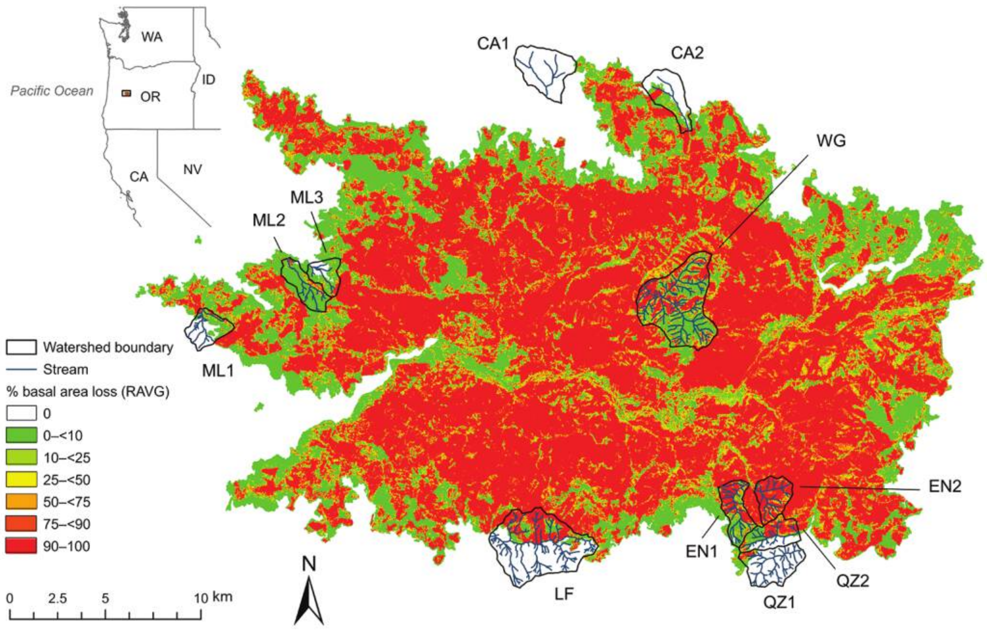

Figure 1. Eleven watersheds within and just outside the Holiday Farm Fire boundary in western Oregon (OR), USA, that burned in September 2020. Rapid Assessment of Vegetation Condition after Wildfire (RAVG) provides estimates of % of basal area loss across 7 fire-severity classes. See Table 1 for stream subbasin codes. State abbreviations are: CA = California, WA = Washington, ID = Idaho, NV = Nevada.

when the wildfire started (United States Geological Survey gauge 14161500).

Characterization of fire treatment

To characterize fire severity in our study watersheds, we used geographic information system analysis in ArcGIS® (version 10.8.1; Environmental Systems Research Institute™, Redlands, California). We determined watershed area and Strahler stream order with available flow accumulation and flow direction raster layers from the National Hydrography Dataset Plus (version 2; https://www.epa.gov/waterdata/get-nhdplus-national-hydrography-data set-plus-data) by using the Hydrology toolset in Spatial Analyst in ArcGIS. To characterize fire effects across these watersheds, we quantified 1) % watershed burned using the Holiday Farm Fire boundary (National Interagency Fire Center; https://data-nifc.opendata.arcgis.com/datasets/nifc::wfigs-wildland-fire-perimeters-full-history/about); 2) watershed and riparian means of Rapid Assessment of Vegetation Condition after Wildfire (RAVG), which provides estimates of % basal area lost relative to the pre-fire condition (United States Department of Agriculture; https://burnseverity.cr.usgs.gov/ravg/data-access); and 3) watershed and riparian means of soil burn severity (SBS; United States Department of Agriculture; https://burnseverity.cr.usgs.gov/baer/baer-imagery-support-data-download). We calculated watershed means of both RAVG and SBS as the mean of raster values within watershed perimeters, and we calculated riparian means within 100-m buffers around stream reaches, reclassifying unburned cells as 0. We used these metrics collectively (% watershed burned, mean watershed and riparian RAVG, and mean watershed and riparian SBS) to characterize the fire treatment at each stream. To understand how fire severity (as described by the 5 burn severity metrics) relates to fire and smoke effect measures (stream temperature, air temperature, DO, and PAR responses), we conducted Pearson's correlation analysis. Although SBS data are ordinal, we treated means across severity classes within watersheds and riparian buffers as continuous values for this analysis.

For simplicity of analysis, we categorized sites by the degree to which their riparian zones were burned, despite a continuous spectrum in burn severity. Because we anticipated fire effects to be more pronounced at streams where fire burned directly near the stream, we categorized sites as burned riparian if the riparian RAVG value exceeded 2, burned watershed if the riparian RAVG value fell below 2 but the watershed RAVG exceeded 1, and unburned if both the riparian and watershed RAVG values fell below 1 (Table 2).

Characterization of smoke treatment

To determine the timing and magnitude of smoke as an experimental treatment, we quantified fluxes of PAR. In addition to the Holiday Farm Fire, 4 other major fires occurred in the region in September 2020, creating an immense layer of smoke that reduced local air quality substantially (Figs 2A, B, 3). We deployed data loggers at the 11 streams (see below) and 2 local clearcut ridgetops (in

Table 2. Summary statistics for the effects of smoke and fire on stream temperature at 11 study sites in the western Cascade Range, Oregon, USA, as well as for 2 local ridgetop photosynthetically active radiation (PAR) loggers in the region. Fire effect is the change in maximum stream temperature on the day of the fire compared with the mean of the previous week's pre-fire maxima. Smoke effect is the change in the maximum stream temperature during the smoke period compared with the mean of the previous week's pre-fire maxima. Negative values indicate decreases in daily maxima. WS = watershed, SBS = soil burn severity, RAVG = Rapid Assessment of Vegetation Condition after Wildfire, FE − ST = fire effect − stream temperature, SE − ST = smoke effect − stream temperature, Max = maximum stream temperature during fire, S max = maximum stream temperature during summer 2020, mod = moderate, − = no data.

Category	Site	WS burned (%)	Mean WS RAVG	Mean Riparian RAVG	Mean WS SBS	Mean Riparian SBS	SBS class	PAR Δ (%)	FE − ST	SE − ST	Max	S Max
Unburned	CA1	0.6	0.01	0.00	0.01	0.00	None	−69.4	−1.14	−1.5	13.6	15.9
	QZ1	7.8	0.33	0.00	0.21	0.00	None	−60.8	−0.04	−2.2	14.6	16.5
Burned watershed	ML3	69.8	1.41	0.38	1.38	0.52	None	−32.8	0.3	−0.8	15.0	16.4
	LF	38.8	1.9	0.81	1	0.24	None	−[a]	0.1	−1.6	14.8	16.0
	QZ2	67.4	2.76	1.34	1.71	1.52	Low	−69.9	−0.1	−1.6	14.5	15.8
Burned riparian	CA2	39.0	1.07	2.23	0.82	1.30	Low	−86.8	−0.6	−0.8	11.4	13.0
	ML2	98.7	1.57	2.18	1.29	2.24	Low	−60.5	0.4	−1.0	15.5	16.8
	ML1	41.3	0.91	2.76	0.87	2.33	Low	−25.2	0.6	−0.5	14.4	15.7
	EN1	97.6	3.89	3.78	2.47	2.69	Low	−51.8	0.8	−1.2	15.6	17.0
	WG	100	3.21	5.94	2.41	3	Low/mod	−[a]	0.3	−1.4	15.9	17.5
	EN2	100	6.17	6.23	3.03	3	Mod	−82.4	4.5	−0.6	19.3	16.4
Ridgetop PAR	Mill Creek	–	–	–	–	–	–	−80.2	–	–	–	–
	Calapooia	–	–	–	–	–	–	−77.3	–	–	–	–

[a] PAR was not measured at LR or WG sites.

the Mill Creek and Calapooia basins, each within 30 km of all sites) in full sun, which were set to log every 10 min (Odyssey® Submersible PAR logger or Odyssey Xtreem PAR logger; Dataflow Systems, Christchurch, New Zealand) (Fig. 3). Odyssey PAR loggers recorded photosynthetic irradiance, which we converted to PAR (mol m^{-2} s^{-1}) using a miniPAR® side-by-side calibration (Precision Measurement Engineering, Vista, California) as a reference meter, set to log every 1 min for 24 h. We calculated an irradiance/PAR relationship forced through the origin for each logger. To calculate a smoke effect on PAR, we summed the total daily PAR values, then subtracted the mean of the post-fire values from the pre-fire values.

Logger deployment and stream data collection

We deployed data loggers at 11 streams that were used here to capture the repsponses of stream air temperature and DO to fire and smoke treatments. Differences in instrumentation and logging intervals across sites reflect the original designs of 2 different studies. Two streams (WG and LF) were each instrumented with 1 HOBO® TidbiT® v2 temperature logger (accuracy ±0.2°C; Onset®, Bourne, Massachusetts) set to log at 1-h intervals and placed in PVC housing to ensure that recorded temperature measurements were not affected by direct sunlight. The protected loggers were submerged ~10 to 20 cm deep in pools in the center of the channel. In the other 9 streams, we established 200- to 300-m study reaches, each with 5 sampling locations 50 to 75 m apart (m 0 at the downstream end and m 200 or 300 at the upstream end). The spacing differed corresponding to the length of stream reaches, which were delineated by planned timber harvest units. TidbiT v2 water temperature data loggers secured inside PVC housings were installed at these locations and submerged ~10- to 20 cm deep in pools but were set to record at 30-min intervals. At the 0-m location of the 9 more intensively sampled streams, we also installed miniDOT® loggers (Precision Measurement Engineering) recording DO concentration, DO % saturation, and stream temperature (accuracy ±0.1°C) alongside PAR loggers, both logging at 10-min intervals. Only recent versions of PAR loggers (Odyssey®) record air temperature (±0.5°C), which we installed at 5 sites (ML1, ML2, ML3, CA1, CA2). At the other 4 sites (EN1, EN2, QZ1, QZ2), we installed HOBO Pendant® Temperature/Light 64K data loggers (±0.53 from 0–50°C) attached to rebar placed in the center of the stream channel ~20 cm above the water's surface. These loggers captured light intensity and air temperature every 2 h.

We performed accuracy checks as described in product manuals on all sensors with side-by-side logging before

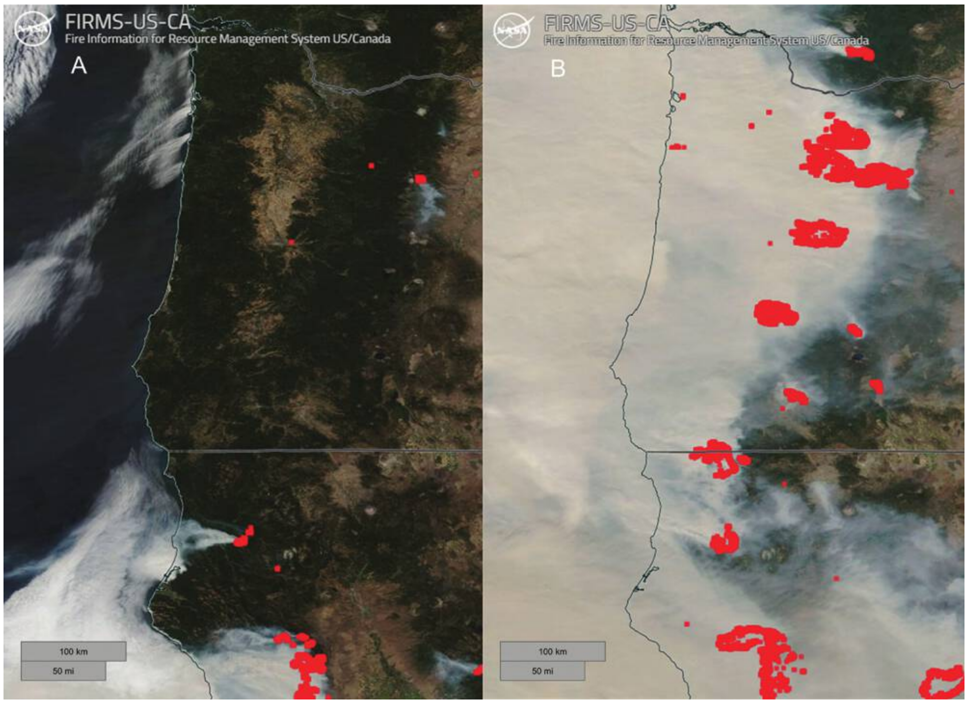

Figure 2. Satellite images of western Oregon, USA, before the Holiday Farm Fire started (1 September 2020) (A) and after this and other nearby fires burned for 9 d (10 September), creating a dense layer of smoke over the region (B). We acknowledge the use of data and imagery from NASA's Fire Information for Resource Management System (FIRMS; https://firms.modaps.eosdis.nasa.gov/usfs/map/#t:adv;d:2020-09-08;l:noaa20-viirs,viirs,modis_a,modis_t,country-outline,viirs_crtc;@-121.6,43.7,8z), part of NASA's Earth Observing System Data and Information System.

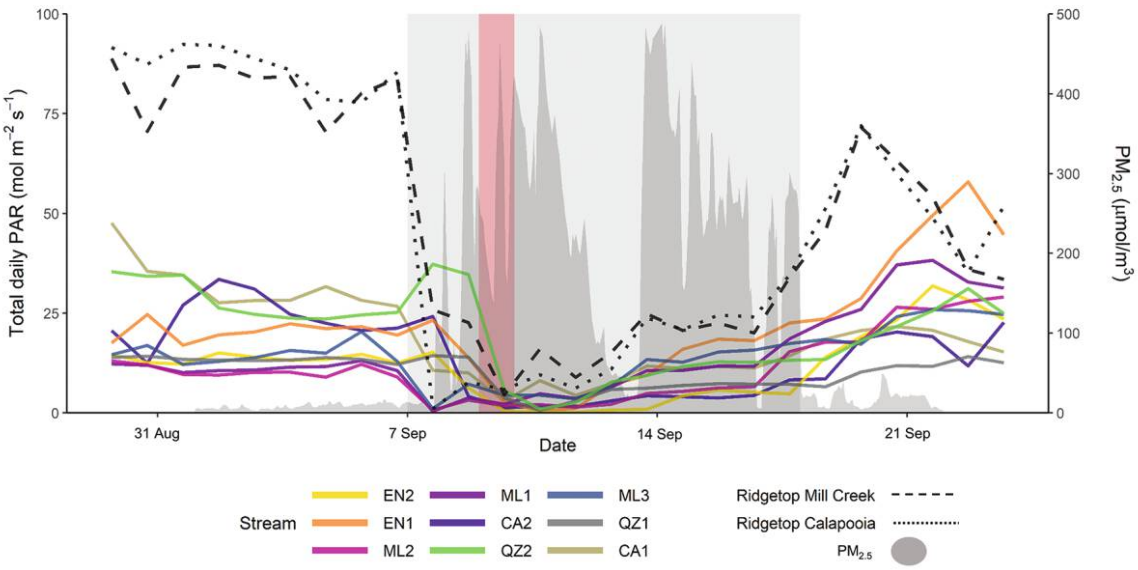

Figure 3. Total daily photosynthetically active radiation (PAR; mol m^{-2} s^{-1}) measured at 2 local ridgetops (dashed [Calapooia] and dotted [Mill Creek] black lines) and at the downstream end of 9 sites (colored lines) before, during, and after the Holiday Farm Fire in September 2020 in the western Cascade Range, Oregon, USA. Concentration of particulate matter smaller than 2.5 µm (PM$_{2.5}$), shown by the dark gray area, is a measure of air quality in nearby city Springfield, Oregon. The red rectangle indicates when the fire burned the affected streams (9 September [Sep]), and the light gray rectangle indicates when smoke was present in the region. See Table 1 for stream subbasin codes and stream characteristics. Aug = August.

and after deployment to check for drift. Though DO sensors did not have wipers, fouling is very unusual in these low-productivity streams, and examining data from the previous months did not indicate that fouling conflated the fire response.

Air and stream temperature responses to fire and smoke

To evaluate responses to the fire, we identified 3 time intervals for analysis: 1) pre-fire (1–6 September), 2) during fire (9 September, 7 September for WG), and 3) post-fire (11–17 September). These intervals reflect acute responses that occurred during the fire and immediate responses that occurred 2 to 8 d after the fire. From time-stamped burn maps along with air temperature loggers at most sites, we determined that all burned sites were affected on 9 September (except WG, which burned on 7 September). For the pre-fire period, we excluded the few days before the fire reached the sites to reflect true pre-smoke conditions, though some smoke from other regional fires was present in this period. To reduce seasonal changes that may underlie temporal comparisons, we chose to analyze only the week before the fire began for pre-fire conditions, acknowledging that streams would have changed slightly regardless of wildfire, but these changes were likely minimal. The post-fire period captures the time after the fire had passed at the burned sites but while the Holiday Farm Fire and other large fires in the western Cascades continued burning, maintaining a heavy layer of smoke. We excluded the day immediately following the fire (10 September, 8 September for WG) in the post-fire assessment to account for residual smoldering at the burned sites. Precipitation data for this region indicate that the 1st rain of the season occurred on 18 September, which reduced the smoke layer substantially (Fig. 3).

To capture the acute change in stream and air temperature due to fire and smoke, we calculated fire and smoke effects for each stream. The fire effect was defined as the difference between the maximum temperature on the day of the fire (during) and the mean maximum daily temperature from the pre-fire period (pre). We used maximum daily stream temperature because this temperature metric has the most relevance to biota in a fire disturbance (Dunham et al. 2003). To calculate the fire effect at unburned sites, we used 9 September as the day of fire to parse fire effects from smoke effects at burned sites. The smoke effect was defined as the difference between the mean maximum daily temperatures in the post-fire and pre-fire periods (Fig. 4A, B). We calculated fire and smoke effects for other metrics (means, minima, ranges) similarly to the maxima, but the terms fire effect and smoke effect hereafter refer to maxima unless otherwise specified. For summary statistics of longitudinal variation within sites, data from all transects were included; however, in other analyses, the logger that recorded the most extreme response was selected to represent the sites because not all sites had longitudinal replication.

To determine whether the means of daily maximum stream and air temperatures during the pre-fire period differed from the post-fire period ($n = 11$ streams for stream temperature, $n = 9$ streams for air temperature) because of the effects of smoke, we performed 1-sided, paired t-tests.

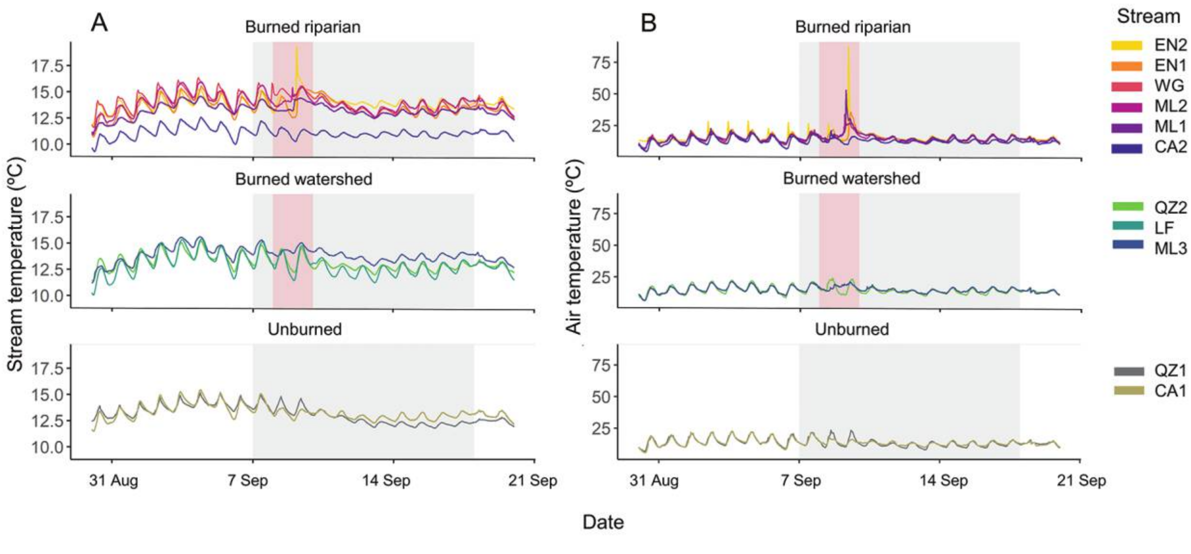

Figure 4. Stream (A) and air (B) temperature across 11 streams in the western Cascade Range, Oregon, USA, including burned riparian sites where the Holiday Farm Fire in September 2020 reached the streams, burned watershed sites where fire was present in the watershed but not near the streams, and unburned watersheds. The red rectangle indicates the period when the fire burned affected sites (7 September [Sep] for stream WG, 9 Sep for all others), and the gray rectangle indicates the period when smoke from this and other fires was present in the region. The legend groups streams by category: top 6 are burned riparian, middle 3 are burned watershed, and bottom 2 are unburned. Aug = August.

DO responses to fire and smoke

We evaluated DO concentrations at 7 sites through the pre-fire, during-fire, and post-fire periods (i.e., WG and LF did not have DO loggers, QZ2 and QZ1 were omitted because of issues with logger reliability). We quantified changes in the daily maximum, minimum, mean, and range of both DO concentration (mg/L) and DO % saturation during the 3 time periods, following the same fundamental methods as our assessment of stream temperatures. However, for DO we focused on minimum % saturation values in quantifying fire and smoke effects, in contrast to maximum values used in the temperature analysis, because biota in these systems are more sensitive to declines, not increases, in DO. We performed 1-sided, paired *t*-tests on means of daily minima and ranges (diurnal variation) in DO % saturation in the pre-fire and post-fire periods to detect changes attributable to smoke ($n = 7$).

RESULTS

Fire treatment metrics

Many of our study sites were on the margins of the Holiday Farm Fire and experienced low to moderate severity burn patchiness in their watersheds and riparian zones (Fig. 1). Percent watershed burned ranged from 0.6% at CA1 to 100% at EN2 and WG (Table 2). Mean watershed RAVG scores ranged from 0.01 to 6.17 (out of a maximum scale of 7), but 8/9 burned streams had watershed RAVG < 4, indicating low burn severity. Mean riparian RAVG scores ranged from 0 to 6.23, exceeding respective watershed values at most of the burned riparian sites. Similarly, the watershed SBS values reflected low severity fire at 8 burned sites (RAVG < 3) and moderate at 1 burned site (RAVG = 3.03). Mean riparian SBS scores ranged from 0.01 to 3.03 and, like RAVG scores, exceeded respective watershed values at the burned riparian sites. Fire metrics were strongly correlated ($r > 0.72$, $p < 0.01$), with the strongest relationships between watershed SBS and watershed RAVG ($r = 0.95$, $p < 0.0001$), riparian SBS and riparian RAVG ($r = 0.92$, $p < 0.0001$), and % watershed burned and watershed SBS ($r = 0.9$, $p = 0.0001$; Table 3).

Stream and air temperature fire effects were strongly correlated with fire severity (i.e., RAVG and SBS; Table 4). The strongest correlations for the stream temperature fire effect were mean watershed RAVG ($r = 0.84$, $p = 0.001$) and mean watershed SBS ($r = 0.73$, $p = 0.01$), followed by mean riparian RAVG ($r = 0.69$, $p = 0.02$). The air temperature fire effect was strongly correlated with mean watershed and riparian RAVG and SBS but was most closely correlated with mean riparian RAVG ($r = 0.9$, $p = 0.001$). DO fire effects correlated best with both mean watershed RAVG ($r = 0.85$, $p = 0.02$) and watershed SBS ($r = 0.81$, $p = 0.03$; Table 4). The DO % saturation smoke effect was strongly correlated with mean watershed and riparian RAVG ($r = -0.82$ and -0.93, $p = 0.03$ and 0.003, respectively), as well as mean riparian SBS ($r = -0.86$, $p = 0.01$); however, the air temperature, stream temperature, and PAR smoke effects did not correlate well with any fire severity metrics (Table 4).

PAR

In the consistently sunny week before the fires (1–6 September), the mean total daily PAR at the 2 ridgetop loggers was 86.2 and 80.5 mol m^{-2} s^{-1}. As smoke lingered in the region in the period from 11 through 17 September, mean total daily PAR declined to 17.1 and 18.3 mol m^{-2} s^{-1}, a change of −80 and −77%, respectively (Table 2). Similarly, although the streams were all heavily shaded, they also experienced reductions in mean total daily PAR by an average of 11.77 mol m^{-2} s^{-1} (Fig. 3). Across streams, mean total daily PAR prior to the fire ranged from 10.4 to 30.5 mol m^{-2} s^{-1} and declined from 2.4 to 10.2 mol m^{-2} s^{-1}. We compared these patterns with air-quality data measured in Springfield, Oregon, a city ~80 km from the origin of the fire. These data reflect the concentration of air particulate matter smaller than 2.5 μm ($PM_{2.5}$) for public health purposes (Springfield City Hall monitoring station 5; https://oraqi.deq.state.or.us/home/map). There was a slight lag in $PM_{2.5}$ because of the distance between Springfield and our PAR loggers; however, $PM_{2.5}$ reached a maximum of 557 μmol/m^3 during the fire period on the same day PAR declined (Fig. 3).

Temperature

Acute localized stream warming during the fire was only detected at some burned riparian sites, and where warming occurred, it was limited in magnitude and brief in duration.

Table 3. Correlations (Pearson's *r*) between burn severity metrics at 11 study sites in the western Cascade Range, Oregon, USA. All correlations had *p*-values ≤0.01. RAVG = Rapid Assessment of Vegetation Condition after Wildfire, SBS = soil burn severity, WS = watershed, – = redundant correlations or correlations between the same metric.

	WS SBS	Riparian SBS	WS RAVG	Riparian RAVG	% WS burned
WS SBS	–	0.81	0.95	0.85	0.90
Riparian SBS	–	–	0.72	0.92	0.83
WS RAVG	–	–	–	0.81	0.78
Riparian RAVG	–	–	–	–	0.75

Table 4. Correlations (Pearson's r-values with corresponding p-values) between burn severity metrics and physical stream responses for 9 to 11 study sites in the western Cascade Range, Oregon, USA. RAVG = Rapid Assessment of Vegetation Condition after Wildfire, SBS = soil burn severity, T = temperature, DO = dissolved oxygen, PAR = photosynthetically active radiation, WS = watershed.

	Burn severity metric									
Response	WS SBS	p	Riparian SBS	p	WS RAVG	p	Riparian RAVG	p	% WS burned	p
Stream T: Fire effect	0.72	0.01	0.59	0.06	0.84	0.001	0.69	0.02	0.56	0.07
Stream T: Smoke effect	0.37	0.26	0.54	0.09	0.33	0.33	0.45	0.16	0.41	0.21
Air T: Fire effect	0.72	0.03	0.82	0.007	0.77	0.02	0.90	0.001	0.58	0.10
Air T: Smoke effect	0.30	0.44	0.49	0.19	0.16	0.68	0.32	0.40	0.46	0.22
DO: Fire effect	−0.81	0.03	−0.53	0.23	−0.85	0.02	−0.71	0.07	−0.65	0.11
DO: Smoke effect	−0.74	0.06	−0.87	0.01	−0.81	0.03	−0.93	0.003	−0.64	0.12
PAR: Smoke effect	0.16	0.69	0.27	0.49	0.04	0.92	0.14	0.73	0.32	0.40

The highest maximum stream temperature on the day of the fire was 19.3°C, which occurred at EN2 (Fig. 4A, B), the most intensely burned site by all fire severity metrics (Table 2). At this site, 1 logger recorded a fire effect of 4.5°C. This peak in stream temperature was brief, with only 1 value in the 30-min logging interval reaching that magnitude. Fire effects at the other burned riparian sites were much smaller (+0.8°C at EN1 to −0.6°C at CA2) and only slightly larger than the fire effects at burned watershed sites (+0.3 to −0.1°C; Table 2). Fire effects at unburned sites were neutral (−0.04°C) to moderately negative (−1.1°C) on the day of the fire, likely because of smoke. The change in mean stream temperature on the day of the fire at burned riparian sites was between 0.3 and 1.2°C and was mixed and less severe at burned watershed sites (+0.3 to −0.4°C), including 3 sites where means decreased. Results were mixed for daily stream temperature minima and ranges across all burned sites on the day of the fire. Daily minima decreased by <0.7°C or increased by <0.9°C. Daily range increased by <4.5°C and decreased by <0.1°C on the day that the fire burned through each watershed and stream riparian area (Table S1). Acute effects of fire on stream temperature were consistent along reaches (200–300 m reaches with loggers 50–75 m apart) at all burned sites, except at site EN2, which experienced the most severe burning and responded with the highest fire effect (Fig. 5A, B). At this site there was a difference of 2.8°C between the highest (m 150) and lowest (m 300) fire effects.

Air temperature was elevated on the day of the fire across most burned riparian sites (except CA2; Fig. 4B) and corresponded in magnitude with increases in stream temperature (Fig. 4A), also evidenced by a strong correlation between their fire effects ($r = 0.92$, $p = 0.0005$). Peaks in air temperature were brief, not extending for more than a few hours before returning to a normal range. The hottest air temperature measured was at EN2 at the upper end of the reach, m 300, with a maximum of ~87.3°C (although this value exceeded the sensor's range of linear calibration) but did not correspond with the greatest change in stream temperature, which was observed at m 75. This recorded air temperature maximum at EN2 was nearly 61.5°C warmer than the mean of pre-fire daily air temperature maxima at this site. At EN1 and ML1, air temperatures over the stream showed similar but less dramatic peaks during the fire, reaching 41.9 and 53.0°C, respectively.

The cooling effect of smoke was evident at all sites (Table 2, Fig. 4A, B). Means of daily maximum stream temperatures in the post-fire period were 1.2°C less than the pre-fire period across sites ($t = -7.6$, $p < 0.0001$). More broadly, with the exception of a few sites that experienced small increases in minima, nearly all temperature metrics (daily maxima, minima, means, and ranges) decreased across all sites during the post-fire smoke period (Table S1). Means of daily air temperature maxima also decreased at all sites compared with means of pre-fire daily maxima by 4.6°C on average ($t = -7.4$, $p < 0.0001$).

DO

We observed a range of responses in DO corresponding with burn severity. In the most severely burned stream (EN2), which also had the largest temperature response, we observed a rapid decline in DO concentrations as soon as fire reached the site. At this site, the average of the daily DO concentration minima was 8.8 mg/L before the fire and fell to 7.1 mg/L on the day of the fire (Table S1). There was also a substantial decline in DO % saturation from 92.4 to 75.6 (fire effect of −16.9%), indicating that this decline was not solely due to changing oxygen solubility associated with increasing temperatures. A notable decline in DO concentration and DO % saturation was also apparent in other burned riparian and burned watershed sites (EN1, ML1, ML2, ML3), though the responses were much smaller (fire effects of −4.0 to −7.8% saturation). Even though fire burned the riparian zone at CA2, DO % saturation decreased only slightly (−1.3%). Unlike temperature, DO % saturation recovered slowly over multiple days in these sites

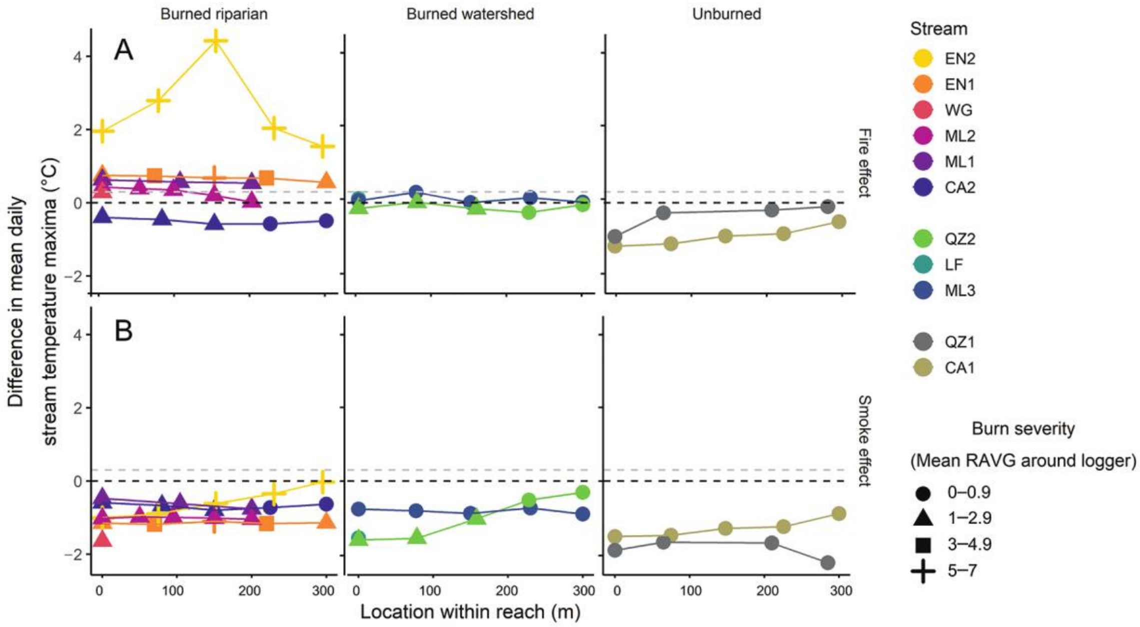

Figure 5. A longitudinal comparison of the fire effect (change in mean daily maxima from the pre-fire period to the day of fire; A) and smoke effect (change in mean daily maxima from the pre-fire to post-fire period; B) across 5 logger locations within each of 9 study streams (and 1 logger location in 2 streams) in the western Cascade Range, Oregon, USA, affected by the Holiday Farm Fire in September 2020. Reach location extends from m 0 at the downstream end to m 300 at the upstream end. Shapes represent average Rapid Assessment of Vegetation Condition after Wildfire (RAVG) score in a 100-m buffer around each logger location to denote burn severity. The dashed black line at 0 indicates no effect, and the dashed gray line indicates the forest practices regulatory standard for temperature change in Oregon (0.3°C). In the legend, streams are grouped by category: top 6 are burned riparian, middle 3 are burned watershed, and bottom 2 are unburned.

(Fig. 6C). The 1st rain event that occurred ~9 d after the fire also elicited abrupt drops in DO that occurred in all the burned streams and in 1 unburned stream (Fig. 6A–F). DO % saturation at burned streams returned to a relatively normal diurnal pattern following these storms, but they never quite returned to their pre-fire magnitudes, transitioning from a maximum of ~100% saturation to ~90%, compared with an unburned stream (CA1) where DO % saturation returned quickly after the smoke cleared (Fig. 6D).

Across all 7 DO sites, the mean DO % saturation daily minima and ranges (diurnal variations) declined during the period of intense smoke, though signals at burned riparian streams are conflated with the fire response. Means of daily minima decreased by a mean of 2.2 % saturation ($t = -2.9$, $p = 0.01$, 1-sided paired t-test), with slightly larger declines in the burned sites (Fig. 6C, D). Similarly, daily mean ranges decreased by 0.8 % saturation on average ($t = -3.7$, $p = 0.005$). In the pre-fire period, the daily range in DO % saturation had a mean of 1.9% and varied from 1.2 to 2.6% (Table S1). In the post-fire/high-smoke period, mean daily range in DO % saturation shifted to 1.1% and varied from 0.6 to 2.4%.

DISCUSSION

Few studies have concurrent measurements of stream conditions from immediately before, during, and after fire across multiple streams that experienced a range of burn severities from a single large wildfire (Hitt 2003, Mahlum et al. 2011, Schultz et al. 2017). In this study we sought to assess acute changes in key stream parameters during wildfire and during the weeks after as smoke lingered, as well as to determine which fire severity metrics may be most useful for predicting stream responses to fire. We used data from 11 headwater streams that recorded air temperature, stream temperature, DO, and PAR as the Holiday Farm Fire burned the adjacent uplands to various extents and severities (or not at all at 2 streams). Although we expected the fire itself to have more substantial acute effects on our study systems, we found that overall, smoke was likely more impactful on changing stream conditions because of stark reductions in PAR availability. Smoke was regionally synchronous, covered a larger area, and lasted longer than the acute effects of the fire. In addition, acute temperature responses were transient, rarely exceeded previously observed maxima, and were not detected consistently along reaches, whereas DO declines were more pronounced and prolonged. At the burned sites, biological consequences from immediate temperature and oxygen changes were likely minimal. Of the fire metrics tested, mean watershed and riparian RAVG, and to lesser extents mean watershed and riparian SBS, were more useful in relating remotely sensed data to observed fire effects

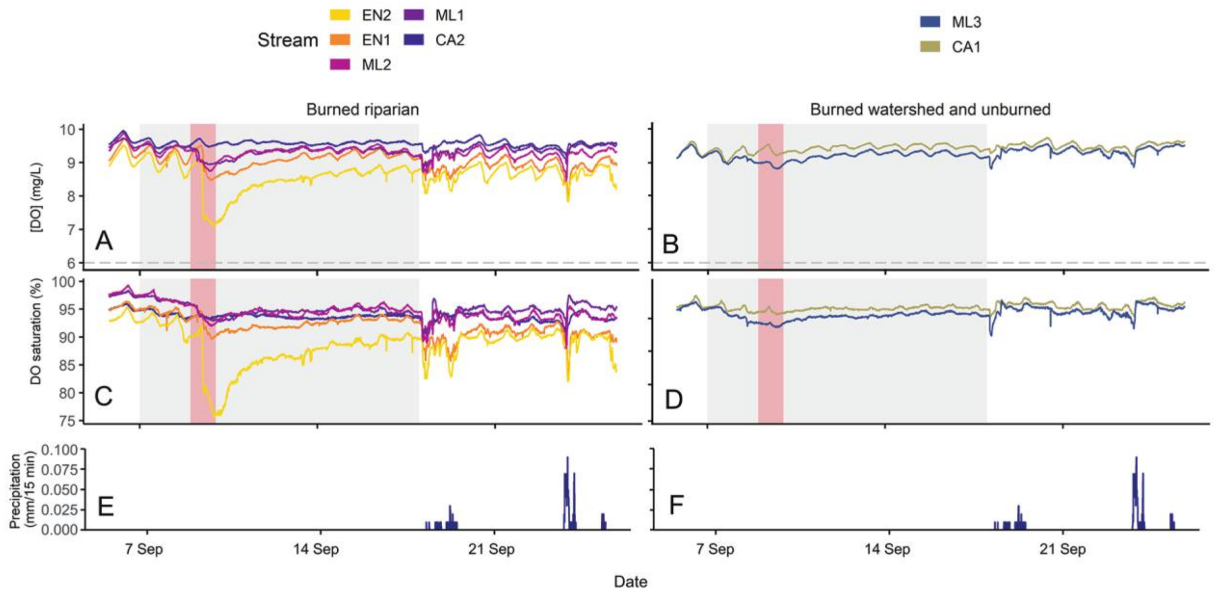

Figure 6. Dissolved oxygen (DO) concentration (mg/L) and % saturation for burned riparian sites (A, C) and both burned watershed and unburned sites (B, D), with data available for 7/11 streams in the western Cascade Range, Oregon, USA, affected by the Holiday Farm Fire in September (Sep) 2020. Red rectangle indicates the period when the fire was burning affected sites (9 Sep), and the gray rectangle indicates the period when smoke was present in the region. The gray dashed line represents the 7-d mean minimum threshold (6 mg/L) to maintain adequate conditions for coldwater species in Oregon (ODEQ 2019). Dark blue lines in panels E and F show precipitation at the Cougar Dam Meteorological Station near Rainbow, Oregon (elevation: 391 m a. s. l). Precipitation data is available from the United States Geological Survey's National Water Information System: https://waterdata.usgs.gov/or/nwis/uv/?site_no=440752122143200&PARAmeter_cd=00045.

than % watershed burned, a more commonly used metric in assessing the degree of damage to watersheds.

Stream and air temperature responses to fire

Stream warming from wildfire at the study sites was minimal, in part because of their locations on the perimeter of the fire boundary and within primarily lower severity burn patches. EN2, a 3rd-order stream that experienced moderate severity fire in the riparian zone and adjacent uplands, was the exception, where we observed stream temperatures beyond a typical daily maximum. However, these temperature increases were short lived. Stream size may play a role in buffering temperature increases, as evidenced by the small fire effect at WG, a low/moderately burned stream larger than EN2, though more replication is needed to understand the mechanisms responsible for these responses. Light and air temperature are the main drivers of stream temperature change (Johnson 2004), but because light at the stream surface was greatly reduced by smoke (Fig. 3) and stream and air temperature peaks aligned temporally during the fire (Fig. 4A, B), we conclude that warm air was the central mechanism in the stream heating we observed. However, we found the greatest change in stream temperature at a reach location that had only a moderate change in air temperature, not the location with the greatest air temperature increase. Either there are other confounding mechanisms, or parcels of heated water were quickly mobilized and detected a short distance downstream. Further, the longitudinal heterogeneity in stream temperatures we observed demonstrates that in severely burned watersheds, acute stream temperature responses can be patchy at small scales. This variability suggests that although it is valuable to retrieve any logger that recorded during a fire, the degree to which single measurements of stream temperature represent responses of entire streams should be interpreted with the acknowledgement that acute effects elsewhere in the stream may be much larger or much smaller.

Our results fit within findings from the limited number of other studies that captured stream temperature during a wildfire. The 4.5°C increase in maxima (peak 19.3°C) at EN2 (3.4 km^2) was similar in magnitude to the 7.8°C difference (peak of 17.2°C) found at a 3rd-order stream (25.2 km^2) in Montana when compared with a larger neighboring unburned stream (Hitt 2003) and to a ~2 to 4°C increase in the headwaters of a dry drought-prone basin in southeastern Oregon where the riparian zone burned completely, with the difference that, in that case, temperature increases persisted well after the fire (Schultz et al. 2017). Observations at our less affected burned streams were more consistent with the findings by Mahlum et al. (2011), where even a high severity fire in the uplands resulted in no change in maximum stream temperatures, though many of these streams were larger than those in our study.

In Oregon, the forest practices' regulatory threshold is 0.3°C for increased stream temperature, which was exceeded by most burned riparian sites during the day of the fire (ODEQ 2019). However, given the short duration of the impact, literature on temperature stress suggests that it is unlikely that acute heating alone would have harmed thermally sensitive biota, such as Coastal Cutthroat Trout, in our study systems (Bjornn and Reiser 1991). Studies assessing Coastal Cutthroat Trout thermal tolerances found that marked stress is observed when temperatures exceed 22°C, with mortality typically occurring ~28 to 30°C (Johnson et al. 1999), but maximum stream temperatures in this study did not rise above 20°C in any of the study streams and were very short lived (peaked within 1 h and exceeded 15°C for 6 h). Stream temperature at all sites except EN2 did not exceed that year's summer maxima in late July and mid-August (Table 2). Taken collectively, these results lead us to expect that maximal temperature effects during a fire will occur in the smallest streams, with reduced effects in the larger downstream rivers where the volume of water increases, and in our study area, where federally threatened salmonids occur (Rieman and Clayton 1997, Mahlum et al. 2011, Gido et al. 2019, Swartz et al. 2020).

Stream and air temperature responses to smoke

In contrast with the highly localized effects of the Holiday Farm Fire on stream temperatures, smoke effects from this and other Labor Day fires had a regional dampening effect on stream and air temperatures that encompassed sites within and outside the burned area and persisted for >1 wk (Fig. 2A, B). The lack of strong correlation between stream PAR and burn severity metrics indicates that all sites experienced similar air and stream temperature smoke effects regardless of proximity to the fire (Table 4). Persistence of smoke is dependent upon local weather patterns but can last for weeks to months (Scordo et al. 2021). Because the fires were punctuated by a rain event that cleared much of the smoke, we cannot estimate how long PAR reductions would have persisted, though we know that light was still ~½ of its normal intensity 1 wk after the fires began (Fig. 3).

The smoke effect at all sites averaged −1.2°C, a difference larger in magnitude than the mean temperature increases observed during the fire at the burned riparian sites (+1.0°C). Not only did maxima decrease, but means, ranges, and most minima did as well (Table S1), highlighting the importance of solar radiation on stream thermal regimes. Smoke effects have not been widely explored in the literature, but a study in the Klamath basin in California, USA, also reported substantial declines in daily stream temperature maxima and range, attributable to wildfire smoke (David et al. 2018). Overall, although heat from fire in our study affected local stream temperatures at some streams, smoke and associated reductions in light had a more uniform effect on stream temperatures across the landscape; therefore, smoke, rather than heat from the fire itself, warrants consideration as the dominant effect on stream and air temperatures associated with fire, at least for low to moderately burned streams.

DO responses to fire

The quantification of stream DO responses during and shortly after wildfire is rare, and monitoring during the Holiday Farm Fire offered a novel opportunity to assess the effects of fire largely independent of post-fire rain events, which are known to depress DO via pulses of sediment (Lyon and O'Connor 2008, Dahm et al. 2015, Reale et al. 2015). During the fire we found distinct and persistent DO declines in both concentration and % saturation in all burned riparian and burned watershed sites except 1 of the least severely burned sites (CA2). In the Rio Grande near Los Alamos, New Mexico, USA, Dahm et al. (2015) similarly found rapid and severe declines in DO as a post-fire sediment pulse moved through the network, which they suggested could occur through both biotic and abiotic pathways. Biologically, they suggest that reductions in DO are associated with increased demand for oxygen by aerobic heterotrophs capitalizing on the rapid influx of dissolved and fine particulate organic matter. Biologically caused DO reductions were likely occurring in our streams during the 1st few rain events following the fire (18, 19, and 24 September), but these DO declines related to rain events were short lived and less severe relative to responses during the fire. In addition, the sheer magnitude and rate of DO decline at EN2 suggests that 1 or more abiotic drivers may have a larger influence than biotic drivers. Results from Dahm et al. (2015), as well as an in-situ ash addition experiment (Earl and Blinn 2003), suggest enhanced chemical oxygen demand associated with ash could explain this response. In addition, temperature is well known to influence DO, but the similar responses in both DO concentration and % saturation indicate that the DO declines we found likely occurred independent of temperature change.

Another study monitoring streams during a fire, Spencer and Hauer (1991), found that nutrient concentrations increased because of ash deposits and diffusion of smoke into stream water in the absence of precipitation. Given the exchange of air and stream water, we suggest that an additional potential mechanism at play in burned riparian areas could be the formation of an oxygen diffusion gradient created from the fire's demand for oxygen as it burned over the stream, affecting the concentration of oxygen in the air incorporated into water via reaeration (Wilhelms et al. 1993). In systems with high rates of reaeration, such as small, steep headwaters, the diffusion gradient could lead to the rapid loss of oxygen from the water without replenishing oxygen from the air. This response would likely be quite ephemeral; although this mechanism may help explain some of the rapid initial responses, it would not account for the slow recovery.

In contrast, others hypothesize that reaeration can instead buffer the drops in DO associated with increased respiration (Reale et al. 2015). It is likely that the changes in DO observed here are some combination of the above processes, and more empirical work is needed to understand acute impacts of fire on oxygen demand.

Declines in DO were greatest in burned riparian sites with coincident stream temperature increases and higher mean RAVG scores, indicating that basin-scale overstory mortality may aid in identifying locations susceptible to DO declines during wildfire. Acute reductions in DO associated with fire can have important implications for stream biota (Gresswell 1999), but our results revealed minimum DO did not surpass regulatory thresholds for the state of Oregon (7-d absolute minimum of 6 mg/L) and were well above the lethal thresholds for salmonids (<3.9 mg/L for 1 d) (ODEQ 2019, WSDE 2002). Where higher severity conditions occur, more substantial declines in DO are possible, and although fish can often withstand short-term increases in temperature or find thermal refuge, they are more sensitive to long periods with severely depleted oxygen (WSDE 2002). Furthermore, we focused on local effects of fire on DO, but longitudinal impacts on DO following post-fire monsoon pulses beyond the burn scar have been documented and modeled (Dahm et al. 2015, Ball et al. 2021, Reale et al. 2021), and there is clear potential for impacts to accumulate and affect biota along the river continuum.

DO responses to smoke

As with stream temperature, changes in stream DO patterns in the smoky post-fire period occurred regionally in burned and unburned sites alike. Across all streams, the diurnal patterns of stream DO % saturation were muted during the smoke period, though some of these declines were conflated with residual fire effects. In the week prior to the fires, all but 1 site (CA2) exhibited clear daily fluctuations in DO, with daily maxima and nightly minima in % saturation, reflecting dominant processes of photosynthesis and respiration. When smoke covered the region, we saw major declines in these daily fluctuations, presumably because of reductions in temperature and light that govern both autotrophic and heterotrophic microbial processes. This finding is contrary to observations by Sherson et al. (2015), where smoke had little effect on DO fluctuations; however, our sites were subject to smoke from a complex of high severity fires, which may have amplified the effect. Although we did not have the ancillary data needed to fully model changes in gross primary production or ecosystem respiration at these sites, the mean 40% decline in the daily range of DO % saturation across sites may reflect a change in benthic primary production at the landscape scale. Unlike acute effects that occur only locally, or longitudinal impacts on DO that have been shown to extend beyond the burn scar (Dahm et al. 2015, Ball et al. 2021, Reale et al. 2021), the presumed effects of smoke on ecosystem processes extend beyond the western Cascades and can affect streams as far as smoke effects occur, which substantially increases the overall ecological impact of wildfire on streams.

Evaluation of burn-severity metrics

Fire exerts a range of effects on aquatic ecosystems that vary with fire extent, severity, vegetation (e.g., species, age), and watershed size, resulting in different responses among streams within the same fire boundary (Pettit and Naiman 2007). Most studies report % of watershed affected by fire (Mahlum et al. 2011, Beakes et al. 2014), but the more detailed and publicly available SBS and RAVG metrics should be added to the suite of resources evaluated by aquatic resource managers and research entities. RAVG is released within 30 to 45 d after fire containment, which makes it particularly useful to rapidly initiate post-fire monitoring. Across the study watersheds, we found that the stream temperature fire effect was most strongly positively correlated with the percentage of basal area mortality as estimated by mean watershed RAVG (Table 4). Interestingly, RAVG and SBS metrics were more useful overall in accounting for variability in stream temperature, air temperature, and DO responses than % watershed burned, which was not correlated with any response despite being strongly correlated with the fire metrics themselves. Despite the fact that RAVG and SBS incorporate different parameters and time scales and are developed for different purposes, they are very strongly correlated at both the watershed and riparian scales ($r > 0.9$ for both), suggesting that though RAVG was the best predictor of change in stream temperature, air temperature, and DO in this analysis, SBS is also informative and predictive of the effects of fire on streams.

Broader implications and future directions

In contrast to our expectations, we found that even at moderately burned sites, acute fire effects on physical stream parameters were muted, with reductions in light from heavy smoke potentially causing more widespread ecological consequences. More observations are needed from streams that experience high severity burning, and our data suggest that higher severity fire conditions may lead to higher peaks in stream temperature, especially for small streams. However, it should be noted, that >95% of the riparian landscape in the Holiday Farm Fire burned at the same fire severity as observed across our sites, therefore fire effects beyond our observations in this specific fire were probably rare. We conclude that in landscapes that experience similarly patchy fire severity, temperatures are unlikely to exceed thermal tolerances for extended amounts of time and may be localized to individual reaches in streams. In contrast, DO responses may be more sensitive to higher fire severity, but more research is needed to confirm what drives immediate DO reductions and if mechanisms change with fire severity.

Our study illustrates the far-reaching and ubiquitous effects of smoke from wildfires on stream and air temperatures, light, and DO, but there are aspects of stream process and function that we did not investigate. As large-scale wildfires and complexes of multiple wildfires continue to burn extensive areas of forested land in the western USA, more work is needed to quantify the magnitude and spatial extent of smoke on stream gross primary production and influences on aquatic food webs. Further research should also address the finer-scale changes in vertebrate bioenergetics that are likely to take place when stream temperatures are quickly reduced during the wildfire season (i.e., typically late summer), a time when stream temperatures would otherwise be at their peak. In addition, sites in this study were all on private timberland with relatively similar upland forest stand structures, and we do not fully understand how fires move through riparian zones in managed stands compared with unmanaged late-succession forests. Additional research with observations across a management gradient is needed help to inform best management practices for mitigating fire effects to streams in the future.

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LITERATURE CITED

Abatzoglou, J. T., D. E. Rupp, L. W. O'Neill, and M. Sadegh. 2021. Compound extremes drive the western Oregon wildfires of September 2020. Geophysical Research Letters 48:e2021GL092520.

Abatzoglou, J. T., and A. P. Williams. 2016. Impact of anthropogenic climate change on wildfire across western US forests. Proceedings of the National Academy of Sciences 113:11,770–11,775.

Albin, D. P. 1979. Fire and stream ecology in some Yellowstone tributaries. California Fish and Game 64:216– 238.

Alizadeh, M. R., J. T. Abatzoglou, C. H. Luce, J. F. Adamowski, A. Farid, and M. Sadegh. 2021. Warming enabled upslope advance in western US forest fires. Proceedings of the National Academy of Sciences 118:e2009717118.

Ball, G., P. Regier, R. González-Pinzón, J. Reale, and D. Van Horn. 2021. Wildfires increasingly impact western US fluvial networks. Nature Communications 12:2484.

Beakes, M. P., J. W. Moore, S. A. Hayes, and S. M. Sogard. 2014. Wildfire and the effects of shifting stream temperature on salmonids. Ecosphere 5:63.

Bêche, L. A., S. L. Stephens, and V. H. Resh. 2005. Effects of prescribed fire on a Sierra Nevada (California, USA) stream and its riparian zone. Forest Ecology and Management 218:37–59.

Bixby, R. J., S. D. Cooper, R. E. Gresswell, L. E. Brown, C. N. Dahm, and K. A. Dwire. 2015. Fire effects on aquatic ecosystems: An assessment of the current state of the science. Freshwater Science 34:1340–1350.

Bjornn, T. C., and D. W. Reiser. 1991. Habitat requirements of salmonids in streams. American Fisheries Society Special Publication 19:83–138.

Burke, M., A. Driscoll, S. Heft-Neal, J. Xue, J. Burney, and M. Wara. 2021. The changing risk and burden of wildfire in the United States. Proceedings of the National Academy of Sciences 118:e2011048118.

Cooper, S. D., H. M. Page, S. W. Wiseman, K. Klose, D. Bennett, T. Even, S. Sadro, C. E. Nelson, and T. L. Dudley. 2015. Physicochemical and biological responses of streams to wildfire severity in riparian zones. Freshwater Biology 60:2600–2619.

Dahm, C. N., R. I. Candelaria-Ley, C. S. Reale, J. K. Reale, and D. J. Van Horn. 2015. Extreme water quality degradation following a catastrophic forest fire. Freshwater Biology 60:2584–2599.

David, A. T., J. E. Asarian, and F. K. Lake. 2018. Wildfire smoke cools summer river and stream water temperatures. Water Resources Research 54:7273–7290.

Dunham, J. B., M. K. Young, R. E. Gresswell, and B. E. Rieman. 2003. Effects of fire on fish populations: Landscape perspectives on persistence of native fishes and nonnative fish invasions. Forest Ecology and Management 178:183–196.

Earl, S. R., and D. W. Blinn. 2003. Effects of wildfire ash on water chemistry and biota in South-Western U.S.A. streams. Freshwater Biology 48:1015–1030.

Gido, K. B., D. L. Propst, J. E. Whitney, S. C. Hedden, T. F. Turner, and T. J. Pilger. 2019. Pockets of resistance: Response of arid-land fish communities to climate, hydrology, and wildfire. Freshwater Biology 64:761–777.

Gresswell, R. E. 1999. Fire and aquatic ecosystems in forested biomes of North America. Transactions of the American Fisheries Society 128:193–221.

Halofsky, J. E., and D. E. Hibbs. 2008. Determinants of riparian fire severity in two Oregon fires, USA. Canadian Journal of Forest Research 38:1959–1973.

Hitt, N. P. 2003. Immediate effects of wildfire on stream temperature. Journal of Freshwater Ecology 18:171–173.

Johnson, O. W., M. H. Ruckelshaus, W. S. Grant, F. W. Waknitz, A. M. Garrett, G. J. Bryant, K. Neely, and J. J. Hard. 1999. Status review of Coastal Cutthroat Trout from Washington, Oregon, and California. National Oceanic and Atmospheric Administration Technical Memorandum NMFS-NWFSC-37. Conservation Biology Division, Northwest Fisheries Science Center, National Marine Fisheries Service, Seattle, Washington. (Available from: http://vvww.krisweb.com/biblio/gen_nmfs_johnsonetal_1999_tm37cutthroa.pdf)

Johnson, S. L. 2004. Factors influencing stream temperatures in small streams: Substrate effects and a shading experiment. Canadian Journal of Fisheries and Aquatic Sciences 61:913–923.

Klose, K., S. D. Cooper, and D. M. Bennett. 2015. Effects of wildfire on stream algal abundance, community structure, and nutrient limitation. Freshwater Science 34:1494–1509.

Lyon, J. P., and J. P. O'Connor. 2008. Smoke on the water: Can riverine fish populations recover following a catastrophic fire-related sediment slug? Austral Ecology 33:794–806.

Mahlum, S. K., L. A. Eby, M. K. Young, C. G. Clancy, and M. Jakober. 2011. Effects of wildfire on stream temperatures in the Bitterroot River Basin, Montana. International Journal of Wildland Fire 20:240–247.

Marañón-Jiménez, S., J. Castro, E. Fernández-Ondoño, and R. Zamora. 2013. Charred wood remaining after a wildfire as a reservoir of macro- and micronutrients in a Mediterranean pine forest. International Journal of Wildland Fire 22:681–695.

Minshall, G. W., C. T. Robinson, and D. E. Lawrence. 1997. Postfire responses of lotic ecosystems in Yellowstone National Park, U.S.A. Canadian Journal of Fisheries and Aquatic Sciences 54:2509–2525.

ODEQ (Oregon Department of Environmental Quality). 2019. Water quality standards: Beneficial uses, policies, and criteria for Oregon. Chapter 340 in Oregon State Archives. Oregon Department of Environmental Quality, Salem, Oregon. (Available from: https://secure.sos.state.or.us/oard/displayDivisionRules.action?selectedDivision=1458)

Pettit, N. E., and R. J. Naiman. 2007. Fire in the riparian zone: Characteristics and ecological consequences. Ecosystems 10: 673–687.

Reale, J. K., T. P. Archdeacon, D. J. Van Horn, E. J. Gonzales, R. K. Dudley, T. F. Turner, and C. N. Dahm. 2021. Differential effects of a catastrophic wildfire on downstream fish assemblages in an aridland river. Aquatic Ecology 55:483–500.

Reale, J. K., D. J. Van Horn, K. E. Condon, and C. N. Dahm. 2015. The effects of catastrophic wildfire on water quality along a river continuum. Fire Ecology 34:1426–1442.

Reid, M. A., M. C. Thoms, and F. J. Dyer. 2006. Effects of spatial and temporal variation in hydraulic conditions on metabolism in cobble biofilm communities in an Australian upland stream. Journal of the North American Benthological Society 25:756–767.

Rieman, B., and J. Clayton. 1997. Wildfire and native fish: Issues of forest health and conservation of sensitive species. Fisheries 22:6–15.

Rupasinghe, P. A., and P. Chow-Fraser. 2021. Relating pre-fire canopy species, fire season, and proximity to surface waters to burn severity of boreal wildfires in Alberta, Canada. Forest Ecology and Management 496:119386.

Schultz, L. D., M. P. Heck, D. Hockman-Wert, T. Allai, S. Wenger, N. A. Cook, and J. B. Dunham. 2017. Spatial and temporal variability in the effects of wildfire and drought on thermal habitat for a desert trout. Journal of Arid Environments 145:60–68.

Scordo, F., S. Chandra, E. Suenaga, S. J. Kelson, J. Culpepper, L. Scaff, F. Tromboni, T. J. Caldwell, C. Seitz, J. E. Fiorenza, C. E. Williamson, S. Sadro, K. C. Rose, and S. R. Poulson. 2021. Smoke from regional wildfires alters lake ecology. Scientific Reports 11:10922.

Sherson, L. R., D. J. Van Horn, J. D. Gomez-Velez, L. J. Crossey, and C. N. Dahm. 2015. Nutrient dynamics in an alpine headwater stream: Use of continuous water quality sensors to examine responses to wildfire and precipitation events. Hydrological Processes 29:3193–3207.

Spencer, C. N., and F. R. Hauer. 1991. Phosphorus and nitrogen dynamics in streams during a wildfire. Journal of the North American Benthological Society 10:24–30.

Swartz, A., D. Roon, M. Reiter, and D. Warren. 2020. Stream temperature responses to experimental riparian canopy gaps along forested headwaters in western Oregon. Forest Ecology and Management 474:118354.

WSDE (Washington State Department of Ecology). 2002. Evaluating criteria for the protection of freshwater aquatic life in Washington's surface water quality standards: Dissolved oxygen. Draft Discussion Paper and Literature Summary. Publication Number 00-1007. Watershed Management Unit, Washington State Department of Ecology, Water Quality Program, Olympia, Washington. (Available from: https://www.ezview.wa.gov/Portals/_1962/Documents/SSRSAG/Hicks%202002%20Edited.pdf)

Wilhelms, S. C., J. S. Gulliver, and K. Parkhill. 1993. Reaeration at low-head hydraulic structures. Technical Report No.W-93-2. Water Quality Research Program, United States Army Corps of Engineers, Waterway Experiment Station, Vicksburg, Mississippi. (Available from: https://apps.dtic.mil/sti/pdfs/ADA284089.pdf)