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WILDFIRE AND HYDROLOGICAL PROCESSES

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Watershed-scale vegetation, water quantity, and water quality responses to wildfire in the southern Appalachian mountain region, United States

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Abstract

Wildfires are landscape scale disturbances that can significantly affect hydrologic processes such as runoff generation and sediment and nutrient transport to streams. In Fall 2016, multiple large drought-related wildfires burned forests across the southern Appalachian Mountains. Immediately after the fires, we identified and instrumented eight 28.4–344 ha watersheds (four burned and four unburned) to measure vegetation, soil, water quantity, and water quality responses over the following two years. Within burned watersheds, plots varied in burn severity with up to 100% tree mortality and soil O-horizon loss. Watershed scale high burn severity extent ranged from 5% to 65% of total watershed area. Water quantity and quality responses among burned watersheds were closely related to the high burn severity extent. Total water yield (*Q*) was up to 39% greater in burned watersheds than unburned reference watersheds. Total suspended solids (TSS) concentration during storm events were up to 168 times greater in samples collected from the most severely burned watershed than from a corresponding unburned reference watershed, suggesting that there was elevated risk of localized erosion and sedimentation of streams. NO_3-N concentration, export, and concentration dependence on streamflow were greater in burned watersheds and increased with increasing high burn severity extent. Mean NO_3-N concentration in the most severely burned watershed increased from 0.087 mg L^{-1} in the first year to 0.363 mg L^{-1} (+317%) in the second year. These results suggest that the 2016 wildfires degraded forest condition, increased *Q*, and had negative effects on water quality particularly during storm events.

KEYWORDS
nitrate, sediment, southern Appalachians, water quality, water yield, wildfire

1 | INTRODUCTION

Large-scale forest wildfires can significantly affect hydrologic processes such as runoff generation and sediment and nutrient transport to streams (Bladon, Emelko, Silins, & Stone, 2014; Hallema et al., 2017; Hallema et al., 2018; Neary, Gottfried, DeBano, & Tecle, 2003; Neary, Ryan, & DeBano, 2005; Smith, Sheridan, Lane, Nyman, & Haydon, 2011). These changes in hydrological processes are driven by the immediate, short-term ecosystem effects of wildfire, such as changes in vegetative cover and structure, followed by mid- and long-term changes in forest structure and composition (Elliott, Vose, & Hendrick, 2009; Lentile et al., 2007), forest floor mass and chemistry (Knoepp, DeBano, & Neary, 2005; Knoepp, Elliott, Clinton, & Vose, 2009), and physical and chemical properties of the mineral soil (DeBano, Neary, & Ffolliott, 2005; Knoepp et al., 2005). The magnitude and duration of these responses depend on the interactions among fire severity, post-fire precipitation regime, topography, soil characteristics, and vegetative recovery rate (Feikema, Sherwin, & Lane, 2013; Niemeyer, Bladon, & Woodsmith, 2020). Hydrological effects of severe wildfires in arid and semi-arid regions of the western United States and globally are well documented (Bladon et al., 2014; Hallema et al., 2017; Hallema et al., 2018; Neary et al., 2003; Neary et al., 2005; Smith et al., 2011). However, water quantity and quality responses to wildfire in forested watersheds of the humid eastern United States remain relatively unknown (Hallema et al., 2018). This lack of information is due, at least in part, to the rarity of large-scale wildfires in eastern forests. Indeed, Swanson (1981) suggested that fire plays a small role in geomorphological and hydrological processes in eastern forests due primarily to very low fire return periods in the region; however, projections of climatic conditions in the eastern United States suggest that wildfires may be more prevalent in the future (Liu, Goodrick, & Stanturf, 2013; Mitchell et al., 2014; Terando, Youngsteadt, Meineke, & Prado, 2018).

Wildfires in the western United States have been shown to significantly affect water quantity and quality. Water quantity typically increases following severe wildfire due to reduced vegetation interception and transpiration, and altered soil physical properties including reduced thickness of the forest floor, soil surface sealing, and increased soil water repellency (Bladon et al., 2014; Hallema et al., 2017; Neary et al., 2003). Severe wildfires have also been shown to significantly affect downstream water quality due to soil erosion and subsequent turbidity and sedimentation in streams, rivers, and water supply reservoirs (Emelko et al., 2016; Neary et al., 2003; Neary et al., 2005; Smith et al., 2011), altered stream cation, anion, and pH (Bodi et al., 2014; Neary et al., 2005), increased N and PO_4 concentrations (Bodi et al., 2014; Rust, Hogue, Saxe, & McCray, 2018), and altered dissolved and particulate organic carbon concentrations, (Rhoades et al., 2019; Smith et al., 2011).

There is a long history of research on the ecosystem and water quality effects of wildland fire in forests of the southern Appalachians of the eastern United States (Clinton, Vose, Knoepp, & Elliott, 2003; Elliott & Vose, 2006; Elliott, Vose, Knoepp, & Jackson, 2012; Vose & Elliott, 2016); however, most research has addressed low-to-moderate severity prescribed fire. These studies have shown that prescribed fires have minimal and short term effects on stream water quality measured as NO_3-N and sediment delivery to streams (Clinton et al., 2003; Elliott & Vose, 2005). However, the applicability of these prescribed fire studies to watersheds impacted by high severity wildfire is largely unknown, and there are few studies evaluating the effects of fire on water quantity in the region. Understanding the effects of wildfire on water quality and quantity in the southern Appalachian mountains is important given the extent of forest cover, the large downstream populations dependent on clean and dependable water supplies (Caldwell et al., 2014), and projected future increases in fire season length and wildfire potential in the region (Liu et al., 2013).

In October and November 2016, prolonged dry conditions and multiple ignitions led to 21 wildfires of mixed severity across the southern Appalachians (NC, SC, GA, and TN). These wildfires were large-scale by historical standards, ranging from 216 to 11,195 ha with a total of more than 61,000 ha burned in the region (MTBS, 2020) and created a mosaic of burned severity not typical of prescribed fires (personal communication, Greg Brooks, Fire Management Officer, USDA Forest Service Nantahala Ranger District). Thus, these wildfires provided us with an opportunity to understand the effects of moderate and high severity burned areas on southern Appalachian forest watersheds. Our overall objective was to evaluate the effects of the 2016 wildfires at the watershed-scale by comparing mixed-severity burned watersheds to nearby unburned reference watersheds. Specifically, we sought to examine the effect of fire severity on post-fire tree and shrub mortality, forest floor characteristics, streamflow, and water quality (suspended sediment, pH, cations, anions, dissolved organic carbon, particulate organic matter, inorganic N, and phosphate) under baseflow and stormflow conditions.

2 | METHODS

2.1 | Site description

The Nantahala National Forest is located in western North Carolina, United States, in the Appalachian Highlands Region, Blue Ridge Province (Fenneman, 1917). The forest encompasses more than 215,000 ha with latitude ranging from approximately 35.0 to 35.4° N and longitude from 82.9 to 84.3° W. Climate is classified as marine, humid temperate (Peel, Finlayson, & McMahon, 2007) with mean air temperature (*T*) of 12.6°C and annual precipitation (*P*) of 1,375 mm yr^{-1} (NCDC, 2020), although *T* and *P* can vary according to elevation and aspect (Laseter, Ford, Vose, & Swift, 2012). Most *P* occurs as rain in frequent, small, low intensity storms in all seasons, while water yield is typically highest in March–April and lowest in September–October and is perennial even during extreme drought (Swift, Cunningham, & Douglass, 1988). Vegetation is southern mixed deciduous forest with the overstory principally consisting of oaks, hickories, red maple, and tulip poplar while evergreen shrubs including rosebay rhododendron and mountain laurel often form a dense mid-story subcanopy. Details on the geology, soils, and vegetation in the southern Appalachian region can be found in Simon, Collins, Kauffman, McNab, and Ulrey (2005). Watersheds burned in the Tellico Fire and the Camp Branch Fire in the Nantahala National Forest were selected for study (Figure 1). The 5,739 ha Tellico Fire started on November 3, 2016, while the 1,310 ha Camp Branch Fire

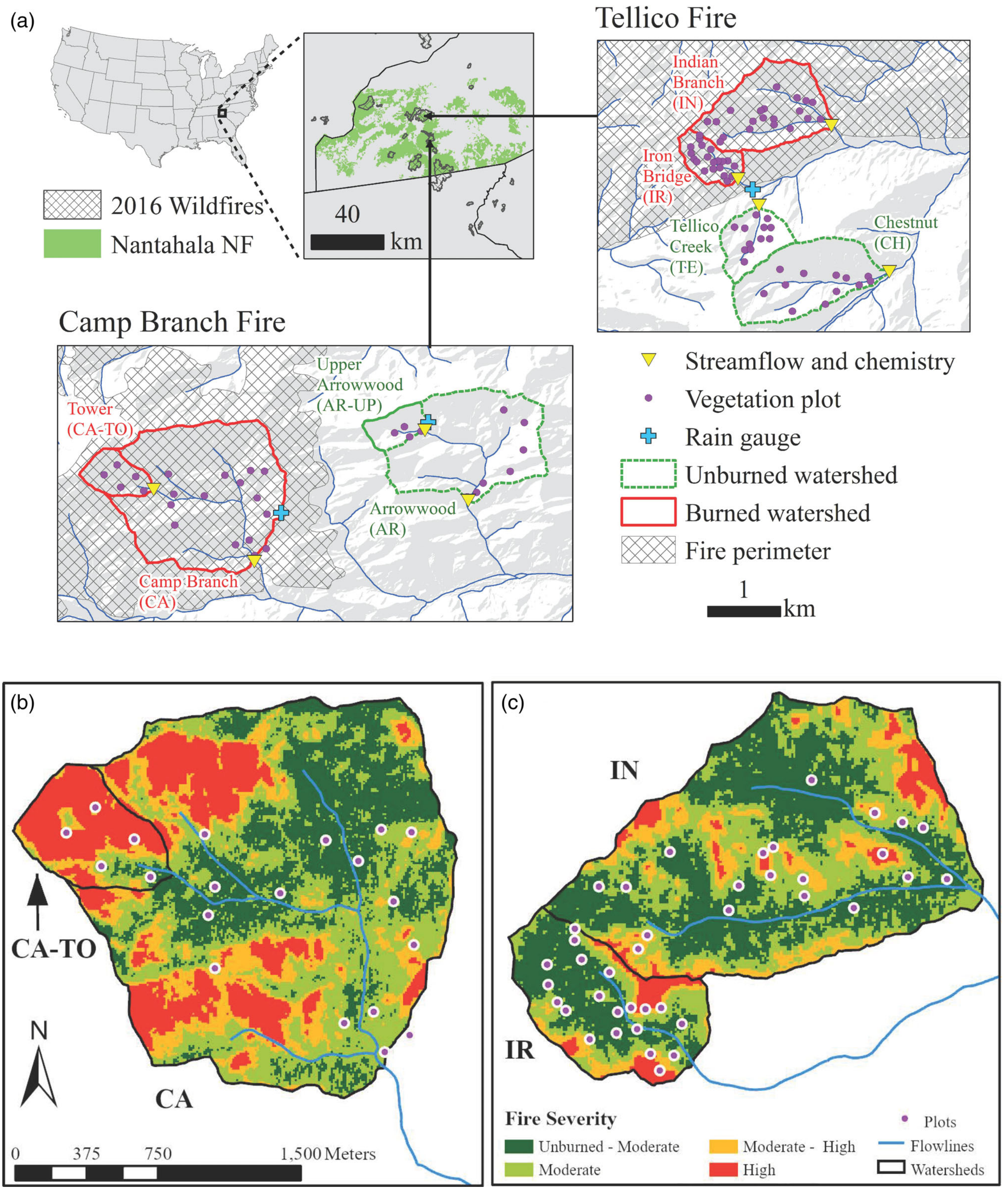

FIGURE 1 Location of the Nantahala National Forest (NF), 2016 wildfires, burned and unburned watersheds and associated burn severity and water quantity and quality measurement locations (a), and fire severity of Camp Branch Fire (b), and Tellico Fire (c) based on the relation between plot burn severity classification and the difference between remotely-sensed June Normalized Burn Ratio (NBR) before and after fire

started on November 23, 2016. An approximately 50 mm precipitation event on November 28, 2016 extinguished most of the wildfires in the region.

We identified burned and unburned watersheds located inside and outside the fire burn perimeters, respectively (Figure 1). Unburned watersheds were selected to be as similar in topographical

TABLE 1 Summary of watershed characteristics

Characteristic	Camp branch fire: Arrowwood (AR)	Camp branch (CA)	Upper Arrowwood (AR-UP)	Tower (CA-TO)	Tellico fire: Chestnut cove (CH)	Indian branch (IN)	Tellico Creek (TE)	Iron bridge (IR)
Watershed condition	Unburned	Burned	Unburned	Burned	Unburned	Burned	Unburned	Burned
Area (ha)	228.4	343.5	28.4	35.1	160.4	158.2	63.1	42.3
Aspect	S	S	SE	SE	E	ESE	NNE	SE
Elevation min (m)	788	819	976	1,185	870	840	1,019	1,042
Elevation max (m)	1,230	1,628	1,230	1,628	1,539	1,355	1,417	1,398
Elevation mean (m)	1,002	1,179	1,119	1,435	1,161	1,095	1,211	1,208
Slope mean (%)	50%	54%	51%	64%	57%	60%	63%	66%
Plot mean (SE) pre-fire tree stem density (stems ha^{-1})	883 (162.1)	683 (43.1)	702 (85.7)	704 (99.9)	834 (60.8)	1,017 (88.6)	953 (108.6)	1,069 (76.5)
Plot mean (SE) pre-fire tree basal area (m^2 ha^{-1})	38.1 (4.84)	33.2 (3.00)	48.5 (8.56)	35.1 (4.69)	39.0 (6.12)	39.3 (3.84)	37.3 (3.91)	45.1 (2.47)
Watershed mean (SE) RdNBR	46.6 (0.23)	292 (1.83)	48.1 (1.30)	729 (6.81)	27.2 (0.12)	139 (1.34)	33.9 (0.25)	176 (3.69)
Unburned – Moderate (%)	–	27%	–	5%	–	43%	–	46%
Moderate burn severity (%)	–	35%	–	12%	–	37%	–	31%
Moderate – High burn severity (%)	–	17%	–	17%	–	15%	–	12%
High burn severity (%)	–	21%	–	65%	–	5%	–	11%

Note: The corresponding reference for each burned watershed is shown in the column to the left of the burned watershed.

characteristics to corresponding burned watersheds as possible (Table 1). Streamflow and chemistry sampling locations for each watershed were selected where equipment installation was feasible. Upstream watersheds from the stream sampling locations were delineated in a Geographic Information System (GIS) using a 1/9 arc-second (~3 m) resolution digital elevation model (U.S. Geological Survey, 2020). All burned and unburned reference watersheds were completely forested and have had intermittent partial harvests since the late 1800s; the watershed area-weighted mean stand age in 2016 ranged from 81 to 127 years old (U.S. Department of Agriculture-Forest Service, 2017a). At the Camp Branch Fire, the 343.5 ha burned Camp Branch watershed (CA) was compared with the neighbouring 228.4 ha unburned Arrowwood Creek watershed (AR) (Figure 1). A second pair of smaller burned (Tower Branch, CA-TO) and unburned (Upper Arrowwood, AR-UP) watersheds were nested within the larger watersheds of CA and AR, respectively. This allowed us to quantify responses in the highest severity burned area in the northwestern portion of the larger CA watershed (Figure 1). At the Tellico Fire, burned watershed Iron Bridge (IR) was compared to nearby unburned watershed Tellico Creek (TE) while burned watershed Indian Branch (IN) was compared to unburned watershed Chestnut Cove (CH) (Table 1).

2.2 | Vegetation, soil, and plot-level burn severity measurements

We established 96 survey plots (7 m radius) across burned and unburned watersheds between January and February 2017 with the former representing a mosaic of fire severities (Figure 1). Specifically, we established 20 plots across each of the three larger burned watersheds CA, IR, and IN, and 12 plots across each of the unburned watersheds AR, TE, and CH. Each plot center location was randomly selected on maps, field identified, and permanently marked and referenced using Global Positioning System (GPS). We tagged all trees in each plot so that we could quantify tree mortality over time and we measured diameter at 1.37 m height (i.e., diameter at breast height) of all live and dead trees ≥5.0 cm diameter. Tree species were grouped by fire tolerance (Vose & Elliott, 2016) to assess the implications of differential tree mortality on forest species composition. Fire-intolerant species were generally mesophytic and included tulip poplar (*Liriodendron tulipifera* L.), black birch (*Betula lenta* L.), blackgum (*Nyssa sylvatica* Marsh.), red maple (*Acer rubrum* L.), and others. Fire-tolerant species were generally xerophytic and included hickories (*Carya* spp.), oaks (*Quercus* spp.), sourwood (*Oxydendrum arboreum* [L.] De Candolle), and others.

Plot-level burn severity was assessed using measurements of forest floor depth (soil Oi and Oe + Oa horizons), bole char height of each tree (a measure of flame height), tree mortality, basal area loss, and mineral soil exposure. Forest floor depth, tree bole char height, and mineral soil exposure were measured in January and February 2017, while tree mortality and basal area loss were assessed in September 2017 (one growing season after the wildfires and before leaf fall). Plot mean char height was based on all trees, live and dead. Basal area was calculated for live and dead trees to estimate percent basal area lost through tree mortality due to the wildfire. Trees that were likely dead prior to the wildfire event were not included in these

TABLE 2 Severity classes for measured characteristics used to compute the plot-level composite burn severity index, and composite burn severity index ranges used to define plot-level categorical burn severity, also shown are the RdNBR value ranges used for each watershed scale burn severity class

		Burn severity class and rank				
Characteristic	Scale	5 High	4 Moderate high	3 Moderate	2 Low moderate	1 Low
Tree mortality (%)	Plot	≥ 50	31 49	21 30	11 20	0 10
BA loss (%)	Plot	≥ 30	16 29	11 15	6 10	0 5
Bole char height (m)	Plot	>1.0	0.8 1.0	0.5 0.7	0.2 0.4	0 0.1
Oe + Oa depth (cm)	Plot	≤ 0.6	0.7 0.8	0.9 1.1	1.2 1.4	> 1.4
Exposed mineral soil (%)	Plot	≥ 25	16 24	11 15	1 10	0
Composite burn severity index range	Plot	>3.5	2.8–3.5	2.0–2.7	1.3–1.9	≤1.2
RdNBR range	Watershed	>541	182–541	62–181	<62	

estimates. We estimated forest floor depth for each plot by averaging 20 point depth measures that were uniformly distributed across the plot. The percentage of mineral soil exposure in each plot was estimated by calculating the proportion of the 20 point locations used for forest floor depth estimation that had mineral soil exposed at the soil surface. Surface (0–10 cm depth) mineral soil samples were collected in three randomly selected locations within each plot using a 10-cm-long, 4.3-cm-diameter PVC core. Samples were composited, air dried, and sieved <2 mm prior to laboratory analysis of soil inorganic N (NH_4-N + NO_3-N) using a Flash EA.

We developed a composite burn severity index for each field plot similar to previous assessments (e.g., Ryan and Noste (1985); Hille and Stephens (2005); Key and Benson (2006); Lezberg, Battaglia, Shepperd, and Schoettle (2008); Metz, Frangioso, Meentemeyer, and Rizzo (2011); see review Keeley (2009)) using a combination of tree mortality, tree basal area loss, bole char height, forest floor depth, and exposed mineral soil. Each of these characteristics was assigned a 1–5 (low to high) rating (Table 2) and the composite burn severity index value was computed as the mean of the ratings over all characteristics for each plot. Categorical plot-level burn severity classes were defined using ranges in the composite burn severity index: Low: ≤1.2; Low-Moderate: 1.3–1.9; Moderate: 2.0–2.7; Moderate-High: 2.8–3.5; and High: >3.5.

2.3 | Watershed-scale burn severity assessment

We estimated watershed-scale burn severity using the satellite-based Relative differenced Normalized Burn Ratio (RdNBR) technique (Miller & Thode, 2007) with categorical watershed-level RdNBR burn severity class ranges informed by plot-level burn severity index values. Copernicus Sentinel-2 MultiSpectral Instrument Level-1C NBR images before (June 9, 2016) and after (July 26, 2017) the fire were obtained for burned area mapping from the Sentinel data access hub (https://sentinel.esa.int/web/sentinel/sentinel-data-access). These images were atmospherically corrected to Bottom of Atmosphere (BOA) reflectance using the algorithm "sen2cor" v2.2.5 with Sentinel-2 Toolbox at 10 m resolution. Growing season images before and after the fire were used to minimize the confounding effect of vegetation phenology. The post-fire image was all clear-sky, while a small portion of the pre-fire image was masked using the cloud information at band "QA60." The homogeneous nature of the vegetation in the study area allowed us to fill the cloud-masked image on June 9, 2016 using a nearest-neighbour interpolation method. The RdNBR was computed from the Copernicus (2020) NBR data according to Equation 1.

$$\mathrm{RdNBR} = \frac{(\mathrm{Prefire\ NBR} - \mathrm{Postfire\ NBR}) * 1000}{\left(\sqrt{|\mathrm{Prefire\ NBR}|}\right)} \quad (1)$$

We established the relationship between plot-level composite burn severity index and RdNBR (within a 10 m buffer on plot center) by first determining which plot measurements were significantly related to RdNBR using the Spearman's Rank correlation coefficient. Only basal area loss, mortality, and char height were useful in predicting RdNBR. We then recomputed the plot-level composite burn severity index using the methods described in section 2.2, but using only ratings for basal area loss, mortality, and char height. The recomputed plot-level composite burn severity index was then fitted to RdNBR using a logistic regression model. We performed a ten-fold cross-validation using 90% of the data to train a logistic model and the remaining 10% to test the model. The resulting R^2 and Root Mean Squared Error (RMSE) for the ten testing datasets were then averaged. Using an approach similar to Miller and Thode (2007), categorical watershed-level RdNBR burn severity class ranges were established by first assigning five severity classes a sequential integer value (i.e., 1 = Low, 2 = Low-Moderate, 3 = Moderate, 4 = Moderate-High, and 5 = High). We then used the midpoint of adjacent values to define composite burn severity index ranges for each class (e.g., Moderate = 2.5–3.5), and lastly used the logistic regression model to calculate RdNBR for the associated composite burn severity index value ranges.

2.4 | Hydroclimatic measurements

2.4.1 | Precipitation measurements

Precipitation (P) was measured at three locations: (a) a central location between burned and unburned watersheds of the Tellico Fire, (b) in the burned area of the Camp Branch Fire, and (c) in the nearby unburned AR watershed (Figure 1). The precipitation measurement sites were all within 1.9 km from the center of corresponding watersheds. At each site, one standard rain gauge (RG202; Stratus, Fergus Falls, MN) was installed at a height of 1 m from the ground surface. Tipping bucket rain gauges (TR-525 M; Texas Electronics, Dallas, TX) were installed at 1 m height at the Tellico Fire precipitation site and the AR precipitation site adjacent to the Camp Branch Fire to allow us to examine *P* intensity during storm events. Each tipping bucket rain gauge was fitted with an event data logger (UA-003-64; Onset Computer Corporation, Bourne, MA). Long-term *P* measurements for RG19 at the USDA Forest Service Coweeta Hydrologic Laboratory (Miniat, Laseter, Swank, & Swift, 2017), located approximately 16 and 26 km from the Camp Branch and Tellico fire sites, respectively, were used to characterize pre- and post-burn precipitation (e.g., average vs. wetter or drier than average).

2.4.2 | Streamflow measurements

Stream stage (*h*) was measured at the outlet of each of the eight watersheds at five-minute intervals between February 2017 and December 2018 using two unvented pressure transducers (U20-001-04; Onset Computer Corporation, Bourne, MA) at each site (one transducer in the stream, one suspended in air to compensate for atmospheric pressure). We measured instantaneous stream discharge (*D*) under a wide range in flow conditions at each watershed outlet using the slug injection salt (NaCl) dilution gauging technique as described in (Moore, 2004, 2005). Discharge-stage rating curves were developed for each watershed (Figure S1) using a power function fitted to the *D* and associated *h* measurements for each watershed (World Meteorological World Meteorological Organization, 2010). Watershed water yield (*Q*) was computed by dividing *D* by watershed area estimated in a GIS. We separated total *Q* (Q_{tot}) into baseflow (Q_{bf}) and stormflow (Q_{sf}) components using the flow separation technique described in Hibbert and Cunningham (1966) and Brantley, Miniat, Elliott, Laseter, and Vose (2014). Briefly, an event begins when two successive flow measurements exceeds the slope of the flow separation line (slope = 0.5465 L s^{-1} km^{-2}). Stormflow is accumulated separately from baseflow until the separation line intersects the recession limb of the hydrograph.

2.5 | Stream water quality measurements

Water samples were collected both as weekly grab samples under baseflow conditions from December 2016 through December 2018 and during sampled storm events from December 2016 through August 2017. Both grab and storm samples were collected using automated samplers (6712; Teledyne ISCO, Lincoln, NE) installed at each site. For storm sampling, the samplers were programmed prior to each sampled storm to collect up to 24 one-litre water samples beginning at a pre-determined time and sampling interval based on National Weather Service forecasted times of storm onset and duration. Water samples were collected from the samplers on the day following the storm event and delivered immediately to the analytical lab to ensure that all samples met shelf-life requirements.

All water samples were analysed for NO_3-N, NH_4-N, SO_4, PO_4, Cl, dissolved organic carbon (DOC), pH, Ca, Mg, K, Na, total suspended solids (TSS), and total volatile solids (TVS) at the Coweeta Hydrologic Laboratory, Analytical Chemistry Lab according to standard analytical protocols (U.S. Department of Agriculture-Forest Service, 2017b). Solution pH for grab samples was determined using a Thermoscientific Orion A211 pH benchtop meter (Thermo Fisher Scientific, Madison, WI) with a Broadley James pH combo probe (Broadley James Corp, Irvine, CA). Solution K, Na, Ca, and Mg concentration were analysed by Optical Emission Spectroscopy using a Thermo Fisher iCAP 6,300 (Thermo Fisher Scientific, Madison, WI). Solution NO_3-N, PO_4, Cl, and SO_4 concentrations were analysed by Micro-membrane Suppressed Ion Chromatography using a capillary AS 18 column using a Thermo Scientific ICS 4000 capillary Ion Chromatograph (Dionex, Sunnyvale, CA). NH_4-N concentration was measured using an automated Phenate method with an Astoria 2 Autoanalyser (Astoria-Pacific, Clackamas, OR). Solution DOC was analysed in samples filtered with a Whatman GF/C glass 1.5 microfiber filter using a catalytically-aided platinum 680°C combustion technique for sample oxidation with a Shimadzu DOC-VCPH TNM-1 analyser (Shimadzu Scientific Instruments, Columbia, MD). TSS concentrations were calculated using a gravimetric method by filtering samples with a pre-weighed Whatman GF/C glass 1.5 micro fibre filter muffled at 550°C, drying filters for 2.5 hr at 65°C, and reweighing. After computing TSS, the weighed filter and samples were placed in a pre-muffled crucible, muffled at 550°C, and re-weighed to compute TVS. TVS is often used as a surrogate for particulate organic matter (Jensen, Scanlon, & Riscassi, 2017). Where analyte concentrations for a given sample were below the Method Detection Limit (MDL) (Table S1), one-half of the MDL was used when computing summary statistics and for export calculations.

Watershed exports of analytes between February 2017 (when flow measurements began) and the end of the study in December 2018 were computed using daily *Q* for each watershed in combination with analyte concentrations in weekly watershed grab samples, assuming that the concentration for a given analyte applied to the day the sample was collected and all days since the previous sampling day. Results were computed using February 2017 through December 2017 as "first year" responses, February 2018 through December 2018 as "second year" responses, and February 2017 through December 2018 as "entire study" responses given that flow measurements did not begin until February 2017. Analyte concentrations prior to February 2017 were also evaluated to provide insight into early responses immediately after the fires.

2.6 | Data analysis

We used one-way analysis of variance (PROC GLM, SAS v9.4, 2002–2012) to compare differences in pre-fire vegetation characteristics, post-fire soil Oe + Oa horizon (humus layer) depth, and soil inorganic nitrogen concentration between burned and unburned plots, and differences in mortality and basal area loss between mesophytic and xerophytic trees in burned plots. All correlations among variables were computed using least squares linear regression except where noted.

3 | RESULTS

3.1 | Vegetation, soil, and plot-scale burn severity

Pre-fire vegetation characteristics were likely similar across plots in burned and unburned watersheds (Table 1). Mean pre-fire tree density and basal area across all plots, taken as the sum of live and fire-killed trees measured in 2017 for plots in burned watersheds, was 910.7 (±38.2 SE) stems ha^{-1} and 38.8 (±1.6) m^2 ha^{-1}, respectively. Of the 96 plots across all watersheds, 46 (48%) had evergreen shrubs, including 34 plots in burned watersheds (57% of burned plots) and 12 in unburned watersheds (33% of unburned plots). Mean pre-fire evergreen shrub density among plots with evergreen shrubs was 659.9 (±116.1) stems ha^{-1}. Differences between mean tree and shrub density and tree basal area among plots in corresponding burned and unburned watersheds were not significant (p > .05).

Burn severity varied considerably across plots in the burned watersheds. We found that 33% of the burned plots experienced high-severity burns, 42% moderate-high or moderate, and 10% low severity. Mean bole char height ranged from 0.05 m to 7.75 m (mean 1.15 ± 0.17 m) and the percentage of mineral soil exposure ranged from 0% to 65% (mean 13.7% ± 2.1%) among burned plots (Table S2). The surface Oi horizon (litter layer) was absent in all plots in burned watersheds, while Oi horizon depth ranged from 2.2 cm to 7.6 cm (mean 4.5 ± 0.2 cm) in plots in unburned watersheds. The mean Oe + Oa horizon (humus layer) depth was significantly greater (p = .0026) in plots in unburned watersheds (1.77 ± 0.19 cm) than in plots in burned watersheds (1.15 ± 0.10 cm). While we did not detect a significant difference in mean soil inorganic nitrogen (NH_4-N + NO_3-N) between samples collected in burned and unburned watersheds, the soil inorganic nitrogen concentration increased with burn severity within burned watersheds (R^2 = 0.29, p < .001) (Figure S2).

Mean tree mortality for all species and size classes in burned plots was 31.2% ± 3.2% in the first year after the fires and increased to 40.6% ± 3.6% in the second year (Figure 2a, Table S3). Small size class trees (<20 cm diameter) had the highest mortality rates regardless of their fire tolerance classification (Table S3). However, there was significantly greater mortality of mesophytic (mean 48.9% ± 4.2%) than xerophytic (mean 28.6% ± 3.8%) trees of all sizes in the second year (Figure 2a, Table S3). Mortality (top-kill) of evergreen shrubs ranged 0% to 100% among the 34 burned plots that had evergreen shrubs (mean 75.2% ± 6.5%), with only five of these plots having 0% evergreen shrub mortality.

Mean tree basal area loss over all species and size classes in burned plots was 2.88 ± 0.53 m^2 ha^{-1} (8.8%) in the first year after the fire and increased to 5.46 ± 0.91 m^2 ha^{-1} (16.9%) in the second year (Figure 2b, Table S4). Small tree sizes (<20 cm diameter) had the highest basal area loss regardless of fire tolerance classification. As a result of the differences in mortality and basal area loss between mesophytic and xerophytic tree fire tolerance classification, the mean forest tree species composition in burned plots by basal area changed from 42% mesophytic and 58% xerophytic before the fire to 38%/62% and 37%/63% mesophytic versus xerophytic in the first and second years after the fire, respectively (Figure 2b).

3.2 | Watershed-scale burn severity

RdNBR was significantly related to tree mortality, BA loss, and bole char height (r > 0.75, p < .001) which allowed us to scale plot-level measurements to the watershed using a non-linear regression model between RdNBR and composite burn severity index at the plot-scale (Burn Severity Index = 0.92 * ln(RdNBR) – 1.29, R^2 = 0.59, p < .001). The 10-fold cross validation resulted in a mean RMSE of 0.81 and mean R^2 of 0.63. The RdNBR below the Moderate category was quite close to that of the unburned watersheds (Table 1), suggesting that the lower severity burned areas could not be discerned by RdNBR one-year post-fire. We therefore combined the Low-Moderate and Low burn severity categories for the watershed scale burn severity assessment, resulting in four watershed-scale burn severity classes: High, Moderate-High, Moderate, and Unburned-Moderate (Table 2).

Based on the burn severity classified using RdNBR, the CA watershed at the Camp Branch Fire was more severely burned than IN or IR watersheds at the Tellico Fire (Table 1, Figure 1b). About 73% of the CA watershed was at least moderately burned, including 21% at high severity, while 57% of IN watershed was at least moderately burned with only 5% high severity (Table 1). Most of CA-TO watershed (95%) and about half of the IR watershed were at least moderately burned (Figure 1b and Table 1). Sixty-five percent of CA-TO watershed was classified as high burn severity, while 11% of the IR watershed was burned at high severity (Table 1). The high burn severity in CA-TO was generally located at high elevations in the watershed (Figure 1b).

3.3 | Stream water quantity responses

Using the long-term data from RG19 at the Coweeta Hydrologic Laboratory as a reference, annual precipitation (P) during the study period ranged from 1315 mm in 2016 to 2529 mm in 2018 (the sixth driest and the wettest years over the 84-year record, respectively), while 2017 (1610 mm) was the 25th driest (Figure S3A). Monthly P for September and October 2016, the months before and during the wildfires, was 113 and 110 mm less than the long-term mean P for those months (Figure S3A), respectively. Total P for September and October was the lowest consecutive two-month P over the period of record

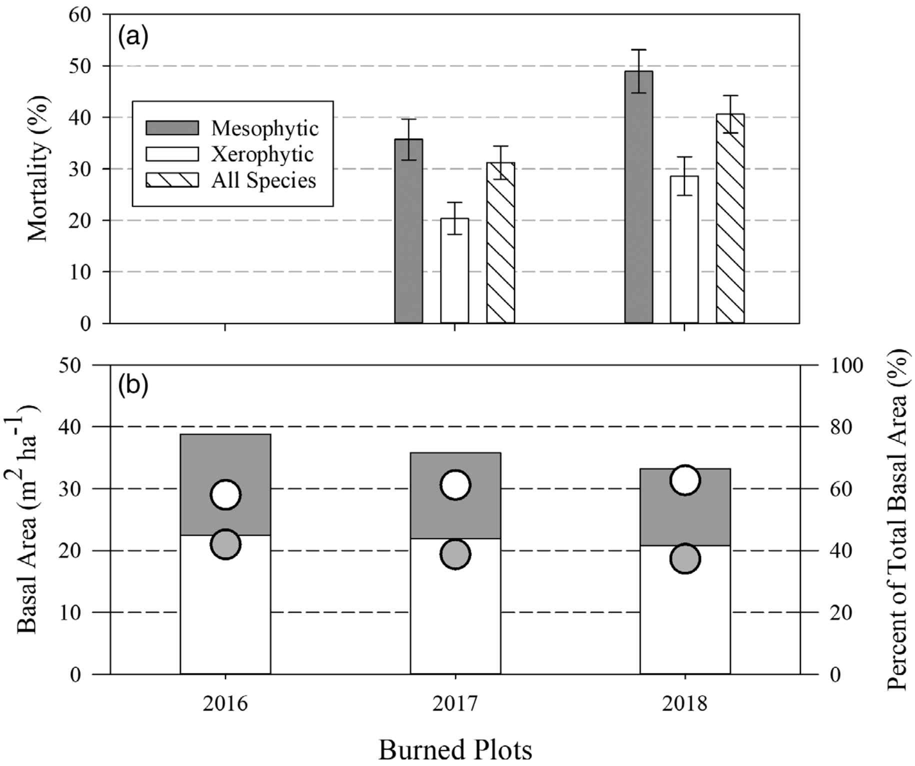

FIGURE 2 Mean (SE; error bars) tree mortality (a), and mean basal area of live trees (bars), and the mesophytic and xerophytic proportion of total basal area (circles) (b) across plots in the three burned watersheds (n = 60) in 2017 and 2018. Mean pre-fire tree basal area for 2016 calculated as the sum of live and fire-killed trees measured in 2017

and during this time was the longest consecutive number of days without rain (66 days from September 19 through November 23) (Figure S4). Monthly *P* was generally near or below the long-term mean until February 2018.

Patterns in water yield (*Q*) reflect the *P* patterns over the study period across all watersheds (Figure 3, Figure S3B) while differences in total *Q* (Q_{tot}) and stormflow *Q* (Q_{sf}) between burned and unburned watersheds varied over time and increased with burn severity. All burned watersheds except IN had greater Q_{tot} and Q_{sf} than corresponding unburned watersheds during the first year after the fires (Table 3, Figure 3). In the second year after the fires, only the most severely burned CA-TO watershed had greater Q_{tot} than corresponding unburned watersheds, while Q_{sf} was greater only in burned watersheds CA and CA-TO. Over the entire study period, the burned Camp Branch Fire watersheds, CA and CA-TO, had 30% and 39% greater Q_{tot} than corresponding unburned watersheds, respectively, while the burned Tellico Fire watersheds, IN and IR, had 26% and 5% less Q_{tot} than corresponding unburned watersheds, respectively (Figure 4a). Similarly, watersheds CA and CA-TO had 29 and 628% greater Q_{sf} than corresponding unburned watersheds, respectively, while the burned Tellico Fire watersheds, IN and IR, had 63 and 30% less Q_{sf} than corresponding unburned watersheds, respectively (Figure 4b). In the first year after the fires, baseflow *Q* (Q_{bf}) was greater in all burned watersheds than corresponding unburned watersheds with the exception of IN. In the second year after the fires, all burned watersheds had lower Q_{bf} than corresponding unburned watersheds. Over the entire study period, Q_{bf} was less in burned watersheds than corresponding unburned watersheds for CA-TO (−3%) and IN (−24%) but was 30% greater in the burned CA watershed and 7% greater in IR than in corresponding unburned watersheds (Figure 4c).

3.4 | Stream water quality responses

Stream water quality responses varied by analyte and burn severity extent in burned watersheds as discussed in subsequent sections. Example responses of select water quality analytes for the most severely burned CA-TO watershed and corresponding unburned watershed AR-UP are shown in Figure 5 and relations of these analytes with flow during the March 1–2, 2017 sampled storm are shown in Figure 6. Complete plots of all analytes across all watersheds are shown in Figures S5–S17.

3.4.1 | Sediment (TSS)

There was not a consistent relation between watershed-scale burn severity and TSS concentration and export during baseflow conditions, but peak storm TSS concentrations and export in the most severely burned CA-TO watershed at the Camp Branch Fire suggested a burn response (Figure 5). In the first year after the fire, mean baseflow TSS concentration for burned watersheds only exceeded unburned watersheds for IR (Figure S5), while first year TSS export exceeded unburned watersheds in both burned watersheds IN and IR at the Tellico Fire (Figure 7). In the second year and over the

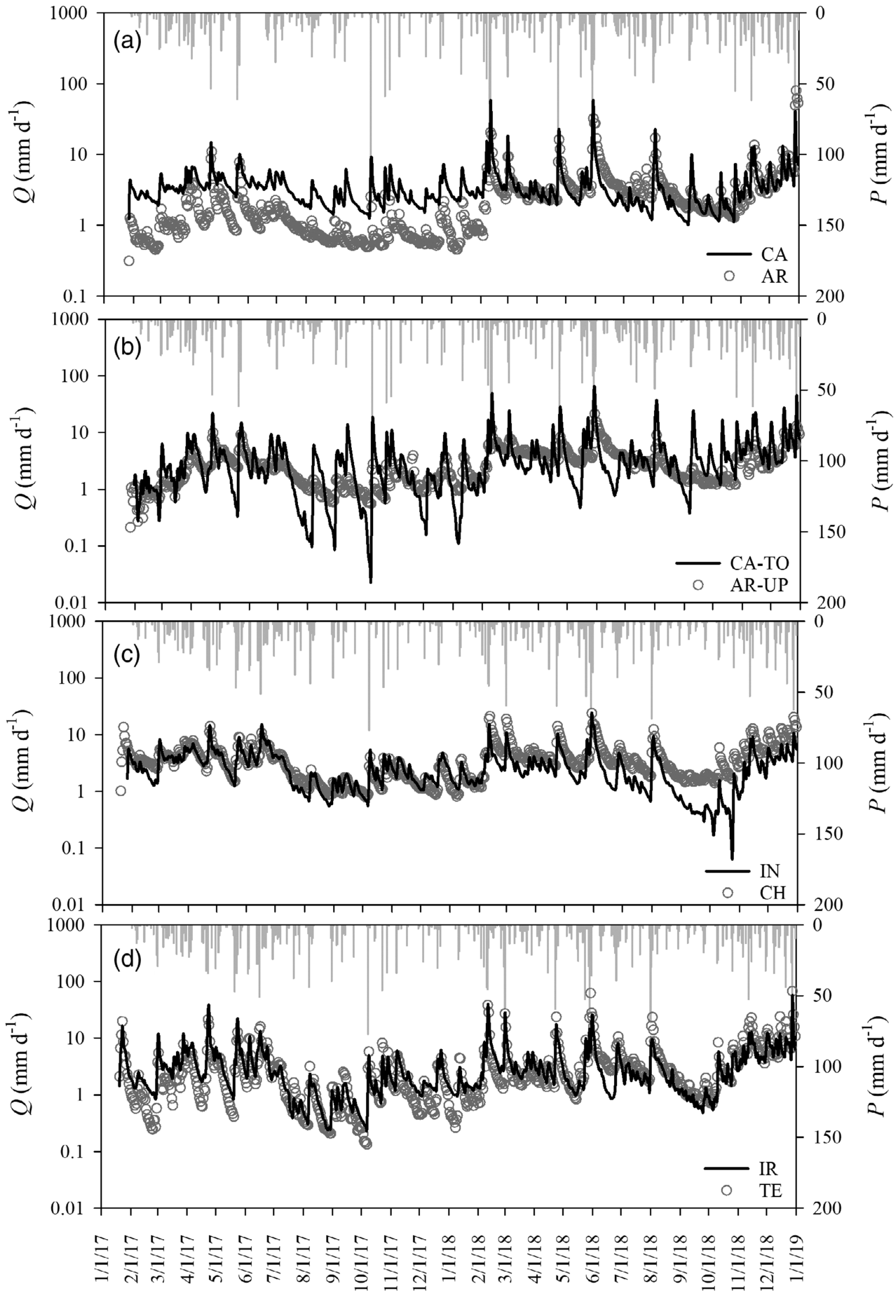

FIGURE 3 Daily Q for burned watersheds CA (a), CA-TO (b), IN (c), and IR (d) (lines) and associated unburned watersheds (circles) and daily P (bars)

entire study period, mean baseflow TSS concentration in burned watersheds CA and IR exceeded that of unburned watersheds (Figure S5), while TSS export was only greater in burned watershed CA (Figure 7). While mean baseflow TSS concentration for the most severely burned watershed CA-TO was 85% less than unburned watershed AR-UP over the entire study period (Table 4) and export was 96% less (Figure 7), CA-TO peak storm TSS concentrations were 2.7 to 168 times that of AR-UP across four of five sampled storms at these sites and was approximately equal for the fifth and last storm sampled on August 30, 2017 (Figure 5). Similarly, CA-TO TSS export during the five sampled storms was 2.4 to 82 times greater than AR-UP across four storms and was equal to that of AR-UP for the fifth and final sampled storm.

3.4.2 | pH, cations (Ca, K, Mg, Na), and anions (Cl, SO_4)

Cation and anion concentrations in burned watersheds were elevated immediately after the wildfires relative to later samples, and anion

TABLE 3 Summary of *P*, total *Q* (Q_{tot}), stormflow *Q* (Q_{sf}), and baseflow *Q* (Q_{bf}) for burned and unburned watersheds for the February 2, 2017–December 31, 2017, February 1, 2018–December 31, 2018, and February 1, 2017–December 31, 2018 periods

	Camp branch fire				Tellico fire			
	AR	CA	AR-UP	CA-TO	CH	IN	TE	IR
Burn/reference	Reference	Burned	Reference	Burned	Reference	Burned	Reference	Burned
February 1, 2017–December 31, 2017								
P at fire site (mm)	1,601	1,445	1,601	1,445	1,666	1,666	1,666	1,666
P at Coweeta (mm)	1,487	1,487	1,487	1,487	1,487	1,487	1,487	1,487
Q_{sf} (mm)	45	96	36	253	83	83	180	217
Q_{bf} (mm)	363	1,023	632	678	930	922	559	728
Q_{tot} (mm)	408	1,119	668	931	1,013	1,005	739	945
Q_{sf}/Q_{tot}	0.11	0.09	0.05	0.27	0.08	0.08	0.24	0.23
2/1/2018–December 31, 2018								
P at fire site (mm)	2,412	2084	2,412	2084	2,106	2,106	2,106	2,106
P at Coweeta (mm)	2,396	2,396	2,396	2,396	2,396	2,396	2,396	2,396
Q_{sf} (mm)	345	401	100	740	282	141	603	330
Q_{bf} (mm)	1,266	1,046	1,261	1,172	1,386	816	1,136	1,076
Q_{tot} (mm)	1,611	1,447	1,361	1912	1,668	957	1739	1,406
Q_{sf}/Q_{tot}	0.21	0.28	0.07	0.39	0.17	0.15	0.35	0.24
2/1/2017–December 31, 2018								
P at fire site (mm)	4,118	3,616	4,118	3,616	3,845	3,845	3,845	3,845
P at Coweeta (mm)	4,017	4,017	4,017	4,017	4,017	4,017	4,017	4,017
Q_{sf} (mm)	391	503	138	1,003	366	224	788	548
Q_{bf} (mm)	1,655	2,157	1941	1884	2,363	1789	1722	1847
Q_{tot} (mm)	2046	2,660	2079	2,887	2,729	2013	2,510	2,395
Q_{sf}/Q_{tot}	0.19	0.19	0.07	0.35	0.13	0.11	0.31	0.23

Note: In addition to *P* measured at the fire site, *P* measured at the Coweeta Hydrologic Laboratory RG19 is shown for reference. The corresponding reference for each burned watershed is shown in the column to the left of the burned watershed.

concentration and export over the study period in burned watersheds consistently exceeded that of corresponding unburned watersheds. For all burned watersheds, pH was 0.2 to 0.6 units lower than corresponding unburned watersheds during baseflow (Table 4) and differences in pH between burned and unburned watersheds were relatively constant over time (Figure S6). Baseflow and storm Ca, Mg, and K concentrations in burned watersheds CA, CA-TO, and IR were elevated during the first three to six weeks after the fires and in sampled storms, but subsequent concentrations did not exceed that of unburned watersheds for the Camp Branch Fire burned watersheds (Figures 5, Figures S7–S10). For example, while mean Ca concentration in the most severely burned CA-TO watershed over the entire study period was 41% less than unburned watershed AR-UP (Table 4), peak storm concentration in the March 1–2, 2017 storm exceeded that of AR-UP by 40% (Figure 5) and concentration was highly dependent on flow through the duration of the storm event (Figure 6). First-year cation export in burned than unburned watersheds generally exceeded that of unburned watersheds (Figure 7). Anion (Cl and SO_4) concentrations were also elevated in burned watersheds during the first three to six weeks of the study relative to subsequent samples at CA and CA-TO (Figures 5, Figures S11 and S12) and anion concentration and export in all burned watersheds exceeded that of unburned watersheds over the entire study period (Table 4, Figure 7). First year Cl and SO_4 concentrations were 19.9% to 66.5% and 38.2% to 195% greater, respectively, in burned than in corresponding unburned watersheds, however the magnitudes and differences in Cl concentrations between burned and unburned watersheds declined from the first to second year of the study. Differences in mean SO_4 concentration between burned and unburned watersheds were similar in the first and second years (Figure S12).

3.4.3 | Dissolved organic carbon (DOC) and particulate organic material (TVS)

The CA watershed had greater DOC concentrations and greater differences in DOC concentration relative to corresponding unburned watersheds than watersheds burned at greater or lesser high severity extent and greater concentrations than all unburned watersheds (Table 4, Figure 5, Figure S13). First-year DOC export in all but the

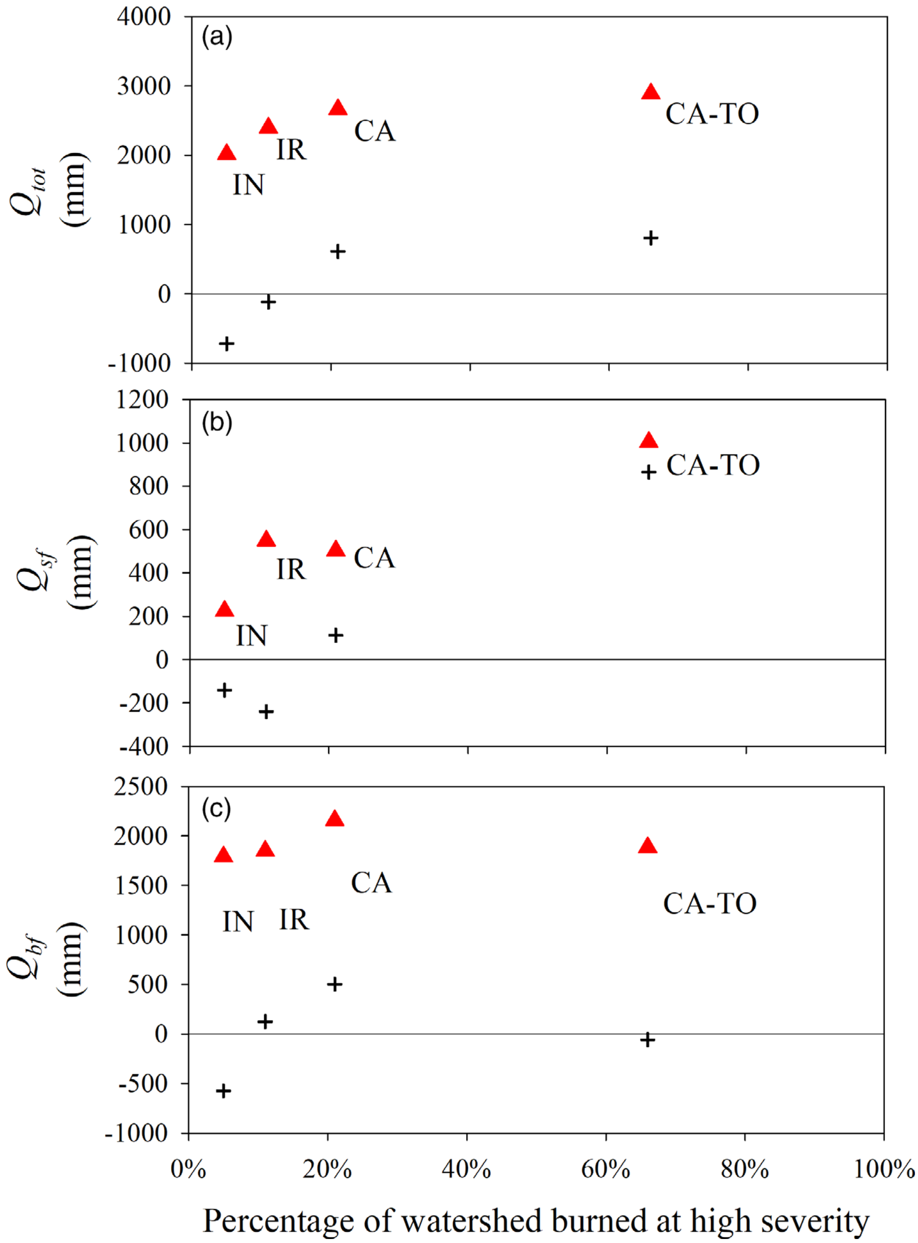

FIGURE 4 Total water yield (Q_{tot}) (a), water yield as stormflow (Q_{sf}) (b), and baseflow (Q_{bf}) (c) over the entire study period as a function of burn severity. Burned watersheds are shown with red triangles, the difference in Q in between burned and unburned watersheds are shown with black crosshair symbols. Burned watersheds CA and CA-TO in the Camp Branch Fire are compared to unburned watersheds AR and AR-UP, respectively. Burned watersheds IN and IR in the Tellico Fire are compared to unburned watersheds CH and TE, respectively

least severely burned watershed IN exceeded that of unburned watersheds by 15% (CA-TO) to 237% (CA) and exceeded unburned watersheds in burned watersheds CA and CA-TO at the Camp Branch Fire over the entire study period (Figure 7). Peak DOC concentration during sampled storms was similar between burned and unburned watersheds (Figure 5, Figure S13), but export during storms was generally greater in burned watersheds except the least severely burned IN watershed at the Tellico Fire. Mean baseflow TVS concentration and export were greater in burned watershed IR, otherwise TVS concentration and export in burned watersheds were generally less than corresponding unburned watersheds (Table 4, Figure 5, Figure S14). Peak storm TVS concentrations in the most severely burned CA-TO watershed were consistently greater than corresponding unburned watershed AR-UP (Figures 5 and 6), otherwise TVS concentrations during storms were similar between burned and unburned watersheds.

3.4.4 | Inorganic nitrogen (NH_4-N, NO_3-N) and phosphate (PO_4)

The NH_4-N concentrations at baseflow were generally very low (65–75% of watershed samples above detection limits) and were similar between burned and unburned watersheds (Table 4, Figure 5 and Figure S15). Similarly, only 7 to 47% of watershed baseflow PO_4 samples were above detection limits (Table 4, Figure 5 and Figure S16), but a larger proportion of samples collected in burned watersheds had detectable PO_4 concentrations than in unburned watersheds. At the

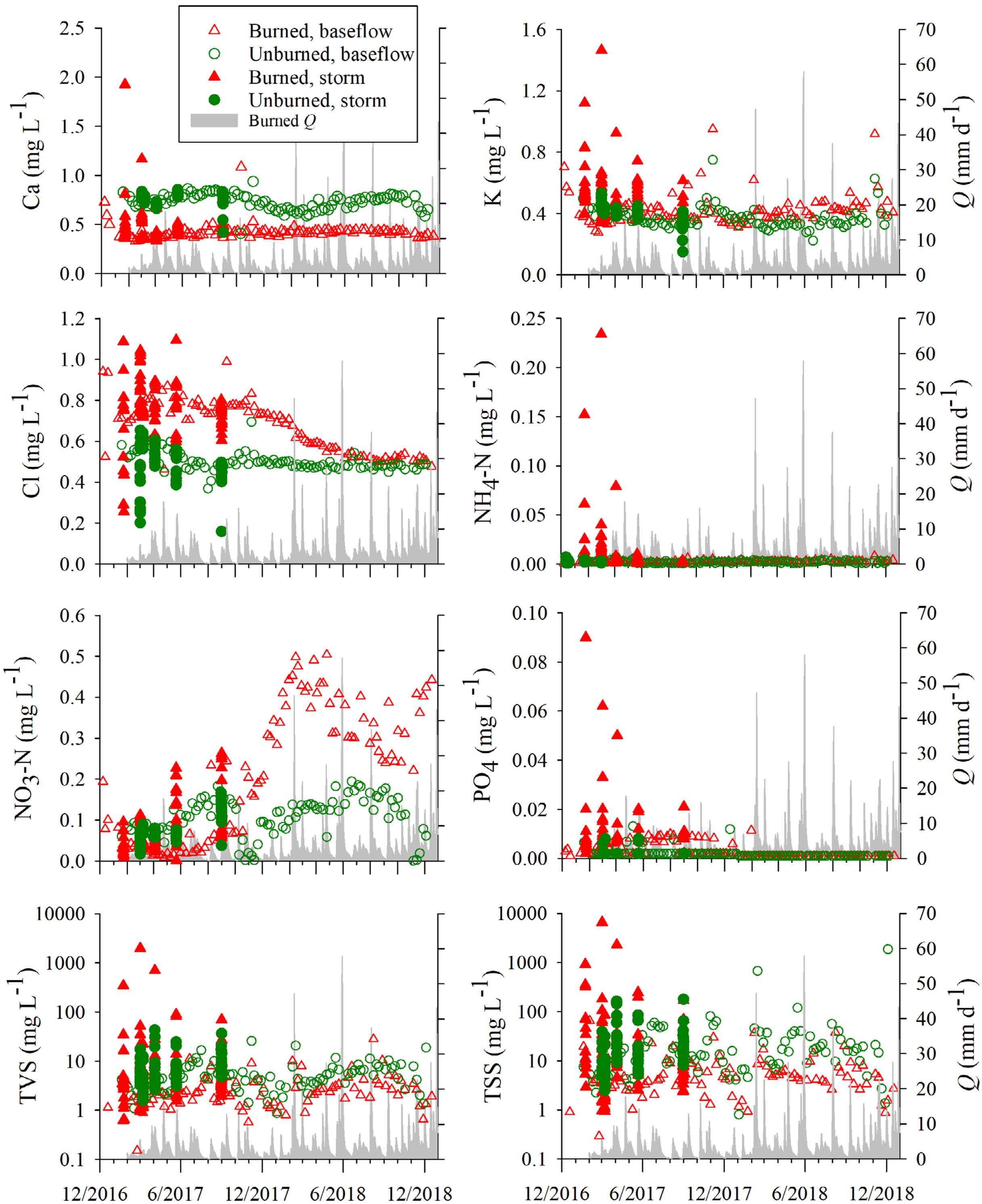

FIGURE 5 Example analyte concentrations for burned watershed CA-TO (red triangles) and corresponding unburned watershed AR-UP (green circles) under baseflow conditions (hollow symbols) and during storms (filled symbols). The daily Q for burned watershed CA-TO (grey) is also shown for reference. Data for other sites are shown in supplemental information

Camp Branch Fire, 24% and 34% of baseflow PO_4 samples were above detection limits for burned watersheds CA and CA-TO, respectively, greater than the 17% and 7% of samples greater than detection limits for corresponding unburned watersheds AR and AR-UP, respectively. Nearly all of the PO_4 detections in CA and CA-TO occurred during the first year after the fire. While baseflow NH_4-N and PO_4 concentrations were generally low, peak concentrations in sampled storms during the first year were elevated in burned watersheds CA and CA-TO relative to corresponding unburned watersheds (Figures 5 and 7, Figures S15 and S16). For example, CA-TO peak NH_4-N

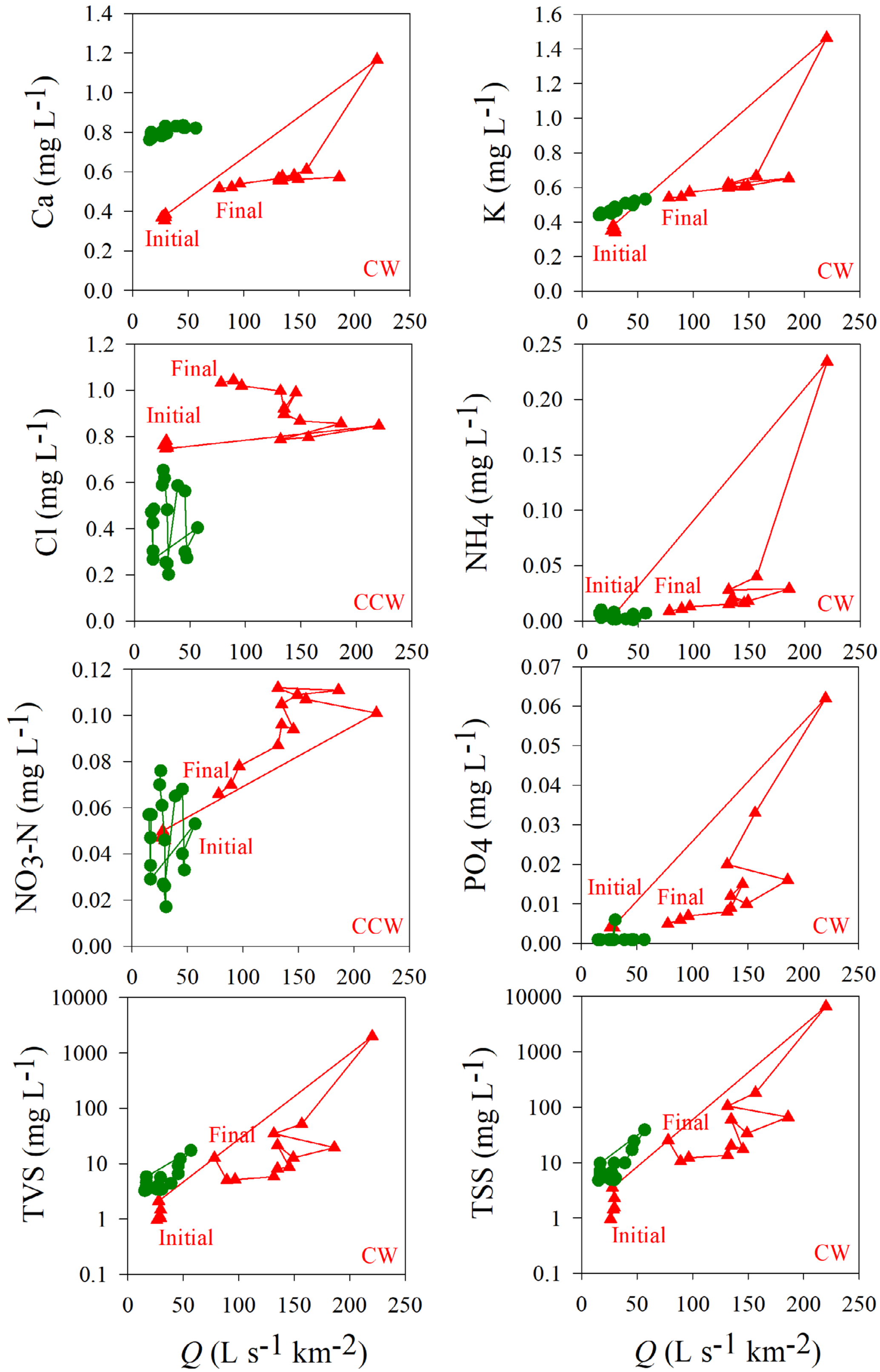

FIGURE 6 Example storm responses for select analyte concentrations in burned watershed CA-TO (red triangles) and corresponding unburned watershed AR-UP (green circles) during the 12.7 hr storm sampling on March 1–2, 2017. Initial and final concentrations for analytes for CA-TO are labelled, and the direction of the hysteretic relation between concentration and flow are labelled clockwise (CW) or counterclockwise (CCW). Precipitation during this two-day storm was approximately 50 mm and was the largest since the fire-ending event in November 2016

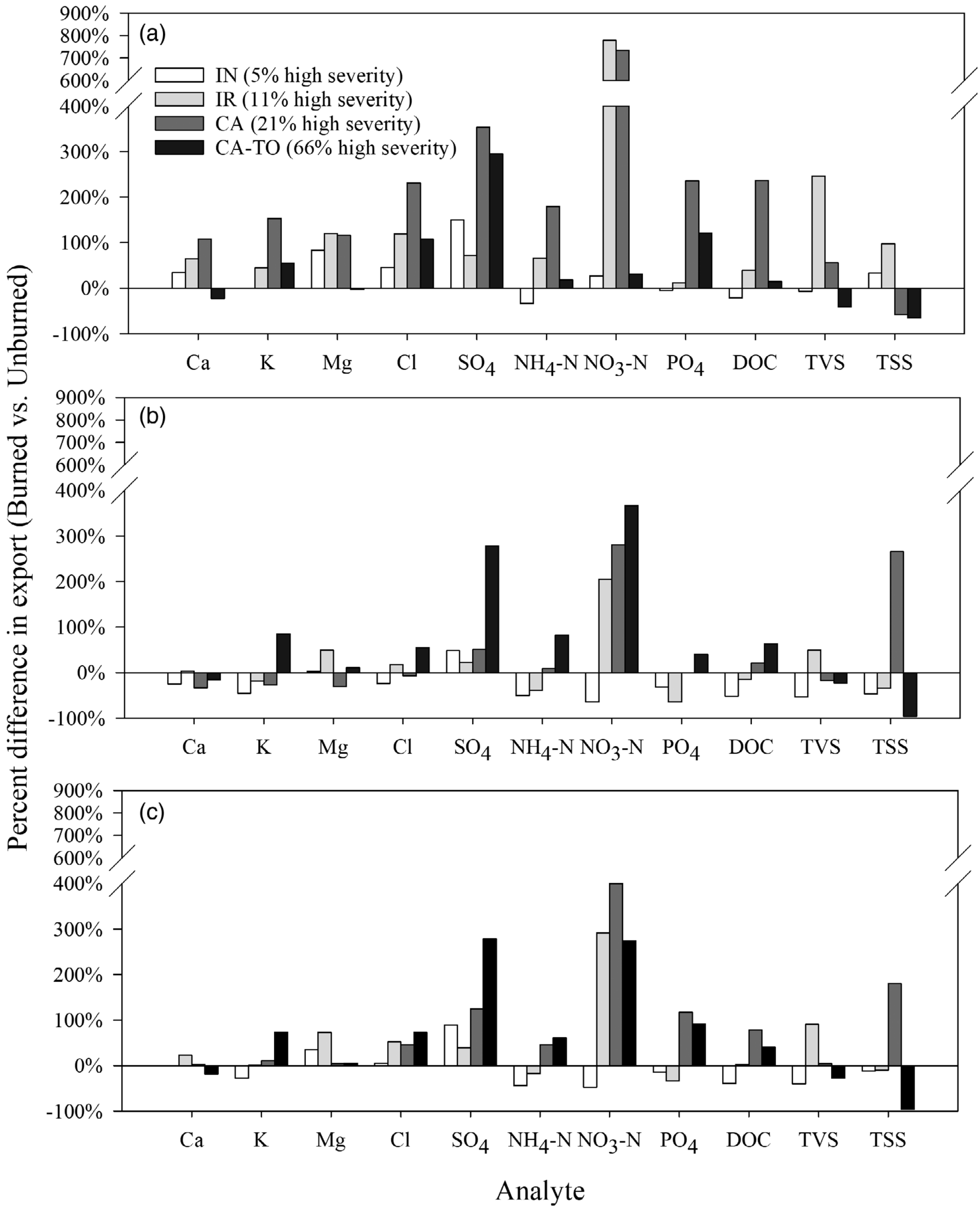

FIGURE 7 Percent difference in analyte export (kg ha^{-1}) between burned and corresponding unburned watersheds over February 1, 2017–December 31, 2017 (a), February 1, 2018–December 31, 2018 (b), and February 1, 2017–December 31, 2018 (c) in order of burn severity. Burned watersheds CA and CA-TO in the Camp Branch fire are compared to unburned watersheds AR and AR-UP, respectively. Burned watersheds IN and IR in the Tellico fire are compared to unburned watersheds CH and TE, respectively

concentration in the March 1–2, 2017 storm was 0.234 mg L^{-1} (23 times greater than AR-UP) while CA-TO peak PO_4 concentration was 0.062 mg L^{-1} (10 times greater).

Baseflow NO_3-N concentration and export were greater in more severely burned watersheds, and responses increased with greater extent of high burn severity (Figure 8). With the exception of the least severely burned watershed IN, mean NO_3-N concentration in burned watersheds over the entire study period exceeded that of unburned watersheds by 104% to 319% (Table 4, Figures 5, 8a, Figure S17). Mean NO_3-N concentration in the most severely burned watershed CA-TO increased from 0.088 mg L^{-1} in the first year to 0.362 mg L^{-1} in the second (+317%), 4.4% lower than corresponding unburned

TABLE 4 Mean (SD) analyte concentrations in baseflow samples collected from December 2016 through December 2018

	Camp branch fire				Tellico fire			
	AR	CA	AR-UP	CA-TO	CH	IN	TE	IR
Burn/ reference	Reference	Burned	Reference	Burned	Reference	Burned	Reference	Burned
N samples	100	103	94	99	102	102	102	102
pH	6.9 (0.1)	6.7 (0.1)	6.9 (0.1)	6.3 (0.1)	6.6 (0.1)	6.2 (0.1)	6.2 (0.1)	6.0 (0.1)
Ca^{2+}	0.84 (0.11)	0.64 (0.11)	0.73 (0.08)	0.43 (0.08)	0.47 (0.04)	0.64 (0.05)	0.50 (0.06)	0.63 (0.06)
K^{+}	0.53 (0.37)	0.46 (0.09)	0.37 (0.07)	0.43 (0.10)	0.38 (0.05)	0.38 (0.05)	0.28 (0.06)	0.28 (0.07)
Mg^{2+}	0.63 (0.07)	0.50 (0.07)	0.66 (0.05)	0.50 (0.07)	0.22 (0.02)	0.40 (0.02)	0.23 (0.03)	0.40 (0.03)
Na^{+}	1.26 (0.12)	0.91 (0.08)	1.09 (0.09)	0.69 (0.08)	0.89 (0.08)	0.95 (0.08)	0.72 (0.07)	0.83 (0.08)
Cl^{-}	0.47 (0.02)	0.54 (0.12)	0.50 (0.05)	0.68 (0.13)	0.41 (0.03)	0.56 (0.07)	0.38 (0.05)	0.59 (0.10)
SO_4^{2-}	0.66 (0.08)	1.09 (0.32)	0.58 (0.16)	1.70 (0.32)	1.06 (0.16)	2.86 (0.22)	2.09 (0.35)	2.98 (0.19)
NH_4^{+}-N	0.002 (0.001)	0.003 (0.001)	0.002 (0.001)	0.002 (0.001)	0.003 (0.002)	0.002 (0.001)	0.003 (0.003)	0.003 (0.002)
% < MDL	27%	27%	30%	32%	25%	35%	20%	22%
NO_3^{-}-N	0.02 (0.02)	0.07 (0.05)	0.11 (0.05)	0.21 (0.16)	0.02 (0.01)	0.02 (0.01)	0.04 (0.03)	0.14 (0.05)
% < MDL	13.0%	1.0%	3.2%	0.0%	25.5%	2.0%	0.0%	0.0%
PO_4^{3-}	0.002 (0.002)	0.003 (0.003)	0.002 (0.002)	0.003 (0.003)	0.004 (0.003)	0.004 (0.004)	0.004 (0.004)	0.003 (0.003)
% < MDL	83%	76%	93%	66%	54%	54%	53%	72%
DOC	0.49 (0.25)	0.62 (0.45)	0.58 (0.25)	0.48 (0.37)	0.39 (0.22)	0.32 (0.17)	0.39 (0.13)	0.40 (0.19)
TVS	4.9 (5.6)	4.0 (5.2)	5.7 (4.3)	3.2 (3.4)	3.0 (3.5)	2.3 (3.9)	2.2 (2.8)	5.2 (4.0)
TSS	38.6 (57.2)	55.5 (203.9)	47.9 (206.0)	7.2 (9.3)	20.2 (35.2)	14.8 (50.3)	10.0 (18.0)	15.6 (20.1)

Note: The corresponding reference for each burned watershed is shown in the column to the left of the burned watershed. pH units are S.U., all others are mg L^{-1}. For some analytes, sample concentrations were at times less than the Method Detection Limit (MDL). For those samples, we set the concentration to equal one-half of the MDL for that analyte before computing the mean concentration; the percentage of all samples less than the MDL is also shown.

watershed AR-UP in the first year and 183% greater in the second year. Total NO_3-N export and the difference in export between burned and unburned watersheds increased with extent of high burn severity over the entire study period (Figures 7, 8b).

NO_3-N concentration increased with increasing Q in the more severely burned watersheds under both baseflow and storm conditions, but generally decreased or did not change with Q in unburned watersheds (Figure 6). The slope of the relation between baseflow NO_3-N concentration and daily Q on the day the sample was collected was positive in the CA, CA-TO, and IR burned watersheds ($p < .005$) but was negative in the IN burned watershed ($p = .013$) (Figure 8c). Baseflow NO_3-N concentration was not significantly correlated to Q in any of the unburned watersheds ($p > .05$). Furthermore, the slope of the NO_3-N concentration vs. Q relation among burned watersheds increased with burn severity in burned watersheds (Figure 8c). Sampling during storm events also revealed a dependence of NO_3-N concentration on Q during some storms in both burned and unburned watersheds with NO_3-N concentration typically increasing in burned watersheds, and decreasing in unburned watersheds (Figure 6, Figure S17).

4 | DISCUSSION

The number of fires, area burned, and fire severity of the 2016 Southern Appalachian wildfires were unprecedented in recent times. Although the wildfires were caused by anthropogenic ignitions, the duration and timing of extremely warm and dry conditions contributed to the number, scale, and severity of the fires. Annual precipitation for 2016 was low (1315 mm), well below the 1800 mm mean annual precipitation recorded at the Coweeta Hydrologic Laboratory (Laseter et al., 2012). Consecutive dry days (CDD) and average maximum temperature in the fall (Sept-Oct) preceding the wildfire events were the highest on record (Figure S4). In addition, leaf fall from mid-October through November provided dry, fine, and highly flammable fuels (Varner, Kane, Kreye, & Engber, 2015) that facilitated the spread of fires over larger landscapes than would have been expected in spring or early summer months. Taken together, these factors provided a rare opportunity to study previously unreported vegetation, soil, water quantity, and water quality responses to wildfire in the humid deciduous forests of the southeastern United States.

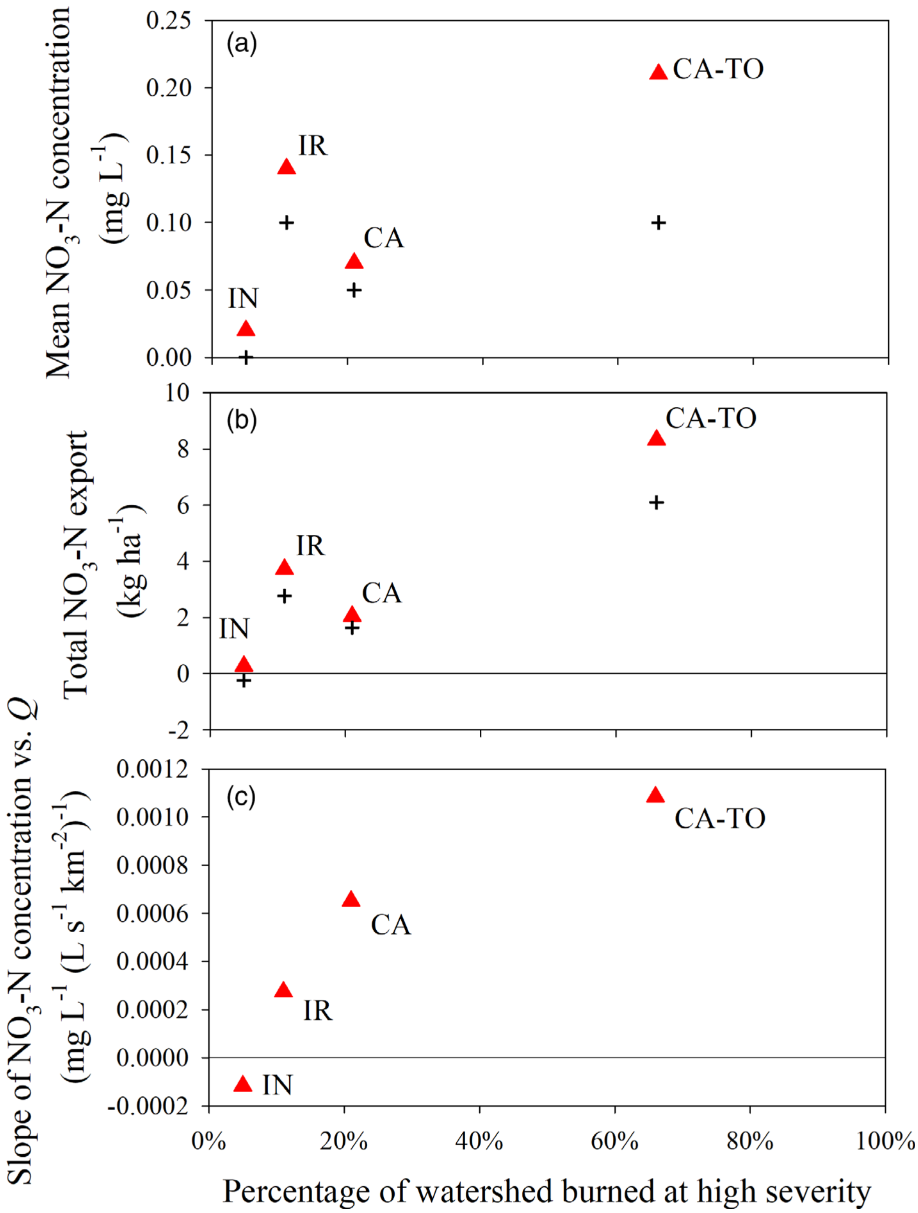

FIGURE 8 Mean NO_3-N concentration (a), total NO_3-N export over the study period (b), and the slope of the relation between NO_3-N concentration and mean daily Q on sampling days (c) for baseflow samples plotted as a function of burn severity. Burned watersheds are shown with red triangles, the difference in mean NO_3-N concentration and export between burned and unburned watersheds are shown with black crosshair symbols. Slopes of NO_3-N with Q were significant ($\alpha = .05$) for all burned watersheds (c), slopes were not significant for all unburned watersheds

4.1 | Vegetation responses

Tree mortality rates were much higher than those reported for low intensity prescribed fire in the southern Appalachians (Elliott & Vose, 2010) and support the notion that future changes in the frequency and severity of drought and fire could favour xerophytic species over the long term (Vose & Elliott, 2016). In addition to potential impacts on productivity and diversity, tree mortality and species composition changes resulting from wildfires could also have cascading effects on hydrology and water yield (Caldwell et al., 2016; Elliott et al., 2017) because mesophytic species, such as maples and poplars, have higher growing season evapotranspiration rates than xerophytic species such as oaks and hickories (Ford, Hubbard, & Vose, 2011; Ford, Laseter, Swank, & Vose, 2011). As a result, water yield may increase in burned watersheds due to the fire induced mortality and tree species composition changes, all else being equal.

Evergreen shrub mortality was also greater than previous studies of low severity prescribed fire. Rhododendron is typically difficult to burn (Zedaker, Harrell, & Pearce, 2010); thus, these results highlight the extremely dry conditions at the time of the fires. The reduction in rhododendron extent in burned watersheds could have ecosystem benefits over the long term, including increased biodiversity through herbaceous and tree seedling growth in the understory (Beckage, Clark, Clinton, & Haines, 2000; Clinton & Boring, 1994) and increased tree height, biomass, and productivity (Bolstad, Elliott, & Miniat, 2018; Nilsen et al., 1999). In addition, widespread loss of evergreen shrubs could result in increased water yield all else being equal; water use by these shrubs could comprise up to 10% of total transpiration of a southern Appalachian forest stand (Brantley, Ford, & Vose, 2013).

4.2 | Soil responses

Significant loss of the forest floor was observed in severely burned plots with implications for soil organic matter, nutrient cycling, erosion, and sedimentation. These results are consistent with what would be expected based on previous studies of prescribed fire in the southern Appalachians. For example, Knoepp et al. (2009) found that 82 to 91% of the Oi horizon mass and 26 to 46% of the Oe + Oa horizon mass was consumed in a low-to-moderate intensity prescribed fire, and while the prescribed fire effect on surface mineral soil inorganic N concentration was significant, these effects were short lived ending within one year after treatment. Knoepp, Vose, and Swank (2004) found that soil NH_4-N concentration in burned plots was significantly greater than that of unburned plots for three years after burning in a high intensity (i.e., high aboveground temperature) but low severity (i.e., low heat penetration into the soil) site preparation burn treatment consisting of clear felling all woody vegetation and allowing to dry for one to two months before burning. In the same study, Vose and Swank (1993) found that the Oe + Oa horizon remained largely intact and there was no soil movement to the stream (Swift, Elliott, Ottmar, & Vihnanek, 1993). In our study, significant forest floor loss (especially the Oe + Oa layer) and mineral soil exposure in severely burned plots likely increased surface soil erosion and transport to streams (see Section 4.4). Greater soil inorganic NO_3-N and NH_4-N concentration in severely burned plots suggests increased nutrient availability to support vegetation regrowth, but elevated stream NO_3-N concentrations in burned watersheds indicates increased N mobilization as well.

4.3 | Water quantity

The overall water yield responses to the wildfires were consistent with what has been observed from forest harvesting experiments in the southern Appalachians. Previous work in the region has shown that a minimum of approximately 20% of the watershed would need to be cut to detect a water yield increase (Douglass & Swank, 1972). Similarly, in a north American study, Hallema et al. (2018) identified 19% as a critical burn area ratio threshold above which wildfire impacts on river flow could be detected. In this study, greater water yield was only observed in watersheds with more than 20% of their drainage area burned at high severity including the CA (21% high severity) and CA-TO (65% high severity) watersheds at the Camp Branch Fire.

While the water yield responses to the wildfires were consistent with the literature with respect to the proportion of watershed area burned at high severity, the magnitude of the difference in water yield between burned and unburned watersheds was greater than we would have expected. The burned CA watershed had 321 mm yr^{-1} (30%) greater annual water yield than corresponding unburned watershed AR, while CA-TO had 422 mm yr^{-1} (39%) greater annual water yield than corresponding unburned watershed AR-UP, with annual water yield estimated by computing mean daily Q_{tot} over the 334 days from February through December and then multiplying by 365 days. By comparison, a watershed that was entirely clearcut (using cable yarding) in a paired watershed experiment within Coweeta Hydrologic Laboratory experienced a 266 mm (28%) increase in water yield in the first year after harvest (Swank, Knoepp, Vose, Laseter, & Webster, 2014). Using published regressions of first-year water yield increase versus percent change in basal area based on data from paired-watershed harvesting experiments in the Appalachian region (Douglass & Swank, 1972), we would expect the increase in water yield for burned watersheds CA and CA-TO watersheds to be 34 and 183 mm yr^{-1}, respectively, 68% and 56% less than what we observed.

The observed water yield responses suggest that the impacts of severe fires on hydrological processes differ from that of an equivalent amount of forest cutting. Part of this difference in response could be explained by fire-induced reduction or elimination of the forest floor. While we did not directly measure the effects of fire on soil infiltration and surface runoff, we can infer that observed effects on forest floor thickness and exposed mineral soil may have decreased infiltration and increased overland flow. There are few studies of the effects of severe fire on soil properties in the humid eastern United States; however, Robichaud and Waldrop (1994) found that high severity burn plots had 5 times greater surface runoff than low severity burn plots in a prescribed fire in the southern Appalachians. Our high severity burn plots had similar forest floor losses as reported in Robichaud and Waldrop (1994), thus surface runoff likely increased and contributed to greater stormflow in the most severely burned watersheds. In addition to forest floor losses, fire-induced changes in soil hydraulic properties may have played a role in the observed differences in water yield (Ebel & Moody, 2017; Matosziuk et al., 2020). Lastly, although we matched burned and reference watersheds as closely as possible, the larger than expected water yield responses could also be related to unaccounted for (or unknown) variation in watershed characteristics that influence water yield (Caldwell et al., 2016; Swift et al., 1988).

4.4 | Water quality

The Fall 2016 wildfires were unprecedented in their extent and severity and water quality was affected, but in complex ways not observed in prior work on prescribed fires in the southeastern United States (Clinton et al., 2003; Elliott et al., 2012; Elliott & Vose, 2005; Knoepp et al., 2009; Vose, Swank, Clinton, Knoepp, & Swift, 1999). For example, TSS concentration and export were greater in samples collected during storm events in the more severely burned watershed CA-TO (up to 6571 mg L^{-1}) relative to corresponding unburned watershed AR-UP, but TSS was not consistently greater in less severely burned watersheds during storms or in any burned watershed under baseflow conditions regardless of burn severity. Robichaud and Waldrop (1994) reported 42 times greater sediment loss during storms from plots burned at high severity than low severity in the southern Appalachians, although these plots were relatively small (22.5 m^2) and the results were unreplicated. It is also possible that there was erosion and sediment movement to and within streams prior to our sampling as a result of the ~50 mm fire-ending precipitation event (approximately two weeks before our sampling began). These studies demonstrate that while fire-induced forest floor loss and exposed mineral soil increase surface soil erosion potential, the impacts to streams can be highly variable and are dependent on connected sediment pathways from burned areas to the stream. Further, disturbance-related erosion and sediment delivery to and transport in streams are episodic and it may take considerable time (longer than the duration of this study) for eroded sediment to reach downstream sampling locations (James, 2013). Elevated TSS concentrations associated with wildfire can also increase the concentration and export of other sediment-bound pollutants of concern. For example, in a 5.6 km^2 forested watershed in the southeastern United States burned by a low severity wildfire, Jensen et al. (2017) found elevated levels of particulate mercury per unit TSS for the first eight months post fire. Taken together, these results suggest that there is elevated risk of localized erosion and sedimentation of streams draining severely-burned southern Appalachian watersheds that could impact aquatic ecosystems and water supply (Bladon et al., 2014; Hohner, Rhoades, Wilkerson, & Rosario-Ortiz, 2019; Smith et al., 2011).

Consistent with previous studies in the western United States (Bodi et al., 2014; Neary et al., 2005; Smith et al., 2011), we found that concentrations of cations (Ca, K, Mg) and anions (Cl and SO_4) were elevated during the first three to six weeks after the fires in the more severely burned watersheds and anion concentrations remained elevated through most of the study. Knoepp et al. (2004) documented increases in soil exchangeable Ca, K, and Mg concentrations and soil pH in high intensity low severity site preparation burn plots in the southern Appalachians, however cation concentration increases were not observed in a stream draining the site at baseflow. Overall, the fire-induced effects on cation and anion concentrations in our study were temporary and within applicable North Carolina water quality standards for trout streams (Title 15A of the North Carolina Administrative Code subchapter 02B), and thus would not be expected to result in water quality issues downstream.

Concentration and export of DOC and particulate organic matter (as TVS) were affected by the fires as reported in previous work in the western United States (see reviews by Hohner et al., 2019; Rust et al., 2018; Smith et al., 2011). Similar to Rhoades et al. (2019), watersheds with intermediate levels of high severity burn extent (i.e., CA, 20% high severity) had greater DOC concentrations and export than watersheds burned at greater or lesser high severity extent. Jensen et al. (2017) also found elevated DOC concentrations (accompanied by elevated mercury concentrations) across the flow regime in a forested watershed burned at low severity (8.5% high severity extent) in the southeastern United States. This non-linear relationship between high burn severity extent and DOC concentration and export has been attributed to unburned and partially charred organic material that remains on the forest floor following less severe wildfires that becomes sustained carbon inputs to streams compared to catchments burned at greater high severity extent (Hohner et al., 2019). Concentration and export of particulate organic matter (as TVS) in burned watersheds at baseflow were generally less than corresponding unburned watersheds, but peak TVS concentration during storms was greater in the most severely burned CA-TO watershed than the corresponding unburned watershed AR-UP. Elevated DOC concentrations associated with intermediate burn severity extent can have implications for water treatment due to the formation of disinfection byproducts (Emelko, Silins, Bladon, & Stone, 2011; Hohner et al., 2019), while reductions in DOC concentration and export in watersheds with greater high burn severity extent could alter trophic resources in downstream aquatic ecosystems (Hall & Meyer, 1998; Webster & Meyer, 1997).

We detected inorganic nitrogen (NH_4-N and NO_3-N) and PO_4 responses previously not observed in prescribed fire studies in the region. While NH_4-N and PO_4 concentrations were generally low across all watersheds, concentration during storm events at the more severely burned Camp Branch Fire burned watersheds CA and CA-TO suggest NH_4-N and PO_4 losses that were not observed in corresponding unburned watersheds or watersheds burned at lower high severity extent. Unlike our study, previous studies of forested southern Appalachian watersheds showed little NH_4-N and PO_4 response following disturbance (Clinton & Vose, 2006; Elliott & Vose, 2005; Knoepp & Swank, 1993; Swank & Vose, 1997; Swank, Vose, & Elliott, 2001). The difference in NH_4-N and PO_4 responses between this study and previous work in prescribed fires could be related to the higher overall burn severity associated with the wildfires and/or the larger extent of high burn severity within the watershed drained by the sampled stream.

Nitrate was among the most responsive of the water quality analytes we evaluated. Nitrate is a sensitive indicator of forest disturbance because stream NO_3-N concentration is generally very low in undisturbed forest watersheds, thus small changes in NO_3-N concentration suggest large disturbance-induced changes in internal nitrogen cycling (Swank & Vose, 1997). Nitrate concentration and export under baseflow conditions were generally greater in burned than unburned watersheds and increased with greater high burn severity extent. Nitrate concentration in unburned and less severely burned watersheds was consistent with reference watersheds previously studied in the southern Appalachians (Clinton & Vose, 2006; Elliott & Vose, 2005; Swank & Vose, 1997; Webster, Stewart, Knoepp, & Jackson, 2018). However, the magnitude of NO_3-N responses in the more severely burned CA-TO clearly exceeded that of previous prescribed burn studies in the region (Clinton et al., 2003; Elliott & Vose, 2005; Knoepp & Swank, 1993; Vose, Laseter, & McNulty, 2005), and even exceeded whole-watershed clearcutting experiments (Swank et al., 2001; Webster, Knoepp, Swank, & Miniat, 2016). For example, mean annual NO_3-N concentration in a clearcutting experiment at the Coweeta Hydrologic Laboratory peaked at 0.1 mg L^{-1} (+0.098 mg L^{-1} over expected) in the third year after harvesting (Swank et al., 2001; Webster et al., 2016). By comparison, second-year mean stream NO_3-N concentration in the most severely burned watershed in our study was 0.363 mg L^{-1} (+0.238 mg L^{-1} greater than the corresponding unburned watershed), much greater than the response in the clearcut watershed, and consistent with NO_3-N responses to wildfires in the western United States (Neary et al., 2005; Rhoades et al., 2019; Rhoades, Entwistle, & Butler, 2011; Smith et al., 2011). In addition, NO_3-N concentration in burned watersheds increased with increasing streamflow under baseflow and stormflow conditions whereas concentration generally remained the same or decreased with increasing streamflow in unburned watersheds, a finding supported by soil inorganic nitrogen concentration measurements over a range of burn severity (Figure S2). Similar increasing flow dependence of NO_3-N concentration with increasing disturbance severity has been observed in forested southern Appalachian watersheds (Webster et al., 2016; Webster et al., 2018). While NO_3-N concentrations in severely burned watersheds are well below the EPA drinking water standard of 10.0 mg L^{-1} (U.S. Environmental Protection Agency, 2018), increased loading of NO_3-N in addition to NH_4-N and PO_4 as a result of wildfire to downstream receiving waters could contribute to eutrophication and associated water quality impairment of water supply reservoirs when a large proportion of the watershed is burned at high severity.

4.5 | Looking ahead

Future projections suggest that severe drought, such as what was experienced in the Fall of 2016, will become more frequent and prolonged over the 21st century (U.S. Global Change Research Program, 2017). Drought, coupled with increasing air temperatures and decreasing relative humidity will increase wildfire potential, increase the length of fire seasons, and will limit the number of days in which prescribed fire can be used as a tool to manage fuel loads (Liu et al., 2013; Mitchell et al., 2014). For example, Liu et al. (2013) projected that fire season length in the Appalachians could increase by one month in 2041–2070 compared to 1971–2000 while the change in autumn wildfire potential as measured by the Keetch-Byram Drought Index (KBDI) in the region was the largest projected increase in the eastern United States. As our data suggested, increased fire frequency and severity could alter forest tree species composition by favouring fire-tolerant xerophytic species (e.g., oaks and hickories) over mesophytic species (e.g., poplars and maples) (Vose & Elliott, 2016), with potential indirect implications for water yield (Caldwell et al., 2016; Ford, Hubbard, & Vose, 2011; Ford, Laseter, et al., 2011). In light of these projections and the results of this study, it will become increasingly important to manage fuel loads through prescribed fire within the increasingly restrictive constraints associated with climate change to minimize negative impacts on water quantity and quality.

Assessing the hydrologic effects of wildfire using traditional approaches is challenging because it is rare to have a record of measurements before and after a wildfire event, much less long-term paired burn and unburned reference watersheds (Hallema et al., 2017). This is particularly true in the southern Appalachians where wildfires are typically small and infrequent. Ideally, a wildfire-based hydrologic study would consist of a paired watershed experimental design (Wilm, 1944) with water quantity and quality measurements before and after the fire to truly isolate the effect of the fires. In our study, we were careful in matching burned watersheds with similar unburned watersheds, but we cannot determine how (or if) differences in watershed characteristics influenced the observed effects of the wildfire on water quantity and quality. Despite these uncertainties, the patterns and magnitude of post-burn watershed responses are consistent with what would be expected following high severity fires which gives us confidence in the overall study design and our interpretation of response mechanisms. Future studies including hydrologic modelling of these watersheds and/or paired watershed experiments could help isolate the burn response of high severity wildfires from other potentially confounding watershed characteristics and to identify and quantify the specific mechanisms that drive that response.

5 | CONCLUSIONS

Our study provides new information describing watershed-scale vegetation, soil, water quantity, and water quality responses to severe wildfire in the humid deciduous forests of the southeastern United States. We found that high tree mortality coupled with forest floor loss in watersheds burned at high severity led to increases in water yield, stormflow, and concentration and export of sediment and nutrients relative to comparable unburned watersheds. While previous studies of low to moderate severity prescribed fires in the southern Appalachians showed negligible water quality effects, the results of this study suggest potential for elevated short-term risk of high stormflows and water quality issues where a large proportion of a watershed is burned at high severity.

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DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding authors upon reasonable request.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of this article.

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