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Removing riparian *Rhododendron maximum* in post-*Tsuga canadensis* riparian forests does not degrade water quality in southern Appalachian streams

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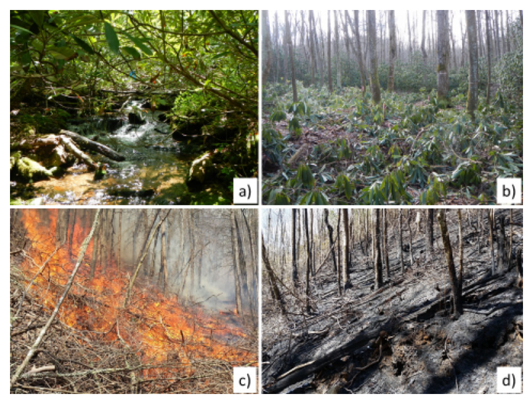
HIGHLIGHTS

- Removing *Rhododendron maximum* from degraded riparian areas may be a management option to restore structure and function.
- Rhododendron subcanopy removal, with and without low severity fire, was examined over three years post-treatment.
- Stream pH, acid neutralizing capacity, and nitrate-nitrogen were lower than expected for several months after treatment.
- We found greater dissolved organic carbon and aluminum concentrations than expected.
- Because water quality was not lowered, removing rhododendron with or without prescribed fire is a viable management solution.

GRAPHICAL ABSTRACT



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ABSTRACT

In the past century, the evergreen woody shrub, *Rhododendron maximum*, has experienced habitat expansion following foundational tree species die-off in eastern US deciduous forests. Rhododendron can potentially alter stream chemistry, temperature, trophic dynamics, and in-stream decomposition rates, given its dominance in riparian areas. Here we conducted two operational-scale (3 ha) riparian treatments that removed rhododendron through cutting alone (CR, canopy removal), or removing both the rhododendron canopy and forest floor using cutting and prescribed fire (CFFR, canopy and forest floor removal). We expected that rhododendron shrub removal, with or without soil organic horizon removal, would increase soil nutrient availability and subsequently alter stream pH, acid neutralizing capacity (ANC), inorganic nitrogen (NO_3-N, NH_4-N), total dissolved inorganic nitrogen, dissolved organic carbon (DOC), calcium (Ca), potassium (K), and magnesium (Mg). We hypothesized that responses would occur more quickly in the CFFR treatment. Treatments reduced shrub-, but not tree basal area. Treatments lowered soil N, but not C. Stream chemistry responses to treatments varied between CR and CFFR and were transient, generally with pH, N, and some cations declining, and aluminum (Al) and DOC showing a pulse increase. By removing rhododendron, the remaining deciduous trees likely accelerated N uptake as soil moisture availability increased. This could partially explain why we observed lower than expected stream nutrients (NO_3-N, Ca, and Mg) after treatments. Initial rhododendron slash on the forest floor coupled with incomplete consumption of the O-horizon on the CFFR treatment likely elevated DOC in the upper soil horizons and

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mobilized Al. From a management perspective, using these treatments to restore structure and function to riparian forests in the wake of eastern hemlock mortality, with or without fire, would most likely not result in short-term diminished water quality that is common when overstory trees are harvested and may even lower stream NO_3-N concentrations long term.

1. Introduction

In the past century, both eastern (Wang et al., 2016) and western (Ansley and Rasmussen, 2005) forested lands in the US have experienced expansions of understory woody shrubs. In moist cove and riparian habitats in southern Appalachian forests of the eastern US, the dominant understory species is rosebay rhododendron (*Rhododendron maximum* L.), a native evergreen shrub. In the past century, rhododendron has experienced a habitat expansion, due to the die-off of American chestnut (*Castanea dentata* (Marsh.) Borkh.) in the early 20th century (Elliott and Swank, 2008; Elliott and Vose, 2012). More recently, it has increased its growth following the decline of eastern hemlock (*Tsuga canadensis* (L.) Carrière) due to hemlock woolly adelgid (*Adelges tsugae* Annand) infestation (Ford et al., 2012; Dharmadi et al., 2019). Landscape-level studies have shown that where evergreen shrubs are present in the understory, forests trees are, on average, 6 m shorter than where they are absent (Bolstad et al., 2018). Rhododendron is an impactful species in these forests as it is highly shade-tolerant, forms a dense shrub layer that strongly attenuates light incident on the forest floor (Clinton, 2003; Elliott and Miniat, 2018), has little to no herbaceous or tree seedling cover below its canopy (Beckage et al., 2000; Elliott and Miniat, 2018), and negatively affects tree seedling densities where they can establish (Hille Ris Lambers and Clark, 2003; Dharmadi et al., 2019). Taken together, these studies suggest that riparian forest structure may be fundamentally altered in the wake of eastern hemlock decline and the concomitant expansion of rhododendron. This has prompted forest managers to advocate for aggressive management strategies involving the removal of rhododendron from riparian areas impacted by hemlock die-off to promote forest recovery (Elliott and Miniat, 2018).

From previous research in the southern Appalachians, we know that rhododendron has unique functional attributes that alter riparian processes. Rhododendron dominates plant-soil interactions in these forests by suppressing decomposition rates (Hunter et al., 2003; Ball et al., 2008; Strickland et al., 2009) and immobilizing N and other nutrients in complex organic compounds that are preferentially utilized by rhododendron's own mycorrhizal symbionts (Wurzburger and Hendrick, 2007, 2009). This immobilization of nutrients, along with attenuation of light, inhibits recruitment of hardwood tree seedlings, thereby influencing forest dynamics (Beckage et al., 2000; Nilsen et al., 2001; Clinton, 2003; Ford et al., 2012; Dharmadi et al., 2019).

Riparian forests also affect stream organisms and ecosystem properties and processes (Snyder et al., 2003; Siderhurst et al., 2010; Webster et al., 2012; Dudley et al., 2020a; Tolkkinen et al., 2020). Thus, the loss of riparian eastern hemlock with a resulting community dominated by rhododendron can potentially alter stream pH, chemistry, temperature, stream flow, quality and quantity of detrital inputs, trophic dynamics, and in-stream rates of decomposition. For soil pH and nutrient chemistry, the presence of evergreen species, particularly rhododendron, is an indicator of acidic coves in the southern Appalachians (see review, Elliott et al., 2014). Rhododendron produces long-lived, sclerophyllous leaves that are composed of lignin and tannins (and other polyphenols), which are highly recalcitrant (Monk et al., 1985; Strickland et al., 2009). Its litter is slow to decompose; and, thus a thick recalcitrant forest floor (soil organic horizon) develops under the rhododendron canopy (Cofer et al., 2018) where much of the nitrogen and cations are bound in complex organic compounds that are unavailable to competitors (Wurzburger and Hendrick, 2007). Tannins found in rhododendron leaf litter bind with organic nitrogen (proteins) in the soil and make this nitrogen source available to rhododendron and unavailable to overstory trees (Wurzburger and Hendrick, 2009). This creates a positive-feedback for rhododendron growth and recruitment and a negative-feedback for other species' growth and recruitment. Subsequently, the effect of rhododendron litter chemistry and decomposition rates also affects nutrient movement to streams, potentially lowering pH and anion neutralizing capacity (ANC). Alternatively, shifts in composition from evergreen to deciduous species may, over time, shift the recalcitrant litter pool to a more labile pool, one characterized by leaf tissue with higher nutrient and lower lignin contents, a pool that would decompose much more rapidly (Cornelissen et al., 2001).

Science-based restoration methods are needed to aid land-managers in the recovery of degraded riparian forests (Messier et al., 2015; Webster et al., 2018), particularly those once dominated by eastern hemlock with a remaining dense rhododendron subcanopy (Roberts et al., 2009; Cofer et al., 2018; Elliott and Miniat, 2018). We conducted a rhododendron and organic soil horizon removal experiment in these affected riparian corridors at the stream reach-scale (3 ha). We expected that: 1) rhododendron removal, with or without soil organic horizon removal, would increase nutrient availability as nutrient pools became more labile; 2) increased nutrient availability in riparian soils along with increased nutrient movement would alter stream pH, ANC, inorganic nitrogen (NO_3-N, NH_4-N), total dissolved inorganic nitrogen (TDN), dissolved organic carbon (DOC), calcium (Ca), potassium (K), and magnesium (Mg); and 3) these changes would occur more rapidly where the soil organic horizon was also removed using prescribed fire.

2. Methods

2.1. Study area

We selected three, perennial, 2nd order streams within the WhiteOak Creek (WOC) watershed (35°20′ N latitude, 83°58′ W longitude), located within the Blue Ridge Physiographic Province, near the southern end of the Appalachian Mountain chain (Fig. 1). For each stream reach, sampled areas were located along a 300-m (length) × 50-m (width) transect on each side of the stream as the treated area (3 ha). For these stream reaches, dead eastern hemlock comprised 37–47% of the overstory basal area and the rhododendron subcanopy was dense (Table 1, Fig. 2a). The three stream reaches were located on Holloway Branch, Split Whiteoak Branch, and Kit Springs (Fig.1). Across reaches, slopes are moderate (30–60%) and elevation is 1160–1390 m. Mean annual precipitation is 1900 mm and mean annual temperature is 10.8 °C, and the growing season is mid-April through mid-October. Sites are located on gneiss and granite bedrock with soils mapped in the Plott, Edneyville, and Chestnut soil series (all Inceptisols) (Thomas, 1996).

2.2. Experimental design

We used a Before-After/Control-Impact experimental design (BACI) (van Mantgem et al., 2001) with three treatments implemented at the stream reach-scale at WOC. The experimental treatments were designed to remove only the rhododendron subcanopy (hereafter, CR), or to remove the rhododendron subcanopy and the soil organic horizon (hereafter, CFFR). We also included a no removal, or reference stream that was left untreated (reference, hereafter, REF). The CR and CFFR treatments included cutting rhododendron by hand, following which

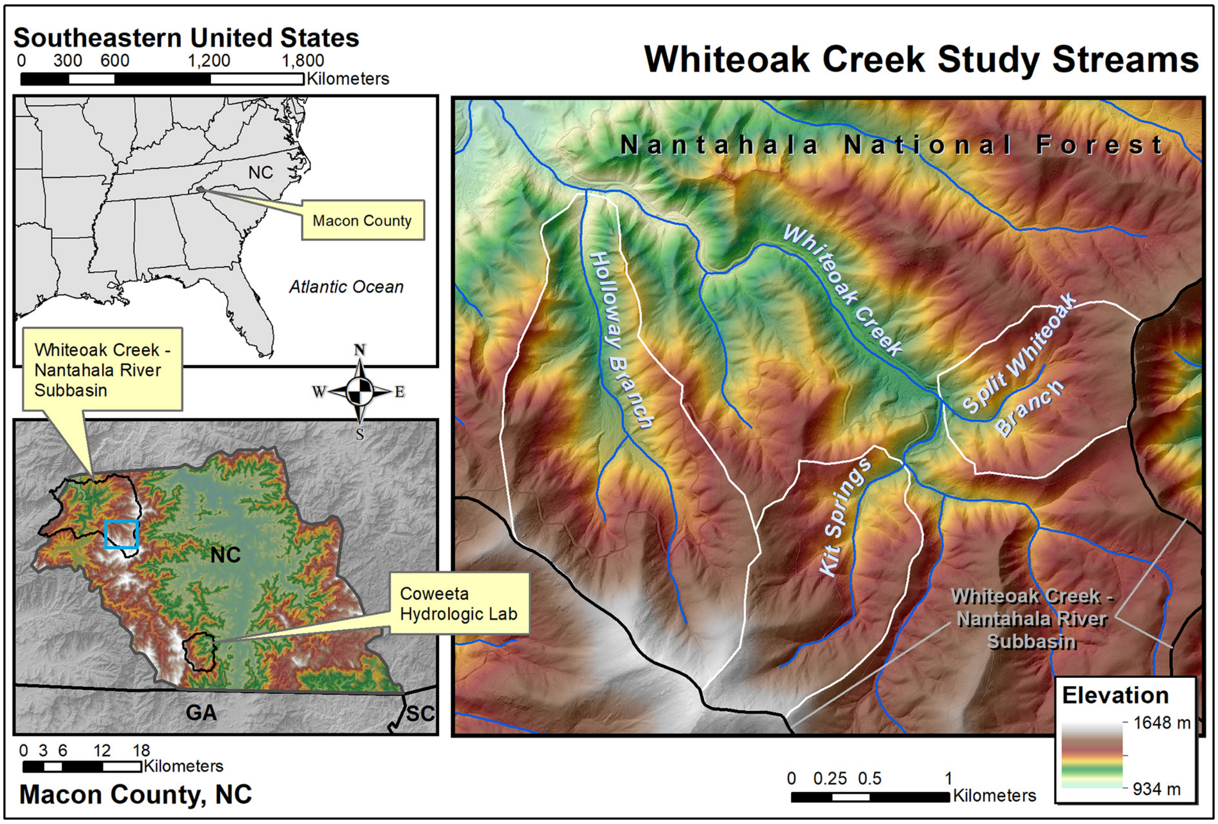

Fig. 1. Map of study site. Three perennial 2nd order streams were within the White Oak Creek (WOC) watershed (35°20′ N latitude, 83°58′ W longitude), within the Blue Ridge Physiographic Province, near the southern end of the Appalachian Mountain chain. The stream reaches were Holloway Branch (CR, remove only the rhododendron subcanopy), Split Whiteoak Branch (CFFR, remove the rhododendron subcanopy and the soil organic horizon) and Kit Springs (REF, untreated, reference).

cut material was either further slashed to the ground not to exceed 1.2 m in height (CFFR) or piled just outside the treatment area (CR). All cut stumps were sprayed immediately with herbicide (Romancier, 1971; Esen and Zedaker, 2004; Harrell, 2006). The herbicide was a triclopyr amine (Garlon 3A®, DOW AgroSciences, Indianapolis, IN) formulation (44.4% Triclopyr Triethylamine Salt) with an aquatic label

Table 1
Overstory (stems ≥2.5 cm dbh) density (stems m^{-2}) and basal area (BA, m^2 ha^{-1}) of trees and evergreen shrubs; pretreatment (2014) and post-treatment (Oct 2016, following the first growing season after cutting and fire) for the three treated stream reaches. The treatments were CR, remove only the rhododendron subcanopy; CFFR, remove rhododendron subcanopy and soil O-horizon; and REF, untreated, no removal. Standard errors are in parentheses.

	CR		CFFR[a]		REF	
	Density	BA	Density	BA	Density	BA
Pre-treatment						
Dead hemlock	392 (67)	20.95 (3.04)	302 (38)	21.07 (2.98)	362 (76)	14.26 (3.22)
Rhododendron	3056 (306)	4.72 (0.64)	2981 (194)	6.02 (0.55)	2887 (448)	4.61 (0.58)
Live deciduous	492 (56)	25.75 (2.22)	**621 (93)**	24.24 (3.43)	981 (208)	21.94 (1.56)
Post-treatment						
Rhododendron	0	0	0	0	2960 (440)	4.24 (0.51)
Live deciduous	494 (55)	25.39 (2.18)	**381 (77)**	21.74 (2.85)	850 (157)	21.08 (1.58)

Bold denotes significant ($p \leq 0.05$) differences between pre-treatment and post-treatment.
[a] At CFFR, most of the trees that died were small (≤10 cm dbh); on average 184 stems ha^{-1} small trees and 69 stems ha^{-1} large trees died after the prescribed fire.

mixed to a ratio of 50% herbicide/50% water (Elliott and Miniat, 2018). Because this herbicide was only sprayed on cut stumps and not soil, and because it breaks down and/or dissipates quickly in soils (~2–4 weeks) (Stephenson et al., 1990), it is unlikely that it affected non-target plants or soil nutrient cycling measurements (described below). Rhododendron cutting (CR, CFFR) occurred in spring (March–May) 2015 (Fig. 2b) and the prescribed fire (CFFR) was implemented in spring (March) 2016 (Fig. 2c). The firing technique included backing fires along the upper ridge and ignitions at 10–25 m intervals, depending on slope steepness, during weather conditions specified in the USDA Forest Service, Nantahala National Forest, Prescribed Burning Plan (USFS, 2011). The prescribed fire was intended to reduce the soil organic horizon in the CFFR treatment, and at the same time, the fire would reduce fuel loads created by cutting rhododendron stems (Fig. 2d).

Each stream reach received one of three treatments: Holloway (CR), Split Whiteoak (CFFR), and Kit Springs (REF). Within each stream reach, we established six transects (three on each side of the stream) extending from stream edge to the 50 m boundary. Transects were aligned perpendicular to, and on each side of, the stream, and were at least 50 m apart. We placed two 20 m × 20 m plots along (or near) each transect line with 10 m distance between plots, yielding 12 plots per stream reach. Within each of these plots, we measured the overstory and sampled organic (Oi, Oe + Oa soil horizons) and mineral soil. The overstory was sampled before (Jul–Aug 2014) and after (Oct 2016) treatments were completed (i.e., cutting rhododendron followed by prescribed fire) on all plots, a total of 36 plots in each year. Organic soil horizons and mineral soil samples were collected in the winter prior to treatment implementation (Jan–Feb 2015) and the first winter after (Jan–Feb 2017) treatments were completed. Soil samples were collected on a random subsample of eight of the 12 plots per stream reach due to

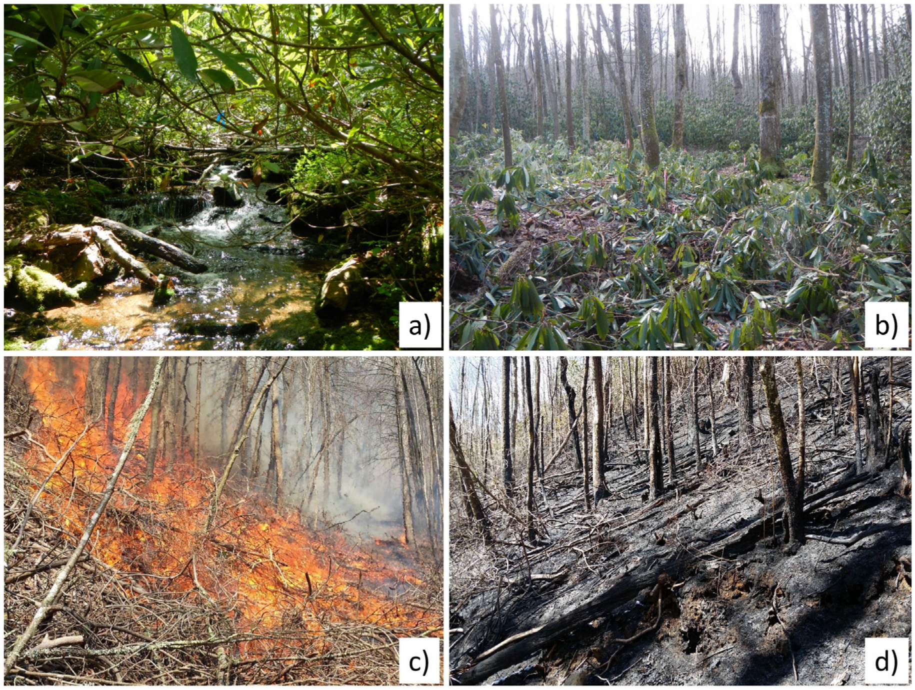

Fig. 2. *Rhododendron maximum* subcanopy was dense before treatments were implemented (a). Rhododendron cutting (CR, CFFR) occurred in spring (March–May) 2015 (b), and the prescribed fire (CFFR) was implemented in spring (March) 2016 (c). The prescribed fire reduced the soil Oi (litter layer) in the CFFR treatment and reduced the fuel loads created by cutting rhododendron stems (d).

the personnel- and time-intensive resources required of these measurements.

2.3. Overstory measurements

To characterize the sites, as well as to ensure that treatments only removed rhododendron basal area and not overstory tree basal area, we measured overstory trees and evergreen shrubs pretreatment (2014) and post-treatment (Oct 2016, following the first growing season after cutting and fire). In each plot, all woody stems ≥2.5 cm diameter at 1.37 m above ground (dbh, diameter at breast height) were measured with a diameter tape to the nearest 0.1 cm. A total of 48 plots were measured in each year, before and after treatments.

2.4. Soil organic horizons and chemical analyses

Soil organic horizon mass and nutrient pools were determined by collecting three samples per plot from a 0.09-m^2 quadrat, across eight plots per stream reach ($n = 8$). Samples were separated into two organic soil horizons: Oi (litter, where senesced leaves and twigs are deposited in the fall) and Oe + Oa (Oe = fermentation, where leaves have fractured and are partially decomposed; Oa = humus, dark and decomposed, no longer recognizable as leaves or twigs). Each horizon was placed in a paper bag, oven-dried to a constant weight at 60 °C, and weighed to the nearest 0.01 g. Dried samples were thoroughly mixed, composited by plot and horizon, ground to <1 mm using a Retsch grinding mill (Retsch, Inc., Newtown, PA), mixed again, and a subsample was placed in a glass vial for storage prior to chemical analyses. Samples were analyzed for total carbon (C) and nitrogen (N) by combustion (Flash EA 1112 series, CE Elantech, Lakewood, NJ). Total Ca, K, Mg, sodium (Na), and aluminum (Al) concentrations were determined by dry-ashing a 0.5-g sample at 500 °C for 4 h followed by digestion in 2.2 M nitric acid (USDA FS, Coweeta Hydrologic Laboratory, 2017), the resulting solution was analyzed using inductively coupled plasma spectrophotometer (ICP, Thermo Fisher iCAP, Madison, WI). Ash-free dry weight of Oe + Oa horizon samples was measured during the dry-ashing/digestion process to allow sample weight correction for mineral soil contamination. All weight and nutrient concentration data presented for Oe + Oa horizon samples are presented on an ash-free basis. Oe + Oa samples contained <5% mineral material and data are presented on an oven-dry weight basis.

2.5. Mineral soil chemistry

Mineral soil samples were collected in winter (Jan–Feb) at the same time as the organic horizon samples. Composite mineral soil samples were collected from eight plots per stream reach ($n = 8$) using a soil probe (Oakfield Apparatus, Inc. Fond du Lac, WI). Composite samples consisted of four individual samples per plot. We collected surface (0–10 cm) and subsurface (10–30 cm) soils by depth. Soil samples were air-dried and sieved to <2 mm for analyses of total C and N and exchangeable cations (Ca, Mg, K, Na, Al, and NH_4-N) (USDA FS, Coweeta Hydrologic Laboratory, 2017). Total C and N were determined by combustion on an elemental analyzer (Flash EA 1112 series, CE Elantech, Lakewood, NJ). Exchangeable cations were extracted from 5.0 g of soil using 50 ml of 0.001 mol m^{-3} NH_4Cl on a mechanical vacuum extractor (SampleTek, Science Hill, KY). Solution concentrations of Ca, Mg, K, Na, and Al were determined using an inductively coupled plasma spectrophotometer (ICP, Thermo Fisher iCAP 6300, Madison, WI). The solution concentration of NH_4-N was analyzed on an Astoria 2 Autoanalyzer (Astoria-Pacific, Clackamas, OR). Total C and N are expressed as a percent and exchangeable cation concentrations are expressed as $cmol_c$ kg^{-1}.

2.6. Stream chemistry

For stream chemistry collections, all three streams were sampled for the duration of the study. Stream water chemistry was analyzed by collecting a weekly grab sample of stream water above and below each stream reach from May 2014 through Dec 2016, and then biweekly thereafter until the end of Oct 2019. Stream water samples were analyzed for nitrate-nitrogen (NO_3-N), ammonium-nitrogen (NH_4-N), ortho-phosphate-phosphorus (PO_4-P), sulfate (SO_4), chloride (Cl), Na, Ca, Mg, K, Al, dissolved organic carbon (DOC), total dissolved inorganic nitrogen (TDN), and pH (USDA FS, Coweeta Hydrologic Laboratory, 2017). Solution pH was determined on an unfrozen, unfiltered sample within 24 h of collection (Orion Research digital A211 pH meter, Broadley James pH probe, Broadley James Corp., Irvine, CA). Cation concentrations (Ca, Mg, Na, K, and Al) were determined on unfiltered samples using an inductively coupled plasma spectrometer (ICP) (Thermo Fisher iCAP 6300, Thermo Fisher Scientific, Madison, WI). NH_4-N was determined on unfiltered samples on an autoanalyzer (Astoria 2, Astoria-Pacific, Clackamas, OR). Anion concentrations (NO_3-N, SO_4, PO_4-P, and Cl), were determined on unfiltered samples using a capillary ion chromatograph (Dionex ICS 4000, Thermo Scientific, Dionex, Sunnyvale, CA). Unfiltered samples were used, as filtering samples does not significantly change concentrations in these surface waters and risks exposing samples to background atmospheric N as a possible contaminant. Analysis of DOC and TDN used filtered samples (0.7 μm Millipore glass fibre prefilters, Shimadzu DOC-VCPH TNM-1, Shimadzu Scientific Instruments, Columbia, MD). Cation and anion concentrations are reported as μeq L^{-1} and DOC, TDN, Al are reported as mg L^{-1}. Stream acid-neutralizing capacity (ANC, μeq L^{-1}) was calculated as the difference of the sum of the acid anions from the sum of the base cations as described in Knoepp et al. (2016).

2.7. Data analyses

To determine whether overstory density and basal area and organic soil (Oi, Oe + Oa) mass were different among the three stream reaches and between years, we used repeated-measures analyses (PROC MIXED, SAS 9.4, 2002–2012) on data collected across the 12 plots per stream reach ($n = 12$). To determine whether soil chemistry (pH, NH_4-N, K, Ca, Mg, Al, SO_4, and total C and N) was significantly different between years (2015 pre-treatment, and 2017 post-treatment) and among treatment reaches (CR, CFFR, REF), we also used repeated measures analyses (PROC MIXED, SAS 9.4, SAS Institute, 2002–2012). Repeated measures takes into account the initial differences (2015, pre-treatment) among stream reaches. We used the unstructured covariance option in the repeated statement because it produced the smallest value for the Akaike's Information Criterion (AIC) and Schwarz' Bayesian Criterion (SBC) (Littell et al., 2004). We evaluated the main effects of year (sample dates), treatment (CR, CFFR, and REF), and year*treatment interactions, with $n = 8$ composite soil and forest floor samples per stream reach. If overall F tests were significant ($p \leq 0.05$) then least squares means (LS-means, Tukey-Kramer adjusted t-statistic) tests were used to evaluate pairwise differences. We used the Satterthwaite option in the model statement to obtain the correct degrees of freedom (Littell et al., 2004).

To determine differences in stream chemistry among all three treatment reaches over time, before and after treatments were implemented, we used a paired watershed approach (Ford et al., 2011). This method uses the relationship between stream chemistry from two closely–located watersheds similar in size and pre-treatment cover conditions. In the subsequent treatment period, one watershed serves as a control (REF, reference) and remains undisturbed while a management treatment (CR or CFFR) is applied to the other watershed. Successive observations in time are considered independent replicates.

Our goal was to predict the stream chemistry response to management (CR or CFFR), given stream chemistry from the reference (REF) watershed and a predictive relationship between them. We fit a model to the average monthly concentration of each analyte (pH, ANC, DOC, NO_3-N, NH_4-N, K, Ca, Mg, Al, SO_4, PO_4-P, TDN) in the treatment watershed (y_T) as a linear function of the average monthly concentration of the analyte in the reference watershed (x_R) using data prior to treatment or management (PROC REG, SAS v9.4, SAS Institute, 2002–2012). We used this relationship to predict the expected average monthly analyte concentration following treatment ($\hat{y}_T$). The observed management response, D, was estimated as the concentration deviation of the observed value from that predicted by the reference without management,

$$D = y_T - \hat{y}_T \tag{1}$$

We interpreted monthly deviations as statistically significant if the value fell outside of the standard errors of the individual predicted values. Autocorrelation among the residuals for each month with those from the preceding six months was also calculated and tested for significance as a test of independent observations in time (PROC ARIMA, SAS v9.4, SAS Institute, 2002–2012). If there was a significant autocorrelation structure in the residuals, we incorporated a first-order autoregressive (AR(1), y_{t-1}) term into the model.

Lastly, we investigated whether deviations in analyte concentrations covaried by estimating Pearson's simple correlation coefficients among all D series (PROC CORR, SAS v9.4, SAS Institute, 2002–2012). All statistical tests were evaluated at $\alpha = 0.05$.

3. Results

3.1. Overstory

Before treatments were implemented, deciduous tree density and basal area were similar among stream reaches. After treatment, deciduous tree density declined only at the CFFR stream reach ($t_{1,44} = -3.93$, $p = 0.0066$) (Table 1). However, there were no differences in basal area between pre- and post-treatment for any of the stream reaches. Even at CFFR, basal area did not significantly decline because most of the trees that died were small (≤10 cm dbh); on average, 171 ± 40 stems ha^{-1} of small trees (basal area reduced by 0.52 ± 0.11 m^2 ha^{-1}) and 69 ± 31 stems ha^{-1} of large trees (basal area reduced by 1.98 ± 1.16 m^2 ha^{-1}) died after the prescribed fire (Table 1). As expected, aboveground stems of rhododendron were absent: 4.72 and 6.02 m^2 ha^{-1} basal area was removed on CR and CFFR, respectively. Rhododendron density and basal area remained high on the reference stream reach, REF (Table 1).

3.2. Organic soil mass (Oi, Oe + Oa horizons)

We visually observed that the fire in the CFFR treatment resulted in the consumption of the cut rhododendron leaves and branches and partial removal of the forest floor (Fig. 2d), temporarily depleting leaf litter (Oi), however, this was replenished by Oi mass the next fall before the post-treatment sampling date (Table 2). As such, there were no statistically significant effects on the Oe + Oa mass measured. The CR and REF

Table 2
Oi and Oe + Oa organic soil horizons mass (g cm^{-2}) before (Jan–Feb, 2015) and one growing season after prescribed fire (Jan–Feb, 2017) for CR, CFFR, and REF (see Table 1). Standard errors are in parentheses.

Treatment	Organic soil horizons	2015	2017
CR	Oi	267.9 (33.3)	191.9 (22.4)
	Oe + Oa	5110 (776)	5645 (1279)
CFFR	Oi	279.4 (21.2)	278.7 (61.5)
	Oe + Oa	3193 (549)	3285 (671)
REF	Oi	260.0 (24.8)	228.4 (31.9)
	Oe + Oa	2145 (578)	2034 (515)

There were no significant ($p \leq 0.05$) differences between years (SAS 2002–2012, PROC GLM).

treatments also showed no difference between pre-treatment and post-treatment years for either Oi or Oe + Oa masses (Table 2).

3.3. Organic and mineral soil chemistry

There were differences in organic (Oi, Oe + Oa horizons) and mineral (0–10, 10–30 cm depths) soil chemistry among the three stream reaches before treatments were implemented (Table S1). For example, shallow soil pH was greater in CFFR site compared to the CR site, K in the Oi layer was greater in the CR site compared to the REF site, shallow soil Al was greater in the CR site compared to the REF site, and shallow soil %N was greater in the REF site compared to the CFFR site (Table 3). The treatments generally resulted in lowering soil N (Table 3; Table S1). Mineral soil pH was relatively low (pH < 4.20) at all stream reaches, indicating acidic soils. This was most likely related to the dense evergreen coverage. Soil pH did not change after treatments on any of the three stream reaches (Table 3). Post-treatment, CR had lower soil NH_4-N than CFFR and REF and NH_4-N declined on CFFR and CR (Table 3). Soil K and Ca concentrations were not affected by treatments. Post-treatment Mg and Al concentrations of the Oe + Oa and 10–30 cm soil depth were higher on CFFR than CR. Oe + Oa total N was lower on CFFR than CR and total N of the 10–30 cm depth was lower on CFFR than REF. There were no differences in soil total C among treatment reaches (Table 3).

3.4. Stream chemistry

Stream concentrations of analytes in the REF streams generally predicted concentrations in the two treatment streams well (Table S2). The strongest models were those predicting DOC, ANC, Mg, and Al ($0.57 < R^2 < 0.90$, $P < 0.001$ for all), while those predicting NO_3 and Ca were somewhat weaker ($0.11 < R^2 < 0.67$, $0.13 < P < 0.001$), but generally statistically significant. All final models had no autocorrelation among residuals, indicating assumptions in our approach were not violated.

Stream chemistry responses to treatments varied between CR and CFFR. Most responses were transient, generally with pH, N, and some cations declining, while Al and DOC showed a pulse increase. There were differences among the three stream reaches in all stream chemical analytes before the treatments were implemented; however, our statistical approach allowed us to account for these differences (Table S1, Figs. 3–8, Figs. S1–S6). Stream pH was relatively high (above 6.1, Fig. S1) over the entire study period and it declined after treatment for several months on both CR and CFFR. For the first two post-treatment years, DOC was elevated at the CR stream, then subsided to pre-treatment levels by 2018 (Fig. 3). Stream ANC declined by 6.0 μeq L^{-1} for several months in 2018 and 2019 at CR, while the ANC responses at CFFR were more variable with fewer months having a significant response (Fig. 4). Stream NO_3-N concentrations declined by 6.0 μeq L^{-1} or more for several months at CR; however, fewer months were only slightly lower at CFFR (Fig. 5). Declines in N tended to coincide with declines in stream Ca and Mg concentrations, which were also were strongly correlated with each other (Table 4). Both Ca and Mg declined after the treatments for both CR and CFFR with the stronger response at the CFFR stream (Figs. 6, 7, Table 4). Stream Al concentrations were elevated, with the stronger response at CR (Fig. 8). The Al response was positively correlated with DOC, with increases in Al generally coinciding with increases in DOC (Table 4).

Table 3
Organic (Oi, Oe + Oa horizons) and mineral soil (0–10, 10–30 cm depths) chemistry before (2015) and after (2017) treatments were implemented for CR, CFFR, and REF. Total C and N are expressed as %, and exchangeable cation (NH_4-N, K, Ca, Mg, Al) concentrations are expressed as $cmol_c\ kg^{-1}$. N = 8 in both years, same plots sampled. See Table S1 for ANOVA results of main effects.

		Pre-treatment 2015			Post-treatment 2017 (cut+burn)		
		CR	CFFR	REF	CR	CFFR	REF
pH	0–10	3.67 b (0.09)	4.02 a (0.03)	4.00 ab (0.04)	3.80 (0.13)	4.10 (0.04)	4.10 (0.09)
	10–30	4.12 (0.04)	4.20 (0.02)	4.16 (0.02)	4.12 (0.07)	4.31 (0.02)	4.28 (0.04)
NH_4-N	0–10	13.72 x (0.79)	13.42 x (0.76)	11.97 (0.65)	4.58 b, y (0.51)	8.61 a, y (1.27)	8.93 a (1.24)
	10–30	10.75 x (1.10)	11.44 (0.56)	9.94 (0.47)	4.10 b, y (0.48)	9.15 a (0.89)	7.78 a (1.06)
K	Oi	5.087 a, x (0.46)	3.892 ab (0.31)	3.232 b (0.24)	3.161 y (0.16)	2.942 (0.32)	3.816 (0.28)
	Oe + Oa	2.503 (0.17)	2.714 (0.23)	2.534 (0.08)	2.079 (0.12)	2.354 (0.16)	2.423 (0.22)
	0–10	0.368 (0.04)	0.375 (0.04)	0.415 (0.04)	0.278 (0.04)	0.416 (0.07)	0.376 (0.03)
	10–30	0.172 (0.02)	0.165 (0.01)	0.172 (0.01)	0.172 (0.02)	0.268 (0.06)	0.204 (0.02)
Ca	Oi	60.72 (4.90)	50.78 (1.96)	60.49 (5.20)	62.99 (4.84)	50.45 (2.17)	65.67 (5.32)
	Oe + Oa	17.47 (1.08)	24.71 (5.99)	26.99 (5.20)	17.68 (2.75)	33.90 (6.18)	33.33 (5.48)
	0–10	0.379 (0.12)	0.325 (0.09)	1.012 (0.29)	0.449 (0.18)	0.804 (0.18)	1.579 (0.62)
	10–30	0.065 (0.01)	0.070 (0.01)	0.206 (0.08)	0.127 (0.03)	0.288 (0.12)	0.422 (0.17)
Mg	Oi	14.78 x (1.28)	14.11 (0.64)	14.84 (1.11)	12.91 y (1.46)	12.60 (0.70)	14.00 (1.17)
	Oe + Oa	6.362 (0.56)	9.640 y (0.51)	9.128 (0.50)	6.367 b (0.82)	13.57 a, x (1.71)	10.52 a (1.04)
	0–10	0.365 x (0.06)	0.380 (0.04)	0.504 (0.06)	0.088 b, y (0.02)	0.390 ab (0.09)	0.441 a (0.12)
	10–30	0.126 (0.02)	0.145 (0.01)	0.152 (0.01)	0.038 b (0.01)	0.225 a (0.06)	0.160 ab (0.04)
Al	Oi	1.298 (0.21)	1.965 (0.28)	1.067 (0.13)	1.528 (0.31)	9.222 (5.04)	1.659 (0.28)
	Oe + Oa	28.48 (3.44)	40.34 y (5.65)	39.58 (7.56)	52.55 b (5.98)	123.33 a, x (19.96)	69.66 b (13.22)
	0–10	8.84 a, x (0.88)	7.27 ab, x (0.45)	5.30 b, x (0.70)	1.72 y (0.44)	3.25 y (0.67)	2.94 y (0.68)
	10–30	5.41 x (0.76)	4.58 (0.24)	3.52 (0.24)	1.47 b, y (0.25)	3.58 a (0.39)	2.88 ab (0.51)
N (%)	Oi	0.945 (0.14)	0.740 (0.04)	0.964 (0.09)	0.871 (0.05)	0.856 (0.03)	0.884 (0.08)
	Oe + Oa	1.631 (0.03)	1.452 (0.04)	1.520 (0.04)	1.795 a (0.07)	1.396 b (0.09)	1.599 ab (0.09)
	0–10	0.459 ab (0.04)	0.302 b (0.02)	0.512 a (0.05)	0.412 (0.04)	0.332 (0.02)	0.513 (0.07)
	10–30	0.237 (0.04)	0.170 (0.01)	0.288 (0.04)	0.210 ab (0.04)	0.124 b (0.01)	0.282 a (0.05)
C (%)	Oi	47.87 (0.84)	48.09 (0.39)	48.09 (0.39)	48.40 (0.12)	48.22 (0.62)	49.09 (0.18)
	Oe + Oa	44.14 (1.04)	43.89 (0.77)	41.80 (1.05)	44.55 (0.72)	40.09 (1.92)	40.77 (2.50)
	0–10	9.335 (0.69)	7.417 (0.47)	9.078 (0.62)	8.344 (0.68)	7.930 (0.41)	8.811 (0.89)
	10–30	4.917 (0.62)	4.235 (0.36)	5.267 (0.56)	4.779 (0.72)	3.692 (0.23)	5.307 (0.68)

Samples were collected in the winter before (Jan–Feb 2015) and the winter after (Jan–Feb 2017) treatments were implemented. Within year, values followed by different letters (a, b) denote significant differences (adjusted $p \le 0.05$) among treatments according to repeated measures analyses (SAS 2002–2012, PROC MIXED) with pairwise comparisons (LS-means, Tukey-Kramer adjusted *t*-statistic). Within treatment, values followed by different letters.
(x, y) denote significant differences (Adjusted $p \le 0.05$) between years (2015 vs. 2017).

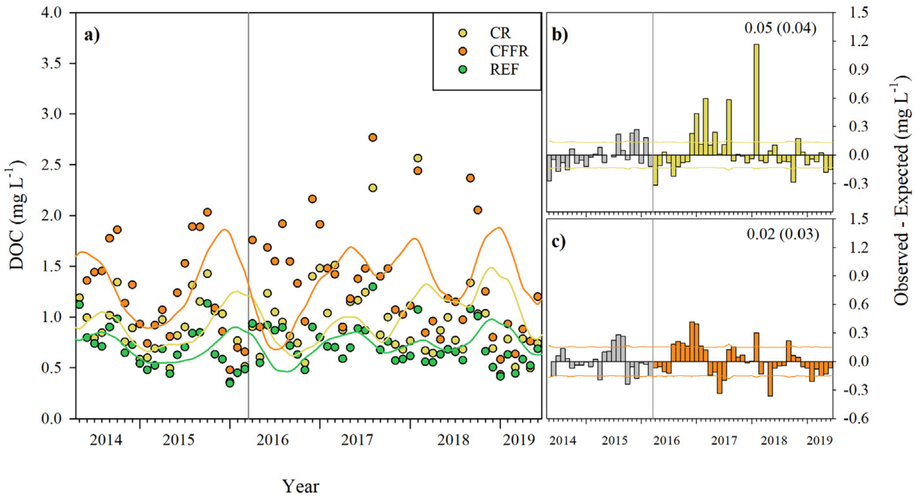

Fig. 3. Stream dissolved organic carbon (DOC, mg L^{-1}) concentration in three experimental watersheds (a). Data are smoothed with a LOESS regression with 10% sampling (lines). Relationships between pretreatment DOC concentration in the CR or CFFR and REF were developed and used to predict expected DOC concentrations post-treatment. Expected DOC concentrations were subtracted from observed concentrations (b, c) in the pre- (grey bars) and post-treatment periods (colored bars in b, c). Deviations outside the standard error of the prediction interval (solid colored lines in b, c) are statistically significant. Post-treatment means (SE) are also given. Vertical grey solid lines in all panels denote timing of treatment completion.

4. Discussion

4.1. Terrestrial responses

We expected to find changes in soil and stream chemistry after the rhododendron removal treatments were fully implemented, with a faster response in the CFFR treatment that used prescribed fire to reduce the rhododendron slash and the soil organic horizons (Oi, Oe + Oa). In the CFFR treatment, we did not detect a decrease in Oi horizon mass, likely because of sampling timing. Our winter sampling occurred after fall leaf inputs. Had we sampled earlier, the O-horizon mass may have shown a modest decline similar to other studies that sample immediately after fires (Hubbard et al., 2004; Elliott et al., 2012).

It is not clear if our treatments influenced soil N availability. In our study, however, we found lower mineral soil NH_4-N in the CR treatment than CFFR and REF and no difference between CFFR and REF (Table 4). We sampled soils in the winter (Jan–Feb, 2017) when inorganic N is typically at its seasonal low (Hubbard et al., 2004; Durán et al., 2016) and several months following the completion of the treatments (cutting + prescribed fire). However, in a parallel study, Osburn et al. (2018) found that the combination of rhododendron subcanopy and Oi layer removal (CFFR) increased mineral soil C and N availability, resulting in increased soil microbial biomass and increased production of key microbial extracellular enzymes. Even though Osburn et al. (2018) collected soil samples during the second growing season (April and July 2017) after the prescribed fire, they also observed increases in soil inorganic N (NO_3-N, NH_4-N) in CFFR plots compared with other treatments (i.e., CR and REF). Direct conversion of organic N to inorganic N by fire (Certini, 2005; Neary et al., 2005), or increased inorganic-N transformation rates following burns, have been observed at other southern Appalachian sites (Knoepp et al., 2009; Elliott et al., 2012). These effects, in combination with reduced inorganic N uptake by roots and mycorrhizae

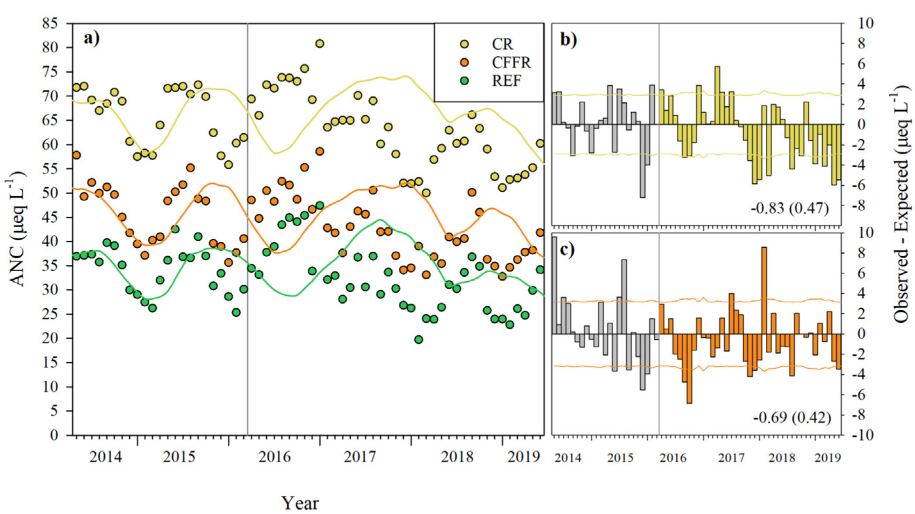

Fig. 4. Stream acid neutralizing capacity (ANC, μeq L^{-1}) concentration in three experimental watersheds (a). Data are smoothed with a LOESS regression with 10% sampling (lines). Relationships between pretreatment ANC concentration in the CR or CFFR and REF were developed and used to predict expected ANC concentrations post-treatment. Expected ANC concentrations were subtracted from observed concentrations (b, c) in the pre- (grey bars) and post-treatment periods (colored bars in b, c). Deviations outside the standard error of the prediction interval (solid colored lines in b, c) are statistically significant. Post-treatment means (SE) are also given. Vertical grey solid lines in all panels denote timing of treatment completion.

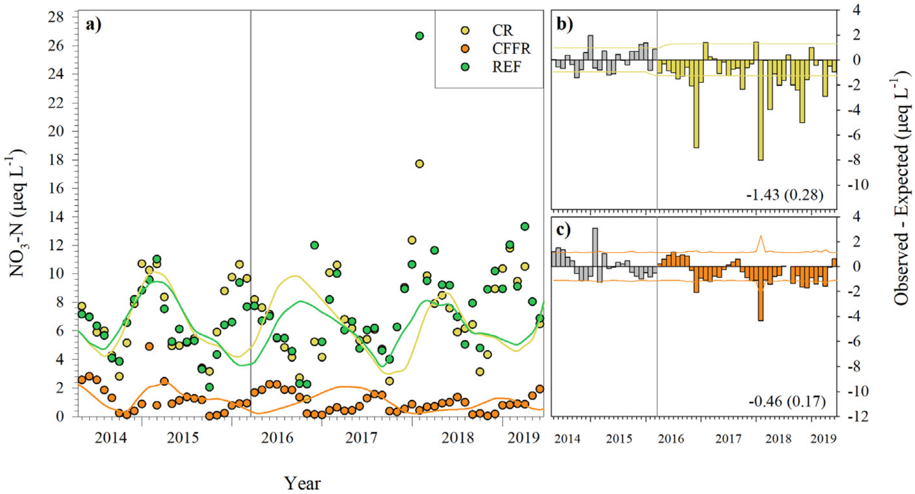

Fig. 5. Stream nitrate-nitrogen (NO_3-N, µeq L^{-1}) concentration in three experimental watersheds (a). Relationships between pretreatment NO_3-N concentration in the CR or CFFR and REF were developed and used to predict expected NO_3-N concentrations post-treatment. Expected NO_3-N concentrations were subtracted from observed concentrations (b, c) in the pre- (grey bars) and post-treatment periods (colored bars in b, c). Deviations outside the standard error of the prediction interval (solid colored lines in b, c) are statistically significant. Post-treatment means (SE) are also given. Vertical grey solid lines in all panels denote timing of treatment completion.

following evergreen removal, could have produced their observed trend, where both evergreen removal and prescribed fire were necessary to increase concentrations of soil inorganic N (Osburn et al., 2018). Others have found that soil nitrogen responses can be immediate and short-term following fire disturbance (see review, Knoepp et al., 2005); however, with repeated fires, cycling of soil carbon and nitrogen can shift more dramatically by changing plant inputs and decomposition rates (Pellegrini et al., 2020). In our study, with a single, spring fire, any immediate release of soil inorganic N or cations (Mg, Ca, K) due to the fire could have been taken up by tree and herbaceous species during the growing season months after treatment. In a companion study, Elliott and Miniat (2018) found that herbaceous flora and tree seedling recruitment rapidly increased after treatment on these same sites.

4.2. Stream DOC was a sensitive indicator

Stream DOC, a sensitive indicator of changes in land management or terrestrial net primary production (Meyer et al., 2014; Lajtha and Jones, 2018), was elevated in both treatments. While the typical seasonal patterns in DOC were unaltered (Chow et al., 2013; Meyer et al., 2014), and the magnitude of DOC fell within the typical range for southern Appalachian Mountain streams (Elliott et al., 2008; Meyer et al., 2014; Singh et al., 2016), both treatments elevated stream DOC in expected ways. We found greater DOC than expected in the stream that had rhododendron stems removed but left as slash nearby without burning (CR) than the stream where the slash was scattered and burned (CFFR). We expected that biogeochemically-active solutes, such as DOC, would respond to removal and that the responses would be faster where

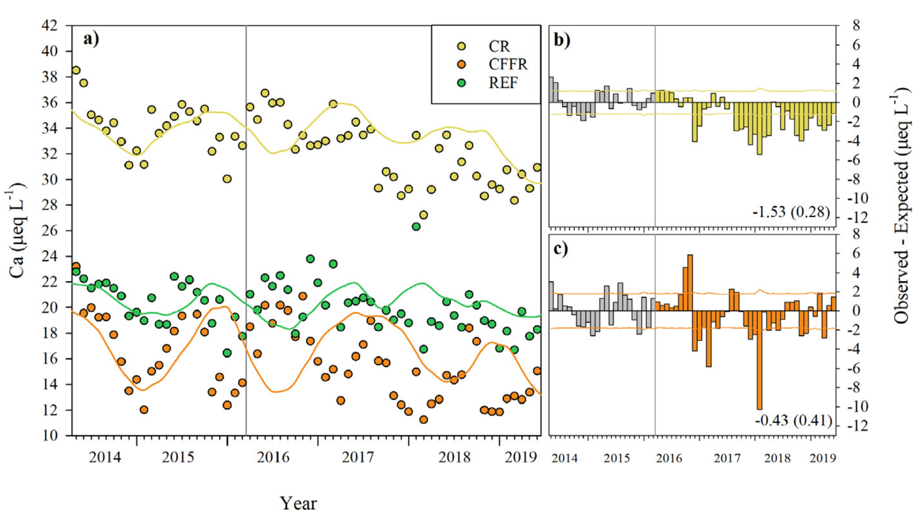

Fig. 6. Stream calcium (Ca, µeq L^{-1}) concentration in three experimental watersheds (a). Data are smoothed with a LOESS regression with 10% sampling (lines). Relationships between pretreatment Ca concentration in the CR or CFFR and REF were developed and used to predict expected Ca concentrations post-treatment. Expected Ca concentrations were subtracted from observed concentrations (b, c) in the pre- (grey bars) and post-treatment period (colored bars in b, c). Deviations outside the standard error of the prediction interval (solid colored lines in b, c) are statistically significant. Post-treatment means (SE) are also given. Vertical grey solid lines in all panels denote timing of treatment completion.

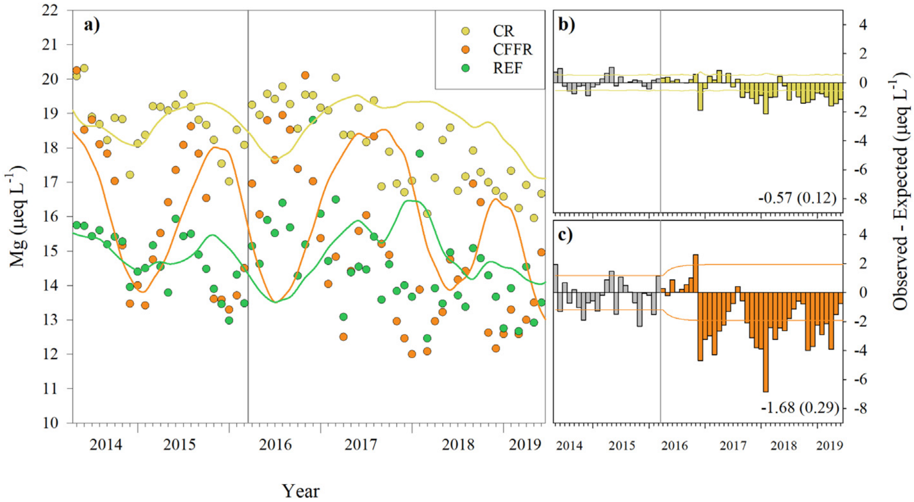

Fig. 7. Stream magnesium (Mg, μeq L^{-1}) concentration in three experimental watersheds (a). Data are smoothed with a LOESS regression with 10% sampling (lines). Relationships between pretreatment Mg concentration in the CR or CFFR and REF were developed and used to predict expected Mg concentrations post-treatment. Expected Mg concentrations were subtracted from observed concentrations (b, c) in the pre- (grey bars) and post-treatment periods (colored bars in b, c). Deviations outside the standard error of the prediction interval (solid colored lines in b, c) are statistically significant. Post-treatment means (SE) are also given. Vertical grey solid lines in all panels denote timing of treatment completion.

prescribed fire was used to reduce the organic soil horizons (i.e., CFFR). Indeed this was the case, with the CFFR treatment showing elevated DOC in the first year. The CR treatment, however, had a much greater response in DOC, but this occurred during the second year when responses in the CFFR treatment had mostly returned to baseline. Our responses were consistent with previous studies that showed greater responses from slash residue than from fire (see below).

Both fire and forest cutting can alter terrestrial DOC exports into stream water (Meyer et al., 2014; Lajtha and Jones, 2018; Rhoades et al., 2019b). Stream DOC concentrations are related to the amount of terrestrial biomass available for decomposition and the hydrologic transport of that carbon to the stream (Brooks et al., 1999). In general, wood biomass left on the forest floor after forest harvesting more consistently elevates stream DOC than fire, particularly if the fire is low intensity. For example, stream DOC exports increase in proportion to the amount of forest floor wood biomass left after harvesting (Lajtha and Jones, 2018), particularly for the smallest molecular size fraction of DOC (Törmänen et al., 2020). Residue piles created by harvesting can trigger decomposition through biochemical means even though physically they tend to increase soil moisture and decrease soil temperature (Ojanen et al., 2017). We did find it to be true that the rhododendron slash increased soil moisture and decreased soil temperature in this study (Elliott and Miniat, 2018). Fire, on the other hand, can produce inconsistent responses for stream DOC. For example, in forested wetland watersheds with high DOC, stream DOC can be elevated in stormflow following fire (Olivares et al., 2019), or minimally affected in mountainous watersheds following wildfire (Mast and Clow, 2008).

Interestingly, the stream DOC responses in both treatments were coupled with changes in stream Al concentration (Table 4). Stream Al concentrations increased after treatment with greater increases at CR than CFFR. The solubility of monomeric Al in the range of soil pH at our sites is controlled by organic matter in the upper soil horizons and

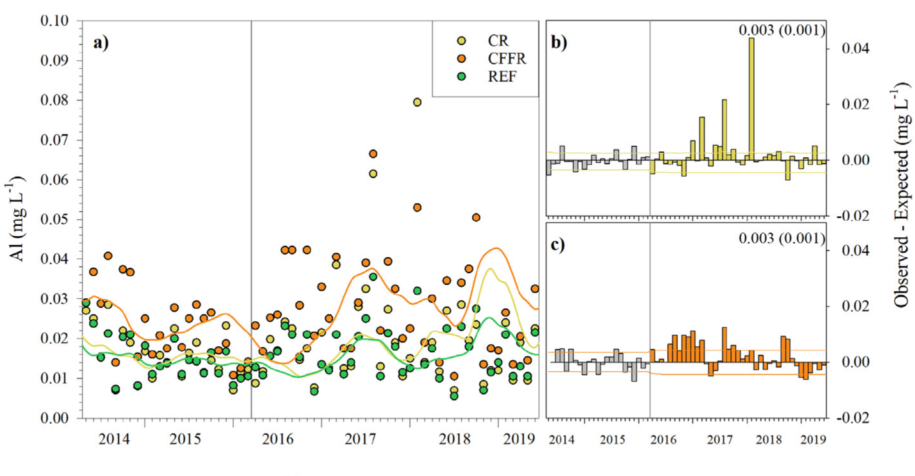

Fig. 8. Stream aluminum (Al, mg L^{-1}) concentration in three experimental watersheds (a). Data are smoothed with a LOESS regression with 10% sampling (lines). Relationships between pretreatment Al concentration in the CR or CFFR and REF were developed and used to predict expected Al concentrations post-treatment. Expected Al concentrations were subtracted from observed concentrations (b, c) in the pre- (grey bars) and post-treatment periods (colored bars in b, c). Deviations outside the standard error of the prediction interval (solid colored lines in b, c) are statistically significant. Post-treatment means (SE) are also given. Vertical grey solid lines in all panels denote timing of treatment completion.

Table 4
Pearson's simple correlation coefficients among stream chemistry values predicted during the post-treatment period at the Holloway Branch (CR) stream reach (A) and the Split Whiteoak Branch (CFFR) stream reach (B). Only significant correlation coefficients are shown, those in normal type are significant at $p \leq 0.05$, and those in bold are significant at $p \leq 0.01$.

A. Holloway Branch (CR)

	SO_4	K	Ca	Mg	Al	ANC	DOC	TDN
pH		0.55					0.33	
NO_3	0.39		0.44	0.45	−0.44	−0.35	−0.45	**0.95**
NH_4							−0.33	
SO_4						−0.36		
K				0.34	0.52	**0.71**	**0.66**	
Ca				**0.90**		0.38		**0.57**
Mg						0.47		**0.58**
Al							**0.90**	−0.36
ANC							0.44	
DOC								−0.34

B. Split Whiteoak Branch (CFFR)

	NH4	PO4	SO4	K	Ca	Mg	Al	ANC	DOC	TDN
pH			−0.36				**−0.60**		**−0.57**	
NO_3	0.43				**0.76**	**0.80**		−0.47		**0.94**
NH_4		−0.31			0.37	0.36				0.48
SO_4				0.33						
K							0.37	0.30		
Ca						**0.94**		−0.42		**0.86**
Mg								−0.40		**0.85**
Al									**0.83**	
ANC										−0.44

in-part by DOC in the mineral soil horizons (McDowell and Wood, 1984; Dethier et al., 1988). Although the prescribed fire was intended to reduce the rhododendron slash and the soil organic horizons (Oi, Oe + Oa) in the CFFR treatment, our sampling did not show a reduction in the soil Oi horizon mass because litterfall replenished the Oi before the winter samples were collected. This organic matter in the O-horizon likely mobilized Al. In the southern Appalachians where streamflow is generated through infiltration and percolation of rainwater rather than by overland flow (Hibbert and Troendle, 1988), the positive relationship between DOC and mobilization of Al in soils would be seen in the stream water, as well. In other forested watersheds with similar hydrology, DOC and Al concentrations are positively related in soils and streams (McDowell and Wood, 1984; Dethier et al., 1988). Lastly, the pattern of increased DOC and Al was likely not a dilution effect at this site for greater streamflow. While we did not measure streamflow directly in this study, a companion study did find that total plot evapotranspiration (ET) did not change pre- vs. post-treatment, with the canopy trees showing a compensation effect. Mean (SE) monthly precipitation at a nearby gage at similar elevation (35°01′57.89/83°28′05.24, 1366 m), (Laseter et al., 2012) during pretreatment was 215 (27) mm, and during posttreatment it was 214 (19) mm, which was slightly greater than the long-term monthly average 198 (3) mm ($n = 933$ months). With similar precipitation regimes among the treatments, and no ET response, we would expect streamflow (the balance) to remain unaffected.

4.3. The perplexing case of stream N declining

One of the most intriguing responses we observed was that stream concentrations of NO_3-N declined along with Ca and Mg. Decades of research have shown that forest harvesting mobilizes base cations, inorganic monomeric aluminum, and NO_3-N from soils to surface waters, and the magnitude and duration of responses depend on the harvest intensity (Siemion et al., 2011). For example, where riparian woody vegetation was removed over an entire stream network (i.e., 5-km length), disturbing 21% of the watershed, the greatest stream NO_3-N pulse occurred in the second spring after the wood removal (Larson et al., 2019). Overstory tree mortality of eastern hemlock also caused dissolved inorganic nitrogen concentrations in the streams to be elevated for the first few years in nearby streams (Miniat et al., 2020). Our results are not only inconsistent with these increases, but they actually show a significant change in the opposite direction. In our study, rhododendron cutting resulted in 15.5% and 19.9% of the total basal area removed, with residual basal areas of 25.39 and 21.74 m^2 ha^{-1} of live deciduous trees for CR and CFFR, respectively. We observed lower than expected stream NO_3-N post-treatment. Although, the length of stream (300 m) treated in our study, was much less than the 5 km length treated in Larson et al. (2019), we speculate that we would have seen the same results had a larger area been treated. Rhododendron has been shown to bind soil inorganic N that is available to deciduous trees into protein-tannin complexes that co-occurring, non-ericaceous trees are unable to access (Wurzburger and Hendrick, 2007, 2009). By removing rhododendron, the remaining deciduous trees likely accelerated their N (and Ca and Mg) uptake along with the increased soil moisture availability (Elliott and Miniat, 2018), which could partially explain why we observed lower than expected stream nutrients (NO_3-N, Ca, and Mg, Table 4) post-treatment.

In addition to increased terrestrial uptake of NO_3-N, Ca, and Mg, in-stream processes were likely altered in ways that would also support the observed declines (Musetta-Lambert et al., 2020; Plont et al., 2020). Removal of rhododendron along these riparian corridors increased summer stream temperatures by 0.9 to 2.6 °C even though the deciduous canopy cover remained above 80% (Raulerson et al., 2020). Warmer streams accelerated uptake activity by in-stream organisms (Eliason, 2017; Dudley et al., in review). However, stream temperatures did not increase to deleterious levels for trout and other cold-water organisms, with average post-treatment summer temperatures still below 17 °C (Raulerson et al., 2020). This fell below the 20 °C threshold for cold-water organisms (McDonnell et al., 2015), another indication that stream organisms could have increased their NO_3-N uptake capacity.

Not all water quality parameters changed as expected following rhododendron removal treatments. We expected that removing rhododendron would increase stream pH and ANC (i.e., decrease stream acidity); however, both were lower than expected for several months after treatment on both treated streams. Even though soil pH was low (<4.2), stream pH was relatively high (>6.1) pre- and post-treatment, and ANC was within the median range for southern Appalachian Mountain streams (Burns et al., 2020). This latter metric indicates that these were not initially acidic streams (Driscoll et al., 2001). ANC likely decreased because Ca and Mg were also lower than expected after treatment. Nonetheless, ANC remained well above 50 μeq L^{-1} on CFFR and ranged from 40 to 55 μeq L^{-1} on CR throughout the course of this study. ANC with chronic levels below 50 μeq L^{-1} are considered potentially harmful for aquatic biota (McDonnell et al., 2015). The reference stream (REF) had the lowest ANC levels, ranging from 25 to 45 μeq L^{-1}.

From a management perspective, using these treatments to restore structure and function to riparian forests in the wake of eastern hemlock mortality would most likely not result in short-term lowered water quality that is common when overstory trees are harvested. In a study where ca. half of the riparian trees had been harvested with no buffer, Clinton (2011) showed an initial two-fold increase in stream NO_3-N that declined within the first year after harvest. Yeakley et al. (2003) reported a small, short-term increase in stream NO_3-N where a hillslope had been disturbed by a combination of cutting rhododendron (0.16 ha) and storm damage (0.21 ha) from a hurricane blowdown. No other post-disturbance changes in stream DOC, cations (Ca, Mg, K) or anions (PO_4, SO_4, Cl) were detected in either study (Yeakley et al., 2003; Clinton, 2011). More significant and longer-lasting responses have been documented where entire watersheds were clearcut (Swank and Webster, 2014; Jackson et al., 2018; Fakhraei et al., 2020). The magnitude of responses can vary depending on region (see review, Muwamba et al., 2019). For example, Meyer et al. (2014) showed that a whole watershed clearcut harvest resulted in reduced stream DOC for decades. In our study, we found higher than expected stream DOC concentrations and lower than expected NO_3-N after treatments at both CR and CFFR. Contrary to our expectations, however, we found that the stronger responses were at the CR stream for both of these analytes.

We expected more rapid responses in the treatment that employed fire because numerous studies show short-term declines in water quality; however, these largely depend on fire severity, spatial scale, and region. In the eastern US, stream inorganic nitrogen concentrations tend to be either unaffected or only temporarily enhanced immediately following fire (Hahn et al., 2019). Within the southern Appalachian Mountains, water quality responses to prescribed fire are also limited and short-term (Vose et al., 1999; Clinton et al., 2003; Elliott and Vose, 2005), and studies that employ cutting and fire in combination show slightly larger responses than low-intensity prescribed fire alone (Knoepp and Swank, 1993). In contrast, severe wildfires can impact stream chemistry and water quality (Neary et al., 2005; Smith et al., 2011; Hohner et al., 2019; Rhoades et al., 2019a; Uzun et al., 2020; Caldwell et al., 2020) and have long-lasting effects (Rhoades et al., 2019b); prescribed fires are typically less severe, particularly in the eastern US, and have limited or short-term effects on stream chemistry (Knoepp and Swank, 1993; Vose et al., 1999; Hahn et al., 2019; Majidzadeh et al., 2019). However, none of these studies examined the combined effects of cutting rhododendron followed by prescribed fire on stream chemistry. From a management perspective, employing fire to remove the rhododendron shrub layer did not result in lower water quality, making it a desirable operational tool to restore these forests.

5. Conclusions

As pests and pathogens eliminate foundational tree species in eastern deciduous forests, land managers need tools to restore the structure and function of these forests. For the restoration of many degraded riparian areas in the southern Appalachian Mountains to take place, the rhododendron shrub canopy must be reduced. This species inhibits tree recruitment and competes with remaining trees for water and nutrients. In two operational-scale rhododendron removal treatments, we show soil N increased, depending upon sampling time. The resulting soil increases were likely utilized by new tree and herbaceous growth and recruitment during the growing season a few months post-treatment. Because N, Ca, and Mg availability increased along with increased remaining tree water use (Dharmadi et al., unpublished data) and because in-stream processing of these increased (Dudley et al., in review), stream concentrations of these nutrients declined. This is an important finding as operational-scale terrestrial disturbances tend to increase stream dissolved inorganic N concentations. The fact that we found the opposite shows that restoration of these riparian corridors with these techniques may not lower water quality, and in fact may raise some, but not all water quality parameters. Namely, while we expected the acid-neutralizing capacity of these streams to increase, because these treatments lowered pH and some cations, we didn't see improvements in ANC during the study period. Finally, we found that DOC and Al increased initially in the CFFR treatment, and then the CR treatment. These generally returned to baseline levels in a year in the treatment that employed prescribed fire, however. While longer-term responses are unknown, we feel that this study shows short-term improvements in stream conditions restored with these techniques.

CRediT authorship contribution statement

Katherine Elliott: Conceptualization, Methodology, Formal Analysis, Writing-Original Draft, Supervision, Project Administration. **Chelcy Miniat:** Conceptualization, Formal Analysis, Writing-Review & Editing, Project Administration.

Declaration of competing interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2020.143270.

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