

Mini Review

From Ash to Algae: Wildfire-Driven Water-Quality Change and Emerging Cyanotoxin Risk in Aquatic Systems

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Abstract: Wildfire can alter downstream water quality, aquatic productivity, and drinking-water treatability. Evidence for chemical and physical water-quality change is substantial, whereas evidence connecting those changes to harmful algal blooms (HABs) or cyanotoxins is less developed. This mini review examines evidence from freshwater receiving waters for post-fire water-quality change, algal or cyanobacterial responses, and cyanotoxin endpoints. Wildfire acts as a conditional amplifier of aquatic risk. Outcomes vary with fire severity, hydrologic connectivity, nutrient bioavailability, optical conditions, residence time, and receiving-water susceptibility. Evidence is strongest for changes in turbidity, sediment, nutrients, dissolved organic matter, and treatment-relevant chemistry; moderate for algal productivity and community responses; and limited for direct attribution of post-fire cyanotoxin occurrence. Event-coupled studies should link fire characteristics and hydrologic transport with water chemistry, cyanobacterial composition, cyanotoxins, and drinking-water outcomes in the same receiving water.

Keywords: wildfire; post-fire water quality; harmful algal blooms; cyanobacteria; cyanotoxin

1. Introduction

Wildfire is an important watershed disturbance because combustion and post-fire erosion alter the amount, timing, and form of material delivered to surface waters. Ash, sediment, nutrients, dissolved organic matter (DOM), major ions, and metals can be mobilized during rainfall, snowmelt, and debris-flow events [1–3]. These changes affect aquatic habitat and primary production. They also create operational problems for drinking-water utilities through high turbidity, greater filter loading, altered coagulation demand, and changes in disinfection by-product precursors [4,5].

Freshwater harmful algal blooms (HABs) can impair ecosystem condition and recreational use, while cyanotoxins create additional concerns for source-water monitoring and drinking-water treatment. Bloom development depends on nutrient availability, light, temperature, mixing, residence time, and community composition. Cyanotoxin occurrence also depends on the

abundance and physiological state of toxigenic cyanobacterial populations [6,7]. A post-fire nutrient pulse may favor algal production in a susceptible water body, whereas turbidity or rapid flushing may prevent biomass accumulation. A chemical perturbation alone does not establish a HAB or cyanotoxin event.

The post-fire water-quality and HAB literatures rarely resolve this problem in the same receiving water. Wildfire reviews provide extensive evidence for erosion, sediment transport, nutrient export, DOM alteration, and treatment challenges [1,3,8]. HAB studies instead examine the environmental and biological controls on cyanobacterial accumulation and cyanotoxin production [6,7,9]. Direct studies that connect a defined wildfire exposure with water chemistry, algal community change, cyanotoxins, and drinking-water outcomes remain scarce. This gap limits causal attribution and leaves water-sector operators without a clear basis for deciding when post-fire surveillance should expand from routine water-quality monitoring to cyanobacteria and cyanotoxin measurements.



Causal attribution requires evidence that material mobilized after fire reaches the receiving water during a period favorable for algal growth. Rapid flushing or sediment-induced light limitation may prevent a nutrient pulse from increasing biomass. Even when chlorophyll *a* increases, diatoms or green algae may account for the response. Cyanobacterial abundance or dominance does not establish cyanotoxin production, and source-water cyanotoxin detection does not establish finished-water occurrence or

human exposure. Linking wildfire to cyanotoxin risk therefore requires evidence at each transition.

This mechanistic mini review examines how wildfire alters freshwater chemistry and physical habitat, and whether those changes influence algal communities and cyanotoxin risk. Figure 1 summarizes these relationships, the environmental conditions that influence them, and the associated research and monitoring priorities.

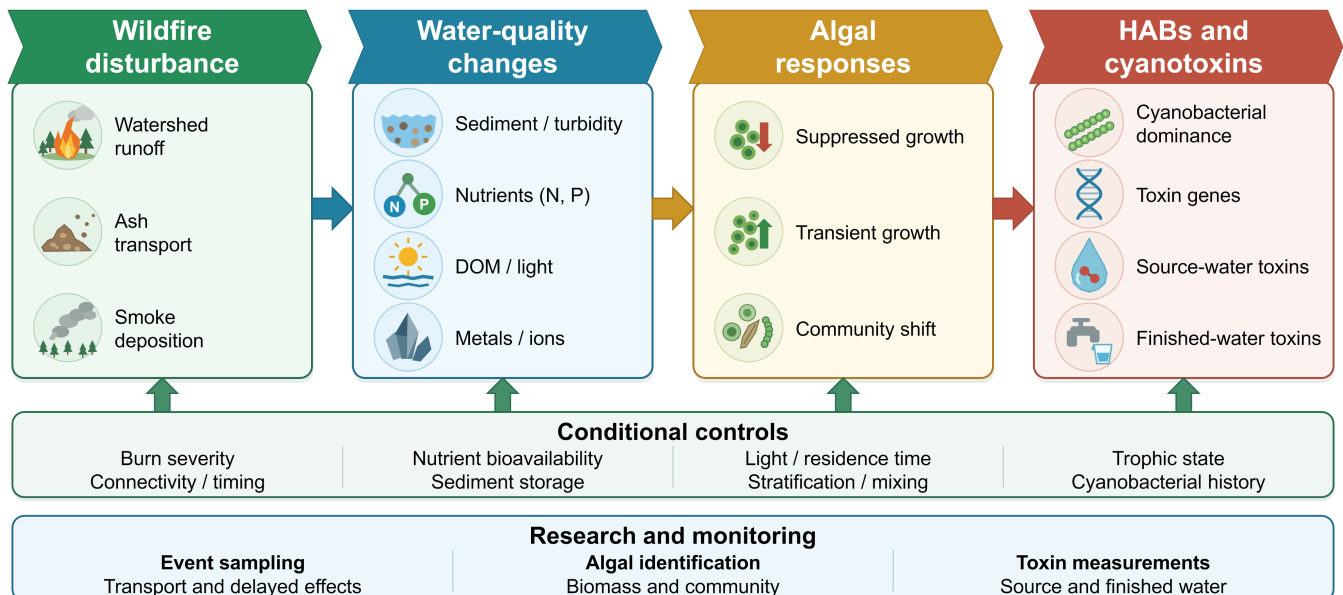


Figure 1. Potential pathways linking wildfire disturbance to cyanotoxin risk in freshwater systems.

2. Wildfire-Driven Changes in Water Quality

The first link in the risk chain, wildfire-driven water-quality perturbation, has the strongest empirical foundation. Across regions and study designs, wildfire can increase sediment transport, turbidity, nutrients, DOM, ions, and metals, with responses shaped by burn severity, rainfall timing, slope, soil properties, vegetation recovery, and hydrologic connectivity [2,8,10]. Post-fire change includes both short transport episodes and persistent watershed legacies. Material stored on hillslopes or in channels may be exported when storms, snowmelt, or debris flows connect burned areas to streams. Some chemical signatures pass within hours or days, while stored material and altered nutrient retention can sustain effects for years [11–13].

The magnitude of these responses can be biologically and operationally relevant. A regional analysis of observations from 1984–2021 in 245 burned watersheds in the western United States, compared with 293 unburned basins, found significantly elevated organic carbon and phosphorus for 1–5 years post-fire and elevated nitrogen and sediment for up to 8 years. In peak response years, average carbon, nitrogen, and phosphorus concentrations were 3–103 times pre-fire levels, whereas sediment concentrations were 19–286

times pre-fire levels [11]. The regional comparison documents large and persistent post-fire water-quality change. It does not resolve how much of the exported nutrient was bioavailable to algae.

The biological effect depends on the form and fate of the transported material. Suspended sediment can limit light and scour benthic substrates, while the same runoff event may deliver dissolved and particle-bound nutrients, DOM, black carbon, and metals. Riverbed sediment may become a post-fire phosphorus reservoir, linking hillslope erosion to later in-channel nutrient release [14]. Fine sediment sampled six and seven years after fire retained substantial phosphorus, showing how sediment storage and exchange with the water column can sustain a downstream nutrient legacy [15]. Post-fire reviews also report increases in nitrogen, phosphorus, dissolved organic carbon, suspended sediment, metals, and disinfection by-product precursors across diverse ecosystems, although the magnitude varies strongly among events and constituents [3,8,10]. Along river continua, downstream responses can be modified by sediment settling, nutrient uptake, tributary dilution, and channel storage [16]. A concentration measured at a single site or time may therefore miss the exposure most relevant to algae or source-water management.

High-frequency monitoring in burned Colorado basins documented turbidity spikes and delayed nitrate responses, showing that post-fire signals can be both event-driven and temporally lagged [17]. Similar work in high-elevation and alpine systems demonstrates that weekly sampling can miss short-duration concentration peaks during snowmelt or rainfall [18,19]. Continuous sensors can resolve stage, discharge, temperature, conductivity, dissolved oxygen, turbidity, and fluorescent dissolved organic matter (fDOM) at subdaily intervals. Laboratory analysis of discrete samples remains necessary for nutrient speciation, metals, cyanotoxin biosynthesis genes, cyanotoxins, and DOM composition [20,21]. Continuous observations can trigger autosamplers and define exposure duration. A brief pulse of bioavailable nutrients may be missed by a seasonal mean, whereas a brief turbidity pulse may temporarily suppress algal growth even while transporting nutrients [6,22].

Post-fire changes in DOM affect ecological conditions and drinking-water treatment. Fire can alter DOM concentration and composition, with implications for light attenuation, microbial metabolism, coagulation demand, and disinfection by-product formation [3,4]. High post-fire sediment loads increase solids-removal demand, while elevated metals can create toxicity concerns [4,5]. Ash and fire retardants can also alter short-term water chemistry, complicating attribution when wildfire-derived materials and suppression activities overlap [23,24]. Whether these chemical changes affect HAB risk depends on the bioavailability of the delivered material and the time available for biomass accumulation.

Persistence varies across systems. Some responses are concentrated in the first post-fire storms, whereas others persist for years because of slow vegetation recovery, channel sediment storage, altered hydrologic pathways, and repeated remobilization. Severe fires can affect water quality at large basin scales [25], and regional analyses suggest multi-year degradation in parts of the western United States [11]. Fourteen years after the Hayman Fire, nitrate remained more than an order of magnitude higher in extensively burned catchments than in unburned catchments, while catchment nitrogen retention remained substantially reduced [12]. Changes in DOM composition and disinfection by-product formation potential have also persisted during slow forest recovery [26]. Persistence also varies among biomes and climates. Boreal catchments can differ from Mediterranean, montane, or savanna systems because organic soils, wetlands, decomposition rates, and fire recurrence influence carbon and nutrient export [27,28]. The variables most commonly measured after fire are not always the variables most directly tied to HAB or cyanotoxin risk. Turbidity and suspended sediment are frequently monitored because they respond strongly and matter for treatment, but they can either suppress algae through light limitation or transport nutrients that

become available later. Total nitrogen and total phosphorus provide useful loading information, but they do not indicate bioavailability, stoichiometry, or the timing of uptake. DOM is relevant to light climate, microbial metabolism, and disinfection by-products, but its ecological effect depends on composition as well as concentration. Post-fire monitoring therefore needs to resolve timing, chemical form, and bioavailability. Turbidity and bulk nutrient concentrations alone cannot determine whether conditions have become more favorable for a HAB [3,21].

3. Algal Responses to Post-Wildfire Water-Quality Shifts

Algal responses are harder to generalize because nutrient enrichment is one component of a broader post-fire disturbance. Nutrient inputs can relieve nitrogen or phosphorus limitation, and canopy loss can increase benthic light availability. Turbidity, sediment deposition, scouring flows, and flushing can instead reduce light, remove biomass, or shorten the time available for growth. The observed response depends on which conditions persist during the relevant growth window [22,29,30].

Stream studies show that wildfire can change algal abundance, community structure, and nutrient limitation [29]. Experimental work using river beads also indicates that severe wildfire can alter algal nutrient limitation and that local physical structure can mediate the response [30]. Nutrient concentration alone did not explain the range of algal responses. Post-fire light, substrate stability, grazing, season, and flow disturbance also influenced whether enrichment increased chlorophyll *a*, shifted community composition, accelerated recolonization, or produced little biomass change.

Long-term stream observations extend this evidence beyond initial disturbance. Severely burned Colorado catchments had greater autotrophic biofilm biomass and primary productivity than unburned streams five and fifteen years after fire. Elevated nitrate reduced nitrogen limitation, although catchment nitrogen supply still exceeded in-stream demand [31]. This result concerns benthic algae in flowing water and does not establish a phytoplankton HAB.

Lakes and reservoirs have different controls. Longer residence time can allow phytoplankton to exploit nutrient pulses, but sediment and colored DOM can reduce light and constrain growth. Sediment resuspension studies show that particles can supply nutrients while reducing underwater irradiance [22]. Reservoirs trap sediments and nutrients, undergo stratification, and often function as drinking-water sources. Available post-fire studies of lakes and reservoirs document changes in turbidity, DOM, nutrients, and thermal or optical conditions more often than matched changes in chlorophyll *a*, taxonomy, or cyanotoxins [32–34]. Hydrologic connectivity, erodibility,

vegetation recovery, and pre-existing trophic state can reduce or delay material delivery to the water column. Retention and stratification provide plausible mechanisms for a delayed response when nutrients and light remain available. The existing evidence does not establish a general post-fire chlorophyll a response.

Smoke deposition can affect lakes outside burned watersheds. Wildfire smoke can transport nutrients associated with fine particulate matter across watershed boundaries and may contribute to downwind cyanobacterial blooms [35]. In one smoke-intersection analysis, the satellite-derived cyanobacteria index increased by a mean of 73.0 across the investigated lakes, with a range of 0.8–409.8. This unitless relative index is scaled from 1 to 1000. It does not measure cell abundance directly. Increases occurred two to seven days after intersection with high-nutrient smoke. The study inferred a dry-deposition pathway. Water-column measurements of deposited nutrients or cyanobacteria were not included [35]. A broader remote-sensing analysis including more than 2000 lakes reported increased algal blooms after wildfire smoke events across the contiguous United States [36]. Smoke can also reduce light and alter heat exchange, so the net response should depend on nutrient deposition, optical effects, mixing, and trophic state [37]. The satellite analyses establish temporal associations between smoke exposure and bloom metrics. They do not measure nutrient delivery, taxonomic change, or cyanotoxin occurrence. Direct tests require paired measurements of deposition, water-column chemistry, algal composition, and cyanotoxins.

Ash-addition experiments show that fire-derived material can stimulate diverse plankton groups [38], and observations during the Thomas Fire documented changes in coastal phytoplankton community composition [39]. Salinity, mixing, food-web structure, and nutrient limitation differ among coastal and freshwater systems, so these observations cannot be transferred directly. They motivate freshwater tests of nutrient release from ash and subsequent changes in plankton composition.

Chlorophyll a is a proxy for photosynthetic biomass and is not taxon-specific. Identifying a cyanobacterial HAB requires evidence that cyanobacteria increased and had sufficient time to accumulate. Cyanotoxin assessment requires targeted chemical or bioanalytical measurements [9,40].

The freshwater evidence is also uneven across algal habitats. Benthic algae in streams respond to substrate disturbance, grazing, canopy loss, and flow events, whereas phytoplankton in lakes and reservoirs respond more strongly to water-column residence time, stratification, mixing depth, and nutrient retention. HAB and cyanotoxin concerns are most often associated with standing or slow-moving waters where biomass can accumulate [6,41]. Stream studies show that wildfire can alter resources and algal communities. Applying those

results to reservoirs requires information on residence time and thermal structure. Lake and reservoir observations are more directly relevant to cyanobacterial accumulation, although many lack high-resolution information on storm-driven nutrient delivery. Studies intended to connect these systems must follow event-scale transport into the seasonal ecology of the receiving water.

Many algal studies report biomass proxies without taxonomic or functional endpoints. Chlorophyll a and satellite bloom metrics are useful screening tools, but they cannot distinguish among diatoms, green algae, cyanobacteria, and mixed assemblages without additional measurements [33,42]. Taxonomic or molecular community data are therefore needed to identify the responding groups and determine whether cyanobacteria increased. A productivity response alone supports altered primary production. A cyanobacterial HAB interpretation also requires cyanobacterial identity and evidence that biomass accumulated.

4. Implications for Harmful Algal Blooms and Cyanotoxin Risk

The link between wildfire-associated algal responses and HABs or cyanotoxins has the least direct evidence. Post-fire changes in nutrients, DOM, light, thermal structure, and residence time overlap with established controls on cyanobacterial blooms. Outside the wildfire context, disturbance-mediated cyanobacterial bloom formation has been documented in reservoir settings [41], and broader HAB literature shows that microcystin distributions depend on nutrient status, cyanobacterial composition, hydrology, and other environmental controls [6,7,9]. The shared controls make a post-fire effect mechanistically plausible. They do not establish wildfire attribution, which requires post-fire measurements of cyanobacterial composition, cyanotoxin biosynthesis genes, cyanotoxins, and exposure-relevant outcomes.

Five endpoints require separate interpretation. These are total algal biomass; cyanobacterial abundance or dominance; genetic potential for cyanotoxin biosynthesis; cyanotoxin production or measured occurrence in source water; and cyanotoxin occurrence in finished water. A productive post-fire stream may support non-cyanobacterial biofilms, and a cyanobacterial bloom may consist largely of nontoxic strains. Detection of cyanotoxin biosynthesis genes indicates genetic potential. It does not demonstrate gene expression or cyanotoxin production. Measured cyanotoxin concentration also depends on genotype abundance, physiological state, cellular release, degradation, and transport [6,40,43]. Source-water concentration cannot predict finished-water occurrence without treatment-performance data [9].

For drinking-water systems, risk assessment must determine whether post-fire conditions increase cyanobacteria or cyanotoxins at the intake and whether

treatment limits cyanotoxin transfer to finished water. Work linking source-water HAB indicators to treated-water cyanotoxin occurrence shows that treatment context and monitoring design shape finished-water detections [9]. This source-to-finished-water framework identifies relevant endpoints for post-fire surveillance. Wildfire attribution still requires matched fire-exposure data. Bloom occurrence is not evidence of cyanotoxin production. Post-fire studies therefore need measurements of community composition, toxigenic genotypes, cyanotoxins, and treatment performance during the same event and subsequent bloom-development period.

Current monitoring designs constrain attribution. General reviews of post-fire receiving waters synthesize chemistry and physical habitat far more often than HAB or cyanotoxin endpoints [3]. HAB monitoring, in turn, is often triggered by visible blooms or recreational concern, which can miss early bloom development, benthic cyanobacteria, or cyanotoxin release after cell senescence. Remote sensing can screen large areas for bloom signals, but satellite chlorophyll *a* or cyanobacteria products cannot determine cyanotoxin presence or congener composition [33,42]. Current evidence supports targeted surveillance in susceptible systems. It remains insufficient to attribute individual cyanotoxin events to wildfire.

Wildfire may also influence cyanotoxin risk indirectly. DOM loading can alter light, oxygen demand, microbial activity, and treatment performance. Sediment pulses can initially suppress photosynthesis but later leave nutrient-rich deposits. In reservoirs, retention, stratification, mixing, and intake location can shape whether these indirect changes affect source-water quality. Post-fire HAB studies have rarely tested these mechanisms directly [4,22,34,41]. Background eutrophication, warming, hydrologic alteration, reservoir management, and pre-existing cyanobacterial communities also shape the response to fire. Reservoir eutrophication assessments show that tributary inputs, hydrologic setting, and existing nutrient status can determine bloom susceptibility before a fire-related pulse arrives [44].

Water-sector operators can use transport and biological indicators to trigger sampling before a cyanotoxin event is confirmed. Connectivity between burned areas and channels, forecast mobilizing events, and observed water-quality change can trigger biological sampling. Chlorophyll *a* indicates photosynthetic biomass, whereas phycocyanin fluorescence or cyanobacterial cell counts provide more specific evidence of a cyanobacterial response. Cyanotoxin biosynthesis genes indicate genetic potential. Measured cyanotoxins document occurrence in source or finished water. Screening programs should use these indicators to trigger confirmatory community and cyanotoxin measurements [9,21].

Attribution is weakest when post-fire blooms are evaluated without pre-fire trophic history, concurrent nutrient and optical measurements, or cyanotoxin data. In

eutrophic or warming lakes with recurring HABs, a bloom after fire may reflect background susceptibility. In other systems, sparse cyanotoxin monitoring may miss a post-fire contribution. Studies can separate a fire contribution from background susceptibility only when pre-fire trophic history and event-resolved transport, chemistry, biological composition, and cyanotoxin data are available [36].

5. Future Research Priorities and Monitoring Strategies

The main research need is to connect post-fire transport, biological response, and cyanotoxin endpoints within the same receiving water. Study designs must also account for differences between rapidly flushed streams and lakes or reservoirs that retain material and support phytoplankton accumulation [6,17,34].

5.1. Research Priorities

Future work should determine when transported material becomes a biologically effective exposure and whether stored material produces delayed effects. Studies should quantify hydrologic connectivity across burn severities and mobilizing events to test whether runoff or snowmelt reaches channels and receiving waters. Burned area alone cannot resolve that transport step.

Studies should compare total event load with the duration of dissolved and exchangeable nutrient exposure after peak turbidity. Brief particle-bound loads may deliver substantial mass without sustaining resource availability. Nutrient speciation should be paired with light, retention, and biological response to identify exposures that alter algal production. Sediment studies should also trace fire-derived particles from deposition to later desorption, mineralization, or release under low-oxygen conditions [14,15].

Future studies should measure algal biomass, cyanobacterial composition, genetic potential for cyanotoxin biosynthesis, cyanotoxins, and treatment outcomes sequentially. A well-powered null result can help locate where the measurable fire response ends, provided sampling spans the relevant response window.

5.2. Event-Coupled Study Design

Event-coupled studies should follow mobilized material through the period in which a biological response could occur. Sentinel stations, archived samples, utility records, and remote-sensing records can provide pre-fire context. Reference catchments or upstream sites provide controls for seasonal change [1,21]. Sampling should capture the rising limb, peak transport, and recession of the first connecting storms or snowmelt events, then continue after suspended solids decline because deposition and biological growth occur on different time scales.

Sampling should include the water column and deposited sediment. Water samples should resolve nitrogen and phosphorus forms, DOM, and suspended solids. Sediment observations should quantify deposited mass, phosphorus fractions, exchange behavior, porewater phosphorus, and redox conditions. Biological sampling should begin with the chemical measurements and continue through the period of possible bloom development. In lakes and reservoirs, vertical profiles should determine whether delivered material is retained and whether bottom-water conditions favor later release.

Models and remote sensing can extend field observations across unsampled times and locations when field measurements provide calibration and confirmation. Burned-area water-quality models remain limited by sparse calibration data and uncertain post-fire hydrologic connectivity [45]. Uncertainty in climate forcing propagates into flow forecasts and can obscure short transport windows [46]. Machine-learning approaches can predict stream temperature and turbidity, but transferability depends on local climate, land cover, burn severity, and hydrologic response [47]. Satellite remote sensing can identify lake optical changes and bloom patterns, but clouds, smoke, mixed pixels, shallow-water complexity, and the inability to detect cyanotoxins limit interpretation [33,42,48]. With field calibration, these tools can help schedule and spatially target sampling during periods of rapid change. Field plans must also account for delayed site access, power and telemetry failures, autosampler capacity, and sample preservation during rapidly changing events.

5.3. Monitoring and Management

Post-fire screening should first establish whether a burned area or smoke plume is connected to a susceptible

receiving water and whether rainfall or snowmelt is likely to mobilize material. Confirmed transport should prompt measurements of nutrient forms, turbidity, DOM, residence time, and water-column structure. A rise in chlorophyll *a* or a satellite bloom metric should trigger community analysis. Increased cyanobacteria or a history of toxic blooms should trigger cyanotoxin measurements. Matched source-water and finished-water samples are needed to assess treatment performance and the potential for exposure [9,21,49].

Each proposed pathway makes a different spatial or temporal prediction. If runoff is the dominant pathway, biological responses should covary with burn severity, hydrologic connectivity, storm timing, and nutrient export. A smoke-deposition response should occur within plume footprints, including lakes outside burned catchments, and should be strongest where deposited nutrients relieve limitation. Sediment storage should produce delayed responses after measurable nutrient release during later stratified periods. These predictions can distinguish the proposed mechanisms from simple co-occurrence [11,34,36].

Agencies and utilities can use these criteria to prioritize monitoring when they cannot sample every downstream water body. Priority sites include receiving waters below high-severity burns when rainfall is forecast, especially where cyanobacteria recur. Satellite screening can cover smoke-exposed lakes with recurring summer blooms even when their catchments did not burn. Reservoirs with long residence times, stratification, or treatment vulnerabilities require monitoring beyond the first storm season [11,21,36].

Table 1 pairs the main uncertainties with testable questions, study comparisons, and the measurements needed for direct tests.

Table 1. Priority questions and study designs for resolving transitions in the wildfire-to-HAB and cyanotoxin risk chain.

Research Gap Testable Question		Study Comparison or Design	Required Measurements
Mobilization and connectivity	Does receiving-water exposure increase when runoff or snowmelt connects severely burned source areas to channels [1]?	Compare burned catchments across severity and hydrologic-connectivity gradients during matched mobilizing events, with upstream or unburned references.	Burn severity and extent, ash or soil composition, precipitation or snowmelt, discharge, suspended sediment, and source tracers.
Bioavailable exposure	Does algal response follow the duration of dissolved and exchangeable nutrient exposure more closely than total event load [10,14]?	Compare events and receiving waters across gradients of nutrient persistence, light, flushing, and residence time. Include null biological responses.	Discharge, suspended solids, dissolved and particulate nitrogen and phosphorus, DOM, light, residence time, and chlorophyll <i>a</i> .
Nutrient stoichiometry and taxa	Do nutrient form and measured limitation predict biomass and community response more consistently than the molar N:P ratio alone [29,30]?	Conduct factorial nitrogen and phosphorus additions across hydrograph phases and habitats, with matched unburned or upstream references.	Nutrient speciation, limitation assays, light, temperature, grazers, chlorophyll <i>a</i> , and community composition.
Sediment legacy	Does stored fire-derived material produce a delayed biological response after measurable release of bioavailable nutrients [14,15,22]?	Track event-derived sediment from deposition through oxic and low-oxygen periods, and compare retained waters with rapidly flushed sites.	Sediment source and mass, phosphorus fractions, porewater phosphorus, exchange behavior, redox conditions, and primary production.

Table 1. Cont.

Research Gap Testable Question	Study Comparison or Design	Required Measurements
Cyanobacteria and cyanotoxins Does comparable fire-derived nutrient exposure produce larger cyanobacterial and cyanotoxin responses in waters with long residence times, favorable light and nutrient conditions, and a pre-fire record of cyanobacteria or cyanotoxins [6,7]?	Compare exposed and reference waters across residence time, stratification, and pre-fire cyanobacterial history, then follow community and cyanotoxin endpoints.	Fire-derived nutrient exposure, residence time, stratification, pre-fire cyanobacterial community and cyanotoxin records, cyanobacterial abundance and taxa, cyanotoxin biosynthesis genes, intracellular and extracellular cyanotoxins, and source-water and finished-water cyanotoxin concentrations.

6. Conclusions

Wildfire can substantially alter the chemistry and physical habitat of fresh waters. Evidence for subsequent algal responses is moderate and system-dependent, and direct evidence connecting wildfire exposure to cyanotoxin occurrence remains scarce [11,31,36].

A defensible attribution of increased cyanotoxin risk requires evidence that fire-derived material reached a susceptible water body, altered conditions relevant to toxigenic cyanobacteria, and preceded cyanotoxin production or occurrence. Evidence from one transition cannot establish the full pathway.

Event-coupled studies that pair storm or smoke exposure with cyanobacterial composition, cyanotoxin measurements, and drinking-water outcomes offer the clearest test of these relationships. Until such evidence accumulates, source-water surveillance should treat wildfire as a conditional amplifier of aquatic risk whose importance depends on post-fire transport, receiving-water susceptibility, and treatment context [9,21].

Author Contributions

Z.C.: Investigation, formal analysis, visualization, and writing of the original draft. X.X.: Conceptualization, supervision, and manuscript review and editing. All authors have read and agreed to the published version of the manuscript.

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No new data were generated or analyzed in this review. The evidence discussed is available in the cited publications.

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Conflicts of Interest

Xiaying Xin is an Editorial Board Member of Aquatic Ecosystems Health. She had no involvement in the peer review of this manuscript and no access to the peer-review process in her editorial capacity. Full responsibility for the editorial process was delegated to another editor of the journal. The authors declare no other conflicts of interest.

Use of AI and AI-Assisted Technologies

During the preparation of this work, the authors used ChatGPT (OpenAI), including Codex, and Claude (Anthropic) to assist with literature searching and manuscript drafting. The authors reviewed and edited the content and take full responsibility for the published article.

References

1. Nunes, J.P.; Doerr, S.H.; Sheridan, G.; et al. Assessing water contamination risk from vegetation fires: Challenges, opportunities and a framework for progress. *Hydrol. Process.* **2018**, *32*, 687–694. <https://doi.org/10.1002/hyp.11434>.
2. Smith, H.G.; Sheridan, G.J.; Lane, P.N.J.; et al. Wildfire effects on water quality in forest catchments: A review with implications for water supply. *J. Hydrol.* **2011**, *396*, 170–192. <https://doi.org/10.1016/j.jhydrol.2010.10.043>.
3. Paul, M.J.; LeDuc, S.D.; Lassiter, M.G.; et al. Wildfire Induces Changes in Receiving Waters: A Review with Considerations for Water Quality Management. *Water Resour. Res.* **2022**, *58*, e2021WR030699. <https://doi.org/10.1029/2021wr030699>.
4. Hohner, A.K.; Rhoades, C.C.; Wilkerson, P.; et al. Wildfires Alter Forest Watersheds and Threaten Drinking Water Quality. *Acc. Chem. Res.* **2019**, *52*, 1234–1244. <https://doi.org/10.1021/acs.accounts.8b00670>.
5. Writer, J.H.; Murphy, S.F. *Wildfire Effects on Source-Water Quality-Lessons from Fourmile Canyon Fire, Colorado, and*

- Implications for Drinking-Water Treatment*; No. 2012-3095; US Geological Survey: Reston, VA, USA, 2012.
6. Beaver, J.R.; Tausz, C.E.; Scotese, K.C.; et al. Environmental factors influencing the quantitative distribution of microcystin and common potentially toxigenic cyanobacteria in U.S. lakes and reservoirs. *Harmful Algae* **2018**, *78*, 118–128. <https://doi.org/10.1016/j.hal.2018.08.004>.
 7. Okello, W.; Kurmayer, R. Seasonal development of cyanobacteria and microcystin production in Ugandan freshwater lakes. *Lakes Reserv. Res. Manag.* **2011**, *16*, 123–135. <https://doi.org/10.1111/j.1440-1770.2011.00450.x>.
 8. Raelison, O.D.; Valenca, R.; Lee, A.; et al. Wildfire impacts on surface water quality parameters: Cause of data variability and reporting needs. *Environ. Pollut.* **2023**, *317*, 120713. <https://doi.org/10.1016/j.envpol.2022.120713>.
 9. Abu-Ali, L.; Albert, R.J.; Davis, A.B.; et al. From bloom to tap: Connecting harmful algal bloom indicators in source water to cyanotoxin presence in treated drinking water. *Harmful Algae* **2025**, *145*, 102846. <https://doi.org/10.1016/j.hal.2025.102846>.
 10. Hampton, T.B.; Lin, S.; Basu, N.B. Forest fire effects on stream water quality at continental scales: A meta-analysis. *Environ. Res. Lett.* **2022**, *17*, 064003. <https://doi.org/10.1088/1748-9326/ac6a6c>.
 11. Brucker, C.P.; Livneh, B.; Rosario-Ortiz, F.L.; et al. Wildfires drive multi-year water quality degradation over the western United States. *Commun. Earth Environ.* **2025**, *6*, 489. <https://doi.org/10.1038/s43247-025-02427-6>.
 12. Rhoades, C.C.; Chow, A.T.; Covino, T.P.; et al. The legacy of a severe wildfire on stream nitrogen and carbon in headwater catchments. *Ecosystems* **2019**, *22*, 643–657. <https://doi.org/10.1007/s10021-018-0293-6>.
 13. Rust, A.J.; Hogue, T.S.; Saxe, S.; et al. Post-fire water-quality response in the western United States. *Int. J. Wildland Fire* **2018**, *27*, 203–216. <https://doi.org/10.1071/wf17115>.
 14. Son, J.H.; Kim, S.; Carlson, K.H. Effects of Wildfire on River Water Quality and Riverbed Sediment Phosphorus. *Water Air Soil Pollut.* **2015**, *226*, 26. <https://doi.org/10.1007/s11270-014-2269-2>.
 15. Emelko, M.B.; Stone, M.; Silins, U.; et al. Sediment-phosphorus dynamics can shift aquatic ecology and cause downstream legacy effects after wildfire in large river systems. *Glob. Change Biol.* **2016**, *22*, 1168–1184. <https://doi.org/10.1111/gcb.13073>.
 16. Reale, J.K.; Van Horn, D.J.; Condon, K.E.; et al. The effects of catastrophic wildfire on water quality along a river continuum. *Freshw. Sci.* **2015**, *34*, 1426–1442. <https://doi.org/10.1086/684001>.
 17. Clow, D.W.; Akie, G.A.; Murphy, S.F.; et al. Dynamic water-quality responses to wildfire in Colorado. *Hydrol. Process.* **2024**, *38*, e15291. <https://doi.org/10.1002/hyp.15291>.
 18. Mast, M.A.; Murphy, S.F.; Clow, D.W.; et al. Water-quality response to a high-elevation wildfire in the Colorado Front Range. *Hydrol. Process.* **2016**, *30*, 1811–1823. <https://doi.org/10.1002/hyp.10755>.
 19. Sherson, L.R.; Van Horn, D.J.; Gomez-Velez, J.D.; et al. Nutrient dynamics in an alpine headwater stream: Use of continuous water quality sensors to examine responses to wildfire and precipitation events. *Hydrol. Process.* **2015**, *29*, 3193–3207. <https://doi.org/10.1002/hyp.10426>.
 20. Akie, G.A.; Clow, D.W.; Murphy, S.F.; et al. An Intercomparison of DOC Estimated From fDOM Sensors in Wildfire Affected Streams of the Western United States. *Hydrol. Process.* **2024**, *38*, e70023. <https://doi.org/10.1002/hyp.70023>.
 21. Murphy, S.F.; Alpers, C.N.; Anderson, C.W.; et al. A call for strategic water-quality monitoring to advance assessment and prediction of wildfire impacts on water supplies. *Front. Water* **2023**, *5*, 1144225. <https://doi.org/10.3389/frwa.2023.1144225>.
 22. Schallenberg, M.; Burns, C.W. Effects of sediment resuspension on phytoplankton production: Teasing apart the influences of light, nutrients and algal entrainment. *Freshw. Biol.* **2004**, *49*, 143–159. <https://doi.org/10.1046/j.1365-2426.2003.01172.x>.
 23. Brito, D.Q.; Santos, L.H.G.; Passos, C.J.S.; et al. Short-Term Effects of Wildfire Ash on Water Quality Parameters: A Laboratory Approach. *Bull. Environ. Contam. Toxicol.* **2021**, *107*, 500–505. <https://doi.org/10.1007/s00128-021-03220-9>.
 24. Crouch, R.L.; Timmenga, H.J.; Barber, T.R.; et al. Post-fire surface water quality: Comparison of fire retardant versus wildfire-related effects. *Chemosphere* **2006**, *62*, 874–889. <https://doi.org/10.1016/j.chemosphere.2005.05.031>.
 25. Emmerton, C.A.; Cooke, C.A.; Hustins, S.; et al. Severe western Canadian wildfire affects water quality even at large basin scales. *Water Res.* **2020**, *183*, 116071. <https://doi.org/10.1016/j.watres.2020.116071>.
 26. Chow, A.T.; Tsai, K.P.; Fegel, T.S.; et al. Lasting Effects of Wildfire on Disinfection By-Product Formation in Forest Catchments. *J. Environ. Qual.* **2019**, *48*, 1826–1834. <https://doi.org/10.2134/jeq2019.04.0172>.
 27. Granath, G.; Evans, C.D.; Strengbom, J.; et al. The impact of wildfire on biogeochemical fluxes and water quality in boreal catchments. *Biogeosciences* **2021**, *18*, 3243–3261. <https://doi.org/10.5194/bg-18-3243-2021>.
 28. Townsend, S.A.; Douglas, M.M. The effect of a wildfire on stream water quality and catchment water yield in a tropical savanna excluded from fire for 10 years (Kakadu National Park, North Australia). *Water Res.* **2004**, *38*, 3051–3058. <https://doi.org/10.1016/j.watres.2004.04.009>.
 29. Klose, K.; Cooper, S.D.; Bennett, D.M. Effects of wildfire on stream algal abundance, community structure, and nutrient limitation. *Freshw. Sci.* **2015**, *34*, 1494–1509. <https://doi.org/10.1086/683431>.
 30. Graziano, A.M.; Fegel, T.S.; Rhea, A.E.; et al. Effects of River Beads on Algal Nutrient Limitation Following Severe Wildfire. *Freshw. Biol.* **2026**, *71*, e70220. <https://doi.org/10.1111/fwb.70220>.
 31. Rhea, A.E.; Covino, T.P.; Rhoades, C.C. Reduced N-Limitation and Increased In-Stream Productivity of Autotrophic Biofilms 5 and 15 Years After Severe Wildfire.

- J. Geophys. Res. Biogeosci.* **2021**, *126*, e2020JG006095. <https://doi.org/10.1029/2020jg006095>.
32. De Palma-Dow, A.; McCullough, I.M.; Brentrup, J.A. Turning up the heat: Long-term water quality responses to wildfires and climate change in a hypereutrophic lake. *Ecosphere* **2022**, *13*, e4271. <https://doi.org/10.1002/ecs2.4271>.
33. Caroni, R.; Pinardi, M.; Free, G.; et al. Investigating the Impact of Wildfires on Lake Water Quality Using Earth Observation Satellites. *Appl. Sci.* **2024**, *14*, 2626. <https://doi.org/10.3390/app14062626>.
34. Nitzsche, N.; Nunes, J.P.; Parente, J. Assessing post-fire water quality changes in reservoirs: Insights from a large dataset in Portugal. *Sci. Total Environ.* **2024**, *912*, 169463. <https://doi.org/10.1016/j.scitotenv.2023.169463>.
35. Olson, N.E.; Boaggio, K.L.; Rice, R.B.; et al. Wildfires in the western United States are mobilizing PM_{2.5}-associated nutrients and may be contributing to downwind cyanobacteria blooms. *Environ. Sci. Process. Impacts* **2023**, *25*, 1049–1066. <https://doi.org/10.1039/d3em00042g>.
36. Olson, N.E.; Brehob, M.M.; Sabo, R.D.; et al. Algal Blooms in Lakes Increase After Wildfire Smoke Events in the Contiguous United States. *Glob. Change Biol. Commun.* **2026**, *1*, e70004. <https://doi.org/10.1002/gcb4.70004>.
37. Farruggia, M.J.; Brahney, J.; Tanentzap, A.J.; et al. Wildfire smoke impacts lake ecosystems. *Glob. Change Biol.* **2024**, *30*, e17367. <https://doi.org/10.1111/gcb.17367>.
38. Ladd, T.M.; Catlett, D.; Maniscalco, M.A.; et al. Food for all? Wildfire ash fuels growth of diverse eukaryotic plankton. *Proc. R. Soc. B Biol. Sci.* **2023**, *290*, 20231817. <https://doi.org/10.1098/rspb.2023.1817>.
39. Kramer, S.J.; Bisson, K.M.; Fischer, A.D. Observations of Phytoplankton Community Composition in the Santa Barbara Channel During the Thomas Fire. *J. Geophys. Res. Oceans* **2020**, *125*, e2020JC016851. <https://doi.org/10.1029/2020jc016851>.
40. Pierce, E.F.; Schnetzer, A. Microcystin Concentrations, Partitioning, and Structural Composition during Active Growth and Decline: A Laboratory Study. *Toxins* **2023**, *15*, 684. <https://doi.org/10.3390/toxins15120684>.
41. Ruhl, N.; Sullivan, N. Disturbance-mediated Cyanobacterial Harmful Algal Bloom (CHAB) formation in a small Ohio reservoir. *Lakes Reserv. Res. Manag.* **2017**, *22*, 161–167. <https://doi.org/10.1111/lre.12172>.
42. Pinardi, M.; Stroppiana, D.; Caroni, R.; et al. Assessing the impact of wildfires on water quality using satellite remote sensing: The Lake Baikal case study. *Front. Remote Sens.* **2023**, *4*, 1107275. <https://doi.org/10.3389/frsen.2023.1107275>.
43. Bittencourt-Oliveira, M.D.C. Detection of potential microcystin-producing cyanobacteria in Brazilian reservoirs with a mcylB molecular marker. *Harmful Algae* **2003**, *2*, 51–60. [https://doi.org/10.1016/s1568-9883\(03\)00004-0](https://doi.org/10.1016/s1568-9883(03)00004-0).
44. Xiang, C.G.; Huang, W.; Wang, Z.W.; et al. Eutrophication risk assessment of tributaries in the Three Gorges Reservoir: A case study of the Pengxi River. *J. Environ. Inform.* **2024**, *43*, 92–104. <https://doi.org/10.3808/jei.202400512>.
45. Basso, M.; Serpa, D.; Mateus, M.; et al. Advances on water quality modeling in burned areas: A review. *PLOS Water* **2022**, *1*, e0000025. <https://doi.org/10.1371/journal.pwat.0000025>.
46. Wu, H.J.; Ye, X.D.; Zhang, B.Y.; et al. Assessment of uncertainty propagation from climate modeling to hydrologic forecasting under changing climatic conditions. *J. Environ. Inform.* **2025**, *45*, 27–41. <https://doi.org/10.3808/jei.202500530>.
47. Chen, J.; Chang, H. Predicting Post-Wildfire Stream Temperature and Turbidity: A Machine Learning Approach in Western U.S. Watersheds. *Water* **2025**, *17*, 359. <https://doi.org/10.3390/w17030359>.
48. Rocha, V.N.L.; Santos, C.A.G.; Silva, R.M. Estimation of water surface reflectance and total suspended solid using MODIS images for a reservoir in the Brazilian semiarid region. *J. Environ. Inform.* **2024**, *44*, 112–125. <https://doi.org/10.3808/jei.202400527>.
49. Elliott, S.M.; Hornberger, M.I.; Rosenberry, D.O.; et al. A Conceptual Framework to Assess Post-Wildfire Water Quality: State of the Science and Knowledge Gaps. *Water Resour. Res.* **2024**, *60*, e2023WR036260. <https://doi.org/10.1029/2023wr036260>.