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Summer stream temperature changes following forest harvest in the headwaters of the Trask River watershed, Oregon Coast Range

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Abstract

The Trask River Watershed Study in the northern Oregon Coast Range was designed to examine physical, chemical, and biological effects of contemporary forest management practices on aquatic ecosystems. We measured stream temperature for 11 summers in 15 small watersheds, eight of which were harvested in 2012. Three riparian buffer treatments, which varied by landowner, were implemented. Using half-hourly data, we characterized summer water temperature distributions with five percentiles: 5th, 25th, 50th, 75th, and 95th. Each percentile was analysed as a separate response variable using a linear mixed model. After harvest, streams without overstory buffer requirements showed shifts in all the percentiles of the temperature distribution; the largest increase (3.6°C) occurred at the 95th percentile. Sites with narrow riparian buffers showed little to no change. We also calculated changes in duration of thermal exposure above 15°C and 16°C for two species of native stream amphibians; these temperatures occurred 4.7% and 1.3% of the time postharvest in the sites clearcut with no buffer. Analysis of distributions of summer temperatures preharvest and postharvest enabled us to more fully characterize site-to-site variability and responses to forest management.

KEYWORDS

amphibians, data distributions, forest harvest, percentiles, stream temperature, thermal regimes

1 | INTRODUCTION

Stream temperature response to natural and anthropogenic disturbances can be complex and nonlinear (Davis, Reiter, & Groom, 2016; Groom, Dent, & Madsen, 2011; Groom, Dent, Madsen, & Fleuret, 2011; Steel, Beechie, Torgersen, & Fullerton, 2017). However, the complexity of thermal responses to disturbance can be oversimplified when responses are represented by a single metric, such as a daily, weekly, or monthly maximum or minimum temperatures. Whereas these single metrics may be useful to characterize thermal environments in certain instances and are simple to administer in a regulatory programme, a more complete assessment of temperature data may provide greater insight into the biological significance of changes in the thermal regime of a system (Arismendi, Johnson, Dunham, &

Haggerty, 2013; Benjamin, Heltzel, Dunham, Heck, & Banish, 2016; Gomi, Moore, & Dhakal, 2006; Steel et al., 2017). Assessing shifts in the entire summer temperature regime can also elucidate the behaviour of physical processes. For example, shifts in the entire distribution can indicate whether temperatures are experiencing a simple response to weather patterns or whether changes in symmetry indicative of increased variability or a skewed shift in extremes is occurring (Arismendi, Johnson, & Dunham, 2015; Lavell et al., 2012; Lewis & King, 2017). In addition, a limited suite of temperature metrics does not represent the full thermal regime experienced by stream biota (Steel et al., 2017) because they are exposed to a range of daily temperatures and across seasons. Further, small but chronic changes in temperatures may influence species that are sensitive to temperatures, including ectotherms, whose physiological processes are connected to environmental conditions. Changes in temperatures can alter energy budgets of ectotherms, which could potentially influence vital rates including survival and reproductive output (Benjamin et al., 2016; Homyack, Haas, & Hopkins, 2011). By examining various facets of an entire temperature distribution (magnitude, frequency, and duration), a broader and ecologically relevant understanding of acute and chronic changes in the thermal regime is possible (Arismendi et al., 2013).

Temperature changes in streams, rivers, and lakes have received considerable study over the last two centuries (Webb, Hannah, Moore, Brown, & Nobilis, 2008), especially changes from land use practices. In particular, the potential for forest harvesting to increase stream temperatures has been well studied due to concerns about effects on aquatic species and ecological processes (Burton and Likens, 1975; Greene, 1950; Levno and Rothacher, 1969; Meehan, Swanson, & Sedell, 1977). In response to the early concerns over forest harvest, conservation measures were mainly focused on minimizing stream temperature increases on fish-bearing streams (Groom, Johnson, Seeds, & Ice, 2017; Hairston-Strang, Adams, & Ice, 2008). Although temperature amelioration measures have been enacted over the years, concerns over warmer stream temperature due to increased solar radiation after forest harvest still continue (Groom, et al., 2011a; Groom et al., 2011b; Wilkerson, Hagan, Siegel, & Whitman, 2006), especially given potential interactions with a warming climate. Further, the concern over temperature changes in aquatic ecosystems has expanded from an initial focus on fish-bearing streams to small, fishless headwater streams.

Headwater streams, especially those near the topographic divide, have a direct connection to terrestrial environments. Due to their narrow width and small volume, they are differentially influenced by the near-stream environment compared with wider, deeper downstream systems. As a result, whereas controls on stream thermal processes may be similar between headwater and downstream systems, the relative magnitude and rate of energy fluxes may vary. For example, headwater streams can be completely exposed to direct solar radiation yet the magnitude and direction of subsurface–surface exchange of water can mediate stream temperature response due to the high proportion of hyporheic or groundwater relative to surface water (Johnson, 2004; Moore, Spittlehouse, & Story, 2005; Story, Moore, & MacDonald, 2003). This variability is evidenced by the contrasting temperature responses to forest harvest, where increases, no change, and decreases in stream temperatures have been observed across experimental and observational studies. Stream temperatures increased, where reductions in overstory canopy cover increased solar radiation reaching the water surface (e.g., Bladon, Cook, Light, & Segura, 2016; Gomi et al., 2006; Jackson, Sturm, & Ward, 2001; Kibler, Skaugset, Ganio, & Huso, 2013). In contrast, temperature changes were minimized by the shade from residual riparian vegetation such as shrubs (Gravelle & Link, 2007) or from logging residue (slash) covering the stream channel (Jackson et al., 2001; Kibler et al., 2013). Further, immediately after timber harvest, evapotranspiration generally is reduced, and summer streamflow can increase, which also can minimize stream temperature changes following harvest (Gravelle & Link, 2007; Surfleet & Skaugset, 2013). Local variation in channel characteristics and geomorphology can mitigate stream temperature response to harvest or other disturbances, where streams flow subsurface and through hyporheic areas (Janisch, Wondzell, & Ehinger, 2012; Johnson, 2004) or amplify temperatures, where streams flow over bedrock (Johnson, 2004).

Prior studies that addressed management-related temperature responses in small, fishless streams of the Pacific Northwest (e.g., Bladon, Segura, Cook, Bywater-Reyes, & Reiter, 2018; Janisch et al., 2012; Johnson & Jones, 2000; Kibler et al., 2013) did not assess changes in the context of thermal requirements for resident aquatic species. Headwater streams provide important habitats for some temperature-sensitive species, including some amphibians (Homyack, 2010; Huff, Hubler, & Borisenko, 2005). The Trask River Watershed Study (TRWS), which occurred 2006–2016, examined the physical, chemical, and biological effects of contemporary forest management on aquatic ecosystems at varying spatial scales, from small, fishless headwaters to larger, fish-bearing streams (https://traskstudy.forestry.oregonstate.edu/). To accomplish the goals of the study, headwater stream protection practices of private, state, and federal forest landowners in Oregon were applied experimentally to small watersheds in the Coast Range. Here, we examine the effect of these practices on summer stream temperature distributions in small, headwater streams. Our basic study approach compares the thermal variability with temperature distributions among reference (REF) and harvested watersheds during preharvest and postharvest periods.

Two species of amphibians inhabit these small streams: coastal tailed frog (*Ascaphus truei*) larvae are dominant grazers and the aquatic stage of coastal giant salamanders (*Dicamptodon tenebrosus*) are dominant predators in headwater streams of the TRWS (Chelgren & Adams, 2017). To place changes in stream temperature distributions as a result of harvest in an ecological context, we also assessed changes in duration of thermal exposure against field-derived temperature preferences for coastal tailed frogs and coastal giant salamanders (Huff et al., 2005). We focused on summer because it is when stream temperatures have the potential of reaching the upper thermal tolerances for aquatic organisms. We predicted that stream temperature distributions would warm in treatments with forest harvest,

especially where little overstory was retained along streams, but that summer temperatures would still be lower than critical temperatures for coastal tailed frogs and coastal giant salamanders in the northern Oregon Coast Range.

2 | STUDY AREA

The TRWS area comprises the upper 25.0 km^2 of the East Fork of the South Fork of the Trask River in the northern Oregon Coast Range (Figure 1). The East Fork of the South Fork of the Trask joins the Trask River, one of the five major rivers draining into Tillamook Bay, Oregon. The TRWS is on the coastal side of the north Oregon Coast Range, and elevation ranges from 275 to 1,100 m. The proximity of the study area to the Pacific Coast and its location at the topographic divide strongly influences precipitation and seasonal temperatures. Moist air masses moving off the Pacific Ocean during the winter months produce high precipitation, whereas air temperatures stay mild throughout the year compared with areas further inland. During the study period (2006–2016), the mean annual precipitation was ~2,100 mm, with the majority falling in November and December. Most precipitation falls as rain; however, snowfall is frequent above 760 m during winter. Summers were typically warm and dry with a mean of 42 mm of precipitation during July and August. Mean maximum air temperature was 23.6°C for July–August, whereas the mean minimum air temperature was 8.7°C.

The geology of the TRWS consists of a mix of volcanic and sedimentary formations with numerous ancient earthflow deposits (Wells, Snavely, MacLeod, Kelly, & Parker, 1994). Streams in the TRWS are a geomorphic expression of underlying geology and surficial processes with lower gradient channels in the earthflow and alluvial terrain and higher gradient channels in the steep, highly dissected terrain underlain by the Siletz River volcanic formation and plutonic bedrock (Turner, Ward, James, & Reiter, 2007).

The overstory vegetation of the TRWS primarily consisted of ≈50-year-old Douglas fir (*Pseudotsuga menziesii)* with some areas of hardwood, predominantly red alder (*Alnus rubra*), along stream channels and in some localized hillside areas. The current vegetation and land ownership pattern resulted from a series of large forest fires between 1933 and 1951 known as the Tillamook Burn (Neiland, 1958). The eastern 11.0 km^2 of the study watershed was owned by Weyerhaeuser Company, a large, private industrial timber company. The state of Oregon owned the western 14.3 km^2 of the watershed; the state forest was managed by Oregon Department of Forestry (ODF). In addition, there was a small area (0.7 km^2) of federally owned land in the northeastern portion of the study watershed managed by Bureau of Land Management (BLM).

Upslope management practices and stream protection requirements along small, fishless headwater streams in the TRWS varied by ownership. Overstory tree retention along small, nonfish streams in the Oregon Coast Range is not required for private forest landowners. However, best management practices encourage landowners to retain understory vegetation, nonmerchantable trees and leave trees from in-unit requirements along small, nonfish streams (ODF, 2010a). For the TRWS area, the in-unit live tree wildlife retention requirements are approximately five tree per

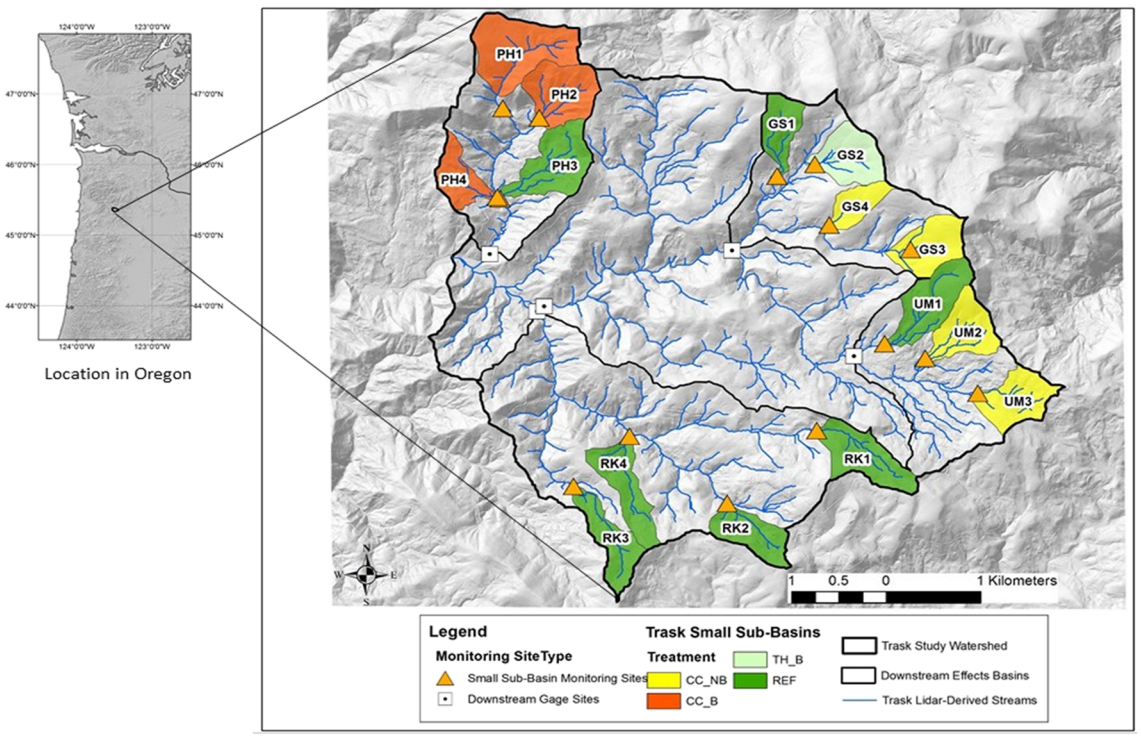

FIGURE 1 Trask River Watershed Study in western Oregon, USA

hectare of area harvested (ODF, 2010a). In this paper, we generally refer to the wildlife retention trees along small, nonfish streams on private land as leave tree areas, as opposed to true regulatory stream buffers. In addition, private forest landowners were required to minimize slash (logging residuals) accumulations in channels or within adjacent riparian areas. On these state forest lands, Oregon Forest Management Plan requirements for small, non-fish perennial and seasonal streams included a minimum 10 m no-harvest buffer on each side of the stream (ODF, 2010b). Harvest treatments on state forest lands in the TRWS included clearcuts, modified clearcuts and retention cuts, which left varying numbers of live trees within the harvest unit. BLM management practices included thinning to a remaining basal area of 30–55% of the original stand and a 15.2-m no-harvest riparian buffer left along each side of the small, fishless streams (U.S. Department of Interior–BLM, 2004).

The fish communities in the larger streams of the broader study area were dominated by resident coastal cutthroat trout (*Oncorhynchus clarkii clarkia*) and sculpin (*Cottus* spp.), with juvenile steelhead (*O. mykiss*) and juvenile coho salmon (*O. kisutch*) at the most downstream sites (Penaluna et al., 2015). The dominant aquatic vertebrate species in the fishless headwaters of the TRWS were coastal giant salamanders and coastal tailed frogs with occasional torrent salamanders (*Rhyacotriton spp.*; Chelgren & Adams, 2017).

3 | STUDY DESIGN AND METHODS

The TRWS was a manipulative field experiment conducted from 2006 to 2016 on small (<50 ha), nonfish-bearing headwater stream watersheds, in a replicated before–after control–impact (BACI) design with a range of riparian buffer treatments (Figure 1). The four treatments in the TRWS were clearcut with no required overstory buffer (CC_NB; n = 4), clearcut with a required buffer (CC_B; n = 3), thinning with a buffer (TH_B; n = 1), and unharvested, REF streams (n = 7; Figure 2a–d; Table 1). The four CC_NB treatments represent the variability of the treatment across Oregon private forestland ownership. Although the Oregon forest practice rules do not require overstory buffers for these types of streams, landowners are encouraged to place their required leave trees along the channels. As a result, two of the watersheds (GS3 and GS4) had no retention of overstory trees along the streams and two (UM2 and UM3) had minimal trees along the downstream portion of the stream. At both of these sites, harvesting up to the stream edge would have caused unacceptable resource damage, due to site specific conditions of saturated soils near the stream and unstable stream banks. The CC_B treatment on state forest lands, where there was a requirement of a 10-m no-harvest buffer for each side of the stream along the entire perennial portion of the channel, was implemented with additional wildlife trees adjacent to the buffers. The resulting buffers were slightly wider than required (Table 1). The TH_B treatment (GS2) had a 15.2-m no-entry riparian buffer along the small, nonfish-bearing stream. Seven unharvested small watersheds were designated as REF and were distributed across ownerships, proximal to the treated watersheds and in the headwaters of a larger REF catchment (Figure 1). Harvest units were designed to cover entire watersheds to the extent practicable and served as experimental units. The small watersheds drain to four larger, fish-bearing catchments. One of the four larger streams did not receive any treatments to allow for an unharvested downstream, fish-bearing REF site to compare with downstream of harvested watersheds fish-bearing sites. Additional information on field methods for site characteristics and for harvest details are available online: https://traskstudy.forestry.oregonstate.edu/.

We measured summer stream temperatures at half-hour intervals during summer low flows (early June–late September) 2006–2016, using continuously recording thermistors (HOBO Water Temp Pro V2, model U22-001, Onset Computer Corporation, Bourne, MA, with an accuracy of ±0.2°C). Thermistors were housed in eight-inch sections of polyvinyl chloride pipe to protect the instrument and prevent direct radiation on the sensor. We placed thermistors in areas of the stream where we observed complete hydraulic mixing at the most downstream portion of the designated small watershed. These specific deployment locations were used annually except where there was significant channel change during the previous winter, in which case, we located the thermistor as close as possible to the previous location. Accuracy checks were performed annually on the thermistors according to protocol in *Water Quality Monitoring Technical Guide Book* (Oregon Watershed Enhancement Board, 1999). We did not deploy any thermistor that varied more than ±0.3°C from the National Institute of Standards and Technology temperature.

4 | STATISTICAL ANALYSIS

We focused our evaluation on half-hourly temperatures measured between July 1–August 31, because streams are the warmest during this period, and our data record was most complete during this period. The analysis accounted for two time periods: 2006–2011 (preharvest) and 2013–2016 (postharvest). We omitted data from 2012 because forest harvest occurred from spring through fall 2012. We included analysis of REF watersheds to document stream temperatures in unharvested watersheds during the full study to account for climatic variability among years and between treatment periods.

We estimated impacts of forest harvest treatments on stream summer temperature distributions, relative to unharvested REF stands. Using the half-hourly data, we characterized summer water temperature distributions with five percentiles: 5th, 25th, 50th, 75th, and 95th to capture wide portions of the temperature distribution. Each percentile was analysed as a separate response variable, using a linear mixed model to incorporate both the design and treatment structure (McDonald, Erickson, & McDonald, 2000) of the TRWS. Specifically, we assumed a model for each percentile of the following form:

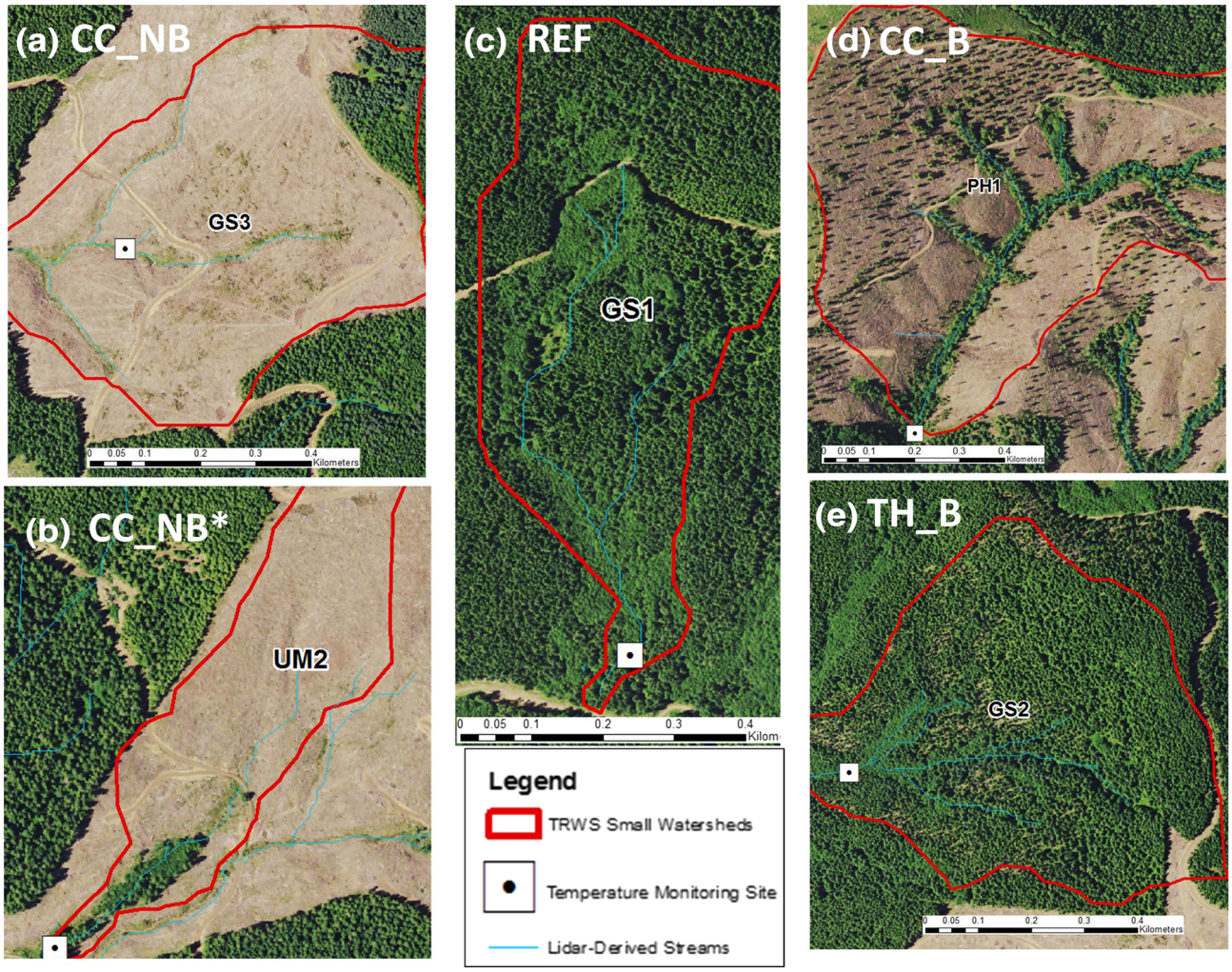

FIGURE 2 Aerial photographs showing treatment types in the TRWS. White squares with black dots in the centre indicate the location of temperature monitoring site in the watershed, whereas redlines indicate watershed boundary, and blue line indicates Lidar-derived stream channel. (a) Oregon Department of Forestry State Forest Lands, PH1 a modified and retention CC_B; (b) Bureau of Land Management, GS2 at TH_B; (c) Weyerhaeuser, GS1 a REF site; (d) Weyerhaeuser, GS3, a clearcut with no overstory retention requirement (CC_NB); and (e) Weyerhaeuser, UM2, a clearcut with wildlife leave trees along the downstream portion of the study stream (CC_NB*). *Note that while there are some trees left along the stream, there are no overstory retention requirements and so these treatments are still considered CC_NB. CC_B, clearcut with buffer; CC_NB, clearcut with no buffer; REF, reference; TH_B, thinning with buffer; TRWS, Trask River Watershed Study

$$y_{ijk} = \mu + b_{0i} + \tau_j + \upsilon_k + (\tau\upsilon)_{jk} + \varepsilon_{ijk},$$

where y_{ijk} represents the relevant temperature percentile for harvest unit i, with treatment j during year k; μ is the overall mean; τ_j is the effect of treatment j; υ is the effect of year k; $(\tau\upsilon)_{jk}$ is the treatment by year interaction; and ε_{ijk} represents the experimental error. Harvest unit random intercepts, b_{0i}, were included to account for watershed-level variation in expected quantile temperature response. The random effects were assumed to be normally distributed with mean zero and independent of the experimental error.

We note that the experimental error in the above model is intended to include sampling variability, measurement error, the watershed by time interaction, and the watershed by treatment interaction. The sums of these "error" terms are assumed to follow a normal distribution with variance σ_e^2.

Estimators for the harvest treatment effects, relative to REF conditions, were constructed from the fitted model:

$$\left(\mu_{Trt,After} - \mu_{Trt,Before}\right) - \left(\mu_{Ref,After} - \mu_{Ref,Before}\right),$$

where $\mu_{Trt,x}$ represents the mean treatment response for the temperature percentile under study, averaged over all years within the period denoted by the subscript x. The analogous REF quantity is denoted by $\mu_{Ref,x}$. The terms in the first set of parentheses estimate the expected pre–post change due to harvest and other temporal changes, whereas the terms in the second parentheses adjust for temporal changes in the absence of harvest.

TABLE 1 Small watershed characteristics including treatment type, monitoring site elevation, drainage area, low flow channel width, and shade (%, calculated as 100—total site factor, using hemispherical analyses) over the stream before and after harvest for Trask River Watershed Study headwaters

Watershed ID	Forest harvest treatment	Thermistor elevation (m)	Drainage area above thermistor (ha)	Average wetted width of stream (m)	Mean overstory 2-sided buffer width (slope distance; m)	Shade (%) preharvest	Shade (%) postharvest
GS1	REF	616.1	26.7	1.3	REF	82.9	83.1
GS2	TH_B	661.4	39.3	1.8	*	81.2	84.0
GS3	CC_NB	789.6	20.6	1.4	0.0	83.6	7.0
GS4	CC_NB	729.2	21.0	1.1	0.0	85.5	10.9
PH1	CC_B	533.2	67.2	2.1	33.2	85.9	82.7
PH2	CC_B	479.5	20.6	1.1	22.6	91.3	89.1
PH3	REF	365.3	48.6	1.0	REF	82.5	88.1
PH4	CC_B	367.5	26.3	0.9	23.9	84.7	82.9
RK1	REF	681.0	44.9	1.6	REF	87.7	86.7
RK2	REF	593.3	32.4	1.4	REF	89.6	87.7
RK3	REF	556.3	35.2	1.5	REF	89.1	93.3
RK4	REF	435.9	38.9	1.7	REF	a	b
UM1	REF	660.0	44.5	1.5	REF	85.1	83.5
UM2	CC_NB	689.8	17.1	1.3	16.0	84.3	65.7
UM3	CC_NB	735.2	39.3	1.5	14.1	80.6	76.6

Abbreviations: CC_B, clearcut with buffer; CC_NB, clearcut with no buffer; REF, reference; TH_B, thinning with buffer.
[a]Unable to determine exact buffer width in thinning because no hard edge to buffer break.
[b]Because RK4 was not monitored for amphibians, detailed habitat data were not collected.

The inferential goal of this analysis was estimating the magnitude and direction of changes in temperature distribution due to harvest, after adjusting for annual variation to understand biologically meaningful changes. This is consistent with an expectation that the impact of harvest on water temperature will not be exactly zero—be it small, large, positive, negative, or variable. Consequently, our analysis and reporting of results avoids the use of testing and *p* values, as these constructs were not consistent with our objectives (Mark, Lee, & Harrell, 2016). We instead report results using estimates and standard errors (*SE*s) for each metric of interest and interpret results with respect to their estimated magnitude and sign and in context of the results for all response variables. Note that we use SE to characterize uncertainty in our estimates and standard deviation (*SD*) to characterize the estimated variation in a population.

For the 15 small watersheds, there were six summers of pretreatment (2006–2011) and four summers of posttreatment (2013–2016) data collected at half-hour intervals resulting in a total of 150 watershed-years possible for the data record. The temperature percentile approach described above requires that there be no missing data and that treatment conditions within a harvest unit are consistent throughout a summer period. Due to the strong dependence of water temperature on within-season and within-day trends, missing data could bias the percentile statistics used in the analysis. Consequently, we dropped 9 watershed-years, where sensors were exposed to air, missing, or contained other data issues, resulting in 141 watershed-years available for analysis. All statistical analyses were performed using the "lme" function from package "nlme" (Pinheiro, Bates, DebRoy, Sarkar, & R Core Team, 2016) in the R programming environment (R Core Team, 2016). We performed residual diagnostic checks to assess discrepancy between the model and the observed data.

To place stream temperature and potential changes as a result of harvesting in an ecological context, we also assessed changes in duration of summer thermal exposure against the upper realized thermal niche (RTN) for coastal tailed frogs (*A. truei*) and coastal giant salamanders (*D. tenebrosus*) based on the work of Huff et al. (2005) in Oregon. A RTN reflects both the temperature of an organism's environment and other factors that influence abundance such as competitive interactions with other species (Huff et al., 2005; Magnuson, Crowder, & Medvick, 1979). To estimate thermal niches for 14 vertebrate species in Oregon, Huff et al. (2005) utilized field-derived relative abundance for those species in combination with actual warm-season temperature data. Similarly, we focused on summer monitoring because it is when stream temperatures have the potential of reaching the upper thermal tolerances for aquatic organisms. Whereas Huff et al. (2005) used a mean 7-day maximum temperature metric, we used half-hourly values to estimate durations above the upper RTN. The values we used for thermal regime preferences for the two

main endemic amphibians in the small watersheds of the TRWS were 16.0°C for coastal tailed frog larvae and 15.0°C for coastal giant salamander larvae (aquatic), which is 0.1°C less than the RTNs for North Coastal sites in Huff et al. (2005). We also evaluated changes in the frequency of stream temperature >18.0°C because tailed frog eggs can experience mortality at this temperature (Bury, 2008). We predicted that stream temperature distributions would warm in treatments with forest harvest, especially where little overstory was retained along streams, but that summer temperatures would be lower than the upper RTN for coastal tailed frogs and coastal giant salamanders in the northern Oregon Coast Range.

5 | RESULTS

For each watershed, the entire July 1–August 31 half-hour temperature record was used to create a distribution, which was inspected in order to determine whether there were shifts in the distribution in response to harvest. Although this is not a quantitative approach, it is informative to the discussion. Examples of preharvest and postharvest temperature distributions for each treatment type are shown in Figure 3. The example of the temperature distribution for CC_NB with no leave trees exhibits a clear shift in the distribution towards warmer temperatures with an increase in total range (Figure 3a). The CC_B (Figure 3d) and REF (Figure 3c) show a much smaller shift to a warmer

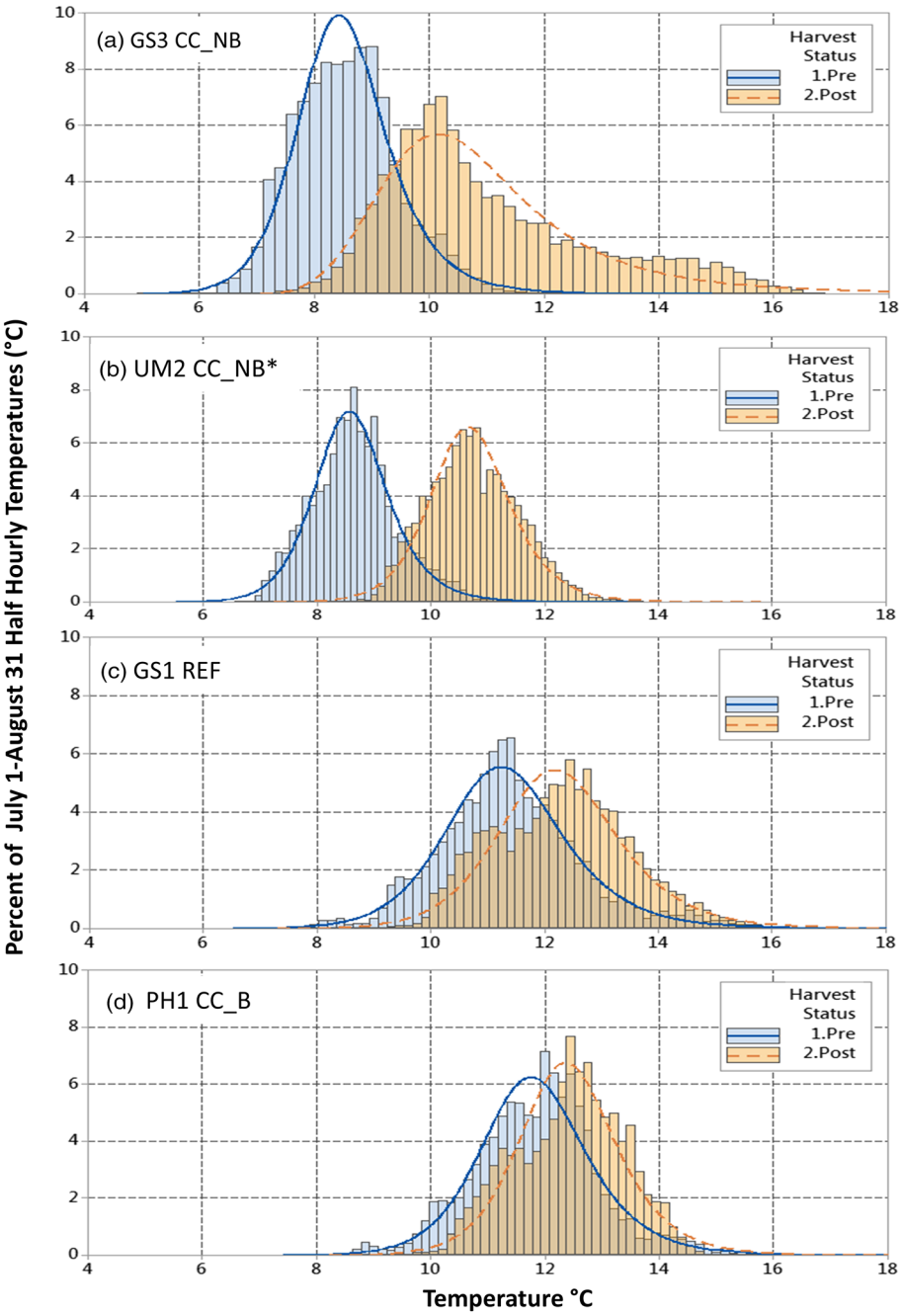

FIGURE 3 Examples of distributions of half-hourly July 1–August 31 stream temperature data during preharvest (2006-2011) and postharvest (2013-2016) periods. (a) GS3, a CC_NB site with no overstory trees left and (b) UM2, a CC_NB site but with leave trees along the downstream portion of the study watershed; (c) GS1, a REF site and (d) PH1, a CC_B site. Not shown is TH_B. CC_B, clearcut with buffer; CC_NB, clearcut with no buffer; REF, reference; TH_B, thinning with buffer

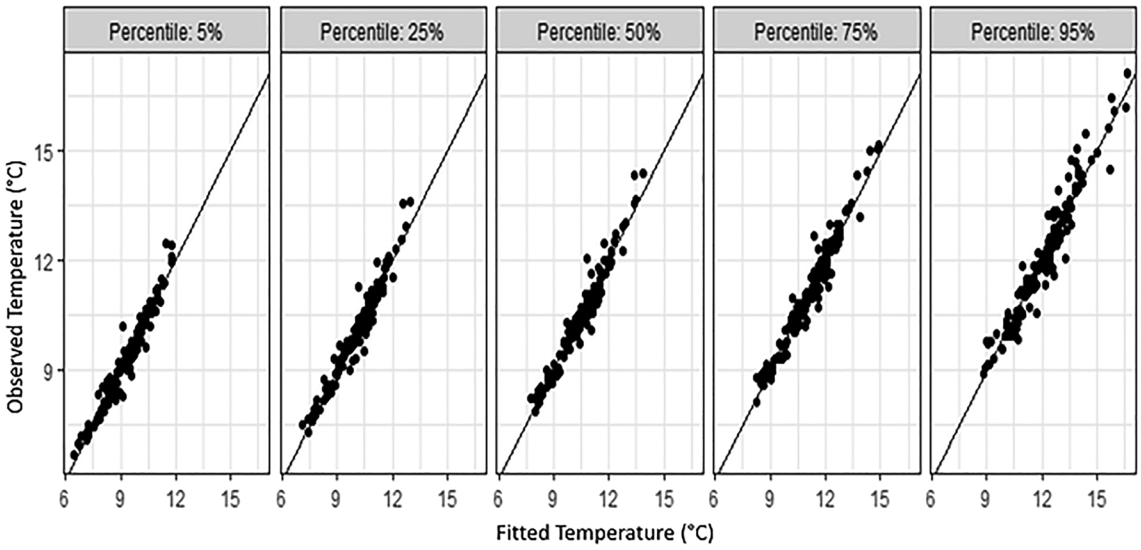

FIGURE 4 Observed versus fitted temperatures for the before–after control–impact model by percentiles across all years and watersheds

temperature distribution and little alteration in overall temperature range. This is indicative of changes in year-to-year climate patterns.

5.1 | Specific response of temperature percentiles to treatments

Data from our replicated reference small watersheds (n = 7) across multiple years allowed us to estimate harvest effects separately from interannual variation preharvest and postharvest. Fitted values from the BACI model showed good overall agreement with observed values across all percentile responses (Figure 4).

Our analysis indicated that stream temperature responses to harvest differed across treatment types and by temperature percentile (Figure 5). Watersheds without an overstory buffer or minimal overstory retention (CC_NB) exhibited the largest temperature increases for all percentiles (Figure 5). The magnitude of the estimated temperature changes for CC_NB watersheds from harvest

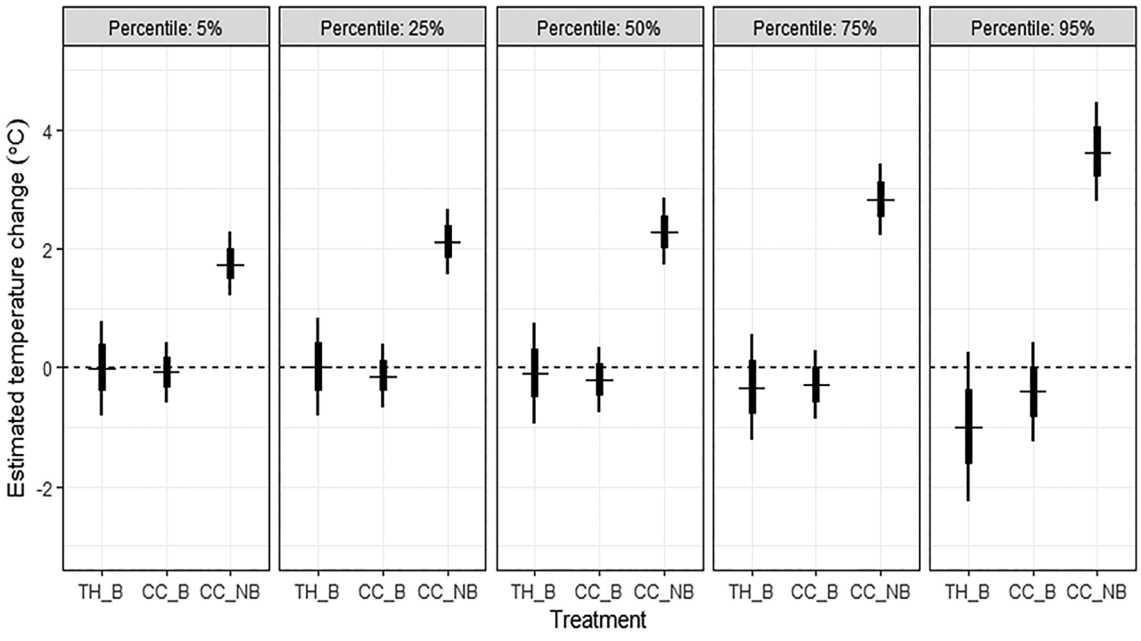

FIGURE 5 Estimated average preharvest to postharvest changes in summer stream temperature percentiles for each harvest treatment type in the Trask River Watershed Study, after adjusting for temporal changes in reference treatment watersheds. Each panel shows results for a different temperature percentile. Positive values indicate a postharvest temperature increase, whereas negative values indicate a decrease. Small horizontal dashes show the mean estimate. Thick vertical bars show ±1 *SE*, and thin vertical bars show ±2 *SE*

increased with increasing percentile (i.e., at higher temperatures the harvest impact increased). For example, the estimated mean CC_NB treatment effect at the 5th percentile was 1.7°C (*SE* = 0.3) and was 3.6°C (*SE* = 0.4) for the 95th percentile. Thus, the warmest segment of temperature distributions (95th percentile) had a greater increase relative to REF sites than lower percentiles. The responses to harvest at the 5th percentile for all four CC_NB watersheds were smaller and more similar (range from 1.5°C to 2.8°C) than at 95th percentile (Figure 5). The two GS watersheds that showed changes of 4.2°C and 4.4°C had no leave trees near the streams, whereas the two UM watersheds both had leave trees in the riparian area after harvest.

Small watersheds with buffers (CC_B) showed little to no apparent postharvest temperature increases across all percentiles (Figure 5). Mean estimates of temperature change with the CC_B treatment were either close to zero or negative for all temperature percentiles. Uncertainty in the estimates for the CC_B treatment effect suggests that we cannot preclude small negative or small positive mean changes in temperature (Figure 5). However, our results indicate that the expected temperature change due to harvest, averaged across study sites and years, for any of the five percentiles was likely less than 0.5°C for CC_B watersheds, after accounting for before–after changes in the reference watersheds. Results for the TH_B treatment, which was only applied to one watershed, were similar to the CC_B treatment.

5.2 | The TRWS thermal landscape

We assessed stream temperature response to annual climactic variability using the seven REF watersheds. For this assessment, we chose to use the 50th percentile (i.e., the median) temperatures for year-to-year comparisons (Figure 6). During the preharvest period (2006 through 2011), REF watersheds had a mean 50th percentile temperature of 9.9°C, with preharvest site 50th percentiles ranging from 8.5°C to 11.3°C (see Table SA1 for individual site information). The postharvest period (2013–2016) had warmer summer weather, and subsequently, all metrics of stream temperature were higher postharvest in REF watersheds. The mean 50th percentile REF water temperature increased to 10.8°C postharvest (Figure 6); the 5th and 95th percentiles increased by 0.9°C and 0.6°C to 9.6 and 12.0, respectively.

To determine whether the treatment watersheds had inherent temperature differences prior to harvest, we calculated selected percentiles for those sites. For the CC_NB, CC_B, and TH_B small watersheds, the preharvest period had a mean 50th percentile summer temperature of 10.2°C similar to the preharvest REF-only sites. The range in 50th percentile summer temperatures preharvest was 8.5 to 11.8°C, a little over 3°C difference across the treatment sites. The mean 5th percentile (i.e., low temperature) summer temperature for the treatment sites prior to harvest was 8.9°C (*SD* = 1.0), comparable, though slightly higher than the REF sites. The preharvest treatment sites had a 5th percentile range from 7.4°C to 10.3°C, also representing almost a 3°C difference. The warmest portion of the distribution that we examined, that is, the 95th percentile, had a mean temperature of 11.9°C for the preharvest treatment sites, likewise higher than for the REF sites. This percentile had the largest range from 10.0°C to 13.5°C, a 3.5°C difference. We further examined thermal variability grouped by treatment type to evaluate inherent variability in treatment groupings even in the absence of harvest. During the preharvest period, stream temperature distributions in REF watersheds and treatment watersheds overlapped all percentiles used in analyses. However, TH_B and CC_B appeared to have warmer preharvest 95th percentile

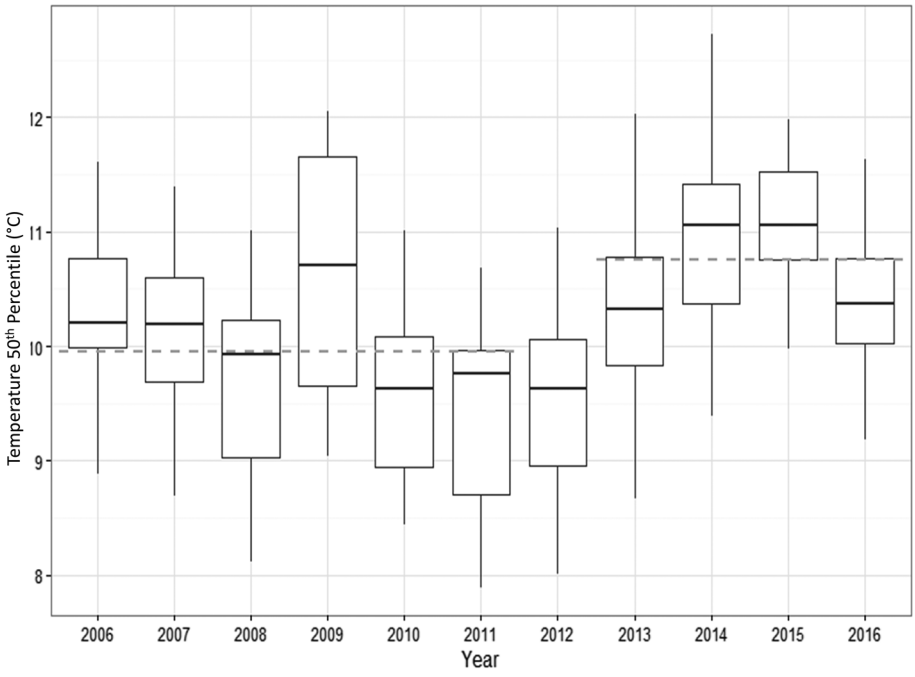

FIGURE 6 Fiftieth percentile (median) temperatures for the reference stream temperatures from July 1–August 31 across summers in the Trask River Watersheds. The horizontal dashed line on the left denotes the mean median value across all reference watersheds for the preharvest period (2006-2011), whereas the dashedline on the right is the mean median value during the postharvest period (2013-2016). The top and bottom of each box represents the 75th and 25th percentiles, respectively, whereas the middle line represents the 50th percentile (median). Whiskers extend from the boxes to the maximum and minimum values

temperatures than the REF or CC_NB watersheds. The mean 50th percentile preharvest temperature for REF watersheds was 10.0°C, whereas the TH_B, CC_B, and CC_NB watersheds had mean 50th percentile temperatures of 10.5°C (*SD* = 0.7), 11.0°C (*SD* = 0.9), and 9.6°C (*SD* = 1.2), respectively. We note that the statistical model described above accounts for preharvest temperature differences when estimating postharvest temperature changes on harvested sites.

Lastly, we combined all REF and treated sites to get the overall landscape thermal variability before and after harvest (Table 2a 2b, respectively). Prior to harvest, the mean 50th percentile temperature was 10.1°C; the estimated among-watershed *SD* across all sites was 1.0°C; and estimated preharvest across-year *SD* was 0.4°C (Table 2a). The mean difference between the 5th and 95th percentiles, an indicator of thermal stability, was 2.9°C preharvest across all watersheds and ranged from 1.6°C (*SD* = 0.64) to a maximum of 4.0°C (*SD* = 0.95). Variation in preharvest summer stream temperatures were more variable across sites, even for watersheds that are in close proximity, than across years, for all percentiles.

During the postharvest period across all the TRWS small watersheds, the mean 50th percentile summer temperature was 11.3°C, with an estimated among-watershed *SD* of 1.2°C and an estimated across-year *SD* of 0.4°C (Table 2b). The range in postharvest mean 50th percentile temperatures was from 8.7°C to 14.4°C, which was 0.7°C greater than the preharvest range. The mean postharvest 5th percentile temperature (i.e., low temperature) was 10.0°C (*SD* = 1.0). The 5th percentile had a range from 7.9°C to 12.5°C; representing almost a 5°C difference, similar to preharvest. The warmest portion of the distribution examined, that is, the 95th percentile, had a mean temperature of 12.9°C (*SD* = 1.8) postharvest. This percentile exhibited the largest range across the landscape with an 8.5°C difference among all the watersheds. The mean difference between the 5th and 95th percentiles, an indicator of thermal stability, was 2.9°C (*SD* = 1.0) preharvest across all watersheds, whereas postharvest it was 2.8°C (*SD* = 1.3). Thus, although there were temperature changes for individual small watersheds across the landscape, the mean thermal stability across all study headwater watersheds did not shift after harvest.

Finally, it was informative to examine the thermal behaviour of the CC_NB small watersheds with and without small areas of leave trees because there is variability in this treatment across the landscape depending on how the buffer requirements are operationally applied on private forestlands. In the harvested watersheds with no overstory buffers, we observed the greatest asymmetrical shift towards the warmer summer temperature distributions and the highest magnitude increases for the upper percentiles with an average increase of 4.3°C for the 95th percentile, whereas CC_NB streams that had even limited areas of leave trees had approximately half the increase in the 95th percentile with an average increase of 2.3°C.

TABLE 2A . Preharvest temperature percentiles for all study watersheds combined

Percentile	Mean	$\hat{\sigma}_{site}$	$\hat{\sigma}_{year}$
5th	8.8	0.8	0.5
25th	9.6	1.0	0.5
50th	10.1	1.0	0.4
75th	10.7	1.1	0.4
95th	11.7	1.3	0.6

Note. Temperature distribution percentiles are calculated for each watershed-year. The statistics show the mean percentile value across all watershed-years, as well as the estimated among-watershed and among-year standard deviations.

TABLE 2B . Postharvest temperature percentiles for across all watersheds combined

Percentile	Mean	$\hat{\sigma}_{site}$	$\hat{\sigma}_{year}$
5th	10.0	1.0	0.4
25th	10.7	1.1	0.5
50th	11.3	1.2	0.4
75th	12.0	1.4	0.4
95th	12.9	1.8	0.4

Note. Temperature distribution percentiles are calculated for each watershed-year. The statistics show the mean percentile value across all watershed-years, as well as the estimated among-watershed and among-year standard deviations.

5.3 | Changes in frequency of thermal extremes

We calculated the duration of exposure to temperatures at or above the RTNs for coastal tailed frogs (RTN = 16.0°C) and coastal giant salamanders (RTN = 15.0°C), the two most common vertebrates in the fishless streams of the TRWS. Using the lower RTN of 15.0°C across all 11 years of analysis for all 15 sites, stream temperatures exceeded 15.0°C for 1.3% of total hours examined (2,704 of 212,772 hr). The low exceedance rate precluded a strictly statistical analysis, and so instead, we summarized the temperature exceedance data in an ecological context. The exceedance duration to upper RTNs for coastal tailed frogs and coastal giant salamanders varied by watershed and treatment (Figure 7). Prior to harvest, all the watersheds combined had less than 1.0% of July 1–August 31 hourly values greater than 15.0°C. Following harvest, the overall mean for all watersheds, including reference sites, was 1.4% of total hours of exposure greater than 15.0°C. These watersheds experienced an increase in the duration of temperatures above 15.0°C from 0.2% preharvest to 4.7% postharvest (Figure 7). The average duration of time when temperatures exceeded 15.0°C increased for the CC_NB treatment from approximately 2 hr before harvest to approximately 70 hr after harvest. One of the CC_NB watersheds, UM3, had a beaver (*Castor candadensis*) pond in its headwaters and showed the largest initial increase in duration of temperatures greater than 15.0°C. However, 4 years after harvest, the duration of temperatures greater than 15.0°C in UM3 were similar to preharvest (Figure 7). Only one REF watershed (GS1) had temperatures greater than 15.0°C in the absence of harvest. The

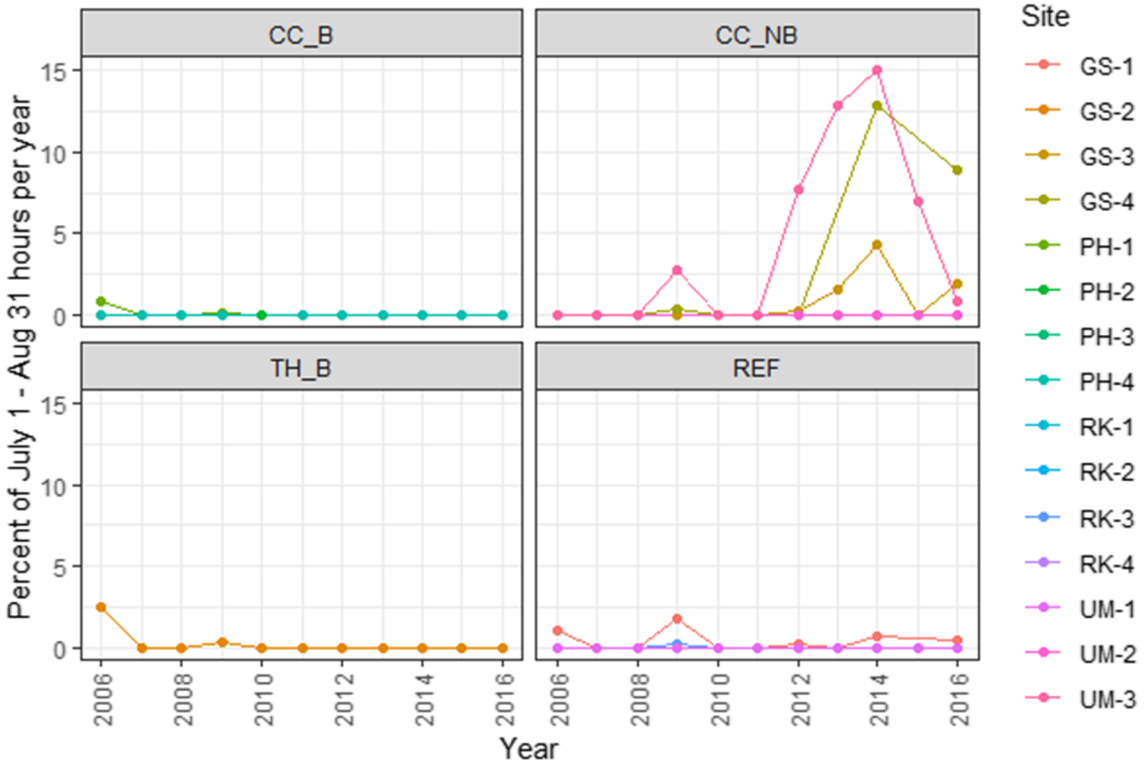

FIGURE 7 Percent of the hours each yearthat temperatures were at or above 15.0°C between July 1 and August 31. Each point represents the exceedance of an individual watershed for a given year. Panels are treatment groupings. Lines are used to connect sites and provide for ease of viewing. CC_B, clearcut with buffer; CC_NB, clearcut with no buffer; REF, reference; TH_B, thinning with buffer

CC_B and TH_B treatments did not experience any change in the percent of time above 15.0°C or 16.0°C. For the RTN of 16.0°C for coastal tailed frogs, only the CC_NB treatment had postharvest temperatures exceeding the RTN. Before harvest, this treatment had no instances of temperatures above 16.0°C, whereas after harvest, 1.3% of the time, this temperature was exceeded. In contrast, the CC_B and TH_B treatments showed no change in duration for either upper RTN. We also note that stream temperatures never exceeded 18.0°C, a threshold for mortality of coastal tailed frog eggs (Bury, 2008), and all temperatures were well below the critical thermal maximum of 30.1°C for coastal tailed frogs (de Vlaming & Bury, 1970) and 28.7–29.3°C for larvae of coastal giant salamanders (Bury, 2008).

6 | DISCUSSION AND CONCLUSIONS

Analysing the entire temperature distribution during the warmest portion of the summer in the TRWS allowed us to gain greater insight into changes to the thermal regimes of small headwater watersheds following harvest rather than simply assessing statistical changes in single metrics. The BACI analysis indicated that there were statistically significant temperature increases for streams with no to limited overstory buffers and that these changes increased at higher percentiles, with the greatest increases at the 95th percentile. However, our objective was to not only examine the data for statistically changes but also potential biologically significant changes. For this, we examined the duration of time that headwater TRWS amphibian species were potentially exposed to temperatures above their upper RTN. This assessment showed that although there clearly were temperature increases for the CC_NB treatment, the amount of time above the RTN was minimal. Finally, the TRWS design, which utilized multiple reference sites, allowed us to better account for steam temperature changes resulting from year-to-year climatic variability, which is critical for disentangling climate signals from management effects (Reiter, Bilby, Beech, & Heffner, 2015).

The thermal landscape of the TRWS was influenced by its geographic and geologic characteristics including elevation, latitude, continentality, and channel morphology. Even within the limited geographic extent of the TRWS, the high variability in preharvest temperatures among watersheds was notable. Preharvest, the TRWS streams had similar overstory characteristics and were in close proximity but had very different temperature distributions, similar to what Leach, Olson, Anderson, and Eskelson (2017) reported for the western Cascades. For example, the 3.0°C difference in the 95th percentile temperatures for small watersheds immediately adjacent to one another (PH1 and PH2) was substantial. We infer that the high thermal stability (i.e., minimal differences between the 5th and 95th percentiles) observed before and after harvest in PH2 suggests greater hyporheic and subsurface flowpaths than PH1. Other aspects of thermal variability occurred in streams that maintained very cool summer

temperatures (e.g., UM1), due to water flowing subsurface for multiple metres, similar to streams studied by Janisch et al. (2012) in Washington State. A third aspect of thermal variability, encountered in UM3, was caused by surface water heating in a shallow upstream beaver pond; temperatures were elevated even prior to harvest. Thus, for small, nonfish streams at the upper extent of perennial flow near the topographic divide, geologic, geomorphic, and climatic influences can impact the magnitude of year-to-year and site-to-site variability and thermal responses to disturbance. This thermal variability of a landscape, both before and after management, contributes to habitat diversity for aquatic species.

The preharvest and postharvest thermal variability in the TRWS was not likely to be a primary factor driving distribution of amphibians in small headwater streams in our study system (Chelgren & Adams, 2017). The distribution of species, especially coastal tailed frogs, were more likely influenced by the sizes of stream substrate materials (Welsh & Hodgson, 2008). The preharvest temperatures at the warmest percentile, that is, the 95th, were below the RTN for both species. Temperatures varied substantially across the landscape, indicating that even in the absence of management, these species are exposed to a range of thermal conditions in adjacent watersheds. During the postharvest period, all small watersheds including the REF sites exhibited an increase in temperature values for all percentiles of the distribution due to a warmer postharvest climate. During the postharvest period, the CC_NB showed the greatest increase in the 95th percentile following harvest, but the resulting average 95th percentile temperatures for this treatment were within the thermal niche preferences for these amphibians.

Our examination of not only a change in the specific percentile of the distribution but the change in duration of time spent at temperatures critical for resident amphibians allowed us to better interpret potential changes in the suitability of amphibian habitat. Postharvest changes in magnitude and duration of elevated temperatures were likely not sufficient to cause changes in survival or movement of headwater amphibians in the TRWS (Chelgren & Adams, 2017). Although temperatures generally stayed below the 15.0°C and 16.0°C thermal preferences for these species, it is not known whether these small changes in temperature might have induced behavioural responses (e.g., Bernardo and Spotila, 2006; Bancroft, Baker, Searle, Garcia, & Blaustein, 2008), including increased movement (Chelgren & Adams, 2017) or changes in survival rates of coastal giant salamanders. For example, Chelgren and Adams (2017) found that timber harvest decreased survival for coastal giant salamanders whereas Leuthold, Adams, and Hayes (2012) found no immediate effect of timber harvest. Further, we do not know whether the site-to-site variability in preharvest stream temperatures, which can be important for thermal adaptation (Hossack, Corn, & Fagre, 2006), could have influenced responses to harvest-related temperature changes.

We placed these findings in context of stream-associated amphibians, which constitute significant biomass and provide many ecological functions in headwater systems. Coastal giant salamanders serve as top predators in streams without fish and transfer energy between aquatic and terrestrial environments (Davic & Welsh, 2004) and were present in all the study watersheds. Tailed frog tadpoles are avid grazers where they occur but were not present at all sites. For ectotherms such as these amphibians, physiological processes from digestion to assimilation and individual behaviours, including habitat selection, are affected by environmental temperatures (Blouin-Demers & Weatherhead, 2001; Pough, 1980). Temperature changes from local (forest harvesting) to global (climate change) scales may influence individual, population, and distributional level parameters of amphibians (Grant, Wiewel, & Rice, 2014; Homyack et al., 2011; Kluber, Olson, & Puettmann, 2009; Peterman & Semlitsch, 2013). Relatively small increases in stream temperatures, especially when they are outside the RTN for a species, can result in biological responses, the directionality of which will depend on availability of food resources and whether individuals are able to optimize physiological functioning (Bancroft et al., 2008). These changes can affect allocation of resources to growth, development and metamorphosis, and potential reproduction for ectotherms (Alvarez & Nicieza, 2002; Homyack et al., 2011). Ultimately, if temperatures remain above the RTN for a given species, fecundity could be reduced until it impacts the local population abundance (Huff et al., 2005). However, both abundance and distributions of amphibians also may be affected by microhabitat characteristics other than temperature, such as fine sediment and woody debris (Welsh & Hodgson, 2008) as well as behavioural factors such as movement (Chelgren & Adams, 2017) and interspecific interactions (Davic & Welsh, 2004).

In agreement with our predictions, summer temperature regimes in headwater streams exhibited variability related to annual weather conditions and local site conditions prior to management. The observed treatment effects were of similar magnitude to other studies of small, nonfish streams in the Pacific Northwest studies of small, nonfish streams (e.g., Bladon et al., 2016; Bladon et al., 2018; Janisch et al., 2012; Johnson & Jones, 2000; Kibler et al., 2013) where streams without riparian buffers experienced increases in maximum and mean temperatures after forest harvest. For the TRWS, the temperature responses to a range of silvicultural harvest prescriptions added variability in headwater stream temperatures at all percentiles of the distribution. Analysing the full distribution of data for the TRWS expanded our understanding of the temperature responses across the full thermal regime, including the magnitude and duration of exposure. This analytical approach could enable researchers to better characterize linkages between aquatic species and their physical environment.

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SUPPORTING INFORMATION

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

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