INFRASTRUCTURE COMMITTEE MEETING

HELD ON THURSDAY, MAY 30, 2024 AT 6:00 PM
COUNCIL CHAMBERS, 400 CENTRE ROAD, LIONS BAY
AND VIA ZOOM VIDEO CONFERENCE

ZOOM INVITE LINK: HTTPS://US02WEB.ZOOM.US/J/2780145720
TO JOIN VIA PHONE, DIAL 778-907-2071 | MEETING ID: 278 014 5720

AGENDA

1. Call to Order

2. Approval of the Agenda

3. Public Questions & Comments

4. Approval of Minutes

A. Infrastructure Committee Meeting Minutes – March 21, 2024 *(page 4)*
B. Infrastructure Committee Meeting Minutes – April 18, 2024 *(page 9)*

5. Business Arising from the Minutes

6. Unfinished Business

Identifier	Description	Responsible	Status
23111	All I.C. members will be provided with a copy of the IMP and the enhanced Asset Management Plan. The document is complicated and requires a dedicated I.C. meeting to fully understand the implications for the Village.	KB/PWM	
23112	Convene a February I.C. Round Table Meeting to focus on a 10 and 20 year horizon plan to identify the new and replacement infrastructure requirements and related expenses.	NTA/All	
23113	CAO and/or Council to be asked to allow members of the I.C. to have selective access to the PW document library.	PWM	
23117	BU will review the SCADA system on behalf of the I.C. and work with the PWM to up-grade the hardware and software. A Requirements Document is anticipated at mid 2024.	BU/PWM	Partial

23121	HM to assist the PWM in preparing REQ/RFP documentation for estimate and work on water main replacement Creekview Place and for the estimate for Highview Place. HM to assist the PWM and CAO in reviewing submissions once received.	HM/PWM	
24011	ASG to approach bigger vendors regarding portable water treatment plants	ASG	
24021	Continue to work with the CAC to prepare a joint recommendation to Council with respect to both short term zone water metering and longer term universal metering.	All	
24022	Assist the PWM/PWD in any way by providing time and expertise in regards to enabling flow from Alberta Creek to be diverted into the Harvey Creek WTP to supplement our potable treated water supplies during the summer of 2024 and beyond.	All	
24041	Once the Village is in receipt of a commissioned report from Carollo Engineers Canada, regarding utilizing Alberta Creek, said report should be reviewed and a recommendation forwarded to Council as how best to proceed with the initiative.	All	
24042	KB to continue his discussions with the drilling companies to determine the full cost of finding sufficient well water supplies, accessing the well sites permanently, providing energy, for pumping etc., classifying the water quality, and treating the water the water if necessary (mineral contamination, GUIDI or GARP classifications).	KB	
24043	KB to continue his discussions with the British Columbia MoE and Ministry of Health to determine the regulatory requirements and possible treatment steps that will be needed to allow any well water supplies to be added to the community water system.	KB	
24044	NTA to contact Staff to gain access to the 2015 water supply and tank fill time data and analysis and allow for joint review of improved real time data in 2024.	NTA	

7. New Business

A. KB: Verbal report on long term additional water supply

B. KB: Verbal report update on ASAP (Alberta Supply Augmentation Project)

C. KB: Verbal update on Water Meters

D. ASG: Emergency WTP availability in the event of a wildfire in our watershed catchments

i. Information Report *(page 15)*
ii. BCWWA – Industry Insights: The Impacts of BC's Wildfires *(page 22)*
iii. Fort McMurray Wildfire *(page 84)*
iv. HealthLinkBC: Wildfire: Its effects on drinking water quality *(page 97)*
v. WRF: Wildfire Impacts on Drinking Water Treatment Process Performance: Development of Evaluation Protocols and Management Practices *(page 100)*

8. Public Questions & Comments

9. Adjournment

10. Next Meeting – June 20, 2024

INFRASTRUCTURE COMMITTEE MEETING
OF THE VILLAGE OF LIONS BAY
HELD ON THURSDAY, MARCH 21, 2024 AT 6:00 PM
COUNCIL CHAMBERS, 400 CENTRE ROAD, LIONS BAY
AND VIA ZOOM VIDEO CONFERENCE

MINUTES

In Attendance:
Mayor Ken Berry – (via conference)
Councillor Neville Abbott – Chair
Committee Member Anthony Greville
Committee Member Brian Ulrich

Absent with regret:
Councillor Jaime Cunliffe
Committee Member Hilary Monfared

Staff:
Public Works Manager, Karl Buhr
Deputy Corporate Officer, Marina Blagodarov (Recorder) (via conference)

1. Call to Order

The Chair called the Infrastructure Committee Meeting order at 18:08.

2. Approval of the Agenda

Moved/Seconded
THAT the agenda of March 21, 2024, Infrastructure Committee be adopted as presented
CARRIED

3. Public Questions & Comments

A. Resident Marek Sredzki offered a comment on the need to put in place a protocol to conserve water in anticipation of a summer drought induced a raw water supply shortage. He had made a similar presentation to Council at its 19 March meeting.

4. Approval of Minutes

A. Infrastructure Committee Meeting Minutes – February 15, 2024

Moved/Seconded
THAT the Infrastructure Meeting Minutes of February 15, 2024, be adopted as presented.
CARRIED

5. Business Arising from the Minutes

6. Unfinished Business

Identifier	Description	Responsible	Status
23087	Locate previous PWM's update report to the IMP indicating the projects that are completed.	PWM	✓
23111	All I.C. members will be provided with a copy of the IMP and the enhanced Asset Management Plan. The document is complicated and requires a dedicated I.C. meeting to fully understand the implications for the Village.	KB/PWM	
23112	Convene a February I.C. Round Table Meeting to focus on a 10 and 20 year horizon plan to identify the new and replacement infrastructure requirements and related expenses.	NTA/All	
23113	CAO and/or Council to be asked to allow members of the I.C. to have selective access to the PW document library.	PWM	
23117	BU will review the SCADA system on behalf of the I.C. and work with the PWM to up-grade the hardware and software. A Requirements Document is anticipated at mid 2024.	BU/PWM	Partial
23118	Assist the PWM in determining the best option for the replacement of the Highway Water Tank. ASG to solicit the expertise of a competent and responsive engineer to allow this project to proceed.	ASG	✓
23121	HM to assist the PWM in preparing REQ/RFP documentation for estimate and work on water main replacement Creekview Place and for the estimate for Highview Place. HM to assist the PWM and CAO in reviewing submissions once received.	HM/PWM	
23122	ASG to assist the PWM in preparing REQ/RFP documentation for estimate for the supply and installation of an alkalinity feed system for each WTP. ASG to assist the PWM and CAO in reviewing submissions once received.	ASG/PWM	✓
23124	Meet with members of the CAC to see if and where the common ground is with respect to water metering in the Village, allowing for possible coordinated recommendations.	NTA/ASG/BU	✓
23125	ASG to find vendors who can supply portable WTP on short notice, in the event of a local wildfire in the watersheds, for rent for up to 36 months.	ASG	✓
24011	ASG to approach bigger vendors regarding portable water treatment plants	ASG	

24021	Continue to work with the CAC to prepare a joint recommendation to Council with respect to both short term zone water metering and longer term universal metering.	All	
24022	Assist the PWM/PWD in any way by providing time and expertise in regards to enabling flow from Alberta Creek to be diverted into the Harvey Creek WTP to supplement our potable treated water supplies during the summer of 2024 and beyond.	All	

23087 The I.C. was notified that such up-dates do not exist and this line item should be struck.

23092 There are no sub-working groups identified and so there is no need to solicit members and so this line item should be struck.

23112 It was agreed the April Meeting would be to consider short and medium term infrastructure projects, and the May meeting will be set aside to consider long term infrastructure requirements.

23118 An engineering firm was contacted, and have subsequently been retained by the Village.

23122 The pH adjustment project has not been approved by Council as a 2024 project, and so this line item can be struck.

23124 ASG and NTA did meet with members of the CAC, and, as a consequence, John Robb will be making a presentation to the I.C. later in this agenda.

23125 ASG reported that he had made contact with three local vendors of portable water treatment plants suitable for the Village needs after a wildfire in our watershed. All three indicated they could meet our needs in terms of custom fabricating such facilities, but they could not provide one for rent on short notice. The recommendation is that the Village approach the leading international water treatment companies who more routinely respond to crisis situations.

7. New Business

A. Confirmation of flow meter accuracy/calibration/fouling at the Harvey Creek WTP.

The PWM reported that the required section for sealing the flow meter after examination had yet to arrive. Further, some of the Works Crew are nervous that a secure seal may not be possible once the system has been opened. The general feeling is that it is unlikely there is significant build up of scale within the line that is causing the flow meter to read incorrectly and this item has lessened in priority.

The PWM noted he has recently fixed a 70,000 usgpd leak within the Village, and that the PWD have determined there is a minimum 200,000 usgpd leak occurring below the Highway Tank. The belief now is that the Highway Tank itself may not be the source of the leak, and the line below the tank is being examined as the most probable source.

It was confirmed that the combination of these two leaks represent over 50% of the winter water demand in Lions Bay and as much as 25% of the peak summer time demand. Overall, the water demand has been reduced significantly since late summer when a focus on leak detection and subsequent fixing was initiated.

B. Water metering – Request for Council

i. Water metering program discussion: https://vancouver.citynews.ca/2024/03/06/bc-water-metering-climate-crisis/

ii. Documents supplied by John Rob: https://drive.google.com/drive/folders/1EAV7dvR12seP9GSz69fnRj7L0HMnHCcJ?usp=sharing

The presentation by John Robb was well received and resulted in a commitment to continue to prepare a joint presentation to Council.

Action Step - Continue to work with the CAC to prepare a joint presentation/recommendation to Council with respect to both short term zone water metering and longer term universal metering. See Item 24021 above.

Action Step – In the interim, make a recommendation to Council that they direct Staff to make an application for some grant monies under the current the program announced in the news article on City News *"In Budget 2024, the provincial government announced $50 million for water metering pilot programs in 21 communities across B.C."*

**** At some point in the meeting, likely during the discussion with the Climate Action Committee, the internet connection with Mayor Berry was lost and therefore the quorum was also lost. In theory, the meeting ceased at the time Mayor Berry left the meeting, however, as the in-person group failed to notice this absence, we did continue with discussions until 19:45.

C. Preparation of a Water Management Plan, physical and communication options, in the event there is a house fire in Lions Bay during the coming summer drought season, or more generally at times of low raw water supply (summer or winter).

The I.C. reviewed and confirmed the presentation made to Council by the PWM on 19 March is valid and we wish to endorse his proposal.

The I.C. believes every effort should be made to be enable to capture the water flow in Alberta Creek and divert said flow into the Harvey Creek in-take or WTP to supplement the flow in Harvey Creek during the anticipated summer source water supply shortage.

Action Step – Assist the PWM/PWD in any way by providing time and expertise in regards to enabling flow from Alberta Creek to be diverted into the Harvey Creek WTP to supplement our potable treated water supplies during the summer of 2024 and beyond. See Item 24022 above.

D. Firefighting/ water use

The I.C. reviewed and confirmed the presentation made to Council by the PWM on 19 March is technically accurate.

E. How can the I.C. assist PW in getting ready for the anticipated drought, and how can we assist PW once we are in the middle of the drought (education/communications from the independent body of experts etc.)

8. Public Questions & Comments

No public comments or delegation were forthcoming.

9. Adjournment

Moved/Seconded
THAT the Infrastructure Committee Meeting be adjourned.

CARRIED

The meeting adjourned at 19:45.

10. Next Meeting – April 18, 2024

The next meeting is scheduled for April 18, 2024.

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Chair

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Corporate Officer

Date Adopted by Committee:	

The Village of LIONS BAY

INFRASTRUCTURE COMMITTEE MEETING
OF THE VILLAGE OF LIONS BAY
HELD ON THURSDAY, APRIL 18, 2024 AT 6:00 PM
COUNCIL CHAMBERS, 400 CENTRE ROAD, LIONS BAY
AND VIA ZOOM VIDEO CONFERENCE

MINUTES

In Attendance:	Mayor Ken Berry Councillor Neville Abbott – Chair Committee Member Anthony Greville Committee Member Hilary Monfared (via conference)
Absent with regret:	Councillor Jaime Cunliffe Committee Member Brian Ulrich
Staff:	

1. Call to Order

The Chair called the Infrastructure Committee Meeting order at 18:15 pm.

2. Approval of the Agenda

Moved/Seconded
THAT the agenda of April 18, 2024, Infrastructure Committee be adopted as amened.
Amendment; to include the addition of the data from 2015 to assist in managing the anticipated 2024 source water supply shortage.
CARRIED

3. Public Questions & Comments

A. Resident Marek Sredzki offered a comment that growth in Lions Bay is inevitable, he cited the possible demolition and development of the townhouse/commercial site, and such growth should not be hampered by a lack of water. He advocated the examination of well water as a possible raw water source to increase raw water supply to the Village.

4. Approval of Minutes

A. Infrastructure Committee Meeting Minutes – March 21, 2024

Due to the concern regarding the lack or a quorum for some of the 21 March meeting, minutes were not presented. The issue was resolved, and the missing minutes will be available for the May meeting.

Moved/Seconded
No vote was taken with respect to the 15 March I.C. Meeting Minutes.

5. Business Arising from the Minutes

6. Unfinished Business

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23121	HM to assist the PWM in preparing REQ/RFP documentation for estimate and work on water main replacement Creekview Place and for the estimate for Highview Place. HM to assist the PWM and CAO in reviewing submissions once received.	HM/PWM	
24011	ASG to approach bigger vendors regarding portable water treatment plants	ASG	
24021	Continue to work with the CAC to prepare a joint recommendation to Council with respect to both short term zone water metering and longer term universal metering.	All	
24022	Assist the PWM/PWD in any way by providing time and expertise in regards to enabling flow from Alberta Creek to be diverted into the Harvey Creek WTP to supplement our potable treated water supplies during the summer of 2024 and beyond.	All	
24041	Once the Village is in receipt of a commissioned report from Carollo Engineers Canada, regarding utilizing Alberta Creek, said report should be reviewed and a recommendation forwarded to Council as how best to proceed with the initiative.	All	
24042	KB to continue his discussions with the drilling companies to determine the full cost of finding sufficient well water	KB	

	supplies, accessing the well sites permanently, providing energy, for pumping etc., classifying the water quality, and treating the water the water if necessary (mineral contamination, GUIDI or GARP classifications).		
24043	KB to continue his discussions with the British Columbia MoE and Ministry of Health to determine the regulatory requirements and possible treatment steps that will be needed to allow any well water supplies to be added to the community water system.	KB	
24044	NTA to contact Staff to gain access to the 2015 water supply and tank fill time data and analysis and allow for joint review of improved real time data in 2024.	NTA	

23112 It was agreed the April Meeting would be to consider short and medium term infrastructure projects and the May meeting will be set aside to consider long term infrastructure requirements.

24011 ASG will attend the up-coming BCWWA Conference in Whistler and has 4 companies to approach, including Veolia, AWC, Bi Pure, and Water n' Waste Tech.

24021 The CAC is currently penning a draft recommendation to Council which will come before the I.C. at our May meeting for review and comment.

7. New Business

A. Discussion around short-term solutions to looming water shortage.

The Village currently does not have a Public Works Manager, and there was no staff representative made available for the meeting.
It was the understanding of the Chair, the major source of water leakage within the Village is now considered to be downstream of the Highway Tank, and likely in the lower section of the Village (and not in the piping immediately below the tank). Work is still being done to confirm the exact locations, as it is suspected there could be several leaks contributing to the losses. The overall non-revenue water losses due to leakage are still being reported to be in the range of 200,000 usgpd, or 50% of overall winter water demand.
The previous PWM believed his findings are that non-revenue water losses of up to 75% of winter demand could be due to leakage, which, if true, would reduce overall consumption to the 300 l/d/p level experienced in Metro Vancouver.

It is possible, if water demand is reduced to 225,000 usgpd, or a third of 2023 demand, then with good management, potable water supply to the residents may not be interrupted during the 2024 summer season. This will require continued vigilance in terms of leak detection and fixing, and an aggressive pubic education programme.

For reference, 1,400 residents × 300 l/p/d = 420,000 l/d = 110,950 usgpd. Current winter demand is close to 400,000 usgpd (with 200,000 usgpd in leakage confirmed to be below the Highway Tank).
The Summer time goal is to establish a daily water demand of 225,000 usgpd (opposite a 2023 consumption rate of 740,000 usgpd). 225,000 usgpd = 850,000 l/d which calculates out to be 600 l/d/p, or twice the Metro Vancouver average.

B. Update Water metering – Request for Council.

The I.C. confirms the most significant benefit associated with water metering is the identification, and subsequent fixing, of water leaks. When considering the above agenda item, it is essential, and somewhat obvious, that water leakage must be properly addressed – water metering is the most important part of the solution.
Water metering installation and operations program costs need to be considered, both costs, recovery through rate structure and grant opportunities be fully articulated to the community. In order for water metering to be successful in Lions Bay, the identified water leaks have to be fixed – again somewhat obvious. And this too involves costs to the residents. If the leaks are on the public side, while the costs will be incurred by the PW budget, ultimately the residents will pay. More significantly, when leaks are detected on the private side (which they will be as original infrastructure ages out past its useful life) then the residents will have to personally pay for the repairs to their property. This is perfectly reasonable, but the cost will be in the $1,000 - $10,000 range per occurrence (in 2024 dollars).

The I.C. supports the proposed new water by-law from a technical perspective as long as the community fully understands what is being asked. Further, the I.C. suggest the water metering should be grant funded as we believe there are other projects that also require access to limited funding. Most certainly the proposed new water by-law should be on the table for discussion within the community.

The I.C. fully endorses the current initiative to proceed with zone metering, and to expand this to include a pilot programme which will focus on the larger water consumers such as commercial, multiplex dwellings and the “large leak repeat offenders”.

C. Options for Alberta Intake.

The I.C. believes using the Alberta Creek water license flow to enhance the Harvey Creek and Magnesia Creek flows during times of low creek flow is workable. A preliminary review suggests water can flow by gravity from the Alberta Creek intake to the Harvey Creek WTP, while the increase in head from the Alberta Creek intake to the Harvey Creek in-take in approximately 3 meters.
For the summer of 2024 the I.C. would recommend Council instructs staff to continue with an aggressive leak detection programme, when appropriate (do not “call wolf” too

early) a technical based appeal to residents to conserve water, good management of the water resource, and finally , the introduction fo Alberta Creek supply into the Harvey Creek in-take as the best option for ensure a safe and continuous water supply.

Action Step – Once the Village is in receipt of a commissioned report from Carollo Engineers Canada, regarding utilizing Alberta Creek, said report should be reviewed and a recommendation forwarded to Council as how best to proceed with the initiative. See Item 24041.

D. Drilling Program – short or long-term?

There was considerable discussion surrounding the option to drill for well water, both within the Village boundaries, and immediately outside on Crown lands.
The Mayor, and one resident, have been in discussions with a mining sector drilling company and have also had contact with the MoE concerning the anticipated raw water supply shortage. Council and I.C. members were not aware of, nor have been made privy to the outcomes, of these meetings.
There was a debate over the urgency of initiating well drilling and whether any flow can be successfully introduced into our potable water system before the August/September anticipated supply shortage deadline.
All agree the mechanical requirements can be reached to drill a test well within the Village boundaries, but questions remain as to the volume of water obtained that could be pumped from one test well, the number of wells that will be needed to make any material difference to our supply, access to the well sites, electrical power and pumping requirements, especially from wells currently on Crown lands, and with the regulatory requirements that will have to be met to allow for the well water to be added to our community water supply.
Well water is clearly an option that has to be considered as it is not as vulnerable to contamination in the event there is a forest fire in our surface water watersheds. Whether or not wells can be put on line in time to meet the 2024 demand is to be determined, but regardless, wells should be properly considered as a potential raw water supply source in the medium term.

Action Step - KB to continue his discussions with the drilling companies to determine the full cost of finding sufficient well water supplies, accessing the well sites permanently, providing energy, for pumping etc., classifying the water quality, and treating the water the water if necessary (mineral contamination, GUIDI or GARP classifications). The cost to drill one test well is much different than the cost to put several wells of fully treated potable water on line. See Item 24042.

Action Step – KB to continue his discussions with the British Columbia MoE and Ministry of Health to determine the regulatory requirements and possible treatment steps that will be needed to allow any well water supplies to be added to the community water system. See Item 24043.

E. Learning for the 2015 experience to manage anticipated 2024 raw water supply shortage.

Discussion centred on learning from the 2015 season, when the Village experienced a similar raw water supply situation. It was recalled flow into the in-takes declined rapidly in July and early August, but at some point in mid August, this decline levelled off and remained reasonably constant until the fall rains arrived. At the time it was determined the water from the saturated ground might have supplied the Village for close to 300 days.

Having this information available to the PWM and I.C. could allow for a similar curve to be drawn in real time in 2024, and based on superior flow monitoring in the Creeks would allow the administration to properly manage our water supply.

If this enhanced water management opportunity can be combined with continued leak detection and fixing, consumer education and conservation and with Alberta Creek possibly available to supplement the Harvey Creek supply, the Village should be able to ensure continued potable water supply all summer and into the early fall.

Action Step – NTA to contact Staff to gain access to the 2015 water supply and tank fill time data and analysis and allow for joint review of improved real time data in 2024. . See Item 24044.

8. Public Questions & Comments

No public comments or delegation were forthcoming.

9. Adjournment

Moved/Seconded

THAT the Infrastructure Committee Meeting be adjourned.

CARRIED

The meeting adjourned at 20:17.

10. Next Meeting – May 16, 2024

The next meeting is scheduled for May 16, 2024.

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Chair

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Corporate Officer

Date Adopted by Committee:	

02 May, 2024.

Information Report to the Infrastructure Committee.

Emergency Water Treatment Plant Options in the event there is a major forest fire in the Harvey Creek and Magnesia Creek watersheds in 2024 or 2025.

Potential Action Steps.

ASG - Continue discussions with the City of West Kelowna to get a better understanding, in real time, of their challenges and the lessons they learn throughout 2024 (after the 2023 fire).

KB - Continue his discussions with the mine drilling companies to determine the full cost of finding sufficient well water supplies, permanent access to the well sites, providing energy, pumping costs etc., classifying the water quality, and treating the water the water if necessary (mineral contamination [Fe, Mn etc.], GUIDI or GARP classifications). The cost to drill one test well is much different than the cost to put several wells of fully treated potable water on line.

KB - Continue his discussions with the British Columbia MoE and Ministry of Health to determine the regulatory requirements and possible treatment steps that will be needed to allow any well water supplies to be added to the existing community water system.

KB & JC – Consistent with the I.C. assigned responsibility criteria, begin the process of determining the best option to secure a long term potable water source for Lions Bay. The options include filtration of the existing three surface water license sources, the drilling of sufficient wells for a bountiful and secure supply of groundwater, or else the construction of an extension pipeline from Metro Vancouver. Also, consider the options for “peak shaving” supply, which will include well water and desalination. The Village needs credible information to make a community wide informed decision, as any option will incur significant cost to the residents.

Council and Staff – Make contact, at the political level, with other small towns, and particularly the First Nations, to pressure the Provincial Government to purchase 6 – 10 portable water treatment plants of various sizes, and have them available for any community that might require one, for a period of up to 12 months free of charge.

Council and Staff – Direct Staff to put in place a contingency plan that will allow the Village to enter, at short notice, into a “rent to own” arrangement with a supplier of portable water treatment plants. This might be a direct rent to own scheme with one of the major international players, or else a combination of assistance from the Provincial Government while a local specialist fabricator assembles a custom designed plant over a 6 - 8 month period.

Staff – Under direction from Council, open negotiations with Veolia (or similar) as a larger supplier of crisis WTP to execute a commercial arrangement whereby, should Lions Bay need a portable WTP, one can be ready for shipment within a week, and commissioned within 4 weeks (before the first heavy rain arrives). Primarily for the 2024 and 2025 fire seasons.

Marc Larivière	Vice President – Municipal Affairs.	
	514 229 5610	marc.lariviere@veolia.com.
Jasmin TamDoo	Regional Sales Manager – Municipal, Western Canada.	
	604 562 0301	jasmin.tamdoo@veolia.com.

Infrastructure Committee – Once the information from all the Action Steps listed above is available, review and make an "informed decision" recommendation to Council as to the best option to manage disturbed surface water supplies in the 1 – 5 year horizon (be that wells for groundwater or a treatment plant for surface water), and to secure a safe, robust and reliable raw water supply to Lions Bay for the longer term.

Discussion.

In the event there is a significant forest fire above the Village, within the Harvey Creek and Magnesia Creek watersheds, we can expect to see a very real impact on the raw water quality in our surface water sources. Due to the absence of any water treatment facilities beyond disinfection, the impact on raw water quality will be such that Lions Bay could not supply potable water that meets the GCDWQ and the Village WTP operating licence regulations for an extended time period, maybe years.

Lions Bay would have to source, install, and commission, a temporary water treatment facility, and perhaps enter into some sort of rent to own commercial contract. Considering the increased number of visitors to our unprotected watersheds, and the number of minor forest fires experienced in 2023, it would be prudent for the Village to put in place an arrangement(s) to allow for the delivery of portable coagulation and filtration technologies as soon as possible.

As previously reported, experience in other jurisdictions tells us the initial impact (1 – 3 months) will be minimal. The immediate short term impact (3 – 9 months) impact would include elevated raw water turbidity levels throughout the first rainy season after the fire, especially during the heavy rains experienced in the first two wet months. The longer term impact (2 – 5 years) would include higher than acceptable TOC/DOC loadings in the raw water which will severely impact any disinfection protocol and cause increased levels of carcinogenic THM and HAA in the treated water. Finally, at 5 years and beyond, as the hydrophobic coating (caused by the fire) on the soil surface begins to break down, high turbidity loading can be expected with any heavy rainfall, regardless of the season.

This scenario described above played out well in 2003 in the Kelowna area after the major Okanagan Mountain Park fire. Similar protocols were experienced in several west coast States in recent years. Please see the attached a recent BCWWA presentation notes.

Locally, and more recently, West Kelowna is beginning to see the effects of the 2023 fire in their region. I was able to speak with their Utilities Supervisor and Water Treatment Manager at the April 2024 BCWWA Conference. They reported a significant increase in turbidity at both their Powers Creek and the newly commissioned Rose Valley WTP last fall. They struggled to maintain potable water quality in facilities designed to treat DOC. As of the end of April, the spring freshet is early, and they are already experiencing higher than normal TOC/DOC loadings in the creeks. I have arranged to meet with them again in the late fall, once their season settles down, for a full debrief and review of the first year after the fire.

While these effects to the raw water chemistry are very real, and in some cases, significant, in all instances where there is a functioning water treatment plant in place, the operators were able to maintain potable water quality at all times. Increased chemical consumption, energy usage, sludge volumes, and in some instances, addition treatment steps were required, but the "job was done."

The problem Lions Bay faces is that we do not have a functioning water treatment plant in place. Lions Bay does have s a functioning disinfection facility and it works well for most of the year. The Ministry of Health has raised questions about the raw water quality during times of heavy rain, and the increased number of summer hikers continues to threaten our supplies from a bacterial and viral perspective. But, in general, we can manage, although the margins are getting slimmer each year!

However, in the absence of any coagulation, flocculation, filtration and possibly organic adsorption protocols, any residual effects of a large forest fire would readily overwhelm our limited facilities and render all "treated water" as being non-potable.

Some degree of TOC adsorption (possibly activated carbon filter, PAC or GAC) chemical treatment, solids separation (ballasted flocculation or membrane filtration) before our VU and chlorine disinfection processes will be required to ensure the Village can supply potable water to our residents on a continuous basis.

Apart from talking with the staff at West Kelowna, I also made contact with several potential vendors of portable water treatment plants. Summary notes are found below. I have known most of the players for many years; I made it clear I had a Village of Lions Bay hat on during our conversations and that as a member of the I.C. all I can do is advocate for a recommendation to Council to instruct Staff to open some sort of commercial discussion. However, the contacts have been made, and hopefully Staff will receive a positive directive.

Veolia.

As expected from previous inquiries, Veolia proved to be the most promising option. I met with Marc Larivière, Vice President – Municipal Affairs and Jasmin TamDoo, the local Regional Sales Manager for municipal business.

It would appear as though Veolia can help us, as they do have units available for such purposes, but based mainly in the Montréal - Ottawa – Toronto triangle. Marc seemed very keen to be of assistance (maybe we might be a beachhead in western Canada). His feeling is that for a 5 year term it would be much better to purchase rather than rent, as over the 5 years, we would easily pay for the unit. A rent to own option could be negotiated, as could some sort of "contingency agreement" whereby Lions Bay would have first option to receive a portable WTP unit should we had an on-going commercial agreement in place.

The technology we discussed would be similar in concept for all vendors, although Veolia does have a patent and are market leaders in the ballasted flocculation/sedimentation (Acti-Flo). They would propose coagulation, flocculation, ballasted sedimentation, likely employing both silica sand and PAC, and finally conventional filtration. The filtered water would be fed into our existing disinfection process and distribution system.

Other Local Suppliers.

The difference between Veolia and the other local vendors is the Acti-Flo process, which is proprietary (but well established in B.C. and Aberta) would be replaced with pre-filtration with either PAC or GAC, followed by UF or MF membranes. Otherwise, coagulation and flocculation are mandatory, and conceptually the processes are doing the same thing is a slightly different way.

The local suppliers I spoke with were mainly long time business associates who were honest and gave me straight answers. The companies include

AWC Solutions	John Sainas	Vice President – Business Development
	778 558 5994	johns@awcsolutions.com.
Bi Pure Water	Jonathan Boughen	Business Development Manager
	604 882 6650	jonathanb@bipurewater.com.
Waste n' Watertech	Dominic Janssen	President
	403 616 4498	dominic@watertech.ca

The main thrust of the local suppliers is that all of them (and others) are more than willing to build Lions Bay a portable water treatment plant that will meet our needs. That is not the issue. The issue is that the smaller companies will readily build a plant given a 4 – 6 month lead time (so really 8 months!), but none will have a fully constructed plant built on spec sitting in their warehouse looking for a home.

This is the primary reason we should be talking to Veolia (or similar) in preparation for the 2024 and 2025 fire seasons. If, by a miracle of nature, we are able to drill test wells, find sufficient volume and put in place all the regulatory requirements to allow well water to be the primary raw water source for Lions Bay for 2024 and/or 2025, then this too becomes a viable option.

Perhaps the most enlightening comments, from both AWC and Bi Pure, is that they, independently, have had preliminary discussions with smaller Frist Nations communities, and the Provincial Government, proposing that the Government inventory 6 – 10 portable WTP of various capacities to have available in times of emergency.

It would appear as though these discussions have somewhat died on the vine, and have not got the full attention of Government. Maybe this is something for Lions Bay to re-invigorate, and include other smaller non First Nation communities as well as senior government agencies such as ISC.

Clean potable water for Indigenous communities and other smaller towns is a positive and well received topic, and some sort of Provincial Emergency Plan surrounding treating forest fire compromised surface waters would be a legacy for the individual, community or group, that can spearhead such a programme. Worth a try for sure.

There is a small group of communities and private companies who are talking about such an initiative, and Lions Bay could do worse than become a part of the solution – not to mention we might be an initial beneficiary should such a concept become a reality.

Plenty to discuss in regards to being prepared to supply potable water to the residents of Lions Bay in the months and years after any fire of a substantial size occurs in our local watersheds.

Even more importantly, actions must be taken to ensure Lions Bay is ready, not only for the immediate duration of any fire, but to ensure the long term sustainability of the Village well after the fire has been extinguished and the cameras have moved on!

To save you reading all 122 pages of the attached Water Research Foundation 2018 paper entitled *Wildfire Impacts on Drinking Water Treatment Process Performance : Development of Evaluation Protocols and Management Practices,* here are a few sections (the last section is probably the most significant for this discussion);

The most apparent influence of wildfires on watersheds is an altered hydrologic regime characterized by flashy events and subsequent sediment transport. Effects can include increased total runoff volume, increased peak flow, flooding, and increased sediment mobilization (Sham et al. 2013). A wildfire can result in loss of vegetation, decreased soil infiltration (higher soil hydrophobicity), and decreased evapotranspiration, all contributing to the effects listed above. Consequently, erosion and sediment mobilization can be elevated post-fire, particularly during storm events and when discharge is greatest in the first year following a wildfire (Silins et al. 2009).

From a water supply and treatment perspective, these changes can greatly influence total suspended solids (TSS) and turbidity levels. Rhoades et al. (2011) found turbidity levels were four times higher in a basin with high severity burn compared to basins that were not burned as severely. Other research has shown elevated turbidity levels in wildfire impacted waters can reach extreme values under specific hydrological conditions (e.g., heavy rainfall, rapid snowmelt) (Emelko et al.

2011, Writer et al. 2012). The elevated level of TSS is a concern to utilities, which are restricted regarding the turbidity levels that are permitted for potable water. It has been shown that the enhanced turbidity associated with a rain event following a wildfire can be effectively coagulated, however the required coagulant doses are, as expected, higher (Hohner et al. 2016). The higher coagulant doses may be beyond what a WTS could handle and will also result in increased costs for disposal of material.

One large research gap regarding the short and long-term effects of a wildfire on water quality is in the effect on the mobilization and changes to the chemical properties of dissolved organic matter (DOM) after a wildfire. ……… Wildfire changes to the landscape and forest floor may alter the quantity and quality of DOM in source waters. ……… Elevated DOM levels, commonly quantified as dissolved organic carbon (DOC - mgC/L), following wildfires have been documented (Emelko et al. 2011, Hohner et al. 2016, McEachern et al. 2000, Minshall et al. 2001), while others found minimal effects on DOC (Mast and Clow 2008). It is suggested that background DOC concentrations may dampen the contributions from wildfire (Smith et al. 2011), emphasizing the importance of local hydrology, burn severity, and extent, when interpreting effects on water quality.

Although DOM is naturally occurring, it poses a concern in source water supplies because it can react with disinfectants (e.g., chlorine, chloramines) to form disinfection byproducts (DBPs) as an unintended consequence of water treatment (Christman et al. 1983, Rook 1977, Stevens et al. 1976). DBP exposure has been linked to cancer risks and reproductive developmental effects (Muellner et al. 2007; Plewa et al. 2004; Richardson et al. 2007; Villanueva et al. 2004, 2006).

There are limited reports on the effects of wildfires on source water quality coupled with the water treatment process performance (Emelko et al. 2011, Hohner et al. 2016, Hohner et al. 2017, Majidzadeh et al. 2015, Tsai et al. 2015, Wang et al. 2015a, Wang et al. 2015b, Writer et al. 2014). Several lab-based leaching studies of material burned in wildfires have been conducted to assess the effects on DOM and drinking water treatment (Revchuk and Suffet 2014, Wang et al. 2015a). In a leaching study of material burned in California wildfires, a 10-fold increase in the concentration of DOC leached was observed when compared to unburned material (Revchuk and Suffet 2014). A study evaluating water extractable material of burned detritus from the Rim Fire revealed elevated N-DBP reactivity (HAN and NDMA) compared to unburned detritus (Wang et al. 2015b). Other work has shown that waters impacted by wildfire could be effectively treated by coagulation (Emelko et al. 2011).

Following the High Park wildfire, the City of Fort Collins was forced to shut down the water intake along the CLP River due to high sediment loads that posed a threat to water conveyance infrastructure. Fort Collins relied solely on an alternate water supply for 100 consecutive days.

Previous work conducted immediately following the wildfire showed evident post-fire effects on CLP water quality (turbidity, TP, TN, TOC) (Writer et al. 2014). Post-fire changes in source water quality were most pronounced following rainstorms in the fire-affected area of the watershed.

Wildfires and other extreme weather events present great challenges and risks to water utilities throughout the United States. In many cases, such events have caused treatment plants to either shut down, reduce flow, or deliver water that was of inferior quality or failed to meet regulations. These risks are due to changes in the frequency, duration, magnitude, and speed of raw water quality changes, specifically increases in turbidity and NOM. As presented in previous chapters, wildfires often result in increased solids loadings to plants in terms of ash content and runoff from soils due to loss of groundcover, and increased transport of terrestrial NOM of altered character. These two parameters – turbidity and NOM – drive the design and operation of water treatment plants. Other water quality parameters that can influence process selection and plant operations after a wildfire include algae, iron, manganese, and taste and odors.

Treatment challenges expected from wildfires include both short term and long-term issues. Short term issues that may occur in the weeks and months after an event are mostly due to increased turbidity due to ash from post-fire erosion, spikes in NOM, and pH and alkalinity changes. Longer term issues that can occur over several years include:

- Increased turbidity from soils due to loss of vegetation – this can continue for years,
- Increased NOM,
- Changed character of NOM,
- Algal growth and associated byproducts: algal toxins and taste and odors that result from release of nutrients after a fire and warmer temperatures due to loss of shade cover,
- Iron and manganese.

BC Water and Waste Association

Industry Insights: The Impacts of BC's Wildfires

Preparing for Impacts of Wildfires on Source Water

Stephen Horsman, PEng, PE, PMP

Alan Domonoske, PE

Jude Grounds, PE

November 29th, 2023

Drinking Water Community of Practice

- Robyn Casement (Associated Engineering) and Megan Wood (Metro Vancouver)
- Looking for willing, able and interested water professionals looking to engage and collaborate on topics relevant to the water industry
- Contact Angelene Fellows
  - 604-630-5346
  - afellows@bcwwa.org

Introductions

Stephen Horsman, Peng, PE, PMP

Regional Lead, Associate Vice President

Jude Grounds, PE

Water Practice BD Manager, Senior Vice President

Alan Domonoske, MSc PE

Senior Project Manager, Vice President

Agenda

1. History of Wildfires in BC
2. Impacts of Wildfires on Watersheds
3. Case Study Series 1: Southwest
4. Case Study Series 2: Cascadia
5. Best Practices to be Fire Ready
6. Question & Anwser

01

History of Wildfires in BC

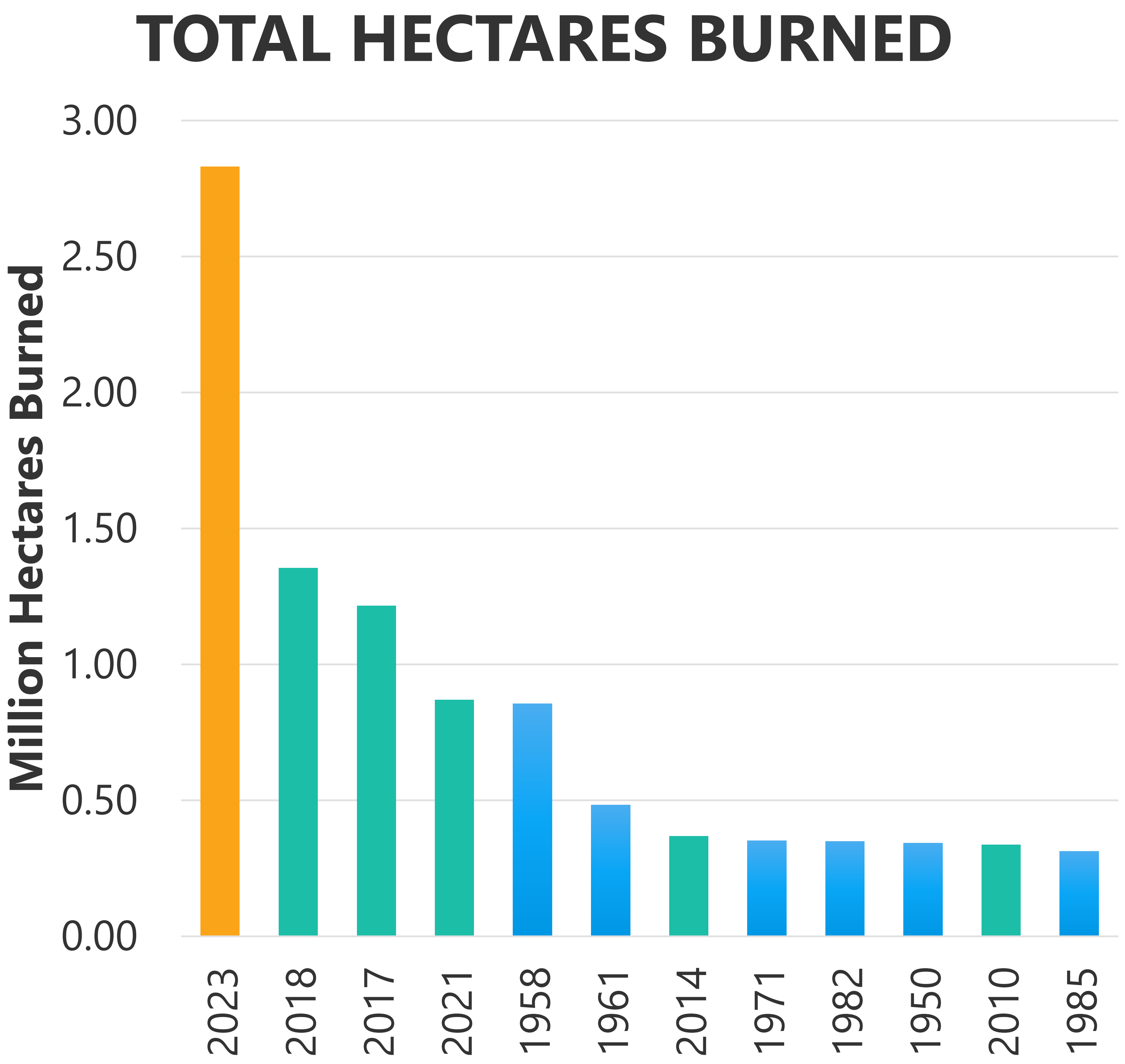
Responses can be wildfire and watershed specific

BC Watersheds and Wildfires

- Worst wildfire seasons on record (2003, 2009, 2017, 2018, 2021, 2023)
- Over 70% of our drinking water supplies in BC rely on surface water sources
- 2003, 2017, 2018, and 2023 wildfires threated Okanagan, Vancouver Island and Central BC drinking water supply watersheds

2023 Wildfire Season

- Worst wildfire season on record (circa 1917)
- Significant wildfires near:
  - » Fort St John
  - » West Kelowna and Kelowna
  - » Shuswap and Adams Lake
- Donnie Creek wildfire represented the single largest wildfire on record



May 1, 2023

–

▼ Overlays

Base map

GeoGratis

☐ Fire Weather Index ⓘ

☐ Fire Danger ⓘ

☑ Fire Perimeter Estimate ⓘ

Season-to-date buffered hotspots (> 1000 Ha)

☐ Fire M3 Hotspots ⓘ

☐ Season-to-date Hotspots ⓘ

☑ Active Fires ⓘ

Out of control
- 0 to 100 Ha
- 101 to 1000 Ha
- > 1000 Ha

Being held
- 0 to 100 Ha
- 101 to 1000 Ha
- > 1000 Ha

Under control
- 0 to 100 Ha
- 101 to 1000 Ha
- > 1000 Ha

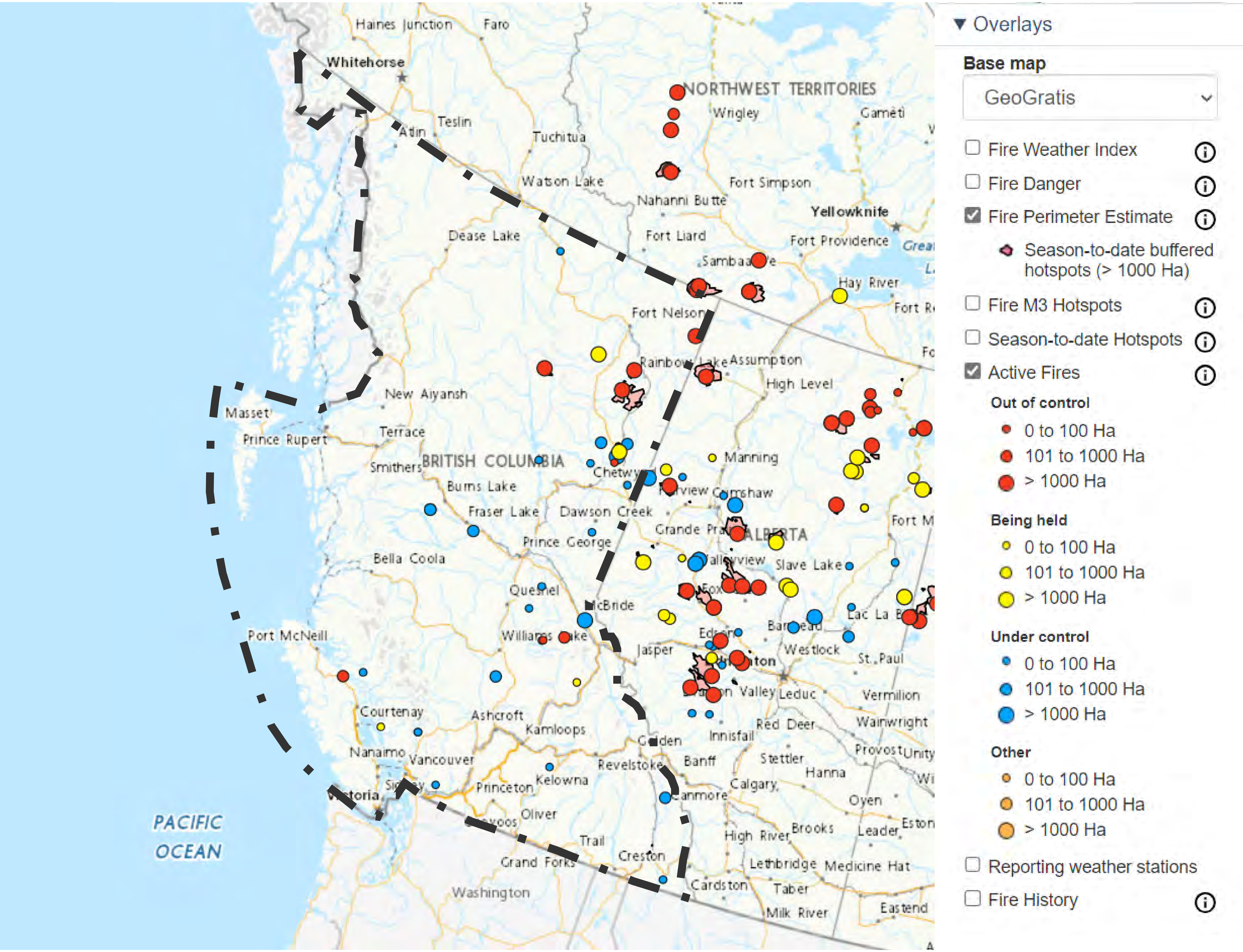
Other
- 0 to 100 Ha
- 101 to 1000 Ha
- > 1000 Ha

☐ Reporting weather stations

☐ Fire History ⓘ

June 1, 2023

–



▼ Overlays

Base map

GeoGratis

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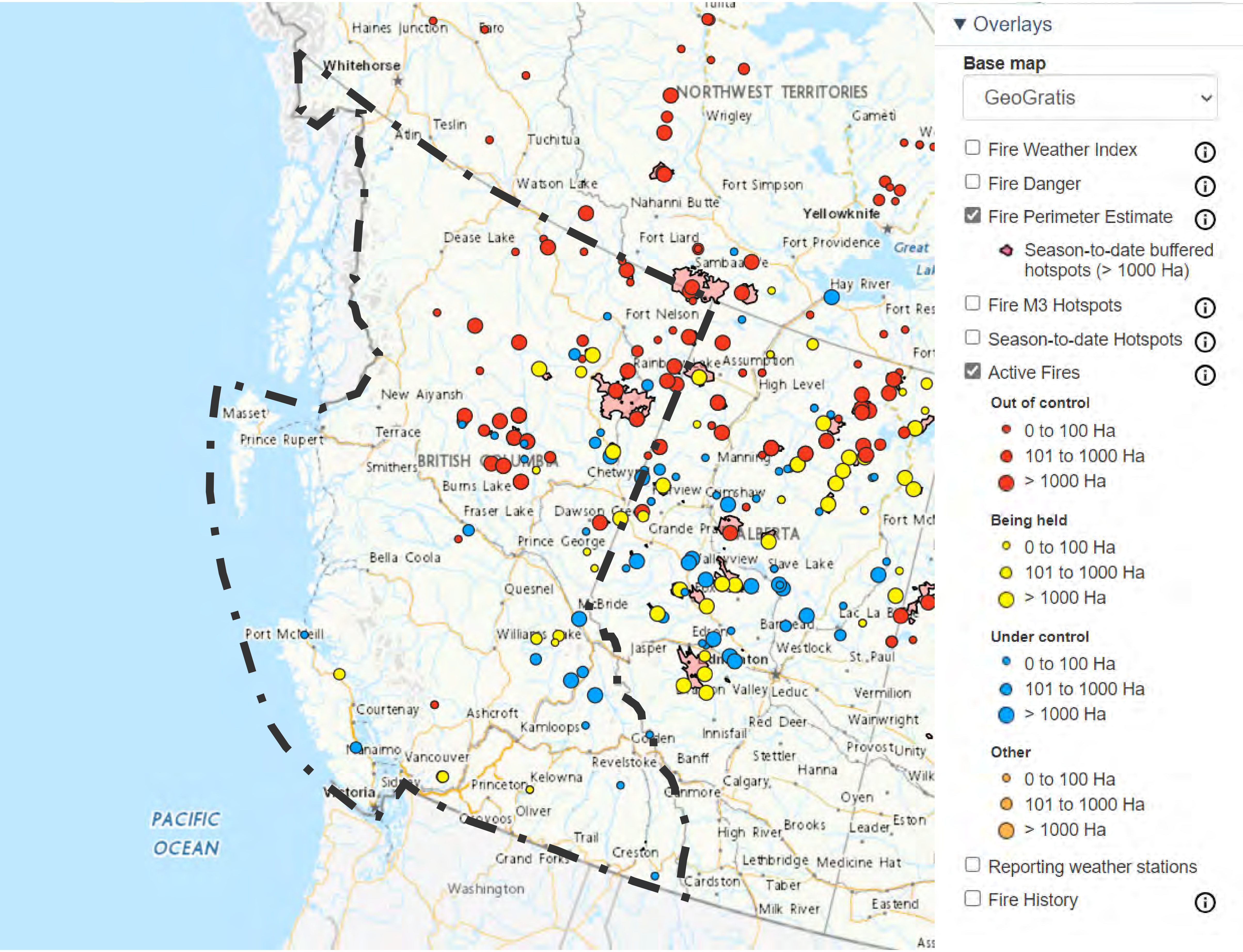
- > 1000 Ha

☐ Reporting weather stations

☐ Fire History ⓘ

July 1, 2023

–



▼ Overlays

Base map

GeoGratis

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- ● > 1000 Ha

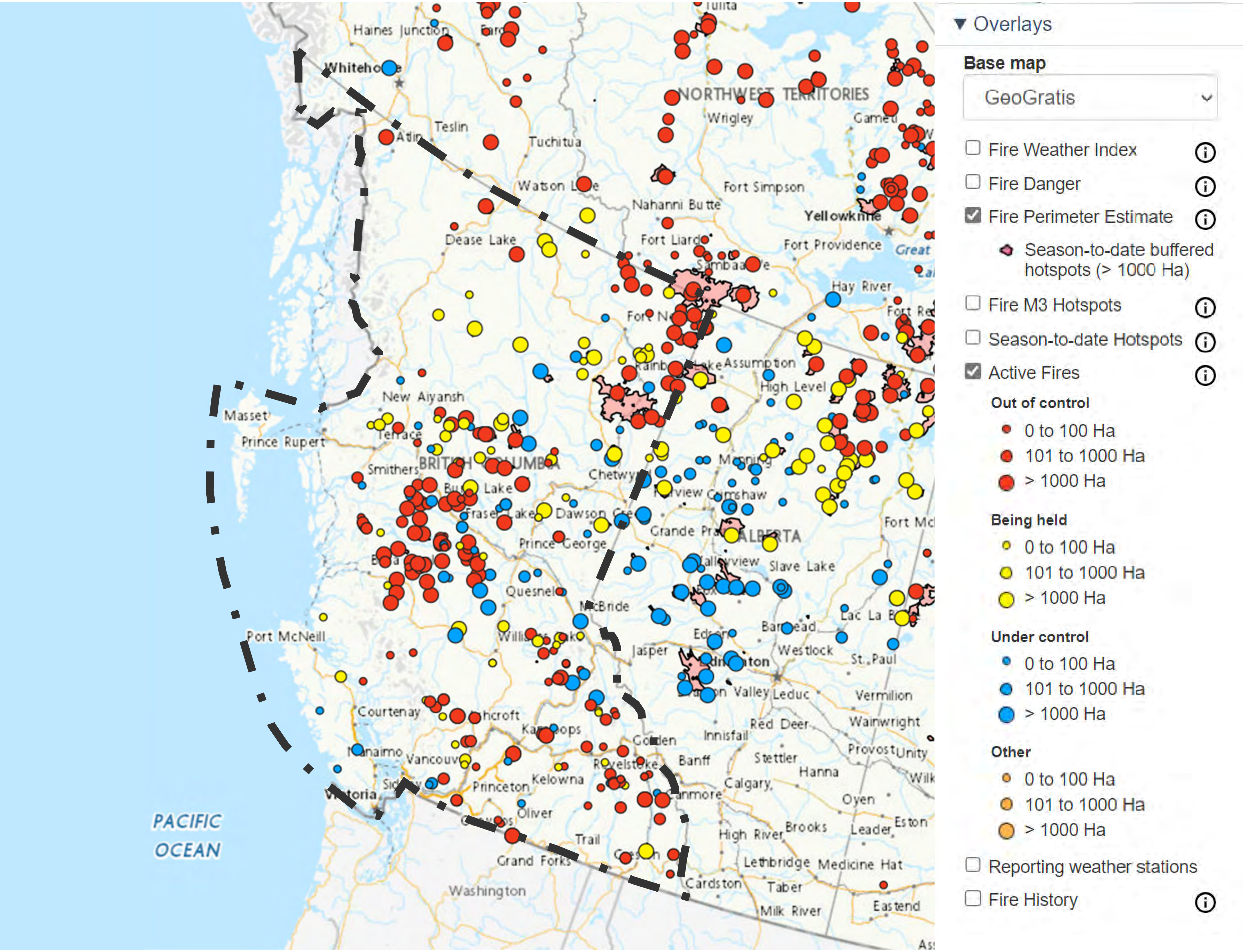
Other
- ● 0 to 100 Ha
- ● 101 to 1000 Ha
- ● > 1000 Ha

☐ Reporting weather stations

☐ Fire History ⓘ

Aug 1, 2023

—



▼ Overlays

Base map

GeoGratis

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Season-to-date buffered hotspots (> 1000 Ha)

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Under control

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- 🔵 > 1000 Ha

Other

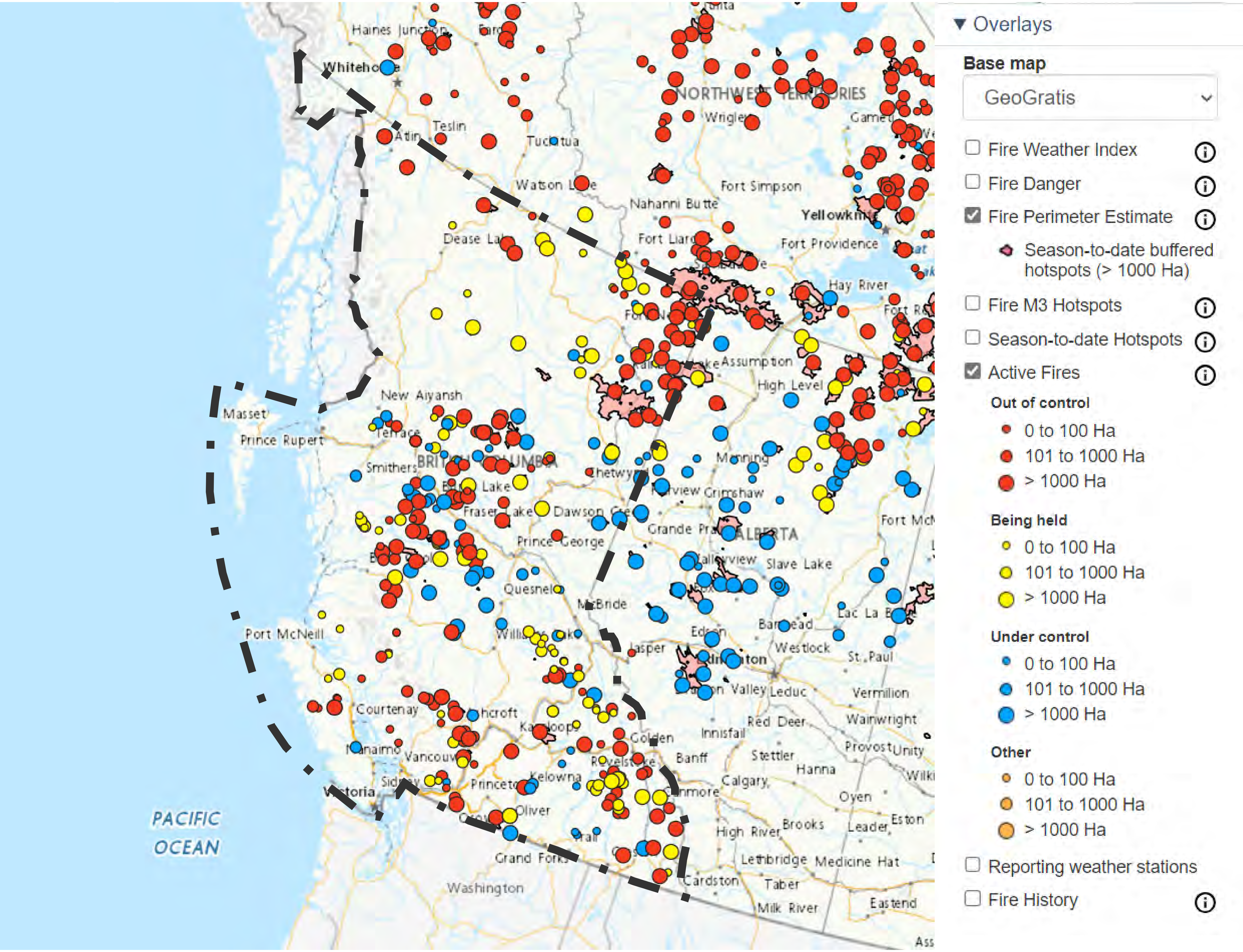
- 🟠 0 to 100 Ha
- 🟠 101 to 1000 Ha
- 🟠 > 1000 Ha

☐ Reporting weather stations

☐ Fire History ⓘ

Sept 1, 2023

–



▼ Overlays

Base map

GeoGratis

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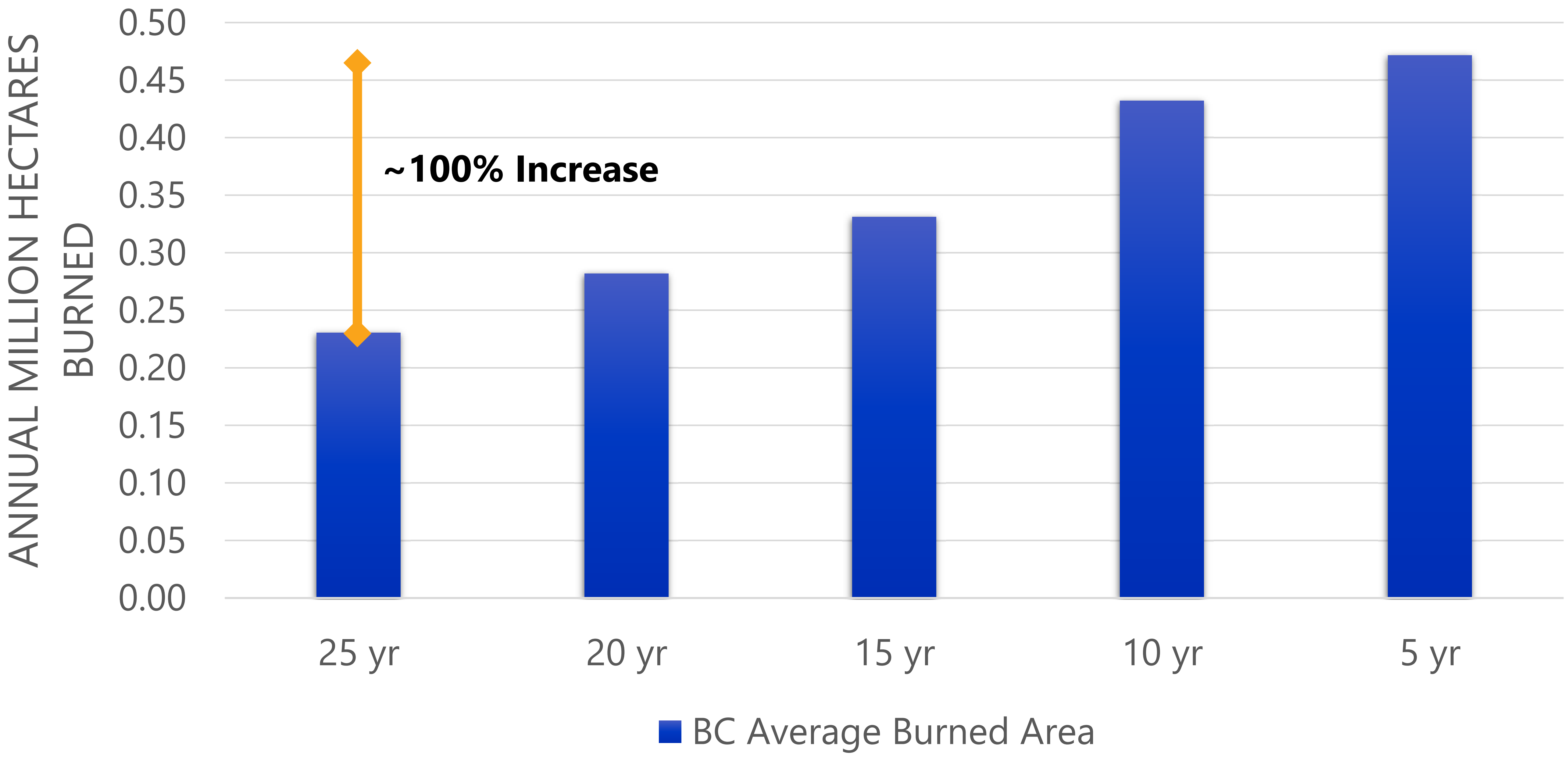
- ● > 1000 Ha

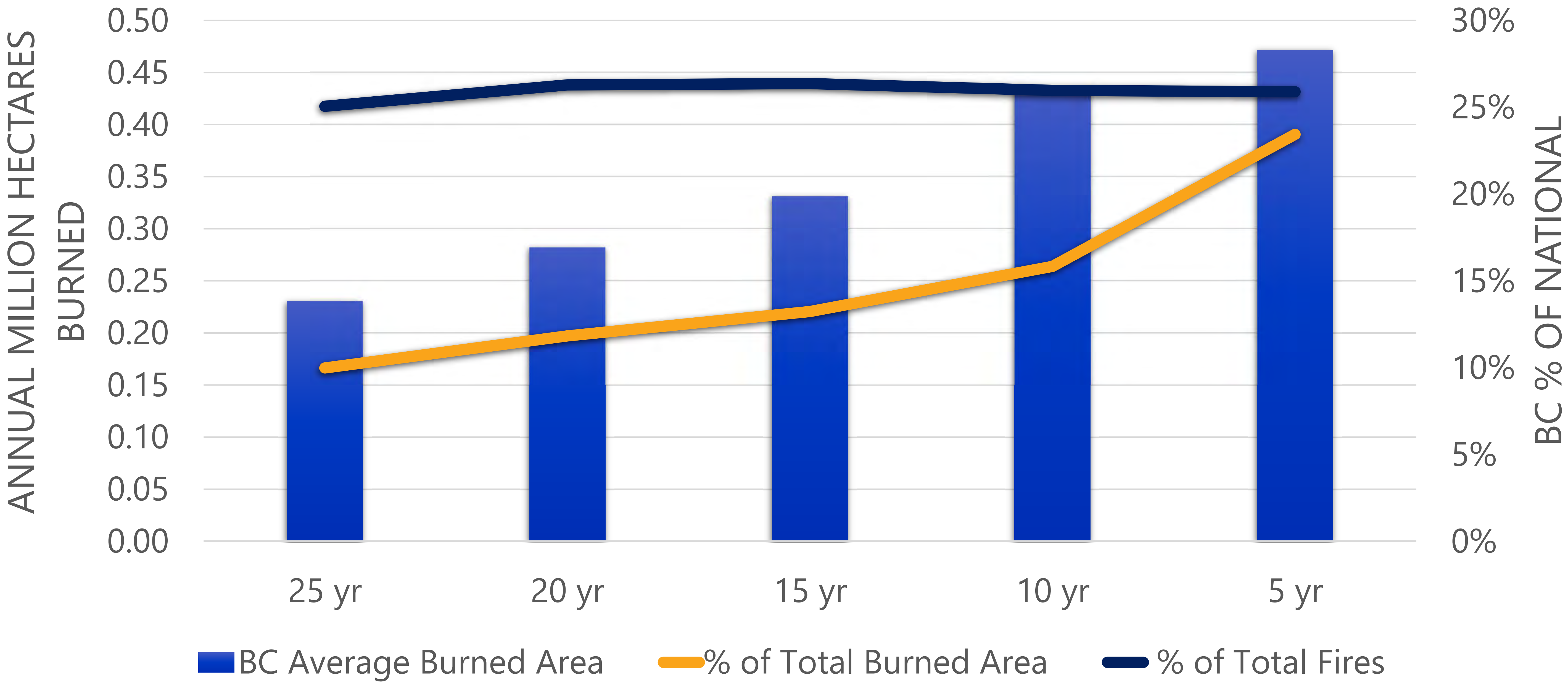
☐ Reporting weather stations

☐ Fire History ⓘ

British Columbia Fire Trends over 25 years

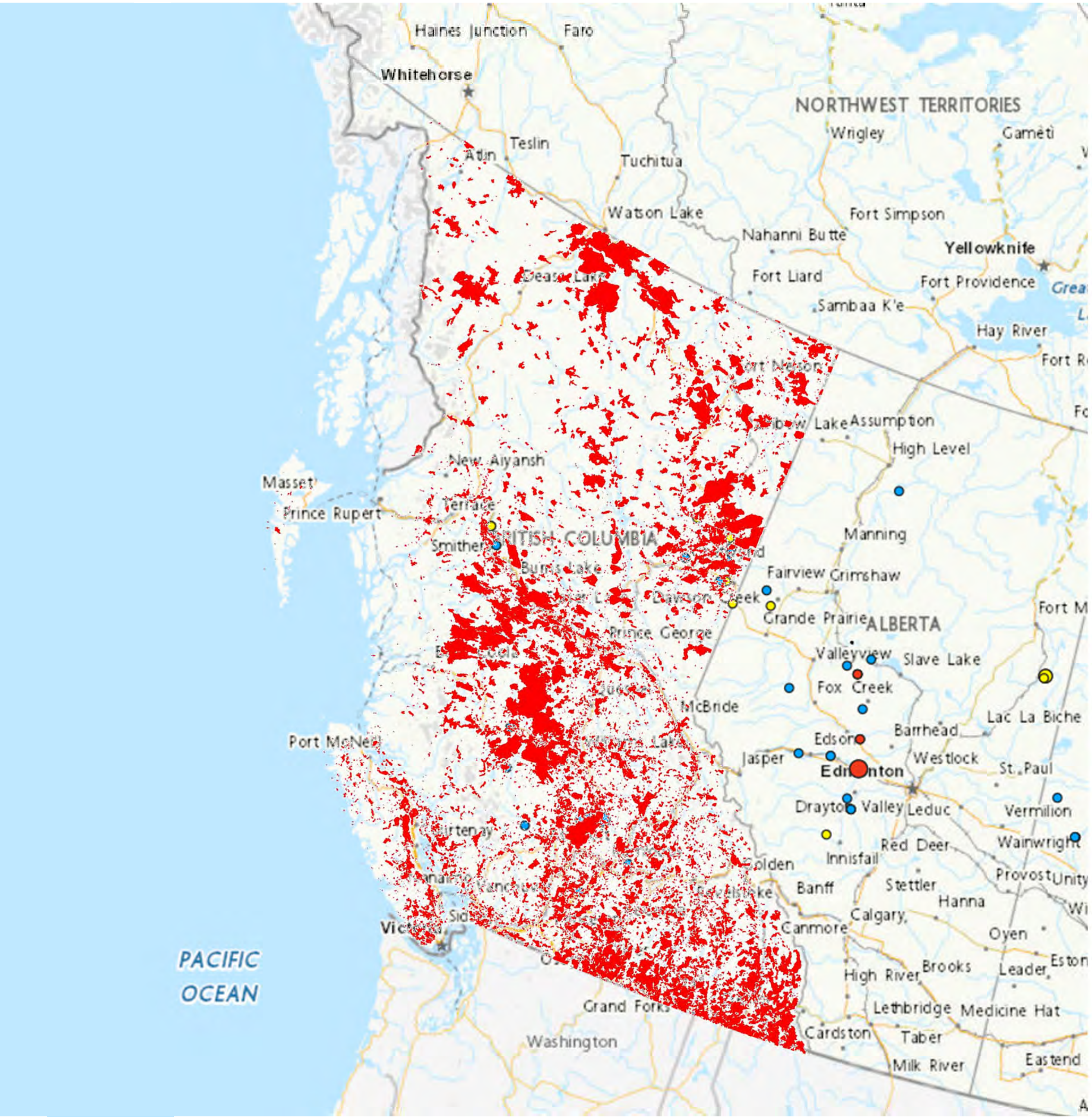


Comparing British Columbia to National Fire Trends



Historical burn area since 1917

- Significant coverage throughout the province
- Affects mange utilities
- Shows the resilience of nature watersheds to dampen

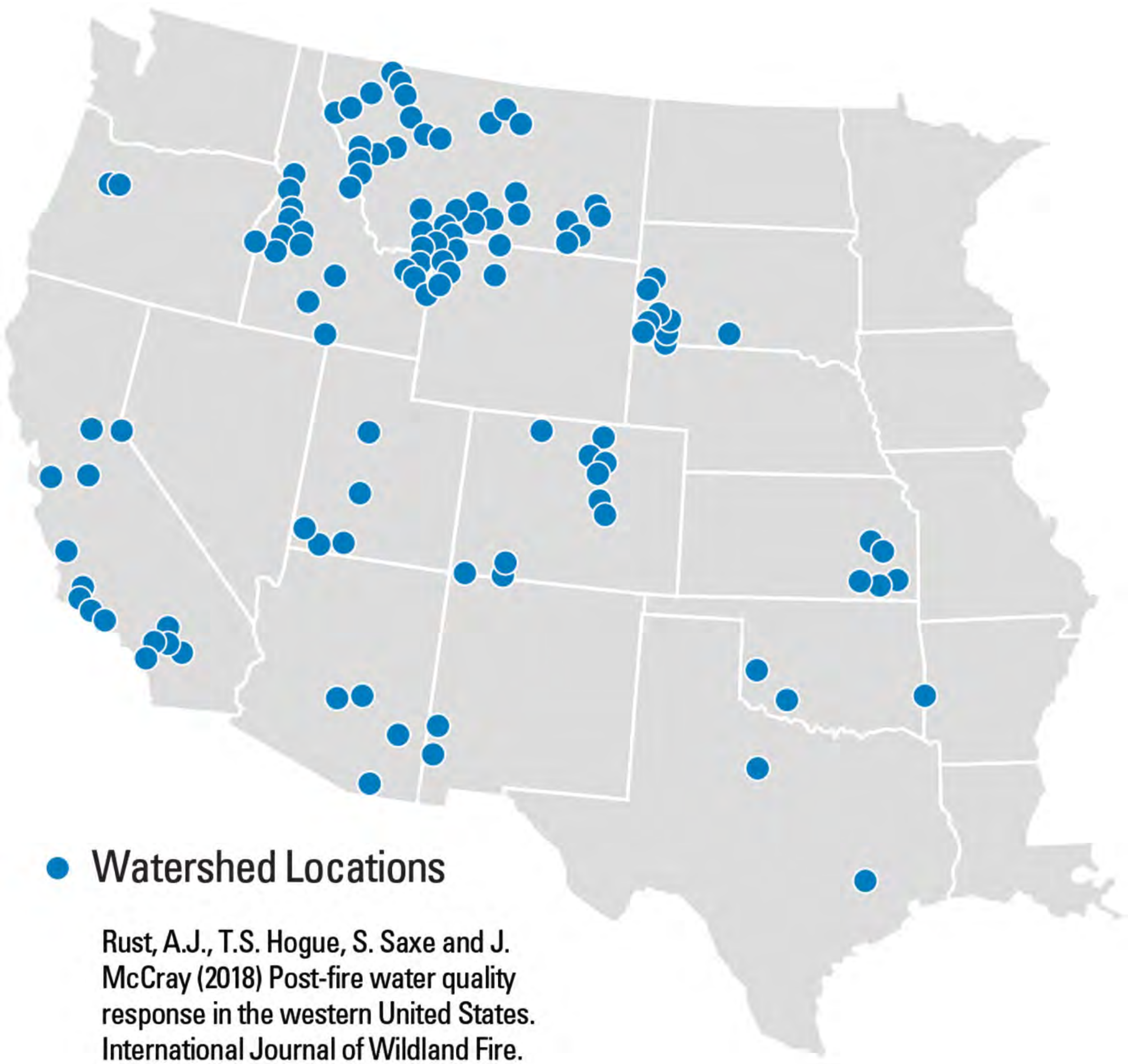


02

Wildfires and Watersheds

Responses can be wildfire and watershed specific

Western US Wildfires Had Three Common Responses



Rust, A.J., T.S. Hogue, S. Saxe and J. McCray (2018) Post-fire water quality response in the western United States. International Journal of Wildland Fire. Doi.org/10.1071/WF17115.

Most common water quality responses

- ● Sediment / Turbidity
- ● Organics / Nutrients
- ● Metals

Increase in base and peak flows (3-5 years)

Most watersheds recover (hydrology and WQ) within 5 years

Factors with strong correlations to water quality

Soil Organic Content

Wildfire Severity

Postfire plant cover

Water Quality

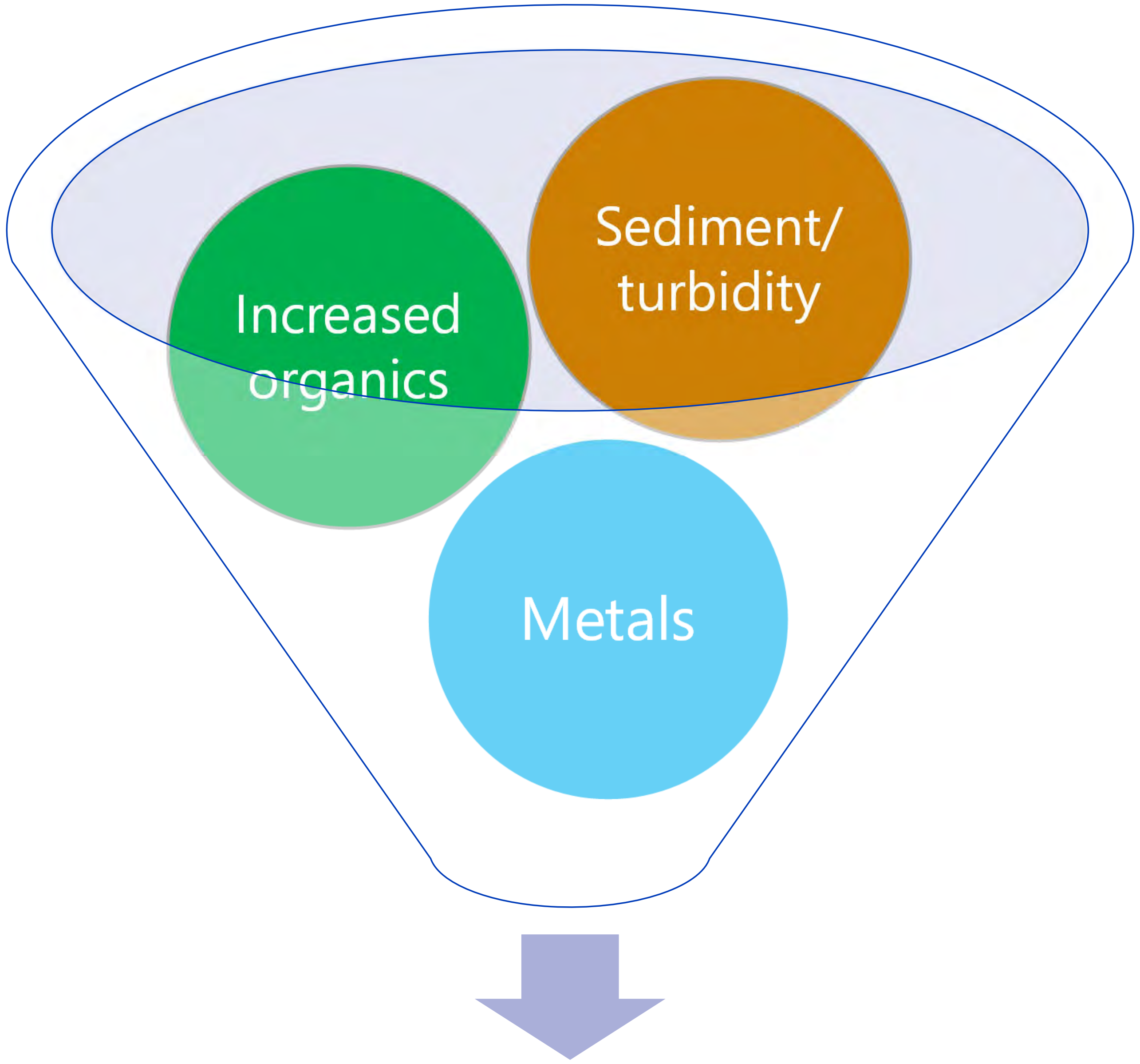
Water quality impacts are uncertain

A tale of two fires….

Percent change
(One year post-fire)

+++	> 1000%
++	100% -- 1000%
+	0-100%
–	-100% - 0%

Constituent	Salt River	Verde River
Total Organic Carbon	++	
Dissolved Organic Carbon	++	+
Total Phosphorus	+++	–
Dissolved Phosphorus	++	–
Total Nitrogen	+++	–
Ammonia	+++	+
Nitrate-Nitrite	++	
Suspended Sediment	++	–
Stream Discharge	–	–



Potential impacts are fire-specific and watershed-specific

03

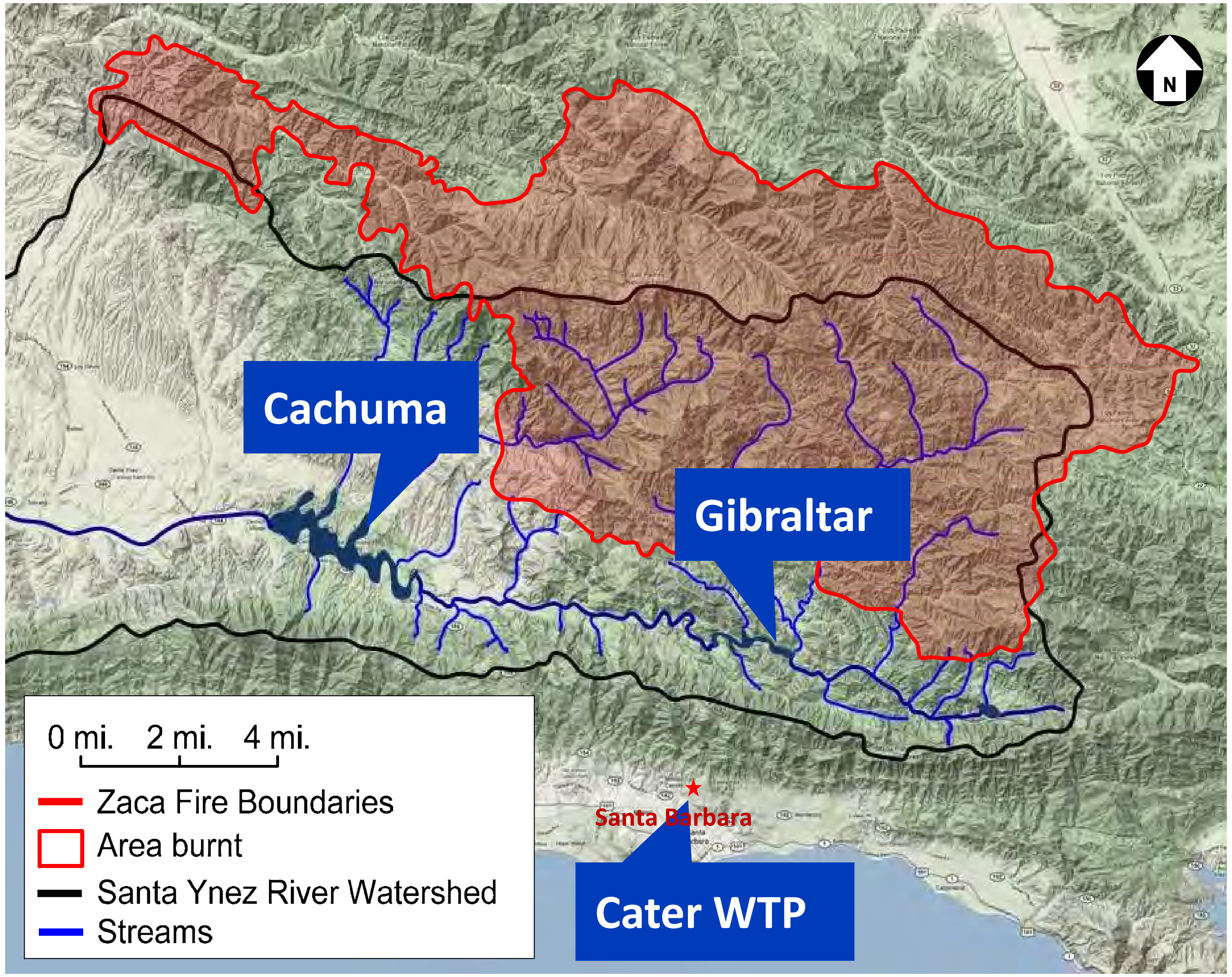
Case Study Series 1: Southwest

Southern Coastal to Central Rockies

Zaca Fire: Santa Barbara, California

Wildfire affected approximately 30% of Cater WTP Catchment Area

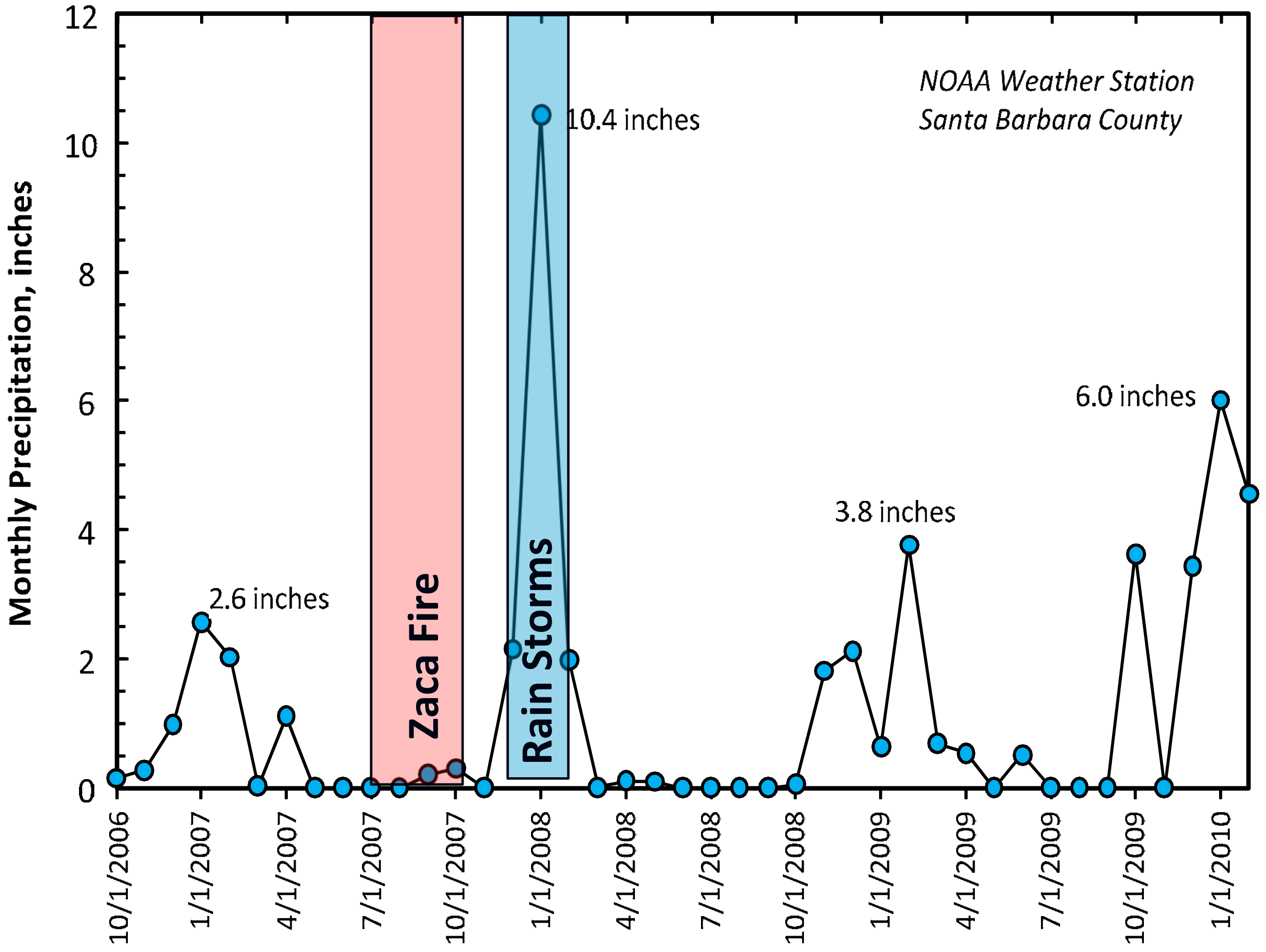
- Summer of 2007
- ~100,000 Ha
- Cater Water Treatment Plant Watershed



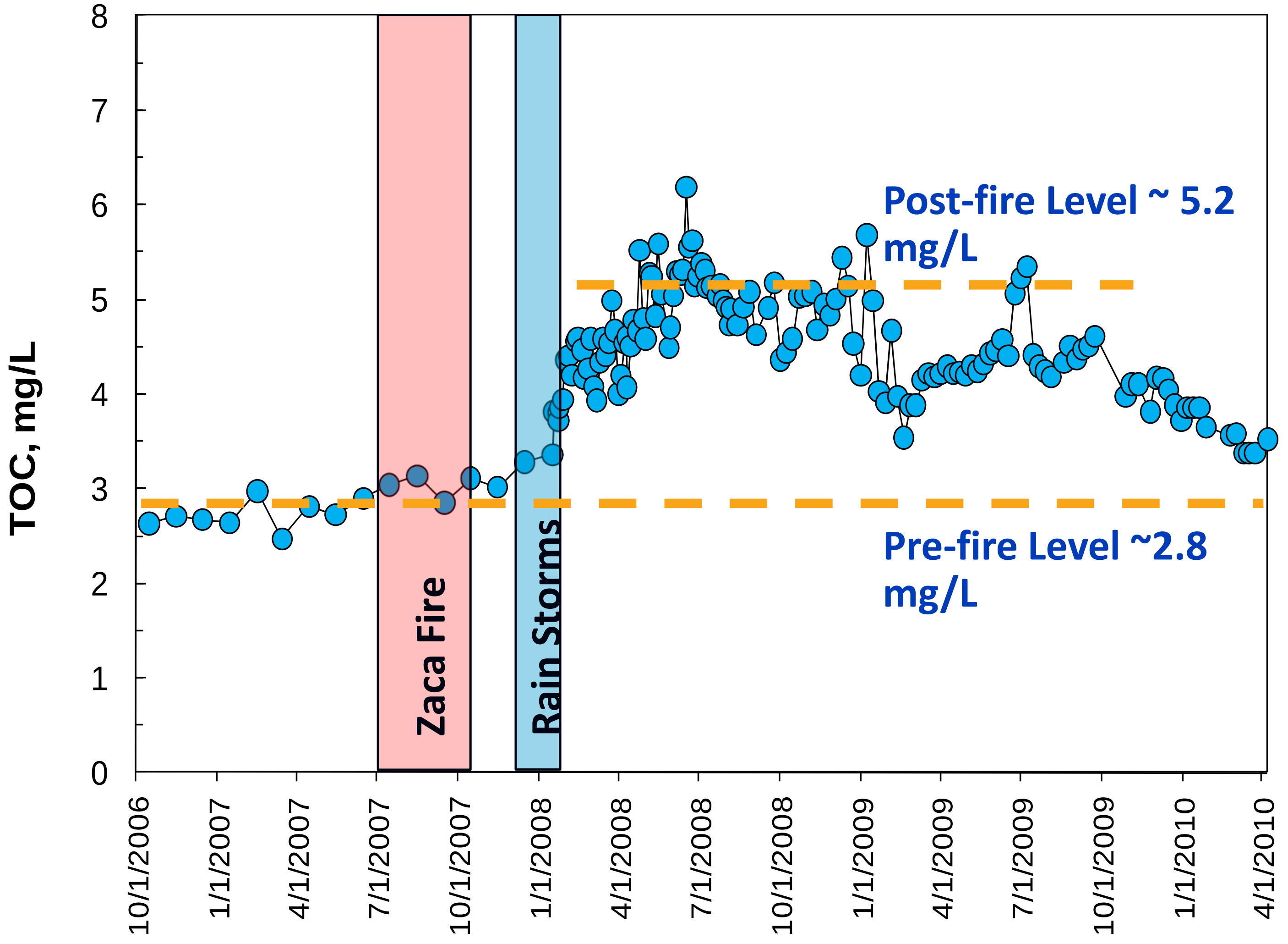
Cater WTP – Santa Barbara, CA

Zaca Fire: Santa Barbara, CA

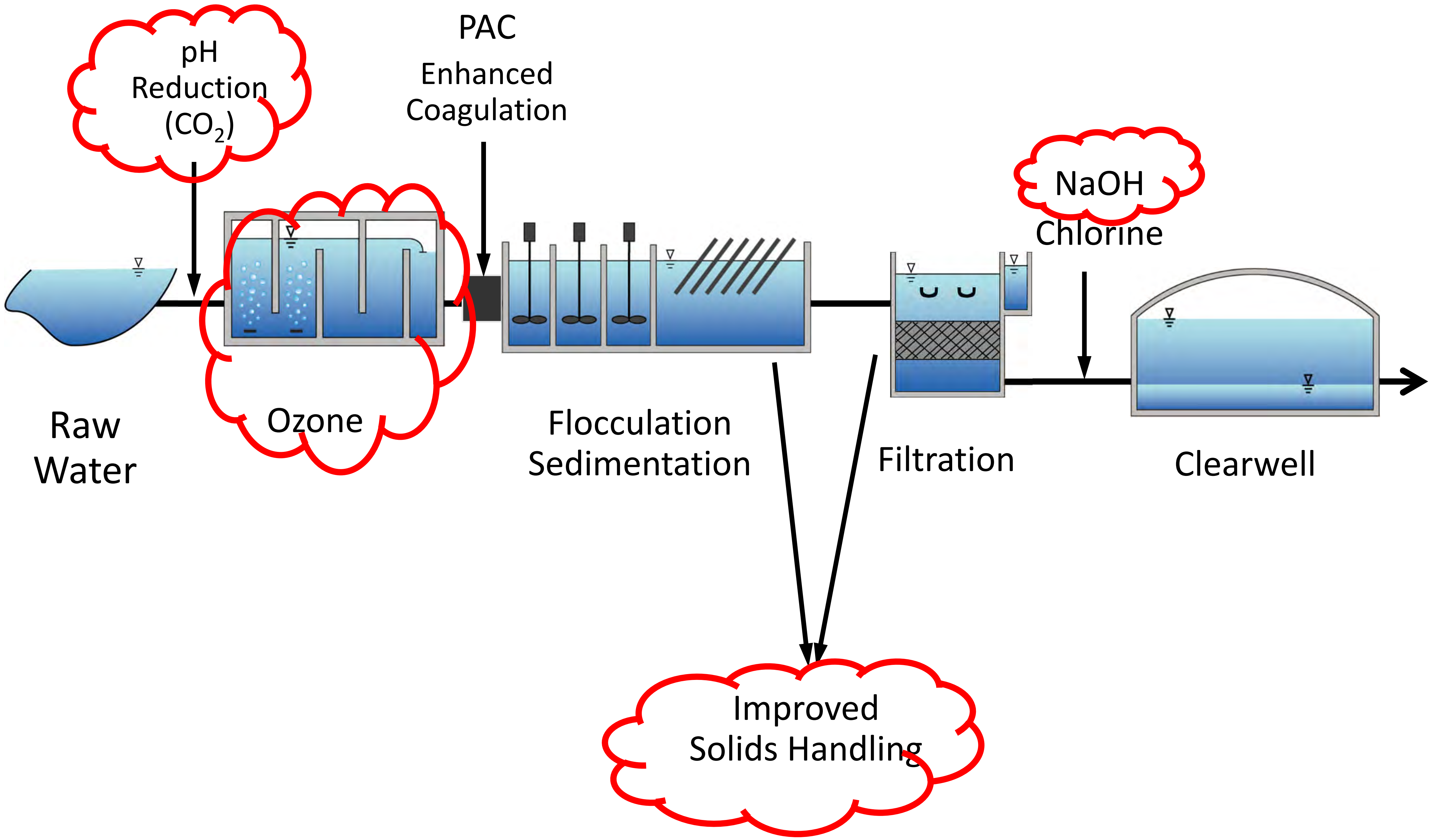
Monthly Rainfall (2 years post wildfire)



TOC (2 years post wildfire)



2013 William B Cater WTP – More Tools



pH Reduction (CO_2)

PAC

Enhanced Coagulation

NaOH

Chlorine

Raw Water

Ozone

Flocculation Sedimentation

Filtration

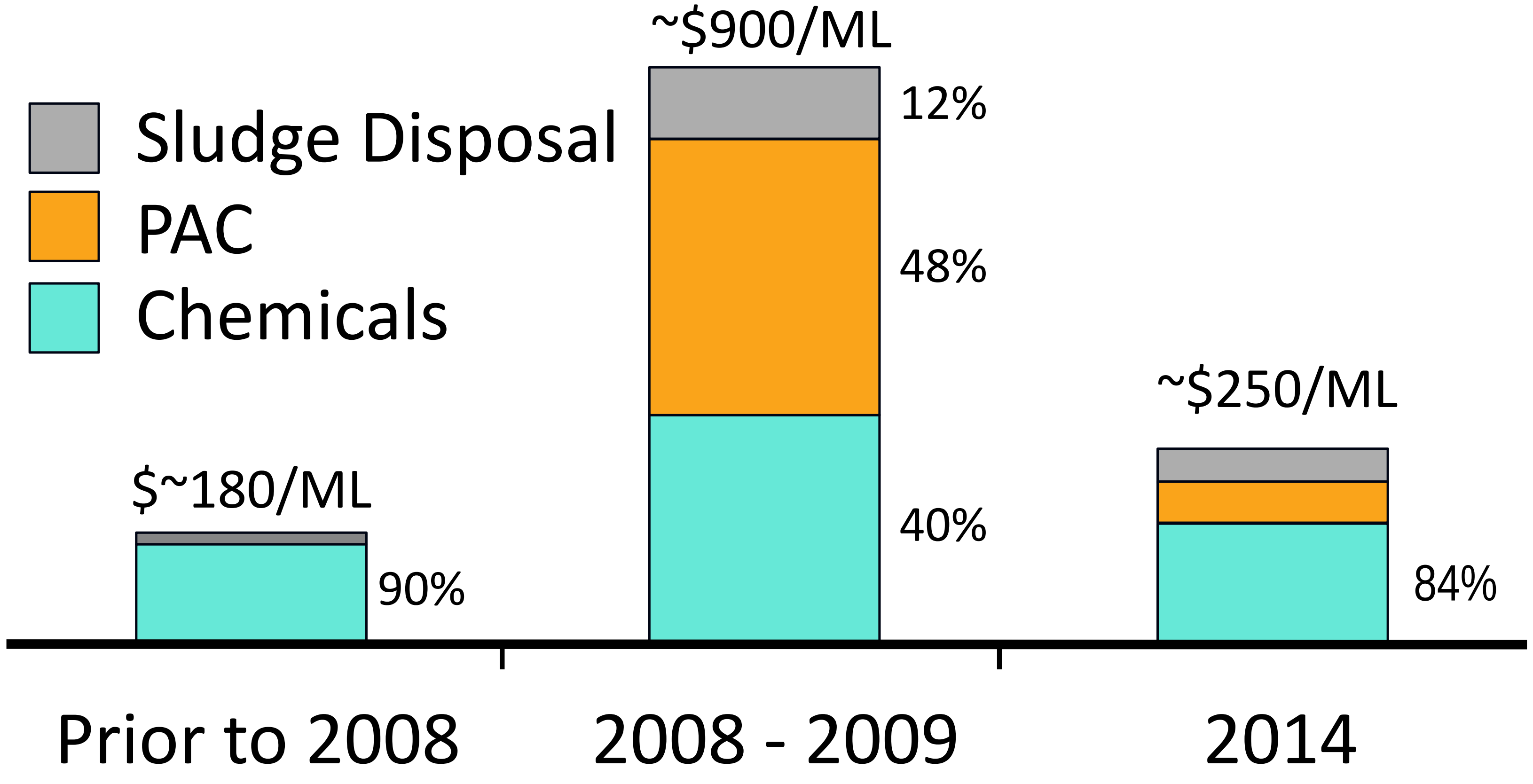
Clearwell

Improved Solids Handling

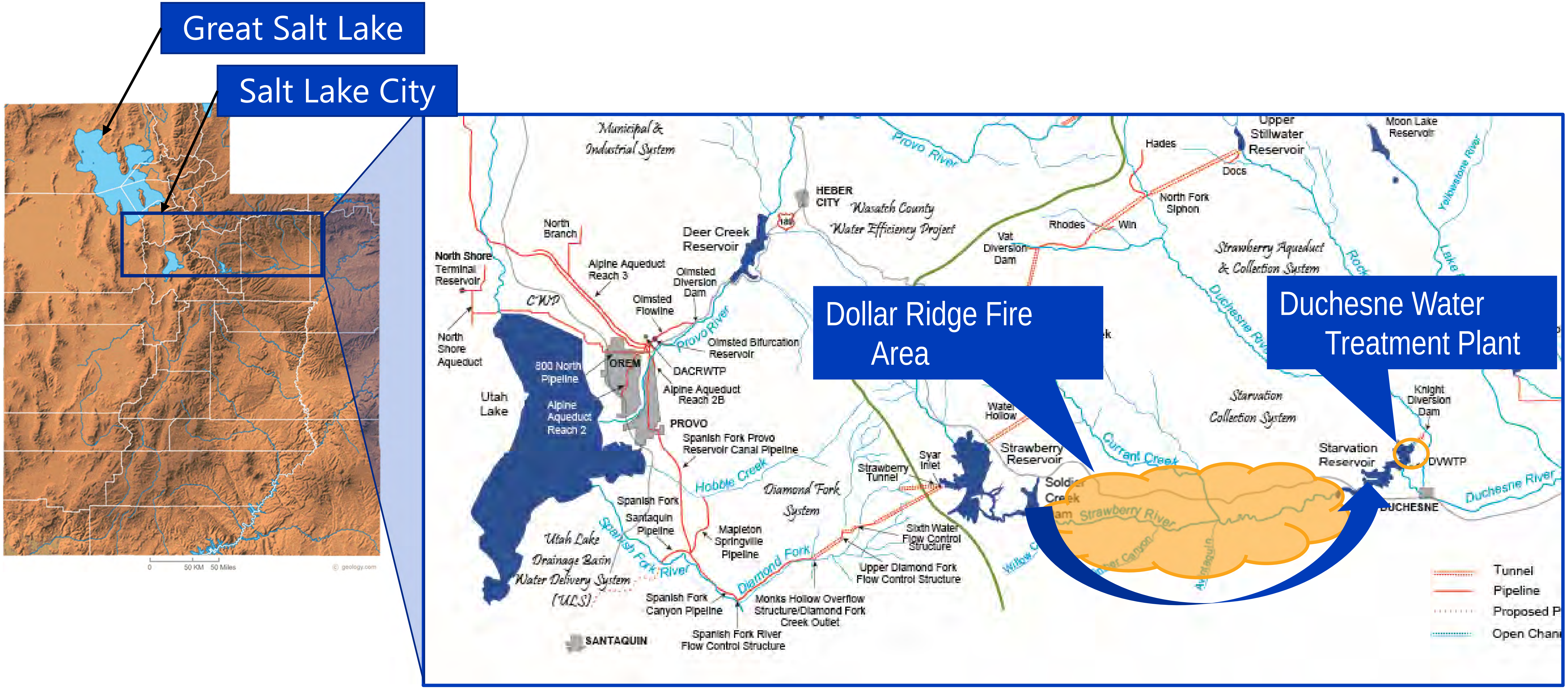
Zaca Fire: Santa Barbara, CA

Cater Water Treatment Plant Cost Impacts

- 140 ML/D Conventional Treatment
  - » 75 ML/D practical max
- Coagulant Usage
  - – 2-4 mg/L PACl
  - – Increased up to 25 mg/L
- Limited tools immediately available – resorted to PAC



Dollar Ridge Fire: Central Utah Water Conservancy District



Great Salt Lake

Salt Lake City

Dollar Ridge Fire Area

Duchesne Water Treatment Plant

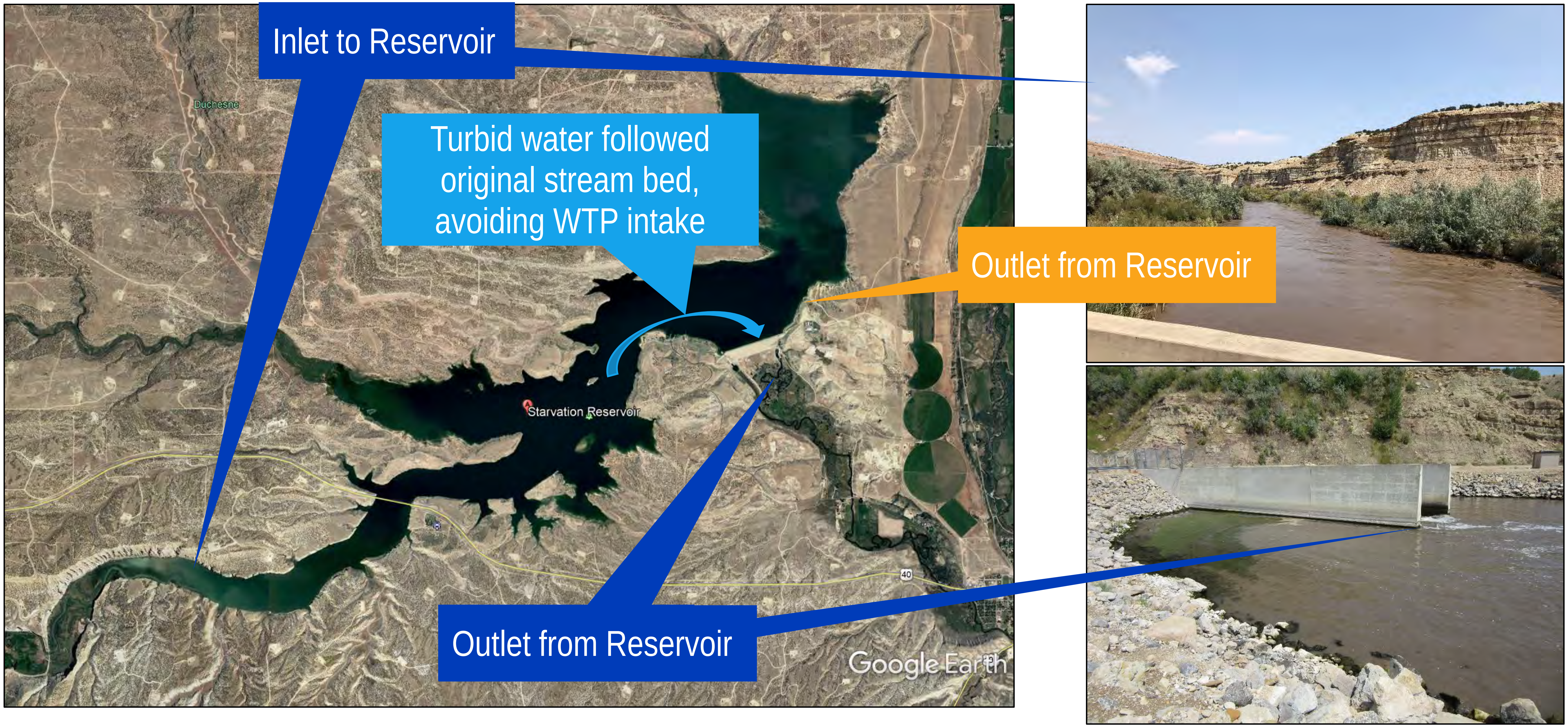
2018 Dollar Ridge Fire and Duchesne Valley WTP: Burned 28,000 Ha (12,000 Ha in the first two days)

Strawberry River Watershed

Strawberry Reservoir releases to Strawberry River and onto Duchesne WTP intake Supply

Heavy rain event following wildfire

Starvation Reservoir configuration saved the WTP from the worst turbidity



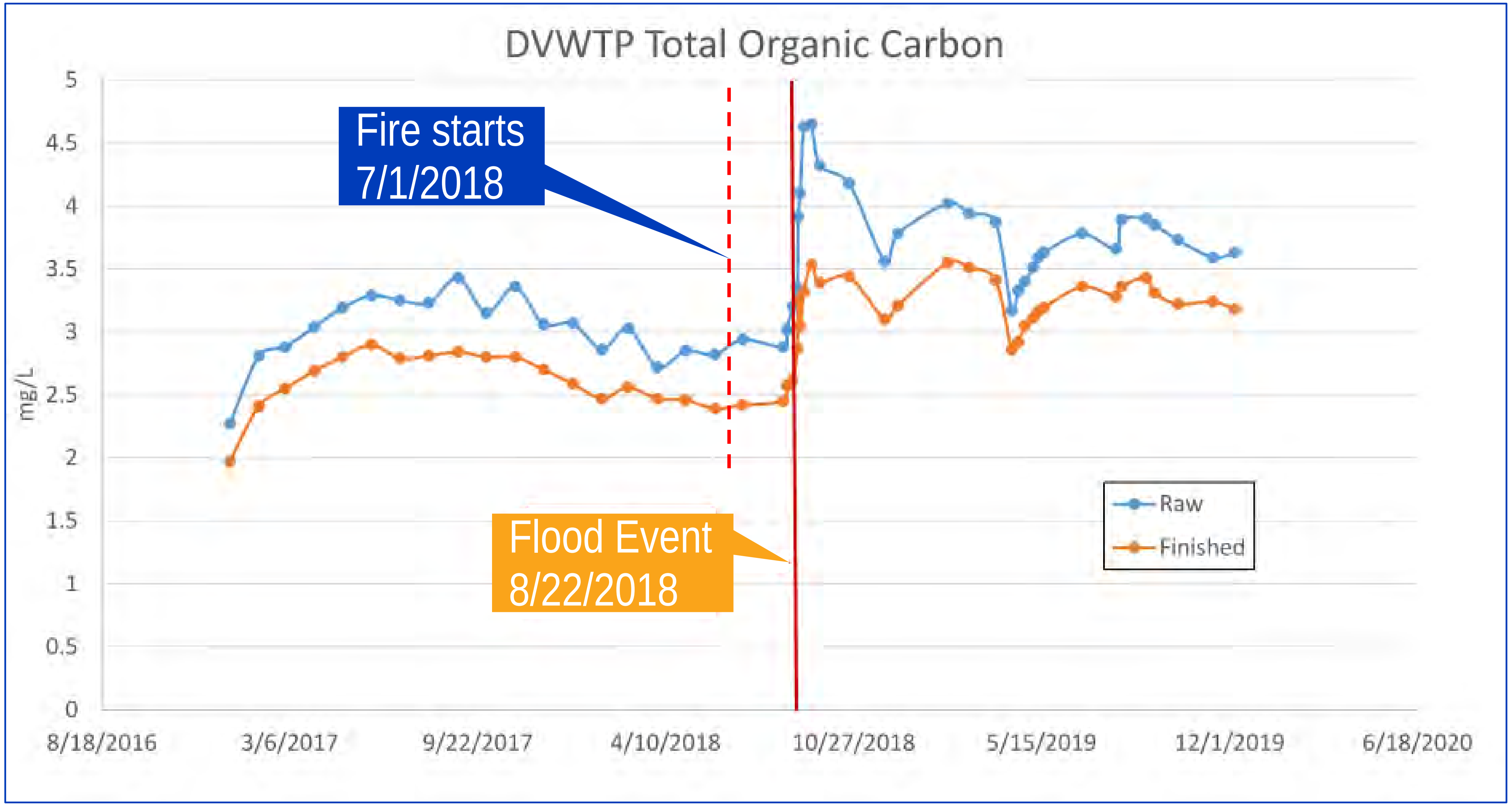
Inlet to Reservoir

Turbid water followed original stream bed, avoiding WTP intake

Outlet from Reservoir

Outlet from Reservoir

The largest impact to this direct filtration plant was TOC related: taste and odor, maintaining residual, DBPs



Fire starts 7/1/2018

Flood Event 8/22/2018

Treatment Responses

Ozone dose:

- 1.0 to 7.2 mg/L

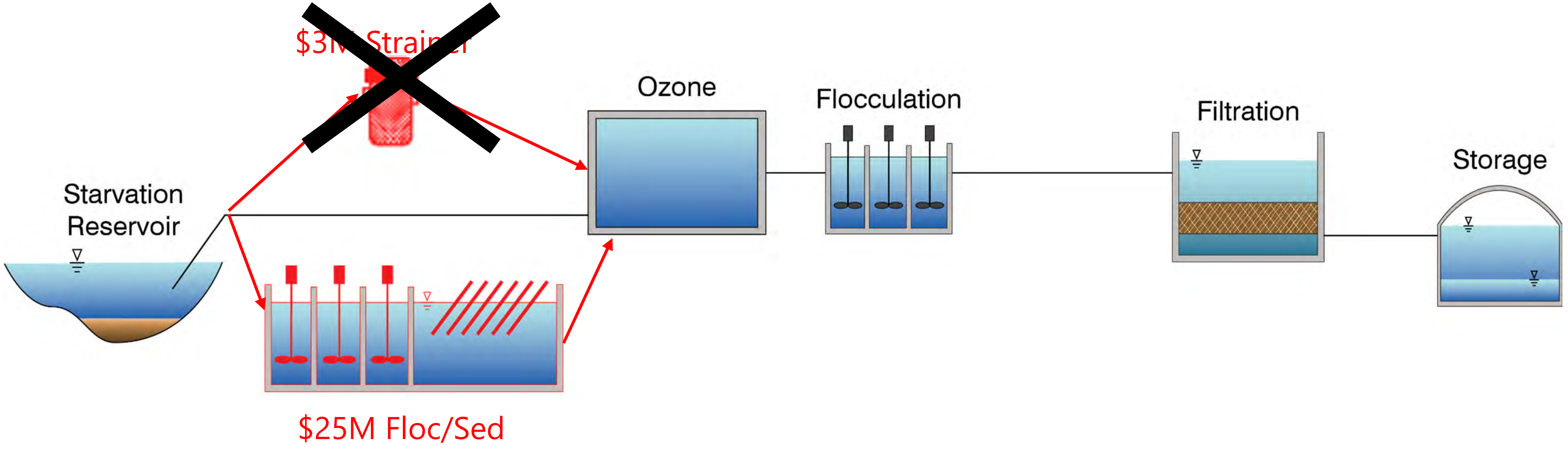
Chlorine dose:

- 3.1 to 7.9 mg/L

Limited coagulation tools as direct filtration

The Result:

- A $3M strainer project to address filter clogging algae was abandoned at 90%. Instead, a $25M project to add floc/sed is under construction



1 year later… Conventional with Int. Ozone is a Success.

- Algae event 1 week before startup:
  - Filter runs were 12 hours @ 8.2 m/hr (3.3 gpm/sf) resulting in UFRVs of 98 m^3/m^2 (2,400 gal/sf), triggered by headloss (looking for >300 m^3/m^2)
- Similar algae event 1 year later:
  - Filter runs were 140 hours @ 7 m/hr (2.9 gpm/sf) and UFRVs of 980 m^3/m^2, triggered by time.
  - Went from backwashing each of their 6 filters twice per day to backwashing 1 filter per day whether it needs it or not.
- This equips them to deal with the lingering affects of the previous fire (increased algae events – frequency and duration, and slightly elevated TOC) – and any future fire events.

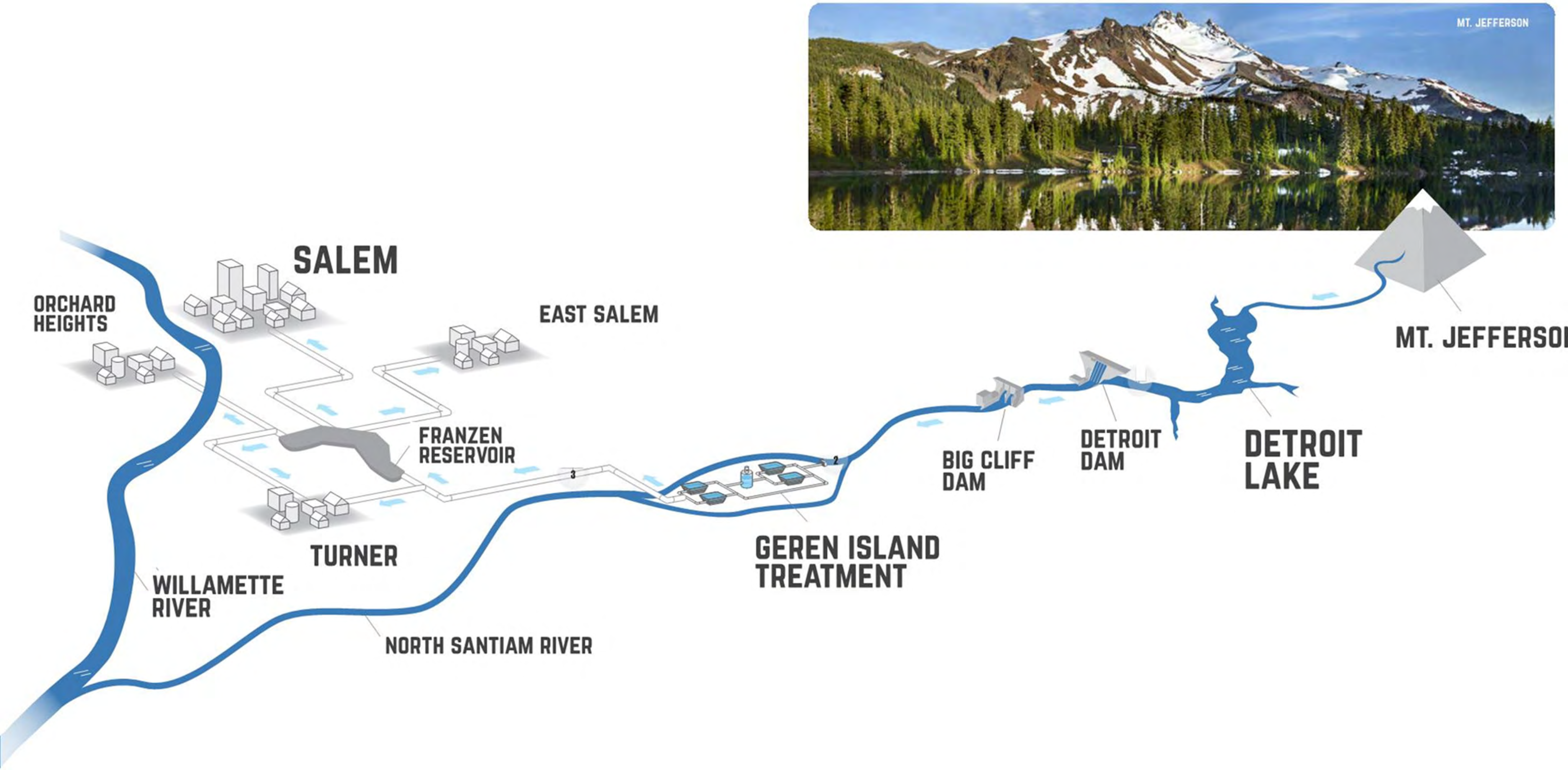
04

Case Study Series 2: Pacific Northwest

Pacific Coastal and Interior

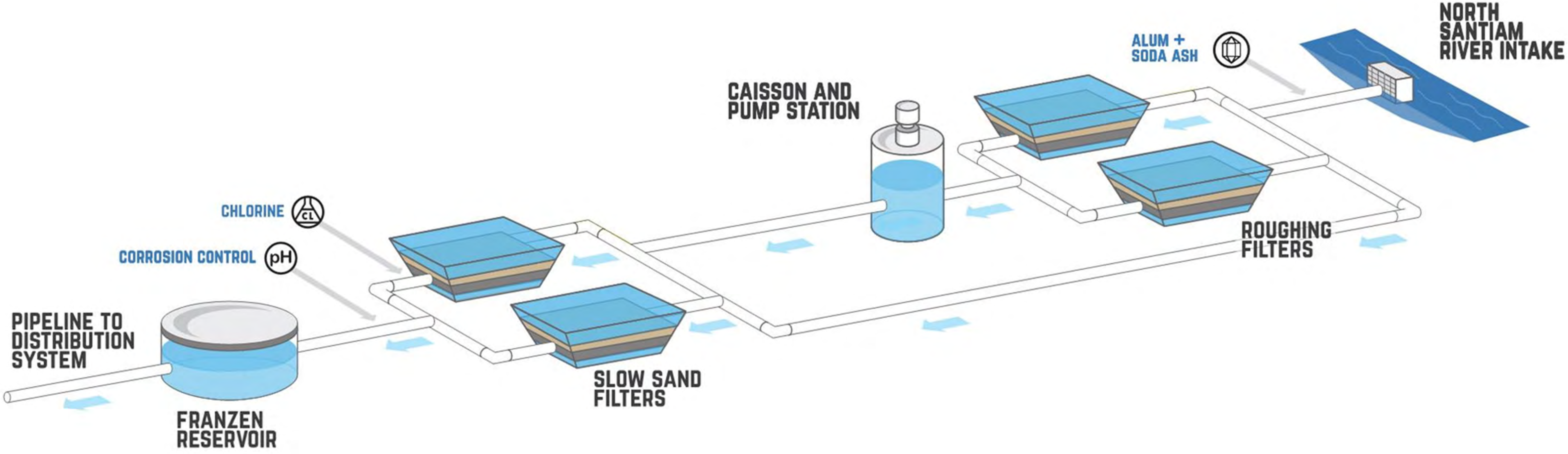
Salem Geren Island Water Treatment Plant

2020 Wildfire Response


MT. JEFFERSON
SALEM
ORCHARD HEIGHTS
EAST SALEM
FRANZEN RESERVOIR
3
2
TURNER
WILLAMETTE RIVER
NORTH SANTIAM RIVER
GEREN ISLAND TREATMENT
BIG CLIFF DAM
DETROIT DAM
DETROIT LAKE
MT. JEFFERSO

Geren Island WTP:

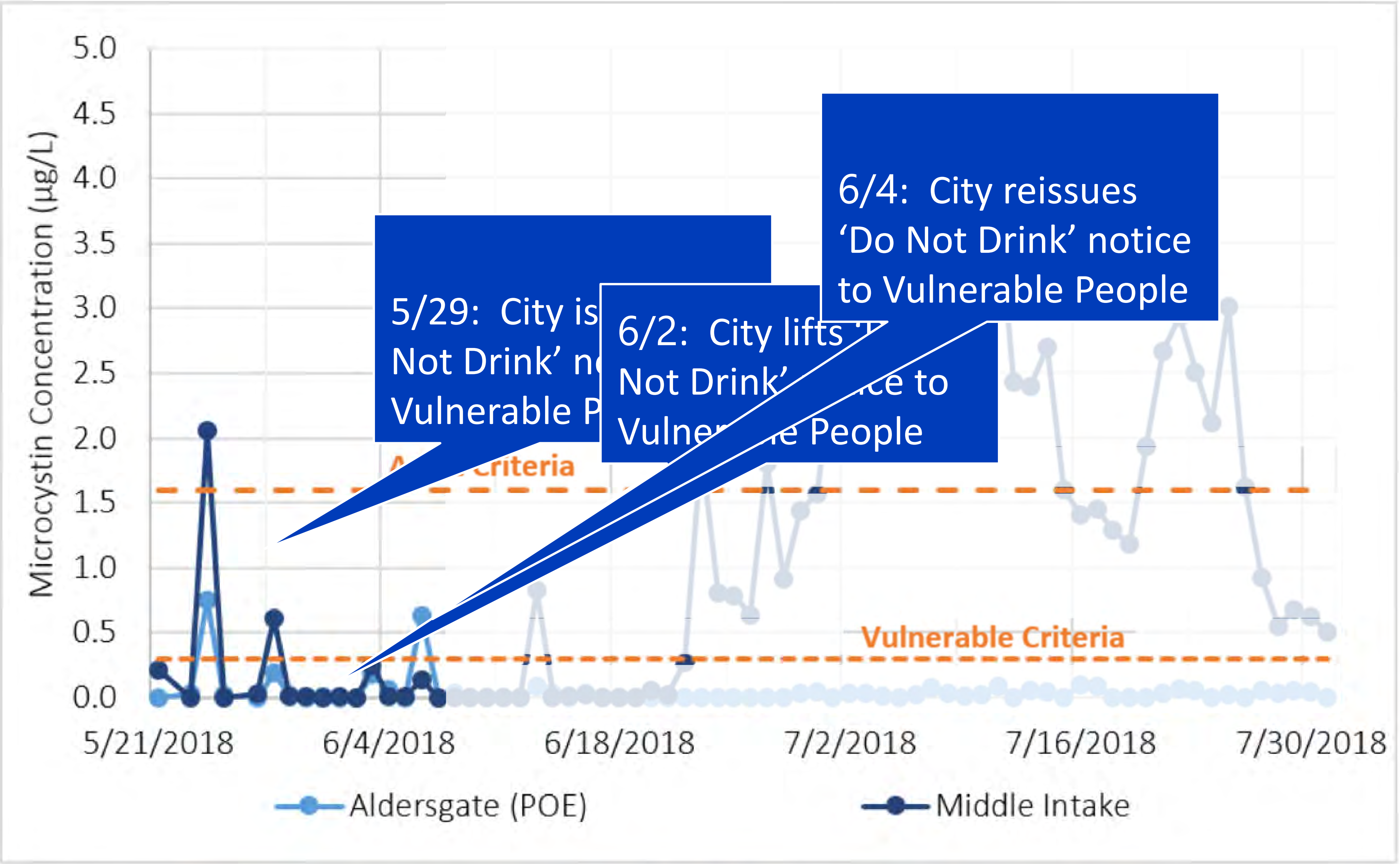
An elegant treatment strategy



Wildfires Increase Algal Toxins Detection Year over Year

	Cylindrospermopsin (ELISA) (<0.10)					Cylindrospermopsin (LC/MS/MS) (<0.005)				
Year	No. of samples	No. of Detects	Min (µg/L)	Max (µg/L)	Average (µg/L)	No. of samples	No. of Detects	Min (µg/L)	Max (µg/L)	Average (µg/L)
2013	13	0	-	-	-	-	-	-	-	-
2014	4	2	0.06	0.238	0.149	33	32	0.03	0.15	0.0625
2015	-	-	-	-	-	28	23	0.01	0.12	0.0535
2016	-	-	-	-	-	15	9	0.04	0.13	0.07
2017	-	-	-	-	-	38	15	0.09	0.43	0.163

May Detections Exceed Health Reference Levels

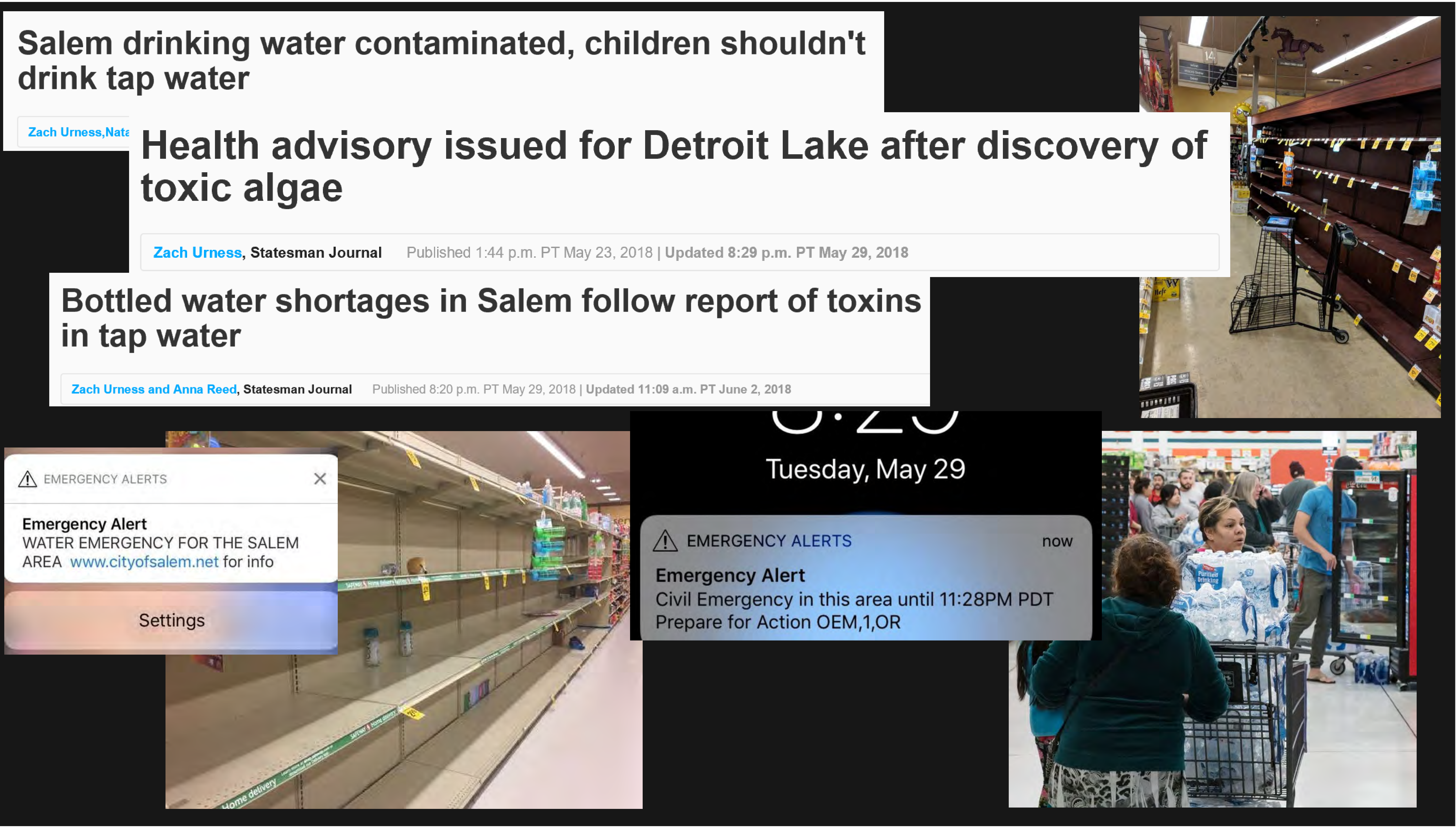


5/29: City is
Not Drink' n
Vulnerable P

6/2: City lifts
Not Drink' ce to
Vulner le People

6/4: City reissues
'Do Not Drink' notice
to Vulnerable People

Figure 2.2 Microcystin Concentrations at the GIWTF Intake and Aldersgate (POE)



Salem drinking water contaminated, children shouldn't drink tap water

Zach Urness,Nata

Health advisory issued for Detroit Lake after discovery of toxic algae

Zach Urness, Statesman Journal Published 1:44 p.m. PT May 23, 2018 | Updated 8:29 p.m. PT May 29, 2018

Bottled water shortages in Salem follow report of toxins in tap water

Zach Urness and Anna Reed, Statesman Journal Published 8:20 p.m. PT May 29, 2018 | Updated 11:09 a.m. PT June 2, 2018

EMERGENCY ALERTS

Emergency Alert
WATER EMERGENCY FOR THE SALEM AREA www.cityofsalem.net for info

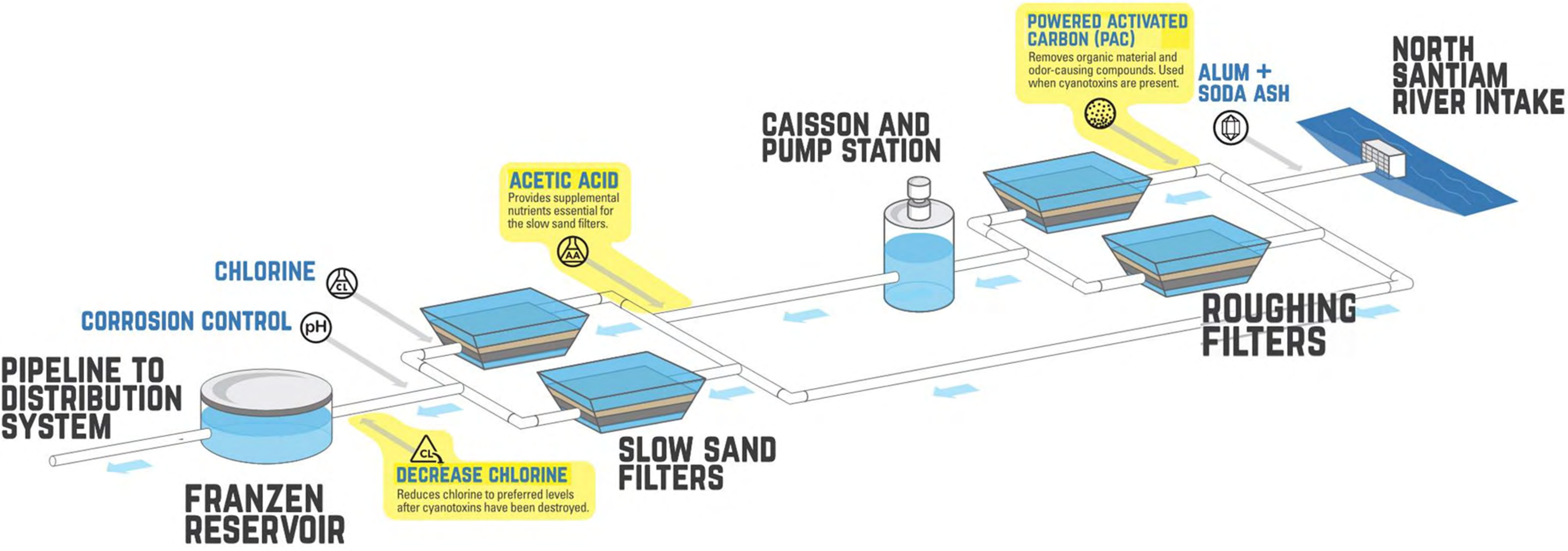
Tuesday, May 29

EMERGENCY ALERTS now

Emergency Alert
Civil Emergency in this area until 11:28PM PDT
Prepare for Action OEM,1,OR

Near-term Response:

Focus on Emergency Response



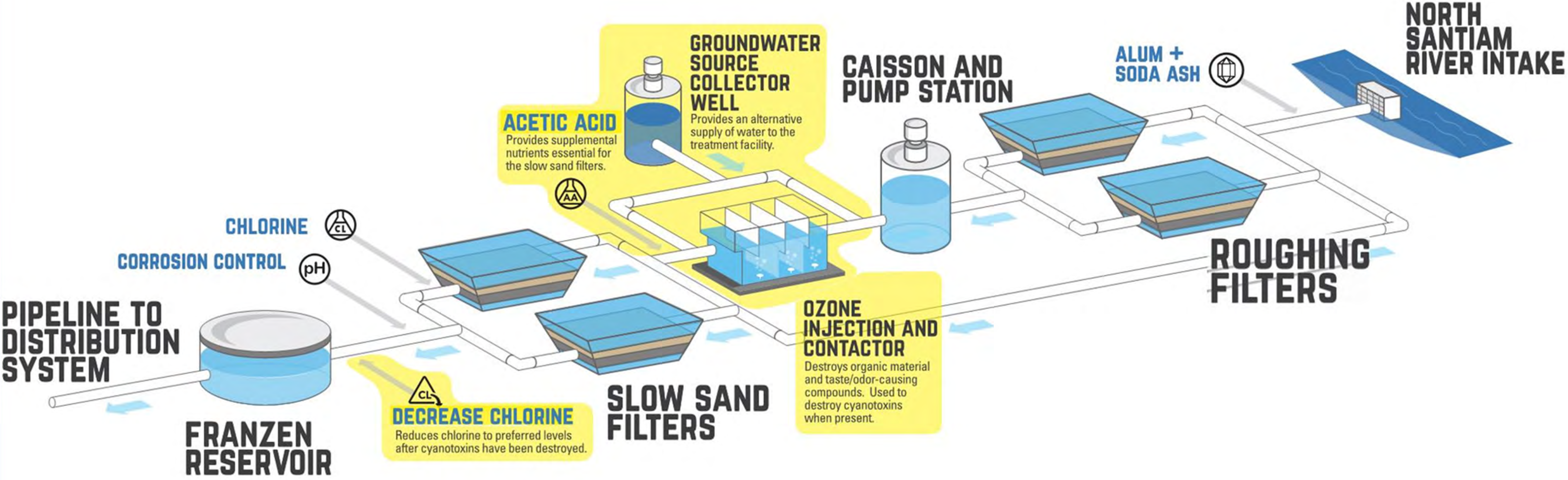
POWERED ACTIVATED CARBON (PAC)
Removes organic material and odor-causing compounds. Used when cyanotoxins are present.

ACETIC ACID
Provides supplemental nutrients essential for the slow sand filters.

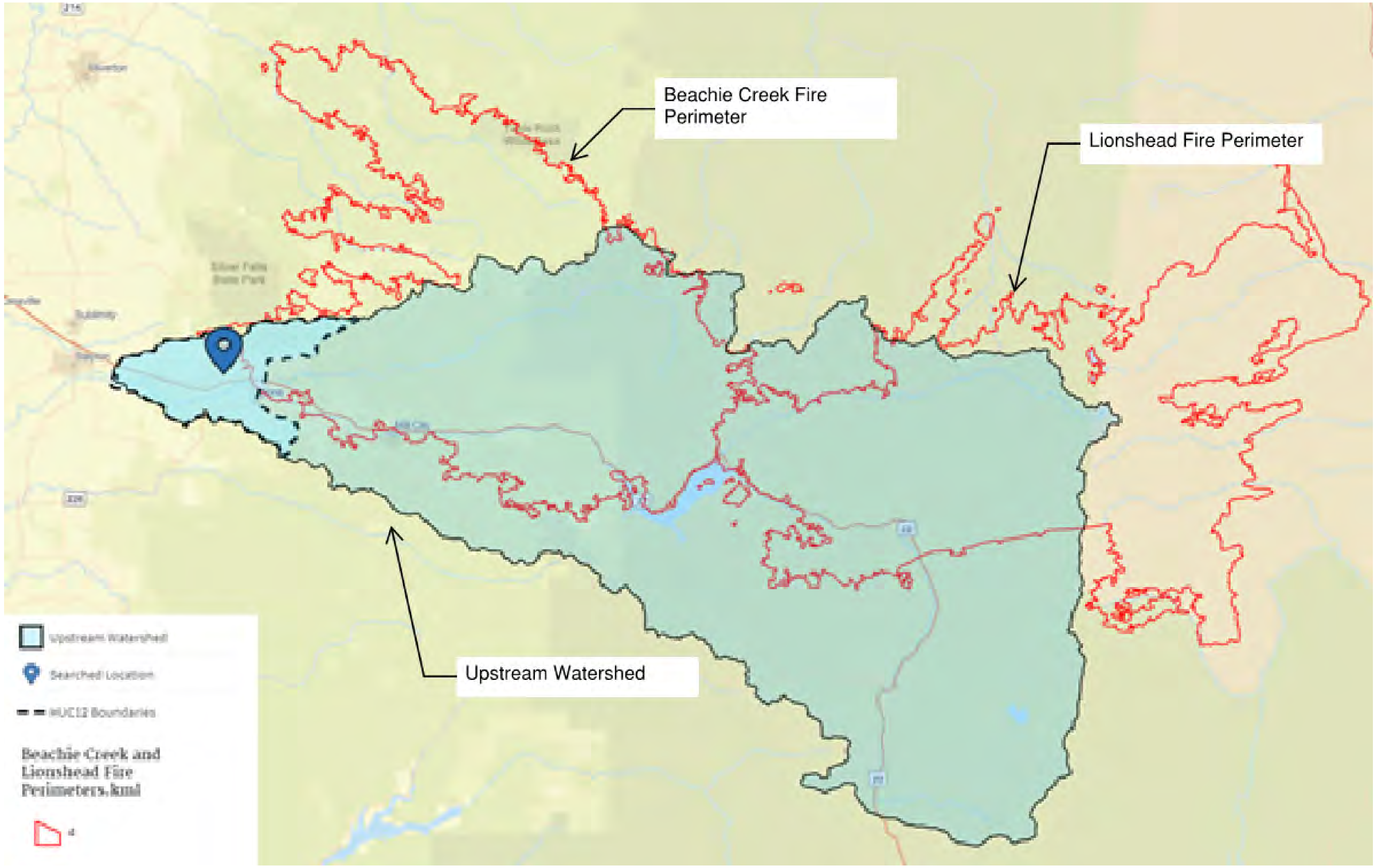
DECREASE CHLORINE
Reduces chlorine to preferred levels after cyanotoxins have been destroyed.

Long-term Response:

Focus on Resiliency



The Best Laid Plans: Additional Wildfires

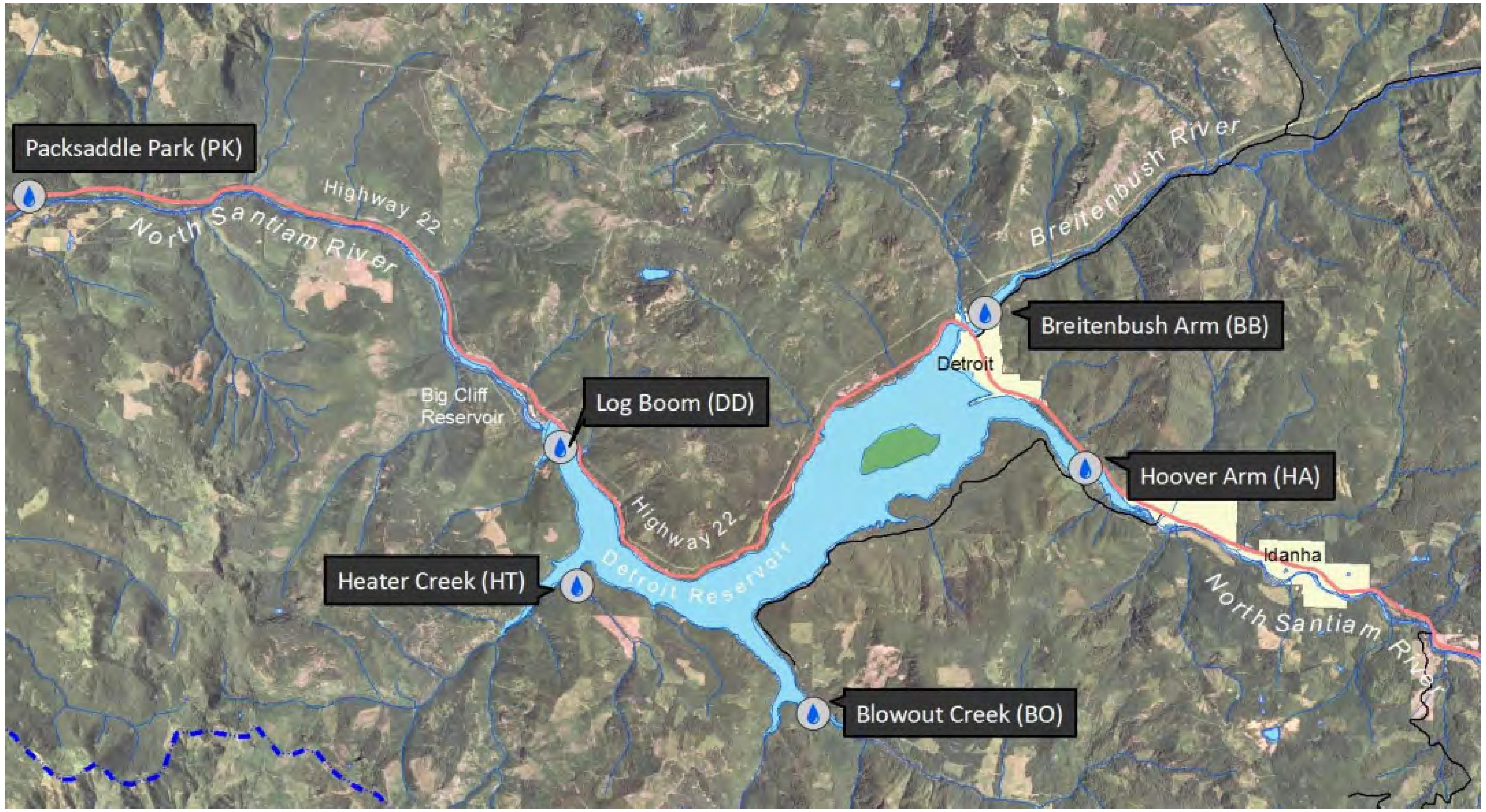


- Fire Reached within Miles of GIWTP
- Over 50% of Watershed within Burn Extents

Source: *U.S. Geological Survey (Dec. 17, 2021) and National Interagency Fire Center (Jan. 28, 2022).*

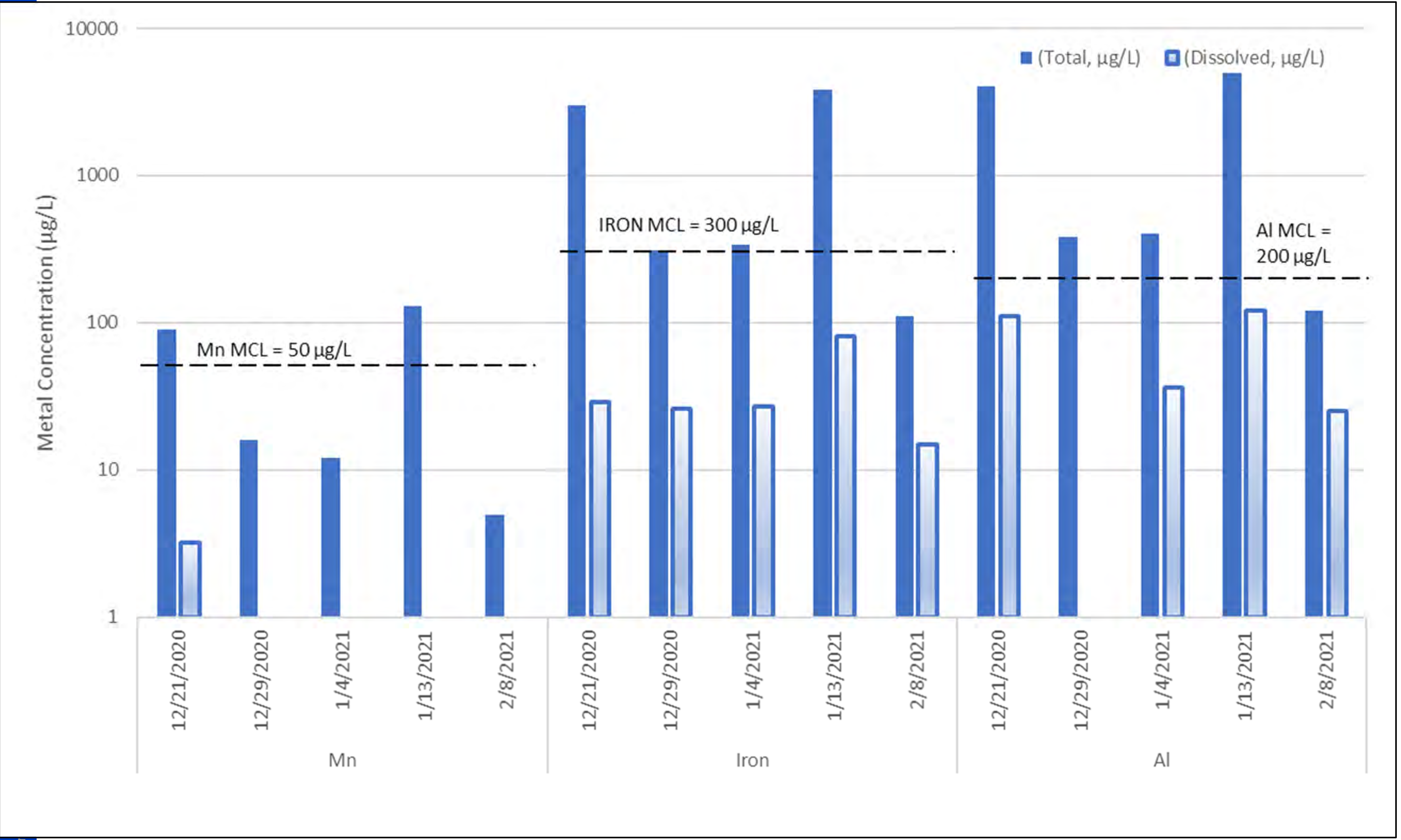
Increased Monitoring

- Increased monitoring throughout watershed, GIWTP, and in distribution system
- Analyzed for variety of parameters, including TOC, nutrients, metals, and disinfection by-products



Source: *City of Salem Drinking Water Monitoring Program (cityofsalem.net/drinking-water-monitoring-program.aspx)*

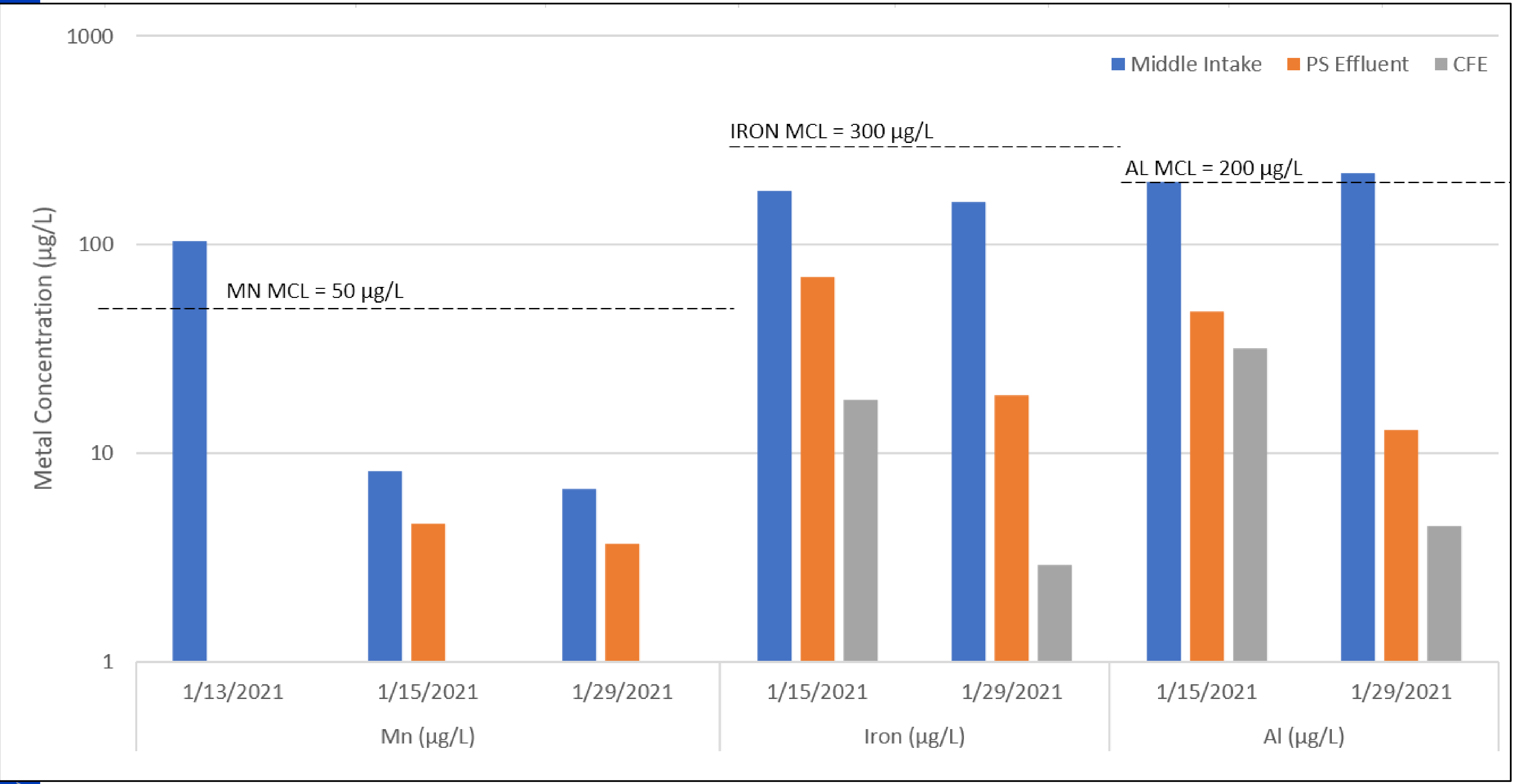
High Metals Concentration



- High metal concentrations following wet weather events
  - » *Manganese, iron, and aluminum*
- Dissolved metal concentrations much lower than total
  - » *Note graph in logarithmic scale*

Source: *City of Salem Drinking Water Monitoring Program (cityofsalem.net/drinking-water-monitoring-program.aspx)*

Particulate Metals Removed with Filtration



- Good removal of metals observed through filtration
  - » *Middle intake = plant intake*
  - » *PS effluent = following first-stage filtration*
  - » *CFE= combined filtered effluent*
- GIWTP effectively treated water

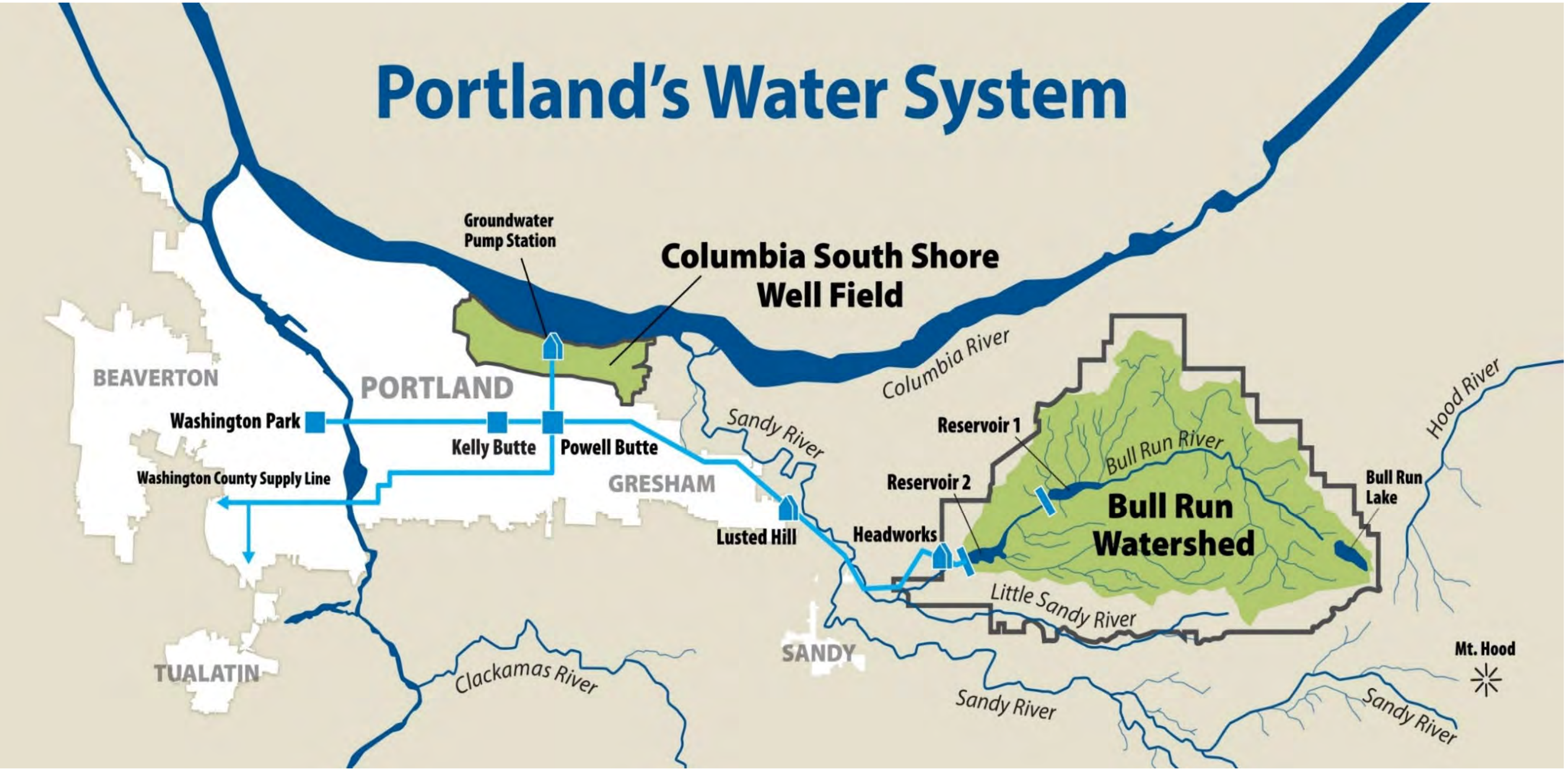
Wildfire Lessons Learned

- Huge increase in metal concentrations following wet weather events
  - *Manganese, iron, and aluminum*
- Vast majority of metals not dissolved
- Particulate metals are filterable
- For immediate response, no need for oxidant

Conclusion

- Resiliency planning prepared the City of Salem for unforeseeable weather and supply challenges
- Current data following wildfires shows high concentrations of filterable particulate metals
- City of Salem continuing to monitor watershed for water quality effects from wildfires, which can take years to surface

Portland Water Bureau

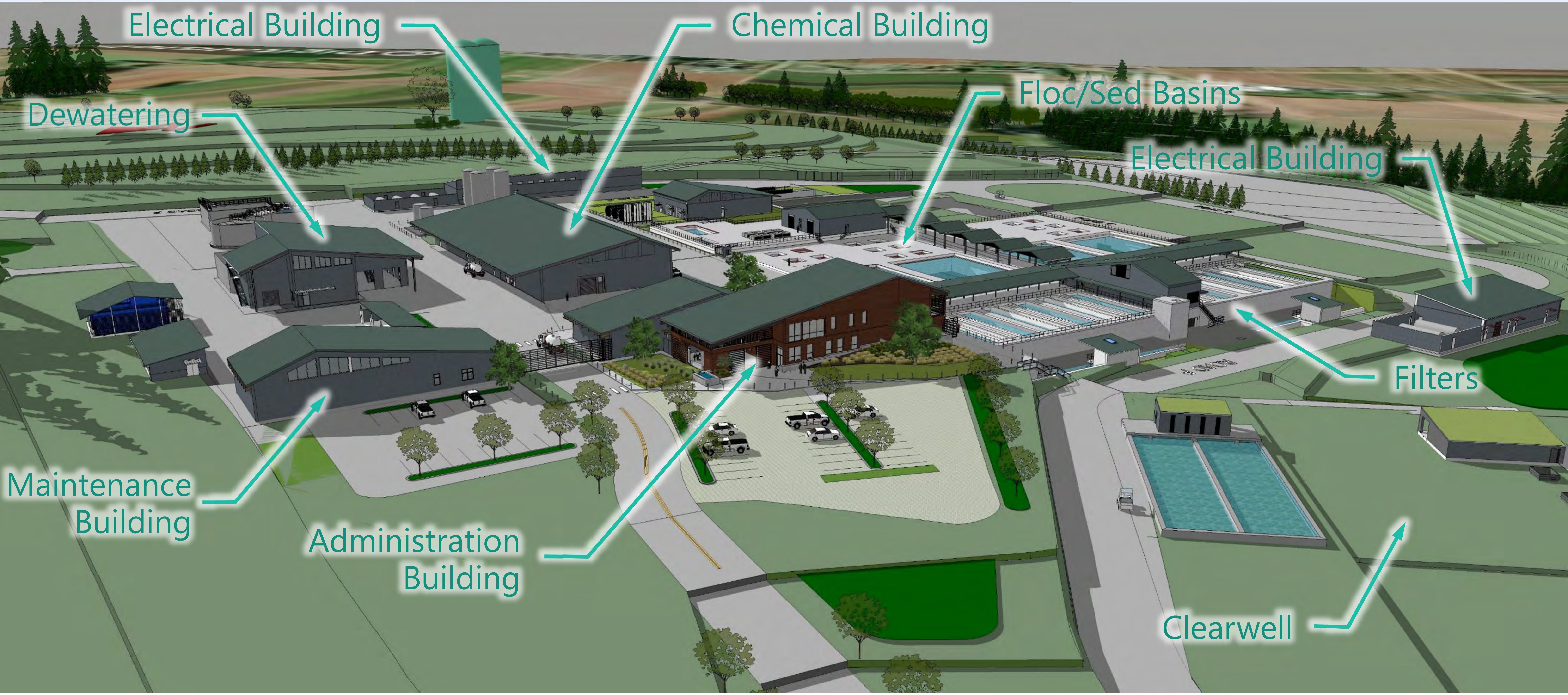


🟦	Water Storage Facility	☐	Portland Water System Distribution Area
—	Water Supply Pipes	🟩	Water Source
⌂	Water Treatment Facility	▬	Protected Area
▬	Dam		

- Serves almost a million people
- Uses 100 million gallons of water on an average day
- Serves the City of Portland and 19 wholesale water districts

New Filtration Facility Layout

Currently in Design (500 mld)

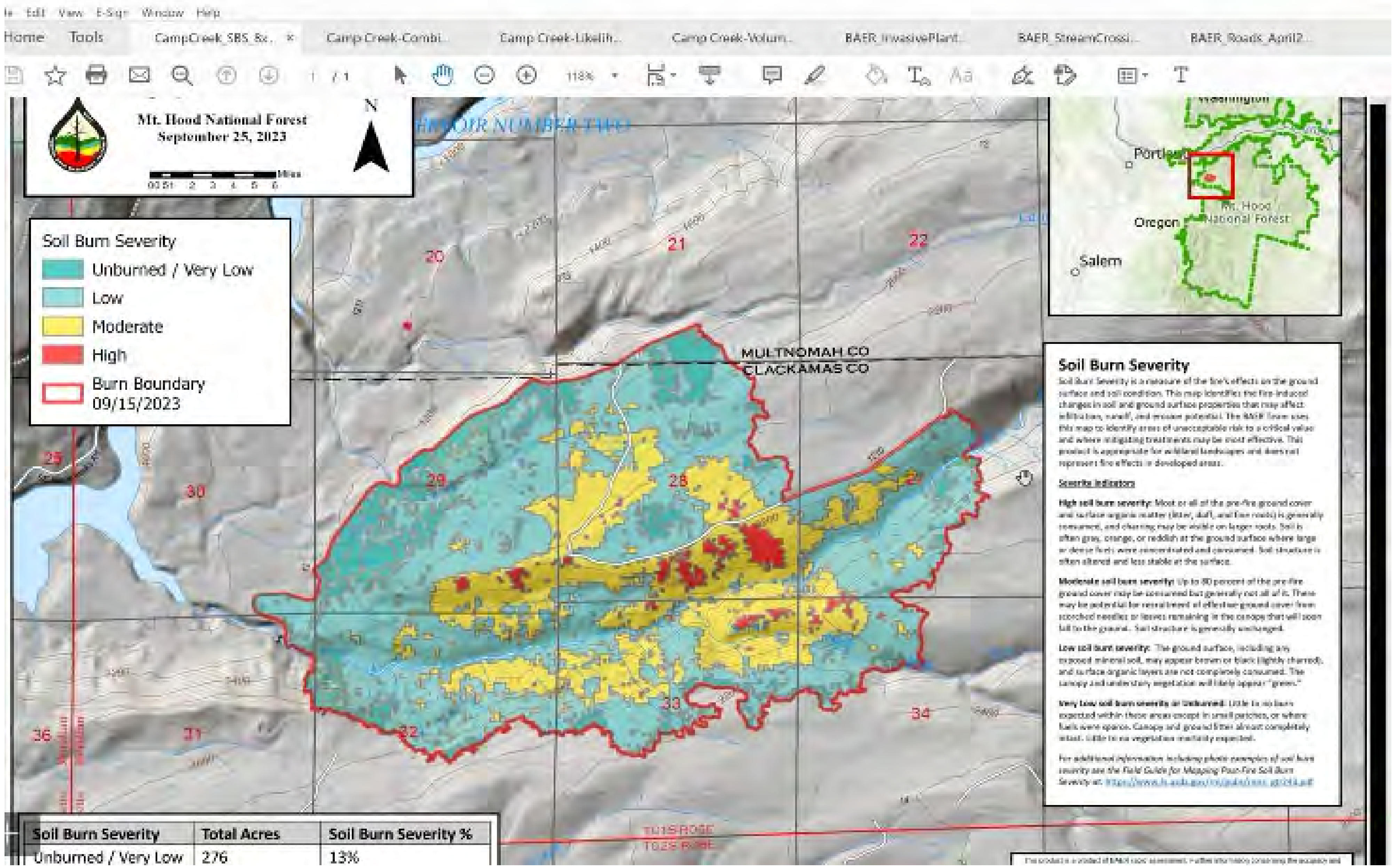


Portland Water Bureau's

Camp Creek Fire (first fire since ~1490)

Visual of Burn Mosaic

Camp Creek Fire Burn Severity



Mt. Hood National Forest
September 25, 2023

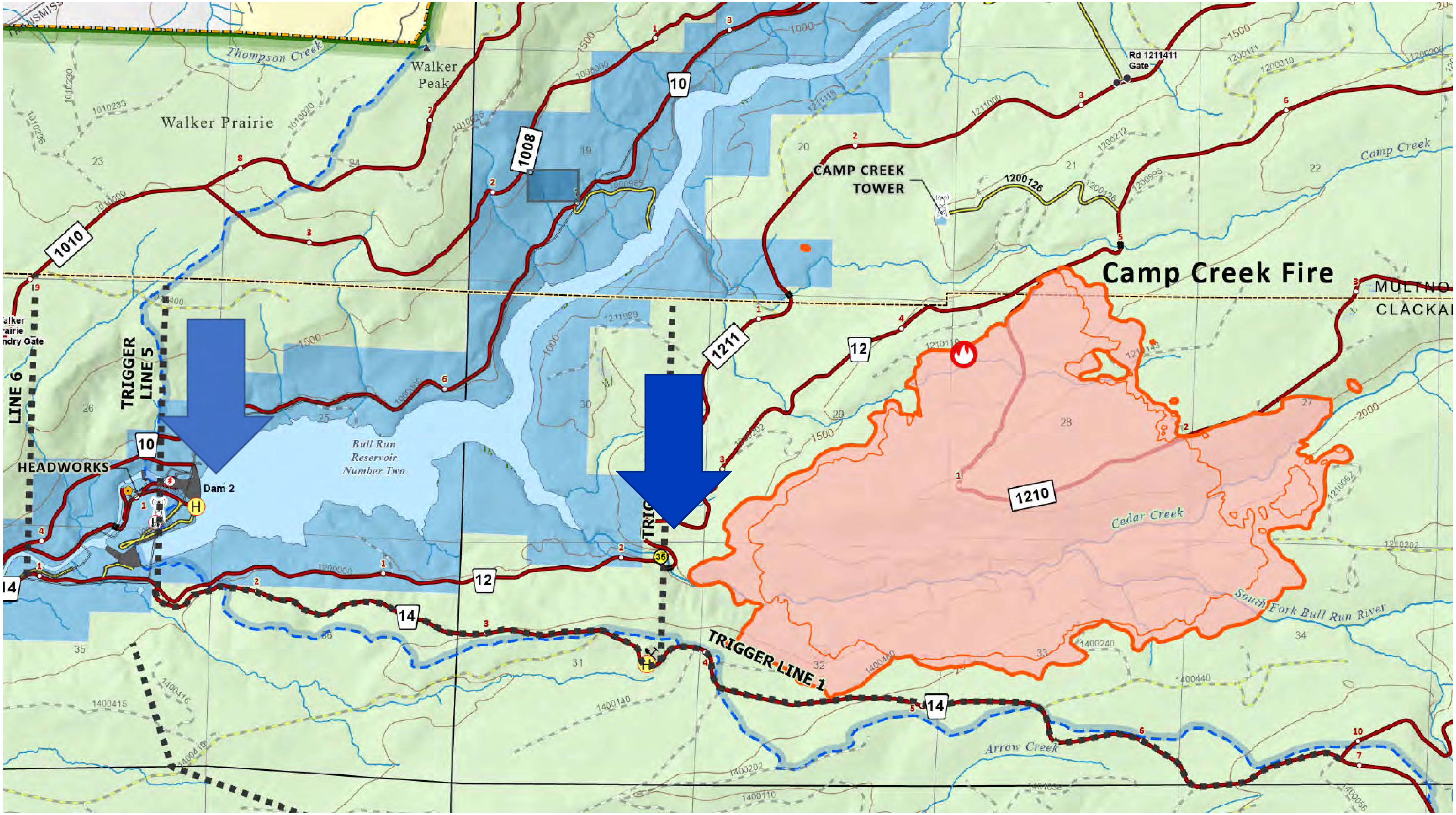
Soil Burn Severity
- Unburned / Very Low
- Low
- Moderate
- High
- Burn Boundary 09/15/2023

Soil Burn Severity

Soil Burn Severity	Total Acres	Soil Burn Severity %
Unburned / Very Low	276	13%

830 Ha fire perimeter (3% of drinking supply drainage)

- **Unburned** 13%
- **Low** 56%
- **Moderate** 28%
- **High** 3%


Camp Creek Fire
Walker Peak
Walker Prairie
Thompson Creek
CAMP CREEK TOWER
Rd 1211411 Gate
Camp Creek
MULTNO
CLACKA
TRIGGER LINE 5
LINE 6
HEADWORKS
Dam 2
Bull Run Reservoir Number Two
TRIGGER LINE 1
Cedar Creek
South Fork Bull Run River
Arrow Creek
1010
1008
10
1211
12
1210
14
1200126

Preliminary Post Fire Actions

Burned Area Emergency Response (BAER) Recommendations

- Invasive species treatment and monitoring on burned and suppression impacted areas
- Targeted early warning systems for potential land slides based on weather
- Increased water quality monitoring
- Erosion mitigation not needed based on sediment delivery modeling and needle cast mulch that has fallen over most of the bare soil
- Storm monitoring of culverts

PWB Take aways

- Leveraged lessons learned from neighbor agencies
- Initial response supported by the National Interagency Fire Centre and BAER Program
- Leverage ash for bench- and pilot-scale investigation to validate provisions at the new Filtration Facility

Best Practices to be Fire Ready

Impacts to Water Treatment

Sediment/Turbidity

- Increased solids disposal and handling requirements
- Decreased filter run times (increased filter backwashing frequency)

Organics and nutrients

- Chlorine demand
- Taste and odor
- Disinfection by-products
- Changing charge neutralization chemistry

Metals

- Typically iron and manganese

A Range of Practices Can Increase Fire-Readiness

GOOD

- ✓ Low (or no) capital
- ✓ Focus on preparatory planning

BETTER

- ✓ Minor capital investments
- ✓ Operational adjustments

BEST

- ✓ Capital investments
- ✓ System improvements

GOOD

Evaluate potential wildfire impacts (watershed size, slopes, amount of fuel, soil characteristics)

Evaluate alternate source(s) of supply and/or alternate treatment facilities to implement post-fire

Confirm instrumentation & lab equipment is calibrated and reliable for changing water quality and chem feeds

Conduct pre-fire planning (connect with partner agencies, follow established planning checklists)

Assess WTP capabilities (reliable capacity at high turbidity/oxidant demand, solids capacity, T&O and Fe/Mn capacity, chem feeds)

Assess ability to reduce algal toxins via oxidants/adsorption

BETTER

Install remote water quality monitoring instruments
and plan for how to react - alternate sources, operational changes

Use zeta potential for monitoring and guiding coagulant doses
for highly charged and turbid source water

Add ability to **feed high charge low molecular weight cationic polymer**

BEST

🏆

Construct necessary infrastructure to access and use **Alternate source of supply**

Construct treatment improvements to harden processes to reliably treat fire-impacted water

Install Hypolimnetic Aeration on supply reservoirs and/or forebays to prevent anaerobic conditions that release Fe/Mn and organics

THANK YOU!

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Vice President
email: adomonoske@carollo.com

Resources and References

WSSF-Intentions-Paper-March2023.pdf (gov.bc.ca)

Wildfire: Its Effects on Drinking Water Quality - HealthLinkBC File #49f - Printer-friendly version

Water quality – typical post-wildfire ranges

(HDR, 2016)

Constituent	Expected Range	Comment
Turbidity (NTU)	Peak 100s-10,000s	May produce flashy water
pH (SU)	Increase ~0.5	Due to dissolution of ash
Total Alkalinity (mg/L)	300s	Due to leaching cations from ash and burned litter
Cations (e.g., Ca, Mg, Si)	Increase in 100s	Carbonate form after dissolution of ash
Total Organic Carbon (mg/L)	15-20	Major fraction may be dissolved
Nitrate-N (mg/L)	< 5	Often peaks during spring snowmelt; slow decline 3-5 years post-fire
Ammonia-N (mg/L)	< 10	
Phosphorus (PO4, mg/L)	< 20	80% soluble; from dissolution of ash in stream; declines 1-2 years post-fire
Iron and Manganese (mg/L)	< 10	
Cyanide (ug/L)	< 120	From fire retardants

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journal homepage: www.elsevier.com/locate/watres

Severe western Canadian wildfire affects water quality even at large basin scales

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Suspended sediment
Boreal
Continuous monitoring

ABSTRACT

Wildfires can have severe and lasting impacts on the water quality of aquatic ecosystems. However, our understanding of these impacts is founded primarily from studies of small watersheds with well-connected runoff regimes. Despite the predominance of large, low-relief rivers across the fire-prone Boreal forest, it is unclear to what extent and duration wildfire-related material (e.g., ash) can be observed within these systems that typically buffer upstream disturbance signals. Following the devastating 2016 Fort McMurray wildfire in western Canada, we initiated a multi-faceted water quality monitoring program that suggested brief (hours to days) wildfire signatures could be detected in several large river systems, particularly following rainfall events greater than 10 mm. Continuous monitoring of flow and water quality showed distinct, precipitation-associated signatures of ash transport in rivers draining expansive (800–100,000 km^2) and partially-burned (<1–22 percent burned) watersheds, which were not evident in nearby unburned regions. Yields of suspended sediment, nutrients (nitrogen, phosphorus) and metals (lead, others) from impacted rivers were 1.2–10 times greater than from those draining unburned regions. Post-fire suspended sediment concentrations in impacted rivers were often larger than pre-fire 95% prediction intervals based on several years of water sampling. These multiple lines of evidence indicate that low-relief landscapes can mobilize wildfire-related material to rivers similarly, though less-intensively and over shorter durations, than headwater regions. We propose that uneven mixing of heavily-impacted tributaries with high-order rivers may partially explain detection of wildfire signals in these large systems that may impact downstream water users.

1. Introduction

Communities within the wildland-urban interface of forested regions are experiencing more frequent and destructive wildfires (Hanes et al., 2018; Mell et al., 2010). This changing wildfire regime is associated with a warming climate that is intensifying dangerous fire weather and drought conditions (Diffenbaugh et al., 2015; Jolly et al., 2015), exposing these communities to increased risk of catastrophic impacts (Radeloff et al., 2018). Besides the evident consequences to lives, natural capital and infrastructure, wildfires often impact downstream surface water resources resulting from changes in runoff and degraded, increasingly variable water quality that can affect aquatic ecosystem functioning and drinking water treatability (Bladon et al., 2014; Emelko et al., 2011; Gresswell, 1999; Hohner et al., 2019; Smith et al., 2011; Writer et al., 2014).

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0043-1354/

Wildfires can enhance meteoric runoff from watersheds through chemical soil sealing and reduction in evapotranspiration due to vegetation removal (Larsen et al., 2009; Neary et al., 2005). Depending on fire severity and the timing of storm events, delivery of ash and sediment (Moody and Martin, 2001; Santín et al., 2015; Silins et al., 2009), organic matter and nutrients (Burd et al., 2018; Earl and Blinn, 2003; Lane et al., 2008; Rhoades et al., 2011), major ions (Bayley et al., 1992; Chanasyk et al., 2003; Mast et al., 2016), and trace contaminants (Kelly et al., 2006) into receiving waters can occur and cause deteriorated water quality (Bladon et al., 2014; Smith et al., 2011). Ensuing biogeochemical processing of this mobilized wildfire-related material can induce secondary ecological impacts on aquatic ecosystems resulting from eutrophication and sedimentation (Earl and Blinn, 2003; Gresswell, 1999; Martens et al., 2019; Silins et al., 2014). However, our understanding of fire-related impacts on surface waters has been overwhelmingly established from investigations conducted at relatively small catchment scales (e.g., <10s–100s km^2) and in catchments with responsive hydrologic connectivities between landscapes and receiving waters (Belillas and Rodà, 1993; Earl and Blinn, 2003; Gerla and Galloway, 1998; Inbar et al., 1998; Lane et al., 2008; Minshall et al., 2001; Neary et al., 2005; Prepas et al., 2003; Scott et al., 1998; Smith et al., 2011).

Landscape disturbances and related impacts on downstream river conditions are difficult to detect as watershed and river sizes increase (Blöchl et al., 2007; Elhadi et al., 1984). High-order river systems aggregate water and material from broad drainage networks and dampen biogeochemical signals from upstream landscapes through lateral mixing, dilution, and other in-stream processes (Temnerud and Bishop, 2005). Consequently, assessments of wildfire-related impacts on water quality in rivers draining diverse watershed sizes—especially very large ones—are scant. Watershed hydrologic connectivity also influences river response to landscape disturbance (Bracken and Croke, 2007). For instance, steeper topographic settings and well-drained soils can promote rapid hydro-geochemical connections between disturbed landscapes and rivers, and contribute to sustaining downstream and temporal wildfire impacts on water quality (e.g., Rust et al., 2018). Though a few studies have described wildfire impacts on water quality in larger (Burke et al., 2005; Emelko et al., 2016; Rhoades et al., 2011) or low-relief (Townsend and Douglas, 2000) watersheds, these investigations have been conducted in landscapes that are hydrologically well-connected. Critically, wildfire impacts on water are not well understood in large river systems with relatively low hydrologic connectivity, in which contaminants may be more efficiently retained on land surfaces conferring greater apparent storage or watershed buffering capacity (Ebel and Mirus, 2014; Prepas et al., 2003). These attributes are characteristic of many areas across the circumpolar Boreal Forest, where low topographic relief, heterogeneous glacial deposits and extensive peatland landscapes result in relatively low landscape-water coupling (Devito et al., 2017). While wildfires have been investigated at plot- and subwatershed-scales in these Boreal landscapes (e.g., Depante et al., 2018; Olefeldt et al., 2013; Prepas et al., 2003; Wilkinson et al., 2018), wildfire impacts on water quality at large basin scales in these regions have not been previously reported. This is a critical knowledge gap considering the high future probability of more severe fires occurring across the circumpolar Boreal Forest.

The primary objective of this study is to assess the impacts of the May 2016 Fort McMurray (Horse River) wildfire in northeastern Alberta, Canada on the water quality of large (>1,000 km^2 drainage areas) and low-relief (<5% drainage area slope) rivers in the region. Using a multi-tiered monitoring approach to target signal detection in these large rivers, we first compared suspended sediment, dissolved material (ion), nutrient, and metal concentrations and yields in rivers draining burned areas with an unburned control station. Second, at each station, we assessed if precipitation events and subsequent river flow mobilized more wildfire-related material relative to non-precipitation periods. Finally, we evaluated suspended sediment, dissolved material and flow relationships in rivers draining burned landscapes with pre-fire data from the same river. We hypothesized that rivers draining burned watersheds would be resistant to unbuffered overland flow and therefore show no detectable differences in water quality relative to rivers draining unburned landscapes or extensive historical data.

2. Methods

2.1. Study region

Our study region is contained within the Boreal Plains ecozone, which is characterized by flat and rolling landscapes with thick glacial till soils, and extensive coverage of wetlands and coniferous forests (Chanasyk et al., 2003; Devito et al., 2017). Due primarily to the Rocky Mountains barrier ~600 km to the west, much of this ecozone is relatively dry (419 mm annual precipitation at Fort McMurray; 1981–2010) with a cold, continental climate (Environment and Climate Change Canada, 2018). Human disturbances in the region are mostly from local urbanization in Fort McMurray and oil sands strip-mining and exploration-related impacts to landscapes (Fig. 1).

The Athabasca River drains approximately 140,000 km^2 of western Canada and occupies 23% of the province of Alberta (Fig. 1). This Strahler-order nine river is large and voluminous (~500 m wide; >500 m^3 s^{-1} mean annual flow) in its lower reaches and exhibits a spring snowmelt-freshet seasonality in flow (Fig. 2). In this portion of the river, concentrations of suspended sediment and associated water quality parameters (e.g., organic carbon, nutrients, metals) derived from upstream tributaries and channel re-suspension can be high relative to national water quality guidelines during much of the year (Glozier et al., 2018). Several high-order river systems empty into the lower Athabasca River including the Clearwater and Hangingstone rivers (Figs. 1 and 2).

The 2016 Fort McMurray wildfire (Fig. 1; May 01/16–August 02/17) was the most costly natural disaster in Canadian history ($8.9 billion in direct costs; Alam et al., 2018). By May 03/16, the wildfire advanced quickly from the southwest of Fort McMurray and triggered the dramatic evacuation of ~90,000 people, resulting in a loss of 3,000 structures and eventual consumption of ~600,000 ha of forest and peatland. Within the final fire perimeter (nearly established by May 17/16), 77% of the area was impacted by fire, half of which burned at a high severity (see supplementary material for methodology).

2.2. Monitoring and historic data

Post-fire river water quality monitoring in this study focused on three rivers: the Athabasca, Clearwater and Hangingstone. Three burned (impacted) stations were established: (1) on the Athabasca River upstream of Fort McMurray (left bank); (2) the Clearwater River near the mouth; and (3) the Hangingstone River at Fort McMurray (Fig. 1). These locations drain a gradient of watershed size (Hangingstone < Clearwater < Athabasca) and reflected a contrasting gradient of relative burned area and area burned at a high severity in their watersheds (Athabasca < Clearwater < Hangingstone; Table 1). An unburned (control) station was also initiated on the Athabasca River at Grand Rapids, upstream of the wildfire perimeter. Each of these four water quality stations incorporated multiple measurement approaches and, where possible, were co-located near meteorological and flow stations to better understand hydrological influences on post-fire water quality.

2.2.1. *Water quality measurements*

A multiple-method approach for sampling river chemistry of each of the four water quality stations was implemented after the fire to assess differences between unburned and burned locations (Fig. 1, Table S1). Nearly two weeks after the fire began, we deployed multi-probe data sondes (Hydrolab DS5X, OTT Hydromet, USA) just offshore at the two Athabasca stations (burned and unburned), and near mid-channel at the smaller Clearwater and Hangingstone stations. Two additional sondes were deployed downstream of Fort McMurray just offshore of each bank of the Athabasca River to assess river mixing conditions (Fig. 1). Each calibrated sonde recorded general water quality including water temperature, pH, specific conductivity, turbidity and dissolved oxygen at 15-min intervals. Sondes were retrieved and replaced with calibrated instruments every week to limit impacts from fouling and sensor drift. All sonde monitoring concluded in mid-October 2016.

In addition to the sondes, we collected daily water quality samples at the four main stations (Fig. 1, Table S1). Surface water autosamplers (Teldyne ISCO, USA) were deployed on the river bank at each station (burned: May–August 2016; control: June–August 2016) with water intakes placed mid-column just offshore at the two Athabasca locations, and mid-channel in the Clearwater and Hangingstone rivers. Each autosampler collected equal aliquots of river water every 3 h into 1 L ProPak® plastic bags in holders for a time-integrated sample each day. Weekly grab samples of surface waters were manually collected at each station from May to August 2016 to supplement automated sampling. After wading waist-deep at each location, water samples were collected at 30-cm depth directly into pre-cleaned or new plastic or glass bottles, depending on chemical methods. At the three burned stations (Athabasca-Fort McMurray, Clearwater-mouth, Hangingstone-Fort McMurray), grab sampling continued monthly through 2016, 2017 and 2018 via other programs.

All water samples were analyzed at accredited laboratories for suspended sediment, total ion (Na, K, Mg, Ca, Cl), total nutrient (total organic carbon, total Kjeldahl nitrogen, total phosphorus) and total metal (29-metal scan) concentrations. Only total measurements were considered for autosampler water samples to limit sample chemical transformation concerns during storage.

2.2.2. *Precipitation and river flow measurements*

To identify periods when wildfire-related material may be mobilized in burned watersheds, we delineated notable local precipitation events and subsequent river flow response periods at each water quality station. Total daily precipitation data from meteorological stations nearest to sampling locations (Fig. 1, Table S1) were used to identify noteworthy rainfalls equalling or exceeding 10 mm (94th percentile of daily rain events at Fort McMurray A, 1999–2015). Precipitation events at each station (see supplementary material) are defined in time here as the day of the noteworthy rainfall plus five subsequent days to encompass flow and biogeochemical responses of each river. Days not within a precipitation event period are defined as non-event periods.

Hourly and daily flow data before and after the wildfire were retrieved from hydrometric stations close to water quality locations, where possible (Figs. 1 and 2, Table S1). Discharge at the Athabasca River upstream of Fort McMurray (left bank) station was calculated by difference in measured daily flows between the nearby Athabasca River downstream of Fort McMurray and Clearwater River at Draper stations. Flow at Athabasca River at Grand Rapids was calculated using daily runoff yields from the upstream Athabasca River at Athabasca hydrometric station scaled to the drainage area at the Grand Rapids location (Table 1, Fig. 1).

2.2.3. *Historic water quality data*

Historic water quality data from long-term provincial and oil sands monitoring programs (2001–2016) were used to quantify pre- and post-fire water quality conditions at the three impacted stations (Table S1). Monthly surface water sampling has been ongoing in the Athabasca River upstream of Fort McMurray for many decades, while the Clearwater and Hangingstone rivers have been monitored for several years. Monthly grab sampling also occurred in 2015 (pre-fire) and 2016 (post-fire) at burned (High Hills, Christina, Clearwater, Hangingstone) and unburned (Calumet, Mackay, Ells, Firebag) river stations via regional monitoring networks (Table S1, Fig. S1a). Water collection and analytical methods from historical programs were consistent with those used during the post-fire sampling program.

2.3. *Quantitative analyses*

2.3.1. *Water quality assessment*

River turbidity and conductivity are typically negatively correlated during high-water events due to flow-related sediment concentration and ion dilution (Schlesinger, 2007). However, after wildfires, ash-laden runoff pulses during storms can trigger concurrent increases in turbidity and conductivity in receiving waters (Dahm et al., 2015; Earl and Blinn, 2003). To identify existence of concentration-dilution and concentration-concentration periods in each river, mean hourly sonde and flow data were included in an unconstrained principal components analysis (PCA; Canoco v.5.03; Biometris, The Netherlands). Six-hour covariances (using a 60-min moving window) of turbidity and conductivity were quantified at each station (including sonde-only locations) to directly compare the magnitude and frequency of suspected ash mobilization periods (i.e., positive covariances) between stations. We also performed a laboratory ash-mixing experiment to demonstrate how ash affected turbidity and conductivity measurements in an aqueous solution. We collected loose grey and black ash from the soil surface of burned forested plot (0.25 m^2) in the Clearwater River watershed (N56.6849; W-111.2560) on May 13/16, prior to post-fire rainfall events. Later, in the laboratory, we incrementally added ~500 mg of ash to 10 L of deionized water and continuously stirred the solution while turbidity, conductivity and pH were measured using a calibrated sonde.

Descriptive statistics were calculated to quantify differences in yields of selected particle-bound and dissolved chemicals between impacted and control river stations on the Athabasca River. To link assessments of selected chemicals at each station to other measured parameters, we correlated all parameter concentrations with concurrent measurements of flow, turbidity, conductivity and pH to organize parameters as particle-bound type (positive correlation with flow and turbidity) or dissolved type (positive correlation with conductivity and pH).

2.3.2. *Precipitation and flow assessment*

All water quality data (i.e., sonde and samples) at each station were binned into precipitation event ($\geq$10 mm) and non-event (<10 mm) periods for comparisons. This approach was used on sonde turbidity-conductivity covariance data and concentrations and yields of sampled chemicals. Assessment of hydrologic responses of impacted rivers before and after the fire is described in the supplementary material.

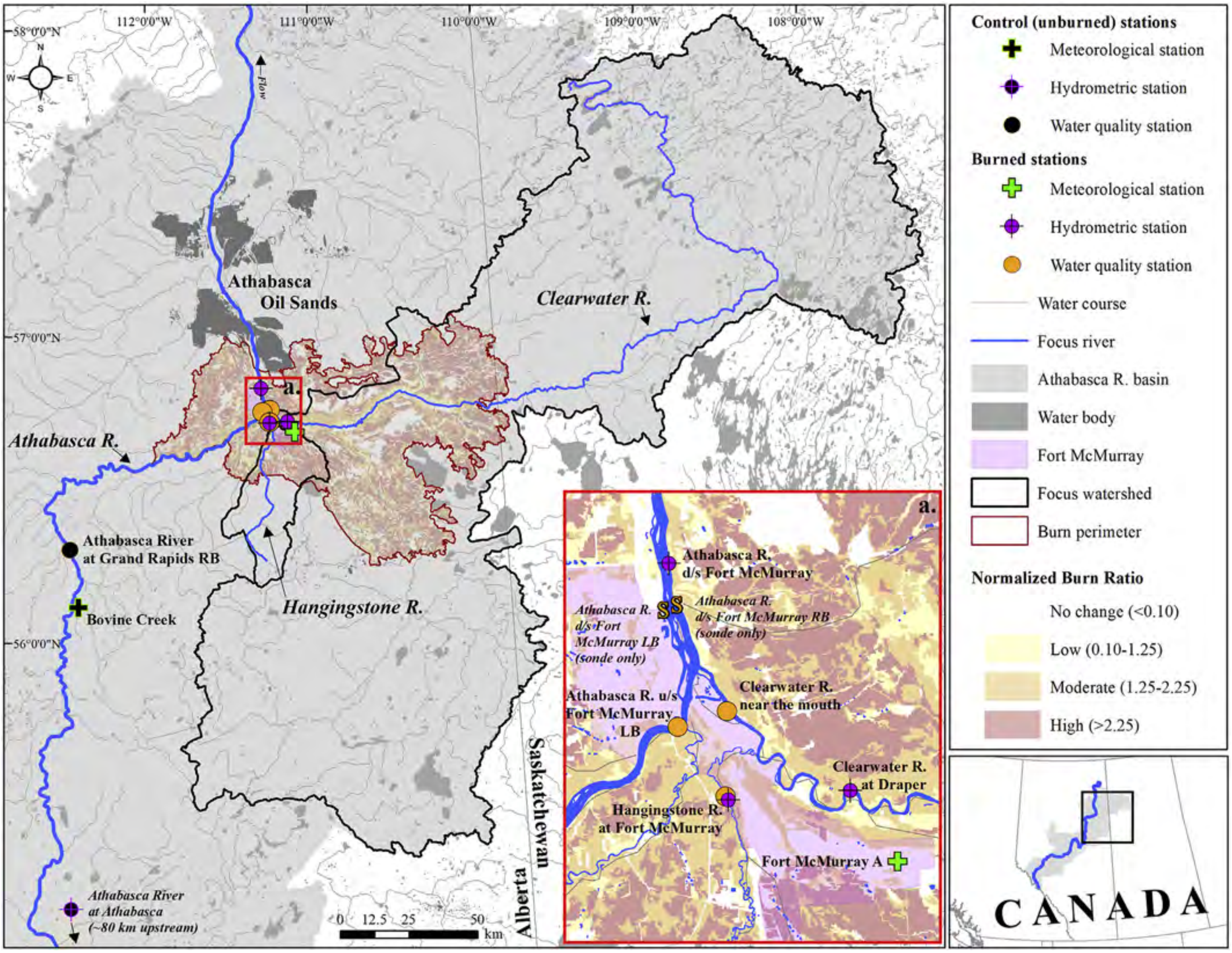

Fig. 1. Map of the lower Athabasca River in northeastern Alberta showing the extent and severity of the May 2016 Fort McMurray wildfire (see supplementary material for methods) and water monitoring stations. Inset (a.) shows monitoring station clustering around Fort McMurray.

2.3.3. Historic water quality assessment

Post-fire changes in particle-bound and dissolved river water chemistry, relative to historic records, were evaluated in the three rivers draining burned watersheds (Table S1) using concentration-discharge (C-Q) relationships. Monthly, pre-fire water quality data (suspended sediments as a proxy of particle-bound chemicals, total calcium as a proxy for specific conductivity/dissolved chemicals; see Table S2) from each river station were log-transformed and linearly regressed with log-transformed, pre-fire daily discharge data. Subsequently, 95% prediction intervals were calculated (Sigmaplot v13, Systat, USA) to serve as a boundary of what C-Q relationships would be expected in the river based on historic conditions. When post-fire C-Q relationships (2016–18) were compared with historic relationships, data points that were outside of the bounds of the prediction intervals were considered noteworthy. Other monthly grab sample and flow data from four unburned and four burned river stations were used in a repeated measures ANOVA (SPSS v.25; IBM, USA) to assess lower-resolution pre-fire (2015) and post-fire (2016) impacts on water quality.

3. Results

3.1. Sonde water quality

High-frequency water quality monitoring by sondes at the four main stations typically showed flow-concentration for turbidity and flow-dilution for conductivity (Fig. 3) and pH, as well as contrasting seasonal changes for water temperature and dissolved oxygen. These trends were evident in PC1 and PC2 scores of the PCA (Table S3). However, sharp peaks in turbidity and conductivity together occurred during short periods (hours to days) often along the rising limb of the hydrograph at burned stations only. The third principal component at each burned station reflected this covariation between turbidity and conductivity in the absence of association with flow. At the Athabasca River control station, PC3 scores also showed periods of concurrent increases in turbidity and conductivity, but were positively associated with flow, perhaps suggesting an influence of saline groundwater (Gibson et al., 2016). Mean 6-h covariance scores of turbidity and conductivity more distinctly showed differences between stations than did the PCA. Covariances from burned stations were 3–25 times higher than the

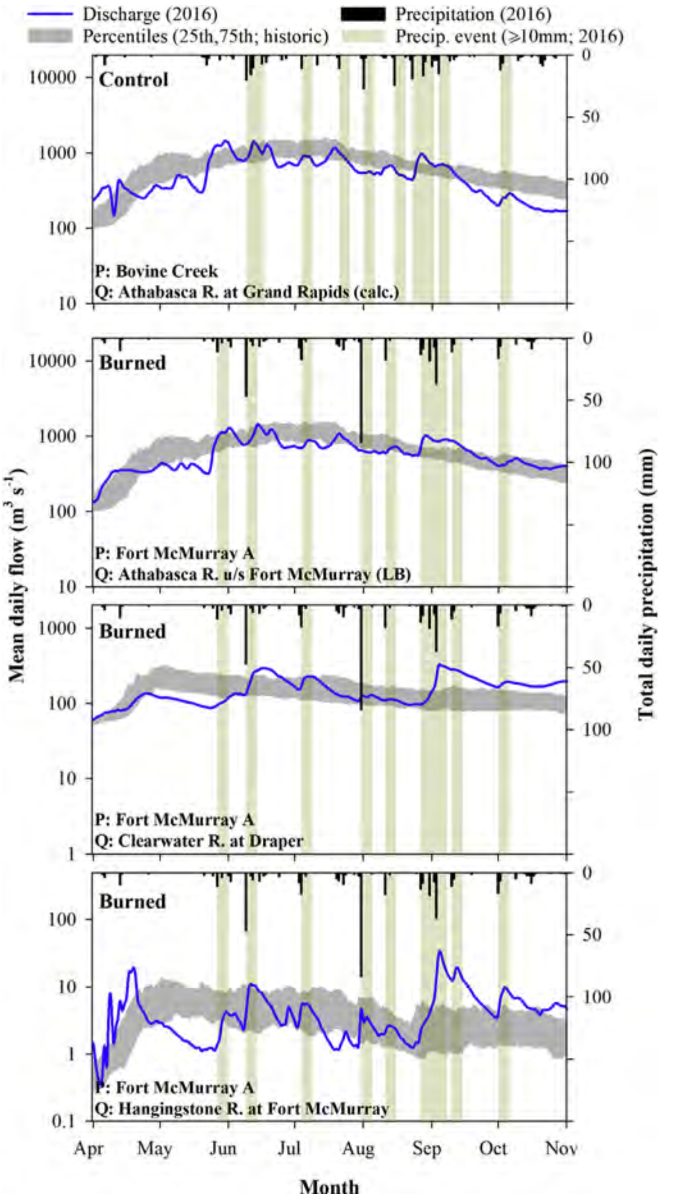

Fig. 2. Log-scaled historic flow quantiles and 2016 mean daily hydrographs and hyetographs of monitored hydrometric stations (Q) and their closest meteorological stations (P) of unburned (top panel) and burned watersheds in the lower Athabasca River of northeastern Alberta. Daily precipitation events ≥10 mm and five subsequent days are highlighted in yellow. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

control station, though means were influenced by the highest values (Fig. 4). Highest covariances at impacted stations typically occurred during precipitation events (mean: 14–113) and were 2–22 times higher than non-event periods. These observations were consistent with results of the ash mixing experiment that demonstrated concurrent increases in turbidity, specific conductivity and pH as ash was added to deionized water (Fig. S2). Further, though pre-fire grab sampling in impacted rivers showed consistent negative associations between turbidity concentrations and specific conductivity, 11–38% of post-fire, precipitation event turbidity and conductivity samples were greater than the upper historic 95% prediction interval, suggesting a more positive association between the parameters during those periods (Fig. S3).

Within the burned reach of the Athabasca River, turbidity and conductivity covariances on the left bank decreased from a mean of 7.7 (maximum: 1,071) upstream to a mean of 7.0 (maximum: 574) downstream. However, the downstream right bank station observed much higher covariance values (mean: 17.2; maximum: 3,782) than either station on the left bank, which together suggests an unmixed chemical profile longitudinally and laterally in the river (Fig. 4).

3.2. Sample water quality

Daily automated and grab sample water quality results indicated only slight differences between burned and control stations when aggregated to the entire sampling season (Fig. 5). For example, seasonal mean daily sediment yields (June 22–August 30/16) from the directly comparable stations of the Athabasca River at Grand Rapids (unburned; GR) and the Athabasca River upstream of Fort McMurray (burned; FMM) were within one standard error of each other (GR: 0.68 ± 0.10 kg ha^{-1} d^{-1}; FMM: 0.75 ± 0.09 kg ha^{-1} d^{-1}). Results of other sediment-associated yield comparisons including total lead (GR: 8.1 ± 1.0; FMM: 10.3 ± 1.8 mg ha^{-1}), total Kjeldahl nitrogen (GR: 3.8 ± 0.3; FMM: 3.3 ± 0.2 g ha^{-1} d^{-1}), and total phosphorus (GR: 0.54 ± 0.07; FMM: 0.45 ± 0.04 g ha^{-1} d^{-1}) were consistent with the suspended sediment results post-wildfire and likely other correlates (Table S2). Similarly, monthly suspended sediment yields across a subset of several rivers draining burned and unburned watersheds in the region before and after the fire indicated no fire effect (Fig. S1b). Total calcium, used as a proxy measure for dissolved material (Table S2), also showed similar yields between the impacted (0.23 ± 0.01 kg ha^{-1} d^{-1}) and the control station (0.21 ± 0.01 kg ha^{-1} d^{-1}) on the Athabasca River (Fig. 5). Differences in monthly calcium yields between a subset of rivers draining burned and unburned watersheds were also statistically-similar (Fig. S1b). Alternatively, total organic carbon (TOC), most of which is in the dissolved form in the Athabasca River (Glozier et al., 2018), had mean yields greater at the control (91 ± 5 g ha^{-1} d^{-1}) relative to the impacted (67 ± 3 g ha^{-1} d^{-1}) station.

Substantial differences occurred within and between stations when classifying data by precipitation event and non-event periods. At impacted stations only, peaks of concentrations (Fig. 6) and yields during precipitation events were observed relative to non-event periods for suspended sediment (concentrations: 1.7–6.6 times higher; yields: 1.8–10.3x), total phosphorus (1.9–2.6x; 2.0–5.3x), total Kjeldahl nitrogen (1.3–1.4x; 1.2–2.4x),

Table 1
Drainage area and burn metrics at water quality stations operating on rivers draining unburned (U) and burned (B) areas following the 2016 Fort McMurray wildfire.

Station	Watershed area (km^2)	Mean watershed slope (%)	Burned area: watershed area (%)	High severity burned area: burned area (%)
Athabasca R. at Grand Rapids (U)	94,464	3.3	0	–
Athabasca R. u/s Ft. McMurray (B)	98,013	3.3	0.4	45.6
Clearwater R. near the mouth (B)	31,936	3.4	10.2	52.1
Hangingstone R. at Ft. McMurray (B)	903	3.5	21.5	57.1

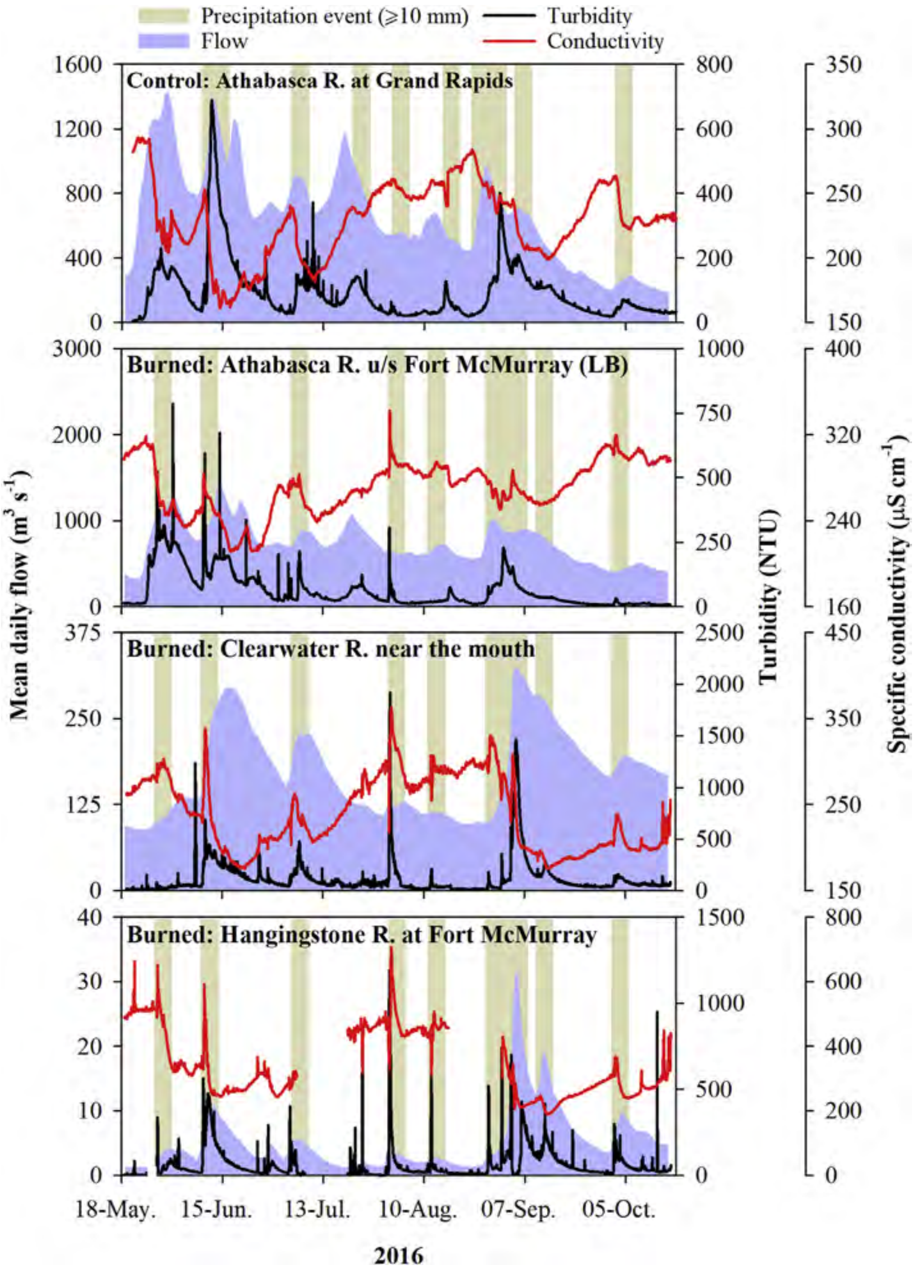

Fig. 3. Hourly time series of sonde turbidity and specific conductivity, and discharge in unburned (top panel) and burned river stations in the lower Athabasca River region following the 2016 Fort McMurray wildfire. Precipitation events (daily ≥10 mm + 5 days) are highlighted in yellow. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

and total lead (2.1–3.3x; 2.0–6.3x). In contrast, such ratios at the control station were lower than impacted stations and were often near unity (concentrations: SS = 0.9, TP = 1.3, TKN = 0.9, Pb = 1.1x; yields: SS = 1.0, TP: 1.4; TKN: 0.9; Pb: 1.2x). Total calcium and TOC yields were largely unchanged between precipitation and non-event periods (0.8–1.1x) at all stations except for the burned Hangingstone River (Ca: 1.7x; TOC: 1.5x).

Despite observing measureable impacts to water quality at burned stations, most monitored parameters rarely exceeded national surface water quality guidelines for the protection of aquatic life (Government of Canada, 2019), or the relative number of exceedances were comparable to pre-fire conditions or those at control stations (Table S4).

3.3. *Historic water quality*

Elevated suspended sediment concentrations observed at burned stations did not result from higher flows than sampled historically, nor wholesale changes in the hydrological conditions of impacted river systems (see supplementary material; Figs. S4–S6). Specifically, the prediction intervals calculated using historic C-Q relationships for suspended sediment from the burned Athabasca River upstream of Fort McMurray station were exceeded on four occasions after the wildfire (Fig. 7). Two of these exceedances occurred on June 09 and July 31/16, when the two largest recorded rainfalls in the Fort McMurray region occurred during that summer. Sampling during subsequent years showed no similar notable concentrations. Five exceedances of the upper historic prediction interval occurred in the

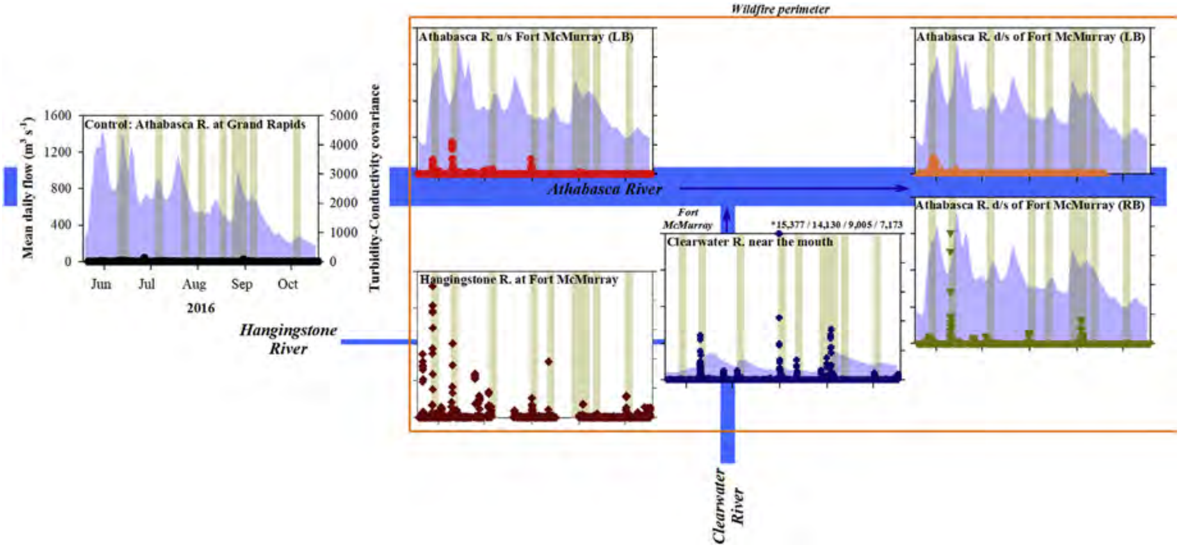

Fig. 4. Schematic of sonde turbidity and specific conductivity covariance, mean daily flow and precipitation events (≥10 mm and five subsequent days) measured at an unburned control station and five downstream stations within burned landscapes. All graphs share the same axis metrics and time periods as the control station graph. LB and RB denote left bank and right bank of the river. *denote four, 15-min measurements beyond the scale on the covariance axes.

Clearwater River during precipitation events in 2016, but none during the following years. The Hangingstone River experienced 17 exceedances relative to its historic concentration-discharge relationship. However, these exceedances occurred during precipitation event and non-event periods in 2016, as well as across multiple years after the fire, including during spring runoff in 2017. These C-Q results were likely similar to other particle-bound measures (Table S2). Compared to suspended sediment concentrations, only four exceedances of the upper historic prediction interval for total calcium concentration were observed across all rivers, and likely similar for other dissolved-type chemicals (Table S2). Collectively, suspended sediment concentration exceedances of historic 95% prediction intervals in impacted rivers became more frequent as watershed size decreased, and the proportion of burned area and area burned at high severity across watersheds increased (Table 1, Fig. 7).

4. Discussion

4.1. Episodic detections of fire-related material in large river systems

Despite (a) naturally turbid and organic-rich water qualities prior to disturbance (Figs. 3, 5 and 6; Glozier et al., 2018), (b) the potential for considerable dilution including from substantial subsurface contributions to regional river flow (Gibson et al., 2016), and (c) poor hydrologic connectivity between the landscapes and low-relief rivers (Chanasyk et al., 2003; Devito et al., 2017; Gibson et al., 2016; Ireson et al., 2015), wildfire impacts on river water quality were detectable at the very large basin scale following the 2016 Fort McMurray wildfire. Three independent monitoring approaches reported brief, precipitation-related increases in wildfire-related material (i.e., suspended and dissolved material, nutrients, metals; Burton et al., 2016; Earl and Blinn, 2003; Reale et al., 2015) in rivers draining burned watersheds relative to a control station or historic, pre-fire data (Figs. 3–4,6-7). These findings were also supported by an ash mixing experiment (Fig. S2) and other historic data. Together, this weight-of-evidence suggests rainfall-induced surface erosion delivered ash and soil from local, burned hillslopes into large rivers that briefly impacted water quality without shifting overall metrics of river water quality relative to unburned regions (Fig. 5). The role of precipitation events in mobilizing particle-bound chemicals across burned landscapes is well-established in the literature (e.g. Bladon et al., 2014; Burton et al., 2016; Langhans et al., 2016; Townsend and Douglas, 2000), and in more extensively-burned regions of the Boreal Plains (Burke et al., 2005). However, detection of suspended chemicals in large, wetland-dominated, low-slope river watersheds with low proportions of areas burned (Table 1) was a key outcome of this study. Alternatively, total measurements mostly in dissolved form (e.g., total calcium, total organic carbon; others; Table S2) showed inconsistent changes between impacted and control stations, as well as between precipitation and non-event periods, perhaps reflecting uneven burning of peat at depth where non-erosive flows dominate (Devito et al., 2012; Wilkinson et al., 2018).

4.2. Factors shaping wildfire signal detection in large river systems

Following wildfire, loss of ground cover, decreased surface roughness, and soil crusting and sealing (Larsen et al., 2009) can impede infiltration and expose soils to splash and fluvial erosion (Shakesby and Doerr, 2006). These processes often allow storms to mobilize loads of sediment, nutrients and contaminants downstream (Bayley et al., 1992; Betts and Jones, 2009; Moody and Martin, 2001; Neary et al., 2005). However, the magnitude and duration of these water quality impacts are shaped by the intensity of precipitation events and the erosive potential of impacted landscapes (Langhans et al., 2016). Headwater systems, with typically small, extensively-burned, steep and flashy landscapes, often show substantial sediment yields from burned watersheds relative to those unburned (typically >100x; Smith et al., 2011) with water quality impacts often detected for months to years (e.g., Inbar et al., 1998; Owens et al., 2013). In this study of lower-relief and wetland-dominated landscapes, catchments are large, burn less-extensively and runoff storage is prevalent (Table 1; Devito et al., 2012). These conditions resulted in suspended sediment yield ratios (impacted:control) ≤1.3x with detectable water quality impacts at only hourly and daily scales (Figs. 3–4,6). However, fire-related impacts on water quality in this study (Table 1, Fig. 6), as well as those in headwater regions (Rhoades et al., 2011) demonstrate that differing basin sizes, proportions of area burned, and area burned at high

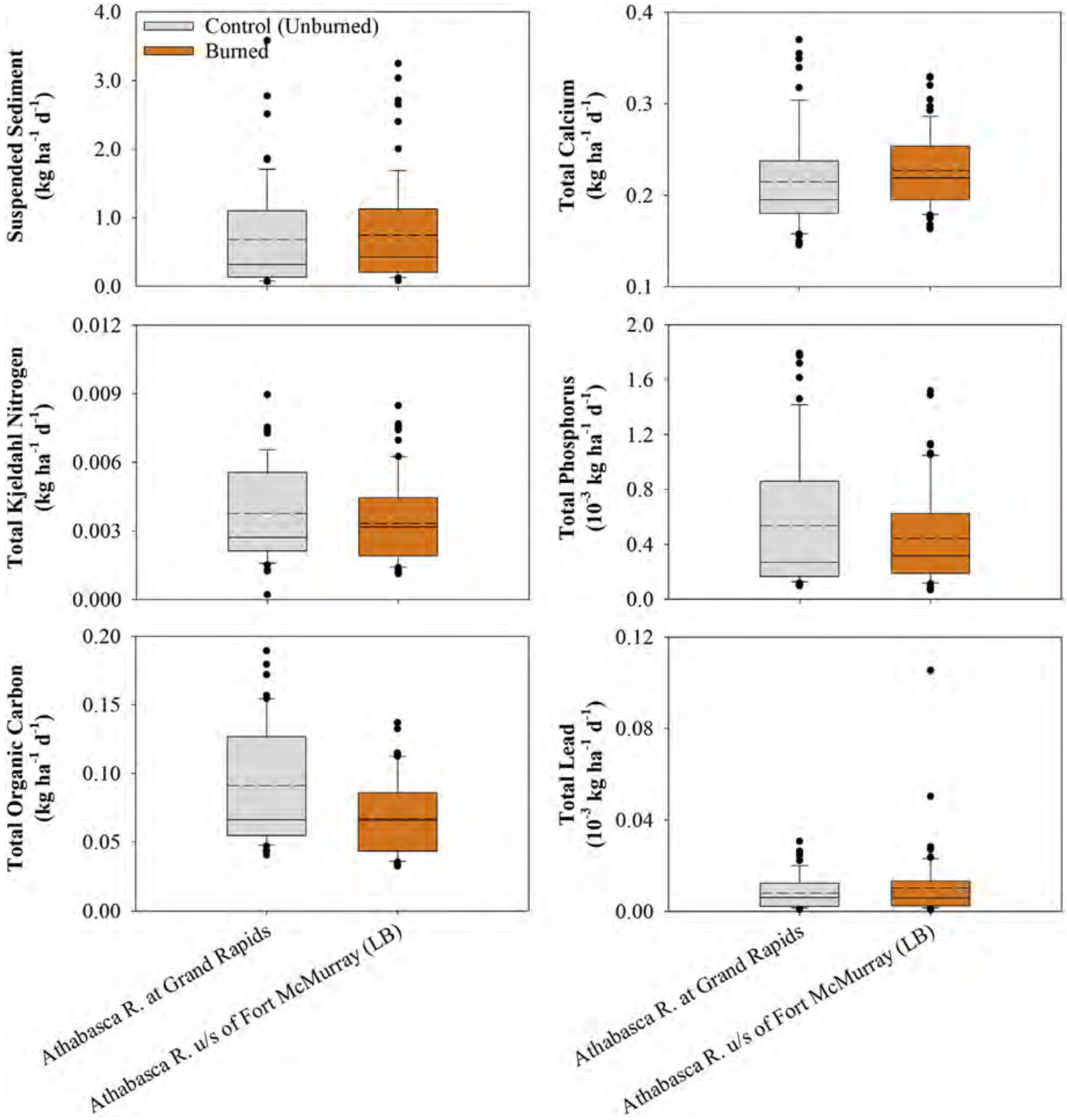

Fig. 5. Box and whisker plots (median-mean [dashed]-25/75–10/90 percentiles-outliers) of Athabasca River watershed yields of selected chemicals from an upstream unburned station and a downstream burned station from daily sampling (May–August) following the 2016 Fort McMurray wildfire.

severity, can impact water quality of river systems differently. This suggests that mobilization of fire-related materials downstream is likely driven by similar processes across differing landscapes, but are shaped by catchment- and fire-related characteristics. However, we postulate that incomplete hydro-chemical mixing may also play an important role in wildfire signal detection in these large rivers expected to buffer landscape chemical signals (Blöchl et al., 2007).

Large rivers can substantially modify upstream and locally-derived terrestrial material via mixing, dilution and sedimentation processes (Temnerud and Bishop, 2005). However, we suggest that the complexities of mixing river waters of different qualities and quantities may have also been responsible for the detection and downstream persistence of wildfire chemical signals observed in these large rivers, particularly in the Athabasca River. Other studies have reported a persistence of plumes/mixing zones downstream of river confluences because of differences in water densities, particle loads, or the physico-chemical composition of sediments (Bouchez et al., 2010; Herrero et al., 2018; Stone et al., 2011). Measurements assessing cross-sectional changes in water quality of the Athabasca River suggest incomplete mixing occurs well downstream of anthropogenic or tributary inputs (Glozier et al., 2018). Following the wildfire and precipitation events, we observed that both low- and high-order tributaries draining severely burned watersheds delivered plumes of ash to the Athabasca River that were visible for several kilometers downstream of confluences (Fig. 8). Empirical evidence from the three Athabasca River sondes supported these visual observations. For example, we observed strong differences in turbidity, conductivity, and turbidity-conductivity covariance values measured on the left and right banks of the Athabasca River at the same longitudinal location (Fig. 4). Higher turbidity values on the right bank were coincident with turbidity and conductivity spikes in the smaller but more

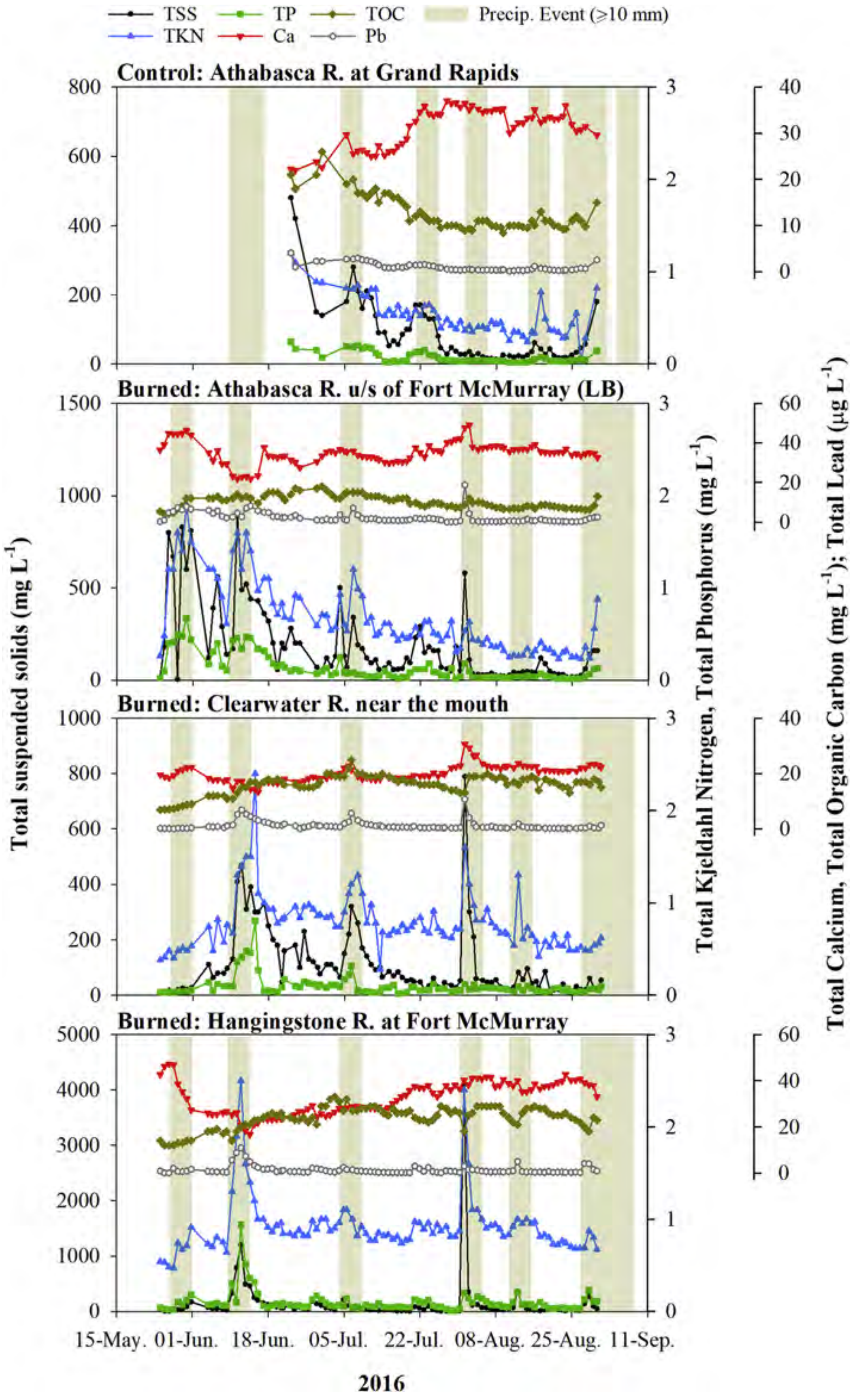

Fig. 6. Time series of mean daily concentrations of selected chemicals collected by automated and grab samples for unburned (top panel) and burned river stations in the lower Athabasca River region following the 2016 Fort McMurray wildfire. Precipitation events (daily ≥ 10 mm + 5 days) are highlighted in yellow. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

extensively-burned Clearwater and Hangingstone rivers that empty upstream from the right bank. The entry of these rivers established a distinct, fire-impacted plume on the right bank of the Athabasca River through to where sonde measurements occurred. Though it was unclear where the right bank plume mixed fully with the Athabasca River downstream, we did observe likely mixing and dilution of fire-influenced plumes on the left bank of the Athabasca. Near the upstream of Fort McMurray station, several severely-burned low-order streams entered the Athabasca River on the left bank and passed though the sonde just downstream, resulting in

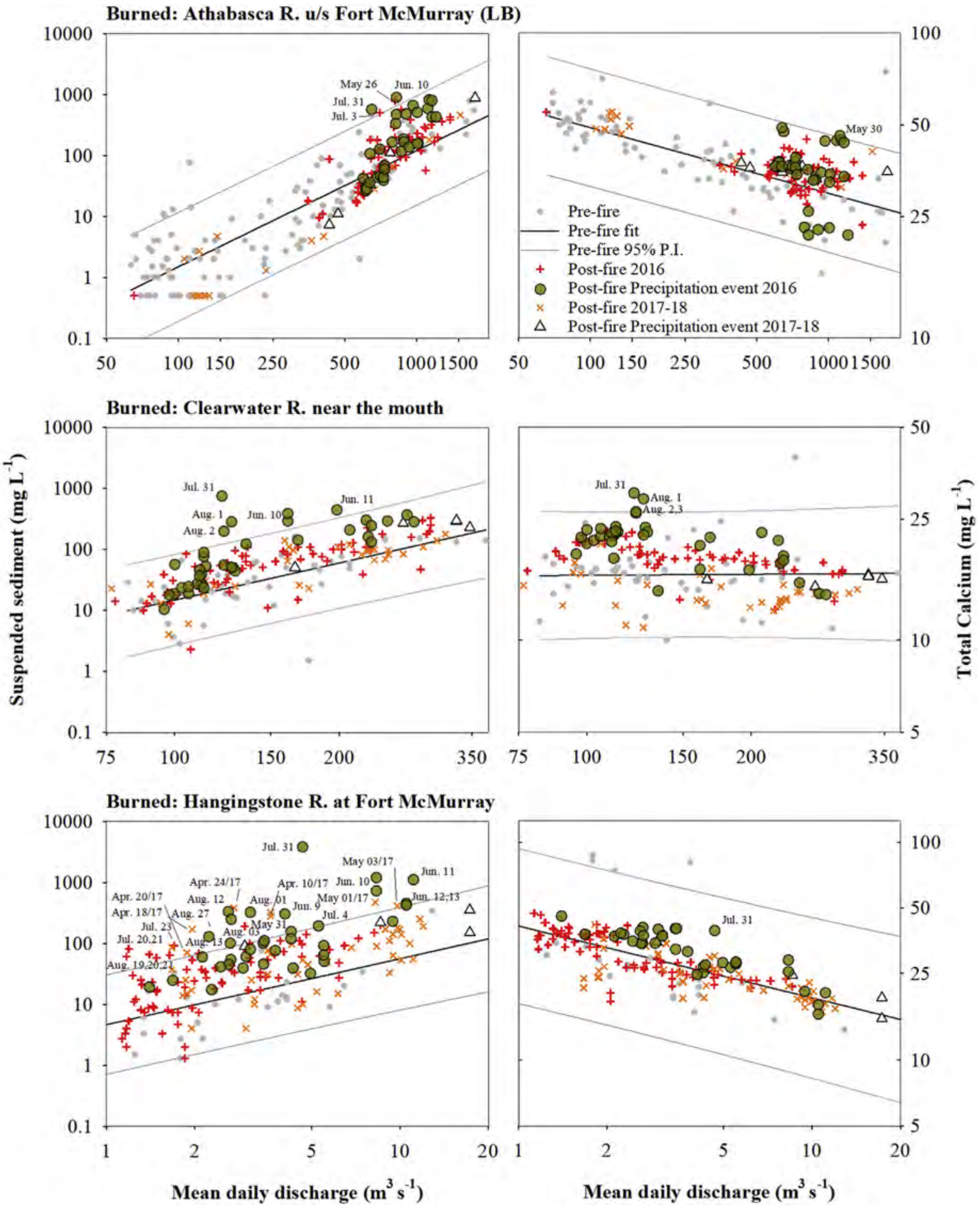

Fig. 7. Log-scaled concentration-discharge relationships of suspended sediment and total calcium (Ca) of historic and post-fire data from three rivers draining burned watersheds in the lower Athabasca River region following the 2016 Fort McMurray wildfire. 95% prediction intervals and linear fit lines on historic data are also shown.

defined, precipitation-related turbidity and conductivity spikes up to 1000 NTU and 350 μS cm^{-1}, respectively (Figs. 3 and 8). Downstream of Fort McMurray on the same bank, another sonde measured turbidity and conductivity spikes often 10–60% as large with lower turbidity-conductivity covariances (Fig. 4) compared to the upstream location, and similar to the unimpacted Grand Rapids station. These results suggest efficient mixing of the small, fire-impacted plumes on the left bank with larger volumes of unimpacted Athabasca River water over just a few kilometers of this reach. Regardless, this uneven mixing of waters from impacted tributaries with the large Athabasca River may simultaneously explain observed downstream persistence of wildfire signals, but also overall signal attenuation due to dilution (i.e., relative to headwater regions), both of which were defining observations of this study.

Fig. 8. Photos of high-order (left panel; Horse River) and low-order (right panel; Little Fisheries Creek) rivers draining recently burned watersheds and discharging into the Athabasca River just upstream of Fort McMurray after a June 2016 precipitation event. (Credit: S. Hustins).

4.3. Human-related impacts of wildfires in large river systems

The occurrence of and damage from wildland-urban interface fires has increased over the previous decade in western North America (Schoennagel et al., 2017; Westerling et al., 2006). As such an event, the Fort McMurray wildfire burned regions both within and surrounding the remote urban center of Fort McMurray, Alberta. However, this wildfire presented a unique situation where very large, low-relief watersheds were expected to buffer wildfire impacts on local surface water resources. Interestingly, though we observed relatively muted water quality changes in these very large rivers compared to headwater regions, these arguably inconsequential basin-scale effects still resulted in impacts to post-fire drinking water treatment costs borne by the local municipality, similar to headwater regions (Bladon et al., 2014; Emelko et al., 2011; Hohner et al., 2019; Thurton, 2017). Preliminary analyses (unpublished) suggest unmixed plumes in the Athabasca River from heavily-impacted, low-order tributaries discharging upstream of the drinking water treatment plant intake on the river may have been responsible. Thus, exclusive focus on the magnitude and persistence of the impacts of wildfire on surface water quality may fail to uncover its true stresses on surface water resources. This consideration may be particularly applicable to future wildland-urban interface fires as export of complex chemicals (e.g., plastics) from residential, commercial, and industrial settings may present unique water quality conditions in receiving waterbodies of human importance (Oliver et al., 2012).

5. Conclusions

This study presents a multi-tiered approach to monitoring the flow and water quality of large rivers in a low-relief region affected by an expansive and severe wildfire. While broader wildfire impacts to water quality of regional rivers were markedly lower than commonly reported in smaller watersheds of greater relief, water quality impacts were nonetheless observed across a range of very large river basin scales. Novel use of continuous flow and general chemical monitoring was key to identifying impacted periods in the rivers studied herein, similar to few other investigations (Cooke et al., 2016; Dahm et al., 2015; Mast et al., 2016). In particular, our study emphasizes the importance of considering the spatial frame of reference (entire basin vs. local) when evaluating wildfire impacts on water quality in large river basins. The proximity and scale of this and other recent wildfires to major urban municipalities further highlights the importance of shifting North American wildfire regimes as increasing threats to community surface water resources.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.watres.2020.116071.

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Wildfire: Its effects on drinking water quality

Wildfires can impact water sources used for our drinking water. For example: nearby streams, rivers and lakes. When wildfire affects the surrounding trees, soil and vegetation, water quality may decline.

During a wildfire, it's important that you have access to enough reliable and safe drinking water. Wildfire smoke can have a harmful effect on your body. Drinking lots of water helps reduce inflammation and protects you from the effects of wildfire smoke. For more information on wildfire and your health, read HealthLinkBC: Wildfires and your health.

How can wildfires affect drinking water quality?

Possible effects of wildfires on drinking water include:

- Changes in the amount and timing of snowmelt and runoff from storms
- Changes in water quality (e.g. nitrate, ammonia, phosphate, total suspended solids, turbidity, temperature) from build-up of ash, soil erosion and fire debris
- Algal blooms in water sources. For more information on cyanobacteria blooms (blue-green algae), read HealthLinkBC File #47 Cyanobacteria blooms (blue-green algae)
- Changes in taste, colour and smell of drinking water
- If fire retardant is present, there may be a possible rise in soil and water chemical levels, such as: ammonium, phosphate, nitrate, and nitrite. Read HealthLinkBC File #05a Nitrate in well water and Ministry of Environment and Climate Change Strategy: Water quality: Fire retardants used to fight wildfires.
- Chemical contamination of drinking water if pipes, meters, and wellheads are burned or exposed to heat. If pipes are damaged and leak, there is also potential for bacterial contamination.

These impacts can make it harder for the local water treatment system to work. They can also cause contamination of water after it's treated.

How do I know if my water quality has been affected?

If you are on a community water system, your water supplier should check your water system and the quality of your drinking water. If there are concerns, they should speak to you.

If your drinking water comes from your own private well or surface water source (e.g. lake), you should check to make sure your system still works properly. A wildfire could cause it to lose power, pressure or integrity. Any of these issues could contaminate the system or lead to stagnant water.

Some signs your water supply may have been impacted by wildfires include:

- Fires at, or upstream, of your water intake
- Changes in water appearance, clarity, colour, smell and/or taste
- Electricity/power outages or fire damage to structures (e.g. building, water intake valve, water well head, treatment system, piping, etc.)
- Burning at or close to your well or piping

Water lines may need to be flushed and water supplies may need to be tested for bacteria and chemicals before being used. In some cases, repairs or replacement of damaged pieces may be required.

Can fire retardant or foams affect my drinking water?

The use of fire retardants and foams to fight wildfires is common in B.C. When good practices are followed, there is little risk to human health and the environment.

Most fire retardants and foams used in B.C. are made of at least 90 percent water and do not contain any PFAS chemicals. Foams are golden-brown and have an orange blossom scent. They may appear liquid. Fire retardants are nitrogen-based and contain small amounts of additives that help suppress fires. If you see a plane dropping fire retardant on a fire, you may notice it is red coloured. This is because dyes are added to the retardant, which helps firefighters see where it lands.

The amount of fire retardant or foams that get into drinking water sources is very small, and levels drop quickly over time. Foams typically get burned-off when used on open flames, making it unlikely that they will enter a drinking water source. Neither of these are human health concerns.

To make sure water is safe, drinking water from sources where fire retardant chemicals were used should be checked to ensure that it meets the Guidelines for Canadian Drinking Water Quality.

People and pets should avoid direct contact with fire retardants and foams. If inhaled or ingested, immediate first aid is likely not required. But, you should contact a physician or B.C. Drug and Poison Control Center for advice at www.dpic.org/ or call 1-800 -567-8911.

If you get fire retardants or foams on your skin or clothing, wash the area with soap and water, and launder any clothing before wearing it. For more information about health and safety around fire retardants, see BC Centre for Disease Control Wildfires notes: Fire retardants/suppressants.

What are the long-term impacts to my water quality?

It is hard to predict the long-term impacts of wildfires on drinking water quality. For some communities, problems may appear long after the wildfire is over (i.e. during intense rainfall events in the fall/winter months). Burned land and forests near water can cause large amounts of hanging or dissolved material (e.g. ash) to wash into downstream drinking water supplies. The following problems may impact a drinking water system long after the wildfire is over:

- More debris in water reservoirs, causing damage and higher maintenance costs
- More algal blooms in reservoirs, causing health effects, taste and odor
- Increased turbidity (water cloudiness). This may lead to larger amounts of sludge stuck in water filters or more chemicals needed for water treatment, all of which would raise operating costs
- Changes in water chemistry, or increased amounts of iron and manganese . These may lead to the need for further treatment
- Damage to water infrastructure like pipes, meters, or wellheads, which need to be repaired or replaced

What can I do if my drinking water has been affected by a wildfire?

- Do not drink, prepare food, or wash with tap water until officials say the water source is safe
- If you are on a well or cistern that has been damaged, assume the water is not safe to drink
- Do your best to conserve water as the water supply may be limited due to power outages or other impacts for an unknown amount of time
- If you have questions about the quality of your drinking water, ask your local water supplier (e.g. municipality, utility provider, well owner, etc.).
- If your water is not safe to drink, you may need to use a different source of drinking water or disinfect tap water. Boil or use disinfection tablets until the water source can be assessed and health officials have assured you that it’s safe for drinking. For more information, see HealthLinkBC File #49b Disinfecting drinking water

- Water lines may need repairing and flushing to remove contaminated water
- Test affected wells or surface water (e.g. lakes, rivers, streams, etc.) to ensure it meets the water quality standards in Guidelines for Canadian Drinking Water Quality. For more information, see www.canada.ca/en/health-canada/services/environmental-workplace-health/water-quality/drinking-water/canadian-drinking-water-guidelines.html
- For information on testing your private water source, refer to the list of Provincial Health Officer Approved Drinking Water Testing Laboratories at http://lmlabs.phsa.ca/Documents/8%203%201%20PHO%20Approved%20Laboratory%20List%2020231231.pdf. For more information, read HealthLinkBC File #05b Well water testing.

For more information

To report a wildfire or for the latest information on the current wildfire situation in B.C., see Public Safety & Emergency Services – Wildfire service at www2.gov.bc.ca/gov/content/safety/wildfire-status.

For information about protecting your community from wildfire, read FireSmart's manual at https://firesmartbc.ca/resource-types/guides-manuals/.

For information about how to prepare in advance for a potential evacuation alert or order, read PreparedBC - Get prepared for a wildfire www2.gov.bc.ca/gov/content/safety/emergency-management/preparedbc/know-your-hazards/wildfires.

For water and food safety information when you return home after a wildfire, read Recover after a wildfire www2.gov.bc.ca/gov/content/safety/emergency-management/preparedbc/know-your-hazards/wildfires/after-wildfire.

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PROJECT NO.
4590

Wildfire Impacts on Drinking Water Treatment Process Performance: Development of Evaluation Protocols and Management Practices

Wildfire Impacts on Drinking Water Treatment Process Performance: Development of Evaluation Protocols and Management Practices

Prepared by:

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EXECUTIVE SUMMARY

KEY FINDINGS

- Post-fire water quality was simulated by heating soil and litter samples in a furnace. Following heating, the samples were leached in low-carbon tap water and the character of the dissolved organic matter was assessed. Bench-scale treatment tests were performed to evaluate the treatability of the leachates.
- Soil and litter released different quantities and qualities of dissolved constituents following heating. In general, low temperatures resulted in enhanced mobilization of carbon from soils, and lower for litter. At higher temperatures, both soils and litter released less carbon.
- The leachates consistently exhibited an overall poor response to coagulation and, even at high coagulant doses, often marginal dissolved organic carbon removal was achieved. Utilities should plan for higher coagulant doses, and the subsequent solids handling implications on downstream processes. Coagulant doses will likely be case specific depending on the wildfire- and watershed-specific factors, and post-fire flow events.

BACKGROUND

The frequency and intensity of wildfires has increased in recent decades, and this trend is expected to continue in the near future, especially in areas where climate change is predicted to result in warmer, drier conditions. In particular, the western United States has observed an increase in the frequency, duration, and amount of burned area from wildfires. Extreme droughts, higher temperatures, earlier snowmelt, and changes in precipitation patterns can all contribute to the likelihood of wildfires. Other factors influencing wildfire occurrence include land use changes, such as livestock grazing and fire suppression. Consequently, wildfires are of increasing concern, and their resulting impacts on the environment must be further investigated.

As wildfire frequency increases, the potential effects on forested watersheds, which commonly serve as high-quality drinking water sources for many communities, become a concern. Drinking water utilities that rely on these water sources are considering the potential impacts of wildfires in their watersheds. These impacts include water quantity and availability, source water quality, and the ability to effectively treat and provide the high-quality water that the public demands. Currently, there are a limited number of reported case studies where post-wildfire water quality and treatability were monitored at drinking water utilities. Therefore, there is a need to better understand the effects of wildfires on source water quality and treatability, while also considering the effects on treatment plant operations and costs.

OBJECTIVES

The overarching objective of this project was to expand the knowledge base regarding the effects of wildfire on drinking water quality, treatment, plant performance, and operations. In order to meet this objective, this project focused on three main aspects. First, in order to evaluate the effects of a wildfire on a particular treatment operation, an approach to simulate the effects of a wildfire on water quality was developed. Second, simulated post-fire runoff was treated using mostly conventional processes. Lastly, an evaluation of the best treatment practices to deal with wildfire-impacted source waters was conducted. An additional objective of this project was to extend the post-fire water quality monitoring at a water intake in a burned watershed.

This project included the collaboration and support from the following utilities:

- Denver Water (DW)
- City of Westminster, CO (WM)
- City of Northglenn, CO
- City of Thornton, CO
- San Francisco Public Utilities Commission (SFPUC)
- New York City Department of Environmental Protection (NYCDEP)
- Truckee Meadows Water Authority
- Metropolitan Water District of Southern California

APPROACH

To complete this project, the team first collected surface soil and litter samples from watersheds serving four water utilities (DW, WM, SFPUC, NYCDEP). The samples were collected from multiple sites in the different source watersheds for the utilities. The samples were transported to the University of Colorado Boulder (CU Boulder) for processing. At CU Boulder, the samples were air dried and heated at a temperature of 225ºC for two hours in a muffle furnace. This temperature was selected as the amount of dissolved organic carbon (DOC) released into solution was greatest compared to other temperatures (350ºC and 500ºC). Therefore, for this study, heating soils and litter materials at 225ºC represents worst-case scenario conditions for disinfection byproduct (DBP) precursors.

After the samples were heated, they were leached into water, followed by an evaluation of the water quality and treatability by coagulation. Unheated (control) soil/litter leachates were also characterized and evaluated with bench-scale treatment studies and compared to the heated leachates.

RESULTS/CONCLUSIONS

Soil and litter samples released different quantities and qualities of dissolved constituents following heating. Litter tended to release more dissolved organic matter (DOM) following heating compared to soil. The release of anions and cations was altered following heating. Anion release into solution showed strong heating dependence, but was not consistent among the measured species. Sulfate concentrations demonstrated the most consistent behavior, increasing with heating of each material, especially litter, which was shown to release nearly ten times more sulfate than soil following heating. Nitrate concentrations generally decreased following heating of both litter and soil. Phosphate release was not constant among the soils, but phosphate release from litter increased after heating at 225°C. Iron and manganese showed similar trends and demonstrated greater release after heating.

Generally, the trends observed for the four utilities were consistent and aid in understanding the effects of heating on water soluble compounds, raw water quality, and the associated treatment challenges. Marginal increases in pH and alkalinity were observed for the heated samples, which may be attributed to the denaturing of organic acids upon heating, with residual alkaline components remaining. An observed decrease in the quantity of DOC leached per gram of material for the heated leachates is consistent with other work indicating partial combustion of soluble organic carbon compounds at 225°C. Alternatively, organic nitrogen has been shown to volatilize at higher temperatures, supporting the observed enrichment of dissolved organic nitrogen (DON) relative to DOC following heating. SFPUC leachates did not follow the same trend as the other utilities, and the DOC leached per gram of soil increased after heating. Only soils were leached for the SFPUC samples (litter samples were not available), and perhaps different organic precursor materials of soils and litter may help explain the difference. Clear and measurable alterations to the soluble DOM character was indicated by increased specific UV absorbance at 254 nm ($SUVA_{254}$). Iron concentrations of the heated leachates were low (< 0.005 mg/L) and did not significantly interfere with absorbance measurements. Consistently higher $SUVA_{254}$ for the heated samples indicates enhanced aromaticity of soluble compounds upon heating, supported by previous soil organic matter studies.

For raw water (not coagulated) carbonaceous DBP (C-DBP) yields, the changes following heating at 225°C varied for total trihalomethane (TTHM) and haloacetic acid (HAA) precursors, but C- DBP yields were often lower following heating. However, this trend was not consistent for all samples. Bromide concentrations were low (< 0.003 mg/L), and primarily chlorinated DBP species were formed. It should be noted that the similar or lower TTHM and HAA precursor reactivity of the heated leachates compared to the control (unheated) samples may not be representative of the precursor load a water treatment facility might receive in its influent supply post-wildfire. Enhanced erosion of terrestrial DOM following a wildfire can significantly increase DOC levels and DBP formation, as observed in field-based studies. Haloacetonitrile (HAN) precursor reactivity of the raw waters also varied following heating. Alternatively, the chloropicrin precursor reactivity was generally higher for the heated leachates, which may be associated with the enrichment of DON relative to DOC, or elevated inorganic nitrogen levels. While the DOC:DON ratio decreased upon heating, and chloropicrin formation and precursor reactivity per unit of carbon increased, HAN4 precursors did not appear consistently altered by heating at 225°C. Previous studies have associated elevated HAN4 reactivity with wildfire.

Following heating of soil and litter, the leachates consistently exhibited an overall poor response to coagulation and, even at high coagulant doses (e.g., > 80 mg/L alum), often marginal DOC removal was achieved (e.g., <30%). The treatability findings are consistent with the results from a field-based post-fire watershed monitoring study when rainstorms transported substantial sediments and debris downstream to a water intake. The adverse effect of heating on the treatability of the leachates might be explained by a lower molecular weight DOM composition. Despite the higher $SUVA_{254}$, a change in DOM quality, such as a shift towards lower molecular weight compounds, may have adversely affected coagulation treatment, resulting in elevated settled water turbidity levels and minimal DOC removal. Further, finished water quality was negatively influenced, including the exceedance of DBP maximum contaminant levels (MCLs) and high chloropicrin concentrations. Heat-induced changes to particle size and characteristics were not explored in this study, but may have negatively affected coagulation processes, possibly due to the presence of finer materials. Following treatment, most heated leachates exceeded DBP MCLs, whereas all control samples were below MCLs. Treated water nitrogenous DBP formation was also higher for the heated leachates, specifically chloropicrin. While findings suggest an altered DOM character, utilities may also experience an increase in influent DOC concentrations coupled with higher, or even extreme, sediment loads, resulting in compounding effects on water treatment.

Lastly, recommendations were made regarding the design and operation of treatment systems for utilities under the threat of wildfire. The following recommendations are presented with the assumption that sufficient space is available.

1. Pre-sedimentation basin
    a. May be required/useful if raw water turbidity exceeds 100 NTU for long periods (i.e., days or weeks)
    b. Include the ability to bypass under normal conditions or potential addition of an oxidant
2. Coagulation
    a. Ensure chemical storage and feed pumps can deliver the higher chemical doses that may be needed after a wildfire
    b. Consider polymer feed facilities that may be needed to treat waters with ash content
    c. Develop operational protocols and install equipment such as streaming current monitors or zeta potential analyzers to help determine optimum coagulant dosages
3. Flocculation
    a. Install a means of removing silty solids that may settle out in flocculation tanks under high turbidity conditions
4. Sedimentation
    a. Use large conventional sedimentation basins if possible to handle large amounts of solids; if not practical, consider the use of lamella plate settlers
    b. Ensure solids can be easily removed from basins via mechanical sludge removal equipment
    c. If in an area where it is not likely that high turbidity will reach the intake, and there is concern that algal blooms could occur, consider dissolved air flotation (e.g., where ash or soil-related turbidity from a watershed after a fire is likely to settle out in an upstream reservoir, but nutrients could be transported to downstream reservoirs)
5. Filtration
    a. Consider the use of deep bed dual-media filters with larger media that can store more solids than conventional filters
    b. Consider granular activated carbon (GAC) in place of anthracite to help with taste and odor
    c. Provide enough backwash water and waste backwash storage so multiple filters can be backwashed at once
6. Membranes
    a. Membrane-based treatment systems should not be used if the raw water will be subject to the impact of firefighting foams that could foul membranes
7. Disinfection
    a. Higher levels of NOM may lead to DBP compliance issues
    b. Attention should be given to maximizing removal of NOM or relying on the use of alternative disinfectants including UV and ozone
8. Advanced treatment
    a. Smoky taste and odor could occur after a fire

b. Nutrient release from wildfires could result in long-term eutrophication and increased algal growth in downstream reservoirs leading to algal toxins and taste and odor issues
c. The installation of powdered activated carbon or post filter GAC contactors should be considered to handle these events
d. The installation of ozone/biofiltration should also be considered when possible

APPLICATIONS/RECOMMENDATIONS

It is recommended that utilities under the threat of wildfires consider the treatment implications of this perturbation in their watersheds. The results from this study indicate that higher coagulant doses will likely be required, with implications for operations and residual handling. If extreme post-fire erosion conditions occur, coagulation alone may not be effective for meeting turbidity and TOC removal requirements. Expanding water storage capacity and diversifying water sources is also recommended to handle worst-case scenario runoff conditions. In addition, a robust water quality monitoring plan is needed to ascertain the specific effects following wildfire and to rapidly and effectively adjust and respond to water quality changes. Utilities should have the capacity to conduct simple treatment evaluation tests in-house to address site-specific effects of post-fire runoff on treatment operations.

RELATED WRF RESEARCH

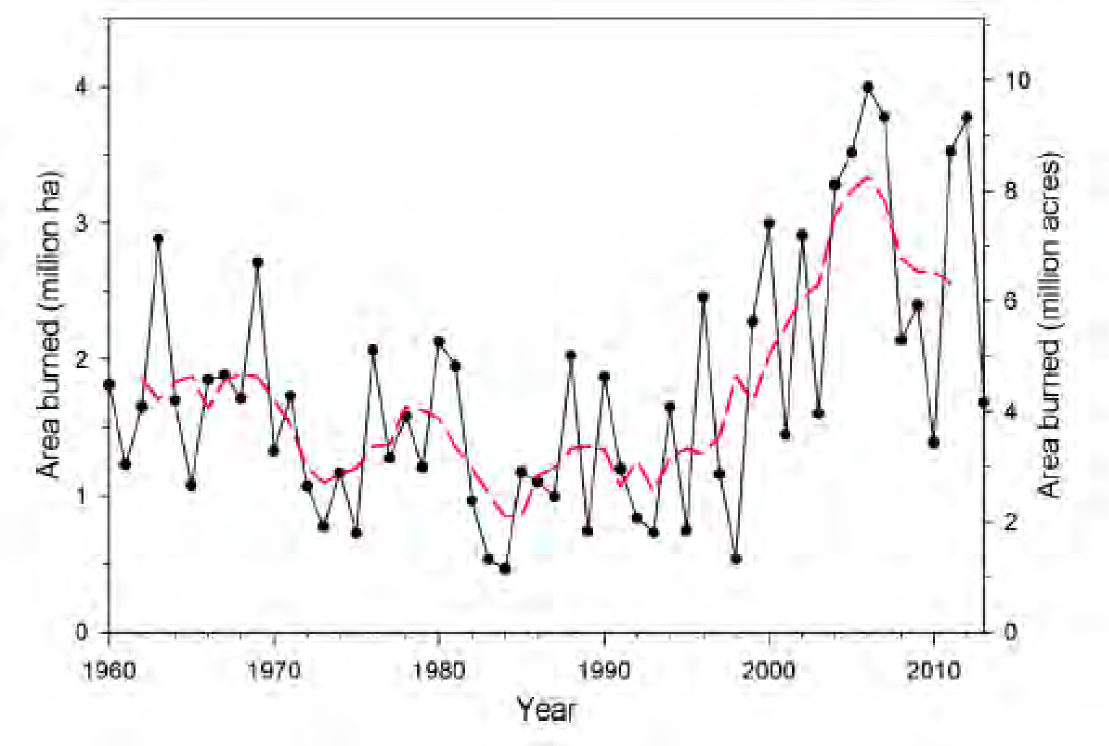
- An Integrated Modeling and Decision Framework to Evaluate Adaptation Strategies for Sustainable Drinking Water Utility Management Under Drought and Climate Change (project 4636)
- Effects of Wildfire on Drinking Water Utilities and Best Practices for Wildfire Risk Reduction and Mitigation (project 4482)
- Impact of Wildfires on Source Water Quality and Implications for Water Treatment and Finished Water Quality (project 4524)
- Utility Guidance for Mitigating Catastrophic Vegetation Change in Watersheds (project 4009)
- Wildfire Impacts on Water Supplies and Potential for Mitigation: Workshop Report (project 4529)

CHAPTER 1
INTRODUCTION

INTRODUCTION

Climate change and population growth continue to stress water resources, resulting in heightened pressures on drinking water supplies. Concurrently, natural and anthropogenic watershed disturbances are rising, potentially impacting the integrity of watersheds and the water quality of potable sources. These disturbances include wildfires, droughts, and floods. Addressing the effects of watershed disturbances on drinking water treatment presents a critical challenge for ensuring delivery of safe and sustainable drinking water.

The frequency and intensity of wildfires has increased in recent decades, and this trend is expected to continue in the near future especially in areas where climate change is predicted to result in warmer, drier conditions (Smith et al. 2011). Globally, many regions are experiencing a marked rise in wildfire activity as a consequence of climate change, increased fuel loads, and drought (Figure 1.1) (Flannigan et al. 2013, 2009; McKenzie et al. 2004; Westerling et al. 2006). In particular, in the Western United States, there has been an increase in the frequency, duration and amount of burned area from wildfires (Brown et al. 2004, Westerling et al. 2006, Flannigan et al. 2009). This increase can be partially attributed to consequences of climate change (Brown et al. 2004, Westerling et al. 2006). Extreme droughts, higher temperatures, earlier snowmelt, and changes in precipitation patterns can all contribute to the likelihood of wildfires. Other factors influencing wildfire occurrence include land-use changes, such as livestock grazing, and fire suppression (Savage and Swetnam 1990, Belsky and Blumenthal 1997). Consequently, wildfires are of increasing concern and the resulting impacts on the environment must be further investigated.



Source: Reprinted (adapted) with permission from Bladon et al. 2014. Copyright 2014 American Chemical Society.
The red dashed line indicates the rolling 5-year average area burned.
Figure 1.1 Annual area burned in the United States from 1960-2013

The increased likelihood of wildfire has created concerns among downstream water utilities regarding the integrity and resiliency of their water systems, including source water watersheds (Sham et al. 2013). While the ecological, forestry, and watershed science communities have investigated wildfires extensively, post-wildfire drinking water effects are relatively unexplored (Emelko et al. 2011, Hohner et al. 2016, Revchuk and Suffet 2014, Writer et al. 2014). A thorough investigation of post-fire treatment challenges is needed to help utilities make informed management decisions and develop mitigation approaches. The vulnerability of watersheds, infrastructure, and process performance must be assessed to provide water utilities with recommendations to effectively prepare for a wildfire.

In recent years, several major wildfires have impacted water treatment systems (WTS) in the Western United States (Sham et al. 2013). In 2002, the Hayman Fire of Colorado burned the forested area of the Upper South Platte River. This watershed serves as the drinking water supply for the City of Denver. The 2012 High Park Fire burned sections of the Cache la Poudre (CLP) watershed, which serves as a drinking water source for three northern Colorado communities, including the City of Fort Collins. Also in 2012, the Waldo Canyon Fire burned Pike National Forest threatening the City of Colorado Springs' drinking water supply. The Rim Fire of 2013 in California burned within four miles of the Hetch Hetchy Reservoir, which serves the City of San Francisco. Other regions of the contiguous United States that have received extensive wildfire damage in the last decade include: Arizona, Montana, Wyoming, and New Mexico. Internationally, wildfires are also a concern in different regions of the world, including Canada, Spain, and Australia.

Wildfire Effects on Soils

Water quantity and quality are inextricably linked to forest soil properties (Neary et al. 2005), and wildfire changes to soils are critical to understanding watershed effects of fire. Research has shown the development of a hydrophobic layer following wildfire (DeBano 2000a, 2000b) that can increase surface-runoff upon rainfall (Neary et al. 2009). Nutrient cycling in soils may also be affected by wildfires. Carbon and nitrogen losses in the organic layer vary depending on fire severity, fuel type, pre-fire conditions (Homann et al. 2011), and time since the disturbance (Prieto-Fernandez 1998). However, several studies have shown an increase in forest floor carbon and nitrogen content following a low or moderate severity fire as a result of partly charred biomass accumulation and integration into the soil (Almendros et al. 1990, 1988; Knicker et al. 2005).

The effects of wildfires on soil organic matter (SOM) have been more extensively studied than DOM and were reviewed by Knicker (2007). Wildfire induced changes to SOM quantity and quality will likely affect the DOM of nearby waters based on soil biogeochemical processes that govern DOM fluxes in watersheds and the hydrologic connectivity through the landscape (Cawley et al. 2018, Cawley et al. 2017, Dick et al. 2014, Sanderman et al. 2009). Specifically, research suggests the importance of rainstorms in the transport of terrestrial debris and SOM to surface waters in unburned (Clark et al. 2007, Inamdar et al. 2012, Inamdar and Mitchell 2007) and burned watersheds (Murphy et al. 2015, Mast et al. 2016).

Wildfire Impacts to Water Quality

There are numerous concerns regarding the impact of a wildfire on source water quality and water treatment systems (WTS) operations. These concerns include water quantity and availability, as well as the impact on source water quality and the ability of WTS to effectively treat and provide high quality water that the public demands (Sham et al. 2013). Currently, there are a limited number of reported studies that have monitored post-fire water quality and treatability at drinking water utilities.

Many confounding and seasonally varying watershed factors, as well as wildfire specific factors, such as burn severity, forest type, location and extent, influence post-fire water quantity and quality responses (Hohner et al. 2016, Neary et al. 2005, Rhoades et al. 2011, Smith et al. 2011). Combined these factors make it challenging to elucidate impacts specifically related to wildfire (Neary et al. 2005). Wide ranges of post-fire responses have been observed and are highly dependent on the timing and magnitude of precipitation in the burned area. Current research shows the primary influences of wildfire on surface water quality are enhanced erosion, increased sediment loads, elevated nutrient concentrations (i.e., nitrogen and phosphorus), and generally more variable water quality (Smith et al. 2011). The potential changes in water quality pose significant challenges to water utilities, however previous work mostly speculates regarding post-fire treatment challenges, lacking a direct evaluation.

The most apparent influence of wildfires on watersheds is an altered hydrologic regime characterized by flashy events and subsequent sediment transport. Effects can include increased total runoff volume, increased peak flow, flooding, and increased sediment mobilization (Sham et al. 2013). A wildfire can result in loss of vegetation, decreased soil infiltration (higher soil hydrophobicity), and decreased evapotranspiration, all contributing to the effects listed above. Consequently, erosion and sediment mobilization can be elevated post-fire, particularly during storm events and when discharge is greatest in the first year following a wildfire (Silins et al. 2009).

From a water supply and treatment perspective, these changes can greatly influence total suspended solids (TSS) and turbidity levels. Rhoades et al. (2011) found turbidity levels were four times higher in a basin with high severity burn compared to basins that were not burned as severely. Other research has shown elevated turbidity levels in wildfire impacted waters can reach extreme values under specific hydrological conditions (e.g., heavy rainfall, rapid snowmelt) (Emelko et al. 2011, Writer et al. 2012). The elevated level of TSS is a concern to utilities, which are restricted regarding the turbidity levels that are permitted for potable water. It has been shown that the enhanced turbidity associated with a rain event following a wildfire can be effectively coagulated, however the required coagulant doses are, as expected, higher (Hohner et al. 2016). The higher coagulant doses may be beyond what a WTS could handle and will also result in increased costs for disposal of material.

Regarding the specific impacts on water quality, the observed changes depend greatly on different parameters including fire intensity, severity, and duration, as well as forest terrain slope, the frequency and intensity of rain storm events, and snowmelt patterns (Landsberg and Tiedeman 2000, Neary et al. 2005). The effects are all driven by the extent of perturbation by the wildfire (e.g., localized low intensity fire versus mixed severity, large size, and location near a water intake). For instance, a high severity fire burns the overstory and understory vegetation and consumes most surface organic matter (Keeley 2009).

Perhaps one of the most studied parameters regarding the effect of wildfire on water quality is the mobilization of nutrients (nitrogen and phosphorous). In general, previous work has shown that nutrient concentrations increased in watersheds after a wildfire. Ranalli (2004) showed that nitrogen levels increased and then eventually stabilized once vegetation regrowth was established. Studies in the Rocky Mountains of Alberta also reported increased nitrogen and phosphorus levels (Bladon et al. 2008, Silins et al. 2009). Other studies found nitrate levels were increased in burned watersheds (Mast and Clow 2008, Writer et al. 2012). Research by Rhoades et al. (2011) found that high severity burns increase nitrate and turbidity levels, whereas a lower intensity fire had less of an effect on water quality. Nutrient levels in streams can also be affected by inputs from atmospheric deposition as well as hillslope soil erosion and transport from burned areas (Smith et al. 2011). Wildfires are also known to enhance the mobilization of metals, including iron, manganese, and mercury (Sham et al. 2013), although the specific mechanisms by which this happens are subject of ongoing research.

Dissolved Organic Matter and Disinfection Byproducts

One large research gap regarding the short and long-term effects of a wildfire on water quality is in the effect on the mobilization and changes to the chemical properties of dissolved organic matter (DOM) after a wildfire. DOM is generally quantified as dissolved organic carbon (DOC). DOM is ubiquitous in the aquatic environment, composed of a complex, heterogeneous mixture of aromatic and aliphatic structures that contain many functional groups (Leenheer and Croué 2003). Terrestrial sources of DOM include plant debris and soil organics flushed into surface waters by rainfall, commonly referred to as allochthonous organic matter. Terrestrially derived DOM is generally more aromatic with higher humic content, and well removed by conventional treatment processes (Edzwald 1993). Autochthonous, or aquatically derived, DOM originates from algal and microbial productivity within a water body. Autochthonous DOM is commonly characterized by higher nitrogen content and a more aliphatic and hydrophilic composition (Leenheer and Croué 2003, Westerhoff and Mash 2002).

There have been numerous studies regarding the mobilization of DOM in watersheds. The background DOM observed in a watershed is from leaching of soils in surface and groundwater (this is for watersheds that do not contain significant proportions of wetlands, peats, or bogs). During baseflow conditions, the DOM is mostly from groundwater and the DOC levels are lower because mineral soils contain less SOM (Thurman 1985). During higher flow (surface) events, including snowmelt and storms, surface SOM is mobilized which typically has higher aromaticity (quantified by WTS as $SUVA_{254}$) (Weishaar et al. 2003).

Wildfire changes to the landscape and forest floor may alter the quantity and quality of DOM in source waters. There are several studies regarding the fate of SOM after a wildfire (Knicker 2007, Lopez-Martin et al. 2016), however there are limited studies on the mobilization to surface waters (Majidzadeh et al. 2015, Tsai et al. 2015, Wang et al. 2015a, Wang et al. 2015b). It has been reported by different groups that, at temperatures associated with low intensity fires, DOM mobilization can increase (Cawley et al. 2017). Alternatively, other studies have concluded minimal effects on DOC levels in burned areas compared to unburned (Lamontagne et al. 2000, Mast and Clow 2008). These seemly contradictory findings highlight the need for further research to address the complexity of wildfire characteristics, environmental influences, and hydrologic effects following wildfires on water quality.

Elevated DOM levels, commonly quantified as dissolved organic carbon (DOC - mg_C/L), following wildfires have been documented (Emelko et al. 2011, Hohner et al. 2016, McEachern et al. 2000, Minshall et al. 2001), while others found minimal effects on DOC (Mast and Clow 2008). It is suggested that background DOC concentrations may dampen the contributions from wildfire (Smith et al. 2011), emphasizing the importance of local hydrology, burn severity, and extent, when interpreting effects on water quality.

Although DOM is naturally occurring, it poses a concern in source water supplies because it can react with disinfectants (e.g., chlorine, chloramines) to form disinfection byproducts (DBPs) as an unintended consequence of water treatment (Christman et al. 1983, Rook 1977, Stevens et al. 1976). DBP exposure has been linked to cancer risks and reproductive developmental effects (Muellner et al. 2007; Plewa et al. 2004; Richardson et al. 2007; Villanueva et al. 2004, 2006). The United States Environmental Protection Agency (EPA) regulates certain DBP species in drinking water, including total trihalomethanes (TTHM) and the sum of five haloacetic acids (HAA5) (EPA 2011). The regulated TTHM and HAA5 maximum contaminant levels (MCLs) under the Stage 2 Disinfectant/Disinfectant Byproduct Rule are 80 μg/L and 60 μg/L, respectively.

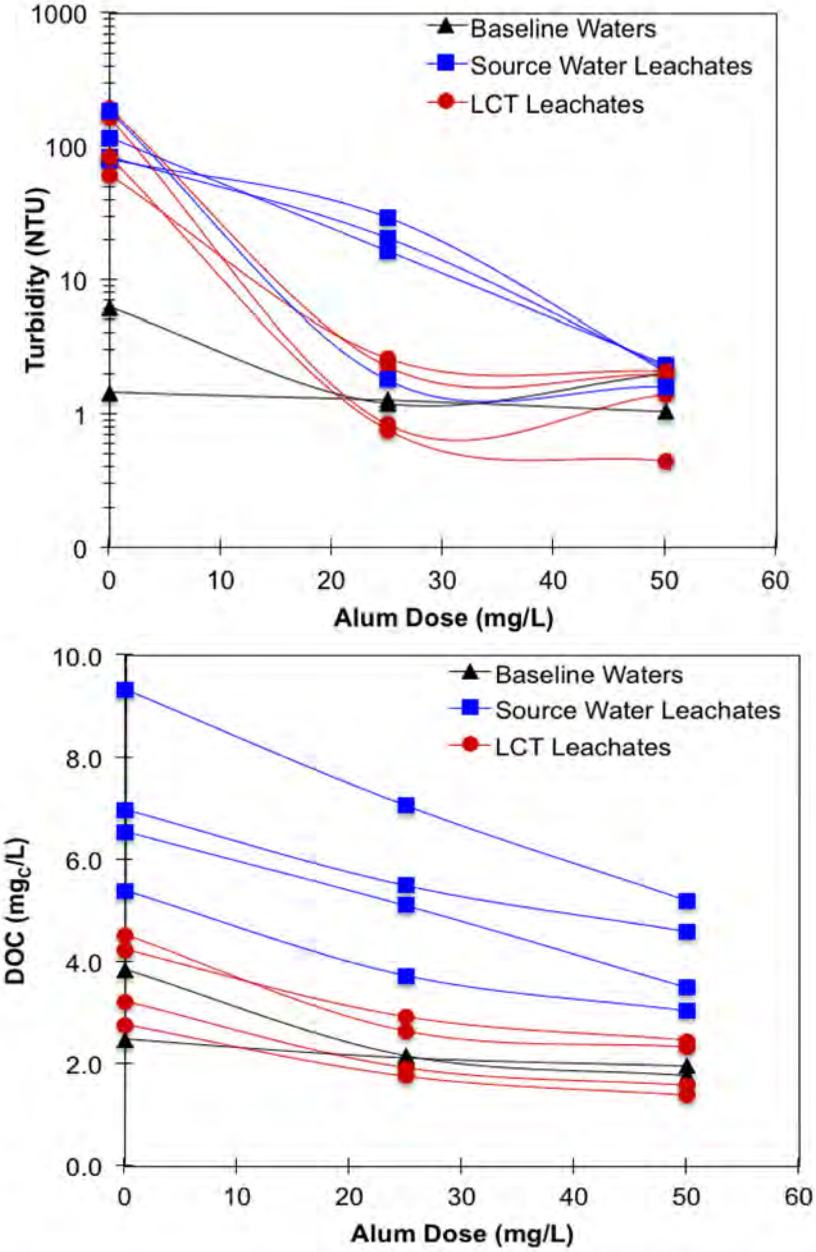
Nitrogenous disinfection byproducts (N-DBPs), such as haloacetonitriles (HAN), halonitromethanes (e.g., chloropicrin), and nitrosamines are not currently regulated, but are of interest as they may be more of a health concern than TTHM and HAA5, even at lower concentrations (Muellner et al. 2007; Plewa et al. 2004, 2008). Minimal information is available regarding the effects of wildfire on DBP formation (Wang et al. 2015b, Writer et al. 2014) and water treatment processes (Emelko et al. 2011). Current research suggests material burned in wildfires may constitute a terrestrial source of N-DBP precursors (Cawley et al. 2017, Wang et al. 2015b). Understanding the sources, characteristics, and reactivity of DOM is essential to control DBP formation in potable water supplies.

Water Treatment Unit Processes

There are limited reports on the effects of wildfires on source water quality coupled with the water treatment process performance (Emelko et al. 2011, Hohner et al. 2016, Hohner et al. 2017, Majidzadeh et al. 2015, Tsai et al. 2015, Wang et al. 2015a, Wang et al. 2015b, Writer et al. 2014). Several lab-based leaching studies of material burned in wildfires have been conducted to assess the effects on DOM and drinking water treatment (Revchuk and Suffet 2014, Wang et al. 2015a). In a leaching study of material burned in California wildfires, a 10-fold increase in the concentration of DOC leached was observed when compared to unburned material (Revchuk and Suffet 2014). A study evaluating water extractable material of burned detritus from the Rim Fire revealed elevated N-DBP reactivity (HAN and NDMA) compared to unburned detritus (Wang et al. 2015b). Other work has shown that waters impacted by wildfire could be effectively treated by coagulation (Emelko et al. 2011).

Our group has conducted detailed treatment studies on wildfire-impacted source waters. Initially, we coagulated water samples collected from the CLP River after the 2012 High Park wildfire in Colorado. The results indicated that post-fire samples required on average a 10 mg/L higher coagulant dose during baseflow and spring snowmelt conditions when compared to water collected from an unburned reference site (Hohner et al. 2016). In a follow up study, post-fire source water quality was simulated using field sediments impacted by the High Park wildfire (Hohner et al. 2017). Sediments were collected from four sites (A-D) along the banks of the CLP River after the wildfire. The sediments were then leached in the laboratory for 24 hours with baseline source water (SW) collected from two utilities. Sediments A-D were also leached in City of Boulder tap water treated with granular activated carbon (GAC) to minimize background DOC (< 0.2 mg_C/L), referred to as low carbon tap-water (LCT). In this previous study, multiple unit processes were evaluated: coagulation, enhanced coagulation, powdered activated carbon (PAC), biofiltration and pre-oxidation with chlorine dioxide or ozone. The different treatment processes were evaluated for DBP precursor removal.

Figure 1.2 shows the coagulation dose response curves for baseline waters, SW leachates and LCT leachates. The baseline samples represent the source waters for the two participating utilities with no sediment added. The baseline waters were effectively treated with 25 mg/L of aluminum sulfate. For the SW leachates, which includes the baseline waters with the addition of sediments, the waters were more difficult to treat, demonstrated by higher finished water turbidity and DOC levels. When the sediments were leached in LCT water, the dose responses were similar to the baseline waters.



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An alum dose of 25 mg/L was used for conventional treatment and 50 mg/L for enhanced coagulation.

Figure 1.2 Raw and coagulated turbidity and DOC values of baseline waters, source water leachates, and low carbon water (LCT) leachates

Table 1.1 presents results from the treatability evaluation of multiple unit processes for baseline waters, SW leachates, and LCT leachates. TTHM and HAA5 MCL values (80 and 60 µg/L, respectively) were divided by the corresponding carbon normalized DBP yield (µg/mg$_C$) determined from the bench scale chlorination tests to calculate *DOC thresholds* (mg$_C$/L). DOC thresholds represent the required DOC concentration at the point of chlorination to meet TTHM and HAA5 regulations for each water sample and treatment process. In each case, the more restrictive DBP yield (TTHM or HAA5) was used. The DBP yield and *DOC* threshold approach allows for a comparison of all samples, independent of raw water DOC concentrations. As shown in Table 1.1, pre-ozonation was most often the best treatment option (highest DOC threshold) for the sediment leachates to minimize DBP formation and meet DBP MCLs. Refer to Hohner et al. (2017) for further detail on the previous study.

Table 1.1
DOC values required at the point of chlorination to meet DBP MCLs (DOC thresholds) determined from treated water TTHM and HAA5 yields and DBP MCLs for the range of unit processes evaluated

	Sample Name	MCL DOC Threshold (mgC/L)							Best Treatment Option
		Conventional Treatment	**Enhanced Coagulation**	**PAC**	**Chlorine Dioxide**	**Pre-ozonation**	**Biofiltration**	**Pre-ozonation/ Biofiltration**	
Baseline Waters	**Fort Collins (FC)**	2.6	2.8	2.3	2.6	2.7	2.6	3.0	Pre-ozonation/ Biofiltration
	Denver Water (DW)	3.1	3.3	2.8	4.8	3.0	2.7	3.3	Chlorine Dioxide
Average increase in DOC threshold			0.2	-0.3	0.8	0.0	-0.2	0.3	
Source Water Leachates	**A- FC**	2.0	2.0	1.8	1.8	2.4	1.4	2.2	Pre-ozonation
	B- DW	1.7	2.1	1.8	1.8	3.0	1.6	2.6	Pre-ozonation
	C- DW	2.1	2.8	2.1	2.1	2.8	2.4	2.1	Enhanced Coag & Pre-ozonation
	D- FC	1.8	2.4	1.3	2.0	2.4	1.8	2.3	Enhanced Coag & Pre-ozonation
LCT Leachates	**A- LCT**	2.0	2.3	1.8	2.1	2.6	1.6	2.4	Pre-ozonation
	B- LCT	1.6	2.1	2.0	2.0	1.7	1.7	2.1	Enhanced Coagulation Pre-ozonation/ Biofiltration
	C- LCT	1.4	1.9	2.1	1.7	3.0	1.5	2.1	Pre-ozonation
	D- LCT	2.1	2.0	1.8	2.2	2.7	1.6	2.5	Pre-ozonation
Average increase in DOC threshold			0.4	0.0	0.1	0.7	-0.1	0.5	Pre-ozonation

Source: Hohner et al. 2017. Reproduced by permission of The Royal Society of Chemistry.

Project Motivation

The work discussed so far offers insight into the potential effects of a wildfire on water quality and treatment processes. However, published literature is primarily based on field samples collected after a select number of wildfires, and it is unknown whether these results apply to other utilities with different treatment processes, or located in different geographical regions. In this regard, there is a need to develop a framework for utilities to evaluate the potential local effects of a wildfire on water quality and treatment performance. Essentially, there is a need to develop an approach to simulate the effects of a wildfire, which presents many challenges and lacks an ideal solution. As part of a previous Water Research Foundation project (WRF 4524), our team conducted preliminary experiments to evaluate lab-based heating experiments to simulate post-fire effects on water quality (Cawley et al. 2017). For this project (WRF 4590), the team continued the development of the laboratory heating approach to simulate post-fire effects on water quality and treatment for four utilities. Soil and litter samples from different watersheds were collected and processed in the lab. In consultation with the participating utilities, samples were heated to 225°C and then leached in LCT water at CU Boulder. The leached samples were evaluated for water quality parameters. In addition, in consultation with the participating utilities, bench-scale treatment tests were developed and conducted. The following sections contain a discussion of the project results.

The following utilities participated in this project:

- Denver Water (DW)
- City of Westminster (WM)
- City of Northglenn, CO
- City of Thornton, CO
- San Francisco Public Utilities Commission (SFPUC)
- New York City Department of Environmental Protection (NYCDEP)
- Truckee Meadows Water Authority
- Metropolitan Water District of Southern California

Further, this project also funded an extension of previous work addressing the changes in CLP River water quality after the High Park fire of 2012. The intent was to expand the monitoring work done as part of previously funded work by both The Water Research Foundation and the Colorado Department of Public Health and Environment. This work is discussed in Chapter 2.

CHAPTER 2
CONTINUED MONITORING OF THE CACHE LA POUDRE WATERSHED FOLLOWING THE HIGH PARK WILDFIRE

In 2012, the High Park wildfire burned approximately 87,284 acres of the Cache la Poudre (CLP) watershed located in northern Colorado. The CLP watershed supplies potable water to three water districts in Colorado, including the City of Fort Collins, serving over 300,000 customers. The immediate and future impacts of the High Park wildfire on source water quality of the CLP River presented a high concern for the local water utilities.

Two previous WRF projects (#4282 and #4524) were conducted by the research team to monitor the water quality and treatability of the CLP River prior to the wildfire (2008 - 2009) (Summers et al. 2013) and for the first year following the wildfire (2012 – 2013) (Writer et al. 2014). As part of this project, to further address post-fire water quality effects, the CLP River monitoring campaign was extended. The CLP watershed was sampled April-October 2015, bi-weekly during spring snowmelt and monthly during baseflow conditions. Samples were collected from the City of Fort Collins water intake located within the burned area, and from an upstream reference site (Figure 2.1). Water quality analysis, chlorination tests and DBP analysis of the monitoring samples were completed following established methods (Hohner et al. 2016). Pre-fire data collected from the CLP River at the water intake (downstream of burned area) and a reference site (upstream of burned area) were used for comparison (Beggs 2010, Summers et al. 2013).

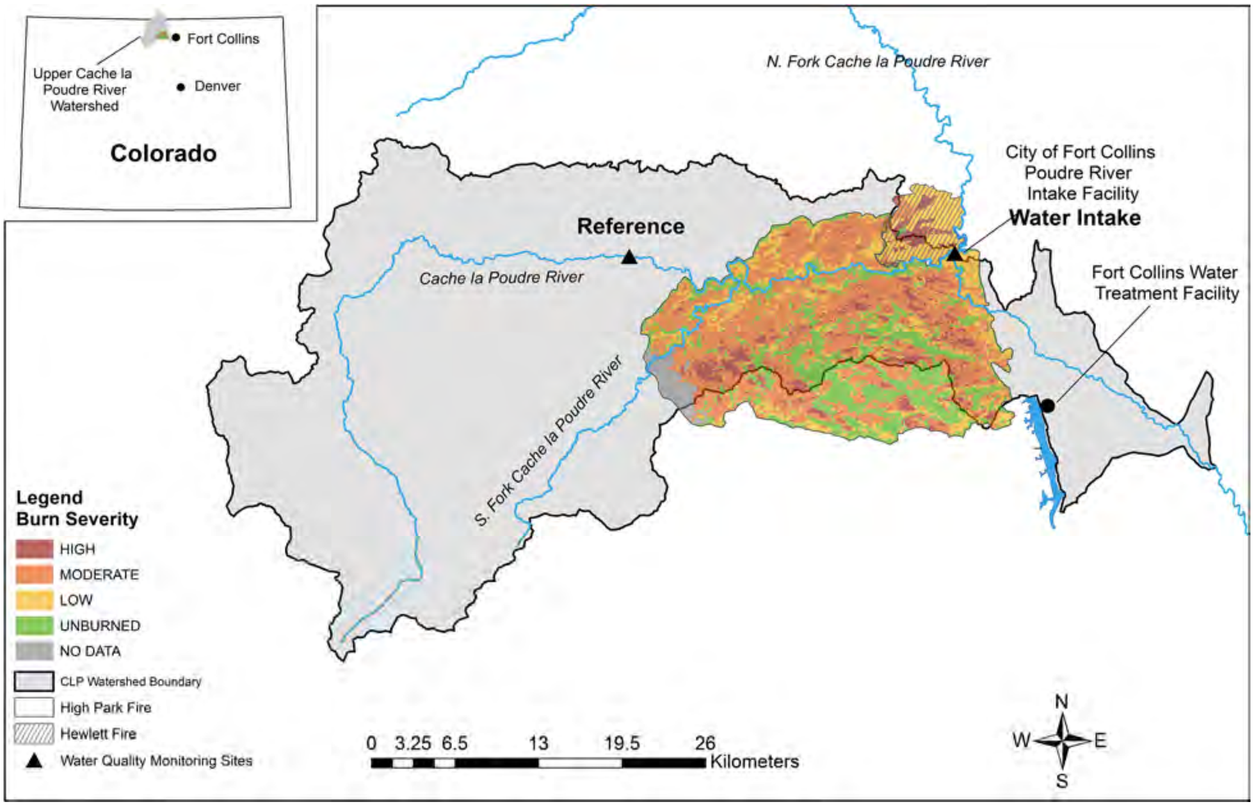

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Figure 2.1 CLP River sampling sites include the City of Fort Collins drinking water intake within the High Park fire burn area, and the reference site, upstream and outside of the burned area

Table 2.1 shows the water quality concentrations for the water intake and reference site including pre-fire years (2008 – 2011), and two post-fire years, 2013 and 2015. Samples collected in 2012 were previously published in Writer et al. 2014 as part of preliminary effort to capture immediate post-fire runoff. The data and text related to the pre-fire and 2013 data sets were taken from Hohner et al. 2016 (Hohner et al. 2016). Unfortunately, samples were not collected during 2014 due to budget constraints and a delayed project start date. A paired data analysis approach was used. For each sampling date the difference (Δ) in water quality concentrations between the water intake and reference site was determined to assess whether any differences occurred following the wildfire. Further, the post-fire spatial differences between the sites were compared to the pre-fire differences.

Pre-fire paired spatial differences are presented in Table 2.1 (pre-fire p values not shown) and reveal the water quality concentrations of the two sampling sites were similar before the wildfire. Total nitrogen (TN) and total organic carbon (TOC) levels were not statistically different ($p > 0.05$) for the two sampling sites before the fire (median Δ: $< 0.01\ \mathrm{mg_N/L}$ and $- 0.12\ \mathrm{mg_C/L}$, respectively). Spatial differences for total phosphorus (TP) were statistically significant ($p < 0.05$). The mean and median TP concentrations for both sites were very close to the reporting limit ($0.01\ \mathrm{mg_P/L}$) and the median difference was $< 0.001\ \mathrm{mg_P/L}$, therefore TP was assumed reasonably similar between the sites prior to the wildfire. The upstream site was chosen as a reasonable reference location for comparison to the water intake due to the minimal differences in pre-fire water quality, which reflect the influence of basin differences (e.g., drainages, elevation) that could impact water quality.

The 2013 post-fire spatial differences for TP and TN were at least an order of magnitude greater than before the fire ($p < 0.01$). In 2015, total dissolved phosphorus (TDP) and total dissolved nitrogen (TDN) were higher at the water intake compared to the reference site. Note that samples collected in 2015 were analyzed as the dissolved fraction, rather than total nutrients (particulate plus dissolved). The 2013 post-fire TOC spatial differences were significantly greater than before the fire (Table 2.1) and showed more variability, likely indicating an increased watershed input of organic matter to the river after the wildfire. In 2015 the spatial differences in nutrients between sites were elevated compared to pre-fire years, but were lower than observed in 2013, suggesting the post-fire effects had mostly dissipated, however the 2015 samples only included the dissolved fraction, which may have influenced the comparison to 2013 nutrient levels. The elevated post-fire TN, TP, and TOC in 2013 likely reflect increased erosion and sediment transport to the stream channel. Whereas, by 2015 most of the fire-impacted materials had been washed downstream by rainfall and snowmelt.

Table 2.1
Pre- and post-fire water quality for the CLP River are shown. The water intake is located downstream of the burned area and the reference site is located upstream and outside of the burned area. Samples collected in 2015 were analyzed as the dissolved fraction for TDP, TDN, and DOC.

Water Quality Parameter		**Pre-fire (2008 – 2011)**				**Post-fire 2013**				**Post-fire 2015**			
		Concentration		**Δ**	**n**	**Concentration**		**Δ**	**n**	**Concentration**		**Δ**	**n**
		Water Intake	**Reference**	**(intake - reference)**		**Water Intake**	**Reference**	**(intake - reference)**		**Water Intake**	**Reference**	**(intake - reference)**	
TP/TDP (mg_P/L)	Mean	0.018	0.014	+0.004	44	0.058	0.013	+0.046	11	0.010	0.006	+0.004	9
	Median	0.013	0.012	+0.000		0.023	0.011	+0.013		0.007	0.004	+0.002	
	Stdev	0.014	0.007	0.008		0.077	0.004	0.077		0.008	0.007	0.005	
TN/TDN (mg_N/L)	Mean	0.29	0.27	+0.02	20	0.66	0.27	+0.4	11	0.28	0.19	+0.09	9
	Median	0.25	0.24	+0.00		0.34	0.28	+0.16		0.21	0.16	+0.02	
	Stdev	0.2	0.18	0.11		0.69	0.1	0.7		0.19	0.08	0.19	
TOC/DOC (mg_C/L)	Mean	4.8	4.8	+0.01	43	4.9	4.2	+0.71	10	5.3	5.1	+0.19	9
	Median	3.5	3.9	-0.12		4.6	4.2	+0.47		4.6	5.1	+0.22	
	Stdev	2.8	2.7	0.8		1.9	1.9	0.91		2.4	2.6	0.62	

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In 2013 and 2015, $SUVA_{254}$ values at the water intake were higher than pre-fire values (Table 2.2). However, post-fire pairwise comparison of the water intake to the reference site indicated $SUVA_{254}$ was statistically the same, suggesting during baseflow and snowmelt conditions the aromaticity of the DOM was not significantly altered by the wildfire. Pre-fire $SUVA_{254}$ data were not available for the reference site, and therefore a pre- to post-fire spatial analysis was not possible.

Carbon-normalized DBP formation yields (DBP concentration/DOC concentration) were used to understand the reactivity of DOM on a per unit carbon basis ($\mu g/mg_C$). TTHM yields at the water intake were similar pre- and post-fire (2013 and 2015), whereas HAA5 yields were considerably higher in both 2013 and 2015, compared to 2009 (Table 2.2). However, the 2013 TTHM and HAA5 yields were similar between the water intake and reference sites, suggesting wildfire impacts on C-DBP reactivity were minimal for this watershed and wildfire. Further, in 2015 HAA5 yields measured at the reference site were higher than the intake. The higher HAA5 reactivity observed in 2013 and 2015 compared to 2009 may be attributed to temporal variability resulting in a shift in the watershed HAA5 precursor material independent of the wildfire, such as increased primary productivity in the late summer, and would likely also be observed for the reference site. Pre-fire data were only available for the water intake, so a pairwise statistical analysis was not possible before the fire, limiting further conclusions. In 2013 paired analysis showed the HAN4 yields were higher at the water intake (median $\Delta = 0.12\ \mu g/mg_C$), while in 2015 the spatial difference was minimal (median $\Delta = -0.04\ \mu g/mg_C$). Chloropicrin yields at the two sites were similar in both 2013 and 2015. Pre-fire N-DBP data were not available.

Table 2.2
Pre- and post-fire DOM properties for the CLP River are shown. The water intake is located downstream of the burned area and the reference site is located upstream and outside of the burned area.

DOM Parameter		**Pre-fire (2008 - 2009)**		**Post-fire (2013)**				**Post-fire (2015)**			
		Water Intake		**Water Intake**	**Reference**	**n**	**Δ (intake - reference)**	**Water Intake**	**Reference**	**n**	**Δ (intake - reference)**
		Value	**n**	**Value**				**Value**			
SUVA$_{254}$ (L/mg$_C$-m)	Mean	2.7	22	3.3	3.3	12	+0.01	3.2	3.3	9	-0.08
	Median	2.9		3.4	3.4		-0.02	3.3	3.3		-0.08
	Stdev	0.5		0.7	0.8		0.21	0.4	0.4		0.12
TTHM Yield (µg/mg$_C$)	Mean	46.5	6	52.4	56.6	12	-4.3	57.7	51.6	9	+1.1
	Median	47.1		50.1	56.4		-5.7	55.3	57.4		-0.9
	Stdev	17.3		15.5	15.8		5	12.5	14.4		11.1
HAA5 Yield (µg/mg$_C$)	Mean	26.7	6	50	53.7	12	-3.7	58.6	69.6	9	-16.4
	Median	20.3		52.2	54.2		-1.1	51.3	58.5		-7.5
	Stdev	14.3		17.2	18.4		8.0	15.0	33.7		22.2
HAN4 Yield (µg/mg$_C$)	Mean	NA		1.7	1.6	12	+0.12	1.33	1.36	9	+0.06
	Median			1.7	1.5		+0.09	1.37	1.32		-0.04
	Stdev			0.32	0.2		0.21	0.28	0.29		0.20
Chloropicrin Yield (µg/mg$_C$)	Mean	NA		1.1	1.2	12	-0.09	1.04	1.02	9	+0.01
	Median			1.1	1.1		-0.09	1.00	1.00		-0.01
	Stdev			0.32	0.42		0.23	0.16	0.22		0.19

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Following the High Park wildfire, the City of Fort Collins was forced to shut down the water intake along the CLP River due to high sediment loads that posed a threat to water conveyance infrastructure. Fort Collins relied solely on an alternate water supply for 100 consecutive days. Previous work conducted immediately following the wildfire showed evident post-fire effects on CLP water quality (turbidity, TP, TN, TOC) (Writer et al. 2014). Post-fire changes in source water quality were most pronounced following rainstorms in the fire-affected area of the watershed. Post-rainstorm samples not presented in Tables 2.1 and 2.2 due to the lack of samples from the reference site. Please refer to Hohner et al. 2016 for post-rainstorm results. Smaller changes in water quality were observed during baseflow and spring snowmelt conditions in 2013, and to a certain extent in 2015. However, by 2015 post-fire effects likely had been washed away. It should be noted that monitoring samples were not collected following rainstorms in 2015, which would likely have captured any erosion of remaining sediments and debris burned in the wildfire. An in-depth study of the post-fire treatability in the CLP watershed is available in previous reports and publications (Hohner et al. 2016, Hohner et al. 2017, Writer et al. 2014).

CHAPTER 3
SAMPLING SITES FOR TREATMENT STUDIES

SOIL SAMPLING AND WATERSHED DESCRIPTIONS

As discussed in Chapter 1, to complete the evaluation of the potential impacts of wildfire on water quality and treatment (Chapters 4 and 5), surface soil and litter samples were collected from the participating utilities. Soil (mostly O-horizon) and litter samples were collected in the watersheds of four municipal water providers including New York City's (NYC) Kensico, Ashokan, Neversink, and Rondout watersheds; San Francisco's (CA) Hetch-Hetchy Reservoir, East Bay Watershed and San Antonio Reservoir; Denver's (CO) Gross Reservoir and Westminster's (CO) Clear Creek watershed. For each watershed or reservoir, a series of sampling locations were selected in consultation with the utilities. Specific locations were sampled using a 0.5 by 0.5 m plot to generate sufficient material for a composite sample for each site. Soil and litter samples from the plot were collected using a clean trowel or large shovel. Large rocks and non-soil materials were removed from the samples in the field. Litter was defined as material that still retains the shape of its precursor material, e.g., pine needles, partially decayed leaves, and sticks. Surface soil was defined as the top 5-cm below the litter. The samples were transported back to the laboratory in gallon sized plastic freezer bags or large trash bags (unscented plain plastic, double bagged). For samples that required shipping, the collected materials were dried first at room temperature for two days prior to transport to the laboratory. Additional methods are described as appropriate in the following chapters.

Denver Water (DW) provides drinking water to 1.4 million consumers in the metropolitan area, relying heavily on high quality water from forested watersheds. The Buffalo Creek fire (1996) and Hayman fire (2002) burned areas which provide water for the region, significantly impacting the sediment load to Denver Water supply reservoirs. Gross Reservoir is part of Denver's water system, located at an elevation of 2202 m in Roosevelt National Forest, northwest of Denver, Colorado. Three soil/litter sites surrounding Gross Reservoir were included in the study. The drainage area is primarily composed of granite bedrock with deposition of some metals, which generated historical mining activity (Murphy 2006). The watershed is a mountainous forested area, consisting of ponderosa and lodgepole pines and mixed confiners, and has been significantly affected by the mountain pine beetle infestation since the late 1990s. South Boulder Creek serves as a channel for transferring water from the western slope of the continental divide to Gross Reservoir, and the City of Denver water treatment facility.

The City of Westminster (WM) relies on the Clear Creek watershed for its water supply. Clear Creek eventually drains to Stanley Lake which provides drinking water to over 250,000 consumers. The Clear Creek headwaters begin near the continental divide, west of Denver, and ultimately meet the South Platte River near Golden, CO. Three soil/litter sampling sites along the upper basin of Clear Creek were sampled for this study. The sites were selected from a wildfire hazard ranking assessment of areas in the watershed (JW Associates 2013). The Clear Creek watershed is steep and mountainous, also located in Colorado's Roosevelt National forest. The geology is made up of crystalline rocks, including granite and gneiss, with quartz and pyrite minerals, among others, and the clay content of soils is 12% (Butler et al. 2009).

The New York City (NYC) water system encompasses 5180 km^2, contains nineteen reservoirs and three controlled lakes, and supplies potable water to half the population of New York State. The West-of-Hudson (WOH) portion of the NYC system is located in the Catskill

Mountains ~160 km northwest of NYC, west of the Hudson River. Three watersheds in the WOH System were included in this study: Ashokan (NYC-EA), Neversink (NYC-NN), and Rondout (NYC-RR). A fourth sample site, Kensico (NYC-KEN), is part of NYC's East-of- Hudson (EOH) District, located east of the Hudson River ~ 24 kilometers due north of NYC. The Kensico watershed is very small (25.5 km^2) compared to the WOH watersheds (227.2-1329 km^2) and the terrain much less mountainous. From each of the studied watersheds five soil/litter samples were collected and composited to provide good representation of each watershed and sufficient material for testing. The geology of the WOH region is characterized by sedimentary bedrock of sandstone and shale, of which ~30% is exposed, while glacial till (60%) and alluvial deposition (10%) make up the remaining surface geology (Mehaffey et al. 2005). In the EOH region, surficial geology is mostly glacial till interspersed with kame deposits. Bedrock is mostly made up of metamorphic gneiss and schist which contain abundant marble formations resulting in soils with higher carbonate content than the WOH soils. Annual precipitation for the NYC water system region ranges from 500-1650 mm, with most rainfall occurring in spring and fall (Mehaffey et al. 2005). All watersheds in the NYC water system are predominantly deciduous although conifers are more prevalent at the higher elevations of the WOH watersheds. Tree species include northern hardwood trees such as maple, oak and birch, and conifers such as white pine and hemlock.

The City of San Francisco (SF) and the greater surrounding Bay Area relies on the Hetch Hetchy (HH) watershed located in Yosemite National Park for its primary source of drinking water. The snowmelt from the Sierra Nevada Mountains supplies water to the Tuolumne River which then drains to Hetch Hetchy Reservoir. The Hetch Hetchy water system is supported by the Alameda and Peninsula watersheds, which provide approximately 15% of water supplies to SF. Together, the SF water system supplies water to 2.6 million customers. Soil samples were collected from three sites within the SF water system: near Hetch Hetchy Reservoir, San Antonio Reservoir located within the Alameda watershed in the East Bay, and San Andreas Reservoir located within the Peninsula watershed in the West Bay. Litter samples were not collected for the SF watersheds. The area surrounding Hetch Hetchy Reservoir was burned in the Rim Fire of 2013, and the HH soil sample was collected in 2015 from the burned area.

CHAPTER 4
EXAMINATION OF CARBON, NUTRIENT, AND METAL RELEASES FROM LABORATORY HEATED FOREST SOILS AND LITTERS

OVERVIEW

The objective of this portion of the project was to investigate the role of heating on the release of DOM, nutrients, and metals from soil and litter into aqueous solution. DOM from the soil and litter samples was measured and these data were analyzed for trends that could help predict DOM release. In a set of sub-experiments designed to evaluate release of dissolved constituents subject to heating, we used samples collected from two watersheds that broadly represent the coniferous forests of the West, and deciduous forests of the East: (1) Kensico watershed in New York City's watershed and (2) Clear Creek watershed in Colorado (referred to as deciduous and coniferous sites, respectively, in results). The soils were characterized for organic matter content, major element composition, and mineralogy. The soil and litter fractions collected from each site were leached independently to examine the source of DOM released. Each of the materials (soil and litter) from the two sites were heated at temperatures of a) 225°C and b) 350°C for two hours. The heated leachates were compared to leachates from unheated material to evaluate the role of heating on the release of dissolved constituents. The authors recognize that the results may not be universally representative of conditions observed in the specific watersheds, but were used as an indication of expected effects.

The analysis of the aqueous release of carbon from the soil and litter was important for understanding the leachates used for the treatment tests (see Chapter 5). The parent terrestrial materials vary such that the organic matter leached into solution was predicted to be different. The litter material is freshly deposited and only minimally decomposed, while soils contain extensively degraded organic matter that can interact with mineral and organic surfaces. These differences were predicted to result in very different responses to heating and leaching.

The temperatures used are in the range that promote browning, but do not cause major mass losses. For example, exposure of conifer litter (pine and fir needles) to 225°C did not produce ash, instead the needles were lightly charred and retained their original structure. At the 350°C, the litter material was degraded such that about 50% of the material lost its initial form and structure, which was observed by the production of dark ash in the samples. In addition to DOM, the aqueous releases of nitrate, iron, and manganese were evaluated to assess the relative effects of heating on commonly measured constituents of interest for water quality.

Lastly, an experiment was conducted to evaluate the effects of solution pH on leaching by examining release of DOM under acidic and basic conditions. We predicted that pH would likely alter the release of DOM as higher pH promotes the release of soil organic matter (SOM) molecules. The effects of pH on the release of DOM from heated and unheated soils were evaluated on a subset of soils only, but provides insight into the character of DOM released. It was hypothesized that heating would alter the size and reactivity of the molecules. Specifically, at low temperature heating (225°C) molecules will be smaller and partially oxidized and high temps (350°C) molecules will be increasingly condensed and less soluble. Size exclusion chromatography was used to evaluate the role of heating on molecular size, an important characteristic for treatment and DOM reactivity.

PROCEDURE

Sample Processing

To generate composite samples, soils from each location were mixed by equal mass and litters were mixed by equal mass, resulting in a composite soil sample and a composite litter sample for each site. In total, approximately 115 individual samples were processed. These were condensed into 14 composite soil and 14 composite litter samples. For this analysis, seven Colorado and four New York State watershed samples were used to evaluate general leaching trends across forest types. Beyond the general analysis, one Colorado site and one New York site were chosen randomly to be investigated in greater details for comparison between coniferous and deciduous forest types and soil characteristics. Soil samples were collected by water utilities and/or by the University of Colorado team and transported or shipped to the University. Once received, soil samples were sifted through 6.25 mm mesh to remove rock and large forest material, then sifted through a #20 sieve. For soils, material that passed the fine sieve was retained for experiments. For litter, the material retained by the fine sieve was used for experiments. The sieved soils were dried at room temperature while being frequently stirred and weighed to determine moisture loss in ambient air.

Heating of soil and litter was carried out in muffle furnace where materials were spread in the bottom of foil boats (or crucibles) to produce evenly distributed heating. Soils were heated for 2 hours at 225°C, a temperature that produces light charring and some structural changes in organic matter. In this section, a subset of samples was also heated at 350°C to examine the role of increased temperature on soil and litter leachates.

Soil and Litter Characterization

Soil and litter samples collected form the deciduous and coniferous sites were characterized for organic content by performing loss on ignition measurements. Each material was weighed in a pre-tared crucible and then heated at 550°C for 4 h. Following cooling in a desiccator, the material was again weighed. The mass difference represents the organic matter that is lost to thermal oxidation and represents the organic matter content of the material (Dean 1974). Soil elemental analysis (Fe, Al, Si, Mn, Na, K, Mg, Ca) was determined by Inductively Coupled Plasma Optical Emission Spectrometry (ICP-OES) after perchloric acid digestion following the methods of Farrell et al. (1980). Soil minerology was determined by quantitative XRD. Samples were prepared for XRD using a method based on Eberl (2003). Samples were analyzed using a Siemens D500 X-ray diffractometer from 5 to 65 degrees two theta using Cu Ka X-ray radiation, with a step size of 0.02 degrees and a dwell time of 2 seconds per step. Quantitative mineralogy was calculated using the USGS software, RockJock (Eberl 2003). Molecular size distribution in solution was determined using size exclusion chromatography (SEC) with a high-liquid chromatograph (HPLC) equipped with a protein-pak column (Waters) and UV detector (280 nm) and calibrated using polystyrene sulfonate standards (PSS, 210, 1000, 4300, 6800, and 17000 Da). A 5 mM sodium sulfate and phosphate buffer solution (pH 6.8) was used as the mobile phase at a flow rate of 0.7 mL/minute. Values are reported as weight averaged molecular weight estimates relative to the PSS standards.

Leaching Procedure and Analysis

Leaching experiments were conducted using the soil and litter collected from each site to assess DOM release. Leaching experiments were conducted on soil and litter independently and also soil and litter mixtures. Leaching tests were performed by adding 1.25 g soil or litter to 100 mL of high purity (18.2 MΩ) laboratory water. Soil to water ratios were varied to evaluate concentration effects on the release of DOM; none were detected. Solids were allowed to equilibrate for 24 h and were then filtered through 0.7 μm Whatman glass fiber filters. A small subset of samples was filtered with 0.2 μm polyethersulfone (Supor®, Pall Life Science) membranes for a comparison of organic matter size fractions during the experiments evaluating solution pH on DOM release. The filtered samples were transferred to amber glass vials (40 mL, I-chem^{TM}, Fisher Scientific, USA) and refrigerated prior to analysis.

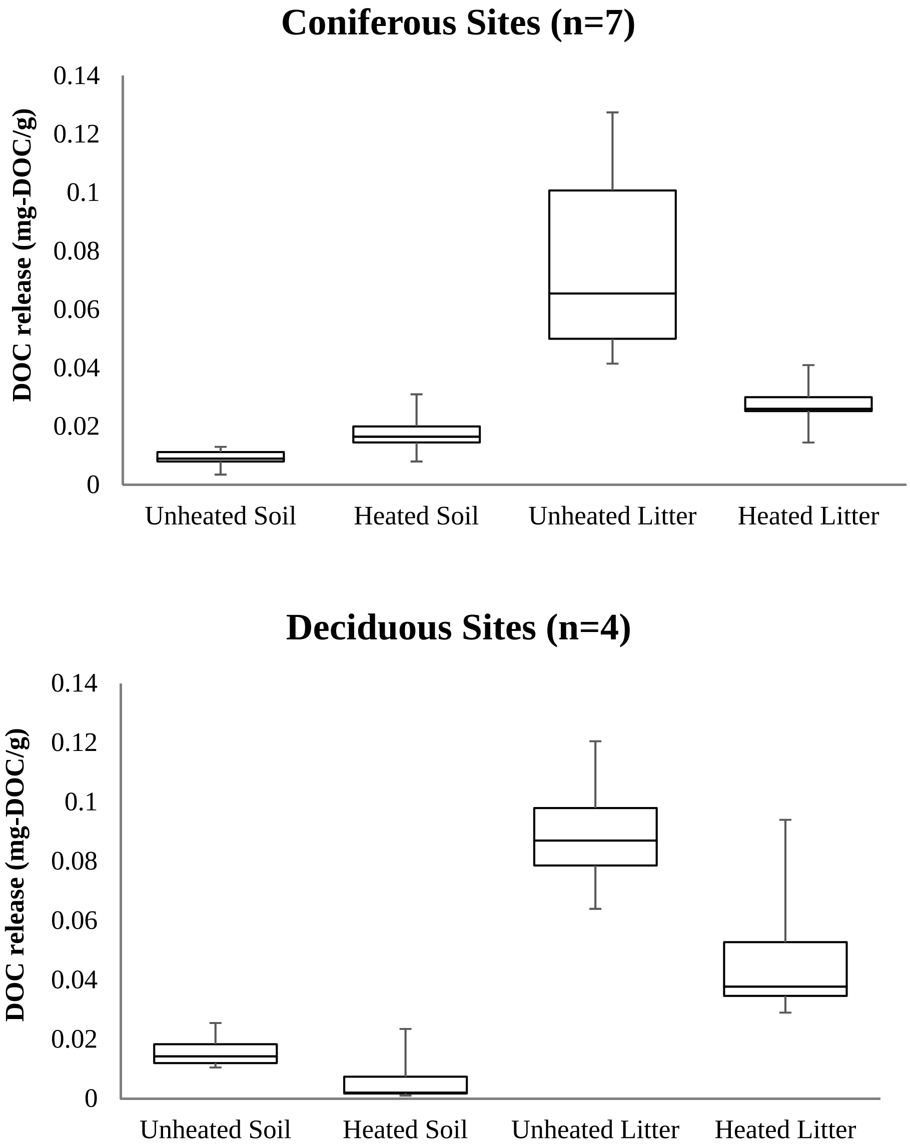
Leachates were characterized by UV absorbance and DOC measurements. UV_{254} was measured using a UV-Vis spectrophotometer (Cary 100, Agilent Technologies) and a 1 cm quartz cuvette. A Siever's 5310 C analyzer (GE Analytical Instruments) was used for DOC measurements. Solution pH was measured using a pH electrode that was calibrated daily using buffers of pH 4, 7, and 10. Samples for metal and ion analyses were stored in low-density polyethylene bottles. Nitrate was quantified by ion chromatography (IC; Dionex IonPac® AS14A column). Total iron and manganese concentrations were determined for filtered samples following acidification using ICP-OES. Select samples filtered through 0.2 μm membranes were characterized for size distribution using size exclusion chromatography.

RESULTS AND DISCUSSION

Figure 4.1 presents the DOC release per mass of soil for the two types of sites evaluated. Soils varied markedly in organic content. The soils had a mean organic content of 13 ± 9.0% and litter contain a mean organic content of 79 ± 11.8%. The release of DOC per unit soil was similar for both coniferous and deciduous sites (approximately 0.015-0.02 mg DOC per g of soil). In the case of the coniferous soil, this increased after heating whereas for deciduous materials there was a slight decrease.

The difference in DOM release behavior between the coniferous and deciduous sites may be related to the difference in the SOM content of the soils; the deciduous materials had a higher SOM content. There was no discernable trend between the litter organic matter (OM) content and the release of DOM (data not shown). It is assumed that non-degraded organic structures, such as pine needles or leaves, will release DOM from the surface of the material only and therefore much of the organic mass measured is part of the physical structure and unlikely to result in mass-dependent leaching. Conversely, soils have much greater surface area per unit mass and less of the organic material is associated with the physical structure of needles and leaves.

There was no discernable relationship between initial organic content of soil and release of DOM after heating. In general, the heated coniferous soils (collected from Colorado) released more organic carbon into solution compared to the respective unheated samples (Figure 4.1). Release of DOM from soils following heating was not consistent across the sites. The deciduous soils (collected from New York City) generally behaved opposite of the coniferous site, releasing less DOM into solution following heating. Whereas, litter from both sites behaved similarly by releasing considerably less DOM following heating. One possible explanation for the lower release of DOM after heating an alteration in the oxygen functionality content of the SOM molecules. It has been shown that organic matter begins to lose carboxylic acids and oxygen functionality around 225°C (Knicker 2007). The results here indicate that deciduous sites generally contained greater organic material, so it is possible that soil release followed the organic-organic interactions similar to litter materials, while the low organic content coniferous soil DOM release was controlled by organic-mineral interactions. Current literature lacks an explanation for this observation, and further research is recommended.

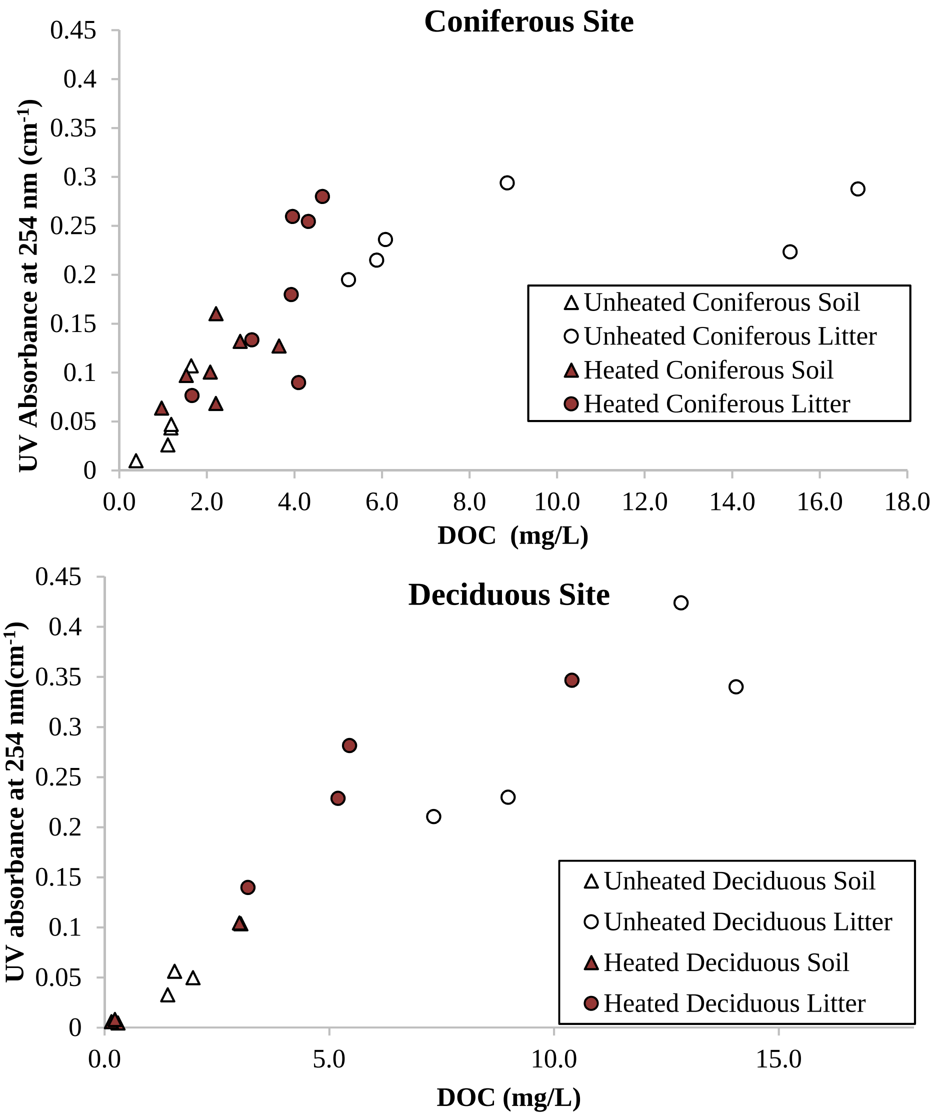


Release from heated material is plotted against the organic content of the unheated material. The coniferous site is from Colorado (Clear Creek Watershed) and the deciduous site from New York City (Kensico).

Figure 4.1 Dissolved organic matter (measured as DOC) released per gram of material as a function of soil and litter organic content

Optical Properties of Soil and Litter Leachate

In addition to quantifying DOM release, the UV absorbance at 254 nm was measured on leachates to optically characterize the DOM (Figure 4.2). Note that regression lines were not added for clarity. In general, there was a relationship between the amount of carbon in solution and the UV absorbance; greater DOM release resulted in greater absorbance by the solution. The absorbance at 254 nm is often normalized to the carbon content of the solution to obtain a $SUVA_{254}$ value, which is an indication of the aromatic content of the DOM (Weishaar et al. 2003). Using absorbance data from soil leachates has resulted in unreliable data in previous studies (Gabor et al. 2015) yielding $SUVA_{254}$ greater than values commonly observed for DOM (i.e., 5.0 L/mg-m). Similarly, in this study $SUVA_{254}$ for several heated leachates was higher than typical values. The general absorbance trends suggest the DOM became more aromatic with heating, which is consistent with the visual indications of darkening of the materials. Average unheated soil leachate $SUVA_{254}$ was 3.4 L/mg-m and average $SUVA_{254}$ of the unheated litter leachates was 2.8 L/mg-m. Following heating the average $SUVA_{254}$ values increased to 4.7 L/mg-m for both soil and litter leachates. The relatively low DOC concentrations measured in soil leachates appeared to lead to elevated (>5 L/mg-m) $SUVA_{254}$ values, and it is likely the increasing colloid-to-DOC ratio causes the overestimation. The increase in aromatic content of DOM with heating is consistent with previous studies showing heating of biomass causes breakdown of woody material and produces carbon gases that mix with O_2 and cause combustion; if the heating stops prior to combustion the material is charred (charcoal; Scott 2010). Charring results in loss of O-alkyl and di-O-alkyl structures that dominate woody materials and an increase in aromatics (Certini 2005, Gonzalez-Perez et al. 2004), which supports the observed results.



The coniferous site is from Colorado (Clear Creek Watershed) and the deciduous site from New York City (Kensico).

Figure 4.2 UV absorbance at 254 nm as a function of DOC

In-depth Analysis of a Coniferous and Deciduous Site

In this section, an investigation of two sites broadly representing the deciduous and coniferous types were evaluated for differences in leachate characteristics. The parent soil material was considered due to the possibility of DOM-mineral interactions causing leaching characteristics beyond changes attributable to solely forest types or thermal alterations. The major elemental composition of the soils and litter from the coniferous and deciduous sites are presented in Table 4.1. The elemental composition and clay content did not differ greatly between the two sites examined, as shown in Table 4.2. The results did not provide evidence to explain difference in DOM release behavior between the two soils. It has been demonstrated previously that DOM export from terrestrial environments is governed by interactions between DOM and soil material, primarily mineral surfaces (Sollins et al. 1996, Kaiser and Guggenberger 2003, Feng et al. 2005).

The extent of DOM sorption depends on the character of soil and DOM, as well as solution chemistry (Tipping 1981, Feng et al. 2005, Gabor et al. 2015). The sorption of DOM is primarily due to carboxyl and hydroxyl functional groups of DOM interacting with oxide surfaces in the mineral soil, specifically under acidic or slightly acidic conditions (Gu et al. 1994, Feng et al. 2005). In this study, the influence of soil clay content on DOM release following heating was not clear, and requires further exploration.

Table 4.1
Major constituents of coniferous and deciduous soils and litters

	OM	Si	Al	Fe	Ca	Mg	Na	K	Ti	Mn	P
	%										
Coniferous Soil	5.6	28.2	7.4	6.4	2.6	2.2	2.4	1.5	0.5	0.1	0.1
Coniferous Litter	87.4	4.3	1.1	1.0	1.5	0.3	0.4	0.3	0.1	0.1	0.1
Deciduous Soil	9.5	29.4	6.4	3.9	1.3	0.9	1.4	1.3	0.4	0.1	0.1
Deciduous Litter	75	3.9	0.7	0.3	1.3	0.2	0.2	0.3	0.0	0.1	0.1

Table 4.2
Mineralogy of coniferous and deciduous soils

	Coniferous Soil	**Deciduous Soil**
Mineral	Weight %	Weight %
NON-CLAYS		
Quartz	15.7	40.2
Kspar (intermediate microcline)	7.0	6.8
Kspar (anorthoclase)	6.0	10.9
Plagioclase (albite, var. cleavelandite)	8.3	3.8
Plagioclase (oligoclase; Norway)	13.1	14.0
Plagioclase (anorthite)	5.6	1.5
Amphibole (ferrotschermakite)	2.5	0.0
Amphibole (actinolite)	30.4	10.3
Total non-clays	**88.6**	**87.6**
CLAYS		
Kaolinite (disordered)	5.0	6.8
Illite (1M; RM30)	6.4	5.6
Total clays	**11.4**	**12.4**

Nutrient and Metal Release into Solution

Analysis of unheated and heated deciduous and coniferous soil and litter leachates (unheated, 225°C, 350°C; 12 leachates total) revealed a wide range of anions and metals in solution (Table 4.3). Sulfate showed the strongest dependence on heating and sulfate concentrations were found to increase markedly with heating. The litter showed a different response, and after heating at 350°C, litter from both the deciduous and conifer forest types released nearly 10 times the amount of sulfate in comparison to the respective unheated litters. The release of sulfate from the forest material is likely a result of organic sulfur oxidation as nearly all sulfur exists in organic form in forest soils (Solomon et al. 2003). Phosphate release from the materials was not as consistent as sulfate. For instance, phosphate release from the coniferous soil was similar to sulfate release, but phosphate release from coniferous litter differed. The release of phosphate from coniferous litter increased following heating at 225°C, but decreased markedly in the litter exposed to 350°C. The leaching response of both litters was similar: phosphate release increased following heating at 225°C with respect to unheated samples, but samples heated at 350°C showed lower phosphate release than either the unheated or 225°C heated litters. Phosphate release could be a result of organic phosphorous oxidation, similar to sulfate. Nitrate release generally decreased following heating. There was one exception to this pattern; deciduous litter released a greater amount of nitrate into solution following heating. It is not clear what caused the general decreases of nitrate following heating. Nitrogen is a common component of organic matter and is likely oxidized when exposed to heating.

The concentration of iron and manganese, measured as total Fe and total Mn, showed similar trends and both metals tended to follow the general trends observed for DOM release (Table 4.3). Heating increased the release of iron and manganese from soils, but only caused minimal release from litter. Further, when heated at 350°C, there was no measurable release of iron and greatly diminished release of manganese. It is unclear exactly what drives this behavior, while the close parallel with DOM release suggest the thermal changes that influence carbon release may be closely tied to metal release.

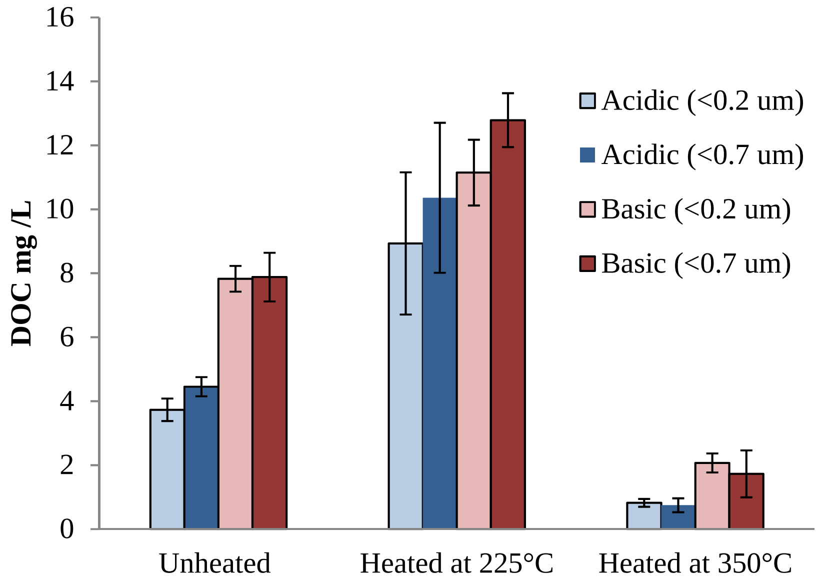
Table 4.3
Release of anions and total iron and manganese from unheated and heated materials. DL indicates below detection limit

		NO_3^-	PO_4^{3-}	SO_4^{2-}	Mn	Fe
		-----------------------	ppm	-----------------------	-------ppb	-------
Coniferous Soil	Unheated	0.3	0.3	0.1	19	17
	225°C	0.1	0.6	0.8	180	33
	350°C	DL	0.9	3.3	9.9	DL
Coniferous Litter	Unheated	0.2	2.4	0.4	92	23
	225°C	0.1	5.0	9.3	120	19
	350°C	0.1	0.9	30	DL	DL
Deciduous Soil	Unheated	0.3	0.3	0.2	22	31
	225°C	DL	DL	1.8	960	220
	350°C	DL	DL	3.6	440	DL
Deciduous Litter	Unheated	0.2	2.3	0.1	270	22
	225°C	DL	2.5	7.1	830	24
	350°C	0.5	1.4	25	5.9	DL
Duplicates						
Coniferous Soil	Unheated	0.5	0.3	0.2	15	21
Coniferous Litter	350°C	0.1	0.8	30		
Deciduous Litter	350°C	0.5	0.3	20	4.9	DL

Effects of pH on Heated and Unheated Soil DOM Release

To examine the influence of pH on DOM release, we leached the coniferous soils at multiple pH levels. Leaching of the coniferous soils revealed that the pH of solution, at the extremes of pH 3 and pH 11, will alter the release, but does not explain the trends of DOM release following heating (Figure 4.3). Note that in Figure 4.3 two filter sizes were used, 0.2 and 0.7 um. The amount of DOM released from unheated soil nearly doubled when leached in a basic solution compared to an acidic solution. The release of DOM from soil heated at 225°C was less sensitive to the change in pH compared to the unheated soil. Wildfires can cause changes in soil solution chemistry, including pH shifts, which can influence DOM mobilization (Molina et al. 2007, Quill et al. 2010, Periera et al. 2012). Increases in pH have been correlated with increasing burn severity due to organic acid denaturation and formation of K and Na oxides, hydroxides, and carbonates (Vega et al. 2013, Revchuck and Suffet 2014, Bodi et al. 2014). The increase in acid extractable DOM following heating and the decrease in sensitivity to base extraction following heating suggests that the molecules available for release into solution tend to be more soluble and smaller. These types of organic acids are referred to as fulvic acids, and are not removed from solution during acidification. From these experiments, it appears the majority of extractable OM is less sensitive to low pH and not greatly increased by basic extraction.

Leachates from the pH experiments were evaluated with size exclusion chromatography (SEC) (Table 4.4), which revealed that heating decreased the average molecular weight (MW) of the released DOM. These results are in agreement with the pH release data and together they provide evidence that heating results in smaller and increasingly homogenous DOM structures. The SEC results also support the results of the pH experiments; the smaller molecules are likely to be increasingly soluble. The effects of base extraction would tend to favor mobilization of larger molecules.



Acidic samples were leached at pH 3 and basic samples were leached at pH 11. Experiments were conducted for the coniferous soil, in triplicate. Bars represent mean concentrations and error bars represent the standard deviation.

Figure 4.3 DOC release for different pH conditions

Table 4.4
Size exclusion chromatography results of dissolved organic matter released from coniferous soil at pH 3 and pH 11. Samples were filtered at 0.2 microns.

pH	Temperature °C	MW (Da)
3	--	2087
3	225	1633
3	350	1434
11	--	2521
11	225	1889
11	350	1528

SUMMARY

The findings of this chapter demonstrate that soil and litter release different quantities and qualities of dissolved constituents following heating. Litter released more carbon compared to soils, and the effect of heating on soils varied. Experiments examining the effects of pH suggest a loss of acid-base functionality during heating that decreased pH effects on DOC release. In additional to decreases in acid-base functional group content, heating produced smaller, more homogenous molecules. The loss of oxygen containing functional groups, particularly carboxylic acids, may explain the decrease in DOM release from heated litter and organic rich materials, while the same loss of oxygen containing functional groups could explain the greater release of DOM from mineral-dominated soils. The loss of carboxylic acid functionality is also supported by the trends in pH dependent release from soil, where loss of acidic functional groups would make the release of DOM less pH dependent.

The release of anions and metals was altered following heating. Anion release into solution showed strong heating dependence, but was not consistent for the measured species. Sulfate demonstrated the most consistent behavior of the anions, increasing with increased heating for all materials. Specifically, the litter released nearly ten times more sulfate than the soils following heating. Nitrate generally decreased following heating for soils and litter. Phosphate release was not consistent for the soils, but phosphate release from litter increased after heating at 225°C and decreased, with respect to the release from unheated material, after heating at 350°C. The metals, iron, and manganese, had similar release trends that generally demonstrated greater release after heating at 225°C, while almost no release of either metal was measured following heating at 350°C.

CHAPTER 5
TREATMENT PROCESS PERFORMANCE EVALUATION OF LABORATORY LEACHATES

INTRODUCTION

To isolate the fundamental effects of heating during wildfire on drinking water treatment, surface soil and litter samples were heated in a furnace and leached in the laboratory. The majority of literature studying laboratory-based heat alterations to forest material is focused in the biogeochemical community, while the impact on water treatment has yet to be fully explored (Wang et al. 2015a). Water providers are concerned about the effects of increased DOM, potentially of altered character, following wildfire. The objective of this work was to address the removal efficacy of heat-altered DOM by conventional treatment and additional processes. The raw and finished water DBP levels were also evaluated. Unheated (control) and heated leachates were coagulated and characterized for water quality, optical properties, TTHM, HAA5, haloacetonitrile (HAN4), and chloropicrin formation before and after treatment.

A laboratory-based approach was used to investigate the effects of forest floor heating during wildfire on drinking water treatment and finished water quality for utilities. Surface soil and litter samples from four watersheds (Denver, New York, Westminster, and San Francisco; described in Chapter 3) were heated in a furnace at 225°C and leached to evaluate changes to water-soluble compounds, and the subsequent treatment process implications. Pairwise data analysis was used to understand the changes in water quality likely associated with heating. Paired differences (Δ) between the heated and control samples (heated – control) were calculated for each site, and the respective parameter analyzed.

LEACHING METHODS

The soil and litter samples collected as described in Chapter 3 were placed in ceramic evaporating dishes or aluminum pans in a pre-heated (225 °C) muffle furnace for two hours under oxic conditions. The soil and litter samples were added to low-carbon tap (LCT) water (treated with granular activated carbon) and manually mixed. The samples were leached in the dark at room temperature to evaluate water-soluble compounds. LCT water was used to maintain a representative inorganic water quality matrix for treatment tests (DOC < 0.3 $\mathrm{mg_C L^{-1}}$; pH = 7.1; alkalinity = 50 mg $\mathrm{CaCO_3 L^{-1}}$; conductivity = 100 μS). Following 24-h the leachates were passed through a 2 mm sieve, decanted, and refrigerated at 4 °C. A previous study showed after 6 hours, 75–99% of the 24-h DOC was leached into solution (Cawley et al. 2017). Soil and litter were leached together for treatment tests, representative of a mixture likely to enter a treatment facility. Soil and litter were added in equal amounts by weight, and diluted with LCT water to a DOC concentration of 5.0 ± 1 $\mathrm{mg_C L^{-1}}$. San Francisco leachates contained only soil (litter samples were not collected) and were diluted to a DOC concentration of 2.5± 0.5 $\mathrm{mg_C L^{-1}}$. Subsamples of the leachates were filtered through 0.45 μm polyethersulfone membrane filters (pre-rinsed) for optical property measurements and DOC analysis, and glass fiber filters (GF/F- combusted 3 h at 550°C) for chlorination experiments and DBP analyses.

ANALYTICAL METHODS

For the leachates, turbidity was measured with a HACH 2100 N turbidimeter. A Siever's 5310 C analyzer (GE Analytical Instruments) was used for DOC measurements. Analysis of total dissolved nitrogen (TDN) was performed with a TOC-V analyzer (Shimadzu Corp., Japan). Ultraviolet absorbance was measured with a 1-cm path length quartz cuvette and a Cary 100 UV-Vis spectrophotometer (Agilent Technologies). Specific ultraviolet absorption ($SUVA_{254}$) was calculated by normalizing UV_{254} absorbance by the DOC concentration and absorbance path length (Weishaar et al. 2003). Nitrate and nitrite were measured with an analytical flow solution IV spectrophotometric analyzer (OI Analytical, USA). Ammonium was analyzed with a BioTek Synergy 2 Microplate Reader. Dissolved organic nitrogen (DON) was calculated from the difference between TDN and inorganic nitrogen. For several samples the inorganic N concentration was high (> 60% of TDN), and the DON calculation may be less accurate (Lee and Westerhoff 2005). EPA Method 551.1 (1995) was followed for the analysis of total trihalomethanes (TTHMs) and haloacetonitriles (HANs). HAN4 analysis included: di-chloroacetonitrile (DCAN), tri-chloroacetonitrile (TCAN), di-bromoacetonitrile (DBAN), and bromo-chloroacetonitrile (BCAN). EPA method 552.2 was used for analysis of haloacetic acids (HAAs) and reported as the five-regulated species (HAA5). An Agilent 6890 Gas Chromatography system with an electron capture detector was used.

TREATMENT METHODS

Denver Water and New York City

Jar tests were performed using a 6-jar programmable jar tester (Phipps &Bird model 7790-901) with 2-liter B-KER^2 jars. Aluminum sulfate ($Al_2(SO_4)_3 18H_2O$, Mallinckrodt Chemicals, 3208-04) was applied at doses ranging from 20-120 mgL^{-1}. The alum doses were selected from preliminary alum dose vs. DOC response curves. Generally, point of diminishing returns (PODR) criteria were used to select the final alum dose (White et al. 1997). Mixing conditions included a rapid mix phase (1 minute at 290 rpm), two flocculation phases (10 minutes at 55 rpm and 10 minutes at 20 rpm) and a 30-minute sedimentation period. Following sedimentation, pH and turbidity were measured and filtered samples were analyzed for DOC and UV_{254}, and chlorinated for DBP analysis.

Westminster

Westminster leachates were treated with ferric chloride and pH adjusted with caustic either pre- or post- mixing and settling. Mixing conditions included a rapid mix phase (1 minute at 290 rpm), flocculation (4 minutes at 50 rpm), and a 1-minute sedimentation period. Four coagulation tests were performed on each leachate (45 and 90 mg/L, pre- and post pH adjustment). Raw and treated samples were filtered and chlorinated following uniform formation conditions (UFC) (Summers et al. 1996) and analyzed for C- and N-DBPs. Treated samples were also analyzed for turbidity, DOC, and UV_{254}.

Chlorination Conditions

DBP formation was evaluated for all samples with bench scale chlorination following uniform formation conditions (UFC), representative of a typical water treatment plant (Summers et al. 1996). Samples were chlorinated with a buffered sodium hypochlorite dosing solution (pH 8) at 20 °C. Borate buffer was added to samples to maintain a pH of 8.0 ± 0.3, as necessary. Preliminary chlorine demand curves were used to determine doses required for a 24-hour free chlorine residual of 1.0 ± 0.4 mg/L. Chlorine residuals were measured with the DPD (N,N-diethyl-p-phenylenediamine) colorimetric method (SM4500-Cl G) and quenched with ammonium chloride immediately following 24 hours.

San Francisco Treatment Tests

San Francisco leachates (soils only) were treated to simulate treatment plant operations for three facilities: East Bay (EB), West Bay (WB), and Hetch Hetchy (HH). Bench-scale treatment of Hetch Hetchy leachates included pre-chlorination (dose = 1.8 mg/L; pH = 8.0; contact time = 16 hours), coagulation (15 mg/L alum and 0.3 mg/L polymer; pH = 6.8), followed by flocculation/sedimentation/filtration. Samples were chloraminated (chlorine = 1.8 mg/L and ammonia = 0.6 mg_N/L; pH = 9.6) and held for 24 hours prior to quenching with sodium thiosulfate and DBP analysis. Treatment of the East Bay leachates included pre-chlorination (dose = 2.5 mg/L; contact time = 6 hours), coagulation (35 mg/L alum and 1.2 mg/L polymer; pH = 7.0), followed by flocculation/sedimentation/filtration. Samples were chlorinated (chlorine dose = 1.0 mg/L; contact time = 30 minutes) and chloraminated (chlorine dose = 1.8 mg/L and ammonia dose = 0.6 mg_N/L; pH = 8.8) and held for 24 hours prior to quenching with sodium thiosulfate and DBP analysis. Treatment of the West Bay leachates included pre-ozonation (dose = 1.5 mg/L; contact time = 10 minutes) coagulation (1.5 mg/L ferric chloride and 3.0 mg/L polymer), flocculation/sedimentation/filtration. Samples were chloraminated (chlorine dose = 2.8 mg/L and ammonia dose = 0.6 mg/L; pH = 9.0) and held for 24 hours prior to quenching with sodium thiosulfate and DBP analysis. Polymer was provided by SFPUC.

DENVER WATER TREATMENT PROCESS PERFORMANCE

DW control and heated samples were generated following heating and leaching, and analyzed for water quality. As shown in Table 5.1, following heating the pH and alkalinity of the leachates increased compared to the control samples (mean Δ = +0.2 and +6.6 $mgCaCO_3$/L). Turbidity levels, inorganic nitrogen, DON, and DOC concentrations are presented, but reflect the amount of material (soil and litter) leached, rather than the effects of heating. Therefore, where appropriate, the sample results were normalized by the concentration of DOC, or the total mass of litter and soil leached. Soil and litter samples were mixed together, leached, and diluted to a DOC concentration of 5.0 ± 1.0 mg_C/L for better comparison of the samples at a realistic DOC concentration. The DOC:DON ratios imply the heated leachates were enriched in organic nitrogen compared to control samples (mean Δ = -20.8 mg_C/m_N). DOC leached per gram of solid material was lower after heating (mean Δ = -1.3 mg_C/g). $SUVA_{254}$ was consistently higher for the heated samples compared to the control leachates (mean Δ = +0.6 L/mg-m).

Table 5.1
Raw water quality for Denver Water control and heated samples. Samples are leached mixtures of soil and litter. Soil and litter samples were collected from three different sites within the DW system.

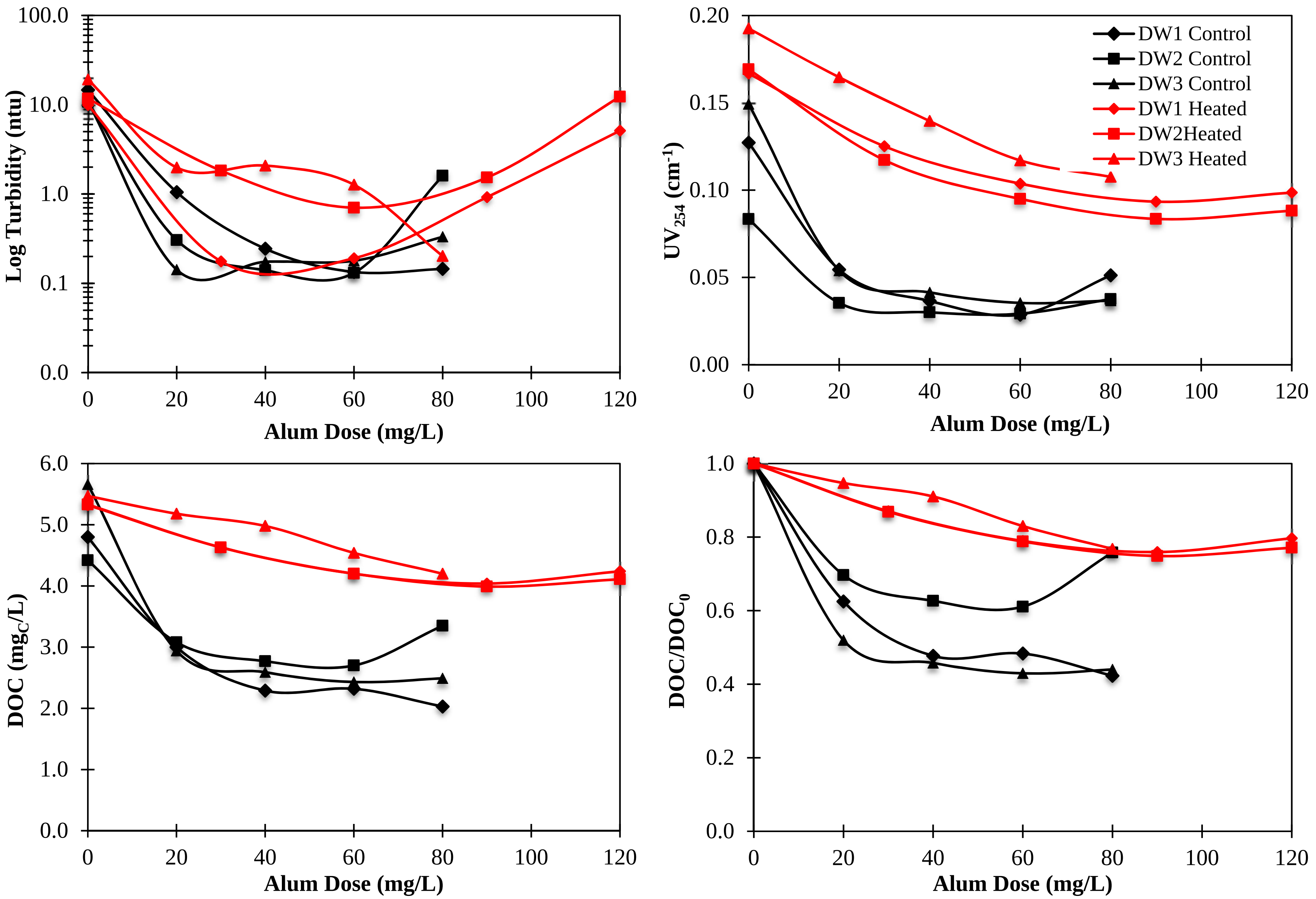
Denver Water	**Alkalinity ($mgCaCO_3/L$)**	**pH**	**Turbidity (ntu)**	**TDN (mg_N/L)**	**NO_3+NO_2 (mg_N/L)**	**NH_4^+ (mg_N/L)**	**DON (mg_N/L)**	**DOC (mg_C/L)**	**DOC Leached (mg_C/g)**	**DOC:DON (mg_C/mg_N)**	**$SUVA_{254}$ (L/mg-m)**
DW1 Control	39	7.6	15.2	0.24	0.105	0.042	0.09	4.8	3.6	54.4	2.7
DW1 Heated	44	7.7	9.9	0.25	0.007	0.020	0.23	5.3	1.6	23.3	3.2
DW2 Control	31	7.4	10.7	0.10	<0.001	0.082	NA	4.9	2.1	NA	2.6
DW2 Heated	45	7.5	11.7	0.29	<0.001	0.071	0.22	5.3	1.1	24.5	3.2
DW3 Control	50	7.5	10.8	0.28	0.069	0.014	0.20	5.7	2.8	29.2	2.6
DW3 Heated	52	7.8	19.2	0.37	<0.001	0.079	0.29	5.5	2.0	18.8	3.5
Average Δ (heated – control)	**+6.6**	**+0.2**	**+1.4**	**+0.10**	**-0.06**	**+0.01**	**+0.12**	**+0.2**	**-1.3**	**-20.8**	**+0.6**
Stdev Δ	**6.4**	**0.1**	**6.8**	**0.09**	**0.05**	**0.05**	**0.03**	**0.4**	**0.6**	**14.6**	**0.2**

Raw leachates were chlorinated and analyzed for DBP formation to address changes to DBP precursors following heating (Table 5.2). TTHM formation showed variable responses to heating, increasing for one sample and decreasing for two samples, compared to the controls (mean Δ = +2.2 μg/L). HAA5 formation of the heated leachates was on average lower than the control samples (mean Δ = -35.3 μg/L), but DW2 showed an increase in HAA5 formation following heating. HAN4 formation of the control and heated leachates was variable, both increasing and decreasing following heating (mean Δ = +0.1 μg/L). However, chloropicrin formation was consistently elevated following heating (mean Δ = +6.9 μg/L). Although samples were diluted to similar DOC concentrations, carbon normalized DBP yields allow for a better comparison of the DBP precursor reactivity of the control and heated samples (Table 5.2). Following heating, on average TTHM yields slightly decreased (mean Δ = -0.7 μg/mg$_{C}$), while mean HAA5 yields showed a larger decrease (mean Δ = -7.9 μg/mg$_{C}$). HAN4 yields showed varied responses for the control and heated samples, and on average showed minimal difference (mean Δ = 0.0 μg/mg$_{C}$), whereas chloropicrin yields increased following heating (mean Δ = +1.3 μg/mg$_{C}$).

Table 5.2
Raw water DBP formation and carbon normalized DBP yields for Denver Water control and heated samples. Samples are leached mixtures of soil and litter. Soil and litter samples were collected from three different sites within the DW system.

Denver Water	Chlorine Dose (mg/L)	TTHM (μg/L)	TTHM Yield (μg/mg$_{C}$)	HAA5 (μg/L)	HAA5 Yield (μg/mg$_{C}$)	HAN4 (μg/L)	HAN4 Yield (μg/mg$_{C}$)	Chloropicrin (μg/L)	Chloropicrin Yield (μg/mg$_{C}$)
DW1 Control	5.5	181.8	37.9	234.9	48.9	4.2	0.9	4.3	0.9
DW1 Heated	5.4	149.4	28.3	197.1	37.4	6.3	1.2	11.4	2.2
DW2 Control	4.6	93.1	19.1	92.4	19.0	5.4	1.1	1.9	0.4
DW2 Heated	6.1	192.4	36.1	130.2	24.4	3.1	0.6	14.0	2.6
DW3 Control	6.8	186.3	32.7	256.0	44.9	5.9	1.0	6.5	1.1
DW3 Heated	6.0	125.9	23.0	150.0	27.4	6.6	1.2	7.9	1.4
Average Δ (heated – control)	**NA**	**+2.2**	**-0.7**	**-35.3**	**-7.9**	**+0.1**	**0.0**	**+6.9**	**+1.3**
Stdev Δ	**NA**	**85.3**	**15.3**	**71.9**	**11.9**	**2.3**	**0.5**	**5.3**	**1.0**

The DW leachates were coagulated to address the effects of heating on conventional treatment processes. Preliminary jar tests were conducted to select the optimal alum dose for DOC removal. Distinct differences in the coagulation response for the control and heated samples were apparent (Figure 5.1). In most cases, turbidity of the heated leachates remained elevated compared to the control leachates for alum doses ranging from 20-60 mg/L. At doses greater than 60 mg/L particles commonly re-stabilized due to a decrease in pH, and turbidity increased relative to lower doses. The control leachates showed typical DOC dose responses (Figure 5.1), with considerable removal occurring at the lowest applied dose (20-30 mg/L). For the control samples the treated water DOC concentrations were generally <3.0 mg/L at doses ranging from 20-40 mg/L. However, the DOC of the heated leachates remained high >4.0 mg/L at doses of 30-40 mg/L, and showed only marginal removal (<20%). Although the heated leachate DOC concentration continued to decrease with increasing alum dose, the DOC remained near 4.0 mg_C/L following treatment with 80–120 mg/L alum. The raw water UV_{254} absorbance of the heated samples was initially higher than the control leachates, despite having similar DOC concentrations. As such, the UV_{254} of the heated samples remained higher than the control samples, at all applied alum doses (20–120 mg/L).



DOC_0 = initial DOC concentration.

Figure 5.1 Alum dose-response for Denver Water control and heated leachates

Following preliminary jar tests, leachates were coagulated at a final dose, filtered, and chlorinated to characterize the DBP levels likely to enter a distribution system. The required alum dose was on average 13.3 mg/L higher for the heated leachates compared to the control samples (Table 5.3). Even at a higher alum dose, the settled water turbidity levels were on average 1.3 ntu greater for heated samples, and treated water DOC concentrations remained higher compared to the control leachates (mean Δ = 2.1 mg$_C$/L). Treated water heated samples showed higher SUVA$_{254}$ values, indicating considerable aromatic material still remained compared to control leachates (mean Δ = +1.1 L/mg$_C$-m). The poor DOC removal for the heated samples resulted in elevated TTHM, HAA5, and chloropicrin levels compared to the control samples, following coagulation treatment. HAN4 formation was similar for the control and heated samples following treatment. As expected from the poor treatability of the heated leachates, the percent decrease for all parameters was lower following conventional treatment. For the heated leachates the percent decrease of DOC (mean Δ = -13%), SUVA$_{254}$ (meanΔ = -7%), C-DBPs (TTHM mean Δ = -18%; HAA5 mean Δ = -22%), and N-DBPs (HAN mean Δ = -9%; chloropicrin mean Δ = -16%) were lower than the control samples (Table 5.4).

The lower DOC removal, and consequently poor removal of DBP precursors for the heated samples impacted the finished water DBP concentrations likely to enter a distribution system. As demonstrated in Figure 5.2, treated water DOC concentrations of the heated DW samples ranged from 3.9-4.5 mg$_C$/L, whereas the control leachate DOC levels were consistently below 2.5 mg$_C$/L. Accordingly, following treatment all heated leachates exceeded both TTHM and HAA5 maximum contaminant levels (MCLs), 80 and 60 μg/L, respectively, whereas all control samples were below MCLs. Chloropicrin concentrations of the heated samples (6.6-10.4 μg/L) were also higher than finished water control leachates (1.1-2.0 μg/L). Alternatively, following treatment, HAN4 concentrations were on average lower for the heated samples (mean Δ = -1.1 μg/L).

Table 5.3
Treated water quality and DBP formation for Denver Water control and heated samples. Samples are leached mixtures of soil and litter. Soil and litter samples were collected from three different sites within the DW system.

Denver Water	Alum Dose (mg/L)	Treated Turbidity (ntu)	Treated pH	Treated DOC (mg$_C$/L)	Treated SUVA$_{254}$ (L/mg$_C$-m)	Treated TTHM (μg/L)	Treated HAA5 (μg/L)	Treated HAN4 (μg/L)	Treated Chloropicrin (μg/L)
DW1 Control	40	0.3	6.5	1.6	1.7	35.6	27.4	4.1	1.1
DW1 Heated	45	1.3	6.6	4.1	2.8	107.2	82.2	3.6	8.1
DW2 Control	30	0.1	7.1	2.1	1.4	37.8	30.0	4.7	1.3
DW2 Heated	45	1.7	6.6	3.9	2.6	112.4	94.5	3.6	10.4
DW3 Control	30	0.1	7.0	2.4	1.8	48.7	41.0	8.3	2.0
DW3 Heated	50	1.4	6.8	4.5	2.9	100.9	110.0	6.4	6.6
Average Δ (heated – control)	**+13.3**	**+1.3**	**-0.2**	**+2.1**	**+1.1**	**+66.1**	**+62.7**	**-1.1**	**+6.9**
Stdev Δ (heated – control)	**7.6**	**0.3**	**0.3**	**0.4**	**0.0**	**12.2**	**7.3**	**0.7**	**2.3**

Table 5.4
Percent removal following conventional treatment for Denver Water control and heated samples. Samples are leached mixtures of soil and litter. Soil and litter samples were collected from three different sites within the DW system.

Denver Water	% Decrease Turbidity	% Decrease DOC	% Decrease $SUVA_{254}$	% Decrease TTHM	% Decrease HAA5	%Decrease HAN4	% Decrease Chloropicrin
DW1 Control	98	67	38	80	88	3	74
DW1 Heated	87	22	12	28	58	42	29
DW2 Control	99	56	45	59	68	14	31
DW2 Heated	89	27	18	26	24	-24	26
DW3 Control	99	57	32	74	84	-39	68
DW3 Heated	93	18	18	20	27	3	16
Average Δ (heated – control)	**-3**	**-13**	**-7**	**-18**	**-22**	**-9**	**-16**
Stdev Δ (heated – control)	**5**	**42**	**35**	**30**	**34**	**22**	**28**

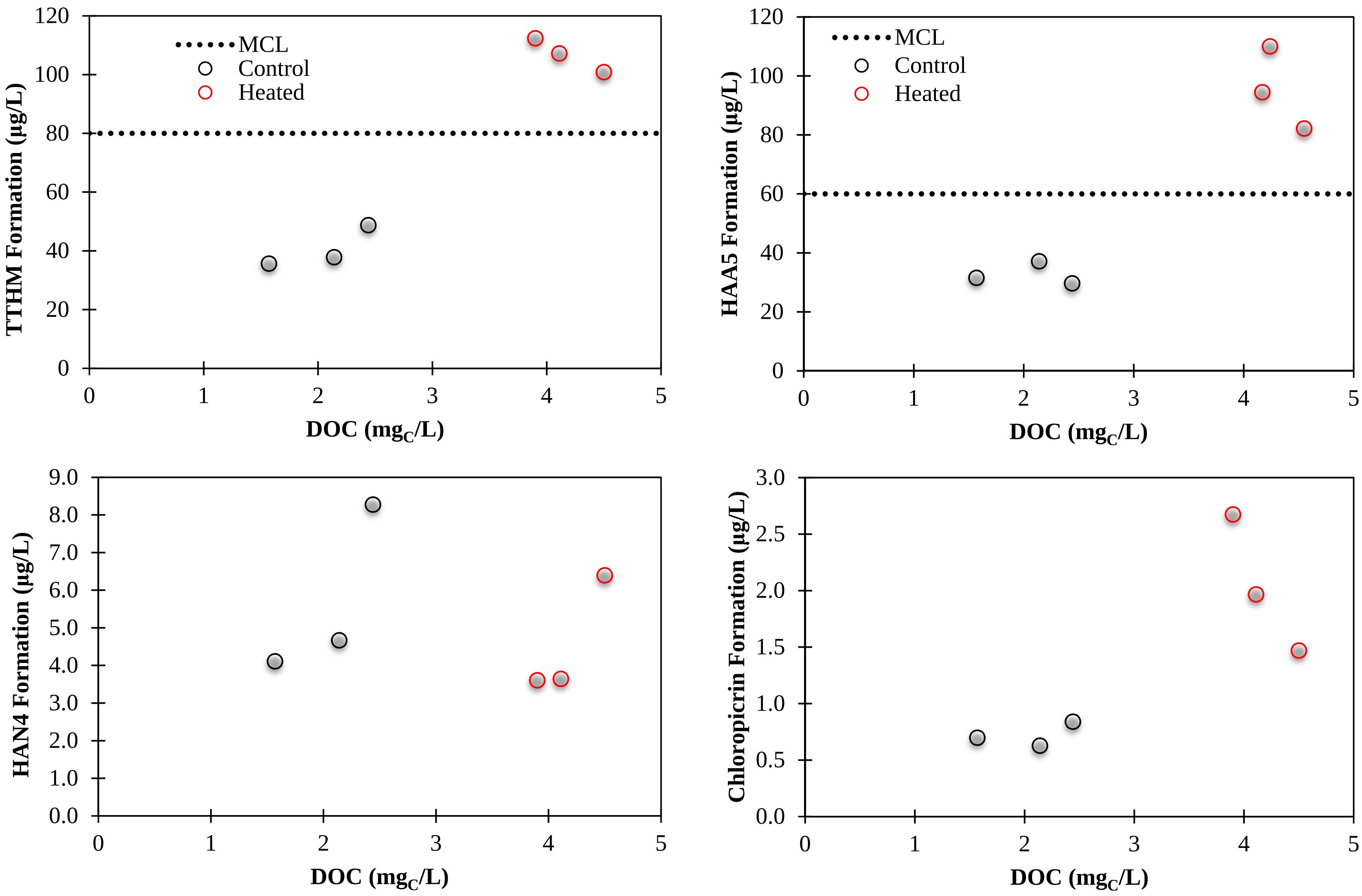


Figure 5.2 Treated water DOC and DBP formation for Denver Water control and heated samples

NEW YORK CITY TREATMENT PROCESS PERFORMANCE

NYC control and heated samples were generated following the heating and leaching methods presented above, and analyzed for water quality. As shown in Table 5.5, following heating the pH and alkalinity increased compared to the control samples (mean Δ = +0.2 and +18.1 $mgCaCO_3/L$). Turbidity levels, inorganic nitrogen, DON, and DOC concentrations are presented, but reflect the amount of material (soil and litter) leached, rather than isolating the effects of heating. Therefore, where appropriate, the sample results were normalized by the concentration of DOC, or the total mass of litter and soil leached. Soil and litter samples were mixed together, leached, and diluted to a DOC concentration of 5.0 ± 1.0 mg_C/L for better comparison of the samples at a realistic DOC concentration. The DOC:DON ratios imply the heated leachates were enriched in organic nitrogen compared to control samples (mean Δ = -4.9 mg_C/m_N). DOC leached per gram of solid material was lower after heating (mean Δ = -2.2 mg_C/g). $SUVA_{254}$ was consistently higher for the heated samples compared to the control leachates (mean Δ = +1.2 L/mg-m).

Table 5.5
Raw water quality for New York City control and heated samples. Samples are leached mixtures of soil and litter. Soil and litter samples were collected from four different sites within the NYC system.

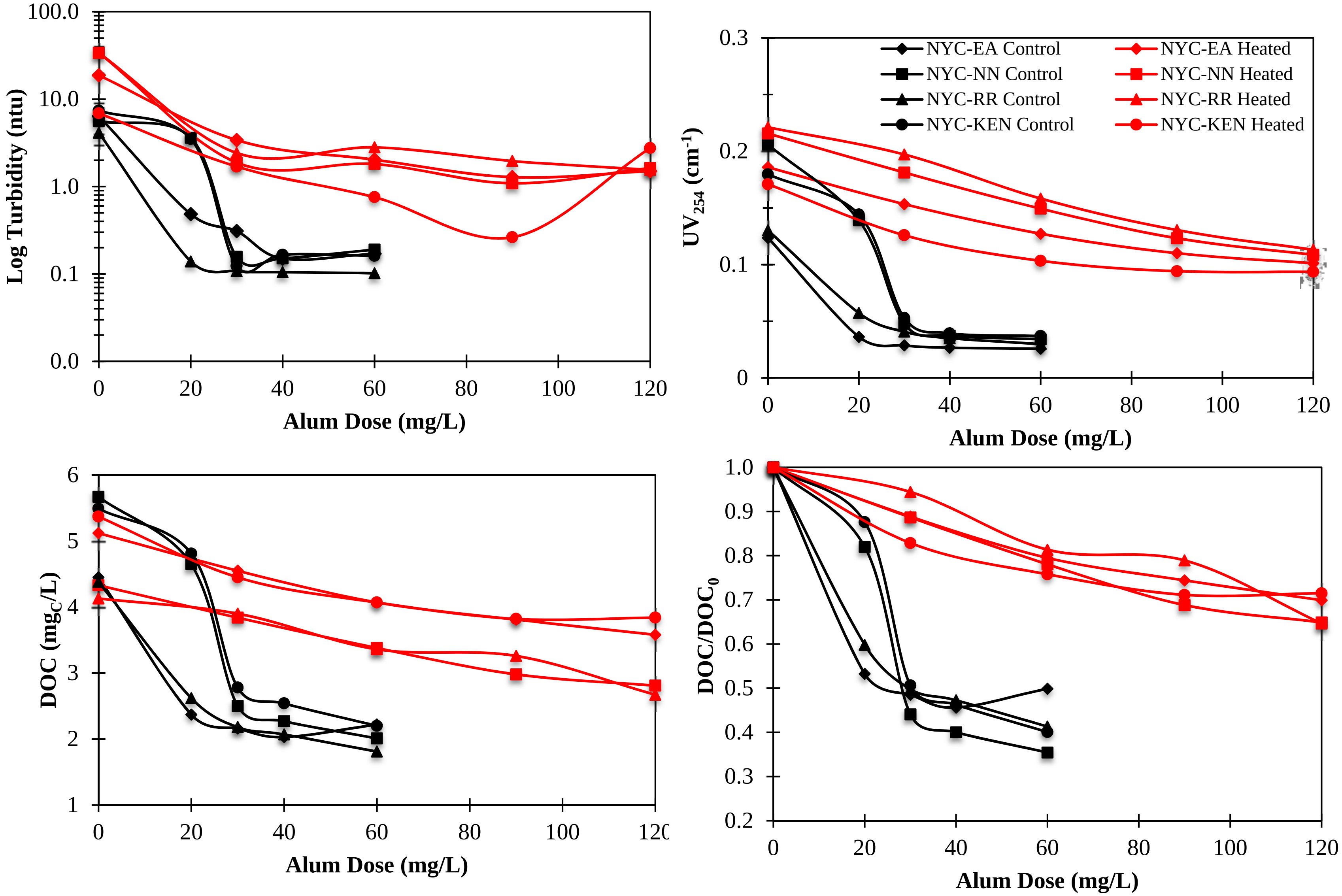
New York City	Alkalinity (mgCaCO$_3$/L)	pH	Turbidity (ntu)	TDN (mg$_N$/L)	NO$_3$+NO$_2$ (mg$_N$/L)	NH$_4^+$ (mg$_N$/L)	DON (mg$_N$/L)	DOC (mg$_C$/L)	DOC Leached (mg$_C$/g)	DOC:DON (mg$_C$/mg$_N$)	SUVA$_{254}$ (L/mg$_C$-m)
NYC-EA Control	41	7.3	8.2	0.59	0.036	0.377	0.18	4.6	3.8	26.2	2.6
NYC-EA Heated	69	7.8	18.7	0.52	0.009	0.289	0.23	5.1	1.2	22.7	3.6
NYC-NN Control	35	7.4	5.7	1.13	0.068	0.877	0.19	5.8	3.4	30.9	3.4
NYC-NN Heated	56	7.6	33.9	0.63	0.009	0.282	0.34	4.3	0.7	12.9	5.0
NYC-RR Control	45	7.6	5.3	0.58	0.037	0.371	0.17	4.3	4.2	24.8	3.1
NYC-RR Heated	60	7.6	33.7	0.66	0.013	0.304	0.34	4.1	0.8	12.1	5.4
NYC-KEN Control	40	7.3	7.4	0.46	0.067	0.141	0.25	5.5	3.0	21.9	3.3
NYC-KEN Heated	48	7.4	6.9	0.24	0.003	0.087	0.15	5.4	3.1	36.4	3.2
Average Δ (heated – control)	**+18.1**	**+0.2**	**+16.7**	**-0.18**	**-0.04**	**-0.20**	**+0.07**	**-0.3**	**-2.2**	**-4.9**	**+1.2**
Stdev Δ (heated – control)	**8.6**	**0.2**	**14.2**	**0.25**	**0.02**	**0.26**	**0.13**	**0.8**	**1.5**	**14.3**	**1.0**

NYC raw water leachates were chlorinated and analyzed for DBP formation to address changes to DBP precursors following heating (Table 5.6). As stated in the methods section, chlorine residuals were 1 mg/L (± 0.4). Following heating, TTHM and HAA5 formation were on average considerably lower compared to the control samples for three of the sites (TTHM mean Δ = -72.2 μg/L; HAA5 mean Δ = -40.9 μg/L). However, for NYC-EA the TTHM formation was higher for the heated leachate compared to the control sample (TTHM Δ = +29.3 μg/L). For HAA5s NYC-RR showed higher formation in the heated sample (HAA5 Δ = +18.1μg/L), whereas the other three heated samples showed lower HAA5s compared to respective controls. HAN4 formation of the heated leachates was also generally lower (mean Δ = -1.0 μg/L), with one exception (NYC-RR Δ = +1.8 μg/L). The different trends may be explained by the variability of the litter material that was leached in solution and may contain substantially different precursor material between sample collection sites. Alternatively, chloropicrin formation was consistently elevated following heating (mean Δ = +5.9 μg/L). Although samples were diluted to similar DOC concentrations, carbon normalized DBP yields allow for a better comparison of the DBP precursor reactivity of the control and heated samples (Table 5.6). Following heating, TTHM yields and HAA5 yields were similar to control samples or decreased. NYC-RR was the only sample that showed an increase in HAA5 yield (Δ = + 2.0 $\mu g/mg_C$), which was relatively small. HAN4 yields both increased and decreased, whereas chloropicrin yields of the NYC heated leachates were higher compared to control samples (mean Δ = +1.2 $\mu g/mg_C$).

Table 5.6
Raw water DBP formation and carbon normalized DBP yields for New York City control and heated samples. Samples are leached mixtures of soil and litter. Soil and litter samples were collected from four different sites within the NYC system.

New York City	Chlorine Dose (mg/L)	TTHM (μg/L)	TTHM Yield (μg/mg$_C$)	HAA5 (μg/L)	HAA5 Yield (μg/mg$_C$)	HAN4 (μg/L)	HAN4 Yield (μg/mg$_C$)	Chloropicrin (μg/L)	Chloropicrin Yield (μg/mg$_C$)
NYC-EA Control	7.7	130.4	29.3	161.2	36.2	5.3	1.2	4.5	1.0
NYC-EA Heated	7.5	159.7	30.2	123.7	23.4	3.5	0.7	10.5	2.0
NYC-NN Control	14.8	279.5	48.3	248.5	42.9	7.8	1.4	7.8	1.4
NYC-NN Heated	7.3	167.6	35.6	179.6	38.1	4.3	0.9	14.3	3.0
NYC-RR Control	8.1	161.7	37.1	166.5	38.2	4.2	1.0	4.1	0.9
NYC-RR Heated	7.3	137.3	29.9	184.6	40.2	6.0	1.3	10.0	2.2
NYC-KEN Control	9.0	316.5	54.5	237.9	40.9	6.9	1.2	5.5	0.9
NYC-KEN Heated	5.9	134.8	25.0	162.8	30.2	6.4	1.2	10.8	2.0
Average Δ (heated – control)	**NA**	**-72.2**	**-12.1**	**-40.9**	**-6.6**	**-1.0**	**-0.15**	**+5.9**	**+1.2**
Stdev Δ (heated – control)	**NA**	**93.3**	**12.9**	**42.6**	**6.7**	**2.2**	**0.4**	**0.5**	**0.3**

The NYC leachates were coagulated to address the effects of heating on conventional treatment processes. Preliminary jar tests were conducted to select the optimal alum dose for DOC removal, following methods described previously. Distinct differences in the coagulation response for the control and heated NYC samples were apparent (Figure 5.3). In most cases, turbidity of the heated leachates remained elevated compared to the control leachates for alum doses ranging from 20-60 mg/L, and particles re-stabilized at 120 mg/L. The control leachates showed typical DOC dose responses (Figure 5.3), with considerable removal occurring at alum doses between 20 and 30 mg/L. For the control samples the treated water DOC concentrations were generally <3.0 mg_C/L at doses ranging from 20-30 mg/L. However, the DOC of the heated leachates remained high >3.9 mg_C/L at doses of 30 mg/L, and showed only marginal removal (<20%). Although the heated leachate DOC concentration continued to decrease with increasing alum dose, the DOC remained near 3.0 mg_C/L following treatment with up to 90 mg/L alum. The raw water UV_{254} absorbance of the heated samples was generally higher than the control leachates, despite having similar DOC concentrations. As such, the UV_{254} of the heated samples remained higher than the control samples, following treatment at a range of alum doses.



DOC₀ = initial DOC concentration.

Figure 5.3 Alum dose-response for New York City control and heated leachates

Following preliminary jar tests, the NYC leachates were coagulated at a final dose, filtered, and chlorinated to characterize DBP levels likely to enter a distribution system. The required alum dose was on average 21.3 mg/L higher for the heated leachates compared to the control samples (Table 5.7). Even at a higher alum dose, the settled water turbidity levels were on average 2.0 ntu greater for heated samples, and treated water DOC concentrations were higher compared to the control leachates (mean Δ = +1.2 mg_C/L). NYC treated water heated samples showed higher $SUVA_{254}$ values (mean Δ = +2.2 L/mg-m), indicating considerable aromatic material still remained compared to control leachates. The poor DOC removal for the heated samples resulted in elevated TTHM, HAA5, and chloropicrin levels compared to the control samples, following coagulation treatment. HAN4 formation was similar for the control and heated samples following treatment. As expected from the poor treatability of the heated leachates, the percent decrease for all parameters was lower following conventional treatment. For the heated leachates the percent decrease of DOC (mean Δ = -29%), $SUVA_{254}$ (mean Δ = -35%), C-DBPs (TTHM mean Δ = -52%; HAA5 mean Δ = -40%), and N-DBPs (HAN mean Δ = -17%; chloropicrin mean Δ = -45%) were lower than the control samples (Table 5.8).

The lower DOC removal, and consequently poor removal of DBP precursors for the heated samples impacted the finished water DBP concentrations likely to enter a distribution system. As demonstrated in Figure 5.4, treated water DOC concentrations of the heated NYC samples ranged from 3.0-3.9 mg_C/L, whereas the control leachate DOC levels were consistently below 2.7 mg_C/L. Accordingly, following treatment all heated leachates exceeded TTHM and HAA5 MCLs, whereas all control samples were below MCLs. Chloropicrin concentrations of the heated samples (8.1-10.8 μg/L) were considerably higher than finished water NYC control leachates (0.8-2.3 μg/L). Following treatment HAN4 concentrations were similar for control and heated NYC samples.

Table 5.7
Treated water quality and DBP formation for New York City control and heated samples. Samples are leached mixtures of soil and litter. Soil and litter samples were collected from four different sites within the NYC system.

New York City	Alum Dose (mg/L)	Treated Turbidity (ntu)	Treated pH	Treated DOC (mg_C/L)	Treated $SUVA_{254}$ (L/mg-m)	Treated TTHM (mg/L)	Treated HAA5 (mg/L)	Treated HAN4 (mg/L)	Treated Chloropicrin (mg/L)
NYC-EA Control	25	0.3	6.8	2.0	1.7	30.9	27.8	6.3	1.7
NYC-EA Heated	45	2.5	6.8	3.9	3.4	109.9	70.2	3.9	8.1
NYC-NN Control	30	0.2	6.9	2.3	2.0	28.1	26.7	3.6	0.8
NYC-NN Heated	50	2.8	7.1	3.2	4.9	115.6	106.1	4.9	10.8
NYC-RR Control	35	0.2	7.0	2.0	1.8	44.2	34.3	3.4	2.3
NYC-RR Heated	60	2.7	6.7	3.0	4.7	94.9	102.5	6.2	9.0
NYC-KEN Control	30	0.1	6.9	2.7	1.5	69.2	57.2	6.8	2.3
NYC-KEN Heated	50	0.8	6.8	3.8	2.7	110.8	97.5	5.7	8.8
Average Δ (heated – control)	**+21.3**	**+2.0**	**-0.1**	**+1.2**	**+2.2**	**+64.7**	**+57.6**	**+0.1**	**+7.4**
Stdev Δ (heated – control)	**2.5**	**0.9**	**0.2**	**0.4**	**0.8**	**22.0**	**19.3**	**2.3**	**1.7**

Table 5.8
Percent removal following conventional treatment for New York City control and heated samples. Samples are leached mixtures of soil and litter. Soil and litter samples were collected from four different sites within the NYC system.

New York City	**% Decrease Turbidity**	**% Decrease DOC**	**% Decrease $SUVA_{254}$**	**% Decrease TTHM**	**% Decrease HAA5**	**%Decrease HAN4**	**% Decrease Chloropicrin**
NYC-EA Control	97	56	37	76	83	-20	63
NYC-EA Heated	87	24	5	31	43	-13	22
NYC-NN Control	97	60	41	90	89	54	90
NYC-NN Heated	92	25	2	31	41	-12	24
NYC-RR Control	96	52	41	73	79	19	45
NYC-RR Heated	92	27	12	31	45	-2	9
NYC-KEN Control	98	51	53	78	76	0.7	58
NYC-KEN Heated	88	28	14	18	40	12	18
Average Δ (heated – control)	**-7**	**-29**	**-35**	**-52**	**-40**	**-17**	**-45**
Stdev Δ (heated – control)	**3**	**5**	**5**	**9**	**6**	**36**	**14**

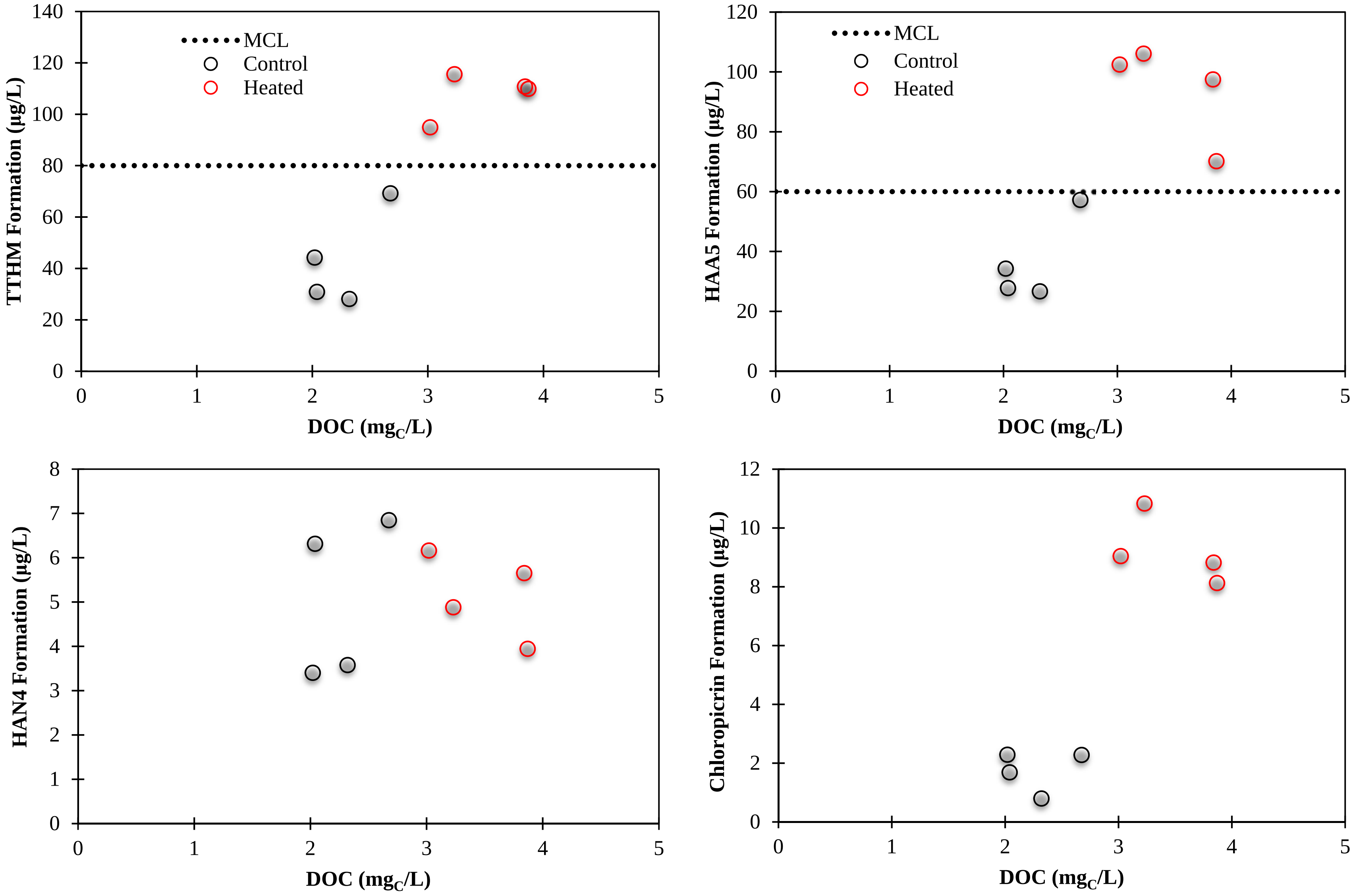


Figure 5.4 Treated water DOC and DBP formation for New York City control and heated leachates

WESTMINSTER TREATMENT PROCESS PERFORMANCE

WM control and heated samples were generated following the heating and leaching methods presented previously, and analyzed for raw water quality. As shown in Table 5.9, following heating the pH and alkalinity increased compared to the control samples (mean Δ = +0.19 and +8.2 $mgCaCO_3/L$). Turbidity levels, inorganic nitrogen, DON, and DOC concentrations are presented, but reflect the amount of material (soil and litter) leached, rather than the effects of heating. Therefore, where appropriate, the sample results were normalized by the concentration of DOC, or the total mass of litter and soil leached. Soil and litter samples were mixed together, leached, and diluted to a DOC concentration of 5.0 ± 1.0 mg_C/L for better comparison of the samples at a realistic DOC concentration. The DOC:DON ratios imply the heated leachates were enriched in organic nitrogen compared to control samples (mean Δ = -4.9 mg_C/m_N). DOC leached per gram of solid material was lower after heating (mean Δ = -1.4 mg_C/g). $SUVA_{254}$ was consistently higher for the WM heated samples compared to the control leachates (mean Δ = +1.2 L/mg_C-m).

Table 5.9
Raw water quality for Westminster control and heated samples. Samples are leached mixtures of soil and litter. Soil and litter samples were collected from four different sites within the WM system.

Westminster	Alkalinity (mgCaCO$_3$/L)	pH	Turbidity (ntu)	TDN (mg$_N$/L)	NO$_3$+NO$_2$ (mg$_N$/L)	NH$_4^+$ (mg$_N$/L)	DON (mg$_N$/L)	DOC (mg$_C$/L)	DOC Leached (mg$_C$/g)	DOC:DON (mg$_C$/mg$_N$)	SUVA$_{254}$ (L/mg-m)
WM35 Control	51	7.8	8.9	0.07	<0.001	0.033	NA	4.0	3.1	NA	2.3
WM35 Heated	55	7.8	21.8	0.38	0.029	0.121	0.23	5.6	1.2	24.3	4.3
WM40 Control	48	7.5	11.3	0.39	0.004	0.178	0.21	4.4	3.3	21.0	1.9
WM40 Heated	54	7.5	8.5	0.17	<0.001	0.023	0.15	5.5	2.0	37.0	2.5
WM49 Control	33	7.3	19	0.18	0.023	0.020	0.14	4.5	2.3	31.9	2.8
WM49 Heated	48	7.7	15.0	0.36	0.003	0.119	0.23	5.4	1.9	23.1	2.9
WM50 Control	37	7.4	10.1	0.10	<0.001	0.002	0.10	4.5	4.5	46.6	2.0
WM50 Heated	45	7.7	9.7	0.27	<0.001	0.049	0.22	5.5	2.5	24.7	3.1
Average Δ (heated – control)	**+8.2**	**+0.19**	**+1.4**	**+0.11**	**0.00**	**+0.02**	**+0.05**	**+1.0**	**-1.4**	**-4.9**	**+1.2**
Stdev Δ	**4.9**	**0.2**	**7.8**	**0.23**	**0.02**	**0.12**	**0.10**	**0.5**	**0.7**	**19.3**	**1.0**

WM raw leachates were chlorinated and analyzed for DBP formation to address changes to DBP precursors following heating (Table 5.10). On average DBP formation increased following heating (TTHM mean Δ = +11.8 μg/L; HAA5 mean Δ = +25.5 μg/L; HAN4 mean Δ = +1.2 μg/L; chloropicrin mean Δ = 8.0 μg/L). However considerable variability was observed and for several samples DBP formation was lower for the heated leachates. TTHM formation was lower for the heated leachates, WM49 and WM50, and HAA5 formation was lower for WM49 following heating. HAN4 formation decreased for two of the leachates (WM35 and WM50) following heating. Chloropicrin formation consistently increased for all heated leachates. Although samples were diluted to similar DOC concentrations, carbon normalized DBP yields allow for a better comparison of the DBP precursor reactivity of the control and heated samples (Table 5.10). Following heating, TTHM and HAA5 yields on average decreased slightly (TTHM yield mean Δ = -3.3 μg/mg$_C$; HAA5 yield mean Δ = -0.7 μg/mg$_C$), again variability following heating was observed. The WM40 heated leachates showed higher TTHM yield compared to the control, and WM35 and WM40 showed higher HAA5 yields compared to unheated samples. HAN4 yields for the heated and control samples varied, both increasing and decreasing following heating (mean Δ = +0.01 μg/mg$_C$), whereas chloropicrin yields of the WM heated leachates were consistently higher compared to control samples (mean Δ = +1.3 μg/mg$_C$).

Table 5.10
Raw water DBP formation and carbon normalized DBP yields for Westminster control and heated samples. Samples are leached mixtures of soil and litter. Soil and litter samples were collected from four different sites within the WM system.

Westminster	Chlorine Dose (mg/L)	TTHM (μg/L)	TTHM Yield (μg/mg$_C$)	HAA5 (μg/L)	HAA5 Yield (μg/mg$_C$)	HAN4 (μg/L)	HAN4 Yield (μg/mg$_C$)	Chloropicrin (μg/L)	Chloropicrin Yield (μg/mg$_C$)
WM35 Control	4.3	121.3	30.1	118.7	29.4	7.1	1.8	2.3	0.6
WM35 Heated	6.5	157.5	28.3	189.7	34.1	6.0	1.1	8.8	1.6
WM40 Control	5.6	149.8	25.2	131.7	22.1	6.2	1.0	5.1	0.9
WM40 Heated	5.5	220.2	39.7	196.1	35.4	10.4	1.9	15.5	2.8
WM49 Control	6.1	226.0	49.9	237.6	52.5	7.4	1.6	9.3	2.0
WM49 Heated	6.6	194.4	36.7	192.7	36.4	9.2	1.7	18.9	3.6
WM50 Control	4.8	175.9	39.4	150.3	33.7	3.9	0.9	4.0	0.9
WM50 Heated	6.4	148.0	26.7	161.3	29.1	3.7	0.7	9.4	1.7
Average Δ (heated – control)	**NA**	**+11.8**	**-3.3**	**+25.4**	**-0.7**	**+1.2**	**+0.01**	**+8.0**	**+1.3**
Stdev Δ	**NA**	**49.9**	**13.0**	**54.0**	**12.6**	**2.3**	**0.64**	**2.4**	**0.5**

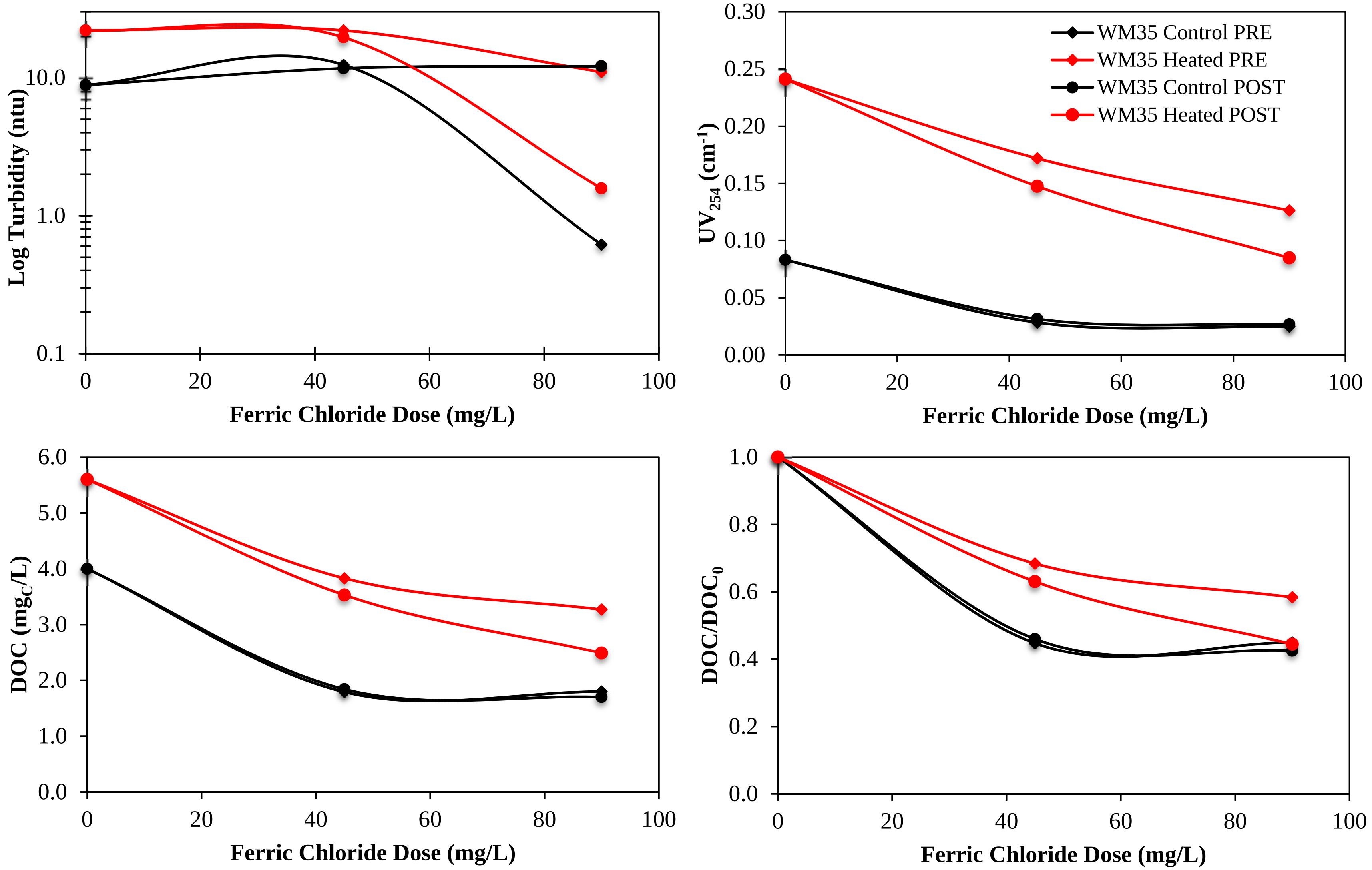
WM control and heated leachates were coagulated with ferric chloride at two doses (45 and 90 mg/L) and pH adjusted with caustic either pre- or post-mixing and settling. Dose response curves for the four sample collection sites (WM 35, 40, 49, and 50) are presented in Figures 5.5-5.8. No clear trends for turbidity were observed. The settling time was limited to 1 minute and as a consequence turbidity levels were high for both WM control and heated samples. Control and heated samples showed similar DOC dose-response curves, however the WM heated samples had higher raw water DOC concentrations (mean Δ = +1.0 mg/L) and also showed higher treated water DOC concentrations at ferric doses of 45 and 90 mg/L, and for both pH adjustment conditions. UV$_{254}$ showed a similar dose-response, with heated samples having considerably higher raw and finished water UV$_{254}$.

Control and heated leachate finished water quality parameters under all treatment conditions are reported in Table 5.11. Minimal differences were observed for different pH adjustment conditions. Treated water DOC concentrations for control samples ranged from 1.3–1.9 mg$_C$/L (ferric dose = 45 mg/L) and 1.5–1.8 mg$_C$/L (ferric dose = 90 mg/L), suggesting that the higher ferric chloride dose did not provide additional DOC removal. For the heated leachates treated water DOC concentrations ranged from 3.3-3.8 mg$_C$/L (ferric = 45 mg/L) and 2.5-3.3 mg$_C$/L (ferric dose = 90 mg/L), with the higher ferric dose providing better DOC removal. Similar trends were observed for finished water SUVA$_{254}$. The control samples showed SUVA$_{254}$ values of 1.5-2.4 L/mg$_C$-m (ferric dose = 45 mg/L) and 1.4 –2.4 L/mg$_C$-m (ferric dose = 90 mg/L). Again, these results indicate the higher ferric dose did not improve the finished water quality of the control samples. WM heated leachate SUVA$_{254}$ values were 2.4- 4.5 L/mg$_C$-m (ferric dose = 45 mg/L) and 2.4 – 3.9 (ferric dose = 90 mg/L), demonstrating improved finished water quality at the higher ferric chloride dose for heated leachates.

Overall, the heated leachates showed a poorer response to ferric chloride coagulation than control samples. On average the treated water turbidity was 1.2 ntu higher than WM control samples, DOC was 1.6 mg$_C$/L higher and SUVA$_{254}$ was 1.2 L/mg-m higher (Table 5.11). Treated water C-DBP formation followed similar trends for the control and heated leachates (Table 5.11) with consistently higher DBP levels following heating (TTHM mean Δ = +49.9 µg/L; HAA5 mean Δ = +56.1 µg/L). N-DBP formation following ferric chloride treatment was also elevated for the heated samples (HAN4 mean Δ = +2.0 µg/L; chloropicrin mean Δ = +6.4 µg/L).

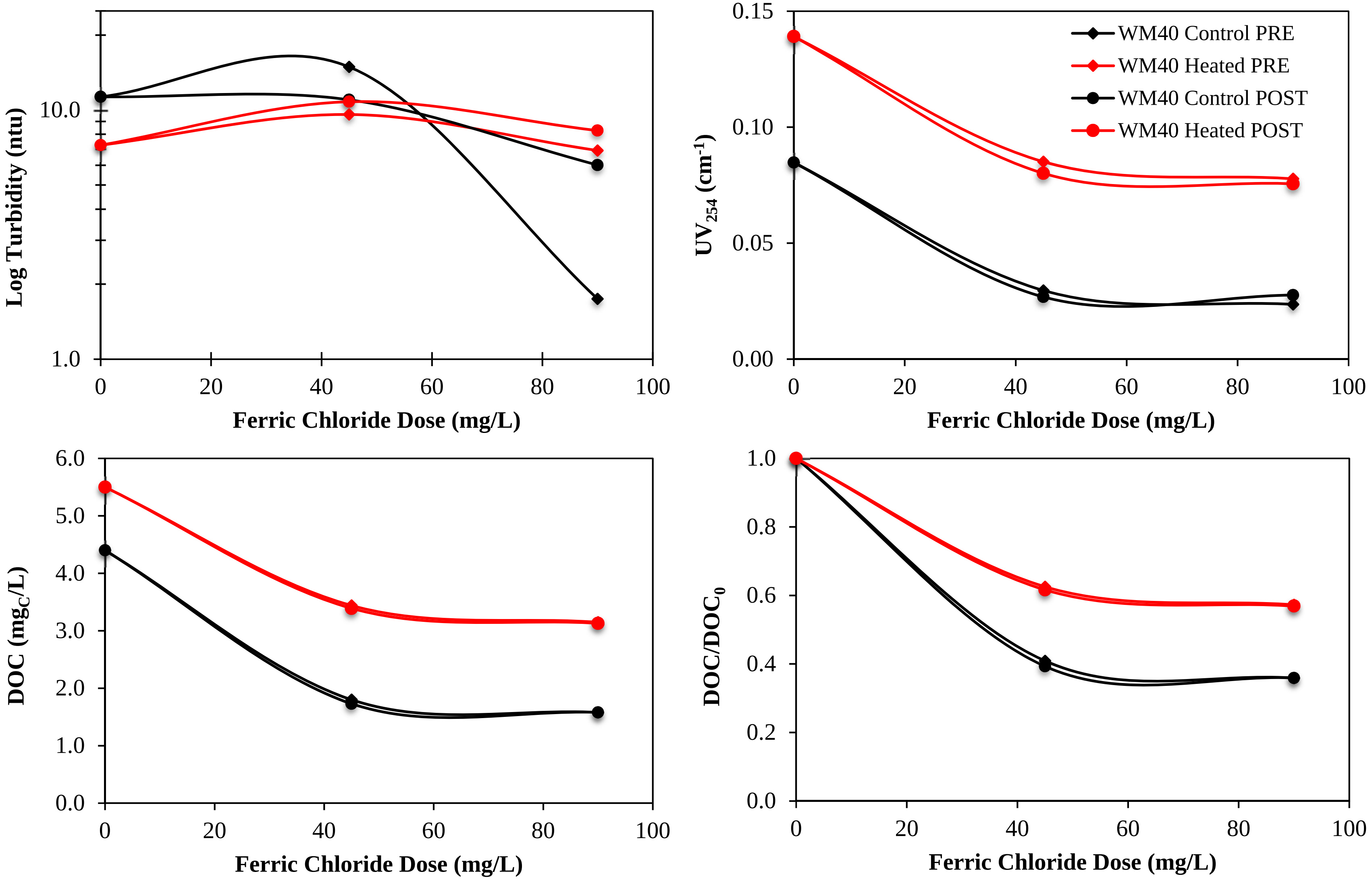
As expected from the poorer treatability of the heated leachates, the percent decrease for all parameters was lower following treatment compared to WM control samples. For the heated leachates the percent decrease of DOC (mean Δ = -21%), SUVA$_{254}$ (mean Δ = -15%), C-DBPs (TTHM mean Δ = -27%; HAA5 mean Δ = -28%), and N-DBPs (HAN mean Δ = -30%; chloropicrin mean Δ = -21%) were lower than the control samples (Table 5.12).

The lower DOC removal, and consequently poor removal of DBP precursors for the heated samples impacted the finished water DBP concentrations. As demonstrated in Figure 5.9, treated water DOC concentrations of the heated WM samples ranged from 2.5-3.8 mg$_C$/L, whereas the control leachate DOC levels were consistently below 2.0 mg$_C$/L. Accordingly, following treatment all except one of the heated leachates exceeded TTHM and HAA5 MCLs, whereas most control samples were below MCLs. HAN4 and chloropicrin concentrations of the heated samples were also higher than finished water WM control leachates (Figure 5.9).



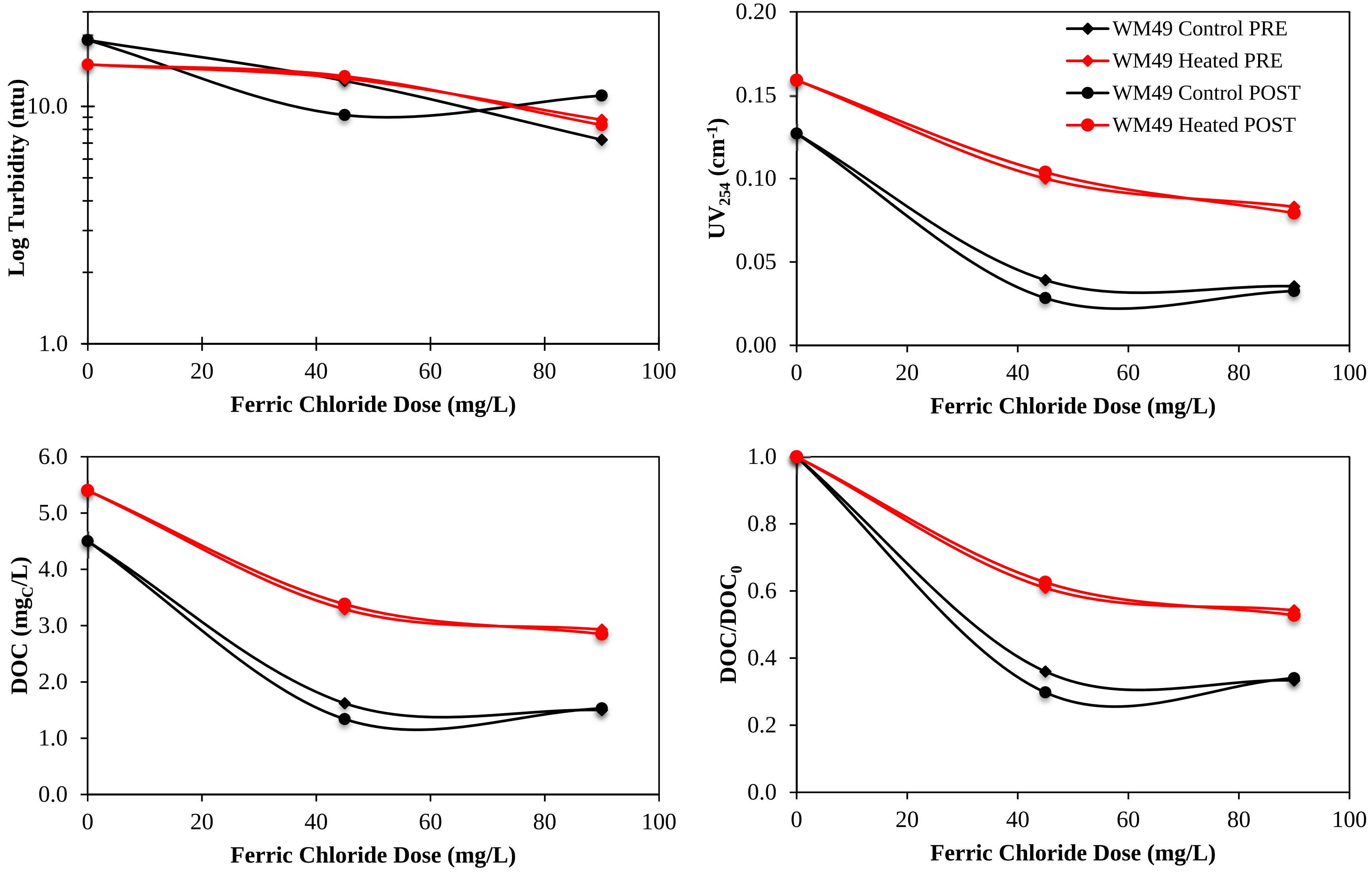
Samples were pH adjusted prior to mixing and settling (PRE) or after mixing and settling (POST). DOC_0 = initial DOC concentration of raw leachate.

Figure 5.5 Dose response curves for WM35 control and heated samples



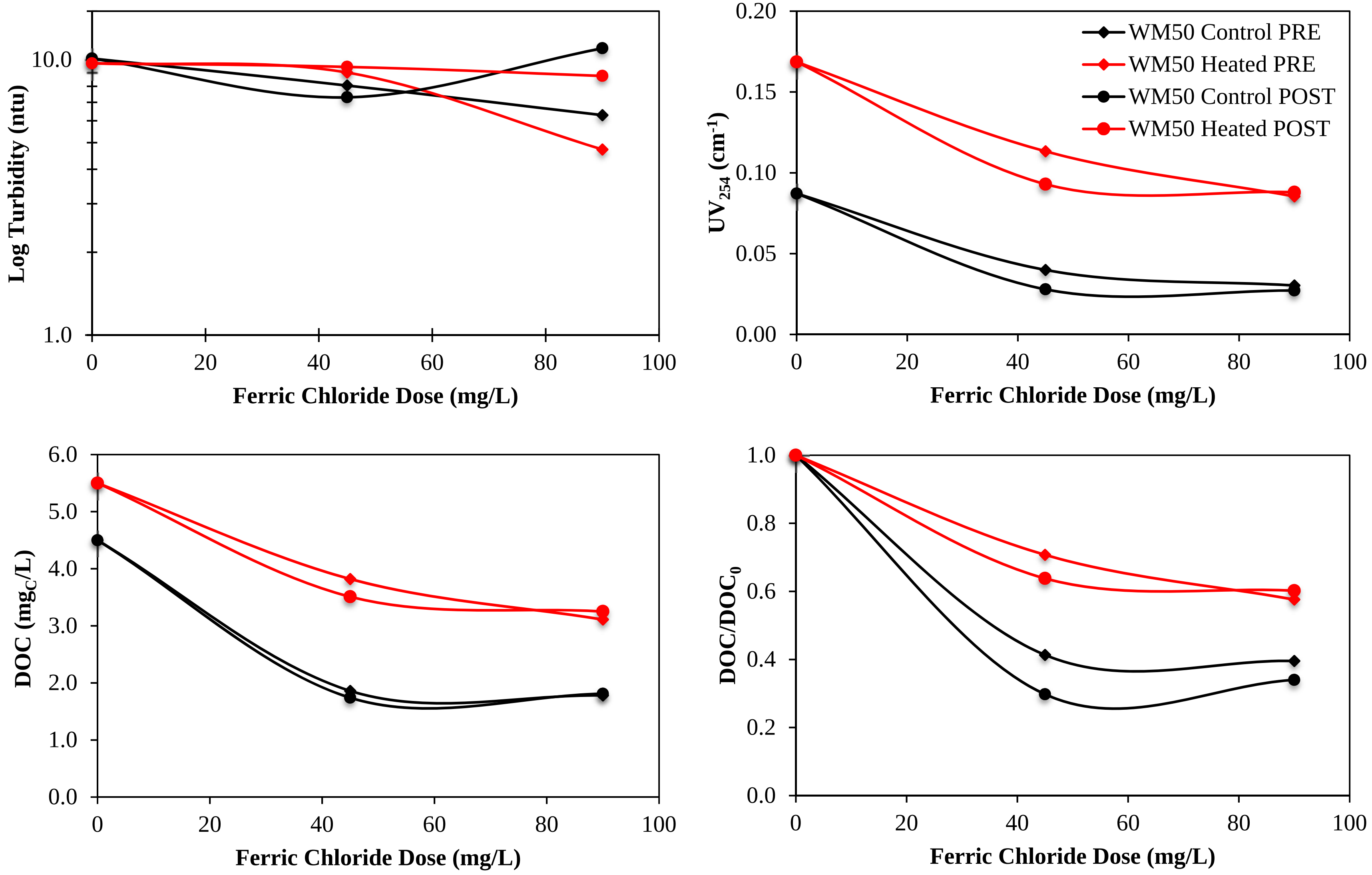
Samples were pH adjusted prior to mixing and settling (PRE) or after mixing and settling (POST). DOC_0 = initial DOC concentration of raw leachate.

Figure 5.6 Dose response curves for WM40 control and heated samples



Samples were pH adjusted prior to mixing and settling (PRE) or after mixing and settling (POST). DOC^0 = initial DOC concentration of raw leachate.

Figure 5.7 Dose response curves for WM49 control and heated samples



Samples were pH adjusted prior to mixing and settling (PRE) or after mixing and settling (POST). DOC_0 = initial DOC concentration of raw leachate.

Figure 5.8 Dose response curves for WM50 control and heated samples

Table 5.11

Treated water quality and DBP formation for Westminster control and heated samples. Samples are leached mixtures of soil and litter. Soil and litter samples were collected from four different sites within the WM system.

Westminster	Ferric Chloride Dose	pH Adjustment	Treated Turbidity (ntu)	Treated DOC (mg$_C$/L)	Treated $SUVA_{254}$ (L/mg$_C$-m)	Treated TTHM (µg/L)	Treated HAA5 (µg/L)	Treated HAN4 (µg/L)	Treated Chloropicrin (µg/L)
WM35 Control	45	PRE	12.4	1.8	1.6	40.3	24.5	3.8	1.7
	45	POST	11.7	1.8	1.7	38.4	26.2	3.9	1.1
	90	PRE	0.6	1.8	1.4	33.7	21.6	2.7	1.3
	90	POST	12.1	1.7	1.6	29.1	17.5	4.3	1.1
WM35 Heated	45	PRE	22	3.8	4.5	106.6	112.4	5.2	7.3
	45	POST	19.6	3.5	4.2	99.3	98.2	5.1	5.1
	90	PRE	11	3.3	3.9	88.2	81.5	5.2	4.9
	90	POST	1.58	2.5	3.4	61.9	55.8	4.7	3.9
WM40 Control	45	PRE	14.9	1.8	1.6	42.7	30.6	4.0	1.7
	45	POST	11.0	1.7	1.5	22.6	26.5	3.3	2.1
	90	PRE	1.7	1.6	1.5	35.8	20.0	3.1	1.6
	90	POST	6.0	1.6	1.7	36.2	27.4	3.9	1.2
WM40 Heated	45	PRE	9.6	3.4	2.5	114.0	70.2	5.08	8.22
	45	POST	10.8	3.4	2.4	100.9	67.7	4.93	8.91
	90	PRE	6.9	3.2	2.5	88.2	108.1	4.81	8.91
	90	POST	8.3	3.1	2.4	97.8	64.8	5.17	7.83
WM49 Control	45	PRE	12.8	1.6	2.4	84.0	49.9	4.5	5.1
	45	POST	9.19	1.3	2.1	59.3	29.1	4.5	4.3
	90	PRE	7.23	1.5	2.4	66.5	42.9	3.9	4.0
	90	POST	11.1	1.5	2.1	63.5	40.6	3.9	3.9
WM49 Heated	45	PRE	13.1	3.3	3.0	138.9	113.8	6.5	15.6
	45	POST	13.4	3.4	3.1	138.6	108.5	6.5	13.0
	90	PRE	8.8	2.9	2.8	115.1	100.6	7.1	13.4
	90	POST	8.3	2.9	2.8	122.1	81.8	5.1	11.9
WM50 Control	45	PRE	8.1	1.9	2.1	75.0	40.5	2.5	1.7
	45	POST	7.3	1.7	1.6	60.0	35.3	2.4	1.6
	90	PRE	6.3	1.8	1.7	66.3	32.7	2.5	1.5
	90	POST	11.0	1.8	1.5	93.0	28.9	2.1	1.5
WM50 Heated	45	PRE	9.0	3.8	3.0	107.5	104.1	5.0	6.1
	45	POST	9.4	3.5	2.6	93.9	85.7	5.7	7.4
	90	PRE	4.7	3.1	2.7	84.7	68.5	4.8	7.6
	90	POST	8.7	3.3	2.7	86.8	70.5	6.1	7.2
Average Δ (heated – control)			**+1.4**	**+1.6**	**+1.2**	**+49.9**	**+56.1**	**+2.0**	**+6.4**
Stdev Δ			**5.4**	**0.3**	**0.8**	**22.8**	**18.3**	**1.0**	**2.1**

Table 5.12

Percent removals following ferric chloride treatment for Westminster control and heated samples. Samples are leached mixtures of soil and litter. Soil and litter samples were collected from four different sites within the WM system.

Westminster	Ferric Chloride Dose (mg/L)	pH Adjustment	% Decrease Turbidity	% Decrease DOC	% Decrease SUVA$_{254}$	% Decrease TTHM	% Decrease HAA5	%Decrease HAN4	% Decrease Chloropicrin
WM35 Control	45	PRE	-39	55	31	67	79	47	24
	45	POST	-31	54	26	68	78	45	51
	90	PRE	93	55	40	72	82	62	44
	90	POST	-36	58	32	76	85	40	53
WM35 Heated	45	PRE	-1	32	-5	32	41	12	17
	45	POST	10	37	2	37	48	14	42
	90	PRE	50	42	10	44	57	14	44
	90	POST	93	56	20	61	71	22	55
WM40 Control	45	PRE	-32	59	15	72	77	35	66
	45	POST	3	61	20	85	80	46	59
	90	PRE	85	64	23	76	85	50	69
	90	POST	47	64	10	76	79	37	77
WM40 Heated	45	PRE	-13	38	2	48	64	51	47
	45	POST	-28	38	6	54	65	53	43
	90	PRE	19	43	2	60	45	54	43
	90	POST	2	43	4	56	67	50	49
WM49 Control	45	PRE	33	64	14	63	79	40	45
	45	POST	52	70	25	74	88	40	54
	90	PRE	62	67	16	71	82	48	56
	90	POST	42	66	24	72	83	47	58
WM49 Heated	45	PRE	13	39	-3	29	41	29	17
	45	POST	11	37	-4	29	44	29	31
	90	PRE	42	46	4	41	48	23	29
	90	POST	45	47	6	37	58	45	37
WM50 Control	45	PRE	20	59	-7	57	73	36	57
	45	POST	28	61	20	66	76	38	59
	90	PRE	38	60	15	62	78	36	62
	90	POST	-9	60	25	47	81	45	62
WM50 Heated	45	PRE	7	30	4	27	35	-34	35
	45	POST	3	36	14	37	47	-52	21
	90	PRE	51	43	11	43	58	-28	19
	90	POST	10	41	12	41	56	-64	23
Average Δ (heated – control)			**-3**	**-21**	**-15**	**-27**	**-28**	**-30**	**-21**
Stdev Δ			**47**	**7**	**12**	**10**	**11**	**37**	**13**

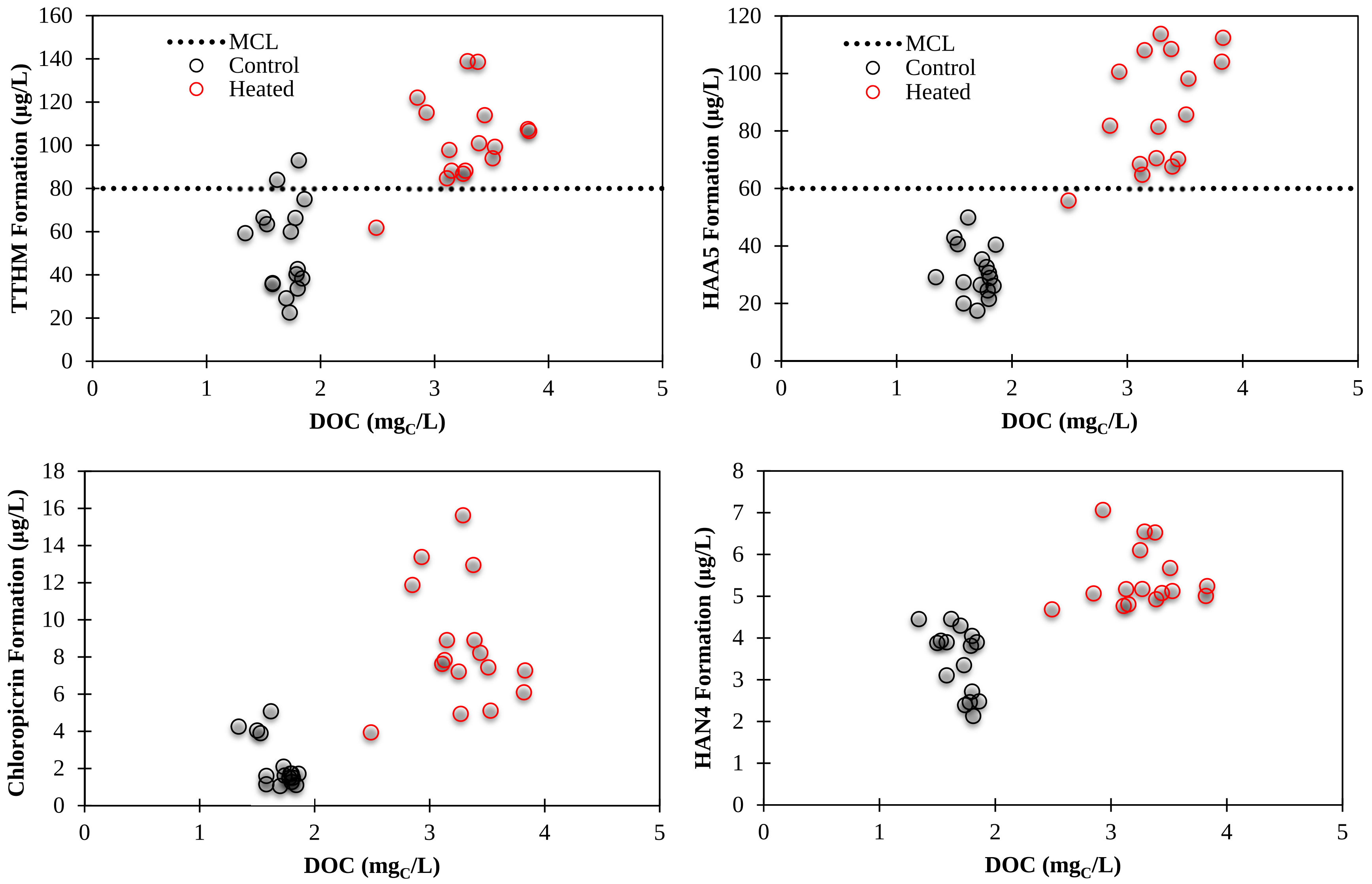


Figure 5.9 Treated water DOC and DBP formation for Westminster control and heated samples

SAN FRANCISCO TREATMENT PROCESS PERFORMANCE

San Francisco (SF) control and heated samples were generated following the heating and leaching methods presented above, and analyzed for water quality. Note that the SF-HH (Hetch Hetchy) sample was collected in 2015 from an area burned in the Rim Fire (2013) and was not heated in the laboratory or included in the calculation of differences between heated and control samples (Δ). However, the soil sample was collected two years after the fire and may have been subject to pre-leaching due to rainstorms. As shown in Table 5.13, following heating of the SF-WB sample, the pH and alkalinity slightly increased compared to the control sample, whereas the SF-EB sample showed minimal change in pH and alkalinity. Turbidity levels, inorganic nitrogen, DON, and DOC concentrations are presented, but reflect the amount of soil leached, rather than the effects of heating. Therefore, where appropriate, the sample results were normalized by the concentration of DOC, or the mass of soil leached. Due to the lack of SF litter samples (only soils were collected), DOC concentrations of the leachates were lower than leachates for the other utilities (DW, NYC, WM), and only soils were used for heating and leaching experiments. The DOC leached per gram of SF soil was higher after heating for EB and WB samples, which differs from the trend observed for the three other watersheds when soils and litter were composited and leached, suggesting different behavior between soils and litter samples upon heating. The SF-HH sample leached the lowest DOC per gram of soil, compared to all other SF samples. The DOC:DON ratios for the EB and WB samples imply the heated leachates were enriched in organic nitrogen compared to respective control samples. The DOC:DON ratio for SF-HH was within the range of the SF-EB and SF-WB heated and SF control leachates. $SUVA_{254}$ was consistently higher for SF-EB and SF-WB heated samples compared to the control leachates. For SF-HH, the $SUVA_{254}$ was between the values for the two SF heated leachates (EB and WB).

Table 5.13

Raw water quality for San Francisco control and heated soil samples (leachates). Soils were collected from three different sites within the SF system. SF-HH was impacted by the Rim fire and not heated in the laboratory.

San Francisco	Alkalinity ($mgCaCO_3/L$)	pH	Turbidity (ntu)	TDN (mg_N/L)	NO_3+NO_2 (mg_N/L)	NH_4^+ (mg_N/L)	DON (mg_N/L)	DOC (mg_C/L)	DOC Leached (mg_C/g)	DOC:DON (mg_C/mg_N)	$SUVA_{254}$ (L/mg_C-m)
SF-EB Control	46	7.7	17.3	0.51	0.357	0.034	0.12	2.8	0.4	23.0	2.4
SF-EB Heated	46	7.6	1.3	0.36	0.001	0.189	0.17	3.0	3.2	17.9	3.2
SF-WB Control	46	7.6	5.7	0.27	0.013	0.119	0.13	2.5	0.8	18.3	3.1
SF-WB Heated	54	7.8	3.3	0.62	0.003	0.350	0.27	2.9	1.3	11.0	4.5
SF-HH	58	7.6	98.2	0.14	0.001	0.022	0.11	2.2	0.1	18.7	3.7

SF raw leachates were chloraminated and analyzed for DBP formation to address changes to DBP precursors following heating (Table 5.14). Following heating DBP formation generally increased for EB and WB samples. DBP formation for the SF-HH leachate was relatively low, which may be attributed to pre-leaching of the soil organic matter during rainfall, prior to sample collection. However, the HAN4 formation of the SF-HH leachate was higher than both the control and heated samples for the other SF sites. Although the leachates were diluted to similar DOC concentrations, carbon normalized DBP yields allow for a better comparison of the DBP precursor reactivity of the control and heated samples (Table 5.14). Following heating, TTHM and HAA5 yields increased for the EB and WB leachates. For SF-HH, TTHM and HAA5 yields were within the range of the control leachates for SF-EB and SF-WB (Table 5.14), and were lower than the heated leachates. HAN4 yields showed minimal differences for the heated and control samples, increasing slightly for SF-EB and decreasing slightly for SF-WB, while SF-HH showed the highest HAN4 yield (0.9 $\mu g/mg_C$). The chloropicrin yield of the SF-EB heated leachate was higher than the control sample, while SF-WB control was below the detection limit for chloropicrin. The chloropicrin yield for SF-HH was also low (chloropicrin yield = 0.1 $\mu g/mg_C$).

Table 5.14
Raw water DBP formation and carbon normalized DBP yields for San Francisco control and heated soil samples (leachates). Soils were collected from three different sites within the SF system. SF-HH was impacted by the Rim fire and not heated in the laboratory.

San Francisco	TTHM (µg/L)	TTHM Yield (µg/mg$_C$)	HAA5 (µg/L)	HAA5 Yield (µg/mg$_C$)	HAN4 (µg/L)	HAN4 Yield (µg/mg$_C$)	Chloropicrin (µg/L)	Chloropicrin Yield (µg/mg$_C$)
SF-EB Control	5.2	1.9	5.9	2.1	1.1	0.4	1.1	0.4
SF-EB Heated	22.4	7.5	26.3	8.8	1.5	0.5	4.4	1.5
SF-WB Control	12.4	5.0	15.7	6.4	1.4	0.6	<DL	NA
SF-WB Heated	16.3	5.5	31.9	10.9	1.4	0.5	3.3	1.1
SF-HH	6.5	3.0	8.1	3.8	1.8	0.9	0.2	0.1

The SF leachates were treated following the conditions described previously. The samples were filtered and chloraminated to characterize the DBP levels likely to enter a distribution system. The settled water turbidity levels for SF-EB and SF-WB were lower for heated samples compared to the control samples from respective sites (Table 5.15). Following heating, the treated water DOC concentrations for EB and WB samples remained higher compared to the control leachates. For SF-HH, the treated water DOC was the lowest (1.7 mg_C/L; Table 5.16: 22% removal) compared to SF-EB and SF-WB samples, however the raw water DOC was also the lowest (2.2 mg_C/L). The SF-WB heated treated water sample had a slightly higher $SUVA_{254}$ value than the respective control sample, and the SF-EB heated leachate also showed a greater treated water $SUVA_{254}$ than the control. For SF-HH the treated water $SUVA_{254}$ was higher than all other SF samples, suggesting this soil was enriched in more aromatic soluble organic matter. The poorer DOC removal for the EB and WB heated samples resulted in elevated TTHM, HAA5, HAN4, and chloropicrin levels compared to the control samples following treatment. Treated water C-DBP formation for SF-HH was relatively high compared to the other SF samples (second highest overall). SF-HH treated water HAN4 formation was the lowest, while chloropicrin formation was higher than most SF samples. Turbidity and $SUVA_{254}$ consistently decreased following treatment for the control and heated leachates, however DOC and DBP formation showed considerable variability, poor removal, and often increased following treatment (Table 5.16). For instance, DOC decreased for the EB and WB control samples, but increased for the EB and WB heated samples, after treatment. For all samples except SF-WB control, TTHM formation increased following treatment. For the SF-EB samples and SF-HH, HAA5 formation increased after treatment. For all SF-EB and SF-WB samples HAN4 formation increased following treatment. For SF-HH HAN4 formation decreased after treatment (41%), while chloropicrin formation increased substantially. Overall, a wide range of treatment responses and variability in DBP formation were observed for SF samples. This observation is likely attributed to pre-chlorination or the use of an organic polymer, both of which were not used for DW, NYC, and WM treatment tests (treatment with aluminum sulfate or ferric chloride).

Despite the general increase in DBP formation following treatment of the SF control and heated leachates, DBP MCLs were not exceeded for any treated water samples (Figure 5.10), although SF-EB heated was very close to the MCLs. The SF-EB heated sample formed high HAN4 (2.7 μg/L) and chloropicrin (14.1 μg/L) compared to the control samples, SF-HH, and SF-WB heated.

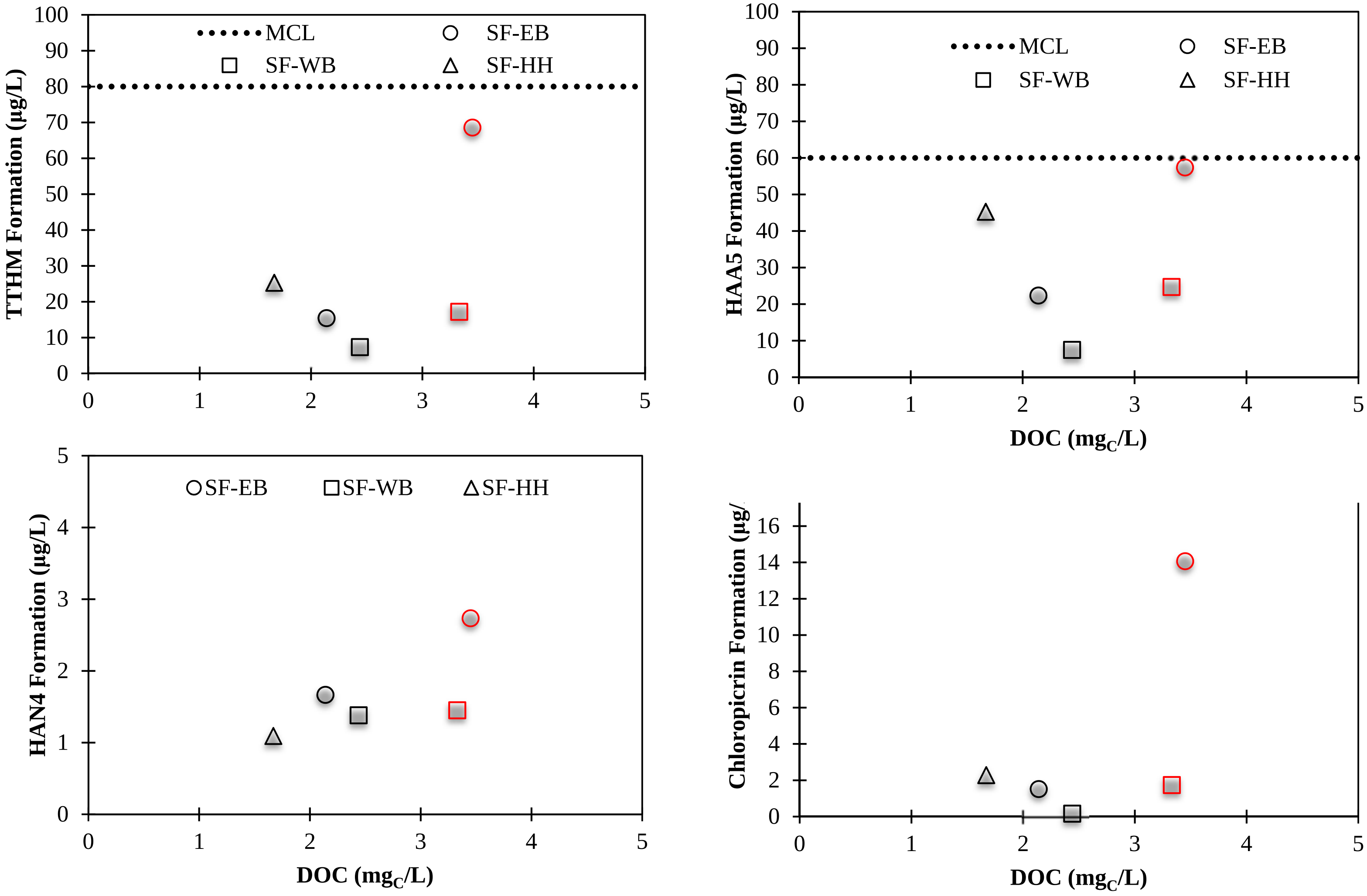
Table 5.15

Treated water quality and DBP formation for San Francisco control and heated soil samples (leachates). Soils were collected from three different sites within the SF system. SF-HH was impacted by the Rim fire and not heated in the laboratory.

San Francisco	Treated Turbidity (ntu)	Treated pH	Treated DOC (mg_C/L)	Treated $SUVA_{254}$ (L/mg_C-m)	Treated TTHM (μg/L)	Treated HAA5 (μg/L)	Treated HAN4 (μg/L)	Treated Chloropicrin (μg/L)
SF-EB Control	4.8	6.9	2.1	1.3	15.4	22.4	1.7	1.5
SF-EB Heated	0.2	6.8	3.5	1.9	68.6	57.4	2.7	14.1
SF-WB Control	3.5	7.5	2.4	2.3	7.3	7.5	1.4	0.2
SF-WB Heated	2.6	7.6	3.3	2.4	17.2	24.7	1.5	1.7
SF-HH	2.3	7.4	1.7	2.6	25.3	45.3	1.1	2.3

Table 5.16
Percent removal following treatment for San Francisco control and heated soil samples. Soils were collected from three different sites within the SF system. SF-HH was impacted by the Rim fire and not heated in the laboratory.

San Francisco	% Decrease Turbidity	% Decrease DOC	% Decrease $SUVA_{254}$	% Decrease TTHM	% Decrease HAA5	%Decrease HAN4	% Decrease Chloropicrin
SF-EB Control	72	23	44	-194	-280	-50	-32
SF-EB Heated	82	-15	41	-206	-118	-85	-217
SF-WB Control	39	1	24	41	52	-1	NA
SF-WB Heated	21	-13	46	-5	23	-1	48
SF-HH	98	22	31	-286	-458	41	-1263



Red indicates the samples were heated prior to leaching.

Figure 5.10 Treated water DOC and DBP formation for San Francisco control and heated soil samples

DISCUSSION

Generally, the trends observed for the four utilities were consistent and aid in understanding the effects of heating on water soluble compounds and the associated treatment challenges. The marginal increase in pH and alkalinity observed for the laboratory heated samples may be attributed to the denaturing of organic acids upon heating, with residual alkaline components remaining (Ulery and Graham 1993). The decrease in DOC leached per gram of material for the heated leachates is consistent with other work indicating partial combustion of soluble organic carbon compounds at 225°C, whereas organic nitrogen has been shown to volatilize at higher temperatures (Hogue and Inglett 2012), supporting the observed enrichment of DON relative to DOC, following heating. SF-EB and SF-WB leachates did not follow the same trend, and the DOC leached per gram of soil increased after heating. SF-HH was burned in the Rim fire, was not subject to laboratory heating, and the DOC leached per gram of soil was very low. SF litter samples (not available) were not leached with the soils, and perhaps different organic precursor materials of soils and litter may help explain the difference. Consistently higher $SUVA_{254}$ for the heated samples indicates enhanced aromaticity of soluble compounds upon heating, supported by soil organic matter studies (González-Pérez et al. 2004, Knicker 2007). Iron concentrations of the heated leachates were low (<0.005 mg/L) and did not significantly interfere with absorbance measurements.

Generally, for raw water C-DBP yields, the changes associated with heating were inconsistent, both increasing and decreasing compared to paired control samples. These findings suggest variable alterations in TTHM and HAA5 precursors at 225 °C. Bromide concentrations were low (< 0.003 mg/L), and primarily chlorinated DBP species were formed. An overall decrease in C-DBP reactivity was observed by others (Wang et al. 2015a), and suggests destruction of TTHM and HAA5 water-soluble precursors during heating, despite the observed increase in $SUVA_{254}$, which generally correlates with C-DBP precursor reactivity (Archer and Singer 2006, White et al. 1997). It should be noted, the similar or on average lower TTHM, HAA5, and HAN4 precursor reactivity of the heated leachates compared to the control samples may not be representative of the precursor load a water treatment facility might receive in their influent supply, post-wildfire. Enhanced erosion of terrestrial DOM following wildfire can significantly increase DOC levels and DBP formation, as observed in field-based studies (Hohner et al. 2016, Writer et al. 2014).

The increase in chloropicrin precursor reactivity following heating may be associated with the enrichment of DON relative to DOC, or elevated inorganic nitrogen levels. Studies have identified 2- and 3-nitrophenol (Merlet et al. 1985, Thibaud et al. 1987) as well as glycine (Mitch et al. 2009) as chloropicrin precursors. While the DOC:DON ratio decreased upon heating, and chloropicrin formation and precursor reactivity per unit carbon increased, HAN4 precursors did not appear altered by heating. Previous studies have associated elevated HAN4 reactivity with wildfire (Hohner et al. 2016, Wang et al. 2015b). Enhanced post-fire erosion and mobilization of HAN4 precursors may have been more significant than heat-induced alterations to terrestrial organic matter. Perhaps water-soluble chloropicrin precursors formed at 225 °C, whereas HAN4 precursors may form at a different temperature, or under different conditions existing in the natural environment. A lab-controlled study showed different trends for HAN4 reactivity based on oxygen conditions during heating (Wang et al. 2015a).

Previous work by Cawley et al. showed a consistent increase in HAN4 yields of heated soils (225°C) relative to unheated soils (Cawley et al. 2017), however litter materials were not included in the experiments, possibly explaining the disconnect with the results of this study. Further, the Cawley et al. study suggests a shift upon heating in the composition of soils, towards more N-containing compounds indicated by high resolution mass spectroscopy. Alternatively for the current study, heated litter, the dominant source of DOM in the mixed leachates, may not contribute substantial HAN4 precursors relative to unheated litter, potentially implying heating alters HAN4 precursors in soils and litter differently. Other research showed for plant biomass, the maximum extractable DON occurred from heating at 350°C (Hogue and Inglett 2012), higher than the temperature used for the treatment tests.

The adverse effect of heating on the overall treatability of the leachates might be explained by a lower-molecular weight DOM composition, as smaller compounds are generally less amenable to removal by coagulation (Quang et al. 2015). Further, although small, the increase in raw water pH and alkalinity of the heated leachates may have decreased the efficacy of coagulation. pH adjustment was not optimized in this study, but may have improved process performance of the heated leachates by lowering the pH for more effective turbidity or DOM removal. The raw water turbidity levels of the heated leachates were low relative to the range of observed post-fire particle loads (e.g., > 200 ntu) (Hohner et al. 2016, Writer et al. 2014). The changes heating imparted on particle characteristics (e.g., size, surface charge, density, adsorptive properties) were not explored, but may have also influenced the coagulation response. As expected, the lower DOC removal and consequently poor removal of DBP precursors for the heated samples significantly impacted the finished water DBP concentrations likely to enter a distribution system. Heating appeared to alter the DOM composition, resulting in poor removal of C-DBP precursors and negative implications for finished water quality, despite the generally lower raw water TTHM and HAA5 yields.

The negligible removal of HAN4 precursors by coagulation for the heated leachates, and better, although low, removal for the control leachates is likely attributed to low molecular weight and low anionic charge of N-DBP precursors, resulting in relatively ineffective removal (Bond et al. 2012). Overall, the results suggest chloropicrin precursors are more amenable to coagulation than HAN4 precursors, consistent with previous work (Hohner et al. 2016). Nitrosamines were not considered in this study, however an increase in reactivity was observed from chloramination of wildfire-affected detritus extracts (Wang et al. 2015b).

CONCLUSIONS

Controlled laboratory heating at 225°C of soil and litter samples from varying geographic regions resulted in clear and measurable alterations to the soluble DOM character, indicated by increased $SUVA_{254}$. Despite the higher $SUVA_{254}$, a change in DOM quality, such as shifts towards lower molecular weight compounds, may have adversely affected coagulation treatment, resulting in elevated settled water turbidity and minimal DOC removal. Further, finished water quality was negatively influenced, including the exceedance of DBP MCLs and high chloropicrin concentrations. Heat-induced changes to particle size and characteristics were not explored in this study, but may have negatively affected coagulation processes possibly due to the presence of finer materials.

While the results from this study provide insight into the potential alterations that may occur during a wildfire, important considerations must be noted. Natural fires will likely result in patchiness across the burned landscape and a heterogeneous mixture of post-fire residues, unburned soils, and vegetation. Similarly, wildfire specific factors such as oxygen availability, temperature, and fuel moisture may result in different alterations to terrestrial sources of DOM than observed in this laboratory study. Further, environmental transformations (e.g., photolysis, biodegradation) of soils and litter are possible, but were not considered in the laboratory experiments. Temporal variation in wildfire-burned detritus characteristics have been observed following cumulative rainfall events (Wang et al. 2016). Of particular importance for interpreting post-fire watershed responses is the timing and magnitude of rainfall, and proximity and extent of the burned area relative to drinking water intakes, ultimately influencing the mobilization of post-fire forest floor detritus to source waters.

Both the enrichment of N-DBP precursors and the poor response to coagulation observed in this controlled laboratory study are supported by findings from a post-wildfire field-based study where debris and sediments were mobilized to the CLP river following rainstorms (Hohner et al. 2016). Water providers should be informed that post-wildfire runoff might contain an altered DOM composition significantly less amenable to coagulation than the background watershed DOM. Increased coagulant doses will likely be necessary, and pH may need to be lowered to optimize DOM removal, impacting solids handling processes and filter run times. While findings from this study suggest an altered DOM character, utilities may also experience an increase in influent DOC concentrations coupled with higher, or even extreme, sediment loads, resulting in compounding effects on water treatment due to substantial challenges from high turbidity levels. Exploring alternate water supplies, expanding storage capacity, adding a pre-sedimentation basin, and increasing upstream water quality monitoring with early warning systems are avenues utilities should take into consideration when preparing for, or coping with, wildfire. Additional recommendations for managing post-fire runoff and treatment challenges are discussed in Chapter 6.

CHAPTER 6
THE IMPACT OF WILDFIRES ON TREATMENT PLANT OPERATIONS AND DESIGN

Wildfires and other extreme weather events present great challenges and risks to water utilities throughout the United States. In many cases, such events have caused treatment plants to either shut down, reduce flow, or deliver water that was of inferior quality or failed to meet regulations. These risks are due to changes in the frequency, duration, magnitude, and speed of raw water quality changes, specifically increases in turbidity and NOM. As presented in previous chapters, wildfires often result in increased solids loadings to plants in terms of ash content and runoff from soils due to loss of groundcover, and increased transport of terrestrial NOM of altered character. These two parameters – turbidity and NOM – drive the design and operation of water treatment plants. Other water quality parameters that can influence process selection and plant operations after a wildfire include algae, iron, manganese, and taste and odors.

The purpose of this chapter is to provide guidance to utilities on the impact of wildfires on treatment plant operations and to develop design recommendations to increase treatment resiliency and reliability. Utility managers need to know what new processes or additional facilities they need to install to protect against future water quality excursions. The approach to this chapter is to first describe a conventional water treatment plant, followed by other common plant configurations. Once the major unit processes are described, limitations to each process in terms of water quality and loading rates will be presented. The chapter will conclude with design recommendations for increasing plant resiliency to better address post-wildfire water quality issues.

Treatment challenges expected from wildfires include both short term and long-term issues. Short term issues that may occur in the weeks and months after an event are mostly due to increased turbidity due to ash from post-fire erosion, spikes in NOM, and pH and alkalinity changes. Longer term issues that can occur over several years include:

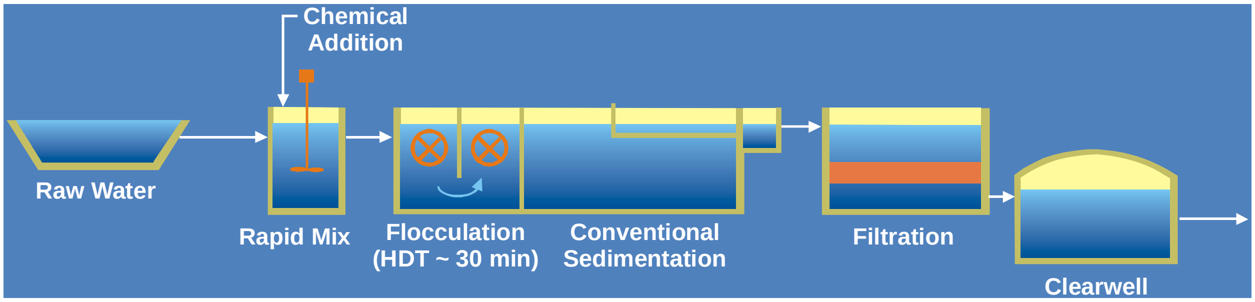
- Increased turbidity from soils due to loss of vegetation – this can continue for years
- Increased NOM
- Changed character of NOM
- Algal growth and associated byproducts: algal toxins and taste and odors that result from release of nutrients after a fire and warmer temperatures due to loss of shade cover
- Iron and manganese

The first focus of this chapter is on the primary initial impacts caused by turbidity and NOM on each unit process. A discussion of treatment challenges expected from wildfires due to other potential secondary impacts is also included.

DESCRIPTION OF MAIN TYPES OF TREATMENT SYSTEMS

Conventional and High-Rate Sedimentation

A conventional treatment plant (Figure 6.1) consists of coagulant addition followed by flocculation, clarification via gravity sedimentation in large rectangular or circular sedimentation basins, and rapid rate granular media filters. Note that high-rate plate and tube sedimentation processes are considered as conventional sedimentation in this discussion as they are governed by the same sedimentation theory as rectangular basins. Coagulant is needed to destabilize particles and to provide metal hydroxide floc for adsorption or co-precipitation of NOM. Since new particles are formed by precipitation, it is important to note that these new particles need to be removed along with particles present in the source water during the sedimentation and filtration processes. Coagulant addition also influences solids handling and disposal practices. It is important that chemical coagulation dosing and pH conditions be controlled such that the removal of both particles and NOM is optimized. Flocculation (slow mixing) is needed to promote particle growth through particle–particle contacts. The goal of flocculation in a conventional treatment plant is to make “settleable” floc, i.e., floc that can be readily be removed in a sedimentation basin. The surface loading (or overflow) rate for conventional sedimentation is in the range of 0.5 to 1 gpm/ft^2, depending on the type of sedimentation process, nature of the raw water turbidity (mineral or non-mineral) and water temperature. Addition of plates or tubes to sedimentation basins will reduce the footprint of the treatment process by providing additional sedimentation surface area which allows for higher hydraulic loading rates, in the range of 2 to 5 gpm/ft^2. Granular media filtration follows sedimentation to remove any remaining particles. Filters typically consist of dual-media (anthracite or GAC over sand) with loading rates of 4 to 8 gpm/ft^2. Filtered water is then collected in a clearwell, where disinfectants and corrosion control chemicals are added, and the water is stored prior to delivery to customers.

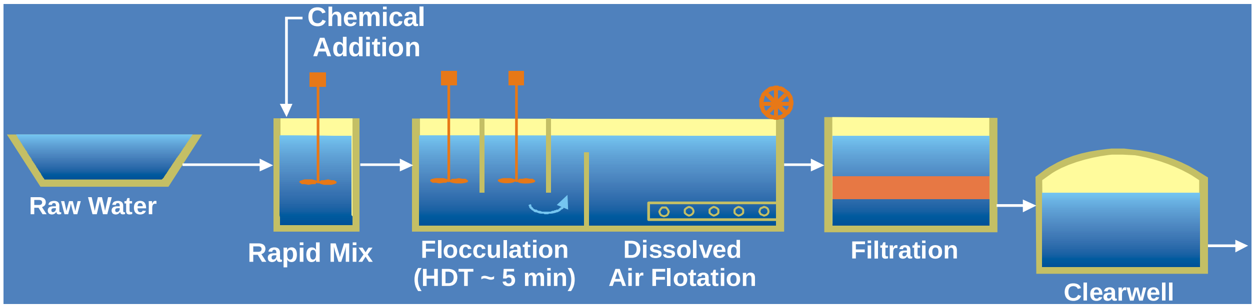


Source: Becker et al. 2018.

Figure 6.1 Schematic of a conventional water treatment plant

Dissolved Air Flotation

A dissolved air flotation (DAF) plant is similar to a conventional sedimentation plant except flocculated particles are removed by attachment to microscopic air bubbles and floating to the water surface (see Figure 6.2). Coagulant addition and flocculation are still used, but the goal of flocculation in this case is to produce a small floc that can be removed by attachment to air bubbles. Flocculation times needed for DAF facilities are on the order of 5-10 minutes (for example, 5 min for the New York City Croton Water Treatment Plant. Two-stage flocculation is commonly used. The typical design surface loading rate of the DAF process usually varies between 6 to 8 gpm/ft^2 -- significantly higher than the loading rate on a conventional sedimentation basin and higher than the footprint loading rates for high-rate plate and tube sedimentation processes. In addition, high-rate DAF processes have recently been developed at rates of 8 to 16 gpm/ft^2. DAF is ideal for treating low turbidity waters and waters with high levels of algae and/or NOM.



Source: Becker et al. 2018.

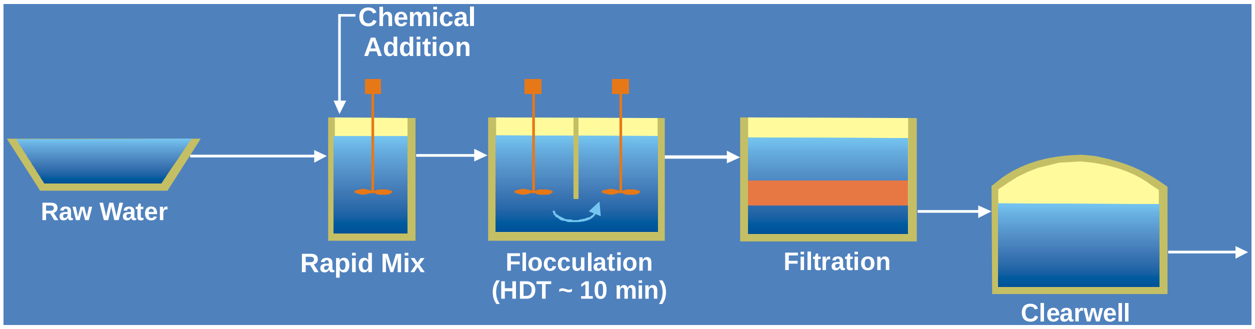
Figure 6.2 Schematic of a DAF plant

Contact Clarification

Contact clarification is a category used for a variety of processes that includes solids-blanket clarification, ballasted-sand clarification, and contact adsorption clarification. Although each of these specific processes is different from a mechanical perspective, they all work by enhancing particle removal via particle–particle or particle–media contacts. Of these, contact adsorption clarification is limited to low turbidity waters with low to moderate TOC levels. Higher turbidity and TOC (and higher coagulant dosages) result in excessive clarifier flushes and reduced plant production. Solids blanket and ballasted-sand clarifiers often have difficulty with algae, but can treat high turbidity source waters.

Direct Filtration

Direct filtration plants are similar to conventional or DAF plants without a sedimentation/clarification step. Flocculation may or may not be included. If flocculation is not included the plant is often referred to as an in-line filtration plant. If flocculation is included, shorter times on the order of 10-15 minutes are normally practiced. A schematic of a direct filtration plant is included in Figure 6.3. Because the filters are the only means of removing solids, direct filtration is limited to very high-quality source waters with low turbidity and low levels of NOM and algae.



Source: Becker et al. 2018.

Figure 6.3 Schematic of a direct filtration plant

IMPACT OF POST-WILDFIRE WATER QUALITY CHANGES ON UNIT PROCESSES

After a wildfire, the range of source water turbidity levels can be quite significant compared to normal operations. For example, in several studies, there have been measurements of turbidities on the order of >>100 NTU (citation needed). Treating these extreme turbidities is unrealistic, and under these conditions, the utility should close down the intake if possible to avoid numerous treatment issues and the possibility of producing unsafe water, contaminating the distribution system. For the sake of discussion, we will examine the case of a plant that needs to treat water at an upper source water turbidity value of 50-100 NTU.

The plant descriptions and unit process loading rates discussed above are applicable to water treatment plants under “normal” source water quality conditions. The performance of each unit process in a plant is dependent on the water quality it experiences and the hydraulic loading rate. In general, poorer water quality requires a decrease in the loading rate to produce acceptable treated water quality, and in severe cases, plants need to shutdown altogether. Since the performance of each unit process is dependent on the performance of prior unit processes, the following begins with a discussion of the impact of raw water quality excursions on coagulation/rapid mix and follows through the plant – flocculation, clarification, filtration, and disinfection. This discussion is presented with the aim of continuously meeting targeted finished water quality goals and regulatory requirements.

Coagulation

The performance of all downstream processes and the quality of finished drinking water is dependent on proper coagulation. Optimum coagulant conditions are a function of the required coagulant dose needed to neutralize the charge on NOM and particles, pH, and the presence of enough alkalinity so hydroxide floc can form. Under- and overdoses of coagulant can result in poor flocculation, high settled water turbidity, high filtered water turbidity and inadequate disinfection (due to pathogen shielding). Likewise, high pH adversely affects coagulation by decreasing the charge on alum and can also lead to high filtered water aluminum levels. Low alkalinity can result in poor floc formation, but very high alkalinity impacts the ability to decrease pH by coagulant or acid addition. Changes in raw water quality due to wildfires can stress the ability of any plant to ensure optimum coagulation.

The impact of increased solids or NOM as a result of a wildfire on coagulation depends on what parameter is driving the optimum coagulant dose. In most cases, NOM is the master variable that dictates optimum coagulant dosing strategies, so increases in turbidity generally have minimal impact, however, there are exceptions. For very high-quality waters treated by direct filtration or adsorption clarifiers, the coagulant dose is often controlled by turbidity. In these cases, post-fire increases in turbidity will likely lead to significant increases in the required coagulant dose. In addition to increased coagulant cost and residuals production, increased turbidity and coagulant doses will lead to shorter filters runs as noted below. For very high-quality waters treated by conventional treatment, an increase in turbidity may enhance the flocculation/sedimentation process such that a decrease in coagulant dose may be feasible. This is because when conventional sedimentation is used on high quality raw waters there are not enough particles for effective flocculation. Higher coagulant doses are often practiced to make enough metal hydroxide precipitate such that flocculation occurs.

Increases in raw water NOM concentrations following a forest fire will almost always require an increase in coagulant dose which equates to increased particulate loading to downstream processes. Failure to use the optimal coagulant dose would result in higher settled water turbidity, and adverse impacts on the filtration process.

Wildfires can also impact raw water pH and alkalinity – both of which affect coagulation. Results from a WRF study that examined the effect of extreme weather events on water quality showed that in some cases heavy rain can result in an increase in alkalinity (when soils were high in limestone content) while in other instances alkalinity decreased (where it was just a dilution impact with no added alkalinity). pH and alkalinity are important parameters to monitor and can be controlled through chemical addition if available.

The following practices are recommended for coagulation:

1. Train utility staff to conduct jar testing on-site in order to support decision making and action during extreme weather events. Research has noted that utilities that were able to perform jar testing on-site were better able to responds to extreme turbidity and water quality excursions and avoid consequences such as turbidity breakthrough and potential boil water advisories.
2. Pay close attention to assure the optimum coagulant dose and pH conditions are maintained. A wildfire that leads to water quality changes will likely lead to increased coagulant requirements in most cases.
3. It may be necessary to increase the frequency of coagulant delivery during or after a wildfire.
4. It may also be necessary to install chemical feed pumps with a higher capacity. Similarly, feed systems to adjust pH and/or add alkalinity may be needed to better respond to turbidity events.

Flocculation

Increased solids loading to a filtration plant will not impact the design or operation of the flocculation process. As mentioned above, with more particles, the flocculation process may improve in some cases – especially when the process is limited by contact opportunities (low turbidity waters). If the raw water turbidity increases dramatically then solids may settle out in the flocculation basin and would have to be periodically removed.

The following practices are recommended for flocculation:

1. Consider installing a means of removing silt that may settle in the bottom of the floc tanks.
2. Evaluate the impact of increased solids production on residuals handling capacity and plan for how to handle additional solids loads during high turbidity events.

Sedimentation

Sedimentation basins remove approximately 90 to 98% of the applied turbidity under typical loading conditions. In order for the filters to function properly, settled water turbidity levels should be less than 2 NTU for dual media filters and less than 3 NTU for deep bed filters. Higher turbidity values will result in higher head loss and shorter filter runs. As an example, if a plant normally achieves 48-hour filter runs at a settled water turbidity of 1 NTU, it is not unrealistic to assume that the filter run length would decrease to 24 hours at 2 NTU and to only 18 hours at 3 NTU, at the same flow rate. Given the typical performance of conventional sedimentation (including plate settlers), raw water turbidity levels of 50 to 100 NTU can be adequately treated without causing downstream filtration issues. Note that increases in coagulant doses due to changes in raw water turbidity or NOM add to the solids loading that must be removed in sedimentation and ultimately handled and disposed of. Also note that post-fire increases in turbidity would require more frequent solids removal from the basins, so automated sludge removal systems should be installed. For example, assuming a turbidity: TSS ratio of 2:1, the following mass loadings of solids are produced as a function of raw water turbidity for a 10 mgd plant. As shown in Table 6.1, increases in raw water turbidity can result in thousands of pounds per day of solids that must be removed, collected, processed, and disposed of.

Table 6.1
Solids produced due to turbidity for a 10 mgd water treatment plant. The numbers shown do not include solids produced from coagulant addition.

Turbidity (NTU)	Mass Loading (lbs/day)
2	83
10	830
100	4200
200	8300

Source: Becker et al. 2018

Recommendations as a function of the type of clarification process are provided below.

1. Conventional sedimentation (including plate settlers)
    a. Conventional sedimentation basins can adequately treat raw waters with turbidity values in the 50 – 100 NTU range. If turbidity is consistently above 100 NTU then a pre-sedimentation basin should be installed.
    b. It should be noted that the maximum turbidity levels that can be treated are a function of the type of turbidity. That is, inorganic clay turbidity settles more readily than turbidity caused by algae or other lighter density material. Elevated levels of non-organic turbidity (i.e., particles) may require an alternative clarification process such at dissolved air flotation (DAF).
2. High-rate clarification
    a. DAF
        i. DAF can treat inorganic turbidities up to 10 NTU for extended periods of time.
        ii. No upper limit for algal based turbidity.
        iii. If (inorganic) turbidities exceed 10 NTU then consideration should be given to installation of a pre-sedimentation basin or converting the plant to conventional settling.
    b. Adsorption clarifier
        i. Limited to 10 NTU or less (regardless of type of turbidity).
        ii. If turbidities exceed 10 NTU then consideration should be given to installation of a pre-sedimentation basin or converting the plant to conventional settling.
    c. Sludge blanket clarifiers
        i. Can handle more than 100 NTU of inorganic turbidity.
        ii. Sludge blanket clarifiers can have difficulty with algae.
    d. Ballasted flocculation

    i. Can handle very high raw water turbidities - greater than 100 NTU.
    ii. Ballasted flocculation clarifiers can have difficulty with algae.
3. Enhance solids removal – in all cases, elevated turbidity levels will increase solids production.
4. If turbidity is greater than 100 NTU, consider a pre-sedimentation basin.

Filtration

Granular media filters are the final particle removal process in drinking water treatment plants. As particles are removed in a granular media filter head loss accumulates and eventually the filter needs to be backwashed. If the time between backwashes is too short, then the plant cannot produce enough water to meet demands. In addition, more wasted backwash water will be produced than can be adequately handled.

Historically, rapid granular media filters consisted of monomedia sand, typically 24 inches of 0.5 mm effective size (ES). These filters were designed to operate at a maximum loading rate of 2 gpm/ft^2. They have given way over the years to dual media filters that typically consist of 20 to 24 inches of 1 – 1.1 mm ES anthracite over 10 to 12 inches of 0.5 mm ES sand. Because of the larger media diameter and greater bed depth, these filters can store more particles and can therefore operate at high loading rates – 4 to 6 gpm/ft^2. Deep bed filters, those with 5 to 6 feet of anthracite, an even larger ES (maybe 1.4 mm) and over 12 inches of sand typically operate at loading rates of 8-12 gpm/ft^2.

The effectiveness of any of these filter media designs is predicated on effective coagulation and the applied turbidity. Effective coagulation is needed to destabilize particles so that they can be removed by sedimentation and filtration. Particles not removed in sedimentation enter the filters, where they are removed and contribute to head loss. From an operations perspective, a useful parameter for evaluating filter production is the unit filter run volume (UFRV). The UFRV is the volume of water produced per square foot of filter area over the course of a filter run. The UFRV should average at least 7,500 gallons per square foot under normal water quality conditions when the settled water turbidity is less than 2 NTU for filters operating at a common filtration rate of 4 gpm/ft^2. This means that 7,500 gallons of water are produced per square foot of filter area between backwashes. This equates to roughly 30 hour filter run lengths at a filtration rate of 4 gpm/ft^2. The minimum desired UFRV is greater than 5,000 gallons per square foot under challenging conditions which equates to roughly 20 hour filter runs at 4 gpm/ft^2. Values lower than this result in excessive waste backwater production and frequent backwashing that makes it difficult for the plant to meet its production goals when producing filtered water with less than 0.1 NTU. Table 6.2 shows UFRV values as a function of filtration rate for a 24-hour period.

Table 6.2
UFRV as a function of filtration rate for a 24-hour filter run

	UFRV (gallons/ft^2)				
Filtration Rate (gpm/ft^2)	**20 hours**	**24 hours**	**36 hours**	**48 hours**	**72 hours**
2	2,400	2,880	4,320	5,760	8,640
4	4,800	5,760	8,640	11,520	17,280
6	7,200	8,400	12,960	17,280	25,920
8	9,600	11,520	17,280	23,040	34,560

An increase in raw water NOM following a forest fire would require an increase in coagulant dosage, which equates to increasing particulate loading to downstream processes. Failure to use the optimal coagulant dose would result in higher settled water turbidity. This would result in higher filter head loss and shorter filter runs and hence more backwash water. In addition, inadequate coagulant dose can lead to poor particle removal in filters, and as a result high effluent turbidity and early turbidity breakthrough.

These operational constraints suggest the following for filter media designs:

1. Filtered water turbidity should always be < 0.1 NTU.
2. Maximum applied turbidity to filters to meet these criteria:
   a. Monomedia filter:
      i. Loading rate: 2 gpm/ft^2
      ii. UFRV $\geq$ 3,000 gal/ ft^2 under extended water quality excursions
      iii. Maximum applied turbidity: 1 NTU
   b. Dual-media filter:
      i. Loading rate: 4-6 gpm/ft^2
      ii. UFRV $\geq$ 5,000 gal/ ft^2 under extended water quality excursions
      iii. Maximum applied turbidity: 1-2 NTU
   c. Deep bed dual-media:
      i. Loading rate up to 12 gpm/ft^2
      ii. UFRV $\geq$ 12,000 gal/ ft^2 under extended water quality excursions
      iii. Maximum applied turbidity: 2-3 NTU
3. Recommendations to increase robustness:
   a. Consider deeper bed, larger diameter media filters to increase solids storage.
   b. Address media size depending on pilot testing or state regulations.
   c. Inspect current filters and conduct a filter surveillance to ensure media is in good condition (e.g., right size, no mudballs, etc.).

Membranes

Membranes (microfiltration or ultrafiltration) are commonly used in place of granular media filters at utilities throughout the country and the use is increasing. Particle removal in membrane systems is by straining – not by transport and attachment mechanisms that work in granular media filters. If the particulate loading is too high, the membrane flux declines which can impact plant production. This leads to increase flushes to remove the particulate matter which impacts residuals handling. Finished water quality is not normally affected since the membranes are absolute barriers. NOM also affects the rate of membrane fouling. When membranes become fouled, they need to be chemically cleaned. Post-fire water quality changes can overwhelm membrane plants, especially when sedimentation is not practiced. Increases in turbidity and NOM can result in flux declines and increased fouling. In addition, firefighting foams can sorb to membrane surfaces and cause excessive fouling.

DESIGN RECOMMENDATIONS FOR UTILITIES UNDER THE THREAT OF WILDFIRES

In addition to the raw water quality, the selection of the optimum treatment processes for any given plant is also a function of site-specific conditions (e.g., space limitations) and operational philosophy of the utility. The following recommendations are presented with the assumption sufficient space is available.

1. Pre-sedimentation basin
    a. May be required/useful if raw water turbidity exceeds 100 NTU for long periods (i.e., days)
    b. Include ability to bypass under normal conditions
2. Coagulation
    a. Ensure chemical storage and feed pumps can deliver the higher chemical doses that may be needed after a wildfire
    b. Consider polymer feed facilities that may be needed to treat waters with ash content
    c. Develop operational protocols and install equipment such as streaming current monitors or zeta potential analyzers to help determine optimum coagulant dosages.
3. Flocculation
    a. Install a means of removing silty solids that may settle out in flocculation tank under high turbidity conditions
4. Sedimentation
    a. Use large conventional sedimentation basins if possible to handle large amounts of solids. If not practical, consider the use of plate settlers.
    b. Ensure solids can be easily removed from basins via mechanical sludge removal equipment.
    c. If in an area where it is not likely that high turbidity will reach the intake, and there is concern that algal blooms could occur, consider dissolved air flotation.
5. Filtration
    a. Consider the use of deep bed dual-media filters with larger media that can store more solids than conventional filters.
    b. Consider GAC in place of anthracite to help with taste and odors.
    c. Provide enough backwash water and waste backwash storage so multiple filters can be backwashed at once.
6. Membranes
    a. Membrane-based treatment systems should not be used if the raw water will be subject to the impact of firefighting foams that could foul membranes.
7. Disinfection
    a. Higher levels of NOM may lead to DBP compliance issues.
    b. Attention should be given to maximizing removal of NOM or relying on the use of alternative disinfectants including UV and ozone.
8. Advanced treatment

a. Smoky taste and odors could occur after a fire.
b. Nutrient release from wildfires could result in long-term eutrophication and increased algal growth in downstream reservoirs leading to taste and odors and algal toxins.
c. The installation of powdered activated carbon or post filter GAC contactors should be considered to handle these events.
d. The installation of ozone/biofiltration should also be considered.

CHAPTER 7
CONCLUSIONS

This project focused on developing a framework for utilities to assess the impacts of a wildfire on water quality and treatment. To do so, post-fire water quality was simulated by heating soil and litter samples collected from the participating utilities in a furnace. Following heating, the samples were leached in low-carbon tap (LCT) water and the character of the DOM was assessed. In addition, bench-scale treatment tests were performed to evaluate the treatability of the leachates, focusing on turbidity, DOC, and DBP precursor removal. Lastly, utility surveys and interviews were conducted to better understand concerns, challenges, and experiences following wildfires and extreme weather events.

Soil and litter samples released different quantities and qualities of dissolved constituents following heating. The release of anions and cation quantity was altered following heating. Anion release into solution showed strong heating dependence, but was not consistent among the measured species. Sulfate concentrations demonstrated the most constant behavior, increasing with heating of each material, especially litter, which was shown to release nearly ten times that of the soil following heating. Nitrate concentrations generally decreased following heating of both litter and soil. Phosphate release was not constant among the soils, but phosphate release from litter increased after heating at 225°C. Iron and manganese had similar trends that generally demonstrated greater release after heating.

Generally, the trends observed for the four utilities were consistent and aid in understanding the effects of heating on water soluble compounds, raw water quality, and the associated treatment challenges. The marginal increase in pH and alkalinity observed for the heated samples may be attributed to the denaturing of organic acids upon heating, with residual alkaline components remaining (Ulery and Graham 1993). An observed decrease in the quantity of DOC leached per gram of material for the heated leachates is consistent with other work indicating partial combustion of soluble organic carbon compounds at 225°C. Alternatively, organic nitrogen has been shown to volatilize at higher temperatures (Hogue and Inglett 2012), supporting the observed enrichment of DON relative to DOC, following heating. SFPUC leachates did not follow the same trend, and the DOC leached per gram of soil increased after heating. SFPUC litter samples were not leached with the soils, and perhaps different organic precursor materials of soils and litter may help explain the difference. Clear and measurable alterations to the soluble DOM character was indicated by increased specific UV absorbance at 254 nm ($SUVA_{254}$). Iron concentrations of the heated leachates were low (< 0.005 mg/L) and did not significantly interfere with absorbance measurements. Consistently higher $SUVA_{254}$ for the heated samples indicates enhanced aromaticity of soluble compounds upon heating, supported by previous soil organic matter studies (González-Pérez et al. 2004, Knicker 2007).

During a natural wildfire, the litter layer is generally consumed leaving behind an ash layer. Ash can take on many different physical characteristics depending on the temperature and duration of heating as well as the parent material, which was not evaluated in these experiments. Therefore, the temperatures and heating durations used for this study are intended to mimic wildfire-watershed responses, but we cannot claim that the results accurately reflect environmental wildfire conditions and post-fire water quality responses. There are currently no rigorous studies examining the quantities and origins of DOM released from recently burned soils and litters. Future studies should examine the release of DOM from recently burned watershed materials to explore the potential shifts in DOM sources that may occur following fire.

TREATMENT STUDIES

For raw water (not coagulated) carbonaceous DBP yields, generally the changes associated with heating were minimal and suggest relatively small and variable alterations in total trihalomethane (TTHM) and haloacetic acid (HAA) precursors at 225°C. Bromide concentrations were low (< 0.003 mg/L), and primarily chlorinated DBP species were formed. An overall decrease in C-DBP reactivity was observed by others (Wang et al. 2015a), and suggests destruction of TTHM and HAA5 water-soluble precursors during heating, despite the observed increase in $SUVA_{254}$, which generally correlates with C-DBP precursor reactivity (Archer and Singer 2006, White et al. 1997). It should be noted, the similar or lower TTHM, HAA5, and haloacetonitrile (HAN) precursor reactivity of the heated leachates compared to the control (unheated) samples may not be representative of the precursor load a water treatment facility might receive in their influent supply, post-wildfire. Enhanced erosion of terrestrial DOM following wildfire can significantly increase DOC levels and DBP formation, as observed in field-based studies (Hohner et al. 2016, Writer et al. 2014). Alternatively, the increase in chloropicrin precursor reactivity following heating may be associated with the enrichment of DON relative to DOC, or elevated inorganic nitrogen levels. Studies have identified 2- and 3-nitrophenol (Merlet et al. 1985, Thibaud et al. 1987) as well as glycine (Mitch et al. 2009) as chloropicrin precursors. While the DOC:DON ratio decreased upon heating, and chloropicrin formation and precursor reactivity per unit carbon increased, HAN4 precursors did not appear altered by heating. Previous studies have associated elevated HAN4 reactivity with wildfire.

Following heating of soil and litter, the leachates consistently exhibited an overall poor response to coagulation and even at high coagulant doses (e.g., > 80 mg/L alum) often marginal DOC removal was achieved (e.g., <30%). The treatability findings are consistent with the results from a field-based post-fire watershed monitoring study when rainstorms transported substantial sediments and debris downstream to the water intake (Hohner et al. 2016). Despite the higher $SUVA_{254}$, a change in DOM quality, such as shifts towards lower molecular weight compounds, may have adversely affected coagulation treatment, resulting in elevated settled water turbidity and minimal DOC removal. Further, finished water quality was negatively influenced, including the exceedance of DBP maximum contaminant levels (MCLs) and high chloropicrin concentrations for the heated leachates compared to control samples. Heat-induced changes to particle size and characteristics were not explored in this study, but may have negatively affected coagulation processes possibly due to the presence of finer materials and should be considered in future work. Treated water nitrogenous DBP formation was also higher for the heated leachates, specifically chloropicrin.

Overall the heated leachates were more difficult to coagulate, requiring higher coagulant doses, consistent with previous work evaluating post-fire river samples collected following the High Park fire (Hohner et al. 2016). Consequently, utilities should plan for higher coagulant doses, and the subsequent solids handling implications on downstream processes. Coagulant doses will likely be case study specific depending on the wildfire (e.g., burned area size, proximity) and watershed (e.g., slopes, hydrology) specific factors, and post-fire flow events (e.g., rainstorm intensity and location). While findings from this study suggest an altered DOM character, utilities may also experience an increase in influent DOC concentrations coupled with higher, or even extreme, sediment loads, resulting in compounding effects on water treatment due to substantial challenges from high turbidity levels. Exploring alternate water supplies, expanding storage capacity, adding a pre-sedimentation basin, and increasing upstream water quality monitoring with early warning systems are avenues utilities should take into consideration when preparing for, or coping with, wildfire.

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ABBREVIATIONS

BCAN	bromo-chloroacetonitrile
CLP	Cache la Poudre
DAF	dissolved air flotation
DBAN	di-bromoacetonitrile
DBP	disinfection byproduct
DCAN	di-chloroacetonitrile
DOC	dissolved organic carbon
DOM	dissolved organic matter
DON	dissolved organic nitrogen
DW	Denver Water
EOH	East-of- Hudson
ES	effective size
GAC	granular activated carbon
HAA	haloacetic acid
HAN	Haloacetonitrile
HH	Hetch Hetchy
HPLC	high-liquid chromatograph
ICP-OES	Inductively Coupled Plasma Optical Emission Spectrometry
LCT	low carbon tap-water
MCL	maximum contaminant level
N-DBP	Nitrogenous disinfection byproduct
NYC	New York City
NYCDEP	New York City Department of Environmental Protection
OM	organic matter
PAC	powdered activated carbon
SEC	size exclusion chromatography
SFPUC	San Francisco Public Utilities Commission
SOM	soil organic matter
$SUVA_{254}$	specific UV absorbance at 254 nm
SW	surface water
TCAN	tri-chloroacetonitrile

TDN	total dissolved nitrogen
TDP	total dissolved phosphorus
TN	total nitrogen
TOC	total organic carbon
TP	total phosphorus
TSS	total suspended solids
TTHM	total trihalomethane
UFRV	unit filter run volume
WM	City of Westminster
WOH	West-of-Hudson
WTS	water treatment systems

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