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Water Resources Research

REVIEW ARTICLE
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Special Section:
Fire in the Earth System

# Wildfire Induces Changes in Receiving Waters: A Review With Considerations for Water Quality Management

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**Abstract** Wildfires have increased in frequency in many ecosystems, with implications for human health and the environment, including water quality. Increased fire frequency and urbanization also raise the prospect of fires burning into urban areas, mobilizing pollutants few have considered to date. As a result, water quality managers lack information to anticipate, respond to and potentially mitigate wildfire impacts. Here, we reviewed the scientific literature to assess wildfire effects on response endpoints of a conceptual model linking fire to water quality, quantifying response directionality, magnitude and duration. Physically, water yield, sediments, and temperature all increased post-fire. Chemically, nutrients, ions, organic chemicals, and metals increased in burned watersheds, sometimes by orders of magnitude over pre-fire or reference conditions. In select cases, post-fire concentrations exceeded aquatic life criteria or drinking water standards, at times even in the finished drinking water. Biological assemblages commonly declined after post-fire runoff events. The duration of effects was less than 5 yr for most endpoints (e.g., metals) on average following fire, although effects did extend 15 yr or more in some individual cases. We found only a few studies on pollutants mobilized from wildfire impacted urban areas with benzene contamination in drinking water and high metal concentrations in ash prominent exceptions. Overall, this review provides a resource for understanding wildfire impacts on water quality endpoints, with the goal of informing the response of managers and other decision makers to this growing problem.

**Plain Language Summary** Wildfires are increasing in frequency due in part to climate change, and the pollutants mobilized during and after fire threaten water quality. Water quality managers should be aware of these threats so they can plan for and respond to them. We reviewed the scientific literature on wildfire effects on water quality. Streamflow typically increases following wildfire, as do sediments and temperature. Nutrients, ions, metals, and certain organic chemicals often increase as well, sometimes 10–100 times or more above reference levels. Some post-fire chemicals can exceed regulatory limits, even in finished drinking water (e.g., arsenic). Wildfire also commonly affects biological assemblages. Most impacts last less than 5 yr, although they can be much longer. Unfortunately, only a few studies measured pollutants from urban areas impacted by wildfire. This is a major knowledge gap since these events are likely becoming more common, can mobilize pollutants from built materials and may impact many water uses. Finally, our paper describes the implications for water quality programs, such as drinking water, water quality standards, assessment, restoration and non-point source programs. We hope water quality managers and communities can use this information to better respond to this growing problem.

**Key Points:**

- Post-fire increases in many water quality pollutants exceeded pre-fire levels, often by multiple orders of magnitude
- Nitrate, some metals, benzene, and disinfection by-products exceeded regulatory levels in treated drinking water following some wildfires
- Little is known about pollutants from fires burning infrastructure and materials (e.g., houses, cars) within the wildland urban interface

**Supporting Information:**

Supporting Information may be found in the online version of this article.

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## 1. Introduction

Wildfires in the United States (U.S.) have increased in frequency since the mid-1980s (Dennison et al., 2014; Miller et al., 2012; Singleton et al., 2019). This is most pronounced in the western U.S. (Dennison et al., 2014; Singleton et al., 2019) and increases have been observed for both forest and grassland wildfire (Westerling, 2016). Much of the increase in wildfire frequency and area burned has been correlated with drier conditions, earlier snowmelt (especially at mid elevations) and severe droughts associated with climate change (Abatzoglou & Williams, 2016; Dennison et al., 2014; Holden et al., 2018; Westerling, 2016). Greater fire activity is not limited to the U.S., with increases observed in regions and ecosystem types across the globe (e.g., Cochrane et al., 2003; Fairman et al., 2015; Seidl et al., 2014).

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Increases in fire frequency have profound implications for human health and environmental quality. These effects have been relatively well studied for air quality (e.g., USEPA, 2019a), yet fires pose potential risks to water resources as well, including those for drinking water supply, recreation and aquatic habitat. Wildfires alter stream flow response to precipitation (Gao et al., 2017; Hallema, Sun, Bladon, et al., 2017; Hallema, Sun, Caldwell, et al., 2017; Shakesby & Doerr, 2006), and mobilize a variety of pollutants through downwind deposition or subsequent runoff. These pollutants include sediment (Dahm et al., 2015; Smith et al., 2011; Verkaik et al., 2013), dissolved organic compounds (Hohner et al., 2016, 2019), and metals (Bladon et al., 2014; Smith et al., 2011) that threaten drinking water sources. Nutrients released following fires can contribute to downstream eutrophication and harmful algal blooms, adversely affecting drinking water quality and recreational uses, such as boating and swimming (Cook & Holland, 2012; Emelko et al., 2016). The same fire-mobilized pollutants impacting water quality can also be harmful to aquatic life, at least in the short-term (Bixby et al., 2015; Kelly et al., 2006).

When coupled with urbanization, increased fire frequency raises the prospect of direct damage to water infrastructure and mobilization of pollutants few have considered to date (e.g., benzene). In recent decades, urban development has expanded the area of the wildland urban interface (WUI, Martinuzzi et al., 2015). The WUI is the area where structures and other human development interface or intermix with undeveloped wildland, and where wildfires most impact people (Martinuzzi et al., 2015). In 2010, approximately 10% of U.S. land area, 34% of homes, 59% of secondary homes, and 32% of the U.S. population were within the WUI (Martinuzzi et al., 2015). Wildfires burning into urbanized areas within the WUI and damaging houses, vehicles and other human infrastructure have been tragically demonstrated in recent years (e.g., in Paradise, CA and Boulder County, CO), but water quality effects of such fires remain largely unknown.

Here, we review the effects of wildland fire and destructive, urban fires in the WUI on water quality stressors important to many water quality management programs. While wildfire impacts have been reviewed before, most reviews have focused on water quality in general or primarily on drinking water (e.g., Smith et al., 2011). In this paper, we take a broader perspective on water quality management to include many endpoints water quality regulators focus on, not only drinking water. For instance, under the Clean Water Act (CWA) in the US, the Environmental Protection Agency (EPA) approves water quality standards (WQS) promulgated by states and authorized tribes. These WQS are a combination of designated uses for waterbodies, such as aquatic life habitat and recreation, and the necessary water quality criteria to protect those designated uses. Wildfire can impact many designated uses and criteria pollutant concentrations. Though we primarily discuss endpoints of concern for U.S. federal and state regulatory programs, other countries have similar regulatory endpoints and programs. Water quality managers beyond regulators are likely to benefit from this broader management perspective, including drinking water facility operators, soil and water conservationists, watershed stakeholder groups, and land managers—namely, anyone who is likely to face the increasing impacts of fire on water quality.

We anticipate this review will help water quality managers and decision makers better understand the likely impacts of fire, and ultimately inform the design of mitigation and recovery strategies and policies. For researchers, the review highlights the state of current knowledge, while identifying key knowledge gaps for future work.

## 2. Methods

### 2.1. Review

To guide our review, we created a conceptual model of the myriad ways fire can affect endpoints of interest to water quality management programs (Figure 1 provides an overview, while a more detailed figure is provided in the Supporting Information S2). The model was based on existing knowledge and iterated with U.S. EPA experts across a range of water programs whose feedback was used to revise and improve the model (Figure 1). The management goals at the bottom are represented by designated uses, including the common aquatic life, recreation, and drinking water uses. These uses are an important component of WQS, which are the foundation of CWA water quality programs including wastewater permitting, assessment, total maximum daily loads (TMDLs), and non-point source control—all of which are designed to protect and restore designated uses. Concomitantly, drinking water standards enforced under the Safe Drinking Water Act (SDWA) are used by water quality programs to protect drinking water.

Based on the conceptual model, we conducted a literature search to identify relevant peer-reviewed materials published after 1980. Among other techniques, we used the SWIFT-Active Screener web application to screen



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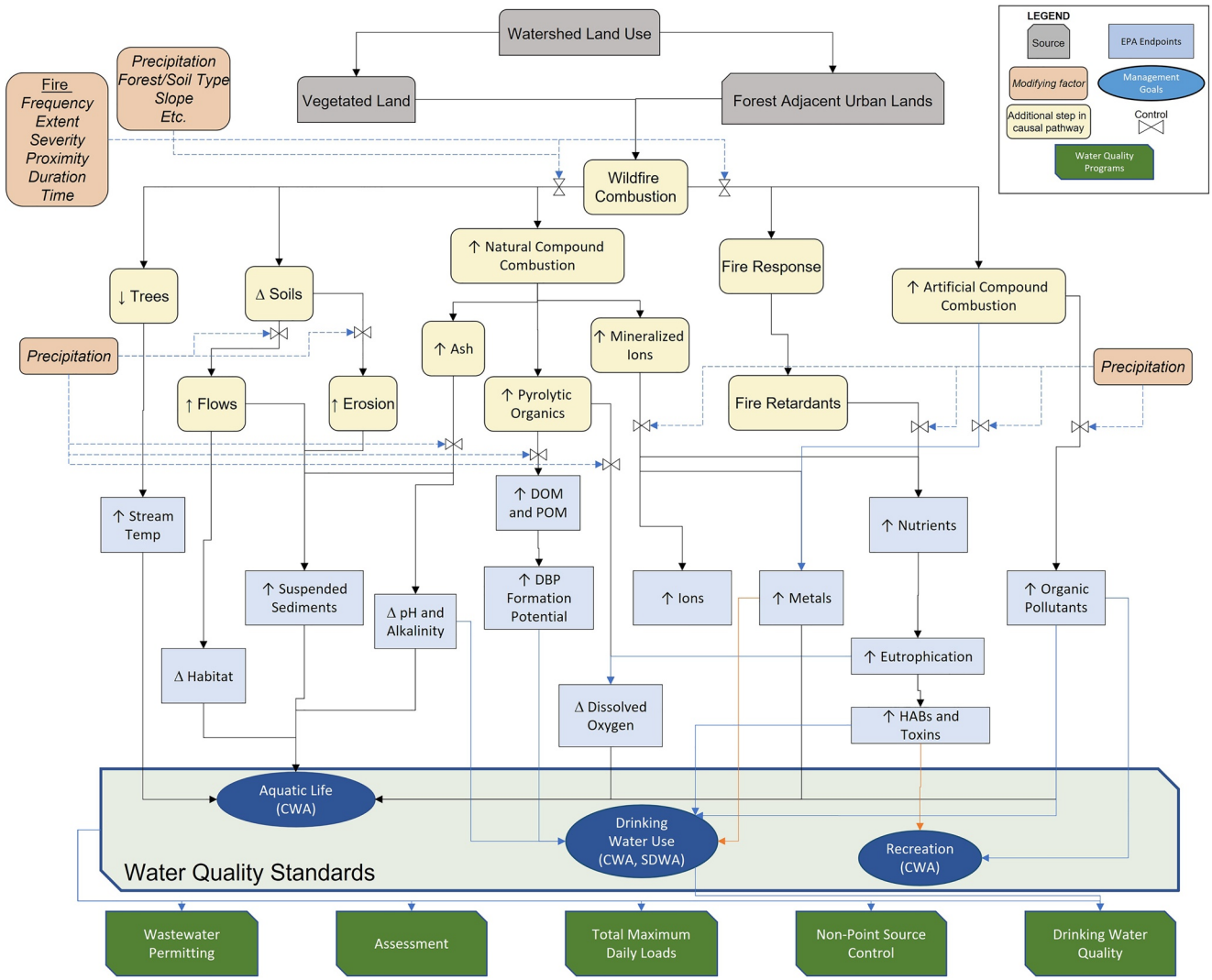



**Figure 1.** Simple conceptual model of wildland fire effects on water quality endpoints, management goals and water quality programs in wildland and urbanized landscapes. Different colored arrows indicate pathways that cross but do not intersect (i.e., join) and dashed lines emphasize the regulating effect of modifying factors such as fire, climate, and landscape characteristics on pollutant export. BOD, biological oxygen demand; DOM and POM, dissolved and particulate organic matter; DBP, disinfection byproducts; HABs, harmful algal blooms; CWA, Clean Water Act; SDWA, Safe Drinking Water Act. A more detailed conceptual model is provided in Supplemental Information.

the initial batch of literature (Howard et al., 2020). SWIFT uses expert decisions to train a machine learning algorithm, automating the reduction of documents to a representative subset. From this subset, we manually identified papers specifically applicable to our conceptual model using EndNote software, for a total of 184 papers (see Supporting Information S1 for more methodological details).

### 2.2. Analysis

Using these papers, we created an extraction database, recording information for each paper (e.g., title, waterbody type studied, fire type, etc.) and the response of the variables measured following fire. We particularly focused on three items for each post-fire response: the *directionality*, *magnitude*, and *duration*. *Directionality* was whether a particular variable increased, decreased or showed no change in response to fire. *Magnitude* was the strength of the response measured as a percent change from pre- or reference watershed conditions, and *duration* was the length of time of the response reported by the study. We recorded instances in the database where the duration



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of the response was limited by study length. There was often more than one response or response variable (e.g., nitrogen and phosphorus) per paper, and each received a separate entry in the database.

Estimates of direction and magnitude were based on what authors reported and the design of their work. Sample timing is important and studies differed in when they first sampled. For example, those missing the first flush events may underestimate the largest magnitudes and bias their averages. We constrained ourselves to what authors reported in terms of magnitude from their designs, recording each value when multiple were reported. Similarly, our duration estimates were constrained by the length of the studies and what they reported. Most studies were of limited duration and did not follow the trajectory of response trends until they returned to reference conditions. As a result, our duration estimates are likely conservative.

From the extraction database, we summarized our findings for this review. For directionality, we calculated the percentage of observations for each response showing an increase, decrease or no-change (e.g., 61% of streamflow/water export measurements following fire reported an increase). We calculated the magnitude of the response as a percent change compared to any unburned or pre-fire reference site(s) used in each paper, and reported the median and full range of values across all observations. Likewise, we calculated the duration for each variable across all observations, and reported the median and full range. We generally referred to short-term durations as those lasting <5 yr, medium duration as 5–10 yr and long-term as >10 yr. This was derived from the distribution of observed responses, but also aligns with assessment program periods, often on 5 yr cycles. All papers reviewed and extracted are provided in the reference section, even those not cited directly in the text but used in the quantitative estimates. The data are included in the extraction database provided as Supporting Information S3.

## 3. Results

When considering the results below, it is important to acknowledge that many factors modify the relationship between wildfire and water quality responses (Figure 1), contributing in part to the high variation observed. Critical modifying factors include: characteristics of the fire, such as severity, extent, and proximity, and post-fire hydrologic responses, including timing and magnitude of precipitation. Most chemical concentrations in water, for example, increase with greater fire severity (Rhoades et al., 2011, 2019; Rust et al., 2019). Notable exceptions include organic carbon and nitrogen, however, which can volatilize at higher temperatures (>450°C) (Hohner et al., 2019). Thus, organic carbon and nitrogen responses can exhibit unimodal responses to severity (Chow et al., 2019; Hohner et al., 2019). Likewise, the extent of the fire in the watershed can also impact concentrations observed, though this is often underreported (Rhoades et al., 2019). Further, water quality responses are driven by hydrology—there is the potential for aerial deposition, but the pollutants generated by wildfire are most often moved by water. As a result, precipitation following fires and the hydrologic regime of the watersheds are primary modifying factors (Moody et al., 2013). Indeed, many responses were not observed until the first major runoff event post-fire (e.g., Rinne, 1996).

Below, we do not dissect the individual context of each reported study, but rather focus on general trends and synthesis important for the water regulatory linkages, while also noting modifying factors where most relevant. The results sections are organized by physical, chemical, and biological response variables. Each subsection of the results starts with the introduction of fire effects, describes the results, and then ends with further context and interpretation.

### 3.1. Physical Responses

#### 3.1.1. Hydrologic Characteristics

Fire impacts many characteristics of the hydrologic regime, important to all subsequent water quality responses since water transports pollutants and affects their concentrations (Bladon et al., 2014; Chanasyk et al., 2003; Moody et al., 2013; Rhoades et al., 2011). Wildfire often kills and partially combusts vegetation and can alter soil hydrophobicity, depending upon the vegetation burned and fire severity, among other factors (Chanasyk et al., 2003; Murphy et al., 2018; Shakesby & Doerr, 2006). Reductions in the interception and infiltration of precipitation in burnt watersheds can result in greater overland flow. This is exacerbated by increased surface velocities and connectivity of flow paths due to bare ground (Moody et al., 2013). These can result in higher peak

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flows and shorter response times between precipitation and discharge (Shakesby & Doerr, 2006), hydrologic dynamics similar to watersheds with more impervious cover (e.g., paved roads, parking lots).

Of the studies we surveyed reporting hydrologic characteristics, 66% found that wildfires generally increased surface runoff, streamflow or water export, and peak flows, 10% reported reductions (in infiltration only), and 24% reported no change in streamflow (Table 1). The increase in post-fire surface runoff was generally more than a 100% to nearly 10,000% compared to before or reference conditions (Figure 2). The magnitude of streamflow and water export increases ranged from no change to nearly 10,000% increases (Figure 2), with a median increase of approximately 60% increase. Conversely, the magnitude of infiltration response was an approximate 40% decline (Figure 2). Hydrologic responses usually lasted until vegetation re-established. The median duration of surface runoff and stream flow response was 5 yr while that for infiltration response was 15 yr, although the sample size was limited ($N = 3$, Figure 3).

Compared to streamflow or peak flows, the effect of fire on groundwater and baseflow measurements remain relatively understudied (Hallema, Sun, Bladon, et al., 2017; Moody et al., 2013). Increased overland flow and decreased infiltration are expected to reduce groundwater recharge and lead to lower baseflows (Morales et al., 2013; Murphy et al., 2018; Parlak et al., 2015) and water quality models predict as much (Rodrigues et al., 2019). Long-term, however, the opposite may also occur: if infiltration recovers, yet vegetation is slow to respond or the recovered vegetation has lower rates of evapotranspiration, recharge may increase and baseflows may not decline as much or may increase (Bart & Tague, 2017; Poon & Kinoshita, 2018).

#### 3.1.2. Suspended and Bedded Sediment (Including Debris Flows and Ash) and Stream Channel Morphology

Given the increase in overland flow, one of the more commonly reported water quality risks after fires is increased erosion, including debris flows, and suspended solid concentrations, especially where post-fire rainfall intensities are high and the hydrogeomorphic regime (sensu Moody et al., 2013) consists of more erodible, high slope conditions (Bodí et al., 2014; Cannon & Reneau, 2000; Emelko et al., 2011; Nyman et al., 2015). Indeed, this effect is so common, the United States Geological Survey (USGS) has developed several tools to predict debris flows following fires (e.g., USGS, 2020a, 2020b), and national Burned Area Emergency Response programs have a significant focus on erosion control and debris flow protection (NIFC, 2020; USGS, 2020c). Elevated suspended solids, including initial pulses of ash from storms following wildfire, can impact water treatment, threatening attainment of drinking water standards for turbidity and increasing filtration and coagulation costs (Emelko et al., 2011; Murphy et al., 2015). Sediments can overwhelm these systems, requiring closure of water intakes in some cases (e.g., following the 2012 High Park wildfire in Colorado, Hohner et al., 2016), and sediments and ash often contain contaminants, including organic pollutants and metals (e.g., Burton et al., 2016).

Of the measurements of suspended solids or turbidity across the studies reviewed, 91% observed an increase in suspended sediments, 9% observed no change and none observed a decrease (Table 1). In magnitude, values ranged from little change to more than a 25,000 fold increase compared to control or prior conditions (Figure 2), with the median of study responses near 1,000 fold. In Arizona, for example, wildfire led to increased total suspended solids in the 1,000s of mg/L, orders of magnitude above background concentrations (Rinne, 1996), and the ash flows measured following wildfire in New Mexico led to turbidities in excess of 1,500 nephelometric turbidity units (Dahm et al., 2015). In duration, the median was less than 5 yr of elevated sediment and in some cases only minor increases in suspended solids occurred and declined quickly (Spencer & Hauer, 1991). Severely burned catchments can take longer to recover (Rhoades et al., 2019) and the longest measured duration was up to 14 yr (Figure 3).

Bedload sediment is presumed to increase following fires, given the presence of erosion and debris flows, but we identified only two studies measuring bedload. One of those, a 3 yr study of sediment from a burned watershed in Wyoming, found heightened suspended loads during post-fire storms, but very little change in bedload from the watershed outlet (Ryan & Dixon, 2008). Bedload response is a longer-term process as heavier sediment requires substantial time to move through the drainage network; it may be that longer duration studies (or space for time substitution studies) are needed.

Changes in the balance of sediment transport capacity (i.e., flow) and sediment supply influence channel form. Not surprisingly, since both drivers are affected by wildfire, channel morphology is often altered following fires, but this process can take years to manifest (Minshall et al., 1997; Shakesby & Doerr, 2006). A study of stream

**Table 1**
*The Number of Studies Reporting Increases, Decreases, or No Change in Conceptual Model Endpoints After Wildfire, With Percentages Given in Parantheses*

| Response endpoints | Increase | Decrease | No change | Total |
|---|---|---|---|---|
| Physical responses | | | | |
| Hydrologic characteristics | 19 (66%) | 3 (10%) | 7 (24%) | 29 |
| Infiltration | | 3 (100%) | | 3 |
| Surface runoff | 6 (100%) | | | 6 |
| Streamflow/water export | 11 (61%) | | 7 (39%) | 18 |
| Peak flows | 2 (100%) | | | 2 |
| Suspended sediment[a] | 31 (91%) | 0 (0%) | 3 (9%) | 34 |
| Temperature | 11 (79%) | 0 (0%) | 3 (21%) | 14 |
| Chemical responses | | | | |
| Nitrogen[a] | 64 (77%) | 6 (7%) | 13 (16%) | 83 |
| Phosphorus | 33 (65%) | 14 (27%) | 4 (8%) | 51 |
| Calcium and potassium | 17 (77%) | 4 (18%) | 1 (5%) | 22 |
| Conductivity | 11 (69%) | 1 (6%) | 4 (25%) | 16 |
| pH[b] | 10 (45%) | 5 (23%) | 7 (32%) | 22 |
| Alkalinity/dissolved inorganic carbon | 3 (33%) | 5 (56%) | 1 (11%) | 9 |
| Organic carbon | 24 (48%) | 13 (26%) | 13 (26%) | 50 |
| Dissolved oxygen | 2 (22%) | 4 (44%) | 3 (33%) | 9 |
| Disinfection byproduct formation potential[a] | 4 (21%) | 4 (21%) | 11 (58%) | 19 |
| Metals[c] | 53 (57%) | 9 (10%) | 31 (33%) | 93 |
| Aluminum[b] | 4 (66%) | 1 (17%) | 1 (17%) | 6 |
| Arsenic[a] | 4 (100%) | 0 (0%) | 0 (0%) | 4 |
| Cadmium[a] | 1 (33%) | 0 (0%) | 2 (67%) | 3 |
| Chromium[a] | 1 (100%) | 0 (0%) | 0 (0%) | 1 |
| Cobalt | 0 (0%) | 0 (0%) | 2 (100%) | 2 |
| Copper[a,b] | 5 (62%) | 1 (13%) | 2 (25%) | 8 |
| Iron[b] | 5 (83%) | 0 (0%) | 1 (17%) | 6 |
| Lead[a] | 6 (60%) | 1 (10%) | 3 (30%) | 10 |
| Mercury[a] | 10 (40%) | 6 (24%) | 9 (36%) | 25 |
| Nickel | 3 (60%) | 0 (0%) | 2 (40%) | 5 |
| Selenium[a] | 2 (100%) | 0 (0%) | 0 (0%) | 2 |
| Vanadium | 1 (50%) | 0 (0%) | 1 (50%) | 2 |
| Zinc[b] | 4 (67%) | 0 (0%) | 2 (33%) | 6 |
| Polycyclic aromatic hydrocarbons[a, c] | 24 (71%) | 0 (0%) | 10 (29%) | 34 |
| Acenaphthene | 1 (100%) | 0 (0%) | 0 (0%) | 1 |
| Acenaphthylene | 1 (100%) | 0 (0%) | 0 (0%) | 1 |
| Anthracene | 1 (100%) | 0 (0%) | 0 (0%) | 1 |
| Benzanthracene | 0 (0%) | 0 (0%) | 1 (100%) | 1 |
| Benzofluoranthene | 2 (100%) | 0 (0%) | 0 (0%) | 2 |
| Benzopyrene[a] | 0 (0%) | 0 (0%) | 1 (100%) | 1 |
| Chrysene | 1 (100%) | 0 (0%) | 0 (0%) | 1 |
| Dibenz-anthracene | 1 (100%) | 0 (0%) | 0 (0%) | 1 |
| Fluoranthene | 1 (100%) | 0 (0%) | 0 (0%) | 1 |
| Indenopyrene | 1 (100%) | 0 (0%) | 0 (0%) | 1 |



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**Table 1**
*Continued*

| Response endpoints | Increase | Decrease | No change | Total |
|---|---|---|---|---|
| Naphthalene | 3 (75%) | 0 (0%) | 1 (25%) | 4 |
| Phenanthrene | 4 (100%) | 0 (0%) | 0 (0%) | 4 |
| Pyrene | 1 (100%) | 0 (0%) | 0 (0%) | 1 |
| Biological responses[d] | | | | |
| Macroinvertebrates | 6 (20%) | 17 (57%) | 7 (23%) | 30 |
| Fish | 2 (7%) | 17 (63%) | 8 (30%) | 27 |
| Algae | 7 (28%) | 6 (24%) | 12 (48%) | 25 |

*Note*. Single studies often reported on multiple endpoints, and therefore the overall number in the right-hand total column (578) is higher than the number of reviewed studies (184).

[a]Primary drinking water regulated contaminant. [b]Secondary drinking water regulated contaminant. [c]The category of total metals and polycyclic aromatic hydrocarbons (PAHs) are noted. Listed under each category are the directional changes for specific metals and PAHs. Some papers indicated changes in metals and PAHs generally without detailing which metal or PAH. Thus, the directional changes for individual metals and PAHs will not sum to the total for these water quality responses. [d]Biological responses are primarily changes in diversity/richness, although a few include changes in abundance, especially for fish.

channels in Yellowstone National Park, for example, found that stream power, channel dimensions, and bank failure rates were still adjusting 12–13 yr after wildfires in 1988 (Legleiter et al., 2003). As expected, fires resulted in higher stream power, increasing with percent burn area, and led to substantial increases in channel dimension and bank failure. In Mediterranean climate regions (e.g., California), morphological recovery may be comparatively faster than in temperate areas due, in part, to earlier native vegetation recovery which reduces erosion rates (Verkaik et al., 2013).

### 3.1.3. Light Availability and Water Temperatures

Light availability and temperature are important physical drivers for aquatic life. Growth of algae and vascular plants is often limited by available light, and aquatic organisms are mostly cold-blooded and their distributions are limited by temperature. Many U.S. states have regulatory targets for temperature based both on aquatic life physiological requirements (e.g., those of cold water fishes), as well as the effect of temperature on other pollutants. Not surprisingly, in-stream light availability tends to increase following wildfires, depending on the amount and severity of riparian vegetation burned (Bixby et al., 2015), and can be even more elevated where debris flows occur, opening up more of the canopy to light (Dunham et al., 2007). The consequences of these light increases may be offset by increases in turbidity, but these are short-lived and the loss of vegetation typically has a greater impact (Beakes et al., 2014). The result of increased direct radiation to streams is increased annual and seasonal stream temperatures, often lasting until riparian vegetation reestablishment (Minshall et al., 1997; Rhoades et al., 2011; Wagner et al., 2014).

Of the studies measuring temperature in receiving waterbodies following wildfire, 79% reported increases and 21% reported no change—no studies reported a decrease (Table 1). The magnitude of temperature responses was, on average, approximately a doubling, but ranged from 0% to a 400% increase (Figure 2). In degrees, this was an average increase of 7.9°C maximum temperature (range: 1–18) and 2.6°C mean temperature (range: 1.5–5). These effects lasted as long as 11 yr (Dunham et al., 2007) with a median across studies between 5 and 10 yr (Figure 3).

The studies surveyed almost exclusively focused on streams. Shifts in thermal regimes have also been observed in wetlands (Gu et al., 2008) and lakes in burned watersheds, but both of these waterbody types have been understudied (McCullough et al., 2019). This is an important knowledge gap given the importance of temperature for a wide range of lake processes (e.g., dissolved oxygen concentration, nutrient cycling, fish energetics).

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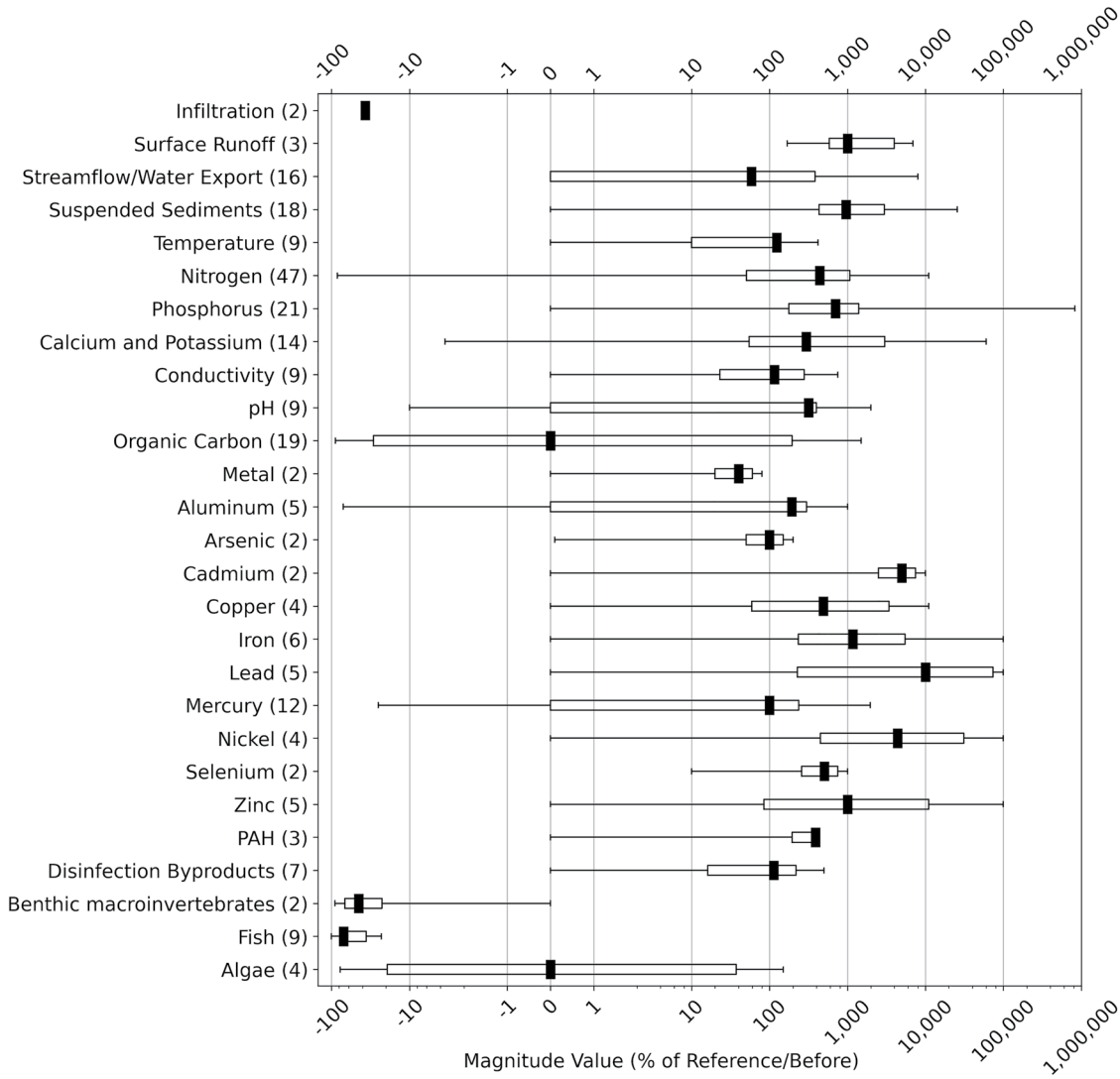



**Figure 2.** Magnitude (percent change compared to reference or before fire condition) of observed water quality change after wildfire. The value in parentheses after the variable is the number of observations from the literature. Shown are minimum, maximum (whiskers), quartiles (boxes) and median (hash) values across studies. *Disinfection byproducts are the formation potential from precursors.

### 3.2. Chemical Responses

#### 3.2.1. Nutrients

Nutrients are one of the more commonly studied chemical responses to wildfire. Both nitrogen and phosphorus generally increase in streams post-fire, and this has been observed in temperate, Mediterranean, and boreal forests as well as grasslands and wetlands (Chanasyk et al., 2003; Emelko et al., 2016; Hohner et al., 2016, 2019; Knoepp & Swank, 1993; Minshall et al., 1997; Ranalli, 2004; Rust et al., 2018; Sherson et al., 2015; Smith et al., 2011). Elevated nutrient inputs can lead to enhanced algal growth and harmful algal blooms, with the potential to negatively affect aquatic life, recreation, and drinking water (see Section 3.3.4). Because of this and other effects, many U.S. states are developing numeric nutrient thresholds to protect against the impacts of eutrophication (USEPA, 2022). Moreover, there is a primary drinking water standard for nitrate.

Across the measurements within the studies reviewed, 77% found increases in nitrogen, 7% decreases, and 16% no changes; 65% found increases in phosphorus, 27% decreases, and 8% no change (Table 1). Response magnitudes were generally greater for phosphorus than nitrogen (Figure 2). Nutrient responses were typically rapid, observed in streams during and especially following storms immediately after fires (Harris et al., 2015; Hauer &

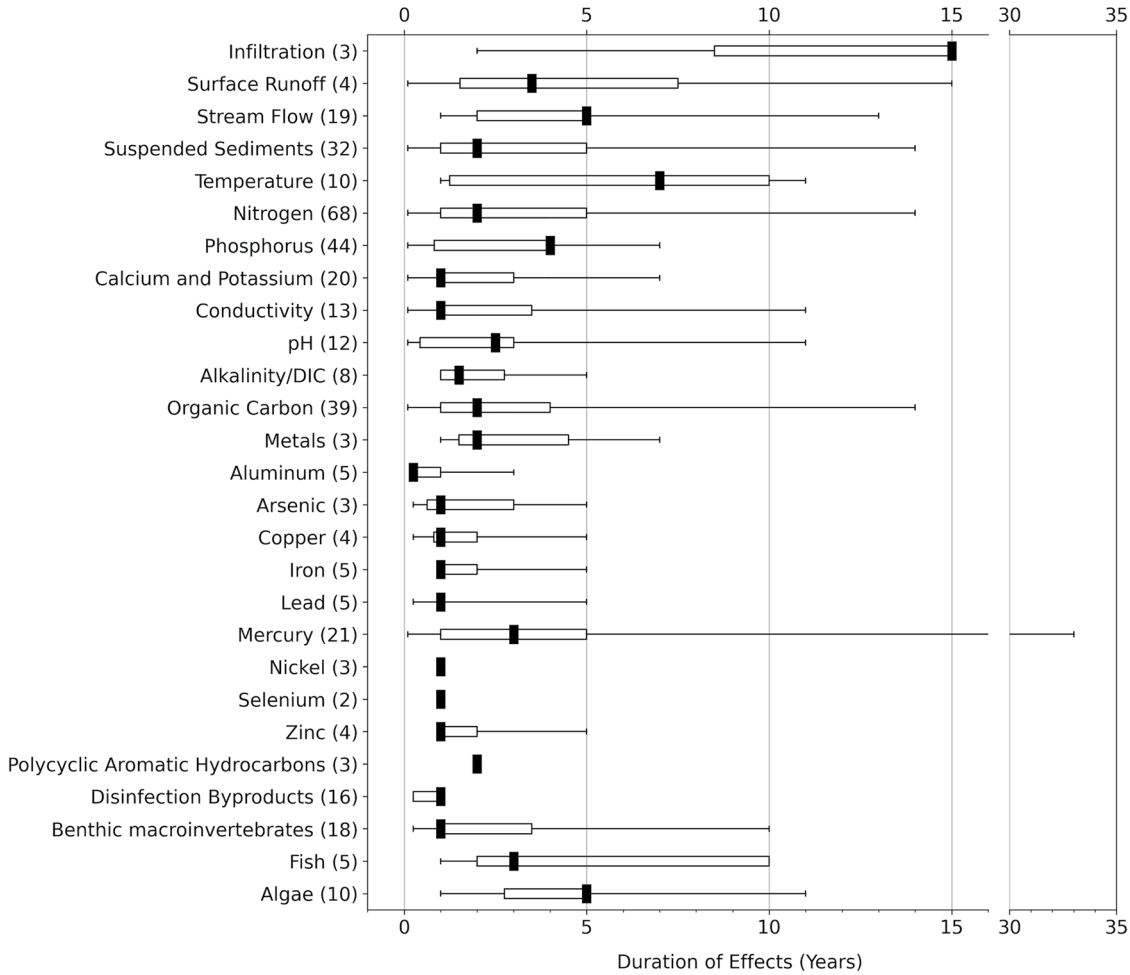



**Figure 3.** Duration (years) of observed water quality responses after wildfire by endpoint as reported in studies. Figure elements as in Figure 2.

Spencer, 1998; Spencer & Hauer, 1991), but delayed responses have been observed in more arid environments (Murphy et al., 2015; Rhoades et al., 2011). Elevated nutrient concentrations can persist, but usually return to reference conditions within 5 yr (median); with phosphorus levels taking marginally longer than nitrogen to return to pre-fire conditions on average (Figure 3). However, the maximum reported duration was for nitrogen (Figure 3), with severe fires in Colorado resulting in elevated nitrate/total dissolved nitrogen concentrations 14 yr after the fire (Rhoades et al., 2019). This was attributed to the slow rate of upland and riparian recovery post-fire at this particular site due to the severity of the fire and increased drought (Rhoades et al., 2019). Nitrogen and phosphorus increases have also been reported for lakes in post-fire watersheds (McCullough et al., 2019).

Proposed mechanisms explaining increased responses of nutrients post-fire include volatilization and deposition, reduced plant and microbial uptake, and increased leaching and runoff into waterbodies (Emelko et al., 2016; Rust et al., 2018; Smith et al., 2011). Of total nitrogen, nitrate from nitrification of ammonium tends to dominate (Ranalli, 2004). A significant increase in public water nitrate violations in the U.S. has been associated with wildfire and should be a concern for water suppliers (Pennino et al., 2022). In contrast, phosphorus is generally not lost through volatilization (Certini et al., 2005), and total phosphorus composition is dominated by soluble reactive phosphorus from the dissolution of ash (Ranalli, 2004).

### 3.2.2. Calcium, Potassium, and Other Ions

Cations contribute to water hardness, which can influence the toxicity of certain metals to aquatic life (e.g., cadmium, chromium III, copper, lead, nickel, and zinc), and to total dissolved solids, for which there is a secondary drinking water standard. Cations like calcium, potassium, and magnesium generally follow nutrient trends,



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increasing after post-fire precipitation events as a function of burned area and burn severity (Bitner et al., 2001; Rust et al., 2018; Smith et al., 2011) as they are primarily generated from ash (Ranalli, 2004). Such responses are consistent with the ash-alkaline hypothesis reviewed in Bayley and Schindler (1991), whereby alkaline fire ash, high in base cations like potassium and calcium, generally leads to an increase in calcium carbonate alkalinity and other associated base cation complexes.

For calcium and potassium, cations with the most reported responses, 77% of studies reported increases, 18% decreases and 5% no trend (Table 1). The magnitude of values included no increases to very large increases (up to 60,000% higher), but the central tendency was an approximately 300% increase from reference conditions (Figure 2). Concentrations of these cations typically remained elevated for less than 5 yr (Figure 3). For example, fires in the Sierra Nevada led to higher ion concentrations in streams for less than a year (Beche et al., 2005).

For cations generally, this pattern of a general increase occurred across biomes (Bayley et al., 1992; Beche et al., 2005; Betts & Jones, 2009; Bitner et al., 2001; Burke et al., 2013; Murphy et al., 2018), but not in all cases. Boreal streams in Alaska, for instance, exhibited a calcium decline downstream of a fire, even though potassium increased (Betts & Jones, 2009). Further, stream reaches draining a burned urbanized area in the WUI in California exhibited calcium and potassium increases, but not sodium (Burke et al., 2013). Such variability in responses was attributed to geology, soils, and changes in subsurface flow (and thus dissolution) affected by soil hydrophobicity and decreased infiltration. In Colorado, an increase in surface flow was associated with stream calcium and potassium increases, but not sodium and silica, since they are derived more from subsurface flow paths in that system (Murphy et al., 2018). Yellowstone Lake exhibited only minor changes in ion concentrations after the 1988 fire, which was attributed to its large volume relative to the area of watershed burned (~25%, Lathrop, 1994).

Sulfate and chloride were often elevated in streams draining burned watersheds—sulfate more than chloride—but the sample size of studies for either was relatively small. Increases for both were relatively short-lived and did not exceed World Health Organization water quality aesthetic use guidelines for sulfate (Beche et al., 2005; Smith et al., 2011). Elevated sulfate concentrations were most often associated with responses in boreal/peatland region lakes (Bayley et al., 1992; Brown et al., 2013; Crilly et al., 2017; Lydersen et al., 2014), but elevated sulfate has also been observed in temperate streams (Beche et al., 2005). Decreases in sulfate were observed in Yellowstone Lake following the 1988 fires (Lathrop, 1994).

#### 3.2.3. Conductivity

Conductivity, the ability of liquid to conduct electricity, and total dissolved solids, the mass of dissolved constituents in water, are correlated measures of the dissolved ion content of water. Increased conductivity can stress biota and require additional treatment to remove. Moreover, as mentioned above, there is a secondary drinking water standard in the U.S. for total dissolved solids (500 mg/L). Like nutrients and cations, conductivity also increased after fires in 69% of observations, decreased in 6% and showed no trend in 25% (Table 1). This pattern generally followed burn severity due to greater ash and mineralization at higher temperatures (Rhoades et al., 2011). Not surprisingly, observed magnitudes followed those for cations, with a median increase of just over 100% (Figure 2). Most conductivity responses tended to be immediate and lasted less than 5 yr, even for severe fires (Figure 3), but they were also dynamic. Stream conductivity response in New Mexico, for example, exhibited large swings over space and time, extending from headwaters to far downstream following precipitation events (Dahm et al., 2015; Sherson et al., 2015). We encountered one study of conductivity in lakes which exhibited a decline post-fire (Allen et al., 2003). Conductivity responses will likely be muted where lake volumes are large (Lathrop, 1994).

#### 3.2.4. pH/Alkalinity

The pH of water directly affects aquatic life, but also indirectly affects it through its influence on mobilization and toxicity of pollutants. It is also important to drinking water treatment facilities, which have a voluntary secondary pH standard to protect taste, treatment process, and infrastructure. pH changes are affected by the acid-neutralizing capacity (often measured as alkalinity), or buffering capacity, of recipient waterbodies. The ash-alkaline hypothesis suggests alkaline ash input leads to an increase in alkalinity and stream pH following fire (reviewed in Bayley & Schindler, 1991). The literature we reviewed, however, ranged from approximately half reporting pH or alkalinity increases in boreal lakes, everglade wetlands, temperate and Mediterranean streams (e.g., Costa et al., 2014; Korsman & Segerstroem, 1998; Son et al., 2015) and half reporting decreases or no

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change in boreal streams and lakes (especially those with excessive sulfur from atmospheric deposition or peatlands) and Mediterranean streams (Table 1). In magnitude, the central tendency of changes in hydrogen ion concentrations was roughly a tripling (300% or 0.5 pH units, Figure 2), but higher increases have been observed. Only one study reported a magnitude for an alkalinity response (a decrease of 50% in an acidic Norwegian lake, Lydersen et al., 2014). In duration, studies report that pH and alkalinity responses were relatively short-lived (median <5 yr, Figure 3), likely linked to short-lived ash exports (Bodi et al., 2014), and rarely violated aquatic life and secondary drinking water standards.

The diverse response in post-fire pH and alkalinity is a function of the buffering capacity of a given waterbody and the acidity or alkalinity of the post-fire inputs. One study of a well buffered stream, for example, in southern California observed no change in pH following fires (Burton et al., 2016). In cases where the buffering capacity was low, decreases in pH observed in the literature were attributed to acidity from export of reduced sulfur and subsequent oxidation (Bayley et al., 1992), increased nitrate (Crilly et al., 2017), application of phosphorus-based fire retardant (Angeler et al., 2004, 2005, 2006) and increased DOC that included organic acids (Bladon et al., 2014). Studies of alkalinity from boreal regions (with peatlands) generally exhibited alkalinity decreases in receiving streams and lakes, and this has been attributed both to peatland release of hydrogen ions as well as oxidation of reduced sulfur (Allen et al., 2003; Bayley et al., 1992; Crilly et al., 2017; Lydersen et al., 2014). By contrast, studies in more temperate, non-peatland areas away from acid deposition reported alkalinity increases, including in two streams draining burned watersheds in Colorado (Murphy et al., 2018; Rhoades et al., 2011). Across both biomes, however, the timing of the response was generally a large change in years 1 and 2, followed by a return to pre-fire or unburned reference levels within 4–5 yr (Lydersen et al., 2014; Rhoades et al., 2011). Studies from lakes were similarly diverse in pH response (McCullough et al., 2019).

### 3.2.5. Organic Carbon and Organic Nitrogen

Organic carbon and nitrogen, especially DOC and DON, are important considerations in drinking water because of their potential to contribute to disinfection byproduct (DBP) formation and metal binding (Regan et al., 2017; Westerhoff & Mash, 2002). Particulate organic carbon can similarly affect metal toxicity. Post-fire nutrient runoff resulting in eutrophication and excess algal growth can also produce organic compounds, causing taste and odor issues downstream of burned sites (Bladon et al., 2014; Hohner et al., 2019).

Slightly less than 50% of studies reporting on this endpoint recorded increases in organic carbon with fires, 26% decreases and the remaining 26% no change (Table 1). The magnitude of organic carbon released during fires ranged from slightly negative values to increases above 1,400%; the median was just slightly positive (Figure 2). The median duration of organic carbon responses was less than 5 yr, but has been reported for as long as 10–14 yr (Rhoades et al., 2019, Figure 3).

Increases in organic carbon have been observed in temperate (Emelko et al., 2011; Murphy et al., 2018; Writer et al., 2014) and boreal regions (Allen et al., 2003, 2005; Crilly et al., 2017). The magnitude of response appears to be fire temperature dependent, according to laboratory heating experiments with water extractable organic carbon and nitrogen exhibiting highest concentrations at medium temperatures (≤350°C; Hohner et al., 2019). By contrast, higher temperatures (≥450°C) volatilized most carbon and nitrogen, suggesting high-intensity fires may result in lower organic carbon and nitrogen export to downstream waterbodies (Hohner et al., 2016, 2019). At least some organic carbon increases in aquatic ecosystems below burned watersheds, especially lakes, are likely attributable to algae. Algal biomass in these downstream lakes, especially oligotrophic lakes, often increased in response to dissolved nutrient export from burned watersheds and contributed organic carbon through exudation or lysis (Allen et al., 2003, 2005; McCullough et al., 2019).

### 3.2.6. Dissolved Oxygen

Oxygen is critical for aquatic life and biogeochemical processes, and there are water quality criteria in the U.S. for a variety of dissolved oxygen (DO) endpoints related to life stage, thermal regime, waterbody type, etc. (USEPA, 2020c). In general, we found a limited number of studies reporting DO concentrations after wildfire. Only two observations recorded increases and four recorded decreases after the wildfire. Only one recorded magnitude (−55%), while for the duration, seven observations were reported with a maximum of 2 yr and a median of 1 yr. This lack of information is a surprising result given the importance of DO to aquatic biota and many biogeochemical processes.

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Given that organic matter drives respiration in receiving waters, large inputs of organic matter (including ash) post-fire likely lead to DO deficits or sags in receiving waters (e.g., Allan & Castillo, 2007). Indeed, in New Mexico, a fire led to large DO declines along streams in the Gila River watershed from the headwaters to mainstem due to ash inputs (Dahm et al., 2015; Reale, 2015; Sherson et al., 2015). Dissolved oxygen dynamics returned to pre-fire or unburned reference levels within 2 yr. Lentic waterbodies would be expected to exhibit even greater effects given their higher residence time and slower reaeration (McCullough et al., 2019). A study of temporary ponds in Spain found that DO decreased following fires and lasted for the duration of the 1 yr study (Cunillera-Montcusi et al., 2019).

#### 3.2.7. Metals and Other Inorganic Chemicals With Water Quality and Drinking Water Standards

Wildfires can mobilize metals and other inorganic chemicals which are then transported by precipitation events into downstream waters (e.g., see Abraham et al., 2017b; Rust et al., 2018; Smith et al., 2011). Many of these chemicals have CWA or SDWA standards, or are listed as Contaminant Candidate List chemicals. The latter are contaminants not currently subject to any proposed or promulgated national primary drinking water regulations in the U.S., but are known or anticipated to occur in public water systems (PWS; USEPA, 2021). Fire mobilized metals can also impact aquatic organisms, with potentially lethal and sub-lethal effects including the bioaccumulation of metals in tissue and decreased mobility (Silva et al., 2015, 2016).

Of the studies reviewed, 57% measured increases in metals, 10% decreases, and 33% no changes following fire (Table 1). Increases in receiving stream metal concentrations from wildfire were striking, spanning from little or no increase to over 100,000% (1,000x) of unburned or pre-burned conditions. Median magnitudes across metals ranged from near 0% for vanadium to 5,000% for cadmium (Figure 2). The reported durations of metal responses were generally less than 5 yr, a finding consistent with that of Rust et al. (2018), which surveyed post-fire water quality responses across multiple watersheds in the western U.S. Mercury was the one exception, with one study finding elevated mercury in lake fishes 10 yr after a wildfire (Figure 3) and a second observed decreased mercury deposition rates in a burned watershed after 33 yr, largely attributed to vegetation differences (Johnson et al., 2007).

Increases in metals were observed following fire across biomes, including boreal peatlands (Brown et al., 2013), temperate forests (Rust et al., 2018), and Mediterranean forests (Burton et al., 2016; Nunes et al., 2017), and from wildfires as well as prescribed burns (Harper et al., 2018). The latter were less represented, and concentrations were typically lower, but Harper et al. (2018) recorded iron and manganese release downstream of prescribed burns. Increased remobilization of some metals and metalloids has also been shown to occur during prescribed burns in fire-prone areas with legacy mining (Abraham et al., 2017a). In general, metal concentrations increased with greater fire severity (Rust et al., 2019). The most common metal measured was mercury, followed by lead and copper. Mercury has been frequently observed in elevated concentrations below fires and is a common target of inquiry given its toxicity and potential for bioaccumulation (see Biological Responses section). Lakes also experience elevated inorganic chemical concentrations, including metals, following fires (McCullough et al., 2019; Smith et al., 2011), but there was comparatively less information on elevated metals in lakes than in streams.

Notably, fires can mobilize metals previously concentrated above background levels by human activities. Atmospheric deposition from anthropogenic sources can increase metals in soils (e.g., lead deposition from legacy fossil fuel combustion and deposition), with the potential for remobilization by fire (e.g., Abraham et al., 2017a; Burke et al., 2013). Historically mined watershed areas can also be at risk for fires, especially in parts of the western U.S. (e.g., see USGS, 2020d). Altered surface and subsurface hydrological flow features may accelerate remobilization and dispersal of arsenic- and metals-containing waste in burned areas surrounding historical mining sites (Murphy et al., 2020). One of the few studies exploring fires burning urbanized areas in the WUI found that ash from building materials in southern California was higher in arsenic, copper, lead, nickel, and zinc than wildland ash (Burton et al., 2016). In this same study, most dissolved metal concentrations were elevated in streams but did not exceed aquatic life use criteria. However, there were 19 observed exceedances of aquatic life use criteria for particulate metal concentrations including for iron, lead, nickel, and zinc, which were 10–1,000x higher than unburned or pre-burn concentrations (Table 2).

Finally, arsenic and cyanide are two, non-metal inorganic chemicals mobilized post-fire. Recent research suggests wildfire may liberate arsenic from soil minerals into a highly mobile form, with the potential for subsequent runoff

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**Table 2**
*Maximum Literature Values Exceeding Magnitudes of U.S. EPA Drinking Water Standards or Ambient Human Health and Aquatic Life Recommended Criteria Values for Water Quality Endpoints in Wildfire Burned Watersheds*

| | Drinking water standards (mg/L) | | Human health recommended criteria (USEPA, 2020e) (mg/L except arsenic as noted) | | Aquatic life recommended criteria (USEPA, 2020c) (mg/L) | | Max observed (mg/L) | |
|---|---|---|---|---|---|---|---|---|
| | Primary (USEPA, 2020a; 2020d) | Secondary (USEPA, 2020b) | Human health for the consumption of water + organism | Human health for the consumption of organism | Acute | Chronic | | Citation |
| Nitrate[a] | 10 | | | | | | 14[b]; 180[c] | d |
| Aluminum | | 0.05–0.2 | | | | | 0.33 | e |
| Arsenic | 0.01 | | 0.018 (µg/L) | 0.14 (µg/L) | | | 0.061 | d |
| Cadmium[f] | 0.005 | | | | 0.0018 | 0.00072 | 0.1 | g |
| Iron | | 0.3 | | | | 1 | 100 | g, h |
| Lead[f] | 0.015 | | | | 0.065 | 0.0025 | 1.0 | e, g, h |
| Manganese | | | 0.050 | 0.100 | | | 0.600 | h |
| Nickel[f] | | | | | 0.47 | 0.052 | 0.10 | g, h |
| Selenium | | | | | 0.0015–0.0031 | | 0.01 | g |
| Zinc[f] | | | | | 0.12 | 0.12 | 1.0 | g, h |
| Cyanide | | | | | 0.022 | 0.0052 | 0.062 | i |
| Benzene | 0.005 | | | | | | 1.25 | j |
| HAA5 | 0.06 | | | | | | 0.66 | d |
| TTHMs | 0.08 | | | | | | 0.23 | d |

*Note.* Values are in mg/L unless otherwise noted, and HAA5 (5 haloacetic acids) and TTHMs (total trihalomethanes) are types of disinfection byproducts. Exceedances for drinking water standards were in source water except for nitrate, arsenic, and HAA5 and TTHMs, which were reported for finished drinking water.

[a]Concentrations for nitrate expressed in mg N/L. [b]Surface water. [c]Groundwater. [d]Pennino et al. (2022). [e]Lydersen et al. (2014). [f]Aquatic criteria are hardness based; values shown for 100 mg/L $CaCO_3$ hardness. [g]Burke et al. (2013). [h]Burton et al. (2016). [i]Gallaher and Koch (2004). [j]Macler et al. (2020).

into surface waterbodies or leaching into groundwater (Johnston et al., 2019). Elevated arsenic concentrations have been reported in multiple watersheds post-fire in the western U.S. (Rust et al., 2018). Pennino et al. (2022) reported significant average increases in public water supply (PWS) arsenic concentrations (0.92 µg/L) and drinking water violations (1.08 violations per PWS) in finished drinking water following wildfire. Similarly, fires can form hydrogen cyanide through the pyrolysis of plant material, and this cyanide may directly enter downwind waterbodies or runoff or leach from ash or soil to nearby waterbodies (Barber et al., 2003). Following a fire in North Carolina and Tennessee, cyanide concentrations averaged 49 µg/L in water droplets at the base of the burned areas, concentrations similar to the median lethal concentration for rainbow trout (Barber et al., 2003). Notably, these concentrations were in water droplets and not in the stream where dilution could occur. The 2000 Cerro Grande fire in New Mexico also resulted in elevated cyanide in runoff samples from burned areas (Gallaher & Koch, 2004; see Table 2). Pyrogenic formation of cyanide or sodium ferrocyanide in fire retardants were suggested as the likely source.

### 3.2.8. Organic Compounds and Persistent Organic Pollutants

Organic compounds with potential health effects may also be released during combustion, mobilized following the fire or used in firefighting. Many of these compounds cause health impacts to humans and aquatic life and are regulated by a range of programs. Of organic compounds in waterbodies post-fire, polyaromatic hydrocarbons (PAHs) have received the most research attention (Table 1), with some notable exclusions (e.g., polychlorinated dibenzodioxins/dibenzofurans). Once in transport, UV radiation in sunlight can activate certain PAH species, enhancing their toxicity to a diverse range of aquatic taxa (Ankley et al., 2003; Hooper et al., 2013).

PAH concentrations generally increased in streams below fires. Of the studies reviewed, 71% measured increases in PAHs and 29% decreases following wildfire (Table 1). Reported magnitudes in receiving water ranged from 0% to 500% of background/reference conditions (Figure 2). Studies suggest PAH increases may be relatively

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short, but these results are uncertain as the database was limited to three studies monitoring only up to 2 yr post-fire (Figure 3). Moreover, even short duration increases in toxics, including those generally persistent and bioaccumulative like PAHs, may cause lasting harm to the health of exposed organisms and populations.

PAH increases occurred in boreal, temperate, and Mediterranean biomes (Campo et al., 2017; Smith et al., 2011; Stein et al., 2012; Wisløff, 2018). In Spain and Portugal, for example, diphenyl ethers and a variety of priority pollutant PAHs, including acenaphthylene, benzopyrene, fluorene, naphthalene, and pyrene have all been observed in surface soils, sediment, groundwater, runoff, and surface water impacted by fires (Campo et al., 2017; Mansilha et al., 2014, 2019; Nunes et al., 2017). Concentrations in overland runoff were generally higher than in streams, suggesting attenuation along the overland flow path or dilution. Still, absolute PAH concentrations post-fire rarely exceeded the World Health Organization guideline for benzo(a)pyrene (0.0007 mg/L, Smith et al., 2011).

In addition to PAHs in runoff, fire can directly impact drinking water infrastructure in WUI urban areas, causing organic compound contamination. A recent paper reported benzene and other volatile organic compounds (VOCs) at concentrations 2–3 orders of magnitude above drinking water standards, likely caused by thermal degradation of plastic materials (Macler et al., 2020). Fires in Santa Rosa, CA and Paradise, CA in 2017 and 2018, respectively, burned houses and melted plastic water service lines. The partial melting of the pipes and back siphonage of smoke, ash, and other materials during depressurization of the water system likely led to the spreading of VOC contamination through the water pipes. Benzene was detected at concentrations exceeding federal and state standards (Table 2). Do not drink orders were issued, and contaminants sorbed to plastic pipes required flushing and, in some cases, wholesale infrastructure replacement to remedy (Macler et al., 2020).

Although generally banned in the U.S., persistent organic pollutants like polychlorinated biphenyls (PCBs) still have drinking water standards because they endure in the landscape due to relatively slow decay rates. Smoke and ash from fires can transport these over longer distances and subsequently deposit them (Smith et al., 2011), yet we found no data reported for these chemicals in the research papers reviewed. Finally, per- and polyfluoroalkyl substances (PFAS) are a large group of persistent, human-made chemicals with numerous uses and exposures linked to adverse human health outcomes. In the U.S., fire fighting foams employed at places such as military bases have contained PFAS compounds, but fire retardants currently approved for federal wildlands do not. A study of fire effects in Spain, including the use of aerial fire retardants, did not find elevated PFAS concentrations in burned sites (Campo et al., 2017).

#### 3.2.9. Disinfection Byproducts (DBPs)

As noted previously, concentrations of carbon and nitrogen compounds often increase following fire, and these compounds can react with disinfectants (e.g., chlorine, chloramine) to form DBPs during drinking water treatment (Bladon et al., 2014; Hohner et al., 2016, 2019). Some DBPs pose health risks, with the ability to cause certain cancers, reproductive issues, and anemia (USEPA, 2020a). Predominantly, the reviewed studies measured DBP formation potential, reacting source water with drinking water treatment chemicals (halogens) in the laboratory.

Most studies (58%) identified little change in the response of DBP formation potential post-fire, with the rest of the studies equally divided between those identifying increases and decreases (21% each, Table 1). On average the magnitude of DBP formation potential responses ranged between 0% and 500% (Figure 2) and these changes were relatively short-lived (<5 yr, Figure 3). However, given the health risk of these chemicals, even short-lived increases may be important.

A recent study on DBPs and wildfire described the formation potential risk as being greatest below fires of medium intensity, since at high temperatures more carbon and nitrogen are volatized and, at lower temperatures, insufficient transformations take place to generate dissolved carbon and nitrogen compounds (Hohner et al., 2019). The authors found formation potential of trihalomethane (THM) and haloacetic acids (HAA), two classes of DBPs, at concentrations above drinking water regulatory levels due to a supply of low molecular weight DOC compounds from runoff in a burned watershed in Colorado (Hohner et al., 2019). The study also observed the formation potential of unregulated haloacetonitriles (HANs), another type of DBP, from low molecular weight DON molecules (Hohner et al., 2019). DBP formation potential from wildfire-generated DON/DOC has been found in several other experimental studies as well, including increased HAA/HAN/THM formation potential from source water collected below other fires in Colorado and exposed to chlorine; dissolved black carbon extracted from burnt leaves of several species in South Carolina; and water extracted organic matter taken

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from California wildfire ash (Majidzadeh et al., 2015; Wang et al., 2015; Writer et al., 2014). Each experimental study found both elevated carbon- and nitrogen-based DBPs.

Beyond the study of DBP formation potential, a recent study of contaminants in public drinking water found actual increases in DBPs (not merely a laboratory based potential, but detection in the delivery system itself) in finished drinking water sourced from burned watersheds (Pennino et al., 2022). The authors found 71% of drinking water facilities below burned watersheds showed an increase in total THM violations (an average of +0.58 violations per PWS) in the first 2 yr post-fire compared to pre-fire levels (Pennino et al., 2022). Total THM concentrations increased an average of 10.4 μg/L in the first year post-fire and remained generally elevated for 5 yr (Pennino et al., 2022). For total HAA, containing 5 haloacetic acid compounds (see USEPA, 2010; USEPA, 2020a), 50% of sites below burned watersheds increased in violations post-fire (an average of +0.82 violations per PWS over 5 yr). Total HAA concentrations increased by 8.5 μg/L in the first year and stayed elevated for 5 yr, while violations remained higher 8 yr post-fire.

#### 3.2.10. Fire Fighting Chemicals

In addition to chemicals mobilized directly or indirectly by fire, chemicals are used to fight wildfire. Wildfire fighting chemicals include three primary types: long-term retardants, foam suppressants, and water enhancers (USFS, 2020a). Long-term fire retardants are typically fertilizers (phosphate, ammonia, and sulfate) and smaller concentrations of colorants, corrosion inhibitors, thickeners, and stabilizers mixed with water (USFS, 2020a). They reduce the flammability of fuel through a chemical reaction and are effective even after the water evaporates and so are often used ahead of the fire to decrease or stop fire from spreading. Fire suppressant foams are detergents used to suppress fire directly and prevent reignition. They are used with water and often applied directly on burning fuel. Water enhancers are composed of polymers and other thickeners that reduce aerial dispersion and increase application thickness and adherence, improving water's fire suppression effectiveness (USFS, 2020a).

The United States Forest Service (USFS) is the lead agency overseeing the use of wildland fire chemicals on U.S. federal wildlands, maintaining a Qualified Products Lists (QPL) of fire chemical products (USFS, 2020a). Unacceptable ingredients in long-term retardants include sodium ferrocyanide, PCBs, and PFAS chemicals, among others (USFS, 2020b). Sodium ferrocyanide, a corrosion inhibitor, is no longer an acceptable ingredient because this can subsequently degrade into free cyanide in water (Crouch et al., 2006; Giménez et al., 2004). Notably, the list of unacceptable ingredients may not apply in other circumstances outside of federal wildlands, such as in the chemicals used by state and local firefighting entities, or on U.S. military bases where PFAS-containing foams were employed for decades to fight fires (US GAO, 2021).

Beyond the types of chemicals used, the USFS protocols specify avoidance zones for the use of aerially dispersed fire fighting chemicals, including the avoidance of waterways and their surrounding buffers, except in cases where retardant use within avoidance areas is needed for public safety (USFS, 2020c). Nevertheless, the potential exists for misapplication or application and subsequent runoff of these chemicals. Studying four separate wildfires, Crouch et al. (2006) compared concentrations of ammonia, phosphorus, and cyanide in streams draining burned areas where fire retardants were applied vs. burned areas where fire retardants were not applied. They found similar concentrations regardless of the use of fire retardants, suggesting the fire retardants employed at these fires did not affect surface water quality. They further concluded the low amounts of cyanide detected were pyrogenically formed, not originating from the fire retardants.

#### 3.2.11. Radionuclides

Finally, radionuclides have been remobilized by wildfire in locations characterized by elevated background radiation, such as the Chernobyl Exclusion Zone and watersheds in proximity to Los Alamos National Laboratory in New Mexico. Bondar et al. (2014) applied an empirical model to assess the contribution of wildfire activity to radionuclide contamination of two water bodies in the Chernobyl Exclusion Zone. In years with greater number of wildfires, fallout of $^{137}$cesium and $^{239,\ 240}$plutonium to the water surface increased, suggesting that radionuclides were being introduced to water bodies from the wildfire events. The Cerro Grande fire in New Mexico in 2000 impacted the Los Alamos Laboratory site. Four years of water quality monitoring data following the fire showed increased transport of radionuclides in initial runoff events, with median concentrations of total radionuclides increasing by 10–50 times from prefire levels (Gallaher & Koch, 2004). By 4 yr post-fire, however, elevated concentrations had subsided (Gallaher & Koch, 2004). Wildfires have been shown to condense and mobilize radionuclides in soils. In burned plots in Colorado and New Mexico, soils had concentrations

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of $^{137}$Cesium 400%–2,200% of those in unburned plots (Bitner et al., 2001). Although the article did not specify, the presumed mechanism for radionuclide release is combustion of contaminated organic matter or vegetation. In locations not in proximity to sources of radiation, remobilization of radionuclides in runoff from past radioactive global fallout has been applied as a technique to trace sources and transport of sediments in runoff following wildfire events (Blake et al., 2009; Wilkinson et al., 2009).

### 3.3. Biological Responses

#### 3.3.1. Microorganisms/Pathogens

Beyond the physical and chemical effects of wildfire, there is the potential for biological responses as well. There are several microorganisms of concern for drinking water supplies and recreation that are either pathogens or are used as indicators for the possible presence of microbial pathogens (e.g., coliforms, such as *E. coli*, are used as indicators). Contamination of waterbodies by these microorganisms of concern is generally not expected post-fire, unless wastewater infrastructure is damaged. Fire frequently reduces soil microbial populations directly due to heating (Neary et al., 1999). Moreover, toxicological tests have found that ash (i.e., AEA) can inhibit bacterial growth (Silva et al., 2015) and that wildfire residues reduce the persistence of some bacteria, including, *E. coli* (Valenca et al., 2020). Microbial populations in the soil recover over time (LeDuc & Rothstein, 2007), but microorganisms of concern for drinking water and recreation generally originate from human or animal waste. Fire damage to wastewater infrastructure (e.g., wastewater pipes, waste treatment facilities) could lead to subsequent spills or leaching of waste and associated microorganisms into surface- or groundwater. We did not, however, identify any studies characterizing the effects of fire on microbial pathogens or indicators in waterbodies.

#### 3.3.2. Benthic Invertebrates

Invertebrate assemblages are an integral component of water monitoring programs across the U.S. and globally, with declines in species richness, diversity, and density associated with impaired aquatic habitat. Most studies found a negative impact of wildfire on macroinvertebrate assemblages (57% of studies, Table 1), the magnitude of which was, on average, a 45% decline (Figure 2). Increases in ash, sediment, temperature, metals, altered pH, alkalinity and conductivity, and increases in other pollutant loads are all known to adversely impact invertebrate assemblages (Allan & Castillo, 2007). In addition, changes in physical habitat resulting from altered sediment loads, especially debris flows, and concomitant scour and channel adjustments all affect invertebrates (Allan & Castillo, 2007). Shifts in food resources post-fire with changes in light and nutrients (e.g., either more or less algal dominated) also affect assemblage structure (Minshall et al., 1997). Invertebrate assemblages, however, typically recover rapidly following alleviation of stress (e.g., Allan & Castillo, 2007). Most invertebrates generally recovered within 5 yr of disturbance, with individual taxa recovery lasting as long as 10 yr following fire (Figure 3).

In the majority of studies, invertebrate diversity in streams was reduced following wildfire, especially sensitive taxa (e.g., from the Ephemeroptera, Plecoptera, and Trichoptera orders), and this was true in boreal, temperate, and Mediterranean biomes (Beche et al., 2005; Brown et al., 2013; Earl & Blinn, 2003; Harper et al., 2018; Harris et al., 2015; Minshall et al., 2001; Rinne, 1996; Spencer et al., 2003; Verkaik et al., 2013; Whitney et al., 2015). In several studies, invertebrate densities also decreased following wildfires. In New Mexico, for example, invertebrate densities declined in one burned watershed, but recovered within a year (Earl & Blinn, 2003). In the second study in New Mexico, invertebrate biomass declined almost 82% after more severe fires and some declines lasted for nearly 2 yr (Whitney et al., 2015). By contrast, studies also show an increase in invertebrate biomass post-fire (e.g., Malison & Baxter, 2010). For instance, in a survey in the northern Rocky Mountains, invertebrate abundance was two times higher in fire-impacted streams compared to reference streams (Martens et al., 2019). This was 8 yr post-fire, likely allowing time for invertebrates to recover and take advantage of resource expansion. Even still, reference streams were characterized by higher relative abundance of sensitive taxa (Martens et al., 2019). In individual cases, macroinvertebrate impacts can last as long as 5 yr or more, with the duration likely depending on the extent of changes in physical habitat (channel morphology) and food resources. After a large-area fire (>15,000 ha) in Montana, for example, invertebrate carbon and nitrogen isotope signatures 4–5 yr post-fire still reflected greater dependence on algal food resources than those in unburned watersheds (Spencer et al., 2003).

In many studies, initial invertebrate responses are negligible until it rains, and runoff carrying sediment and other pollutants occurs. For example, immediately after a fire in Arizona, invertebrate densities were unchanged from

pre-fire conditions, whereas after the first storm, invertebrate densities declined precipitously (Rinne, 1996). Other factors influence invertebrate responses. For example, in burned watersheds with debris flows, the diversity and biomass of stream invertebrates exported (drifting) from the water was much lower than in burned watersheds without debris flow or unburned watersheds (Harris et al., 2015). As predicted, food webs in streams draining burned watersheds do exhibit shifts toward opportunistic, shorter life-cycle generalists (Minshall et al., 1997) or toward more algal based food resources when fires led to increased algal growth (Cooper et al., 2015; Rugenski & Minshall, 2014). For lakes, we identified only two studies and two reviews discussing littoral invertebrates or zooplankton responses (Allen et al., 2005; Lewis et al., 2014; McCormick et al., 2010; McCullough et al., 2019). Where they have been documented, however, the changes are similar to those in streams post-fire, with shifts in diversity combined with changes in the food web.

There has been some work on the effects of fire fighting chemicals on invertebrates. Foam suppressants were generally reported to be more acutely toxic to *Hyallella* amphipods than fire retardants (McDonald et al., 1997). The authors postulated that surfactants affected oxygen diffusion. In these experiments, ammonia concentrations also exceeded aquatic life criteria by 120x, which was hypothesized as a major cause of acute toxicity. In Spain, studies of vernal pools exposed to the fire retardant FireTrol 934 found that ammonia and conductivity increased and pH decreased, and these were associated with a decline in invertebrate diversity (Angeler et al., 2006). The same chemical was found to exhibit a toxic effect on zooplankters in lab studies (Martin, 2016). Lastly, the fire-retardant GTS-R was found to release cyanide when exposed to ultraviolet light (Calfee & Little, 2003). Neither of these two retardants is on the current USFS QPL (USFS, 2020a). Lastly, toxicological experiments show adverse effects of some retardants and foam suppressants on lentic invertebrates, including both zooplankters (Angeler et al., 2006) and benthic invertebrates (McDonald et al., 1997).

#### 3.3.3. Fish

Wildfire effects on fish populations can be direct, such as localized extirpation occurring during or immediately following a fire event, or indirect such as post-fire physical and chemical alteration of fish habitat (Gresswell, 1999). Reported fish assemblage responses to wildfires are similar to those of invertebrates: the majority of studies (63%) indicate declines in abundance and diversity of taxa (ranging from 30% to 100% declines, Table 1 and Figure 2). Fish biomass can decline rapidly after fire, most commonly after the first storm introduces pollutants. In Arizona, a severe fire resulted in little decline until the first post-fire storm, after which salmonids were completely absent and only showed some recovery after 2 yr and restocking efforts (Rinne, 1996). Notably, longer-term outcomes may be better than these findings suggest. For instance, fires may contribute to improved fish habitat, particularly through increases in woody debris to streams (Flitcroft et al., 2016). Nonetheless, fish generally do not disperse as quickly as invertebrates, so recovery after fires can take longer depending on fish movement and habitat connectivity (Rosenberger et al., 2015). The average duration of effect in studies surveyed was indeed longer than that for invertebrates, yet still generally within 5–10 yr (Figure 3).

One research focus area has been on post-fire thermal effects on fish. Many cold-water taxa (e.g., salmonids) have strict thermal requirements and much of their remaining habitat is at the extent of their tolerance. As discussed previously, water temperatures generally increase following burning, especially when severe fires remove riparian vegetation. This can reduce cold-water habitat, including refugia, leading to extirpation. In Idaho, salmonid densities (age class 1+) were lower 10 yr following wildfires in watersheds that burned and experienced debris flows. This response was likely due to the effect of higher water temperatures from slow riparian recovery (Rosenberger et al., 2015). In California, salmonid declines following fires were attributed to an increase in stream pool thermal refugia temperatures, leading to higher energetic maintenance costs (Beakes et al., 2014).

Besides temperatures, post-fire mercury bioaccumulation is another area of focus, with implications for fish consumption advisories. Some studies on mercury bioaccumulation in fish following burning note no or minimal response while others recorded increases. For example, mercury increased in rainbow trout (*Oncorhynchus mykiss*), brown trout (*Salmo trutta*) and European perch (*Perca fluviatilis*) in Canadian and Norwegian lakes impacted by wildfires (Kelly et al., 2006; Moreno et al., 2016). Factors affecting this increase included both higher water concentrations of mercury and methylmercury following wildfires, as well as a restructuring of the food web and lengthening of trophic pathways, leading to greater potential for bioaccumulation. Still other studies have not observed increases, as in the lack of a mercury increase in yellow perch (*P. flavescens*) following fires in northern Minnesota (Riggs et al., 2017). Many factors affect mercury cycling, and identifying the specific risks for any single waterbody may be necessary.



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Lastly, experimental studies have examined the effects of fire fighting chemicals on fish. In one study, rainbow trout (*O. mykiss*) avoided elevated concentrations of fire retardants (Wells et al., 2004), suggesting these chemicals may affect movement and impose metabolic costs. The authors did not identify specific compounds responsible for the avoidance cue, although they experimentally ruled out colorants or corrosion inhibitors, and suggested that ammonium salts may be a culprit given the sensitivity of this taxon and other fishes to salts. Fire retardants caused acute toxicity to *O. tshawytscha* (chinook salmon) yearlings at concentrations possible during accidental applications (Dietrich et al., 2013). The resulting ammonia concentrations were above those sufficient for acute mortality and were a presumed contributor (Dietrich et al., 2013). Foam suppressants were generally reported to be more acutely toxic to chinook salmon larvae and juveniles than fire retardants (Buhl & Hamilton, 2000).

#### 3.3.4. Algae

As noted previously, algal growth is important to water regulatory programs because excess algal biomass can interfere with water treatment systems, creating compounds that lead to DBP formation and taste and odor issues. Additionally, several harmful algal taxa produce toxins threatening human, livestock, and aquatic life (Chorus & Welker, 2021). Algal growth can also affect food webs and cause chemical responses like hypoxia (low DO), with the potential to produce fish kills, mobilization of sediment metals, and other adverse ecosystem impacts.

Algal assemblages can be impacted by wildfire, both through changes in biomass and composition, depending on waterbody type, context, and timing (Bixby et al., 2015; McCullough et al., 2019). Physical scour of stream substrate by erosion and sediment transport during runoff is one factor leading to decreases in initial stream algal biomass. This has been offset over time by increased light availability from riparian canopy loss, increasing algal biomass and shifting competition toward high light taxa (Bixby et al., 2015; Klose et al., 2015; Verkaik et al., 2013). Wildfire-mediated nutrient enrichment also can increase algal biomass and shift composition (Kelly et al., 2006; McCullough et al., 2019). Some studies of lakes, however, have shown this effect could be dampened by decreased light availability from post-fire dissolved or suspended material (e.g., Allen et al., 2003).

Most studies reported no directional change in algal response to fire (48%), with the balance roughly equally split between increases or decreases (Table 1). The magnitude of the response was also relatively modest (Figure 2), and most reported algal response durations lasted less than 10 yr after the fire (Figure 3). Wildfires in New Mexico and southern California led to initial declines in stream algal biomass, up to an order of magnitude lower than pre-fire levels (Klose et al., 2015; Whitney et al., 2015). In California, however, where the riparian canopy was removed by fire, algal biomass ultimately increased 5–20x over pre-burn levels over time, indicating the importance of light (Klose et al., 2015). The duration of effects on algae in streams will likely depend on how long nutrient concentrations stay elevated and how quickly riparian vegetation recovers to limit light availability.

Algal responses in lakes post-fire were variable as well, with some lakes experiencing increases and others not (Lewis et al., 2014; McCullough et al., 2019). The characteristics of the fire and antecedent lake conditions likely influence the response. For example, already productive lakes might not see an increase in algal growth post-fire, whereas oligotrophic systems would likely respond to nutrient input if sufficient light is available. In Spain, algal biomass increased in ponds following a burn, largely attributed to increased nutrient levels (Cunillera-Montcusi et al., 2019). Post-fire increases in nutrients can lead to increased algal growth in water supply reservoirs, with the potential to impact human health by increasing algal DOC and DBP formation risk (Hohner et al., 2019).

Fire retardants may also be another factor affecting algal responses. In laboratory experiments, fertilizers in fire retardants have increased algal growth (Angeler et al., 2004). As noted previously, best management practices are to avoid riparian areas when spreading fire retardants (USFS, 2020c), yet there is the potential for misapplication or subsequent runoff into waterbodies after a fire.

### 3.4. Key Knowledge Gaps

Although the effects of wildfire on many endpoints have been well-characterized, matching our literature review with the topics in the conceptual model (Figure 1) makes it clear that some endpoints and pathways have not. Unfortunately, some of the topics receiving little research also pose some of the greatest threats to water quality programs.

Most strikingly, little is known about the effects on water endpoints from wildfire in urban areas in the WUI. Downwind air quality aside, these areas are where the largest number of people will be affected now and in

the future. We fully acknowledge that there are legal, safety, and resource constraints on sampling these areas during immediate post-fire runoff events, but the lack of study is glaring. We identified one peer-reviewed study that found ash generated from residential and building materials had higher metal concentrations (e.g., arsenic, copper, lead, nickel, and zinc) than wildland ash (Burton et al., 2016). A second set of articles described increases in benzene and other VOCs due to damaged pipes following two destructive wildland fires affecting WUI urbanized areas in California (Macler et al., 2020; Proctor et al., 2020). No other study we found focused explicitly on the effects of these types of fires on endpoint responses. Major outstanding research questions include: is the mixture of pollutants mobilized different than those of wildland fire as Burton et al. (2016) suggest? Do these chemicals rapidly move into nearby drains or waterways in subsequent storm events given the higher proportion of impervious surfaces in the WUI? And, what priority contaminants should be tested for—downstream, at the tap and/or at the drinking water facility—when these fires occur?

Beyond urban fires, there is a large research gap in wildfire responses for many chemicals, including many regulated pollutants. Of the 82 non-pathogen related primary drinking water standards (USEPA, 2020a), wildfire effects were reported for 13 (Table 1). For the secondary drinking WQS, effects were reported for 5 of 15 (Table 1). Many primarily synthetic chemicals with knowledge gaps may not arise from forested watersheds, but may originate from fires burning urbanized areas. Studies of the risk of HABs and cyanotoxins post-fire are also urgently needed.

Other important knowledge gaps include those surrounding waterbody type. Studies of wildfire effects on streams are far more common than those in lakes, reservoirs, wetlands, or estuaries, creating a knowledge gap. Whether responses in lakes and other lentic waterbodies differ in important ways is an especially critical question given municipalities often rely on lakes and reservoirs for their drinking water. Beyond surface water, Pennino et al. (2022) found post-fire contamination in municipal groundwater sources, yet impacts on groundwater remain little studied. Communities that rely on individual or municipal wells for drinking water need more information on this emerging topic.

## 4. Considerations for Water Quality Programs

The purpose of this review was to synthesize what is known about the effects of wildland fires and fires burning urban areas in the WUI on stressors relevant to water quality programs. Here, we highlight how some of the programs could be affected, a few of the most consequential stressors, and select prevention or mitigation steps that could be considered.

Stressors generated by wildfire can affect drinking water programs by impacting drinking water sources, increasing treatment needs and costs, and affecting drinking water infrastructure directly (Table 3). Across the literature reviewed, post-fire chemical concentrations for several analytes were reported at concentrations exceeding drinking water standards (Table 2). This included samples in finished drinking water in the case of nitrate, arsenic, and DBPs (total haloacetic acids and total trihalomethanes; Pennino et al., 2022). Managers of public water supplies should be aware of the potential for post-fire contamination from these chemicals in particular. Physical damage from fire is also a threat to drinking water infrastructure, such as the cases observed in California (Macler et al., 2020). Areas in the fire-prone WUI may need to consider mitigation measures, such as backflow preventers to keep contaminants from burning infrastructure out of drinking water distribution systems. Moreover, smaller drinking water suppliers are especially vulnerable from sedimentation or damage if they only have one drinking water source, potentially requiring a regional approach to help reduce risk (Bladon et al., 2014).

Programs focused on WQS and wastewater may also need to consider impacts from fire. Under the CWA, states and tribes adopt WQS that assign designated uses (e.g., recreation) to water bodies and water quality criteria to protect those designated uses. These standards form the basis of wastewater permitting, assessment and restoration of waterbodies. Standards and wastewater programs in the U.S. may face increased requests for variances, allowing waterbodies to temporarily exceed criteria due to wildfire. Alternatively, states and tribes may propose site specific criteria for pollutants, justifying wildfires as natural conditions. Other impacts might include the incorporation of natural variability from fires into threshold setting for biocriteria, and like drinking water providers, wastewater facilities should be aware of their susceptibility to physical damage from the direct and indirect effects of fires.



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**Table 3**
*Common Water Quality Programs Potentially Affected by Wildfire, With Example Impacts Given for Each Program*

| Water program | Example impacts |
|---|---|
| Drinking water | • Stressors may adversely affect drinking water supplies, source water quality and treatment efficiency and costs<br>• Drinking water infrastructure may be physically damaged |
| Water quality standards | • Standards programs may have increased requests for variances, allowing waterbodies to temporarily exceed criteria<br>• Natural condition site specific criteria may need to incorporate fire effects<br>• Aquatic life criteria setting based on reference condition may need to include wildfire effects on reference baselines |
| Wastewater | • Altered flows may influence low flows and reasonable potential analyses<br>• Altered background pollutant concentrations may lead to an impact on permitting calculations<br>• Water permitting programs may have increased requests for variances, allowing waterbodies to temporarily exceed limits<br>• Wastewater infrastructure may be physically damaged |
| Assessment | • Assessment programs may see listings/delistings depending on fire impact magnitudes and durations<br>• Exceedance frequencies may need to be adjusted for fire affected watersheds<br>• Beach closures to swimming and to other recreation use may increase due to post-fire harmful algal blooms |
| TMDLs | • Reference targets for total maximum daily loads (TMDLs) using forested watersheds may need to consider fire effects<br>• Upstream fire effects may need to be incorporated into load reduction scenarios |
| Non-point source control | • Increased need for watershed restoration post-fire to reduce sedimentation and other impacts |

Water quality assessment programs may also need to consider fire impacts as they influence variability in physical, chemical, and biological conditions of water in reference watersheds (e.g., Bixby et al., 2015), especially as these impacts can last for many years (e.g., Emelko et al., 2016; Minshall et al., 2001; Rhoades et al., 2019). Across the literature reviewed, post-fire chemical concentrations for several analytes were reported at concentrations exceeding national recommended aquatic life use criteria, including: cadmium, iron, lead, nickel, selenium, and zinc (Table 2). Pathogen-related recreational criteria are likely to be minimally impacted by wildfire, unless physical damage of waste infrastructure occurs. However, increased nutrient loads may contribute to downstream harmful algal blooms and the potential for cyanotoxins to exceed advisory levels for recreation (e.g., swimming).

Efforts to restore impaired waters may also need to account for fire in TMDL planning. For example, TMDL programs may need to be aware of how burning can affect low flow statistics (e.g., Shakesby & Doerr, 2006), as well as the background pollutant levels upon which load allocations may be derived. Restoration may be further confounded if wildfires generate pollutants undergoing management (e.g., sediment or temperature). These effects may need to be incorporated into load reductions. Destructive fires in urbanized areas in the WUI may also make discriminating contributions from human and forested sources difficult.

Additionally, wildfire has the potential to affect non-point source management (CWA Section 319) efforts to reduce stressors, and burned watersheds, in turn, could be targeted for restoration to reduce non-point sources. Revegetation of burned watersheds shortens the duration of many wildfire water quality effects (e.g., nutrients, sediment, and temperature) and it may be important to prioritize non-point source improvements, such as hillslope and riparian revegetation, as well as other management practices in watersheds where wildfire has impacted water quality. For instance, the Pueblo of Santa Clara utilized U.S. EPA's Section 319 resources (https://www.epa.gov/nps/319-grant-program-states-and-territories) to install erosion control measures following the Las Conchas Fire in 2011, with these efforts leading to improved water quality conditions (USEPA, 2019b).

Finally, prescribed fire is a tool by which managers may potentially lessen the risk of extensive, high-severity fires and their subsequent water quality effects. Prescribed fire is likely to be most effective in ecosystems that historically experienced frequent, low severity fires, yet less so in ecosystems characterized by infrequent, high severity fires (Moritz et al., 2014). In this review, we found fewer studies (21) on prescribed burn responses

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than we did on wildfire responses (126). Since prescribed fires are typically low severity fires, they generally elicit decreased magnitude and/or duration of physical, chemical, and biological responses in waterbodies. One study, for example, found similar runoff responses following prescribed burns but for shorter duration when compared to that of wildfire (dos Santos et al., 2017). Similarly, nutrient increases were less under prescribed burns and also of shorter duration than for wildfire (e.g., Knoepp & Swank, 1993), and two studies of prescribed burns on macroinvertebrates found generally moderate responses of <1 yr duration (Beche et al., 2005; Brown et al., 2013). Utilizing prescribed burning in particularly fire-prone watersheds or with vulnerable drinking water or wastewater resources may be an effective means to reduce risks to water programs.

## 5. Summary and Conclusions

Fire is a natural disturbance in many ecosystems, yet the increase in fire frequency poses a risk to water quality in the U.S. and globally. Here, we synthesized the effects of wildfires on the physical, chemical, and biological endpoints relevant to water quality management programs. We acknowledge that for any specific wildfire response, the individual watershed context is important, including factors like the fire regime and post-fire hydrology, and contributes to the large variation in reported responses. Still, general trends emerged from this review.

Physically, wildfires result in vegetation mortality and expose and alter the soil, leading to reduced infiltration, greater runoff, higher streamflows and greater erosion and sediment export. The loss of tree canopy can also increase water temperatures. Increases in physical responses surveyed in this review ranged between 100% and 10,000% of unburned or pre-fire reference conditions, with the highest values for surface runoff and sediment responses. On average, the duration of effects was shortest for suspended sediments (<5 yr), while longer (5–10 yr) for streamflow and temperature. In most cases, duration estimates were constrained by the length of the studies, so any duration estimate should be considered conservative.

For the chemicals reviewed, concentrations generally increased in response to fires. Similar to the other studies (e.g., Rust et al., 2018, 2019), we found that nutrients and metals were commonly elevated post-fire. Nutrient concentrations nearly universally increased, with few exceptions. Metals and polycyclic aromatic hydrocarbons were typically higher post-fire as well. Concentration increases, especially for individual metals, can be quite high compared to reference or before conditions (as much as 100,000%), with studies observing increases of iron, lead, manganese, nickel, zinc, and arsenic above regulatory standards or aquatic criteria (Table 2). On average, chemical effects lasted less than 5 yr post-fire, although individual cases lasting longer did occur.

Biological assemblages most commonly decline with fire, with the exception of tolerant or opportunistic organisms. The most common observations were declines in biomass (or densities) of all assemblages (except for algae, which may increase in biomass in response to increased light and nutrients), declines in diversity due to the loss of sensitive taxa, and, for invertebrates, an increase in opportunistic or tolerant species. The average duration of these effects was shorter for invertebrates (median of ∼1 yr) than fish (median of ∼3 yr).

Through our review, we also identified key knowledge gaps; a principal one being the mobilization of pollutants from fires burning anthropogenic sources in the WUI. This problem is only likely to grow as urbanization expands and fire frequency increases with climate change. Some of these communities rely on a single municipal source or individual private wells that, if damaged or contaminated, could substantially impede recovery.

Finally, we conclude that an array of water quality programs in the U.S. and globally will likely need to consider the effects of fire. Previous reviews have emphasized the problems to water quality posed by sediments, nutrients, and metals, among other chemicals (e.g., Bladon et al., 2014; Rust et al., 2018; Smith et al., 2011). We echo these findings, and also particularly highlight cases where nitrate, arsenic, and two classes of DBPs exceeded U.S. drinking water standards in the finished drinking water itself following fire (Pennino et al., 2022). This occurred in both surface and groundwater, highlighting the vulnerability of the latter to chemicals highly mobile in the environment. Although not a focus of our review, to our knowledge the effects of fire on groundwater quality has received little attention to date. We also note the recent finding of benzene and VOCs in tapwater from infrastructure damage in the WUI as a potentially important, emerging issue. By providing this information, we hope communities, water quality managers, and other decision makers can better plan for, mitigate, and, if necessary, recover from wildfire impacts.

## Data Availability Statement

All extracted information from citations used in this review are being provided as raw data, searchable atomized statements and summary synthetic statements in a keyword searchable Excel workbook provided in a data repository (to be added if/when publication occurs). These are included in Supporting Information S3.

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