Contents lists available at ScienceDirect

Forest Ecology and Management

Stream temperature responses to forest harvesting with different riparian buffer prescriptions in northern California, USA

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ARTICLE INFO

ABSTRACT

Forested riparian zones adjacent to headwater streams provide unique habitat attributes and influence water quantity and quality. Forest harvesting adjacent to and within these zones has led to changes in stream temperature (T_s) and impacts on aquatic ecosystems. As such, policies and regulations have sought to limit forest management activities in riparian zones; however, water temperature responses have been variable, necessitating additional research to improve our understanding of the effectiveness of contemporary riparian harvesting prescriptions in limiting water temperature increases after harvest. We sought to quantify the effects of three different riparian buffer prescriptions with increasing intensity of basal area removal (2–43% area harvested) on T_s in forested headwaters in northern California, USA. We measured T_s in 18 headwater catchments (12 harvested, 6 six unharvested references) during a pre-harvest year and two post-harvest years using 12 thermistors distributed longitudinally down each stream (i.e., 216 total sensors). We analyzed the T_s data to assess (1) the seasonal variability of seven-day moving average of daily maximum stream temperature ($T_{7-day-max}$) and (2) the relative importance of the different harvesting prescriptions and catchment physiographic characteristics in influencing the $T_{7-day-max}$. Our analysis indicated substantial changes in basal area and shade in the riparian areas with the most intensive harvesting treatment (i.e., 50% reduction in canopy cover). However, these changes in riparian canopy were poorly related to stream temperature. Results indicated a median $T_{7-day-max}$ increase of ∼ 2 °C in the sites with the most intensive harvesting treatment. The greatest changes in seasonal $T_{7-day-max}$ occurred during the summer and fall but only during the first year after harvesting. There was no evidence of increases in $T_{7-day-max}$ during the second post-harvest year. While air temperatures in the riparian areas increased by 1–5 °C after harvesting, this warming did not directly transfer to strong warming in stream temperatures. Rather, random forest models revealed that $T_{7-day-max}$ was more strongly related to topography (i.e., elevation) and climatic variability (i.e., changes in precipitation and stream stage) than to the riparian harvesting prescription or the harvesting period. Our study further highlights the challenges in understanding the thermal regimes of headwater streams and their responses to forest disturbances. Predictions of stream temperature responses to forest disturbances are complicated by the heterogeneity of the factors that influence this important physical water quality parameter. However, with global climate change and increasing pressures on water resources and aquatic ecosystems it is increasingly important to continue to provide insights into the relationships between forest management activities and the thermal regimes of headwater streams.

1. Introduction

Forested riparian zones adjacent to headwater streams provide critical biotic and abiotic habitat elements and influence water quantity, water quality, and availability of water for downstream uses (Vannote et al., 1980; MacDonald and Coe, 2007; Bernhardt et al., 2018). For example, forests in the riparian zone provide a source of large wood, erosion control, nutrient filtration, and moderation of direct solar radiation to streams (Clinton, 2011; Kuglerová et al., 2014). The extent and spatial arrangement of riparian forests along streams often impose a first-order control over stream water quantity and quality (Hill, 1996; MacDonald et al., 2014). Moreover, headwater riparian areas can act as

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climate refugia and can contribute to regional species diversity by providing stable terrestrial and aquatic habitat (Isaak et al., 2016; Olson et al., 2017; Krosby et al., 2018). Given the projected effects of climate change on summer low flows (Vander Vorste et al., 2020), stream temperatures (Arismendi et al., 2013; Lisi et al., 2015; Wondzell et al., 2019), and habitat provision (Wohl, 2017), it is increasingly critical to improve our understanding of the function of riparian buffer zones and the response of streams to disturbances in forested headwaters.

Concerns about the response of streams to forest disturbance are not new (Beschta, 1997; Johnson and Swanson, 2009). In the 1950 s and 1960 s, reductions in fish habitat and negative impacts on water quality from forest harvesting led regulatory agencies to add protections to streams from forestry practices (Binkley and Brown, 1993; Richardson et al., 2012). Historical forest management activities, such as removal of streamside vegetation followed by broadcast burning, often resulted in substantial increases in stream temperatures, suspended sediment, and nutrients (Levno, 1967; Brown and Krygier, 1970; Moring, 1975; Beschta and Taylor, 1988). As a strategy to mitigate the potentially negative impacts of forest harvesting on aquatic ecosystems, policy makers proposed the retention of vegetated buffer strips during forest operations (Kuglerová et al., 2014). Hence, riparian protection guidelines have been increasingly implemented in many regions since the 1970 s (Richardson et al., 2012).

Since that time, many have illustrated the effectiveness of riparian forests at providing shade, limiting direct solar radiation to the stream, and mitigating changes in stream temperature after contemporary forest harvesting (McGurk, 1989; Kibler et al., 2013; Bladon et al., 2016, 2018). Alternatively, a reduction in riparian canopy cover during forest management activities or other forest disturbances can lead to increased stream temperature sensitivity to atmospheric energy exchanges (Moore et al., 2005a; Gomi et al., 2006; Simmons et al., 2015). For example, complete removal of the riparian buffer during forest harvesting has resulted in increases in maximum and mean daily stream temperatures and diel variability, especially during the summer (Moore et al., 2005b). Substantial thermal responses are of concern as they can influence in-stream primary productivity (Bernhardt et al., 2018) and habitat (Olson et al., 2007; Brewitt et al., 2017; Armstrong et al., 2021) with detrimental impacts on aquatic species, such as salmon and trout (Wondzell et al., 2019). However, there has been considerable variability in stream temperature responses to forest harvesting, which have been attributed to differences in groundwater discharge (Leach and Moore, 2011; Macdonald et al., 2014), steepness of channel slopes (Kasahara and Wondzell, 2003), bed conductive heat transfer (Story et al., 2003), hyporheic exchange (Magnusson et al., 2012; Moore et al., 2005b; Poole and Berman, 2001), or catchment physiography (Callahan et al., 2015; Ebersole et al., 2003).

Despite observations of catchment-specific drivers of stream temperature, current riparian buffer regulations and best management practices generally prescribe fixed-width buffers (Lee et al., 2004; Richardson et al., 2012). The buffer width is determined by criteria such as waterbody size, presence of fish, whether streams are perennial or intermittent, whether streams contribute to a domestic water source, and slope (Castelle et al., 1994; Lee et al., 2004). While fixed-width buffers remain a common practice, there are still uncertainties associated with their effectiveness across heterogenous landscapes. As such, additional work is needed to assess the influence of different riparian buffer prescriptions on stream temperature in forested headwaters.

In our study, we evaluated the effects of three different riparian buffer prescriptions with increasing intensity of basal area removal on stream temperature from fall 2019 to summer 2022 in northern California, United States (USA). The objectives of our study were to (1) examine the seasonal distribution of seven-day moving average of daily maximum stream temperature ($T_{7\text{-}day\ max}$) in both reference (unharvested) and harvested catchments and, (2) quantify the relative importance of catchment physiographic characteristics (e.g., aspect, topography), changes in stream stage (as a proxy for streamflow),

percent shade from the riparian canopy, and the percent catchment area harvested in driving the observed seasonal $T_{7\text{-}day\ max}$ responses across our study catchments. By addressing these objectives in our study, we sought to improve understanding of the complex interactions between forest management, riparian buffer prescriptions, site-specific characteristics, and stream temperature responses in forested headwater streams.

## 2. Materials and methods

2.1. Site description

We conducted our study in 18 catchments in northern California, US (Fig. 1b). We selected headwater catchments that were tributaries to the lower Klamath River (i.e., Tarup, Ah Pah, McGarvey, and West Fork Tectah, East Fork Tectah; Figs. 1c-1f). Catchments were located on private timber land owned by Green Diamond Resource Company (GDRC). All streams were non-fish bearing step-pool systems (Fritz et al., 2020; Montgomery & Buffington, 1997) with a few small cascades. While the streams were considered perennial, dry weather between 2019 and 2022 led to some stream sections going dry during the summer (June–August; streams 4, 11, 13, 15, and 18; Fig. S1).

The catchments ranged in area from 10 to 66 ha and were located between 200 and 695 m in mean elevation (Table 1). Catchments varied in average aspect with east, southeast, southwest, and south directions. Average catchment slope varied between 31% and 60%, with the steepest catchments located in McGarvey Creek sub-region. The climate in the study region is characterized as temperate (Köppen climate classification: Csb, warm-summer Mediterranean) with a distinct wet-dry seasonality with considerable influence from coastal fog (Dawson, 1998). Summers are warm and dry, and winters are mild and wet, with average annual air temperatures of approximately 12 °C (PRISM Climate Group, 2014). The majority of precipitation occurs as rainfall between October and May, with an average annual rainfall of 2800 mm in Tectah Creek, 1800 mm in Ah Pah Creek, 1980 mm in Tarup, and 1870 mm in McGarvey Creek (overall region average 2110 mm; PRISM Climate Group, 2014). Accordingly, the annual peak streamflow typically occurs during mid-winter (December to January) and the annual low flows occur during late summer (August to September).

The primary vegetation cover consisted of 30–60 year-old second-growth Douglas-fir (*Pseudotsuga menziesii*) and coast redwood (*Sequoia sempervirens*). Lower densities of western red cedar (*Thuja plicata*) and western hemlock (*Tsuga heterophylla*) were also present. Other commonly identified tree species in the riparian areas were red alder (*Alnus rubra*) and tanoak (*Notholithocarpus densiflorus*). Soils were typically well-drained gravelly clay loams of the Coppercreek and Sasquatch series, with depths ranging from 70 to 100 cm (Soil Survey Staff, NRCS, 2016). The uppermost soil layers were composed of well drained organic detritus, and field observation showed that soil clay content increased with soil depth. The lithology of the study area consists of marine-derived sedimentary and metasedimentary rock of the Franciscan Complex (Woodward et al., 2011).

2.2. Experimental design

Our study focused on 18 headwater catchments in northern California, with 12 harvested catchments and six unharvested reference catchments (Table 1). These 6 reference catchments are within a managed landscape and have been logged in the past. They are generally located near the harvested catchments. Both reference and treatment catchments were originally logged around the same time-period—about the 1970 s to early 80 s—and, as such, the stand ages in the study catchments were similar. The 12 harvested catchments included four replicates in each of three riparian buffer prescriptions. The riparian buffer prescriptions included: (a) Anadromous Salmonid Protection (ASP) Coastal Anadromy Zone Class II-L prescription with a 30 foot
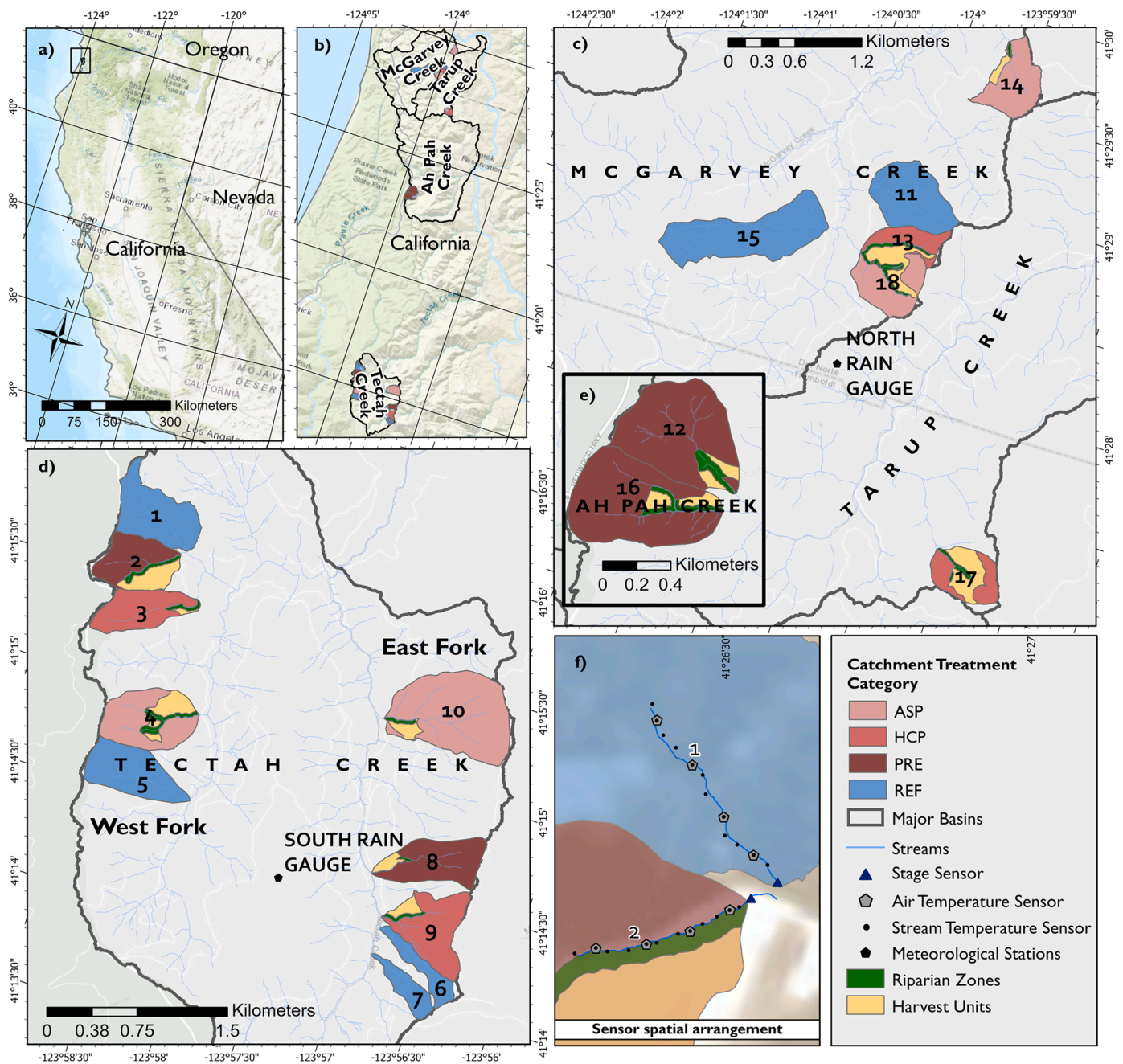
Fig. 1. Maps of (a) the location of the study sites in California and (b) the spatial relation of all study catchments. Additional maps of site identification, harvest units, and riparian buffer prescriptions (reference (REF), Anadromous Salmonid Protection Rules (ASP – 20% of riparian area harvested), GDRC Habitat Conservation Plan (HCP – 30% of riparian area harvested), and before ASP regulations (PRE – 50% of riparian area harvested)) for (c) McGarvey and Tarup Creeks, (d) Tectah Creek, (e) Ah Pah Creek, (f) map providing an example from two sites (sites 1 and 2) illustrating the spatial arrangement of air and stream sensors along the stream channel.

(9 m) unharvested core zone and 70 foot (21 m) outer zone where 80% overstory canopy cover remained and 20% of the riparian area was harvested; (b) GDRC Habitat Conservation Plan (HCP) prescription with a 30 foot (9 m) unharvested core zone and a 70 foot (21 m) outer zone where 70% overstory canopy cover remained and 30% of the riparian area was harvested; (c) an alternative prescription resembling the period before ASP implementation or the pre-ASP (PRE) with a 100-foot (30-meters) riparian zone where 50% overstory canopy cover remained and 50% of the riparian area was harvested.

The pre-harvest period generally occurred between August 2019 and December 2020, but was slightly shorter in four catchments (i.e., 9, 12, 16, 17; Table 1) where it extended until January, March, and May 2020 due to the need for adjustments in the forest harvesting schedule. The

percent clear-cut of the 12 harvested catchments ranged between 2% and 43% with cutblocks only occurring on one side of the stream (Fig. 1). All trees were felled using chainsaws by ground crews and moved to landings using cable yarding systems. In the riparian buffers, individual trees were marked for harvest and were felled into clearings between trees and yarded up to landings by a cable system. Given that we were assessing an industrial operation, there was variability in timing and harvesting intensity across our study catchments that was representative of operational constrains (Table 1).

2.3. Data collection

To quantify the effect of forest harvesting with different riparian

Table 1

Site information, including riparian buffer prescription (REF: Reference; ASP: Anadromous Salmonid Protection Coastal Anadromy Zone Class II-L (20% of riparian area harvested; HCP: GDRC Habitat Conservation Plan – 30% of riparian area harvested; PRE: pre-ASP – 50% of riparian area harvested), catchment area, area harvested, total riparian area, catchment topographic characteristics.

Catchment Number	Catchment Name	Riparian Buffer Prescription	Catchment Area (ha)	Catchment Harvested Area (ha)	Riparian Area (ha)	Catchment Average Elevation (m)	Catchment Average Slope (%)	Catchment Average Aspect	Average Space Between T_s Sensors (m)	Pre-Harvest End Date
1	WF Tectah	REF	32.8	-	-	541	48.1	SE	34	-
2	WF Tectah	PRE	28.5	7.7	2.0	534	47.0	SE	28	2020–11–01
3	WF Tectah	HCP	25.2	0.6	1.1	539	46.4	SE	34	2020–09–14
4	WF Tectah	ASP	37.5	7.1	3.2	539	43.0	SE	30	2020–09–14
5	WF Tectah	REF	30.6	-	-	559	34.4	E	28	-
6	EF Tectah	REF	11.0	-	-	678	32.9	SW	23	-
7	EF Tectah	REF	10.4	-	-	695	34.5	SW	13	-
8	EF Tectah	PRE	30.8	2.4	0.3	647	31.3	SW	32	2020–07–20
9	EF Tectah	HCP	33.5	2.8	1.1	678	33.6	SW	21	2020–05–31
10	EF Tectah	ASP	66.2	2.8	0.9	591	37.1	SW	28	2020–09–26
11	McGarvey	REF	39.8	-	-	209	58.7	S	26	-
12	Ah Pah	PRE	39.0	2.1	2.1	415	50.9	SE	14	2020–03–08
13	McGarvey	HCP	18.8	3.9	2.0	218	47.5	SW	24	2020–10–05
14	McGarvey	ASP	29.1	2.5	-	210	57.1	S	13	2020–08–06
15	McGarvey	REF	61.0	-	-	200	51.2	SE	26	-
16	Ah Pah	PRE	41.9	3.1	2.3	428	43.8	SE	24	2020–03–08
17	Tarup	HCP	28.0	11.9	1.8	316	59.4	S	23	2020–01–02
18	McGarvey	ASP	33.3	6.4	2.9	205	47.7	S	23	2020–10–05

buffer prescriptions on seasonal stream temperature, we installed 12 thermistors (Onset HOBO Tidbit, Bourne, MA; accuracy +/- 0.21 °C) longitudinally along each study stream (i.e., 12 thermistors/stream × 18 catchments = 216 total sensors). Sensors were spaced approximately every 25 m along the thalweg of each study stream (Fig. 1, Table 1). Overall, we were able to space sensors evenly down the stream; however, there were slight deviations due to small barriers such as fallen trees, stream cascades, and boulders (Table 1). Once determined the location, the sensors were enclosed in white PVC tubing with drilled holes to enable a constant flow of freshwater over the sensor and to minimize direct solar radiation. Data were collected at a 15-minute resolution during both the pre-harvest and post-harvest periods at all sites.

Given that streamflow can influence the stream temperature regime (Johnson, 2004), we also quantified stream stage throughout the study. We quantified stream stage at the downstream end of each stream with a pressure transducer (Levelogger Edge, Model 3001, accuracy ± 0.05%, Solinst Canada Ltd., Georgetown, ON, Canada) housed in a PVC stilling well. A barometer (Barologger Edge, Model 3001, accuracy: ± 0.05 kPa, Solinst Canada Ltd., Georgetown, ON, Canada) was placed at the outlet of streams 1 and 11 to quantify atmospheric pressure for compensation of stream stage. Manual measurements of stream stage were taken with a ruler at the base of each stilling well to the nearest half centimeter during field visits to derive continuous stream stage data from the pressure transducers. The pressure transducers and barometers recorded data every 15-minutes throughout the study.

We also installed two meteorological stations (Onset HOBO U30 Data Logger, Bourne, MA) to quantify key climatic variables such as solar radiation, relative humidity, and precipitation. One station was centrally located in the Tectah Creek region while the other station was installed closer to McGarvey, AhPah, and Tarup Creeks region. The stations were located within 3 km of all study streams (Miralha et al., 2023; Fig. 1). All data was collected at a 15-minute resolution.

We collected pre- and post-harvest data on stand structure from six fixed area plots along each of the 18-study stream reaches (i.e., 108 plots total). Fixed area plots ranged between 206 and 260 m² with plot centers established ~18 m perpendicular to the stream. Data was collected on all standing live and dead trees in the riparian area with diameters ≥ 10 cm at breast height (1.37 m above ground). For each tree, we recorded its species, condition (dead or live), diameter at breast height

(DBH), and distance and azimuth from plot center. We used DBH to calculate the basal area per tree (cm²) and the total basal area (m² ha⁻¹) per plot and treatment assigned per stream reach.

We quantified canopy closure and shade using hemispherical photography (Chianucci & Cutini, 2012; Glatthorn & Beckschäfer, 2014). Photographs were taken directly over the center of each of six evenly spaced plots along each of the 18 study streams (i.e., 6 photos per stream reach; 108 photos total). We collected photographs during August each year of the study during both the pre- and post-harvest. Images were taken with a digital single-lens reflex camera (Nikon D7100) equipped with a circular fisheye lens (Sigma 4.5 mm f/2.8 EX DC HSM) mounted on a level tripod 1 m above the surface and facing vertically up into the canopy. We processed the photos using the HemiView software version 2.1 (Delta-T Devices, Burwell, Cambridge, UK) to estimate the percentage of visible sky in each photo and calculate canopy closure (%), effective shade (%), and leaf area index (LAI).

2.4. Data analysis

The continuous monitoring of stream temperature at a 15-minutes resolution enabled us to use a before-after/control-impact (BACI) design to assess changes in stream temperature to harvesting (Moore et al., 2005; Gravelle and Link 2007). Because water temperatures can vary substantially with the occurrence of warmer air temperatures, snowmelt events, humidity, shortwave radiation, air temperature, precipitation events, and streamflow conditions (Johnson and Jones, 2000, U.S. EPA 2001, 2003), we used the 7-day moving average of daily maximum stream temperature ($T_{7-day-max}$) as the main response metric in our study. Research has shown that $T_{7-day-max}$ (a) provides a robust indicator of prolonged warm stream temperatures, (b) is more meaningful to potential impacts on aquatic ecosystems than mean daily stream temperature, (c) and is used as a metric for both regulatory and biota assessments (WA Dept. of Ecology 2002, U.S. EPA 2003; McCullough, Spalding, Sturdevant, & Hicks, 2001). During harvesting activity, sensors were temporarily removed from the stream to prevent damage or loss and were re-installed after harvesting—the time of removal varied between one day to three months. For consistency in our analysis, we removed all data collected during the harvesting period for all sites. We also omitted data for the days when the field crew was offloading the sensors, generally one day every two to three months. After completing

preliminary quality control of the data, we calculated the $T_{7\text{-day-max}}$ for each stream temperature sensor and calculated a single catchment average (i.e., average of 12 sensors per stream reach). Prior to proceeding with the analysis, we also compared the longitudinal gradient of $T_{7\text{-day-max}}$ between the most upstream and downstream stream temperature sensors in each and found that the longitudinal stream temperature signatures were quite stable. As such, we used one averaged $T_{7\text{-day-max}}$ time series per study catchment (Fig. S1).

We investigated the relative influence of forest harvesting, the different riparian buffer prescriptions, catchment climate, and catchment physiography on $T_{7\text{-day-max}}$. For this aspect of the analysis, we included precipitation (mm), catchment area (ha), average catchment aspect, average catchment elevation (m), topographic wetness index (TWI), catchment average slope (%), effective shade (%), percent catchment harvested (i.e., treatment), catchment harvest period (pre- and post-harvest; i.e., timing), and the relative percent change in stream stage.

Relative percent change in stage for a stream i was calculated based on a 7-day moving average of daily maximum stream stage values ($Q_{7\text{-day max}}$):

$$Q_i = \frac{x_{i\ t_2} - x_{i\ t_1}}{x_{i\ t_1}} * 100 \tag{1}$$

where $x_{i\ t_1}$ is the initial value of $Q_{7\text{-day-max}}$ at time 1 and $x_{i\ t_2}$ is the $Q_{7\text{-day-max}}$ at time 2 (i.e., the day following day 1). We opted to use a relative change in percentage metric because it is dimensionless and it allowed easy comparison across all catchments in this study,

Because of the high correlation among basal area, canopy closure, LAI, and effective shade, we elected to include only effective shade in the modeling framework. Effective shade is the percent of incoming solar radiation blocked from reaching a stream surface by physical features of the catchment, such as tree canopies and surface topography (e.g., hills). We quantified effective shade from the global site factor (GSF) output from the analysis of the canopy hemispherical photographs in Hemi-View. GSF is the proportion of global radiation (direct plus diffuse) under a plant canopy relative to that in the open:

$$Effective\ Shade\ (\%) = (1 - GSF) * 100 \tag{2}$$

To calculate the topographic metrics, we used filled 0.25-meter resolution digital elevation models (DEMs) provided by GDRC for each sub-region in ArcGIS Pro version 2.8. We used R (R Core Team, 2020) for data handling and descriptive statistics, and Python version 3.9.7 for all data visualization and random forest modeling tasks.

To address our objectives, we first investigated differences in site characteristics (i.e., precipitation, basal area, and effective shade) before and after harvesting using descriptive analysis and median statistical tests. Then, we tested if there were changes in seasonal $T_{7\text{-day-max}}$ values between the reference and the harvested catchments during the pre- and post-harvest periods. We classified the seasons based on the day of the year. The spring period started in March and ended in June, the summer season started end of June and ended by September 21st, the fall season started in the end of September and ended in December, and the winter was classified between the end of December and beginning of March. We opted to perform individual post-hoc statistical tests per harvest period due to the significant differences in climate conditions between the pre- and post-harvest years. After checking for changes in $T_{7\text{-day-max}}$ and site characteristics, we separated the data into two major modeling datasets: (1) before forest harvesting and during the first year after harvesting (*Before and After 1*), and (2) before forest harvesting and during the second year after harvesting (*Before and After 2*). Once the data was separated by harvesting periods, we ran random forest regression models to assess the most important site factors contributing to the stream temperature patterns observed per season.

2.4.1. Seasonal stream temperature patterns

To account for climatic variability among years and between harvesting periods, we evaluated the seasonal thermal landscape in both the reference and harvested sites (Reiter et al., 2019). To evaluate whether $T_{7\text{-day-max}}$ changed between the pre- and post-harvest periods in the headwater sites, we compared the $T_{7\text{-day-max}}$ seasonal median of each treatment site to the reference $T_{7\text{-day-max}}$ median values. This descriptive comparison allowed us to understand the shift in stream temperature values in both the reference and treatment streams before and after harvesting. For this analysis, we used *tidyverse* (Wickham et al., 2019), *lubridate* (Grolemund and Wickham, 2011), and *stats* (R Core Team, 2020) packages. We tested the seasonal median distribution differences using Kruskal-Wallis, a robust non-parametric test, and performed the Dunn's multiple pairwise comparison test to identify differences among treatment categories in each harvesting period. For this analysis, we used the *pingouin* (Vallat, 2018) and *scikit-posthocs* (Terpilowski, 2019) Python packages.

2.4.2. Stream temperature modeling and variable importance

To model seasonal $T_{7\text{-day-max}}$ and investigate the relative importance of various predictor variables to the thermal regime of the headwater reaches, we used the random forest regression algorithm based on the *scikit-learn* (Pedregosa et al., 2011) package version 1.2.2 in Python. Before analysis, we split the data by seasons (i.e., fall, winter, spring, and summer) and harvest periods (i.e., first set: pre-harvest period and first post-harvest year; second set: pre-harvest period and second post-harvest year). We split the data because of (a) the seasonality in streamflow and (b) the first post-harvest year (post-harvest 1) was slightly wetter (annual precipitation 1765 mm) compared to the second (post-harvest 2) post-harvest year (1505 mm).

We applied a 30–70% train-test-split percentage to the data and calibrated the model with 500 tree splits using our ten climate and catchment physiographic factors as predictor variables (i.e., see Section 2.4.) and $T_{7\text{-day max}}$ as the dependent variable. Our goal was to identify catchment climatic and physiographic characteristics that may help explain some of the variability in $T_{7\text{-day max}}$. To analyze the dependent and independent variables, we used mean absolute error (MAE; °C), mean absolute percent error (MAPE; %), and accuracy (%). The accuracy score (%) expresses the percent of predictions that were closer or the same as the value observed and is a useful metric for assessing random forest regression models (Makridakis, 1993). To check the accuracy, we calculated MAE, which is a metric robust to outliers (Jackson et al., 2019) that expresses the error in the unit of the response variable modeled (i.e., °C). We also calculated MAPE, which is built on the same calculations as MAE, but is expressed as a percentage and makes the comparison among the models per season and harvesting period easier (Makridakis et al., 1979).

In our random forest analysis, we were primarily interested in the feature importance option. However, to avoid overfitting the model due to the intrinsic temporal dependence of our samples, we performed a post-hoc permutation analysis on the validation dataset from the model. This technique, which is known as permutation importance, often solves the overfitting issue because the permutation decreases the importance of features that may be overfitting in the test dataset (Breiman, 2001). We repeated the permutation technique 200 times. If the decrease in accuracy score (i.e., permutation importance) was above zero for a specific input variable, it indicated that the model was more sensitive to that respective variable. A negative score is associated with random noise, implying that the model performance remained the same even when there were changes in that specific input variable, and should be considered as zero. We also plotted the partial dependence plots, which showed the dependence between the dependent variable and each predictor feature in the dataset. These plots enabled us to understand whether the relationship between $T_{7\text{-day-max}}$ and each of the driving factors was linear, monotonic, or complex.

3. Results

3.1. Precipitation

Total annual precipitation (mm) during our study was slightly drier than the long-term average for this region. Our study sites received ~16–30% less average annual precipitation relative to the long-term average (2110 mm) for the region. Annual precipitation during the pre-harvest period (2020 water year) was 1485 mm, during the first post-harvest year (2021 WY) was 1765 mm, and during the second post-harvest year (2022 WY) was 1505 mm (Fig. 2a). Precipitation was generally consistent with the temperate, Mediterranean climate of our study region with wet winters and dry summers. However, during the second post-harvest year the average daily precipitation was slightly greater during both the fall and spring compared to winter (Fig. 2b). Summer average precipitation was also the lowest during the second post-harvest year compared to the previous years.

3.2. Riparian stand characteristics

The median basal area in the riparian areas during the pre-harvest period was 2.8 m^2 ha^{-1} (mean ± SD: 2.9 ± 1.8 m^2 ha^{-1}) in the REF sites, 2.4 m^2 ha^{-1} (2.5 ± 1.3 m^2 ha^{-1}) in the ASP sites, 2.9 m^2 ha^{-1} (2.7 ± 1.3 m^2 ha^{-1}) in the HCP sites, and 3.6 m^2 ha^{-1} (3.2 ± 1.2 m^2 ha^{-1}) in the PRE sites (Fig. 3). Notably, in the sites with the most intense riparian prescription (PRE sites with a target of 50% harvesting) the median basal area was 1.8 m^2 ha^{-1} in the first post-harvest year and remained ~1.8 m^2 ha^{-1} in the second post-harvest year. Statistical analysis of the median difference in basal area indicated very strong evidence that basal area was different between the pre-harvest period and the post-harvest years in sites with the most intense riparian treatment (i.e., the PRE sites; Kruskal Wallis: H =15.4; p < 0.01). Comparatively, we found no evidence for differences in basal area between the pre-harvest period and the post-harvest period in the REF sites (Kruskal Wallis: H = 1.0; p = 0.60) as well as in both HCP (Kruskal Wallis: H = 0.7; p = 0.72) and ASP (Kruskal Wallis: H = 3.7; p = 0.15) treatments (Fig. 3).

Statistically, there was no evidence (Dunn's Z = −1.88 to 1.09; p = 0.36–1.00) for differences in mean basal area between any of the site types during the pre-harvest period (indicated as letter *a* in Fig. 3; Table S1). However, after harvesting, we found strong evidence that basal area in the PRE sites differed from the REF sites in both of the two post-harvest years (letter *b*; Z = −3.08; p = 0.01 and Z = −3.78; p < 0.01). Comparatively, there was no evidence that basal area in the ASP and HCP sites differed from REF sites after harvesting (Z = −1.19 to −0.01; p = 1.00). Our statistical results also provided suggestive evidence of a difference in basal area between the PRE and HCP sites in the
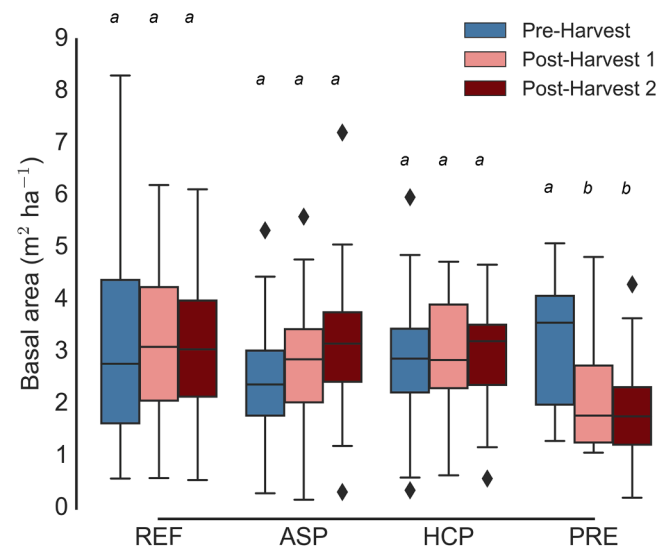
**Fig. 3.** Box plots of basal area (m^2 ha^{-1}) within each of the riparian buffer types during the pre-harvest period and the first and second year after harvesting. Site types: unharvested reference (REF), Anadromous Salmonid Protection (ASP – 20% of riparian area harvested) prescription, GDRC Habitat Conservation Prescription (HCP – 30% of riparian area harvested), and the pre-ASP prescription (PRE – 50% of riparian area harvested). Distinct letters represent the outcomes of Dunn's post-hoc analysis and indicates statistical differences among groups at a significance level of 0.05.

first year after harvesting (Z = 2.47; p = 0.08; letter *a* in Fig. 3). During the second year after harvesting, there was strong evidence for differences in basal area between the PRE and HCP sites (Z = 3.10; p = 0.01) as well as the PRE and ASP sites (Z = 3.38; p < 0.01).

The mean effective shade in the riparian areas during the pre-harvest period was 97.5 ± 2.9% in the REF sites, 96.3 ± 3.1% in the ASP sites, 95.5 ± 5.2% in the HCP sites, and 94.0 ± 6.3% in the PRE sites (Fig. 4a). Statistically, there was no evidence for differences in effective shade between REF and harvested sites during the pre-harvest period (Kruskal-Wallis: H = 4.9; p = 0.18). After forest harvesting, there was 17.5% less effective shade in the most intensive harvesting treatment sites (i.e., PRE) compared to the REF sites. Comparatively, there was only 0.2% less effective shade in ASP sites compared to REF sites and 1.2% less effective shade in the HCP sites compared to REF sites. Statistical tests suggested that effective shade in the post-harvest period was lower in the PRE sites compared to the REF sites (Dunn's Z = −5.26; p < 0.01). During the post-harvest years, the PRE sites had 17.3% less effective
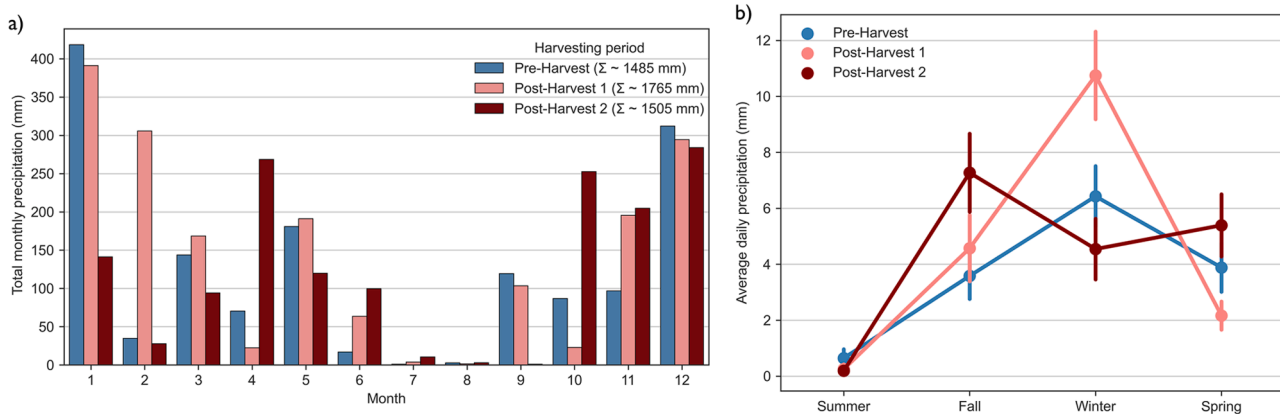
Fig. 2. (a) Total monthly precipitation (mm) during each of the three time periods in our study (pre-harvest, first post-harvest year, second post-harvest year) with the total annual precipitation indicated in the legend labels and (b) average daily precipitation and standard error during each of the seasons and each of the three main study periods.
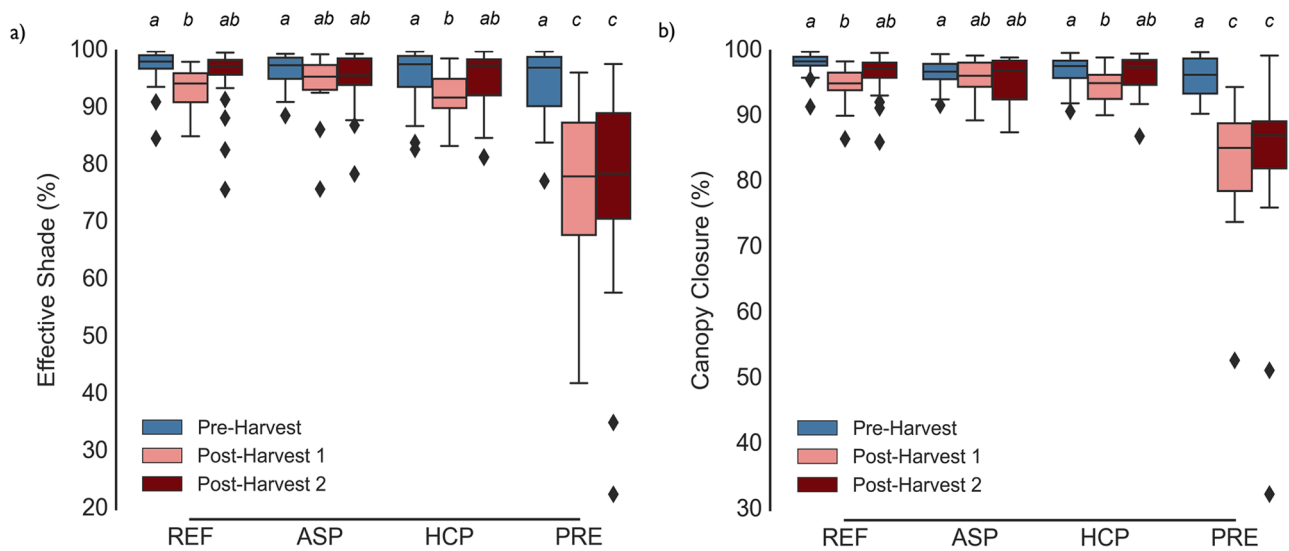
Fig. 4. Box plots of (a) effective shade and (b) canopy closure for each of the site types and study time periods. Site types: unharvested reference (REF), Anadromous Salmonid Protection (ASP – 20% of riparian area harvested) prescription, GDRC Habitat Conservation Prescription (HCP – 30% of riparian area harvested), and the pre-ASP prescription (PRE – 50% of riparian area harvested). Distinct letters represent the outcomes of Dunn's post-hoc analysis and indicate statistical differences among groups at a significance level of 0.05.

shade relative to the ASP sites and 16.2% less effective shade compared to the HCP sites. Statistically, there was strong evidence that effective shade was lower in the PRE sites than in both the HCP (Dunn's $Z_{post\,1} = 3.68$ and $Z_{post\,2} = 4.74$; $p < 0.01$) and ASP ($Z_{post\,1} = 5.81$ and $Z_{post\,2} = 4.33$; $p < 0.01$) treatment types. There was no evidence for differences in effective shade among HCP ($Z_{post\,1} = -1.22$ and $Z_{post\,2} = -0.40$; $p = 1.00$), ASP ($Z_{post\,1} = 1.33$ and $Z_{post\,2} = -0.80$; $p = 1.00$), and REF sites during the post-harvest period.

3.3. Seasonal stream temperature patterns

Before harvesting, the mean seasonal stream temperature ranged from 8.8 °C to 13.3 °C in the REF sites, 9.1–13.3 °C in the ASP sites, 8.8–13.6 °C in the HCP sites, and 8.4–13.7 °C in the PRE sites (Table 2). Statistical tests indicated there was strong evidence for differences in $T_{7\text{-}day\text{-}max}$ among the seasons and riparian harvesting treatments combinations (Kruskal-Wallis $H = 21.3\text{–}102.0$; $p < 0.01$). However, during the summer season, the multiple comparison test revealed that $T_{7\text{-}day\text{-}max}$ before harvesting was not different in the ASP sites ($Z = 0.02$; $p = 1.00$) compared to the REF sites. The stream temperature was cooler in the PRE sites compared to all other site types in all seasons, except during the summer when $T_{7\text{-}day\text{-}max}$ was generally warmer compared to $T_{7\text{-}day\text{-}max}$

$_{max}$ in REF and ASP sites. For all the other seasons, $T_{7\text{-}day\text{-}max}$ in ASP sites did not differ from the REF sites (i.e., fall (Dunn's $Z = -0.25$; $p = 1.00$), spring ($Z = 0.58$; $p = 1.00$)). Similarly, stream temperature during the fall ($Z = -0.10$; $p = 1.00$) and winter ($Z = -0.76$; $p = 1.00$) seasons in the HCP sites did not differ from the REF sites in the pre-harvest period. Additionally, there was no evidence that $T_{7\text{-}day\,max}$ in the ASP sites differed from the HCP sites ($Z = -0.15$; $p = 1.00$) during the fall.

Generally, results suggested that during the first year after harvesting seasonal stream temperature in the harvested sites significantly differed from the reference sites. Relative to the REF sites and the pre-harvest period, the greatest increase in $T_{7\text{-}day\,max}$ occurred in the HCP and PRE sites during the fall (HCP: 0.3 °C and PRE: 1.6 °C), spring (HCP: 1.4 °C and PRE: 1.3 °C), and summer (HCP: 0.5 °C and PRE: 0.6 °C) seasons (Table S2). There was no evidence for differences in $T_{7\text{-}day\text{-}max}$ in the ASP sites during the fall (Dunn's $Z = -1.09$; $p = 1.00$) and summer ($Z = 0.36$; $p = 1.00$) seasons. Although we calculated an overall increase of 0.3 °C in $T_{7\text{-}day\text{-}max}$ in the HCP sites during the fall relative to the REF sites, this difference was not statistically significant ($Z = 1.87$; $p = 0.37$), the same was observed for the winter season ($Z = -0.88$; $p = 1.00$). Among the harvested catchments in the first year after harvesting, stream temperature in the ASP sites did not differ from the $T_{7\text{-}day\text{-}max}$ observed in the HCP ($Z = -1.51$; $p = 0.78$) and PRE ($Z = 1.75$;

Table 2
Seasonal median, mean, and standard deviation (SD) $T_{7\text{-}day\,max}$ in degrees celsius (°C) for each of the site types and study time periods. Site types: unharvested reference (REF), Anadromous Salmonid Protection (ASP – 20% of riparian area harvested) prescription, GDRC Habitat Conservation Prescription (HCP – 30% of riparian area harvested), and the pre-ASP prescription (PRE – 50% of riparian area harvested).

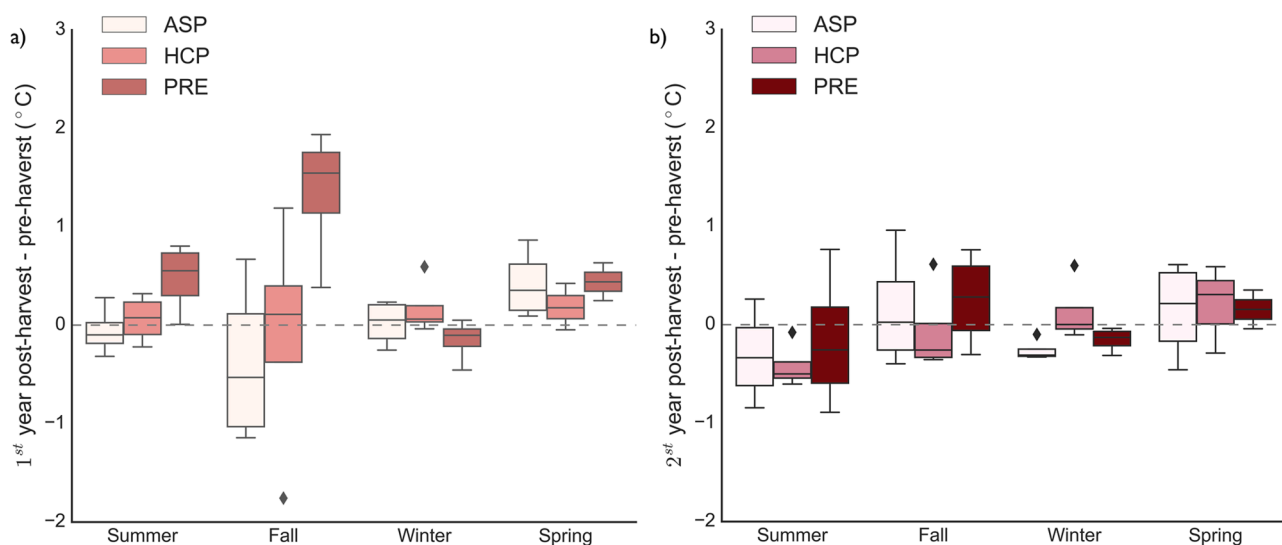| Treatment	Metric (°C)	Pre-Harvest				Post-Harvest 1				Post-Harvest 2			
		Fall	Winter	Spring	Summer	Fall	Winter	Spring	Summer	Fall	Winter	Spring	Summer
REF	*Median*	10.4	8.6	10.2	13.2	10.7	9.2	9.1	13.0	10.6	8.6	9.8	13.1
	Mean	10.5	8.8	10.0	13.3	10.9	9.2	9.3	13.6	10.5	8.6	9.7	13.2
	SD	1.6	1.1	1.2	1.0	1.7	1.0	1.3	1.4	1.0	1.1	1.1	1.2
ASP	*Median*	10.4	9.2	10.4	13.2	10.4	9.6	9.8	13.2	10.9	8.8	9.9	13.0
	Mean	10.4	9.1	10.1	13.3	10.7	9.5	9.8	13.5	10.8	8.8	9.9	12.9
	SD	1.6	0.9	1.2	1.0	1.9	0.8	1.1	1.1	0.9	1.0	0.9	0.8
HCP	*Median*	10.4	8.5	9.6	13.7	11.0	9.1	9.9	14.0	10.6	8.6	10.0	13.3
	Mean	10.4	8.8	9.7	13.6	11.2	9.2	10.1	14.2	10.6	8.6	9.9	13.3
	SD	1.7	1.1	1.2	0.9	2.2	1.0	1.5	1.0	1.2	1.2	1.3	1.0
PRE	*Median*	10.0	8.3	9.3	13.4	11.9	8.6	9.5	13.7	10.4	8.1	9.0	13.3
	Mean	10.1	8.4	9.3	13.7	11.7	8.6	9.7	14.1	10.4	8.1	9.3	13.3
	SD	1.7	1.1	1.2	0.9	2.2	1.0	1.5	1.0	1.2	1.2	1.3	1.0

$p = 0.48$) sites during the spring season. Yet, $T_{7\text{-}day\text{-}max}$ in the HCP sites differed from the PRE sites ($Z = 3.49$; $p < 0.01$) during the spring. In addition, fall stream temperature were on average significantly lower ($\sim0.5\,^\circ$C) in the ASP sites compared to the HCP sites ($Z = -2.88$; $p = 0.02$).

During the second year after harvesting, the mean seasonal stream temperature decreased compared to the first year after harvesting. The $T_{7\text{-}day\ max}$ decreased between $0.2\,^\circ$C and $0.8\,^\circ$C in the catchments that received the most intensive riparian treatment (i.e., PRE; Table 2) compared to the REF sites. There was no evidence that the seasonal stream temperature in the least intensive harvest sites (i.e., ASP) was different from the REF sites during winter (Dunn's $Z = 2.04$; $p = 0.25$), spring ($Z = 1.89$; $p = 0.35$), or summer ($Z = -1.54$; $p = 0.75$) seasons. The same was observed in the HCP sites compared to the REF sites during the fall (Dunn's $Z = 0.89$; $p = 1.00$), winter ($Z = -0.04$; $p = 1.00$), and spring ($Z = 1.76$; $p = 0.47$) seasons. Our tests suggested that the $T_{7\text{-}day\ max}$ in the PRE sites was not different from the REF sites during the fall ($Z = -1.81$; $p = 0.42$) or summer ($Z = 2.25$; $p = 0.15$) seasons. The distributions of $T_{7\text{-}day\text{-}max}$ per site, season, harvesting intensity, and riparian treatment type can be found in the supplementary material (Fig. S2).

We considered the REF sites as climatic and regional controls to assess the changes in stream temperature in the harvested sites. To do this, we first calculated the difference in $T_{7\text{-}day\text{-}max}$ between the pre-harvest period and each of the post-harvest periods, independently for the REF sites and harvested sites. Then, we subtracted the $T_{7\text{-}day\text{-}max}$ change observed in the REF sites from the $T_{7\text{-}day\text{-}max}$ change observed in the harvested sites to remove the change in temperature attributable to climate differences between the study years. This enabled us to isolate the change in $T_{7\text{-}day\text{-}max}$ due to the riparian treatment type. This analysis revealed that the largest difference in $T_{7\text{-}day\text{-}max}$ occurred during the first year after harvesting, principally during summer ($0.5\,^\circ$C) and fall ($1.6\,^\circ$C) seasons, in catchments that received the most intensive riparian harvesting treatment (i.e., PRE; Fig. 5a). However, this difference was greatly reduced in the second year after harvesting, with only a slight increase in stream temperature in the PRE sites relative to the REF sites ($0.3\,^\circ$C) during the fall season (Fig. 5b).

3.4. Stream temperature modeling and variable importance

We fit seasonal random forest models comparing stream temperature

in each of the two post-harvest years with the pre-harvest time period. Results suggested that prediction errors were lower in the winter ($0.3\,^\circ$C) and greater in the fall ($\sim1.0\,^\circ$C) seasonal models (Table 3). Modeling results were approximately the same for the spring and summer models, with accuracy reaching 93% for the spring and 95% for the summer. Both MAPE and accuracy scores were higher for each model fit, with MAPE ranging between 4% and 10% and the general accuracy greater than 90%. These results suggested that our seasonal models usually were reliable in predicting $T_{7\text{-}day\text{-}max}$ based on the selected input variables (i.e., catchment elevation, slope, TWI, shade, aspect, catchment area, precipitation, change in stream stage, timing, treatment).
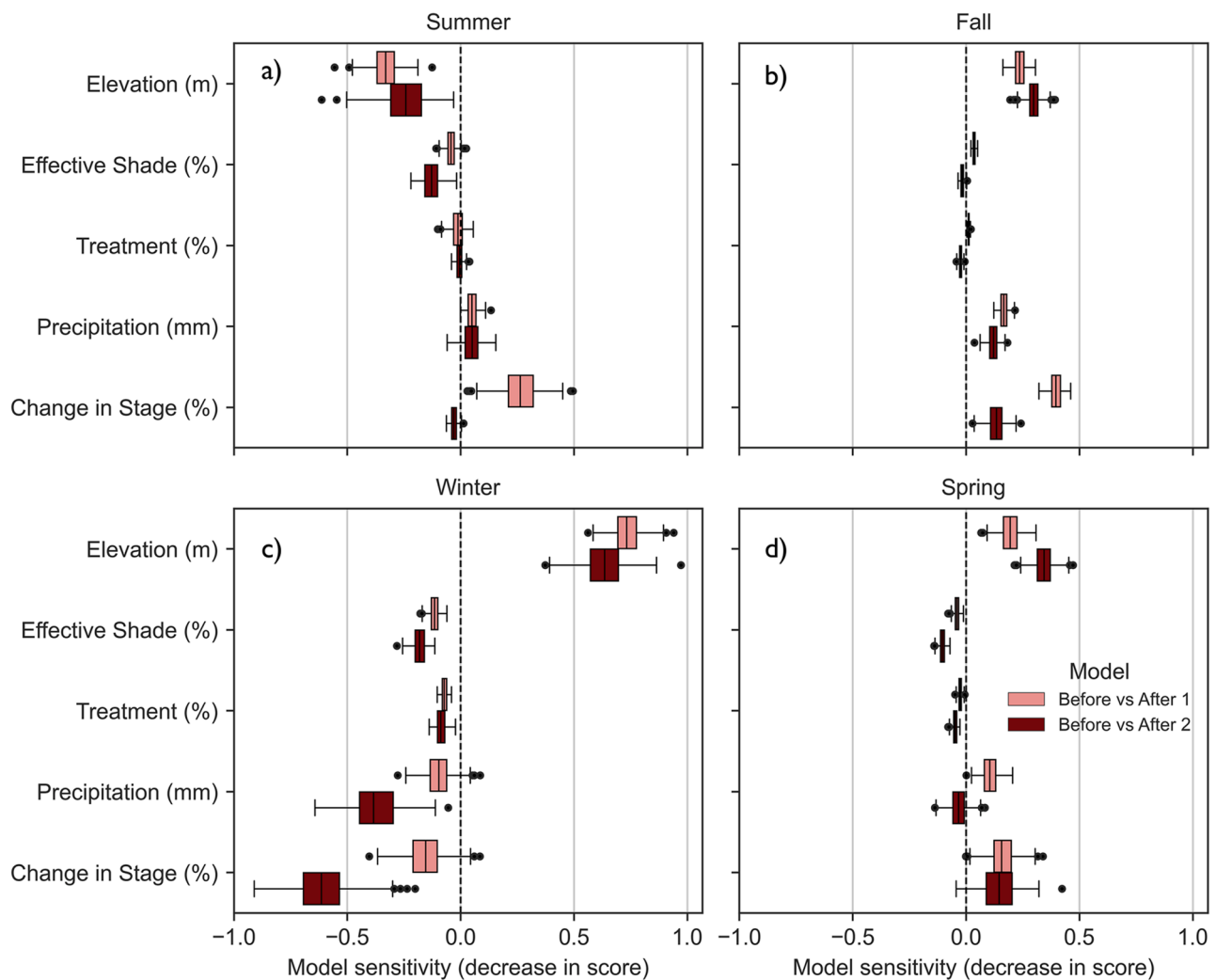
From the empirical correlations, we evaluated the permutation importance of each model input variable when predicting stream temperature per season. Seasonal modeling results for the *Before* vs *After 1* period showed that during the first summer after harvesting, $T_{7\text{-}day\text{-}max}$ was highly sensitive to the changes in stream stage (median change in score: 0.26) and slightly sensitive to precipitation (median change in score: 0.05; Fig. 6a). During the fall, permutation results showed that $T_{7\text{-}day\text{-}max}$ predictions were sensitive to the change in stream stage (median change in score: 0.40), catchment elevation (0.24), and precipitation (0.17), and slightly sensitive to effective shade (0.04) and basin area (0.02; Fig. 6b and Fig. S3). Similar permutation results were observed in the spring season, except for effective shade and basin area input variables. Spring $T_{7\text{-}day\text{-}max}$ model was also sensitive to timing (i.e., pre-harvest and post-harvest time periods; median score: 0.03) while

Table 3

Seasonal random forest modeling results per year after harvesting. *Before* vs *After 1* included data from the pre-harvest period and the first year after harvesting. *Before* vs *After 2* included data from the pre-harvest period and the second year after harvesting. Metrics used to evaluate the models included MAE (mean absolute error in °C), MAPE (mean absolute percent error in %), and accuracy (1-MAPE in %).

Model	Metrics	Summer	Fall	Winter	Spring
	MAE (°C)	0.6	1.1	0.4	0.7
Before vs *After 1*	*MAPE (%)*	4.0	10.0	5.0	8.0
	Accuracy (%)	95.7	89.7	95.4	92.4
	MAE (°C)	0.6	1.1	0.4	0.7
Before vs *After 2*	*MAPE (°C)*	5.0	10.0	4.0	7.0
	Accuracy (%)	95.3	89.8	95.6	92.7

Fig. 5. Box plots of (a) the difference in stream temperature ($T_{7\text{-}day\text{-}max}$) between the first year after harvesting and the pre-harvest period in the harvested sites relative to the reference sites, and (b) the difference in $T_{7\text{-}day\text{-}max}$ between the second year after harvesting and the pre-harvest period. Site types: Anadromous Salmonid Protection (ASP – 20% of riparian area harvested) prescription, GDRC Habitat Conservation Prescription (HCP – 30% of riparian area harvested), and the pre-ASP prescription (PRE – 50% of riparian area harvested).

Fig. 6. Permutation variable importance for (a) summer, (b) fall, (c) winter, and (d) spring $T_{7\text{-}day\text{-}max}$ random forest models per year after harvesting. Lighter boxplots indicate the models for the pre-harvest period and the first year after harvesting *(Before vs After 1)*. Darker boxplots indicate the models for the pre-harvest period and the second year after harvesting *(Before vs After 2)*. For illustrative purposes, we only included the variables that most influenced the model results per season in this figure.

winter $T_{7\text{-}day\text{-}max}$ predictions were most strongly related to topographic metrics (i.e., elevation; Fig. 6c). Complete permutation importance results with all predictive features can be found in the supplementary material (Fig. S3). During the second year after harvesting *(Before vs After 2)*, results differed in the importance of input variables when predicting $T_{7\text{-}day\ max}$ compared to the first-year models. During the summer season, precipitation was the most important predictor for stream temperature (dark color boxplot in Fig. 6a). Similar to the results for the fall season during the first year, catchment elevation, precipitation, and stream stage were important predictors for $T_{7\text{-}day\text{-}max}$ (Fig. 6b). Predictions of $T_{7\text{-}day\text{-}max}$ during the winter were most strongly influenced by elevation followed by a minor model sensitivity to timing (median change in score: 0.06; Fig. 6c). The $T_{7\text{-}day\text{-}max}$ model for the spring season in the second post-harvest year responded to catchment elevation and stream stage, but not to precipitation, which was notably important in the first year after harvest. Overall, we observed that the seasonal variability in $T_{7\text{-}day\text{-}max}$ was not dependent on the harvesting period (i.e., timing relative to the post-harvest period) or the percent of catchment harvested (i.e., treatment). Overall, these results revealed that the seasonal stream temperature responses in the study catchments were most strongly related to the climatic and topographic characteristics of each catchment.

Partial dependence plots indicated that some of the stream

temperature responses may have been related to catchment hydrologic processes. For instance, a positive change in stream stage (%) generally led to a decrease in average stream temperature response during the summer (Fig. 7). We also observed a general decrease in average stream temperature responses with increasing elevation in all seasons—most notably during winter and spring—during the post-harvest period (Fig S4). The average stream temperature response appeared to be slightly dependent on the timing (pre-harvest compared to the post-harvest periods). However, in the model *Before vs After 1*, permutation analysis demonstrated that this change of 0.3 (°C) from before (i.e., timing = 0) to the first year after harvesting (i.e., timing = 1) was poorly related to timing. Rather, we observed a strong inverse relationship between change in stage (%) and stream temperature. Precipitation was also a relevant factor influencing stream cooling and warming (mm; Fig. 6a). Stream cooling generally occurred when precipitation amount was within 10 mm. We observed that stream warming happened when precipitation amount was above 10 mm. However, this relationship between stream temperature and precipitation varied per season. The results also suggested an expected relationship between average $T_{7\text{-}day\text{-}max}$ and effective shade with cooler stream temperatures associated with increases in shade.
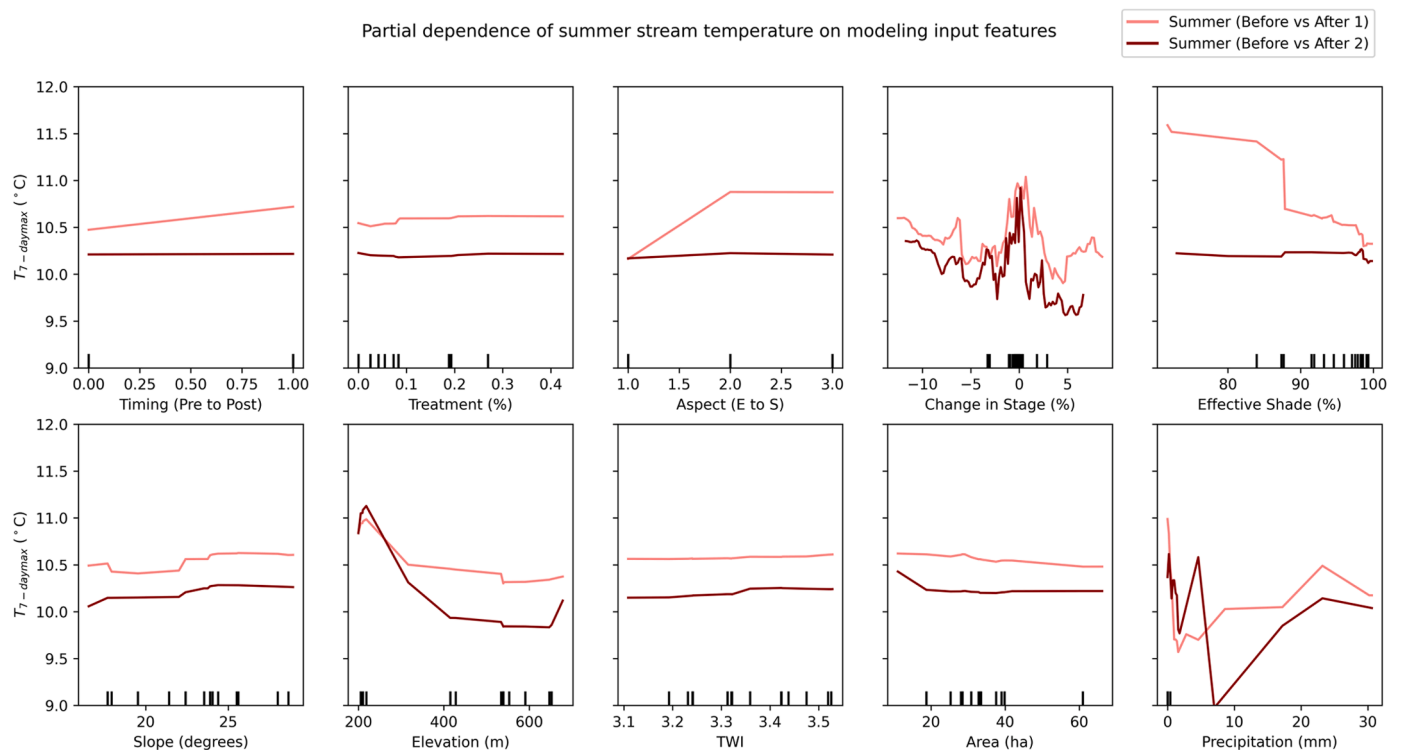
Fig. 7. Example of partial dependence plots for the summer season models between the pre-harvest period and the first and second years after harvesting (*Before* vs *After 1* and *Before* vs *After 2*). These plots illustrate the relationship between average $T_{7\text{-}day\text{-}max}$ (y-axis) and each model input feature (x-axis). Black ticks in the bottom of each plot represent significant breakpoints in the relationship between stream temperature and the input variable.

4. Discussion

In our study, the current riparian area prescriptions (i.e., ASP: 30 ft. unharvested core and 70 ft. outer zone with 80% canopy retention; HCP: 30 ft. unharvested core and 70 ft. outer zone with 70% canopy retention) were generally effective at limiting changes in the 7-day moving average of the daily maximum stream temperature ($T_{7\text{-}day\text{-}max}$) in harvested catchments in northern California. We observed the greatest warming of $T_{7\text{-}day\text{-}max}$ in catchments that were harvested using the historical riparian area prescription (i.e., PRE: 100 ft. 50% canopy retention). Thus, as expected, with our analysis we found a strong relationship between stream temperature and percentage of effective shade, which was consistent with observations on larger stream systems in our study region (Roon et al., 2021). Reduced basal area and effective shade due to harvesting can lead to increased solar radiation and higher stream temperatures (Johnson and Jones, 2000), which may have consequences for aquatic ecosystems and sensitive species such as salmonids (Kibler et al., 2013; Brett, 1952). However, the average canopy cover percent after harvesting in our sites remained high at 83–95%. Garner et al. (2017) explained that under very dense canopy cover (i.e., 70–90%) the majority of solar radiation may be intercepted by the forest canopy, resulting in low incident net energy at the stream surface.

As such, we were also not surprised our modeling results indicated that the increases in $T_{7\text{-}day\text{-}max}$ among all riparian treatment types compared to the reference catchments were strongly dependent on season and were also dependent on other factors, including climatic variability (i.e., precipitation and change in stream stage) and catchment characteristics (i.e., elevation, slope, and area). The additional permutation analysis and partial dependence plots revealed the strong relationship between stream temperature responses after harvesting and the climatic or physiographic characteristics of our study catchments. These results are in agreement with previous research, illustrating relationships between maximum stream temperature and site-specific geology and climate (Johnson, 2003; Bladon et al., 2018). Previously,

topographic controls on stream temperature have been associated with potential groundwater discharge and/or the seasonal shifts in hydrologic connectivity between hillslopes, riparian zones, and streams (Leach and Moore, 2015). During the wettest part of the year (fall to spring), when maximum hydrologic connectivity is achieved, increased groundwater or shallow subsurface flow from upper hillslopes can influence stream temperatures (Uchida et al., 2002). For example, during the fall and spring, stream water in the channel may be dominated by cooler groundwater discharge, resulting in overall stream cooling. Alternatively, in the winter season there may be more shallow subsurface flow and groundwater may be warmer than the water in the stream channel, resulting in stream warming. This hillslope-groundwater interaction may have contributed to the relationship between stream temperature responses and elevation, stream stage, and precipitation in our study catchments.

For example, precipitation inputs varied annually and by season, with the first post-harvest year being slightly wetter than both the pre-harvest period and the second post-harvest year. Variability in precipitation can influence stream temperature through shifts in runoff flowpaths and direct input of precipitation into streams. Miralha et al. (2023) investigated the changes in stream temperature during storm events throughout all seasons in two of our study sub-regions (i.e., McGarvey and West Fork Tectah) and also found strong relationships with season, storm-event rainfall, and catchment characteristics. In that study, the authors found that stream temperature patterns in the Tectah region were associated with meteorological metrics such as 1-hour rainfall intensity, 14-day antecedent rainfall, and 3-day mean solar radiation. Our results also suggested that precipitation was one of the dominant factors controlling stream temperature responses to forest harvesting, especially during summer and spring. In our study, we posit that the greater precipitation during the first post-harvest year may have increased the hillslope hydrologic connection between the drier and warmer portions of the hillslope contributing to increased stream warming in all catchments (Subehi et al., 2010). This is consistent with previous research by

Wilby et al. (2015) who also found relationships between stream temperature and the release of subsurface water from riparian zones during storm events. Increased hillslope hydrologic connection would also be expected following harvesting and the related reduction in evapotranspiration and increased soil water content and groundwater recharge (Surfleet and Skaugset, 2013). However, the muted effect of forest harvesting on stream temperature in our study, suggested that other factors, such as climate and catchment topography modulated the stream temperature response. This is indicative of the increasing effectiveness of current forest management practices, such as smaller clear-cut openings (i.e., not 100% clear-cut), harvesting on only one side of the stream, and greater canopy cover retention in riparian areas, compared to historical forest harvesting practices (Levno, 1967; Brown and Krygier, 1970; Beschta and Taylor, 1988).

It is also possible that other climatic factors, such as shortwave radiation, air temperature, and humidity could have influenced the observed stream temperature changes between the study years; however, we did not quantify these variables in our study. The notion that forest disturbance can influence heat and moisture circulation is not new. Modeling studies have shown that changes in canopy cover leads to decreases in transpiration and increases in soil evaporation resulting in increases in the ratio of sensible to latent heat fluxes (Wiedinmyer et al., 2012). These changes may lead to decreases in longwave radiation at the stream surface potentially facilitating shortwave radiation gains (Klos and Link, 2018). These changes can also be triggered by annual climate variability. More research considering these hydrometeorological influences in stream energy balance is warranted. While our models revealed no dependence of stream temperature on the timing of measurements (pre-harvest vs. post-harvest) or changes in the canopy characteristics associated with the harvesting activity (Fig. S3), it is important to also consider that the summer in the pre-harvest year was wetter compared to both post-harvest summer periods. These drier summer periods after harvesting could have influenced catchment microclimate (i.e., air temperature fluctuations in Fig. S5) and impacted the stream temperatures we observed in our study. For example, modeling studies in Oregon and Washington states illustrated important interactions between catchment microclimate, streamflow, and stream temperature responses (Mantua et al., 2010; Wondzell et al., 2019). Similarly, research from headwater streams in several regions have suggested that the air-water and water-channel bed interfaces may act as both heat sources and sinks, which can influence the variability in seasonal water temperatures (Hannah et al., 2004; Johnson, 2004; Wagner et al., 2014). Similar to our study, these studies highlighted the need to improve our understanding of the complex interactions between non-advective energy exchange (e.g., net radiation, water-channel bed interactions, sensible and latent heat fluxes) and advective energy exchange (e.g., direct precipitation input, sub-surface hillslope runoff, tributary inflows, hyporheic exchange, and groundwater recharge/discharge) in governing the stream temperature response to disturbance. As such, longer term empirical stream energy balance research is needed to improve the characterization of heat flux dynamics in headwater systems and improve our prediction of stream temperature variations over time and space. This type of research is increasingly critical as changes in heat fluxes and microclimate may impact habitat suitability for amphibians, particularly salamander species that are sensitive to microclimate and microhabitat conditions (Huff et al., 2005; Olson et al., 2007). Our study streams are known to support critical habitat for non-fish aquatic species (e.g., southern torrent salamander, coastal giant salamander, coastal tailed frog) and drain into downstream tributaries that support coastal cutthroat trout. However, there is still substantial uncertainty about how alterations in stream thermal regimes impact aquatic ecosystems (Gomez Isaza et al., 2022; Warren et al., 2022), suggesting the need for further research.

Unfortunately, this type of "real-world" empirical research is often complicated and expensive. For example, changes in the forest harvesting schedule and variability in the riparian treatment prescriptions in our study may have impacted our results. Due to real-world constraints on our industrial partner, a few catchments were harvested earlier than expected, limiting our pre-harvest data. In other catchments, the spatial location of the harvested area was shifted at the time of harvesting, reducing the harvest area upstream or adjacent to the installation locations of our sensors. We accounted for these changes in our analysis, but the catchments planned to receive the most intensive riparian harvesting treatment (i.e., PRE) were the catchments that had the most substantial changes in timing and spatial location of harvesting implemented by the landowner. To best address these uncertainties and variability observed in this study, additional research and longer-term monitoring is needed. Further research should consider longer pre- and post- disturbance observation periods to fully understand the recovery and resilience of headwaters after harvesting, principally because of potential interannual climate variability. There is also a need to monitor the long-term integrity of buffers since temperature changes may occur in post-harvest years if trees are dislodged by windthrow. Furthermore, the assessment of other potential factors influencing stream temperature, such as channel morphology, geology, and vegetation composition, which could interact with harvesting practices can be a valuable addition to future research.

5. Conclusions

Our study in 18 forested headwater catchments in northern California provided insights into the stream temperature responses to current harvesting practices, including smaller clearcut areas, forest harvesting on one side of the stream, and greater basal area retention in riparian areas. We were also able to compare current practices with a historical practice with only 50% riparian canopy retention. Our results suggested that current harvesting practices and riparian management were generally effective at limiting substantial stream temperature impacts. Instead, we found that seasonal thermal regimes were strongly related to climatic variability and catchment topographic characteristics. These findings illustrate the challenges in understanding the thermal regimes of headwater streams and their responses to forest disturbances. Model predictions of stream temperature responses to forest disturbances are complicated due to the complex interaction of advective and non-advective energy exchanges that influence a stream thermal regime. Thus, while we did not observe a substantial effect of forest harvesting on stream temperature, we strongly caution against the broad extrapolation of these results as there are likely situations where the current practices could still result in substantial thermal responses due to differences in local catchment characteristics. Additionally, with global climate change and increasing pressures on water resources and aquatic ecosystems, there remains a need for future research to provide further insights into the relationships between forest management activities and the thermal regimes of headwater streams. For example, future research should (1) investigate heat flux responses to disturbances across catchments with a diversity of forest types, geology, physiography, hydroclimatic regimes, and forest management practices, (2) include longer pre-harvest periods to establish stronger linkages between study catchments to enable disentangling the effects of forest management from other drivers, and (3) undertake more holistic interdisciplinary assessments that consider the effects of forest harvesting and riparian management on other important factors, such as terrestrial ecosystem health, aquatic ecosystem health, and socio-economic tradeoffs to inform future forest management policy and decisions.

CRediT authorship contribution statement

LM: original draft, conceptualization, analysis, writing & editing; **KDB:** funding acquisition, conceptualization, writing & editing, supervision; **CS:** funding acquisition, conceptualization, writing & editing, supervision.

Declaration of Competing Interest

The authors declare no conflict of interest.

Data availability

Data will be made available on request.

Acknowledgments

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.foreco.2023.121581.

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