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Do southern Appalachian Mountain summer stream temperatures respond to removal of understory rhododendron thickets?

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Abstract

Dense understory thickets of the native evergreen shrub *Rhododendron maximum* expanded initially following elimination of American chestnut by the chestnut blight, and later in response to loss of the eastern hemlock due to hemlock woolly adelgid invasion. Rhododendron thickets often blanket streams and their riparian zones, creating cool, low-light microclimates. To determine the effect of such understory thickets on summer stream temperatures, we removed riparian rhododendron understory on 300 m reaches of two southern Appalachian Mountain headwater streams, while leaving two 300 m reference reaches undisturbed. Overhead canopy was left intact in all four streams, but all streams were selected to have a significant component of dead or dying eastern hemlock in the overstory, creating time-varying canopy gaps throughout the reach. We continuously monitored temperatures upstream, within and downstream of treatment and reference reaches. Temperatures were monitored in all four streams in the summer before treatments were imposed (2014), and for two summers following treatment (2015, 2016). Temperatures varied significantly across and within streams prior to treatment and across years for the reference streams. After rhododendron removal, increases in summer stream temperatures were observed at some locations within the treatment reaches, but these increases did not persist downstream and varied by watershed, sensor, and year. Significant increases in daily maxima in treatment reaches ranged from 0.9 to 2.6°C. Overhead canopy provided enough shade to prevent rhododendron removal from increasing summer temperatures to levels deleterious to native cold-water fauna (average summer temperatures remained below 16°C), and local temperature effects were not persistent.

KEYWORDS
canopy cover, rhododendron, riparian buffer, riparian vegetation, shade, stream temperature, understory canopy, vegetation change

1 | INTRODUCTION

Across the globe, forest composition is changing in response to climate change, changes in atmospheric deposition, management, and human alterations of disturbance regimes (Cramer et al., 2001; Wu et al., 2015). Altered forest composition and structure can affect nutrient, water, and energy cycling (McLaughlin et al., 2017; Pederson et al., 2014), impacting ecosystem services and driving further trophic changes (Nolan et al., 2018). Consequently, forest composition strongly influences many aspects of stream habitat. There is interest in utilizing forest management to better maintain water-based ecosystem services (Ford, Laseter, Swank, & Vose, 2011). Alteration to forest composition could involve a change to dominant canopy covers, altering basal resources and modifying stream-surface insolation and stream temperatures. Forest managers need to understand the benefits and drawbacks of different management strategies which could be employed to accommodate changes to forest structure and composition.

Vegetation changes that alter the dominant overstory and understory canopies can have significant implications on the amount of solar insolation reaching a stream, allowing for potential warming or cooling of the water temperature. Depending on the catchment and the region of study, radiative fluxes at the water surface are the dominant drivers of seasonal stream temperature variation and diurnal heating (Brown, 1972; DeWalle, 2008; LeBlanc, Brown, & FitzGibbon, 1997). As a result, shade or the lack thereof, strongly affects stream temperature regimes, as shown by stream temperature responses to many experimental clearcuts of small watersheds and their riparian zones (Burton & Likens, 1973; Hewlett & Fortson, 1982; Swift & Messer, 1971). Conversely, preservation of riparian shade can mitigate the stream temperature effects of clearcutting small watersheds (Bladon, Cook, Light, & Segura, 2016; Fraser, Jackson, & Radcliffe, 2011; Gomi, Moore, & Dhakal, 2006; Story, Moore, & Macdonald, 2003). Several studies have shown that any kind of shade will work to moderate solar-driven temperature fluctuations including logging debris (Jackson, Batzer, Cross, Haggerty, & Sturm, 2007; Kibler, Skaugset, Ganio, & Huso, 2013), topographic shade (Li, Jackson, & Kraseski, 2012), and artificial shade (Johnson, 2004; Klos & Link, 2018). In many forests, stream shade is provided by both an overstory canopy as well as a mid-story or understory canopy. Unexplored is the question of how removing a forest understory while leaving the overstory intact might change stream temperature regimes? In the mixed deciduous forests of the southern Appalachians, there is an overstory canopy providing significant shade in the summer, and in many areas, a dense rhododendron understory. The rhododendron understory contributes to overall shade, as well as alters wind and humidity near the channel, but in the context of the larger overstory, the magnitude of its influence on stream temperature is unknown.

Vegetation changes in the southern Appalachians have allowed rosebay rhododendron (*Rhododendron maximum* L.) to expand beyond its historical habitat, particularly along riparian areas where it forms stream-spanning thickets with the potential to play a large role in regulating stream temperature. Rosebay rhododendron is an evergreen clonal shrub that forms dense understories in the southern Appalachians. Rhododendron thickets create microclimates and local habitats characterized by: (a) reduction in sunlight available for photosynthesis [photosynthetically active radiation (PAR)] near the forest floor (Clinton, 2003; Lei, Semones, Walker, Clinton, & Nilsen, 2002; Nilsen et al., 2001); (b) production of allelopathic compounds that retard the germination or growth of other plant species (Baker & Van Lear, 1998; Lei et al., 2002); and (c) nutrient sequestration within rhododendron's recalcitrant forest floor litter (Chastain, Currie, & Townsend, 2006; Monk, McGinty, & Day, 1985; Wurzburger & Hendrick, 2007). Rhododendron has spread in the forest understory due to a combination of factors, including fire exclusion, loss of American chestnut [*Castanea dentata* (Marsh.) Borkh.] due to chestnut blight, and loss of eastern hemlock [*Tsuga canadensis* (L.) Carriere] due to hemlock woolly adelgid (*Adelges tsugae* Annand; Atkins, Epstein, & Welsch, 2018; Bolstad, Elliott, & Miniat, 2018; Elliott & Vose, 2012; Ford, Elliott, Clinton, Kloeppel, & Vose, 2012; Vose, Wear, Mayfield, & Nelson, 2013). Returning frequent fire to these ecosystems has been suggested as a way to control the spread of rhododendron and achieve a more desirable overstory community (Vose et al., 2013).

One concern in its removal is that rhododendron also has beneficial aspects for stream ecosystems. For example, slowly decaying rhododendron leaves provide basal food resources for stream ecosystems in the summertime, which are important for long-lived consumers such as crayfish (Huryn & Wallace, 1987; Schofield, Pringle, Meyer, & Sutherland, 2001). These slow-decaying leaves can also suppress summer algal growth (Dudley, 2018; Kominoski et al., 2015). Additionally, rhododendron thickets along riparian corridors reduce light reaching the forest floor and shade streams, influencing stream temperature dynamics by maintaining cooler water temperatures in the summer (Clinton & Vose, 1996). There are few studies investigating effects of understory removal on stream temperature, but the effect of rhododendron on forest floor light levels has been studied. Clinton (1995) compared forest floor light levels in plots with and without a rhododendron understory and found major seasonal increases in light, measured as PAR, in the plots lacking rhododendron (179 vs. 641 $\mu mol\ m^{-2}\ s^{-1}$). However, this increase was only substantial during leaf-off; once the over-story was fully established forest floor PAR was only slightly greater (14 vs. 65 $\mu mol\ m^{-2}\ s^{-1}$) in the plots without rhododendron. This suggests that rhododendron removal will likely increase direct solar insolation only slightly during the summer months and that resultant increases in stream temperature will be slight, particularly if the overstory canopy remains mostly intact.

Investigations of light levels following hemlock mortality have found that light penetration during the summer months is not greatly increased (Ford et al., 2012) and that thermal impact to streams would be short term (Siderhurst, Griscom, Hudy, & Bortolot, 2010), likely due to the combined influence of dense understory rhododendron and the presence of other trees in the overstory beyond hemlock. Many riparian corridors within the Nantahala and Pisgah National Forests contain dense understory rhododendron covers, and modelling (McDonnell et al., 2015) indicates that enhancing riparian cover with more dense canopy like that produced by rhododendron could help mitigate future stream warming from future air temperature increases forecasted by climate change models. We have found no studies specifically examining the effects of removing forest understories on stream temperatures, but studies of riparian forest overstory thinning have indicated that there are thresholds of canopy reduction that must be reached before there will be a response in the stream temperature (Klos & Link, 2018; Kreutzweiser, Capell, & Holmes, 2009; Rex, Maloney, Krauskopf, Beaudry, & Beaudry, 2012). For example, Kreutzweiser et al., 2009 found that partial-harvest logging in riparian buffers showed minimal effects on stream temperatures if 50% of the shading canopy remained intact, while Klos and Link (2018) found minimal changes to incoming radiation if 50% of the overstory was still intact.

To address the stream temperature questions raised by rhododendron understory management, we conducted two paired watershed experiments. Both watersheds received the same treatment, where the rhododendron understories were experimentally removed within the riparian areas of watersheds where dead and dying eastern hemlock constituted a significant portion of the otherwise intact overhead canopy. By experimentally removing only the rhododendron understories from riparian forests, we asked the following questions: (a) How do summer stream temperatures respond to rhododendron removal? and (b) Will temperature increases associated with rhododendron removal reduce available cold-water habitat?

2 | MATERIALS AND METHODS

2.1 | Site description

Four perennial second-order streams were selected within the White Oak Creek watershed (35°20′ N latitude, 83°58′ W longitude) in the Nantahala River drainage, approximately 10 km west of Franklin, NC, by the USDA Forest Service land managers (National Forest System, Nantahala Ranger District; Figure 1). These stream reaches had similar abundances of rhododendron and dead hemlock (Elliott & Miniat, 2018). Across sites, slopes were moderate-to-steep (30–60%) and elevation ranged from 1,160 to 1,390 m (Table 1). Annual rainfall averaged 1,900 mm and mean annual temperature was 10.8°C. The stream basins were either adjacent to one another or in close proximity, each draining between 0.93 and 2.91 km^2 (Table 1, Figure 1). Streams were similar in their average bankfull channel widths (3.42–4.53 m) and channel slopes (2.53–3.52%). Average bankfull widths were measured in the field. Mean channel slope was determined from site LiDAR, and drainage area was delineated using a LiDAR-derived digital elevation model. Elevation ranges were also determined from the site digital elevation

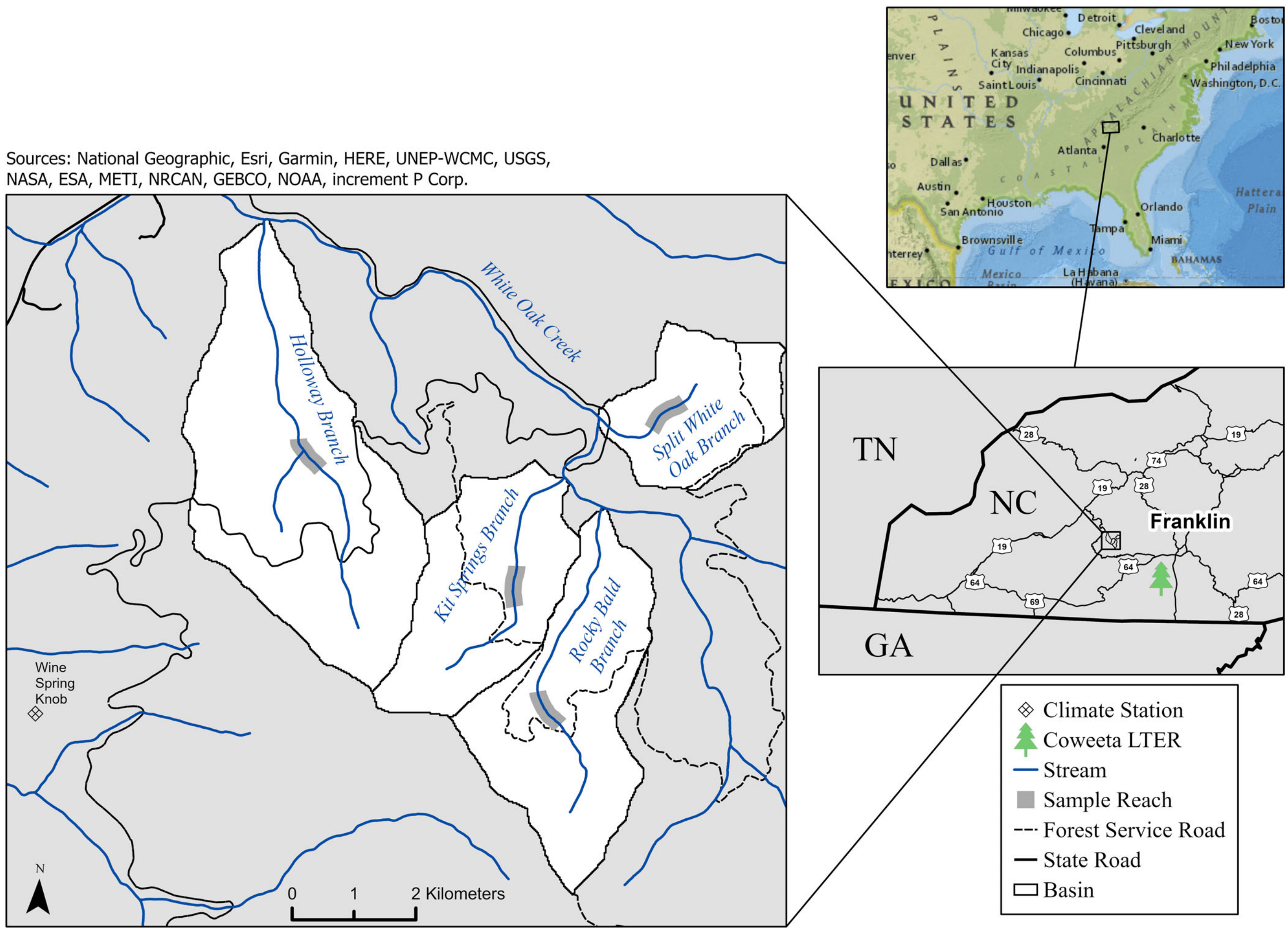

FIGURE 1 Map of study area and focus stream reaches

TABLE 1 Characteristics of the four stream reaches at the White Oak Creek watershed, western NC

Stream	Treatment type	Drainage area at bottom of study reach (km^2)	Elevation range at reach (m)	Elevation range of watershed (m)	Average bankfull width (m)	Mean channel slope(%)	Forested percent (%)	Orientation
KS_R	Reference	0.93	1,255–1,315	1,170–1,627	4.53	3.52	97.7	NE
RB_R	Reference	1.42	1,350–1,390	1,198–1,640	3.92	3.16	99.4	NW
HW_T	Cut, piled	2.91	1,160–1,210	1,024–1,625	3.69	2.66	95.4	NW
SWO_T	Cut, scattered, burned	1.76	1,195–1,265	1,149–1,451	3.42	2.53	99.1	WSW

Note. Drainage area and elevation ranges were determined from LiDAR-derived digital elevation model. Bankfull widths were measured in the field. Channel slope was defined from site LiDAR, and percentages of forest cover were pulled from the 2016 National Land Cover Database (NLCD).

model. Holloway Branch, which drains the largest area, is alluvial in nature, and the other three streams are colluvial in nature. In many locations, the channels are braided, and substantial down valley flow can be heard moving below the floodplain surface. Furthermore, the channels are highly dynamic, with boulders and wood moving during common stormflows. For this reason, flow measurements were not taken to accompany temperature measurements. The basins are fully forested, with less than 5% of land cover in unpaved forest roads. Landcover was determined based on the 2016 National Land Cover Database (NLCD). Reference sites had a north-east or north-west orientation, while the treated watersheds faced north-west and west south-west.

At each of the four stream reaches, a treatment area of 300 m length and 50 m on each side of the stream (≈3 ha) was delineated. The treatment length was limited to 300 m because choosing larger rhododendron patches would have required selecting streams from a much larger geographic area, and 300 m was judged to be sufficient to test the ecological questions posed by the larger study. Each site received one of three treatments as follows: Holloway (HW_T) – rhododendron was cut and scattered; Split White Oak (SWO_T) – rhododendron was cut, scattered, and burned; and both Rocky Bald (RB_R) and Kit Springs (KS_R) were unmanipulated and served as reference streams. To minimize rhododendron regrowth, an immediate application of herbicide (triclopyramine Garlon 3A, DOW AgroSciences, Indianapolis, IN) formulation (44.4% Triclopyr Triethylamine Salt, with an aquatic label mixed to a ratio of 50% herbicide/50% water) was applied to cut stumps, and thereafter, periodically applied to new rhododendron sprouts as needed. At SWO_T, prescribed fires were hand lit across the entire delineated stream reach (3 ha), following the guidance specified in the USDA Forest Service, Nantahala National Forest, Prescribed Burning Plan (USFS, 2011). Throughout the post-treatment monitoring period, the rhododendron canopy was completely absent over HW_T and SWO_T creeks while the rhododendron canopy was unmanipulated in RB_R and KS_R. Rhododendron cutting (HW_T, SWO_T) occurred in spring (March–May) 2015, and the prescribed fire (SWO_T) was implemented in spring (March) 2016 (Elliott & Miniat, 2018).

TABLE 2 Summer climate data during the period of study (2014–2016), June 1 to September 1

	Period		
Variable	Pre-treatment	Post +0	Post +1
Air T_{min} (°C)	9.8	9.6	10.5
Air T_{mean} (°C)	18.1	18.9	20.0
Air T_{max} (°C)	27.8	28.8	28.6
Precipitation (mm)	443	482	371

2.2 | Climatic conditions

Climatic measurements were taken at the Wine Spring Knob station, located in the Nantahala National Forest, within 5 km of all watersheds. The Wine Spring Knob station was maintained by the Coweeta Long Term Ecological Research (LTER) station. Averages of minimum, mean, and maximum air temperature were taken every hour (Table 2). Similarly, precipitation was averaged on an hourly time step and then totalled to give a daily rainfall total. Climatic measurements were not made at each stream because of the dense canopy.

2.3 | Canopy measurements

We measured canopy cover at the reach-scale in August of each year. We had 10 sampling locations within each reach spaced approximately every 30 m within the reach (starting at 30 m). At each sampling location within a given reach, we took canopy cover measurements using digital photographs in four directions (upstream, downstream, left bank, right bank; Canon Powershot SD890 IS (Melville, NY) and Ricoh WG-4 GPS (Rungis, France). Digital photographs were taken perpendicular to the stream bed from a height of 3 m, which enabled us to capture all canopy cover that crossed the stream at each of the photograph locations. For each photograph, cover type (rhododendron leaf or stem, non-rhododendron leaf or stem, or open) was recorded for 200 randomly distributed points using ImageJ software (Schneider, Rasband, & Eliceiri, 2012). Percent rhododendron canopy cover was calculated by dividing the points covered by rhododendron by 200, and percent total canopy cover was calculated by dividing the total number of points covered by either plant type by 200. At each sampling location, we averaged percentages within each cover type for the four photographs.

2.4 | Stream temperature measurements

Sensors were deployed from June 1 to September 1 of each year (2014–2016). Stream temperature was recorded at a 30-min time step at each site with Onset HOBO Pendant (8 and 64 K) temperature data loggers (Onset Computer Corporation, Bourne, MA), resolution: ± 0.14°C at 20°C. To prevent loss and drift during high flow events, loggers were zip-tied to standard modular bricks prior to placement into the stream channel. Nevertheless, some loss and some movement of loggers occurred. Loggers were placed 50 m upstream of the treatment area, 50 and 100 m downstream, and throughout the 300 m treatment area (Figure 2). Sensors were placed in mid-channel pools to lessen the chance of dewatering and to measure temperatures representative of the thalweg. Sensors were placed at the start and end of the treatment area, as well as every 100 m during the pre-treatment and first summer. Deployment was intensified in the treatment streams during the second summer following removal, with loggers placed every 30 m, co-located at positions where canopy measurements were taken. Reference streams were left in the pre-treatment configuration.

Sensor recovery was variable year to year (Supporting Information). Channel beds were largely gravel/cobble and subject to bed mobility and some realignment during high stormflows. There were data gaps due to sensor failure, dewatering, or loss of sensor due to bed-scouring and flushing by high flows. The downstream sensors at SWO_T dewatered or were lost during both years following treatment. The upstream sensor at SWO_T was lost during the second year following treatment. Sensors recovered were typically within 1 m of deployment site. Full sensor recovery is described in the Supporting Information along with data cleaning procedures.

2.5 | Statistical analysis

2.5.1 | Temperature differences between and within streams, and across years

A univariate approach to a repeated-measures ANOVA was conducted to look at temperature differences among streams (both reference and treatment streams), and to look at year to year differences within the reference streams. A linear mixed effect model was fit with a restricted maximum likelihood approach using the *lme* function of the *nlme* package (version 3.1-141). The Tukey's Studentized Range (HSD) Test was used to investigate pairwise differences among means using the *glht* function of the *multicomp* package (version 1.4-10).

2.5.2 | Assessing rhododendron removal

To assess the effects of rhododendron removal on summer stream temperatures, regression relationships were developed for daily minimum, average, and maximum stream temperatures. These regression relationships were based on temperature sensors within the focal 300 m reaches of the reference and treatment streams, during the pre-treatment year. Because of the high variability in stream temperature prior to rhododendron removal and to get around unequal sample design, only the average stream temperature across all sensors in the treatment section response was modelled. We defined the mean treatment response as the average temperature from sensors located within the treatment block, for each metric (i.e., average high across all sensors in the treatment block), at a given point in time. The variation from that response was then evaluated for each of the two

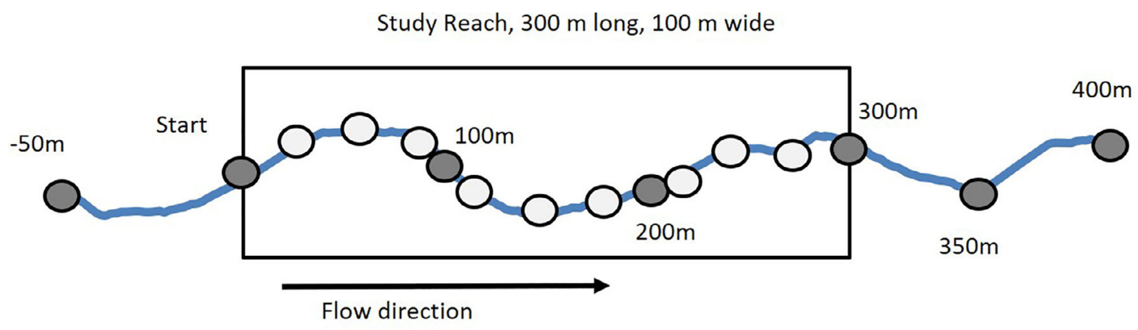

FIGURE 2 Temperature logger layout, with low spatial resolution of 2014 and 2015, high spatial resolution in 2016. For all streams and times, one sensor was placed 50 m above the 300 m study reach, and one sensor was placed 50 m below the end of the study reach. 2014 through 2016, an additional sensor was placed 100 m below the end of the study reach (400 m). In 2014 and 2015, four sensors were placed in the 300 m study reach, at the start, 100 m, 200 m, and the end (300 m). This sensor arrangement is shown with the dark circles. In 2016, spatial sampling frequency was intensified in the treatment streams to more precisely assess the spatial scale of temperature changes. In the treatment streams, temperature sensors were placed every 30 m within the treatment area. This sensor arrangement is shown with light circles. All longitudinal distances are stream distances

treated streams to understand individual stream response to rhododendron removal. The pre-treatment regression model used an autoregressive integrated moving average (ARIMA) model with external regressors, following the methods of multiple studies examining patterns of stream temperature (e.g., Gomi et al., 2006; Guenther, Gomi, & Moore, 2014; Moore, Sutherland, Gomi, & Dhakal, 2005; Equations 1 and 2). Model fitting was completed using a maximum likelihood approach, using the *arima* function of the *stats* package (version 3.5.2) within R.

$$T_t = \beta_0 + \beta_1 R_t + \beta_2 \sin(2\pi d/d_y) + \beta_3 \cos(2\pi d/d_y) + \varepsilon_t \quad (1)$$

where $T_t$is the predicted temperature in the treatment watershed at a given time, t; $\beta_0\beta_1$, β_2, and β_3 are the model parameters fitted in the ARIMA model; R_t is the temperature in the reference watershed at the given time, t; d is the day of the year, while d_y is the number of days in a year (365.25); sine and cosine terms address interseasonal variations in stream temperature (Guenther et al., 2014; Watson, Vertessy, McMahon, Rhodes, & Watson, 2001); ε_t is an autoregressive error term of some order, k. ε_t is further defined as:

$$\varepsilon_t = \rho_1 \varepsilon_{t-1} + \cdots + \rho_k \varepsilon_{t-k} + u_t \quad (2)$$

Where $\rho_k$is an autocorrelation coefficient for the error term, ε_{t-k} in terms of the maximum lag, k. The optimal k values were determined from the ARIMA model as the value with the lowest Akaike information criterion value. u_t is the random disturbance in stream temperature, which is assumed to be normally distributed, with a constant variance. These regression coefficients are stationary and independent of the predictive variable (R_t). This is an important consideration, as other potential system state variables (discharge conditions, air temperature, antecedent moisture conditions, etc.) are also independent of these coefficients, and in this modelling framework will not influence the predicted stream temperature.

Uncertainty from the model was calculated in the form of upper and lower prediction limits at a 95% confidence via Monte Carlo simulations, following the structure of Guenther et al. (2014) and Leach, Moore, Hinch, and Gomi (2012). These limits were predicted to account for uncertainty in model parameters (Table 3). The *rmvnorm* function within the *tmvtnorm* package (version 1.4-10) was used as a multivariate random number generator to develop variance–covariance matrices for each realization of the model.

The models were fit based on the average temperatures of all sensors within the focal 300 m study reach for each pair of reference and treatment streams. Sensors upstream and downstream from those treatment areas were not included in the time series regression (ARIMA). We used separate regressions for each treatment stream with both reference streams and used the pair that provided the best model for the pre-treatment year for analysis of change in subsequent post-treatment years. Model fit varied little between models (Table 3 and Table S2). All models, even those not selected, had very high R^2 values, exceeding .867, and RMSE values below 0.222. We selected the best model for each treatment watershed and temperature metric based on both visual inspection of the predicted and observed temperature time series and the R^2, RMSE, and bias statistics of the models. Temperatures in HW_T were predicted as a function of the temperature in KS_R. Temperatures in SWO_T were predicted as a function of the temperature in RB_R. With the exception of daily maximum temperature in SWO_T, we selected the model pairs which had a lower RMSE. Visual inspection of the SWO_T observed and predicted time series showed that this model better capture diurnal highs and lows, and by pairing SWO_T with RB_R we kept consistent pairs across all metrics. These predictions are based on the pre-treatment year model, so major deviations from these predictions can be attributed to the treatment effect. Across all models, we noted consistent but small low bias. Deviations of each sensor from the mean behaviour of the focal reach were used to assess spatial variations in temperatures and treatment effects.

Deviations from each model were calculated as the difference between the recorded stream temperature (T_{obs}) and simulated stream temperature (T_t) from the pre-treatment model (Equation 3).

$$T_{error} = T_{obs} - T_t \quad (3)$$

These deviations were calculated for each of the treatment watersheds during each period of the study, and for each of the three temperature metrics of interest (daily minimum, mean, and maximum temperature). To test for bias in model predictions during the pre-treatment period, one sample t tests (μ = 0, p < .05) were used to

TABLE 3 Summary of model coefficients from time series regression

Metric	Site	Predictor	*df*	*k*	β_0	β_1	β_2	β_3	ε_t	R^2	Bias	RMSE
Daily maximum	HW_T	KS_R	89	5	2.19	0.90	−0.03	0.22	0.16	.96	−0.034	0.116
	SWO_T	RB_R	89	2	3.98	0.83	−0.44	0.15	0.61	.90	−0.838	0.219
Daily average	HW_T	KS_R	89	8	1.62	0.94	−0.02	0.11	0.38	.98	−0.270	0.085
	SWO_T	RB_R	89	1	2.20	0.95	−0.37	−0.06	0.74	.96	−0.857	0.140
Daily minimum	HW_T	KS_R	89	9	1.47	0.95	0.01	0.12	0.07	.99	−0.254	0.076
	SWO_T	RB_R	89	1	1.73	1.01	−0.25	0.09	0.55	.96	−0.419	0.134

Note. df refers to degrees of freedom, *k* is order of residual autocorrelation, β_x are fitted estimates of model parameters, ε_t is the sum of all autoregressive error terms and autocorrelation coefficients ρ_k.

determine if deviations were statistically significant. Bias estimates from the pre-treatment models indicate that the models underestimated stream temperature in pre-treatment summer from 0.034 to 0.857°C. This pre-treatment bias should be considered when assessing treatment effects calculated from model deviations. During the post-treatment periods, a two-sample *t* test ($\mu_{post\text{-}treatment} > \mu_{pre\text{-}treatment}$, $p < .05$) was used to test if model deviations constituted a significant change in temperature relative to the bias of the pre-treatment model. Without accounting for this pre-treatment model bias (whether the pre-treatment model over or underestimated temperature), we would likely be incorrectly estimating the effect of treatment.

3 | RESULTS

3.1 | Canopy measurements

Prior to treatment, all four streams featured total summer canopy cover in the range of 89.4–93.8%, but rhododendron canopy varied substantially, from a low of 28.8% in KS_R to a high of 54.0% in SWO_T (Figure 3). Proportions of rhododendron contribution to canopy cover were quite different between the two reference streams: 28.8 versus 48.6% for KS_R and RB_R respectively, and both showed inter-annual variability in measurements but no trend through the study period (Figure 3). In both treatment streams, rhododendron canopy was reduced to near-zero both years following removal. However, total canopy cover was reduced only a little by the loss of rhododendron canopy, decreasing by only 3.9% in HW_T and 6.2% in SWO_T in the first year post-treatment, and further to 5.3 and 6.9% less than the pre-treatment average in the second year post-treatment. Total canopy cover was still high in the treatment streams in the second post-treatment year at 84.2 and 82.4% in HW_T and SWO_T, respectively.

3.2 | Environmental variability in stream temperatures

Daily average and daily maximum air temperatures were warmer in the second year post-treatment, an increase of 2°C for the daily average and 1°C for daily maximum (Table 2), relative to the pre-treatment year. Average minimum daily temperatures varied by only 0.9°C among years. The second post-treatment summer received between 60 and 100 mm less rainfall than in the preceding two summers (Table 2).

Stream temperatures varied among streams and within individual streams in the pre-treatment year, and across years in the reference streams. Stream temperatures were significantly different among streams in the pre-treatment year (ANOVA, $p < .0001$; Figure 4), with means ranging from 12.5 to 14.2°C (Table 4). Both reference streams were cooler than the streams designated for treatment (Figure 4). This might reflect the fact that the reference watersheds were a little higher and had lower drainage areas (Table 1). They did, however, have wider channels. Sensors placed in the same stream but at different locations featured significant, 0.6–1.5°C, differences in temperatures during the pre-treatment year (ANOVA, $p < .0001$; Figure 4). During the pre-treatment year, temperatures in HW_T decreased going downstream, but in SWO_T temperatures increased going downstream.

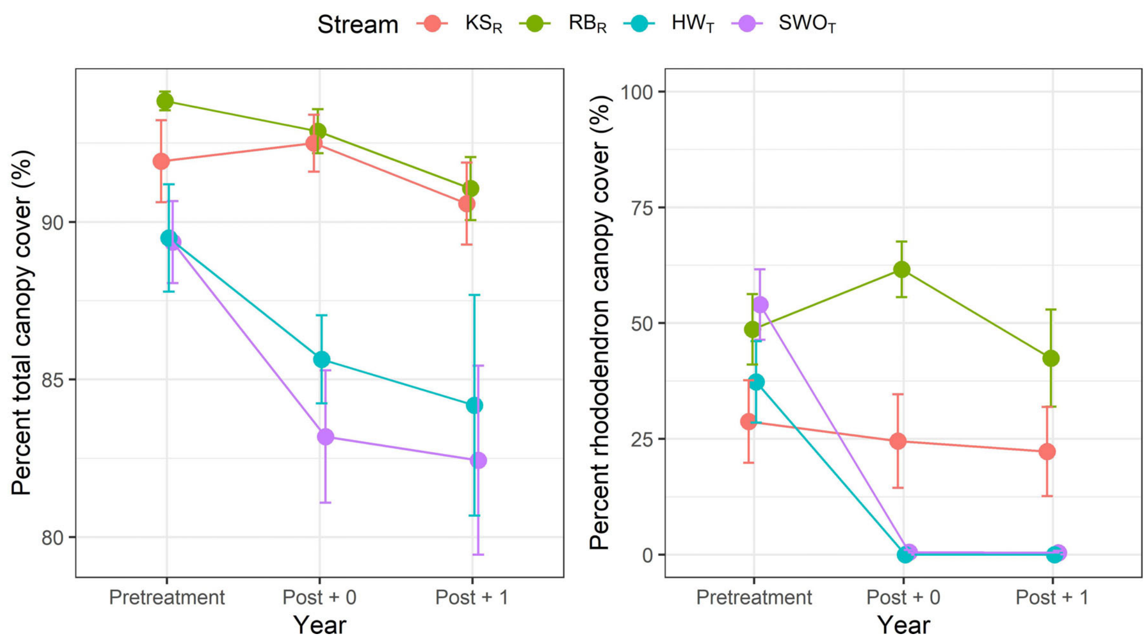

FIGURE 3 Canopy cover measured in August of each year based on digital photographs taken. *Note:* Points on photographs were classified based on cover type, and percentages of rhododendron and total canopy coverage were calculated based on their relative abundance in each photograph. Error bars reflect 1 standard deviation from the mean

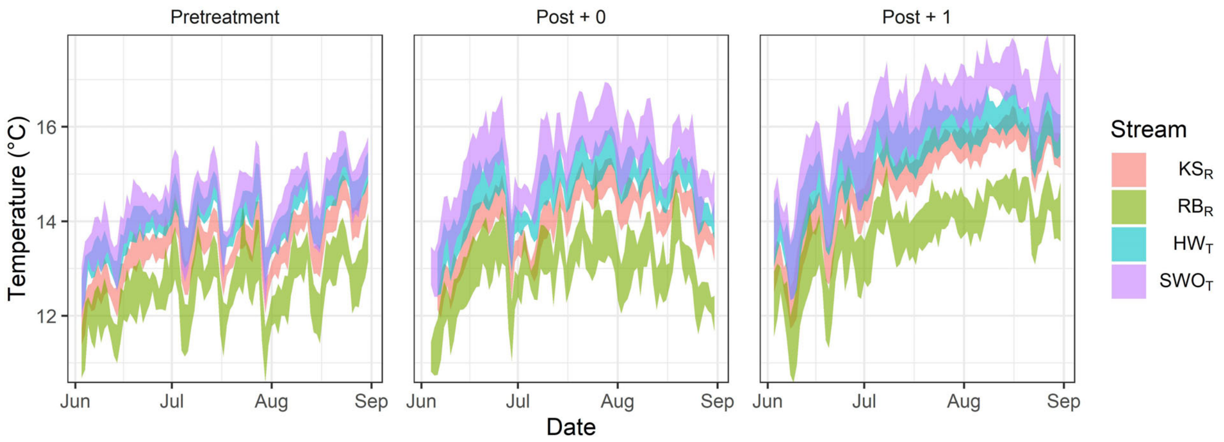

FIGURE 4 Daily variation in summer stream temperature across all reference and treatment sites. Two reference streams (KS_R and RB_R) are cooler than both of our treatment streams (SWO_T and HW_T) across all 3 years. The difference in temperature between streams could be upwards of 4°C at a given time. The vertical width of each band corresponds to the daily range in stream temperature

TABLE 4 Summary of average ($\bar{x}$) summer (June 1–Sept 1) stream temperature across all of the sites and all sensors within each site, with standard deviation (σ)

		Pre-treatment		Post +0		Post +1				
Site	Treatment	$\bar{x}$	σ	$\bar{x}$	σ	$\bar{x}$	σ	δ across streams	Post +0 to pre-treatment	Post +1 to pre-treatment
KS_R	Reference	13.43	0.73	14.01	0.70	14.85	1.05	0.89	0.59	1.42
RB_R	Reference	12.53	0.59	12.92	0.65	13.68	0.79	Coldest	0.39	1.15
HW_T	Cut, scattered	14.00	0.63	14.54	0.67	15.32	0.90	1.47	0.53	1.32
SWO_T	Cut, scattered, burned	14.23	0.72	15.07	0.80	15.87	1.06	1.69	0.85	1.64

Note. δ across streams refers to temperature difference from the coldest site during the pre-treatment period. Changes relative to the pre-treatment year are noted for the post-treatment periods.

Temperatures in the two reference streams (RB_R and KS_R) differed across all years (ANOVA, $p < .001$), getting warmer each year of the study. Average reference stream temperatures were 1.2–1.4°C warmer in the second post-treatment year than in the pre-treatment year (Table 4), consistent with the differences in air temperatures observed across these summers (Table 2). Even with warming across years and treatment effects described below, average summer temperatures never exceeded 15.9°C in any stream.

3.3 | Stream temperature response to rhododendron removal

Based on modelled stream temperature values, T_t, summer stream temperatures significantly increased in the treatment reaches relative to the reference reaches, but the effect varied considerably across post-treatment years, sites, and sensor position (Table 5). In the first post-treatment summer, daily minimums deviated from the pre-treatment model predictions by +1.63°C and + 2.41°C in SWO_T and HW_T (Table 5). Daily mean temperature deviated by +1.28°C and +1.93°C in SWO_T and HW_T. Daily maximums did not significantly increase during this first summer (Table 5; $p > .05$). By the second summer post-treatment summer, daily minimum stream temperature within the treatment reach of HW_T and SWO_T deviated from pre-treatment model predictions by +1.63 and +3.33°C (Table 5). Daily mean stream temperatures deviated from pre-treatment model predictions by +1.26°C in HW_T, and +2.98°C in SWO_T. Finally, daily maximums deviated from pre-treatment model predictions by +0.85 and +2.61°C in HW_T and SWO_T, respectively (Table 5). Due to negative bias in all models for the pre-treatment period, these treatment effects are likely slightly low based.

HW_T and SWO_T differed in their treatment responses. In the first post-treatment year, HW_T displayed limited increases in temperature, within the range of model error, while the temperature in SWO_T showed an increase outside of model error. Spatial characterization of SWO_T temperature responses during this first post-treatment year was limited due to sensor loss, as three of six sensors were not recovered. During the second year post-treatment, both streams had pronounced deviations from pre-treatment across all metrics (Table 5). There was an increase in model residuals during this period across all metrics, but most notably for daily maxima (Figure 7). SWO_T showed a distinct warming of the treatment stream in the second year post-treatment, specifically for daily maximum temperature (Figures 5 and 6). This effect was noted in the daily

TABLE 5 Results of time series regression, indicating deviations from pre-treatment model predictions

				Deviation from T_t		
Metric	Site	Period	T_t	Min	Mean	Max.
Daily minimum	HW_T	Pre-treatment	13.72	−0.32	−0.04	1.19
		Post +0	14.23	−0.54	0.06	1.63
		Post +1	14.94	−0.82	−0.11	1.63
	SWO_T	Pre-treatment	13.84	−0.41	−0.06	1.61
		Post +0	14.66	−0.12	0.31	2.41
		Post +1	15.36	−0.23	0.27	3.33
Daily mean	HW_T	Pre-treatment	14.00	−0.68	−0.04	0.88
		Post +0	14.54	−0.69	0.09	1.28
		Post +1	15.32	−1.18	−0.06	1.26
	SWO_T	Pre-treatment	14.23	−1.03	−0.04	1.12
		Post +0	15.07	−0.35	0.36	1.93
		Post +1	15.87	−0.75	0.45	2.98
Daily maximum	HW_T	Pre-treatment	14.34	−1.13	−0.04	0.35
		Post +0	14.98	−1.16	0.21	0.86*
		Post +1	15.87	−1.68	0.13	0.85
	SWO_T	Pre-treatment	14.69	−1.59	−0.02*	0.58
		Post +0	15.66	−0.94*	0.51	1.47*
		Post +1	16.55	−1.34	0.82	2.61

Note. T_t refers to temperature predicted in the treatment stream based on the accompanying reference stream temperature, R_t. Deviations from T_t refer to how much individual temperature sensors deviated from the temperature predicted from the pre-treatment model. All deviations marked * indicate they are not statistically significant.

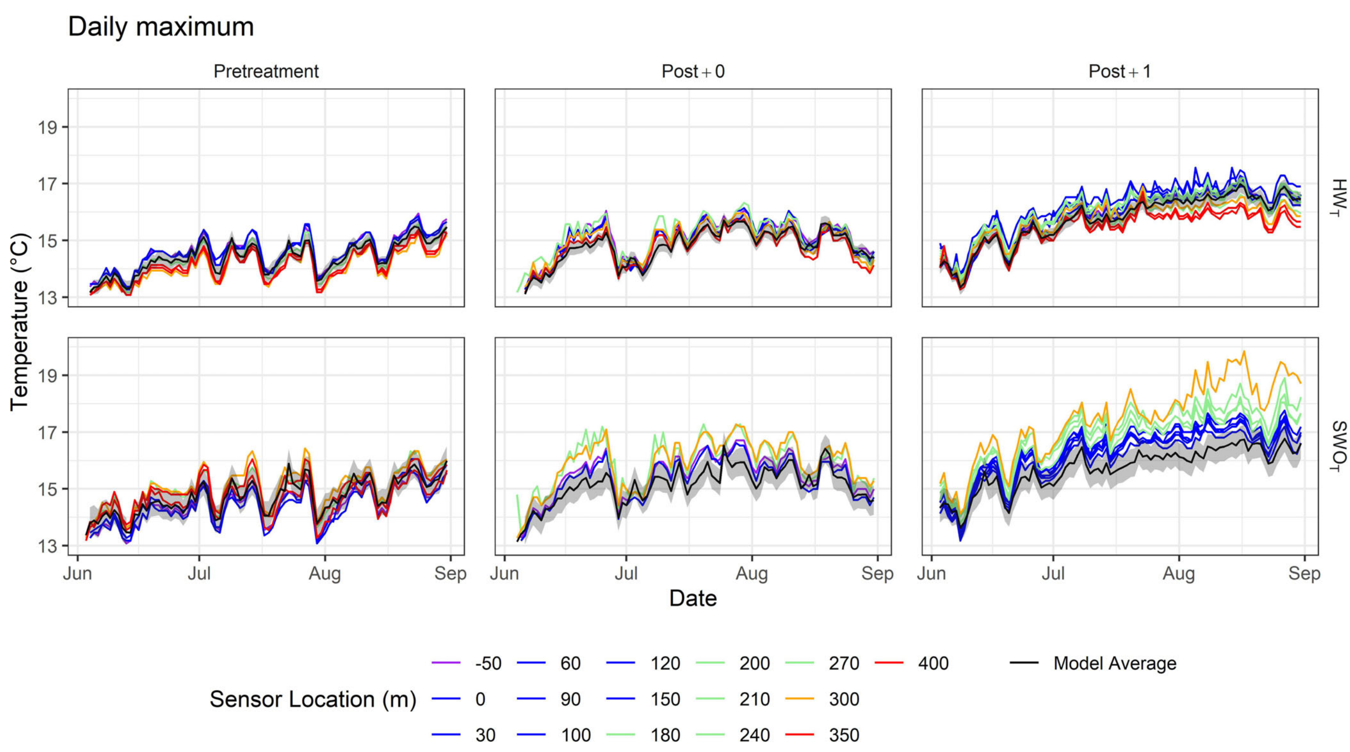

FIGURE 5 Downstream spatial variations in maxima daily stream temperature post-treatment. Grey areas surrounding model average indicate the model uncertainty generated from Monte Carlo realizations

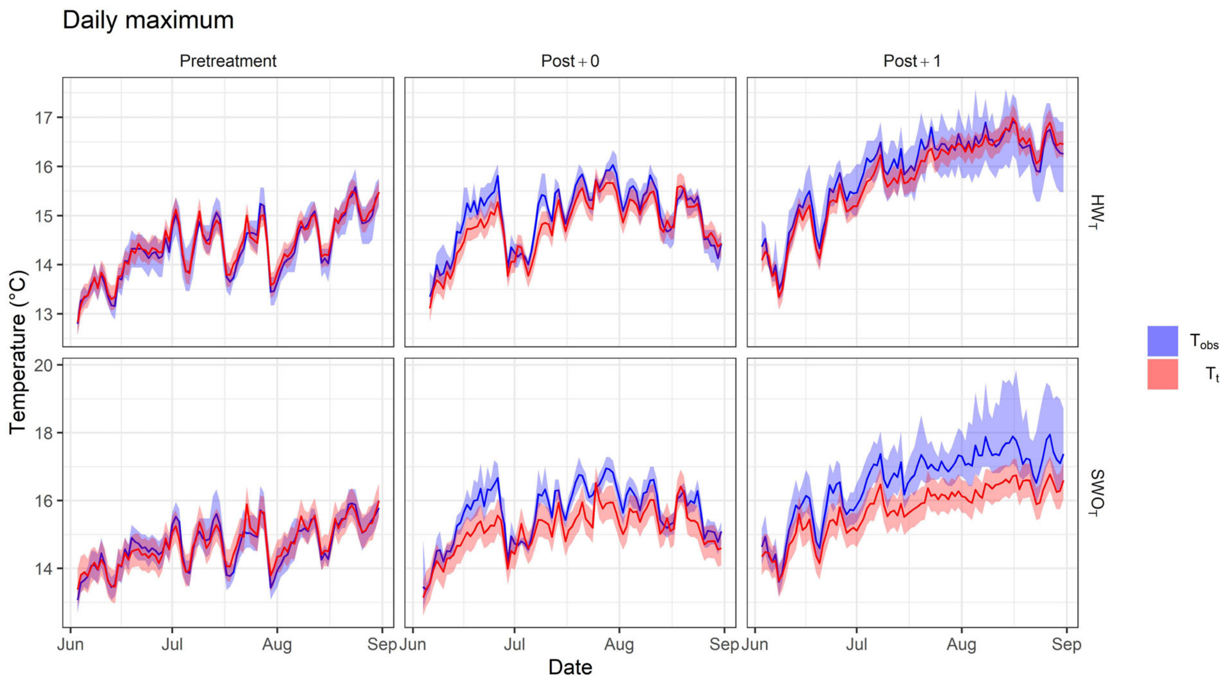

FIGURE 6 Spatial variation in observed stream temperature (T_{obs}) compared to the range in model estimated stream temperature (T_t). Blue solid line represents the average daily maximum stream temperature across all temperature sensors. Light blue ribbon indicates the range in observed (T_{obs}) daily maxima across all temperature sensors. Red solid line represents the model-average daily maximum stream temperature (T_t), with the light red ribbon displaying the upper and lower bounds of model uncertainty obtained via Monte Carlo realizations

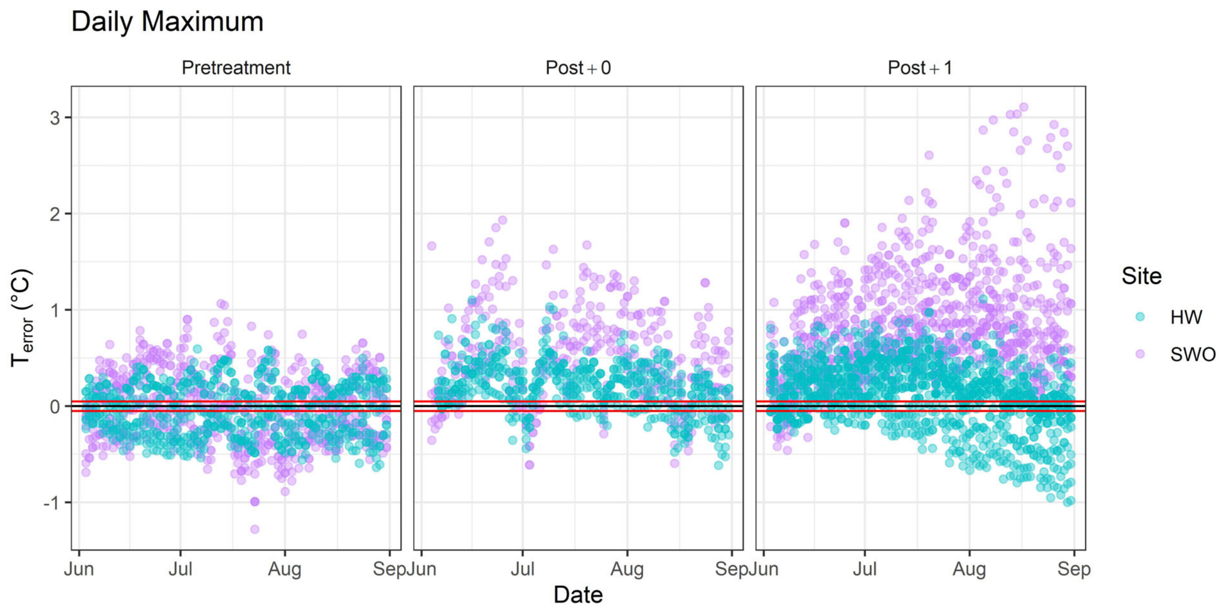

FIGURE 7 Deviations from pre-treatment model across all sensors in each treatment watershed. T_{error} is calculated as the difference in temperature between an individual sensor (T_{obs}) and the predicted stream temperature (T_t). Red horizontal lines indicate 95% prediction intervals for the difference between modelled and observed temperature. These confidence intervals are based on predicting the mean temperature, which is why the pre-treatment point cloud is more narrow and evenly distributed around 0 relative to the post-treatment periods

mean and minimum temperatures, but to a lesser degree than daily maximums (Figures 4 and 5).

Assessment of the downstream treatment effects at SWO_T was hampered by the loss of both downstream temperature loggers during the years following treatment. Stream temperature in HW_T during the first year following treatment displayed limited changes relative to the pre-treatment period (Figure 7). In the second year after treatment, HW_T showed cooling downstream of the treatment area, though there was significant noise in this assessment across all temperature metrics (Figures 5 and 6). These downstream sensors displayed the largest deviations from pre-treatment. Difference in daily maximum temperatures between the most upstream and downstream sensors during the pre-treatment year was 0.77°C for HW_T and 0.96°C in SWO_T (Figures 5 and 6). During the second year post-treatment, this difference increased to 1.05 and 2.85°C for HW_T and SWO_T, respectively. Deviations from pre-treatment increased as the summer progressed, in both treatment sites (Figures 4–6). Model residuals during the pre-treatment year for both sites are relatively uniform across the summer (Figure 7). The distribution of these residuals in the second summer following treatment shifted further away from the one to one line, especially later during monitoring period (August–September).

4 | DISCUSSION

The data indicate that rhododendron thickets moderate summer stream temperatures. Although rhododendron removal only marginally reduced total canopy coverage, it caused spatially and temporally variable, but significant, increases in summer stream temperatures. Rhododendron removal reduced total canopy cover by 3.85–6.92% and consequently increased summer stream temperatures, but the temperature effects varied by stream, by year, and by sensor location in each stream. The maximum estimated treatment effect for daily maximums in SWO_T was 2.61°C, but only 0.86°C for HW_T (Table 5). The variability of responses matched the variability of temperature measurements within the same channel. There were differences of up 4°C at the same time interval at different sensors within SWO_T (Figures 5 and 6). During the pre-treatment period, intra-stream variability was much more constrained, typically closer to 1°C. This indicates that the effect of rhododendron removal on stream light was patchy, with patchy effects on temperature. The greater increase in stream temperature noted at SWO_T could be magnified due to stream orientation when compared to HW_T. SWO_T was more of a south-facing reach, while HW_T was north facing. Daily minimum temperatures were also affected by rhododendron removal. From the pre-treatment to second post-treatment year, mean daily minimums at both treatment sites were considerably higher (1.22°C at HW_T, 1.52°C at SWO_T). Previous work has consistently found daily maxima, rather than daily means or minimums, to be most affected by changes in canopy cover (Isaak & Hubert, 2001; Mellina, Moore, Hinch, Macdonald, & Pearson, 2002; Moore et al., 2005).

4.1 | Response in context of other canopy removal studies

Summer stream temperature in southern Appalachian streams responded to modest canopy reductions (over 80% of total overhead canopy remained) that were created by rhododendron removal. The effects documented here were smaller than those documented for clearcut experiments where canopy loss is much larger. Full canopy removal by clearcutting has resulted in temperature increases of 2–8°C, with rates varying by stream and by year in coastal British Columbia (Gomi et al., 2006). Hewlett and Fortson (1982) documented 6–7°C increases in stream water temperature following the clearcutting of a pine plantation in the Georgia Piedmont. Johnson and Jones (2000) reported a 7°C increase in maximum summer stream temperature following clearcutting, also noting that maximum summer stream temperatures occurred earlier in the summer. Increases of up to 12°C in maximum stream temperatures have been recorded following a clearcut in the Oregon Coast Range (Harris, 1977). Daily maxima responded to a clearcut in western Washington from −0.1 to 3.6°C depending on the type of riparian buffer left in-tact (Janisch, Wondzell, & Ehinger, 2012). Janisch et al. (2012) noted highly variable responses in the temperature sensitivity between streams, as we saw in the two treatment streams here. The temperature effects of rhododendron removal were similar to those reported for a partial forest harvest where stream temperatures increased from 1.6 to 3°C depending on sensor location (Guenther et al., 2014). Another of Harris (1977) study sites underwent a partial harvest (20% harvested) and noted a 2°C increase in monthly maxima temperatures. Harr and Fredriksen (1988) found a 2.5–3°C increase in maxima stream temperature after a partial forest harvest (25%) in the Oregon Cascades. The within and across-stream treatment variability observed in our streams were similar to variability in temperature responses in other studies of full and partial canopy removal.

Solar insolation is a primary driver of summer stream temperature regimes, with studies on clearcutting effects on stream temperature showing that the loss of shade provided by canopy is the primary driver of change (Bladon, Segura, Cook, Bywater-Reyes, & Reiter, 2018; Davis, Reiter, & Groom, 2016; Johnson, 2004). The removal of riparian buffers produces a limited stream temperature response if the dominant shading canopy is left mainly intact (Kreutzweiser et al., 2009). In our study, rhododendron only contributed 3.85–6.17% of the total canopy cover in each of the treatment areas, but it may have had a disproportionate effect on riparian microclimate. Removal of a small portion of total canopy cover would be expected to produce a limited temperature response. The remaining overstory canopy cover provided enough shade to prevent large systemic summer stream temperature effects, regardless of the higher solar insolation from rhododendron removal and the continued decline of hemlocks at the sites that create additional overstory canopy gaps. The spottiness of temperature effects in the treatment streams reflects the fact that while overall canopy cover loss was low, specific locations in the stream saw much higher canopy losses particularly were the loss of hemlock canopy coincided with removal of understory canopy. The sensitivity of stream temperature to rhododendron removal is likely to increase where there are overhead canopy gaps (Warren, Collins, Purvis, Kaylor, & Bechtold, 2017; Warren, Keeton, Bechtold, & Rosi-Marshall, 2013). These responses are consistent with work on forest floor light levels in areas with and without understory rhododendron, where marginal increases in PAR were found in areas without rhododendron compared to areas with rhododendron during the growing season (Clinton, 1995).

4.2 | Freshwater habitat considerations

Despite significant increases in summer daily maximum and average temperatures, stream temperatures did not increase to levels deleterious to cold water organisms. Post-treatment summer average daily stream temperatures, 15.32–15.87°C at treatment sites and 13.68–14.85°C at reference sites (Table 4), all still below accepted upper temperature thresholds (17.9°C) for cold water organisms like trout and amphibians, (McCormick, Hokanson, & Jones, 1972). Southern Appalachian streams support habitat for one native (brook) and two introduced (rainbow and brown) trout species that are the foci of a regionally important recreational fishery. Field studies indicate brook trout presence/absence is a function of duration of time that mean or maximum daily temperatures exceed 20°C (Wehrly, Wang, & Mitro, 2007). In controlled environments, brook trout growth rates are maximized between 12.4 and 15.4°C, and mortality increases above 17.9°C (McCormick et al., 1972). The 15.32–15.87°C average summer stream temperature at the treatment sites following rhododendron removal is within or near the optimal temperature range for maximizing growth rates of brook trout. However, the anticipated warming of air temperatures associated with climate change in the southern Appalachians (Flebbe, Roghair, & Bruggink, 2011) may increase the sensitivity of streams to canopy changes in the future. Beyond specific temperature thresholds for cold water organisms, there are concerns over how increased variability in temperature regime will affect salmonid performance (Malcolm et al., 2008). Fires, whether wild or prescribed, can lead to alteration to physical stream habitat (channel reorganization, additions of debris to the system) which can further disturb the distribution of fish and amphibians (Dunham, Rosenberger, Luce, & Rieman, 2007), and should be another aspect further considered when considering rhododendron removal.

4.3 | Variability in paired-catchment analysis of stream temperature response to rhododendron removal

The responses of our two treatment streams illustrates how inherent variability of similar watersheds can confound responses to watershed manipulations (Groom, Dent, Madsen, & Fleuret, 2011; Johnson & Jones, 2000; Kibler et al., 2013; Terrell, Summer, Jackson, Miwa, & Jones, 2011). The variability of temperature responses by year, stream, and sensor location makes a generalized assessment of treatment effect difficult. Subtle, unexplored differences in inter-site characteristics due to hyporheic exchanges (Arrigoni et al., 2008; Hester, Doyle, & Poole, 2009), groundwater upwelling (Briggs, Lautz, Buckley, & Lane, 2014), differences in lithology and groundwater storage (Bladon et al., 2018), stream aspect (Li et al., 2012), and overstory canopy gaps can play a role in the short-term variations in stream temperature by changing the capacity of the reaches to regulate their energy budget (Malcolm, Soulsby, Donaghy, Hannah, & Youngson, 2010). Localized differences in riparian forest cover, both in type and coverage percentage, have been shown to be a major control on stream thermal dynamics (Warren et al., 2013, 2017; Dugdale, Malcolm, Kantola, & Hannah, 2018). These characteristics can strongly influence the amount of radiation reaching the stream, and the ability of the stream to retain the energy that reaches it. Furthermore, these characteristics can control the exchange between the streambed and groundwater sources, resulting in differences in contributions of cooler groundwater to the stream as you move downstream. There is an inherent difficulty in untangling the degree of influence that watershed, local hillslope, and groundwater exchanges have on stream temperature. In small, high-gradient headwater streams like the ones in this study, subsurface water exchange is driven by processes that occur across these scales (Mallard, McGlynn, & Covino, 2014; Payn, Gooseff, McGlynn, Bencala, & Wondzell, 2009; Wondzell, 2011). These exchanges occur at varying magnitudes at different locations in the stream network; lateral inflows from the hillslope may introduce significantly different temperature water at one point in the stream network, where at another it may be an insignificant inflow (Mallard et al., 2014; Ward et al., 2013).

The seasonality of rhododendron's influence on stream temperature is an important consideration. The contrast in evergreen/deciduous composition of understory/overstory, makes the removal of rhododendrons of greater concern during summer months where there are the serious ecological implications (native species sensitivity to cold water, stream productivity, etc.) While this study was limited to exploring the influence of rhododendron on summer stream temperatures, it is important to note that this likely the most important season for. There could be other temperature effects during deciduous leaf off, an important control noted by Clinton (1995) for the amount of light coming through the canopy. However, in the dormant season, sun angles are low and topographic shading is high in these streams. The influence of this topographic shadow on the temperature regime is going to vary throughout the year (Zhang et al., 2018), and differences in annual weather conditions will also add to the noted variability in temperature response.

5 | CONCLUSIONS

Vegetation changes and fire exclusion in the southern Appalachians have allowed rosebay rhododendron to expand beyond its historical habitat, securing a significant role in riparian canopy coverage. Re-introduction of regular fires to these systems has been proposed as a way both control rhododendron spread and to obtain more desirable canopy. Key to this management proposal is understanding how rhododendron removal will affect riparian and stream ecology, of which summer stream temperatures are a key component.

Our study shows that removal of the rhododendron understory along riparian corridors resulted in variable summer stream temperatures increases of 0.9–2.6°C in these headwater streams even though total canopy cover remained above 80%. 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. This suggests that forest managers could remove rhododendron along riparian corridors and not adversely affect cold-water organisms as long as the overstory canopy remains largely intact. Stream temperature responses varied considerably across streams, years, and sensor location. One of our treated streams (HW_T) showed downstream cooling below the treatment reach and the other treated stream (SWO_T) showed downstream warming. Prior to rhododendron removal, stream temperatures varied significantly among the four study streams and among sensors within streams. Reference stream temperatures also varied significantly across the three study years, likely as a response to increases in air temperature, and differences in rainfall.

Future work could include examining the stream responses if more than 300 m of the stream reach were treated and testing whether rhododendron removal affects stream temperature during other seasons of the year. Additional investigations of any canopy removal should include PAR measurements and hemispherical photography as a way to better understand the influence understories play in regulating light at forest floor level. Additional, localized hydrologic measurements (discharge, local precipitation, shallow and deep groundwater levels, etc.) would help to partition out hydrologic controls on stream temperature. Further years of pre-treatment monitoring (>1 year) would help to better support a before-after, control-impact study design.

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CONFLICT OF INTEREST

The authors declare no conflicts of interest.

DATA AVAILABILITY STATEMENT

Data used in analysis available at: https://doi.org/10.6073/pasta/5f7c8c4f0527e8f6cf757ad9f96a8f41.

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