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Water quality implications of post-wildfire erosion control with polymeric additives†

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Post-wildfire erosion to downstream surface waters can deteriorate water quality to levels that can create challenges for aquatic life and drinking water treatment. Polymeric additives, xanthan gum (XG) and polyacrylamide (PAM), have been demonstrated to be effective for controlling erosion in the presence of hydrophilic ash. However, with repeated rainfall applications, some of the applied XG and PAM may mobilize with the runoff and enter surface waters, which may pose water quality concerns. In this study, indoor rainfall simulation experiments were performed on plots containing wildfire-burned soil overlaid by hydrophilic ash collected after the 2021 Green Ridge Wildfire near Walla Walla, WA. The plots were treated with three concentrations (11, 33, and 60 kg ha^{-1}) of XG or PAM and subjected to three wet–dry cycles. Runoff water samples were collected at 5 min intervals during each wetting event. The pH, electrical conductivity, dissolved organic carbon (DOC), total dissolved nitrogen (TDN), and settled water turbidity (SWT) were measured for runoff water samples. The presence of XG in runoff from XG-treated plots increased SWT by up to 247% and DOC to as high as 16.6 mg_C L^{-1}. PAM treatment also increased DOC (up to 24.5 mg_C L^{-1}) and TDN (up to 5.8 mg_N L^{-1}) in runoff. DOC and TDN concentrations in runoff from treated plots increased with an increase in treatment concentrations and were generally greatest in the first wetting event. The results suggest that benefits of using polymeric additives for erosion reduction should be evaluated together with an assessment of dilution of downstream water bodies to alleviate the negative impacts of the additives on downstream water quality.

Environmental significance

Benefits of using polymeric additives for post-wildfire erosion reduction should be evaluated with an assessment of dilution of downstream water bodies to alleviate the negative impacts of the additives on downstream water quality.

Introduction

Post-wildfire erosion is a concerning effect of wildfires because soil and ash can erode to downstream waterbodies and negatively impact aquatic life.[1,2] For instance, salmonid fish suffered a loss of physiological condition (*i.e.*, mass and length) after the 2009 Lockheed Fire in California.[3] Additionally, the mobilization of burned soil and ash to water supplies can cause challenges for drinking water treatment.[4,5] For example, Hohner *et al.*[6] showed that additional coagulant was required to achieve acceptable levels of organic carbon and disinfection byproducts (DBPs) after the 2012 High Park Fire in Colorado.

Post-wildfire erosion can lead to an increase in concentrations of suspended solids, nitrogen, carbon, phosphorus, and heavy metals, and a reduction in dissolved oxygen in water bodies.[7,8] Several studies have reported an increase in pH,[9,10] electrical conductivity (EC),[11,12] organic carbon,[13] nitrogen,[12,14] and turbidity in surface waters[13,15] following enhanced post-wildfire erosion. Extreme pH levels in streams can stress aquatic animals and may result in mortality.[16] pH also impacts the mobilization and toxicity of pollutants, which in turn impacts aquatic life.[17] EC affects the stability of particles in water, drinking water treatment processes, and aquatic life.[18] Organic matter in aquatic systems control the formation of DBPs and metal binding during drinking water treatment.[19] In addition, high nitrogen concentrations in waterbodies can lead to the generation of algal blooms which can have negative impacts on aquatic life.[20,21] Increases in streamwater turbidity after a wildfire due to the addition of sediment can also cause challenges for drinking water treatment and aquatic life.[6,22]

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Burned slopes in critical locations are often treated after fire to reduce erosion and consequent impacts on downstream waterbodies. Mulching and seeding are the most commonly used methods but have poor efficiency during heavy winds and rainfalls and can introduce invasive plant species.[23,24] Our recent study showed that the polymeric additives xanthan gum (XG) and polyacrylamide (PAM) are effective for reducing erosion in the presence of ash on top of soil, despite some of the applied additives moving with the runoff along with soil and ash.[25] However, for the polymeric additives (XG and PAM) to be considered viable alternatives to the conventional methods of mulching and seeding, the impacts of these treatments on runoff water quality need to be evaluated.

The mobilization of XG and PAM to waterbodies can have potential implications to aquatic life and drinking water treatment as both additives can increase the turbidity of water, total organic carbon, total nitrogen, and acidity.[26] However, the overall impacts of XG and PAM treatment on water quality depend on the actual mass of soil, ash, and additives mobilized to aquatic systems and the dilution of downstream water bodies.[26] In addition, the post-treatment reduction in sediment loss from the landscape may compensate for the potential negative impacts of XG and PAM on water quality. This study evaluated the impacts of various concentrations of XG and PAM treatment on runoff water quality including pH, EC, dissolved organic carbon (DOC), total dissolved nitrogen (TDN), and turbidity. Indoor rainfall simulation experiments were performed on plots containing soil and ash collected after the 2021 Green Ridge Wildfire and treated with three concentrations of XG and PAM (11, 33, and 60 kg ha^{-1}). The runoff from untreated and treated plots was collected and water quality analysis was performed and recommendations on the use of XG and PAM for post-wildfire erosion mitigation are provided.

Study site

The study site was in a moderate soil burn severity area, on 30° slope burned by the 2021 Green Ridge Wildfire, near Walla Walla, WA. The fire started on July 7, 2021 and was contained on October 4, 2021. The soil burn severity was determined based on Burned Area Emergency Response (BAER) maps. BAER teams classify the soil burn severity depending on multiple site factors including fuel type and load, climate, and post-fire vegetation. Moderate burn severity indicates consumption of up to 80% of pre-fire ground cover, incomplete consumption of fine roots, black and gray ash, presence of canopy needles on the ground, and unchanged soil structure.[27] The dominant vegetation in the region includes ponderosa pine (*Pinus ponderosa*), Douglas-fir (*Pseudotsuga menziesii*), grand fir (*Abies grandis*), and lodgepole pine (*Pinus contorta*). The watershed within the burned perimeter contains two major water bodies – Meadow Creek and Panjab Creek which drain into the Tucannon River – a tributary of the Snake River. The Tucannon River flows through the burnt region for 18 km.

Materials

Soil and ash

Soil and ash were collected in the second week of August, 2021 prior to any rainfall. Ash was collected from the surface using a hand shovel. The ash layer was removed and soil was collected from the top 10 cm in bulk evenly along the slope. The samples were stored in the laboratory at room temperature until experiments and analysis. Particle size distribution of soil and ash were measured using sieve and hydrometer analysis (ESI Fig. 1†). A Pario Automated Particle Size Analyzer (Meter Group, Pullman, WA), which calculates particle size using Stoke's Law as particles that are in a suspension settle in a graduated cylinder, was used for particle size distribution. Soil was classified as high plasticity silt (MH) according to the Unified Soil Classification System (USCS). Ash had predominantly silt-sized particles (76.3%).

The Atterberg limits (ASTM D4318), specific gravity (ASTM D792-14), loss on ignition, water droplet penetration time (WDPT), permeability (constant head test, ASTM D2434-19), zeta potential, and exchange complex (ASTM D7503-18) of soil and ash were determined following standard methods (Table 1). Loss on ignition was determined by heating the soil at 550 °C for 4 h in a muffle furnace as described by Scalia *et al.*[28] For the WDPT test soil and ash samples were packed in 2 cm diameter, 1 cm height containers and 10 water droplets were placed on the surfaces using a standard medicine dropper. The average time required for penetration of each drop was recorded as the WDPT and used to classify the water repellency according to King[29] and Chenu *et al.*[30] The zeta potential was measured using suspensions of soil or ash (20 mg solid, 50 mL deionized water) in a Malvern Zetasizer Nano Z.

Polymeric additives

Anionic XG from Sigma-Aldrich (CAS 11138-66-2, St. Louis, MO) and PAM from SNF Inc., Riceboro, GA (FLOBOND™ A 30) were used in this study. Energy dispersive spectrometer (EDS) spectrum of XG and PAM (ESI Fig. 2†) indicated that both XG and

Table 1 Physical and chemical properties of soil and ash from the 2021 Green Ridge Fire site

Property	Soil				Ash			
Liquid limit	52				50			
Plasticity index	7				15			
USCS classification	MH				—			
Specific gravity	2.66				2.54			
LOI (%)	5.8				16.6			
WDPT (s)	<1 (hydrophilic)				<1 (hydrophilic)			
Saturated permeability (cm s^{-1})	1.24×10^{-4}				7.76×10^{-4}			
Zeta potential (mV)	−31.8				−25.4			
CEC (cmol$^+$ per kg)	25.8				20.7			
Cations	Ca^{2+}	Mg^{2+}	Na^+	K^+	Ca^{2+}	Mg^{2+}	Na^+	K^+
Soluble cations (cmol$^+$ per kg)	1.0	0.2	0.1	0.1	1.8	0.8	0.6	6.5
Bound cations (cmol$^+$ per kg)	7.8	0.7	—	0.1	8.1	10.4	—	1.9
	O	Fe	Si	C	Al	Ca, Mg	O	C	Si	Ca	K, Al, Mg	
Elemental composition (%)	29	21	19	14	9	<5	41	30	11	10	<5	

PAM contain carbon and oxygen in their chemical formula and PAM also contains nitrogen. The dominant cation in XG is potassium and in PAM is sodium. EDS spectrums of XG and PAM are given in the ESI materials.†

Methods

Rainfall simulation experiments

Rainfall simulation experiments were performed on custom-made plots as described by Ahmed *et al.*[25] The plots contained soil (S plots), soil overlaid by 1 cm thick layer of ash (A plots), or soil overlaid by ash and surface treated with low (L: 11 kg ha^{-1}), medium (M: 33 kg ha^{-1}), and high (H: 60 kg ha^{-1}) concentrations of XG (X–L, X–M, and X–H plots, respectively) or PAM (P–L, P–M, and P–H plots, respectively). Ash thickness was selected based on typical conditions observed after fires in the study region. The PAM and XG concentrations were selected to test a wide range of concentrations. The highest concentration was selected based on the recommendations of the PAM supplier and lower concentrations were selected to span a range of concentrations evenly. The same concentrations were used for XG for comparison. The plots had a length and width of 68.5 cm, and a depth of 7 cm on the front and 15 cm on the remaining three sides.[31,32] The shorter thickness at the front allowed unconstrained movement of soil or ash with rainfall. Soil was sieved through a custom-made sieve (0.63 cm diameter) to remove large roots and gravel and compacted into plots in three 3 cm layers at the *in situ* water content of 2.75% to a void ratio of 2.1. Ash (0.49 kg) was placed on A plots as an uncompacted layer of 1 cm thickness. Powdered XG and PAM were uniformly sprinkled on the plots. Two plots were prepared for each condition.

The plots were subjected to three wet–dry cycles following Akin *et al.*[32] During each wetting event, plots were inclined at 30° and subjected to a rainfall of intensity 100 mm h^{-1} for 30 min to simulate a high intensity storm, which corresponds to a 15 min rainfall of 50 year return period at the study site.[33] The rainfall was applied using a rainfall simulator equipped with VeeJet nozzles that produce raindrops of 3 mm average diameter at a terminal velocity of 8.8 m s^{-1}, providing a potential energy of 275 kJ ha mm^{-1} at 3 m height.[34] Drying was done under two ultraviolet lights (5700 W) on a horizontal platform. The first drying event was for 4 h and the second drying event was 8 h. The durations were selected to simulate drying between two successive rainfalls in the field.[32] Runoff samples were collected at 5 min intervals for estimation of runoff volume, sediment loss, and water quality. Aliquots of the runoff samples collected at 5 min intervals were separated for water quality analysis. The remaining runoff was filtered through 8 μm filters to determine the runoff and sediment loss. The filtered runoff was used for water quality analysis.

Controlled water quality experiments

Controlled water quality experiments were conducted to evaluate the individual impacts of soil, ash, XG, and PAM on water quality, prior to rainfall simulation experiments. For soil and ash experiments, 25 g of solid was mixed in 75 mL of ultrapure water for 30 min followed by 15 min of settling. pH and EC were measured using a VWR Symphony B10p probe and Orion conductivity meter, respectively. For DOC and TDN analysis, soil and ash samples were prepared following similar methods as Rodela *et al.*[35] In 100 mL of DI water, 1 g of soil or ash was mixed followed by filtration through glass fiber filter with 0.7 μm pore size. DOC (as non-purgeable organic carbon) and TDN were measured using a Shimadzu TOC-V Analyzer with nitrogen module. Settled water turbidity (SWT) was measured by mixing 10 g of soil or ash in 100 mL of DI water for 30 min and allowing the suspension to settle for 60 min. The top 20 mL of supernatant was separated and SWT was measured using Hach TL2300 turbidimeter.

For polymeric additives, the XG and PAM were added to ultrapure water and mixed for 16 to 24 hours using a magnetic stirrer to obtain various concentrations (*i.e.*, 0.01%, 0.03%, 0.1%, 0.25%, and 0.5%) of XG and PAM. The solutions were analyzed for the beforementioned water quality parameters. The impacts of XG and PAM concentrations on pH, EC, DOC, TDN, and turbidity of ultrapure water has been reported by Ahmed *et al.*[26]

Water quality analysis of runoff samples

Aliquot samples were separated from the runoff collected from the plots during the wetting events for pH, EC, and turbidity measurements. Individual 5 min samples were analyzed. All pH and EC measurements were taken at 25 °C. Samples were mixed thoroughly for 5 min using an end-over-end method. After mixing, the samples were settled for 10 min followed by the measurement of pH and EC of the supernatant. For turbidity measurements, the aliquot samples were mixed thoroughly and allowed to settle for 60 min. Aliquots from the filtered runoff samples were again filtered through glass microfiber filters (0.7 μm pore size) to remove particulate organic carbon and nitrogen prior to DOC and TDN measurements.

Statistical analysis

Statistical analysis was done for all the water quality parameters to determine if the differences in water quality parameters among all the plots (soil, ash, and all the treated plots) were statistically significant. Statistical significance of the results was evaluated using the one-way analysis of variance (ANOVA) at a confidence level of 95%. ANOVA produced a p-value of less than 0.05 which suggests that the probability of the null hypothesis (*i.e.*, no difference in mean values for various groups) to be true was less than 5%. Tukey HSD tests were performed to compare the statistically significant difference in the mean values of water quality parameters from untreated and treated plots during each wetting event. Both ANOVA and Tukey HSD tests were performed using R.[36]

Results and discussion

Controlled water quality experiments

The results of the controlled water quality experiments suggest that the presence of ash in water can increase pH, EC, DOC,

Table 2 pH, electrical conductivity (EC), dissolved organic carbon (DOC), total dissolved nitrogen (TDN), and settled water turbidity (SWT) determined from the controlled water quality experiments. Reported values are mean and standard deviation of three replicates

	pH	EC (mS cm^{-1})	DOC (mg$_C$ L^{-1})	TDN (mg$_N$ L^{-1})	SWT (NTU)
Soil	7.1 ± 0.0	0.6 ± 0.0	12.3 ± 0.4	15.1 ± 0.5	523 ± 7
Ash	10.7 ± 0.0	13.4 ± 0.5	15.2 ± 0.4	15.1 ± 0.7	167 ± 3

TDN, and SWT, which is generally supported by previous research.[37] However, the specific effects of wildfire ash on water quality are highly variable and depend on combustion completeness, fuel source, post-fire rainfall, and time-since-fire.[38] Soil, on the other hand, had negligible impacts on pH and EC but increased DOC, TDN, and SWT in water. The study soil pH was 7.1, while the pH of the ash was 10.7. Ash had higher EC and DOC than soil (Table 2), TDN of ash and soil were very similar, whereas SWT of soil was greater than soil. These results suggest that the soil and ash, from this study site, can impact water quality, with different trends for soil and ash, in some instances.

At all concentrations, PAM had negligible impacts on pH (pH ranged from 7.2 to 7.7), whereas 0.25% and 0.50% of XG reduced pH to less than 6.5 (Fig. 1a). Both XG and PAM increased the EC of water and EC increased with increasing XG and PAM concentrations (Fig. 1b). Even at the lowest concentration (*i.e.*, 0.01%), both XG and PAM increased the DOC drastically (Fig. 1c), with PAM introducing more DOC in water than XG at all concentrations. PAM also increased TDN in water,

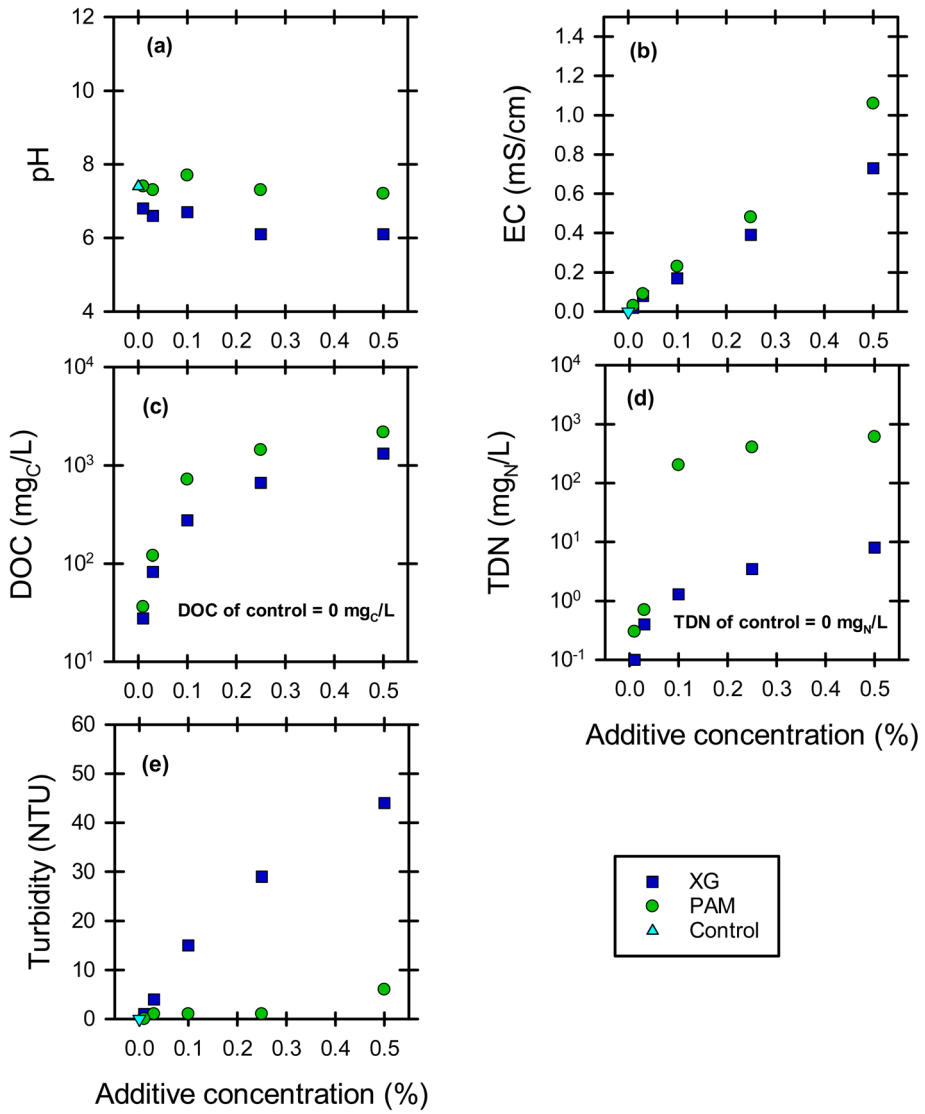

Fig. 1 Effects of XG and PAM concentration on pH (a), electrical conductivity (b), dissolved organic carbon (c), total dissolved nitrogen (d), and turbidity (e).

proportional with PAM concentration (Fig. 1d). At 0.1% and higher concentrations, XG increased the turbidity of water whereas PAM had negligible effects on turbidity (Fig. 1e). These results suggest that the polymeric additives can have potential, negative implications for water quality if they erode to surface waters.

Rainfall simulations: runoff and sediment loss

In the first wetting event, S and A plots had negligible runoff (<100 mL) and sediment loss (<5 g), whereas all treated plots produced runoff and sediment loss, which both generally increased with increasing treatment concentration (Table 3). In the second wetting event, A plots produced 25% less runoff and 56% lower sediment loss than S plots. Further, X–L and P–L plots produced 3% (for X–L) and 11% (for P–L) lower runoff than A plots and the sediment loss was 53% lower (for X–L) and 78% lower (for P–L). Plots treated with medium and high concentrations of XG and PAM produced higher runoff than A plots (7 to 16% higher for XG and 34 to 40% higher for PAM). However, compared to A plots, sediment loss was 69 to 81% lower for XG and 86 to 93% lower for PAM. In the third wetting event, S and A plots had similar runoff but the sediment loss for A plots was 29% lower than S plots. The runoff from all treated plots was lower than A plots (10 to 25% lower for XG and 11 to 30% lower for PAM). Similarly, compared to A plots, sediment loss from XG-treated plots was 9 to 66% lower and from PAM-treated plots was 53 to 94% lower.

The runoff from A plots was lower than S plots in the second wetting event, while in the third wetting event, both S and A plots produced similar runoff (Table 3). The sediment loss from A plots also increased with subsequent wetting events. The trends in runoff and sediment loss suggest that with subsequent wetting events, spatial and vertical redistribution of ash occurred and some of the ash moved with the runoff and the remaining ash migrated into the soil. This resulted in reduced ash availability on top of soil in A plots and consequent reduction in water retained by ash and increase in runoff.

All treated plots produced runoff in the first wetting event due to the surface sealing effects of additives. XG and PAM swell upon hydration which results in reduced infiltration.[32] Furthermore, XG and PAM increase the viscosity of percolating water which causes a reduction in hydraulic conductivity.[39] The surface sealing effects of XG and PAM decreased with subsequent wetting events due to the movement of XG and PAM with the runoff and into the plots. The reduction in surface sealing resulted in lower runoff from treated plots in the third wetting event. The sediment loss from the treated plots was lower than from A plots for all wetting events which indicates not all of the additives moved with the runoff, and that XG or PAM present in the plots bonded with ash and soil and decreased the sediment loss. However, with subsequent wetting events, sediment loss from treated plots increased due to progressive loss of additives. Results also suggest that PAM bonded the ash particles more effectively than XG and therefore the sediment loss from PAM-treated plots was lower than XG-treated plots.

Rainfall simulations: water quality of runoff

The settled water turbidity (SWT) from A plots was 396% higher in the first wetting event, 75% lower in the second wetting event, and 26% higher in the third wetting event compared to S plots (Fig. 2). SWT in runoff from S plots had a very high correlation[40] with the sediment concentration during the wetting events (Pearson's correlation coefficient, $\rho = 0.91$; Fig. 3) suggesting that the turbidity in runoff from S plots was primarily due to suspended soil particles. On the contrary, SWT from A plots had a low correlation with sediment concentration ($\rho = 0.5$). Even though suspended sediment concentration is often used as a proxy for turbidity, a strong correlation between turbidity and sediment concentration is not always true as turbidity depends on several factors such as type, shape, size, and refractive index of particles.[41] As the runoff from A plots contained both soil and ash, turbidity was influenced by both the components. Moreover, since SWT was measured, the difference in the settling rate of soil and ash particles influenced turbidity measurements which helps explain the relatively lower correlation coefficient for A plots compared to S plots.

All concentrations of PAM-treated plots had SWT lower than A plots for all wetting events. SWT for all PAM-treated plots was significantly lower than A plots in the third wetting event ($p < 0.01$). Alternatively, in the first and second wetting events, SWT

Table 3 Cumulative runoff and cumulative sediment loss from plots during the three wetting events. The values in parenthesis show the percent increase or decrease in runoff and sediment loss from treated plots relative to the control (A plots) for the second and third wetting events

	Wetting 1[a]		Wetting 2		Wetting 3	
Plots	Cumulative runoff (L)	Cumulative sediment loss (g)	Cumulative runoff (L)	Cumulative sediment loss (g)	Cumulative runoff (L)	Cumulative sediment loss (g)
S	0.0	2	8.3	910	9.4	745
A	0.1	1	6.2	398	9.3	526
X–L	1.0	9	6.0 (−3%)	193 (−52%)	8.4 (−10%)	477 (−9%)
X–M	4.2	31	7.2 (+16%)	122 (−69%)	7.1 (−25%)	223 (−58%)
X–H	5.7	41	6.6 (+7%)	77 (−81%)	7.1 (−24%)	179 (−66%)
P–L	1.3	9	5.5 (−11%)	89 (−78%)	8.3 (−11%)	248 (−53%)
P–M	4.7	44	8.3 (+34%)	30 (−93%)	7.4 (−21%)	60 (−89%)
P–H	6.4	31	8.7 (+40%)	54 (−86%)	6.4 (−31%)	31 (−94%)

[a] Comparison for the first wetting event not available as runoff and sediment loss from A plots were negligible.

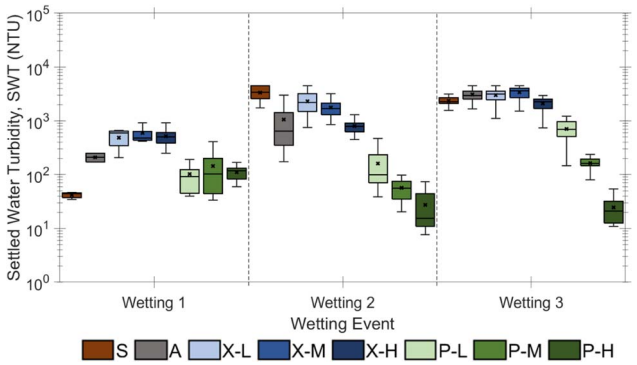

Fig. 2 Settled water turbidity (SWT) of the runoff from the plots. The 'x' mark shows the mean SWT values. S: bare soil, A: soil overlaid by ash, X: A plots treated with xanthan gum, P: A plots treated with polyacrylamide, L: low concentration (11 kg ha^{-1}), M: medium concentration (33 kg ha^{-1}), H: high concentration (60 kg ha^{-1}). Darker blue or green indicates higher concentration. Runoff samples were collected in 5 min intervals and 2 plots were prepared for each condition.

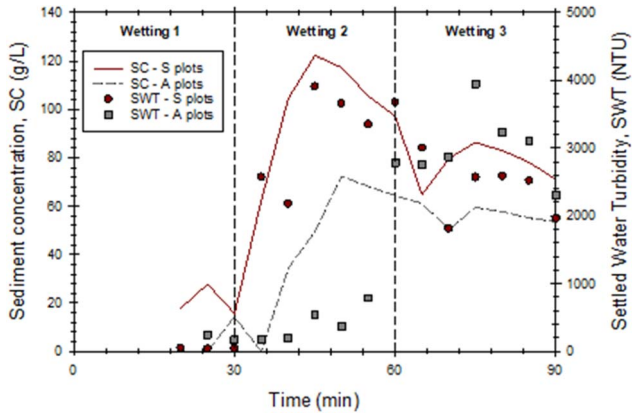

Fig. 3 Correlation between sediment concentration and settled water turbidity (SWT) in runoff from untreated plots (S and A plots) during the three wetting events.

from XG-treated plots was generally higher than A plots even though the sediment in runoff was less than 50 g in the first wetting event and less than A plots (52 to 81% less) in the second wetting event. The only exception was X–H plots in the second wetting event which had 5% lower SWT. The higher turbidity from XG-treated plots was attributed to the influence of XG in runoff on the settling of suspended particles. Fig. 1 shows that XG does not increase the turbidity when dissolved in water at low concentrations (0.03%, or lower). However, XG can increase the viscosity of water[42] and an increase in viscosity leads to an increase in drag force on settling particles and a consequent decrease in the terminal velocity, which could influence the 60 min settled water turbidity measured.

To further investigate the impacts of XG and PAM on settling of soil particles, soil suspensions were prepared by mixing 10 g of soil and 0.2 g of XG or PAM in 250 mL of DI water. A control sample was also prepared without XG or PAM. After mixing, each sample was rested for 60 min and the top 50 mL of suspension was pipetted out and passed through glass microfiber filters to obtain the concentration of suspended solids. The suspended solids concentration of the control sample was 0.5 g L^{-1}, whereas for samples containing XG and PAM, suspended solids concentrations were 39.5 g L^{-1} (7800% higher) for XG and 14 g L^{-1} (2700% higher) for PAM. This suggests that XG and PAM can increase the time required for settling of particles and suspended solids concentration, which explains the higher turbidity from XG-treated plots in the first and second wetting events. Although PAM also reduced the particle settling rate by increasing the viscosity of water, the sediment loss from PAM-treated plots during all wetting events was less than A plots (53 to 94% less), and therefore SWT of runoff was also lower (32 to 99% lower) than A plots.

Results indicate that XG can potentially negatively impact the settling of particles in waterbodies which can have implications on aquatic life such as decreased production and abundance of fishes.[43] However, the dilution of runoff by the receiving waterbodies might alleviate the negative effects of XG on turbidity. PAM treatment, on the other hand, can decrease turbidity in streams by reducing erosion.

The pH of runoff from S plots was relatively constant during the three wetting events (Fig. 4), whereas for A plots, pH decreased with subsequent wetting events, suggesting the ash primarily eroded in the first wetting event. A plots had higher pH than S plots in the first and second wetting events which is attributed to the movement of ash with the runoff. Depending on the type and composition, wildfire ash can increase the pH of runoff water by two to three units.[44] White to light grey ashes, which are formed at temperatures greater than 400 °C, typically have pH of 10, or higher.[35,45] High pH of ash is due to presence of carbonates, oxides, and hydroxides of metals such as calcium, magnesium, potassium, and sodium.[45] The ash used in this study had a pH of 10.7, while the pH of soil was 7.1 (Table

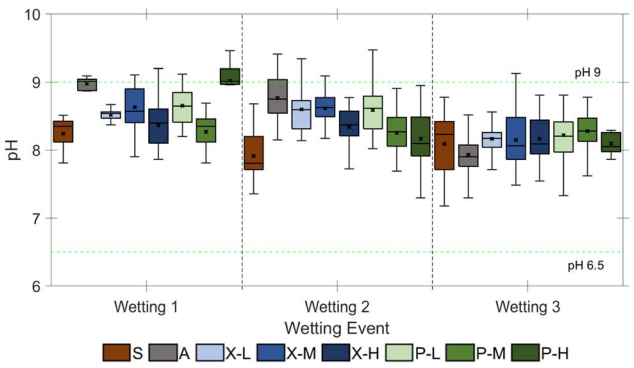

Fig. 4 pH of the runoff from the plots and the EPA recommended surface water ranges of pH for aquatic life (shown with dashed lines). The 'x' mark shows the mean pH values. S: bare soil, A: soil overlaid by ash, X: A plots treated with xanthan gum, P: A plots treated with polyacrylamide, L: low concentration (11 kg ha^{-1}), M: medium concentration (33 kg ha^{-1}), H: high concentration (60 kg ha^{-1}). Darker blue or green indicates higher concentration. Runoff samples were collected in 5 min intervals and 2 plots were prepared for each condition.

2), which also suggests that the higher pH for A plots was due to the presence of ash in runoff. In the third wetting event, pH from A plots was the same as S plots which indicates that the proportion of ash in the sediment from A plots was very low. This could be due to the migration of ash into the soil with the applied rainfall.

The pH of runoff from the treated plots was generally lower than control plots (A plots) in the first and second wetting events (except for P–H plots in first wetting – mean pH 9.1) and slightly higher than A plots in the third wetting event (8.1 to 8.3 pH for treated plots and 8.0 pH for A plots). The trends in pH of runoff from the treated plots are attributed to the combined effect of ash, soil, and XG or PAM. The initially lower pH from treated plots was because of lower sediment loss as well as the presence of XG or PAM in the runoff which reduced the effects of ash on pH. With subsequent wettings, the amount of sediment mobilized from treated plots increased which resulted in an increase in pH in the third wetting event. Results suggest that XG and PAM treatment would not have additional negative impacts on the pH relative to the untreated condition (A plots), and generally were within the pH range considered safe for aquatic life. Moreover, the overall effects of mobilization of soil, ash, XG or PAM on the pH of downstream waterbodies will depend on the dilution and buffering capacity of the receiving waters and the background water quality.

The EC of runoff from A plots was greater than S plots for all wetting events however, the difference decreased with subsequent wetting events (Fig. 5). Higher EC for A plots is because of the presence of ash in the runoff and the decrease in EC with subsequent wettings is attributed to the decrease in ash content in the sediment eroded from the plots. Wildfire ash can contain a variety of inorganic constituents such as calcium, magnesium, potassium, silicon, phosphorus, sodium, and sulfur.[37,46,47] Compounds of these elements dissociate in water and increase EC.[48] The EDS spectrum showed that the ash used in this study contained carbon, calcium, silicon, aluminum, and small

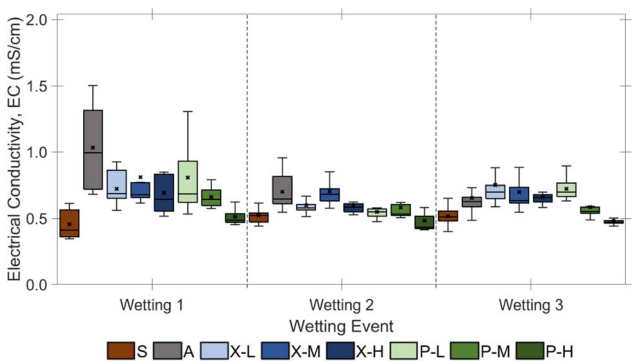

Fig. 5 Electrical conductivity (EC) of the runoff from the plots. The '*x*' mark shows the mean EC values. S: bare soil, A: soil overlaid by ash, X: A plots treated with xanthan gum, P: A plots treated with polyacrylamide, L: low concentration (11 kg ha^{-1}), M: medium concentration (33 kg ha^{-1}), H: high concentration (60 kg ha^{-1}). Darker blue or green indicates higher concentration. Runoff samples were collected in 5 min intervals and 2 plots were prepared for each condition.

concentrations of iron, potassium, and sodium. These elements were likely responsible for higher EC from A plots. Moreover, EC of ash (13.4 ± 0.5 mS cm^{-1}) was greater than soil (0.6 ± 0.0 mS cm^{-1}), which further supports the explanation that higher EC from A plots was due to the presence of ash in runoff.

For plots treated with a low concentration of XG and PAM (*i.e.*, X–L and P–L plots), the EC of runoff was lower than A plots for the first and second wetting events because of the lower sediment loss. However, the presence of XG or PAM in the runoff also contributed to EC as both XG and PAM can increase EC (Fig. 5). In the third wetting event, EC from X–L and P–L plots was higher than the controls likely because of the mobilization of ash and soil due to loss of XG and PAM from the plots with subsequent wetting events. For plots treated with medium and high concentrations of XG and PAM, EC was lower than A plots (0 to 17% lower for XG and 12 to 50% lower for PAM) for all wetting events because of lower sediment loss (58 to 81% lower for XG and 86 to 94% lower for PAM). The difference in the EC of runoff from treated plots and A plots decreased with subsequent wetting because of the progressive increase in sediment loss from treated plots as a result of XG and PAM redistribution.

The runoff volume from S and A plots in the first wetting event was insufficient for DOC measurements. Therefore, DOC from S and A plots was measured only for the second and third wetting events. The high variability observed for DOC is because during each wetting event, the concentration of DOC was greatest in the first 5 min of runoff collection and decreased for the remaining time intervals. For instance, the DOC of runoff collected from S plots at 5, 10, 15, 20, 25, and 30 min intervals during the first wetting event were 22.6, 15.8, 12.3, 11.5, and 9.8 $mg_C\ L^{-1}$, respectively. The same trend was observed for all plots, which suggests that the first few minutes after the generation of runoff can be more critical in terms of DOC export from the landscape. Relative to S plots, DOC from A plots was 34% lower in the second wetting event, and 44% lower in the third wetting event (Fig. 6). DOC in runoff from the untreated plots (S and A plots) was due to the mobilization of burned soil and ash. Even though ash leached more DOC than soil in controlled experiments (15.2 ± 0.4 $mg_C\ L^{-1}$ for ash and 12.3 ± 0.4 $mg_C\ L^{-1}$ for soil), the greater sediment loss from S plots resulted in greater DOC in runoff compared to A plots. Increased DOC concentrations in streams following wildfires have been reported by other studies.[13,49]

XG- and PAM-treated plots generated enough runoff for DOC measurement during all wetting events. For all concentrations of XG and PAM, DOC was not significantly different from A plots. In the first wetting event, DOC from treated plots was primarily due to the presence of XG or PAM in runoff as the sediment loss from all treated plots in the first wetting event was less than 50 g (Table 3). Even at a low concentration (0.01%), XG and PAM induced 27.6 $mg_C\ L^{-1}$ and 36.2 $mg_C\ L^{-1}$, respectively, of DOC to water (Fig. 1). The lower DOC concentration from X–L and X–M plots compared to A plots in the second wetting event is likely because of lower sediment loss. Even though the sediment loss from X–H plots was also less than A plots, mobilization of more XG due to high treatment concentration resulted in an increase in DOC greater than

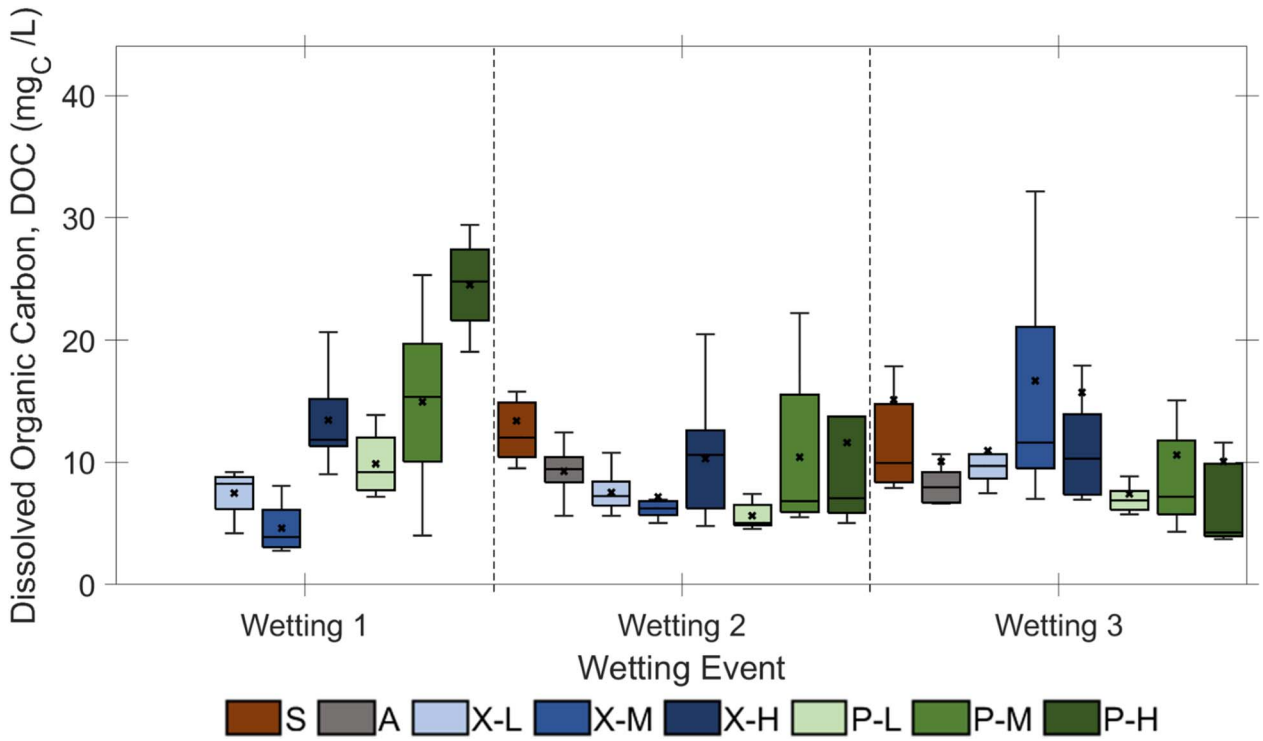

Fig. 6 Dissolved organic carbon (DOC) of the runoff from the plots. The '*x*' mark shows the mean DOC values. S: bare soil, A: soil overlaid by ash, X: A plots treated with xanthan gum, P: A plots treated with polyacrylamide, L: low concentration (11 kg ha^{-1}), M: medium concentration (33 kg ha^{-1}), H: high concentration (60 kg ha^{-1}). Darker blue or green indicates higher concentration. Runoff samples were collected in 5 min intervals and 2 plots were prepared for each condition.

control plots. Similarly, runoff from P–M and P–H plots also had higher DOC concentrations in the second wetting event due to the presence of PAM in runoff. Also, in the third wetting event, DOC from X–M, P–M, and X–H plots were higher than A plots because of the presence of XG or PAM and increased sediment content in the runoff. Overall, XG and PAM treatment can result in an increase in DOC concentrations in aquatic systems beyond the untreated conditions, especially when applied at medium (33 kg ha^{-1}) and high (60 kg ha^{-1}) concentrations.

Even though XG and PAM treatment provided benefits related to erosion control, they produced runoff with DOC greater than 4 mg_C L^{-1}, especially when applied at a concentration of 33 kg ha^{-1} or higher. Therefore, XG and PAM can potentially lead to an increase in DOC concentrations in downstream waterbodies and challenge drinking water treatment. However, the overall DOC concentration and the impacts of increased DOC from post-fire runoff will depend on downstream dilution, the water chemistry of aquatic systems, and the robustness of drinking water treatment plants.

TDN was measured only for the second and third wetting events for S and A plots due to insufficient runoff in the first wetting event. TDN concentrations from A plots were lower than S plots (Fig. 7). Additionally, the TDN in runoff from A plots was less than S plots because of lower cumulative sediment loss from A plots (Table 3) as the TDN of soil and ash were similar (15.1 ± 0.7 mg_N L^{-1} for ash and 15.1 ± 0.5 mg_N L^{-1} for soil). Post-wildfire increases in TDN in streams have been reported in past studies and is attributed to the increase in sediment concentration.[49,51] For instance, Roebuck *et al.*[51] showed a strong coupling between TDN and turbidity in stream water after the 2020 Holiday Farm Fire in Oregon. However, the

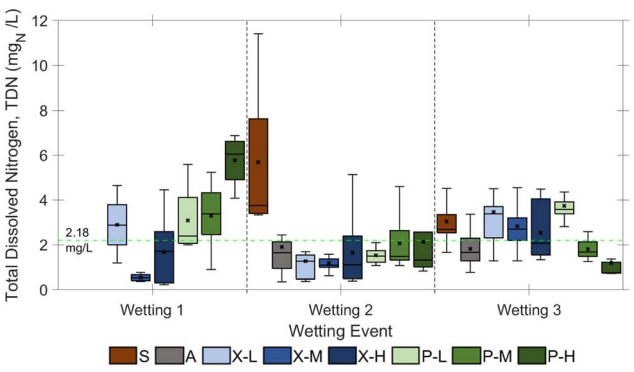

Fig. 7 Total dissolved nitrogen (TDN) of the runoff from the plots and EPA recommended range[50] for aquatic life (shown with dashed line). Safe limit varies based on the region; dashed line shows the recommended concentration for ecoregion 6 which has the maximum upper limit. The '*x*' mark shows the mean TDN values. S: bare soil, A: soil overlaid by ash, X: A plots treated with xanthan gum, P: A plots treated with polyacrylamide, L: low concentration (11 kg ha^{-1}), M: medium concentration (33 kg ha^{-1}), H: high concentration (60 kg ha^{-1}). Darker blue or green indicates higher concentration. Runoff samples were collected in 5 min intervals and 2 plots were prepared for each condition.

reported post-wildfire TDN concentrations in streams range from 0.1 to 1.4 $mg_N\ L^{-1}$.[49,51,52] These concentrations are lower than the TDN in runoff from S and A plots observed in this study, likely because of dilution by stream waters.

For all treatment concentrations, TDN concentrations of runoff from PAM-treated plots were higher than XG-treated plots in the first wetting event (7% for low, 560% for medium, and 287% for high treatment concentration) and in the second wetting events (15%, 75%, and 31% higher for low, medium, and high treatment concentration). Higher TDN in runoff from PAM-treated plots is because PAM contains nitrogen, whereas XG does not. This suggests that TDN sources from PAM-treated plots include both sediment and PAM, while for XG-treated plots, TDN is from only the sediment. Compared to A plots, in the second wetting event, all XG-treated plots had lower TDN (33%, 38%, and 14% lower for X–L, X–M, and X–H plots), whereas plots treated with medium and high concentration of PAM had higher TDN (8% and 12%, higher for P–M and P–H plots). Lower TDN from XG-treated plots was attributed to lower sediment loss. Although PAM-treated plots also produced less sediment than A plots, the presence of PAM in runoff increased the TDN concentration. In the third wetting event, TDN from XG-treated plots was higher than A plots (89%, 61%, and 47% higher for X–L, X–M, and X–H plots) and TDN decreased with increasing XG concentration. This trend was consistent with sediment loss from XG-treated plots, which also decreased with increase in treatment concentrations indicating that higher TDN is due to more sediment in runoff. TDN was higher for P–L (112% higher) and P–M (6% higher) plots in the third wetting event, but lower for P–H plots (49% lower). TDN concentration in the third wetting event reflected the combined effect of sediment (soil and ash) and PAM in the runoff.

Results suggest that PAM treatment can lead to an increase in TDN concentration in runoff and subsequently in downstream water bodies. Moreover, with an increase in PAM concentration, TDN in runoff increased initially, however, with subsequent wettings, TDN from PAM-treated plots decreased due to lower sediment loss. Although TDN was above the EPA[53] recommended limits in rivers and streams for aquatic life (0.12–2.18 $mg\ L^{-1}$, depending on the ecoregion) for PAM-treated plots in the first wetting event, and for XG-treated plots in the third wetting event, the concentration will be affected by the background water quality of streams and dilution effects in receiving waters. Nevertheless, PAM treatment, especially in medium and high concentrations, can potentially increase TDN which can lead to problems including the formation of nitrogen-based DBPs and increases in N-DBP formation have been observed after wildfires.[53] Moreover, increased nitrogen may lead to harmful algal blooms.[20]

Conclusions

XG and PAM treatment at all concentrations did not negatively impact the pH of runoff relative to the untreated control conditions. The presence of XG in water can result in a decrease in pH below 6, the safe limit for aquatic life, and can increase turbidity. DOC increased in runoff due to XG and PAM treatment, especially when the treatment concentration was 33 $kg\ ha^{-1}$, or higher. If sufficient dilution does not occur in downstream waterbodies, increased DOC can lead to increased formation of DBPs during drinking water treatment. PAM treatment can also increase TDN concentrations in runoff. Increased TDN can result in harmful algal blooms in lakes and reservoirs and impact aquatic life. Moreover, high TDN concentration in water can contribute to nitrogen-based DBP formation during drinking water treatment. XG can also lead to an increase in SWT beyond untreated conditions as it slows the settling of particles by increasing the viscosity. Due to the negative impacts of XG and PAM treatment on runoff water quality, an assessment of quality of downstream waterbodies is recommended to be performed before applying XG and PAM on burned slopes to control erosion.

Data availability

Data for this article are presented in the article.

Conflicts of interest

The authors have no competing interests to declare that are relevant to the content of this article.

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