



Stream temperature responses to experimental riparian canopy gaps along forested headwaters in western Oregon



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ABSTRACT

Streamside (riparian) areas in the western United States and across much of North America are dominated by young, regenerating forests with closed canopies, which shade the understory and reduce light to streams. The addition of canopy gaps has been suggested as a management tool to accelerate development of riparian forest complexity, create stream light conditions that mimic those in late successional forests, and enhance in-stream productivity. Although gaps form naturally in late-successional forests, explicitly adding gaps is a concern because the increases in light that accompany a canopy gap may have the potential to increase stream temperature, which is an important ecological driver and regulatory metric in streams. The goal of this study was to determine whether and to what degree riparian forest canopy gaps that reflect localized disturbance events (mortality of one to a few dominant canopy trees) affect stream temperatures. We created experimental gaps in young regenerating riparian forests along six replicate headwater streams in western Oregon. Gaps increased light along 90 m study reaches by $3.91 (\pm 1.63)$ moles of photons $m^{-2} day^{-1}$, similar to that of a naturally occurring gap in a late-successional forest. Using a Before-After-Control-Impact study design, we assessed stream temperature by tracking multiple responses in each reach including: maximum seven day moving average of daily maximums ($T_{7DayMax}$), maximum seven day moving average of daily means ($T_{7DayMean}$), daily maximum, and mean summer temperatures, as well as within-reach (every 30 m) responses, and downstream recovery. Over a 40-day period in summer (July 22nd - August 30th), the mean response in $T_{7DayMax}$ across the six replicate streams was $0.21 ^\circ C \pm 0.12$, and the mean response in $T_{7DayMean}$ was $0.15 ^\circ C \pm 0.14$. Although the mean response in $T_{7DayMax}$ (a key regulatory metric) was small, changes varied across individual study streams, and the magnitude of the relative increase in stream $T_{7DayMax}$ as a result of the canopy gap was strongly negatively correlated with stream size. $T_{7DayMax}$ was not correlated with the size of the canopy opening or change in reach-scale light availability (within the range of gap sizes from $514 m^2$ to $1,374 m^2$ (0.05 to 0.14 ha)). Overall, riparian forest canopy gaps have the potential to increase stream temperatures, but in the western Cascade Mountain headwaters studied here, gap effects were small (all $< 0.5 ^\circ C$), and temperature responses declined as stream size increased.

1. Introduction

Across much of North America, legacies of historic timber harvest have created a landscape dominated by regenerating forests (Pan et al., 2011). Past harvest encompassed both upland and streamside (riparian) areas, but contemporary management regulations commonly restrict timber harvest in the riparian zone (Richardson et al., 2012). Therefore, many riparian forests are currently in the early and middle stages of stand development and will remain uncut (Franklin et al., 2002). The closed canopies of these young riparian forests reduce light flux to

streams (Kaylor et al., 2016, Bechtold et al., 2017), creating conditions that contrast with understory light environments of late-successional forests where gaps in the canopy create heterogeneous light conditions (Canham et al., 1990, Parker et al., 2002, Van Pelt and Franklin, 1999). Because it can take many decades for gaps to form naturally in young forests, the creation of canopy gaps have been advocated as a management tool to help promote restoration or enhance recovery of late-successional forest structure (Keeton, 2006) and associated light environments along streams (Kreutzweiser et al., 2012). However, increased light from canopy gaps could increase stream temperature, a

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key water quality concern in many regions, and therefore, while gap creation has been implemented in terrestrial systems (Keeton, 2006, Tulod et al., 2019), its application in riparian forests is limited. This study explores whether and to what degree creating small, individual canopy gaps in the riparian zone that emulate small-scale natural disturbances affect summer temperatures of headwater streams.

In forested streams, canopy shading is often the dominant control on stream light exposure (Johnson, 2004, Moore et al., 2005b), but the degree of shading by riparian forests changes through time as those forests develop (Warren et al., 2013). Shading generally increases as riparian stands approach maturity and many systems with streamside forests in the middle stages of stand development have stream light levels that are well below those that the system would or did experience when old-growth stands dominated the riparian zone (Kaylor et al., 2016, Bechtold et al., 2017). This difference in light availability to the stream is because old-growth forests have more canopy gaps than mid-succession and early-mature forests (Franklin et al., 2002). Gap development is an important process as forests transition from the stem-exclusion phase toward horizontal diversification stages of stand development (Franklin et al., 2002). In Douglas-fir forests of the Pacific Northwest, gaps occur as a result of tree mortality or crown loss from a range of factors including wind-throw or diseases such as the Douglas-fir bark beetle (*Dendroctonus pseudotsugae* Hopkins). Gaps vary in size as a result of the different mortality events and the size of trees present, therefore, they are often measured in terms of the ratio between gap diameter and average tree height. Smaller gaps generally occur more frequently, and across old-growth stands with highly variable tree heights, gap diameter to tree height ratios are quite variable (Spies et al., 1990, Gray and Spies, 1996). In the Douglas-fir forests of the Pacific Northwest, gap sizes from individual tree mortality events commonly vary from 0.05 to 1 in gap diameter to tree height ratio (Gray and Spies, 1996).

Localized gaps created by smaller disturbance events occur more frequently, particularly in late-successional forests, compared to larger disturbances such as fire, large wind events, and extensive harvests that cover large areas but occur infrequently (Spies et al., 1990). Therefore, in late-successional forests, although the size of canopy openings created by the mortality of a few trees may be small individually, the collective impact of gaps on forest structure and composition can be large (Spies and Franklin, 1989). Individual gaps also have a “footprint” that extends beyond the area of the gap directly below the canopy opening as the sun angle changes increasing light exposure to understory plants and to streams that run through the forest (Franklin and Van Pelt, 2004, Heaston et al., 2017).

Stream exposure is important because light availability often limits primary production in forested headwaters (Hill and Knight, 1988), and changes in light availability can therefore affect resources at the base of stream food webs that transfer energy up the food web (Hetrick et al., 1998b, Kiffney et al., 2004, Wilzbach et al., 2005, Wootton, 2012, Kaylor and Warren, 2017). The increases in primary production that occur beneath canopy gaps may be effective even at small spatial scales because benthic biofilms – dominated by algae – are disproportionately important as a food resource at the base of stream food webs (Thorp and Delong 2002, McCutchan and Lewis, 2002, Cross et al. 2005).

Similarly, temperature conditions in streams are a key regulator of ecological systems and biota through controls on physical, chemical and biological processes (Magnuson et al., 1979, Beschta et al., 1987, Poole and Berman, 2001, Moore et al., 2005b, Caissie, 2006). Increases in temperature can affect aquatic ecosystems via accelerated rates of photosynthesis and growth of primary producers as well as increases in ecosystem respiration (Acuna et al., 2008, Demars et al., 2011, Hill et al., 2014). Similarly, for many ectothermic organisms, metabolic processes and growth rates increase with temperature up to a threshold (assuming adequate food resources), however, once this threshold is exceeded, further increases in temperature can be detrimental, leading to declines in production, condition, and ultimately survival (Sloat

et al., 2005, Bear et al., 2007). Complete removal of riparian forests in headwater ecosystems exposes many streams to high light conditions, and as a result stream temperatures can increase to levels above the thermal thresholds of cold-water adapted species such as salmonid fishes (Brown and Krygier, 1970, Beschta et al., 1987, Sinokrot and Stefan, 1993). Current forest management regulations restrict harvest of riparian forests, thus maintaining riparian buffers that shade streams (e.g. Oregon Forest Practices Act) (Lorensen et al. 1994, Johnson and Jones 2000, McCullough et al. 2001, Moore et al. 2005a, Groom et al. 2011b).

Recent studies have suggested that riparian buffers should be managed to reflect natural disturbance regimes that promote forest structural complexity (Kreutzweiser et al., 2012, Moore and Richardson, 2012, Sibley et al., 2012). In many regions across North America, canopy gaps may become increasingly common as a result of natural stand development processes (Warren et al., 2016). A key question when considering the influence of riparian forest gaps on streams – whether gaps are created naturally or as part of a restoration effort to promote late-successional forest structure – is how much of a change in stream temperature can be expected from a canopy gap that reflects a range of small scale disturbances? Addressing this knowledge gap will allow us to then explore whether creating gaps in a riparian forest is a viable management option. This is particularly important in regions where cold-water adapted species dominate (e.g. salmonid fishes), and timber harvest and water quality regulations explicitly address changes in temperature.

Light exposure is important during summer months because short wave radiation (visible and ultra violet light) is a dominant factor influencing maximum stream temperatures in headwaters across the Pacific Northwest (Brown and Krygier, 1970, Johnson, 2004). Other factors such as long wave (infrared) radiation (Brown and Krygier, 1970, Sinokrot and Stefan, 1993, Johnson, 2004), sensible heat transfer (exchange with air temperature, groundwater, convection or advection), latent heat transfer, and conduction (exchange with the stream bed) (Poole and Berman, 2001, Bogan et al., 2003, Caissie, 2006) can all affect changes in stream temperatures. The relative influence of different components of stream energy budgets depends on stream size, with streams that have smaller discharge being more susceptible, and streams with larger discharge being less susceptible to changes in temperature (Quinn and Wright-Stow, 2008, Poole and Berman, 2001, Hetrick et al., 1998a). The influence of stream size on thermal regimes is well illustrated in a riparian vegetation removal experiment in Alaska where Hetrick et al. (1998a) found that discharge was the dominant factor accounting for the degree of stream temperature change in a single stream over the summer following the removal of 40–70 m sections of streamside forest. Beyond the Hetrick et al. (1998a) study, the effects of small scale canopy openings on stream temperature have received little attention. Given both the fundamental importance of solar radiation to stream heat budgets and the results from Hetrick et al. (1998a) on the influence of stream discharge on stream temperature changes in the context of patchy removal of riparian vegetation, in the current study, we expect both the size of canopy gaps and the size of the stream to influence the response of stream thermal regimes to artificial riparian canopy gaps.

Both natural disturbances and active management designed to emulate local disturbances can create gaps and increase heterogeneity in stream light, yet few replicated field experiments have explicitly assessed the impacts of canopy gaps on streams. To determine the effects of riparian forest canopy gaps and the associated localized increases in light on stream temperature across multiple streams, we created experimental gaps along six replicate headwater streams within young (40–60 year-old) dense riparian forests. We used a Before-After-Control-Impact (BACI) study design to analyze summer stream temperature responses to riparian canopy gap implementation. In this study we created gaps designed to mimic a small natural disturbance in these systems such as a multiple tree mortality or wind throw event. In

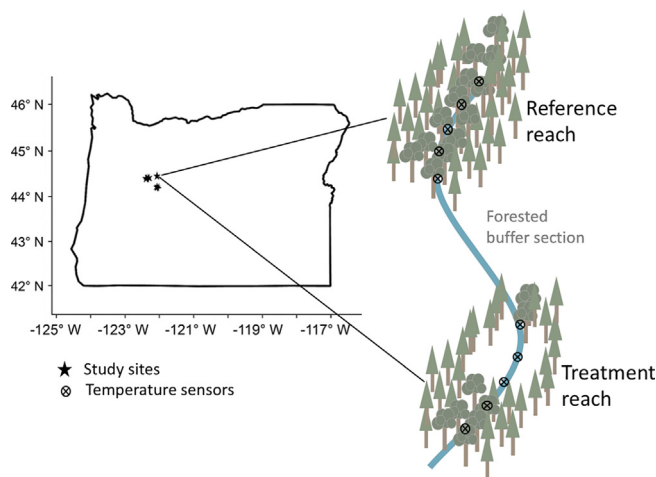


Fig. 1. Map of Oregon with the study stream locations and diagram of a stream containing one reference reach and a treatment reach separated by a buffer reach. Temperature loggers were located every 30 m.

addition to quantifying local temperature responses (e.g. directly beneath gaps), we also evaluated whether and to what degree effects persisted downstream, and evaluated how gap and stream attributes affected potential stream warming. We expected these small gaps to increase temperature locally, but that impacts would be less than 2 °C (Hetrick et al., 1998a) and would dissipate rapidly (within the 120 m reach) downstream of the gap location (Davis et al., 2016).

2. Methods

2.1. Study design and location

This study took place in six replicate streams located within the McKenzie River Basin in the western Cascade Mountains of Oregon (Fig. 1). Each stream consisted of paired reaches, a reference reach and an experimental gap treatment reach. Three of the reach pairs are located on private land owned by Weyerhaeuser Company (W-113, W-100, and W-122) and three are on US Forest Service (USFS) land in the Willamette National Forest (McTE, Loon Creek and Chucksney Mountain Creek). One of the USFS sites (McTE) is located within the HJ Andrews Experimental Forest, a Long-Term Ecological Research site.

The foothills of the western Cascade Mountains of Oregon are characterized by a Mediterranean climate with high precipitation during cool winter months and little to no precipitation during warm summer months. The study systems are mid-elevation (393–867 m) second- and third-order fish-bearing steep step-pool and cascade dominated headwater streams with boulder substrate that ranged from 2.2 to 6.4 m in bankfull width and were lined by 40 to 60 year-old riparian forests (Table 1). Each site's previous harvest left no riparian buffer along the stream. At the initiation of the study, all sites had closed canopies with tree communities that were composed predominantly of red alder (*Alnus rubra*) and Douglas-fir (*Pseudotsuga*

menziesii) with sporadic western red cedar (*Thuja plicata*); the Weyerhaeuser Co. sites also contained bigleaf maple (*Acer macrophyllum*).

Study sites in each stream encompassed two 120 m reaches with no large tributary inputs within or between the study reaches, and reference and treatment reaches were separated by a buffer section of 30–150 m (Fig. 1). At McTE, W-113, Loon and W-100, we applied the experimental canopy modification to the downstream reach and Loon had a buffer section of only 30 m while all others had a minimum of 90 m. Due to concerns about slope stability in the mid-sections of the downstream reaches, the treatments were applied to the upstream sites at Chucksney Mountain Creek (hereafter “Chucksney”) and W-122. At Chucksney and W-122, the buffer sections between reference and treatment reaches are over 100 m long. By using a BACI design we compared reach differences between the pre- and post-treatment years to reduce inherent stream-to-stream environmental variability (e.g. gradient, geology, substrate, etc.) as well as natural variation between pre- and post-treatment periods.

Data for this study were collected during summer (July 22nd – August 30th) in 2016, 2017, and 2018. Pre-treatment data at Loon, W-113, W-100 and W-122 creeks were collected in summer 2017. Due to a large fire in the Willamette National forest and other logistical constraints limiting access to Chucksney and McTE during the summer 2017 field season (the fire did not impact our reaches directly), summer 2017 pre-treatment data were not useable at these two streams. Fortunately, we had deployed loggers in reference and treatment reaches of these two streams during summer 2016 as part of our early site-selection exploration for this study, although only at meters 0 and 90. Therefore, for these two streams, 2016 data were used to characterize the pre-treatment temperature conditions in reference and treatment reaches of McTE and Chucksney.

2.2. Canopy gap treatments

Gaps were created in treatment reaches between late fall 2017 (after leaf fall) and early spring of 2018 (before leaf-out), with the exception of McTE where the gap was cut at in August of 2017. In each treatment reach, we planned for gaps that would create openings in the canopy that were approximately 20 m in diameter over the stream, independent of bankfull width. Gaps were centered on a tree next to the stream at approximately meter 30 along each reach (Fig. 1). The gaps sizes were intended to mimic naturally occurring gaps from an individual large tree mortality or small scale disturbance events found in these systems which range from 0.05 to 1.0 gap diameter to tree height ratio with smaller gaps occurring more frequently (Spies et al., 1990, Gray and Spies, 1996, Gray et al., 2002). Using the Douglas-fir canopy height of 50 m from the HJ Andrews Experimental Forest (Gray and Spies 1996, Gray et al. 2002), we aimed to create gaps in the 0.4–1.0 gap diameter to tree height ratio range (approximately 314 m² – 1,963 m²) because gaps less than a 0.2 gap diameter to tree height ratio have little to no understory light response (Canham et al., 1990, Gray et al., 2002). Because gaps were cut in winter when deciduous trees had no leaves, and due to vagaries of tree fall as well as safety considerations when trees were hung up in felling, some additional, unplanned

Table 1

Site characteristics of the six streams including elevation, coordinates, mean bankfull width, baseflow discharge, mean wetted width, azimuth and aspect and mean gradient.

Stream	Elevation (m)	Latitude	Longitude	Mean Bankfull Width (m)	Baseflow Discharge (L s ⁻¹)	Mean Wetted Width (m)	Azimuth and Aspect	Mean Gradient (%)
McTE	867	44.254544	-122.16672	2.2	5.0	1.6	SW – 221.6°	14
W-113	537	44.192892	-122.51074	3.3	9.1	1.7	SE – 118.2°	11
Loon	721	43.953624	-122.18333	4.1	12.5	1.9	NE – 58.5°	10
Chucksney	833	43.953624	-122.11355	5.2	21.0	2.0	NE – 29.1°	18
W-100	441	44.198130	-122.49298	5.4	43.9	3.7	SE – 136.9°	03
W-122	393	44.195514	-122.46718	6.4	50.2	4.2	SW – 220.7°	02

trees had to be cleared. Actual gap sizes varied across sites from approximately 514 m² to 1,374 m² (0.45 – 0.74 gap ratios) with a mean of 962 m² (mean gap ratio 0.61), which also fall within the gap area distribution of other temperate forests (Foster and Reiners, 1986).

2.3. Riparian shade

To characterize how riparian shade changed due to cutting canopy gaps, we took three to five hemispherical canopy photographs along each reach at 25–30-meter intervals. Photographs were taken in the same location and with the camera at the same height both before and after treatment to determine riparian shading at all sites except Loon and Chucksney which could not be obtained due to a wildfire restricting access in the post-treatment period during the time of summer when photographs needed to be taken. Pictures were taken during low light conditions either dawn or late dusk to avoid sunspots that arises from taking hemispheric photographs in direct sun. We used effective shade (ES) as the metric for riparian canopy cover, which estimates the amount of shade that covers the path of the sun in a given location (Ringold et al. 2003). Changes in effective shade were quantified with the global site factor (GSF) in HemiView™ 2.1 software (Delta-T Devices, Cambridge, UK) and were then calculated as $ES = (1 - GSF) \times 100$, where GSF is the proportion of direct plus diffuse radiation under the canopy relative to that radiation at the given location (latitude and longitude) out in the open. Riparian shade, ES, was averaged for each reach and mean responses in ES were compared between reaches before and after treatment.

2.4. Light

To determine how experimental canopy gaps influenced light reaching the stream, we measured solar radiation by quantifying the 24 h photodegradation of fluorescein dye in vials with 3 vials deployed at each 5 m interval along the length of the stream bed in the center of the stream through control and treatment reaches. Following methods in Bechtold et al., 2012, Warren et al., 2013, and Kaylor et al., 2016, an array of three vials were filled with fluorescein dye (batch concentration of approximately 400 ppm, Turner Designs, Sunnyvale, CA) and were left for 24 h, and then fluorescein vials were collected and concentrations were re-measured. Every fifth array contained a “control” vial that was wrapped in aluminum foil, which prevented any photodegradation and therefore served as a “field blank” that could be used to account for any background drift in fluorescein concentration. All vials were stored in the dark for at least 4 h and brought to room temperature before measurements were taken with a Turner Designs AquaFluor handheld fluorometer. Averaged values from each 5 m interval along a given reach were then averaged to obtain a mean fluorescein decay per reach. Fluorescein decay was converted to daily PAR (photosynthetically active radiation) using the relationship established in Warren et al. (2017) for sites in the HJ Andrews Experimental Forest.

2.5. Site characteristics

Location and elevation values were collected in the field with a hand-held GPS. Azimuth, aspect and gradient values were attained in ArcGIS using digital elevation models of the area. Gradient was calculated for each reach section using changes in elevation divided by stream length and averaged for the two reaches. Mean bankfull width was collected every 10 m along each stream reach and values were averaged. Baseflow discharge values were used from mid-summer measurements from 2017 between July 1st and July 7th using tracer release estimates.

2.6. Stream temperature

We deployed temperature data loggers in the thalweg of the downstream end of each reach at meter 90 to determine how stream temperature responded to experimental canopy gaps. Continuous temperature data were recorded at 15-minute intervals using Onset TidbiT water temperature data loggers (Onset HOBO model UTBI-001, ± 0.2 °C accuracy, ± 0.16 °C resolution) and HOBO Water Temp Pro data loggers (Onset HOBO model H20-001, ± 0.2 °C accuracy, ± 0.16 °C resolution) and were deployed mid-July through early September in order to capture when stream temperatures are most likely to be at their peak. Loggers were removed from the water to prevent potential damage to the sensors during electro-fishing sampling events that occurred once per reach each summer. To avoid skewing daily values, the entire day of data during these events were removed. Temperature loggers were housed in white PVC piping with holes and placed parallel to flow to prevent the influence of direct solar radiation and allow for adequate flow through the piping. Prior to deployment, loggers were validated against one another in a well-mixed ice bath for 1 h recording every 15 s while allowing ice to melt and temperatures to warm. Loggers out of the range of accuracy specified by the manufacturer (± 0.2 °C) were replaced and not included for this analysis. Although the manufacturer provides (± 0.2 °C) as the accuracy for both of these sensors, a study by Groom et al. (2018) found only two instances out of 500 pre- and post- deployment assessments with logger errors > 0.1 °C.

Numerous metrics can be extracted from long-term temperature data (Arismendi et al. 2013). We focused on four key metrics that apply most directly to management regulations in quantifying the stream temperature responses to riparian canopy gaps. We first used the 15-min data to calculate daily mean and daily maximum temperatures for each logger location for the same 40-day period (from July 22 to August 31) where we have consistent data at all locations during both years, and which typically encompasses the time of maximum temperatures in streams of this region (Groom et al., 2011a, Arismendi et al., 2013).

Stream temperature can vary a great deal over the summer, so to focus on the maximum potential effect of the treatments when temperatures are at their peak, we also calculated the maximum of the 7-day moving average of the daily maximums ($T_{7\text{DayMax}}$) and maximum 7-day moving average of the daily means ($T_{7\text{DayMean}}$). These two temperature metrics are recommended by the U.S Environmental Protection Agency as they are not overly sensitive to values on a single day, but still provide an indication of the peak stream warming (McCullough et al., 2001, Bladon et al., 2016, Groom et al., 2017, USEPA, 2001). The $T_{7\text{DayMax}}$ also aligns with many regulatory criteria. For example, in Oregon, the Protecting Cold Water criterion prohibits warming of existing cold waters from anthropogenic increases by prohibiting an increase of more than 0.3 °C in the $T_{7\text{DayMax}}$ where salmon, steelhead and bull trout are present (ODEQ, 2004). Further, common regulatory standards for the $T_{7\text{DayMax}}$ are 16 °C for core cold-water fish rearing habitat, 18 °C for non-core juvenile rearing and migration, and 20 °C for migration of salmon and trout (USEPA, 2003, ODEQ, 2004).

In addition to evaluating the local temperature responses at the downstream end of each 90 m reach, we also quantified finer-scale longitudinal patterns in temperature responses along each study reach. To do so, we placed loggers every 30 m along each reach between meters 0 and 120 (Fig. 1). In Chucksney and McTE we only have data from meters 0 and 90 in the pre-treatment year as previously mentioned. At W-113, the logger at meter 60 of the reference reach failed in the pre-treatment year. In the analysis of longitudinal profiles of stream temperature, following methods described by Arismendi and Groom (2019) we set the value of the most upstream sensor of the reach (0 m) to 0 to look at finer-scale longitudinal patterns through control and treatment reaches during pre- and post-treatment years. We focused on the $T_{7\text{DayMax}}$ temperature metric for this analysis because it characterizes responses during the hottest week in each summer period and

therefore the results of this analysis represent a picture of the largest potential responses observed without overly weighting a single day.

2.7. Statistical analysis

In order to determine the effects of experimental canopy gaps on stream light as well as reach responses in $T_{7\text{DayMax}}$ and $T_{7\text{DayMean}}$, we fit a linear mixed-effects model fit by REML using the nlme package (Pinheiro et al., 2016) in R (R Core Team, 2014) to the data. The following statistical model was used to describe the linear model estimating the response variables:

$$T_i = \text{Period} + \text{Reach} + \text{BACI} + (1 | \text{Stream/Reach}) + \varepsilon_i$$

where ε_i is the random error term for the i^{th} group, where $\varepsilon_i \sim N(0, \sigma_{\varepsilon}^2)$ and ε_i and $\varepsilon_{i'}$ are independent. For all response variables the model included the fixed effects of Period (Pre or Post), Reach (Reference or Treatment), and the BACI effect, which is the interaction term of Period and Reach identifying the effect of the imposed gap. Additionally, random effects for Stream and Reach were included as nested random effects to account for inherent variation between streams. To account for non-constant variance we relaxed the assumption of constant variance in the model using a weights argument.

To assess changes in daily maximum and daily mean temperatures due to the gap rather than summarizing over the summer, we applied the same model as above, however due to the repeated (daily) measurements over the 40-day survey period, we also tested for temporal autocorrelation and evaluated changes in the relationship between reference and treatment reaches in each stream. We tested four correlation structures and chose the best model based on the lowest AIC value, and therefore included the corCAR1 term to account for autocorrelation in the analysis of daily data. Further, to visualize and quantify gap effects over the full 40-day survey period on daily temperature metrics, we used a regression approach to compare the relationships between reaches during each summer period to evaluate the difference in warming within the reference and treatment reaches between years. This approach treats the reference reach as the independent variable and the treatment reach as the dependent variable and has been used in multiple other temperature assessment studies to capture processes occurring over a summer season, rather than compressing data to a single value (Groom et al. 2011a, Kibler et al. 2013, Bladon et al. 2016). In each period – pre- and post-treatment – the regressions evaluated the relationship between daily temperature (maximum and mean) at the downstream end (meter 90) of the reference reach and the downstream end of the treatment reach. The slope of the relationship in the pre-treatment period is assumed to reflect the ambient relationship between reaches and accounts for any warming or cooling that may occur naturally as water moves from one reach to another in these systems. Differences in the slope of the relationship between the pre- and post-treatment periods are attributed to the gap treatment.

Finally, to evaluate our expectations about light and stream size as key environmental factors that could best explain the variability in temperature changes across the six study sites, we explored how the magnitude of the changes observed in the BACI analysis related to metrics of stream size and gap size. The BACI response is the reach

difference in the post-treatment period minus the reach difference in the pre-treatment period. We focused on responses in $T_{7\text{DayMax}}$ for this analysis because of the metric's biological and regulatory relevance noted above and to assess the maximum potential effect of the gap without too heavily weighting a single day. We regressed $T_{7\text{DayMax}}$ responses against four explanatory variables: light exposure (mid-summer daily PAR) response, gap area, baseflow discharge, and bankfull width. These metrics relate to one of two overarching drivers: solar exposure or stream discharge. Changing light exposure is the key mechanistic process that is expected to be related to a response in temperature because solar radiation is a principal factor for stream temperature (Brown and Krygier 1970, Sinokrot and Stefan 1993, Johnson 2004). Therefore, measuring the relationship with a measure of light exposure is important, however, light is not a metric that is commonly included in stream assessments. Riparian gap size can be evaluated more easily by managers and practitioners in the field or even remotely via Light Detection and Ranging (LiDAR), if those data are available, than measuring light reaching the stream bed. Similarly, we consider both stream baseflow (mid-summer) discharge measured using the salt tracer-dilution gauging method where discharge is equal to the volume times the concentration of the injectate divided by the integration of the time-concentration curve. Releases were conducted at all six sites within a one-week period at the end of July, and the mean bankfull width of each stream in the regression analysis. Discharge describes the volume of water moving through the stream and the rate is therefore fundamental in characterizing the energy inputs and exports of the reach. However, it is less common to quantify discharge in rapid assessments and stream discharge can change over time. Conversely, stream wetted widths and stream bankfull widths are easier to measure than discharge. Bankfull width in particular is a useful and consistent measure of relative stream size that is applicable across seasons. Assessments of light and discharge are expected to align more closely with the mechanisms that account for an instantaneous change in stream temperature, while an assessment of the gap size and stream bankfull width are expected to reflect these mechanistic processes in two metrics that are seasonally integrated and more easily assessed. These regressions functionally address the question of whether the changes in temperature that we see in response to the treatment are best explained by factors relating to gap size and light, or whether they are more closely related to underlying stream characteristics for the range of stream and gap sizes evaluated here. Although multiple factors may ultimately affect stream temperature, given low statistical power with only six replicate treatments, we could not perform multiple regression analysis.

3. Results

3.1. Gap size, shading and light

Across the six sites mean gap size was 962 m² (0.09 ha) (SD $\pm$ 316 m²), and individual gaps ranged in size from 514 m² to 1,374 m² (0.05 to 0.14 ha) (Table 2). Light exposure responses, as expected, increased after the canopy gaps. In the pre-treatment period, the light fluxes to the reference and treatment reaches were similar with a mean difference of only -0.10 (SD $\pm$ 0.33) moles of photons m⁻²

Table 2

Gap sizes and BACI responses (changes in reach differences before and after the treatment) of $T_{7\text{DayMax}}$, $T_{7\text{DayMean}}$ and changes in the slopes for the reference versus treatment reach relationships of daily maximums and daily means between periods.

Stream	Gap Size (m ²)	$T_{7\text{DayMax}}$ Response (°C)	$T_{7\text{DayMean}}$ Response (°C)	Δ in Slope- Daily Maxes (°C/°C)	Δ in Slope- Daily Means (°C/°C)
McTE	1374	0.36	0.44	0.34	0.22
W-113	713	0.29	0.07	0.05	-0.01
Loon	514	0.26	0.09	0.06	0.06
Chucksney	1113	0.15	0.10	0.02	0.01
W-100	1164	0.19	0.13	0.06	0.03
W-122	892	0.01	0.08	0.04	-0.01

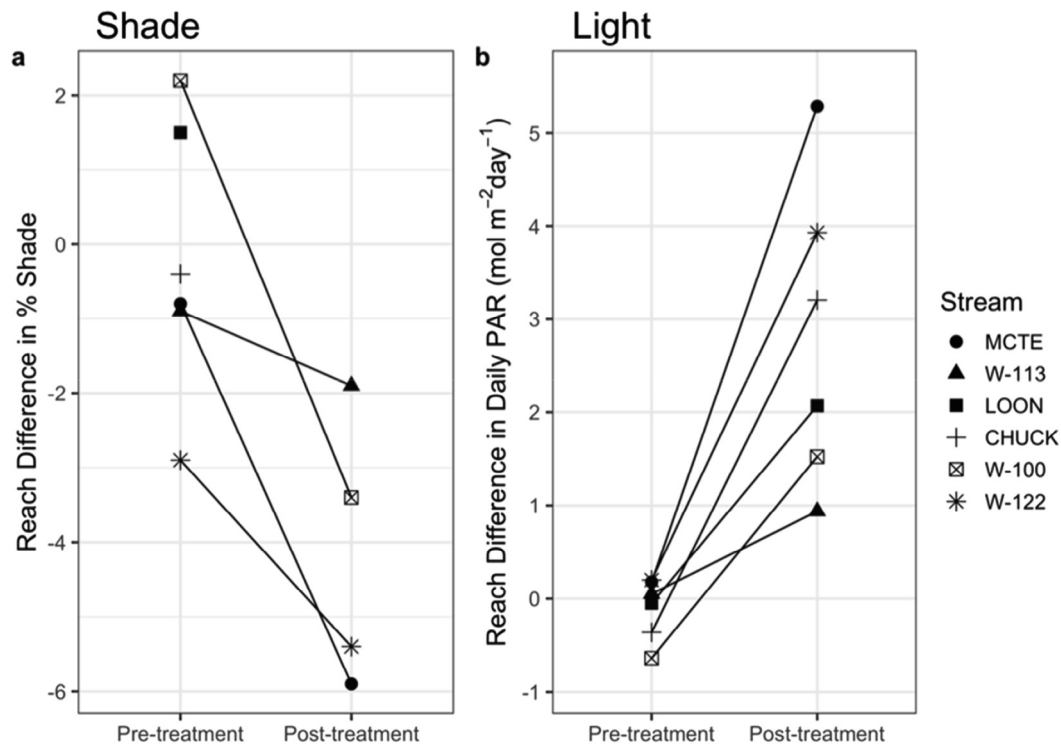


Fig. 2. Differences in effective shade (%) between reaches (treatment reach – reference reach) (a) and reach differences in mean daily PAR in moles of photons $\text{m}^{-2} \text{day}^{-1}$ (b) between the treatment reach and the reference reach for each period. Data were collected for each reach during one day either early morning or at dusk during late August before leaf-fall in the summer in 2017 for the pre-treatment period, and in 2018 for the post-treatment period. Reach differences in shading decreased and reach differences in mean daily PAR increased in the post-treatment period. Data for Loon and Chucksney were unavailable for shade post-treatment.

day^{-1} . After gaps were cut, the BACI analysis showed strong evidence for significant increase in mean reach light ($p < 0.01$, F-statistic = 15.62) to a mean of 3.91 ($\text{SD} \pm 1.63$) moles of photons $\text{m}^{-2} \text{day}^{-1}$ overall resulting in a mean change in light of 2.93 ($\text{SD} \pm 1.50$) moles of photons $\text{m}^{-2} \text{day}^{-1}$. Mean stream shading could not be evaluated in the full BACI analysis because post-treatment hemispherical photographs could not be taken at all sites due to fire impeding access in 2018. Among the 4 sites where we have pre- and post-treatment estimates of shade, pre-treatment effective shade was similar between reference (90.3% , $\text{SD} \pm 0.03\%$) and treatment (90.5% , $\text{SD} \pm 0.03\%$) reaches. After the gap treatment, the reference reaches remained 90.0% ($\text{SD} \pm 0.01\%$) shaded. While the areas beneath each gap had notable localized declines in shade, through the entirety of the treatment reach mean shading declined by only 4% ($\text{SD} \pm 0.02\%$) (Fig. 2). Considering sites individually, McTE had the largest gap ($1,374 \text{ m}^2$), which led to the largest increase in light response ($5.11 \text{ mol of photons m}^{-2} \text{day}^{-1}$), whereas Loon and W-113 had the smallest gaps (514 and 713 m^2 respectively) and the smallest increases in light responses (2.12 and 0.89 and moles of photons $\text{m}^{-2} \text{day}^{-1}$).

3.2. Summer temperature reach responses ($T_{7\text{DayMax}}$, $T_{7\text{DayMean}}$, daily maximums and daily means)

Overall, the gap treatments did not change summer $T_{7\text{DayMax}}$ or $T_{7\text{DayMean}}$ significantly across the 6 study sites (Fig. 3, Table 3). The mean response (change in reach difference before and after the cut) indicated an increase on average across the six sites in $T_{7\text{DayMax}}$ of 0.21°C ($\pm 0.12^\circ\text{C}$) and in the $T_{7\text{DayMean}}$ of 0.15°C ($\pm 0.14^\circ\text{C}$); however, there was not statistical support of the BACI effect for either metric ($p = 0.35$, F-statistic = 0.96 , $p = 0.53$, F-statistic = 0.42). Considering each site separately, McTE had the largest BACI response for both temperature metrics ($T_{7\text{DayMax}} = 0.44^\circ\text{C}$, $T_{7\text{DayMean}} = 0.36^\circ\text{C}$). The smallest response for $T_{7\text{DayMax}}$ occurred at W-

122 ($T_{7\text{DayMax}} = 0.01^\circ\text{C}$) and the smallest $T_{7\text{DayMean}}$ response was at W-100 ($T_{7\text{DayMean}} = 0.09^\circ\text{C}$) (Table 2).

In contrast to the summary values, results from the analysis of individual days throughout the full 40-day summer period identifying differences in the relationships of daily maximums and daily means between reaches showed a statistically significant effect of the gap for average daily maximums ($p < 0.01$, F-statistic = 14.25) and for average daily means ($p = 0.02$, F-statistic = 5.16) (Table 3). Results from the regression comparisons of reach relationships of individual daily maximums and daily means showed that the slopes were greater in the post-treatment year (Fig. 4). However, the increases in slopes for daily maximums were very small with a mean estimated difference of 0.10 (± 0.12) $^\circ\text{C}/^\circ\text{C}$. When comparing the reference and treatment relationships between years for the daily means, an increase in slope was found at only four out of six sites (Appendix A.2). The mean estimated difference in slopes for daily means was 0.05 (± 0.09) $^\circ\text{C}/^\circ\text{C}$. The difference in slope indicates that for every one degree increase in maximum daily temperature in the reference reach, the regression comparison reveals there will be on average an additional $0.12^\circ\text{C}/^\circ\text{C}$ increase in daily maximum temperature in the reach with a gap. Likewise, for the daily mean, for every degree increase in the shaded reference reach, an average additional increase of 0.05°C in a reach with a small gap is expected.

When evaluating individual sites, the differences in slopes for the daily maximums and daily means before and after the cut were largest at McTE which were 0.34 and $0.22^\circ\text{C}/^\circ\text{C}$ respectively. These differences were much greater than the differences for all other sites which ranged between 0.02 and $0.06^\circ\text{C}/^\circ\text{C}$ for the daily maximums and -0.01 and $0.06^\circ\text{C}/^\circ\text{C}$ for the daily means (Fig. 4, Appendix A.2). At W-113 and W-122, the differences in slopes for the daily means between years were negative indicating less warming in the treatment reach than in the reference reach after the gap.

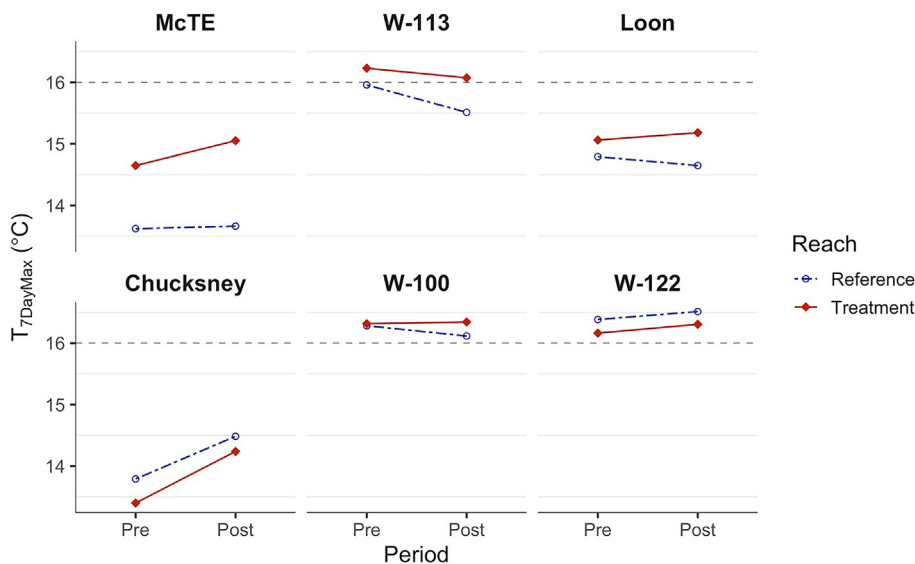


Fig. 3. Maximum of the 7 day moving average maximum ($T_{7\text{DayMax}}$) temperatures between July 22nd and August 30th in the pre-treatment and post-treatment years (before and after the gaps were cut) for each site at the downstream end (meter 90) of each reach for the treatment reaches (red) and the reference reaches (blue). Grey dashed lines are at 16 °C. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 3

BACI statistical results for Mean PAR, reach $T_{7\text{DayMax}}$ and $T_{7\text{DayMean}}$ values, and daily means and daily maximums including estimates, standard errors, F-statistics, degrees of freedom, and p-values for the reach means of PAR, and $T_{7\text{DayMax}}$ and $T_{7\text{DayMean}}$, daily maximum and daily mean reach responses to the gap treatment.

	Model Coefficients		Effect Sizes		
	Estimate	Standard Error	DF	F-statistic	p-value
Mean PAR					
Intercept	0.91	0.22	10	33.42	0.002
Period	0.30	0.21	10	15.11	0.003
Reach	-0.13	0.17	5	0.01	0.918
BACI	3.00	0.76	10	15.62	0.003
$T_{7\text{DayMax}}$					
Intercept	15.14	0.42	10	1555.10	< 0.001
Period	0.02	0.16	10	2.85	0.122
Reach	0.17	0.27	5	1.97	0.220
BACI	0.21	0.22	10	0.96	0.350
$T_{7\text{DayMean}}$					
Intercept	14.45	0.31	10	2729.85	< 0.001
Period	-0.09	0.16	10	0.02	0.883
Reach	0.07	0.25	5	2.87	0.151
BACI	0.15	0.23	10	0.42	0.532
Daily Maximums					
Intercept	14.41	0.46	908	1009.24	< 0.001
Period	0.07	0.05	908	24.60	< 0.001
Reach	< 0.01	0.14	5	0.71	0.438
BACI	0.31	0.08	908	14.25	0.002
Daily Means					
Intercept	13.66	0.34	908	1675.43	< 0.001
Period	0.03	0.05	908	7.99	0.005
Reach	0.02	0.09	5	1.41	0.288
BACI	0.17	0.07	908	5.16	0.023

3.3. Longitudinal patterns/downstream effects

Longitudinal patterns in stream temperature ($T_{7\text{DayMax}}$) generally increased naturally with distance downstream in both reference and treatment reaches pre- and post-treatment (Fig. 5). Localized post-treatment responses in the treatment reach associated with the canopy gaps were variable between sites. In some sites such as McTE, W-113 and Loon, we observed distinct increases in temperature profiles along the stream associated with the gap that ranged between 0.2 and 0.8 °C. In other sites such as Chucksney, W-100 and W-122, we observed minimal to no change in temperature associated with the gap. In the sites where temperatures increased associated with gap, we observed varied downstream trajectories. In McTE and Loon, increases in

temperature remained elevated at 60 meters downstream of the gap, whereas in W-113 increases in temperature dissipated to initial conditions. The magnitude of temperature changes associated with the gap was more pronounced at the sites with smaller discharges and more-narrow bankfull widths.

3.4. Environmental covariates

To explore the potential drivers of variability in $T_{7\text{DayMax}}$ responses amongst sites we tested relationships with each of the following explanatory variables: light response ($\text{mol m}^{-2} \text{ day}^{-1}$), gap area (m^2), baseflow discharge (L s^{-1}), stream wetted width (m) and stream bankfull width (m). The light response was not correlated with $T_{7\text{DayMax}}$ responses ($r^2 < 0.01$, $p = 0.69$), nor was gap area ($r^2 = 0.01$, $p = 0.63$), but there was a significant relationship between discharge ($r^2 = 0.73$, $p = 0.03$), and bankfull width ($r^2 = 0.93$, $p < 0.01$) and the $T_{7\text{DayMax}}$ response (Fig. 6). Wetted width was also highly correlated with $T_{7\text{DayMax}}$ responses, but the relationship was not as strong with this stream size metric as with discharge or bankfull width ($r^2 = 0.65$, $p = 0.05$).

4. Discussion

The objective of this study was to evaluate summer stream temperature responses to experimental canopy gaps along forested headwater streams. Temperature is a master variable influencing aquatic ecosystems processes and biota (Magnuson et al. 1979), and given the dominant role of solar radiation in the stream heat budget (Caissie 2006), even small changes in riparian canopy cover have the potential to increase stream temperature (Johnson and Jones 2000, Moore et al. 2005a). We observed consistently small, non-statistically significant BACI responses in the $T_{7\text{DayMax}}$ and $T_{7\text{DayMean}}$ temperatures in the reaches with experimental canopy gaps across the six study streams after treatment, and small, but statistically significant BACI responses in daily maximums and daily means. Although the magnitude of the response in the common regulatory metric of $T_{7\text{DayMax}}$ was small, with only one of six streams exceeding 0.3 °C, we observed variability in the relative increases in temperature across the six study streams. Within the range of headwater stream sizes and canopy gap sizes evaluated here, temperatures in smaller streams were more responsive than temperatures in larger streams to canopy gaps. This finding is evidenced by a strong negative relationship between stream size and the magnitude of temperature increases, and the absence of a relationship between temperature responses and either gap size or light responses.

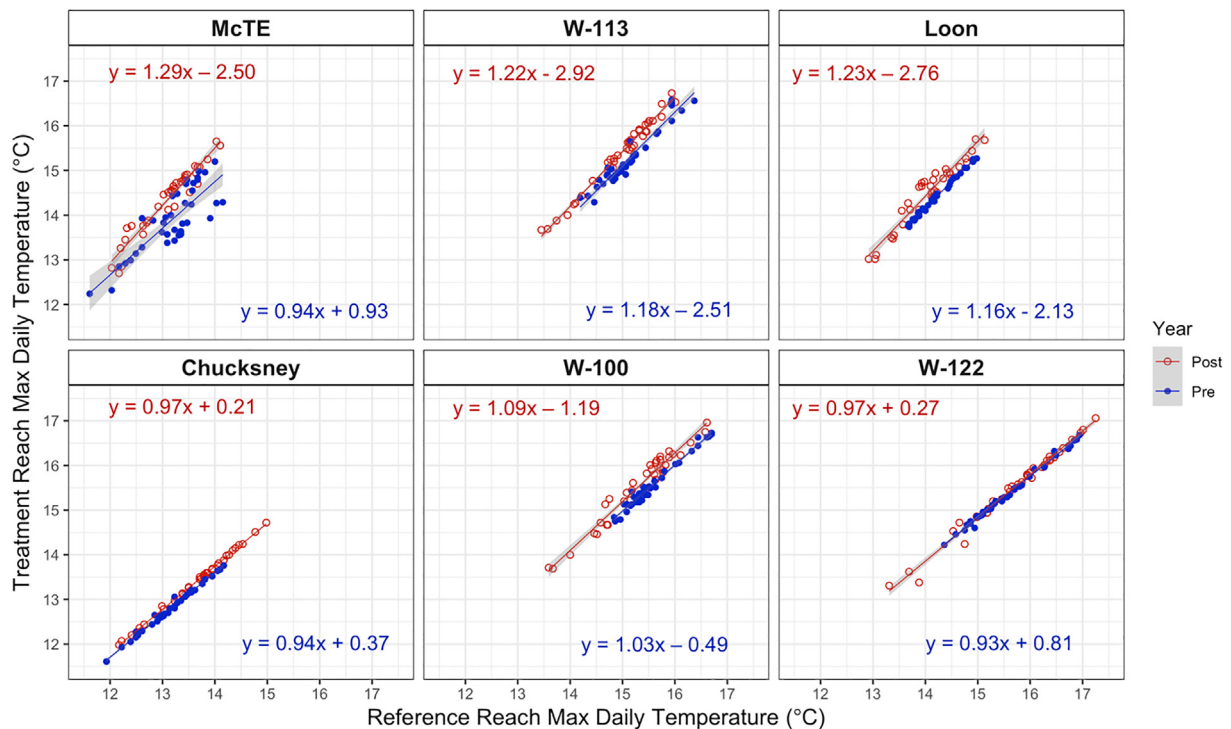


Fig. 4. Maximum daily temperature pre- and post-treatment regressions over the summer 40-day period. Pre- and post-treatment comparisons of regression relationships for the reference reach (x-axis) versus the treatment reach (y-axis) of maximum daily downstream temperatures from July 22nd to August 30th. Sites are ordered by stream size (bankfull width) and 95 percent confidence intervals are in grey.

Water temperature is an important component of water quality regulations associated not only with timber harvest, but all land uses (ODEQ, 2004, USEPA, 2003). The overall mean increase of 0.21 °C in $T_{7\text{DayMax}}$ that we observed is small relative to other studies evaluating changes in riparian canopy cover both from historic and contemporary harvest practices and is below the threshold for Oregon's Cold Water Criteria of 0.3 °C (Brown and Krygier, 1970, Cole and Newton, 2013, Groom et al., 2011b, Johnson and Jones, 2000, Mellina et al., 2002,

Kiffney et al., 2003). Additionally, responses are small compared to the accuracy of the sensors, however a large assessment of these specific sensors (500) showed that 99.6% had error that did not exceed 0.1 °C (Groom et al., 2018). Furthermore, only a few other studies (Hetrick et al., 1998a, Rutherford et al., 2004) have looked at the effects of changes in riparian conditions at the scale of individual gaps as we did here. The limited average temperature response directly below the gap of ~ 0.4 °C is reasonable given our relatively small decreases in canopy

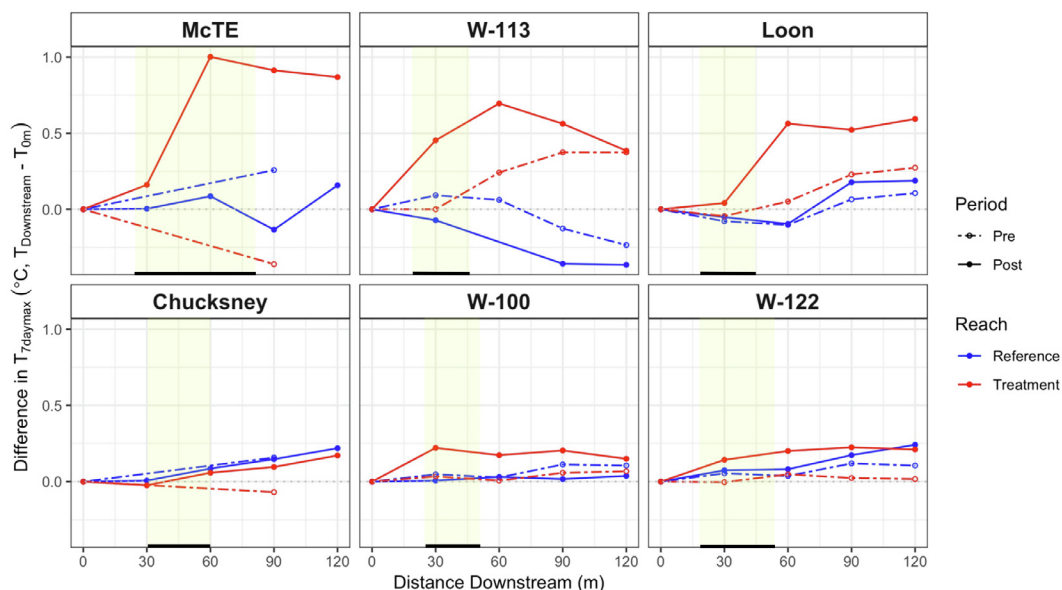


Fig. 5. Longitudinal patterns in $T_{7\text{DayMax}}$ temperatures. Yellow shaded areas with dark lines along the x-axis indicate the extent of the additional light from the canopy gap in the treatment reach only. Pre-treatment and reference reach data show inherent variability in $T_{7\text{DayMax}}$ along reach and between reaches and post-cut treatment reach data show increases due to the gap. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

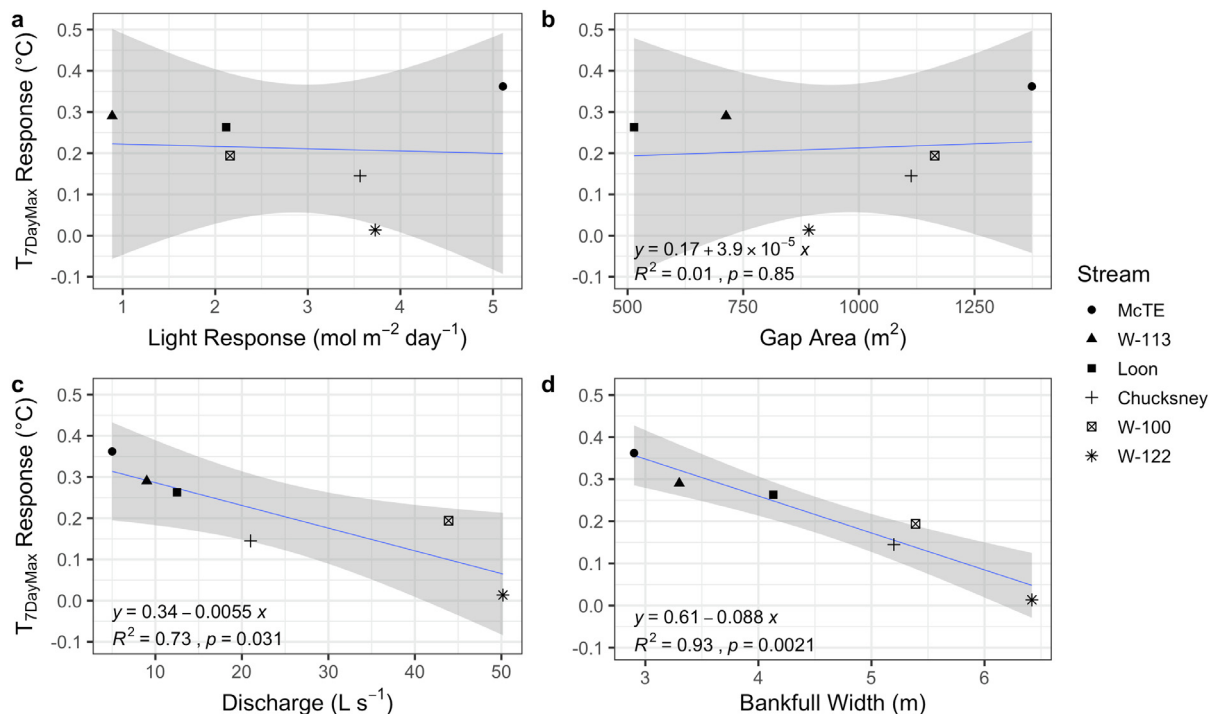


Fig. 6. Explanatory variables of the maximum 7 day moving average maximum temperature BACI responses. Fitted linear regressions and 95% confidence intervals of explanatory variables versus the $T_{7DayMax}$ response show strong relationships with discharge (c) and bankfull width (d), but not with light (a) or gap area (b).

cover as compared to [Hetrick et al. \(1988a\)](#) who observed increases of up to 2 °C in similar, but slightly larger canopy openings (40–70 m) in Alaska during low flow conditions. Additionally, our response is very small relative to the 4–5 °C found in 25 m gaps at Hubbard Brook, NH on a south facing stream, but those streams were smaller than those evaluated here with discharge well below that of the smallest stream in this study ([Burton and Likens, 1973](#)). Maximum temperatures were 4–8 °C higher after canopy removal in [Kiffney et al. \(2003\)](#), [Mellina et al. \(2002\)](#) measured increases of 5 °C, and [Johnson and Jones \(2000\)](#) found up to 7 °C increases, but all of these studies were large canopy removals (100–300 m reaches or larger). Overall, although gap size was not significantly related to our responses, all of these gaps were small relative to most earlier work and the limited temperature responses documented in this study reflect the relatively small change in canopy cover along the riparian zone.

Given the dominant role of solar radiation on stream thermal budgets and the resulting increase in light due to opening the canopy, we expected the gap size and changes in light to explain the magnitudes of the temperature responses. In small headwater streams close to their source, factors such as travel time, hyporheic flow, substrate type, and discharge that buffer stream temperatures can buffer the effects of increased light exposure in large scale harvest operations ([Evans and Petts, 1997](#), [Johnson, 2003](#), [Kasahara, 2003](#), [Moore et al., 2005b](#), [Gomi et al., 2006](#), [Janisch et al., 2012](#)). Therefore in some headwater systems, temperatures may respond rapidly to the loss of riparian shade, while others may remain largely unchanged ([Janisch et al., 2012](#)). Across our six replicates, although the increases in light caused temperature responses, the variability of those responses was not correlated with the magnitude of responses in stream temperature. Site variables relating to stream size (e.g., bankfull width and discharge) rather than solar exposure were better at describing the variability in the magnitude of the temperature increases across the six sites, suggesting temperatures in larger streams are more buffered against changes in light.

Results from the BACI analysis showed that increases in light resulted in increases in temperature as we predicted, but as shown with the $T_{7DayMax}$ regressions, the magnitude of this increase varied with the

size of the stream. We observed larger temperature responses in smaller streams, and minimal to no responses in larger streams. We also showed that proxy metrics quantifying the size of canopy openings (gap size) and stream size (bankfull width) yielded results that were comparable to the metrics of light and discharge that are more closely aligned with the physical processes of interest affecting stream temperatures. In order to further understand the relative influences of additional variables in a multiple regression framework, larger canopy removal studies with more replicates, wider ranges of explanatory variables, and thus statistical power are required.

The analysis of daily maximums and means allowed us to evaluate responses over the full 40-day study period and showed that responses to the gap were small, but statistically significant. Further use of these data using the regression comparison of daily maximums showed increases in slope in the post-treatment year. This increase indicates greater warming within the treatment reach than the reference reach in the summer period after the cut than the summer period before the cut. Therefore on hotter days, treatment reaches warmed slightly more than reference reaches, resulting in greater differences between the treatment and reference reach when compared to cooler days. All sites had steeper slopes in the period after the cut, however, based on this relationship, the thermal responses to the canopy gaps were very small. The average change in slope of 0.12 °C/°C found in this study for daily maximums is substantially smaller than the average change in slope found in a Cascade mountain stream in Oregon 1.05 °C/°C where canopy cover decreased by an average of 20% over streams as compared to the approximately 4% decrease in this study ([Kibler et al., 2013](#)). However, the mean slopes in our study were greater than those observed in the coastal range of Oregon in which no significant temperature increases were observed after contemporary forest harvest that left 15 m buffers along streams ([Bladon et al., 2016](#)). Other recent studies evaluating the effects of contemporary forest management practices that include riparian buffers have also documented limited temperature responses, and when present, the direction and magnitude of these responses tend to be associated riparian buffer width ([Groom et al., 2011b](#), [Janisch et al., 2012](#)). Collectively these studies suggest

that modern riparian forest management practices have been effective in reducing stream temperature responses relative to historical conditions, but light and a combination of a host of factors such as discharge, local climate, underlying geology and hydrology are also influential.

Yearly variation can be an important consideration when assessing stream temperatures. Although we use data from both 2016 and 2017 for the pre-treatment period, our analyses are based on comparisons of differences between reference and treatment reaches within each year across treatment periods (pre-gap versus post-gap). This application of the BACI study design focusing on changes in within-year differences reduces the influence of yearly variation.

In streams, it is important to consider not only local responses, but also how those local responses carry over downstream (Moore et al., 2005a, Garner et al., 2014, Davis et al., 2016). In small headwater streams temperatures often increase longitudinally with distance downstream (Fullerton et al., 2015), but these increases can be highly variable over space and time (Torgersen et al., 1999, Story et al., 2003, Johnson, 2004, Caissie, 2006, Dent et al., 2009). In this study, we documented both variability in the magnitude of local responses and variability in how that local response propagated downstream. Sites where we observed a larger response directly below the gap ($\sim 0.25\text{--}0.8\text{ }^{\circ}\text{C}$) were more likely to have that effect carry over downstream. In some sites, that downstream propagation of temperature remained elevated to our farthest sensor positioned $\sim 80\text{ m}$ downstream of the gap. However, we did not consistently observe evidence of continued downstream warming associated with our canopy gaps due to the small magnitude of increase relative to the high background longitudinal variability. Pre-treatment data before the cut show inherent longitudinal variability and the changes in response to gap formation were small relative to this background variation.

4.1. Management implications

Canopy gaps in riparian forests are small-scale disturbances that will become more common naturally over time as young, dense forests transition to late-successional forests with increased structural complexity (Keeton, 2006, Warren et al., 2016). Given the prevalence of gaps in late-successional forests, the creation of riparian canopy gaps has been suggested as a restoration strategy to increase both aquatic and terrestrial riparian habitat complexity which aligns with the management concept of emulating natural disturbances (Kreutzweiser et al., 2012). Furthermore, in managed landscapes, even when buffers are left, the remaining trees are often not windfirm and may be subject to impact from storm events, creating gaps in the canopy along the stream. Whether considering gaps as an explicit management tool or in the context of unintentional canopy openings along a riparian corridor, localized increases in stream light are an important consideration for forest management along streams. A critical first step to understand the effects of these gaps on aquatic ecosystems is to understand how these small-scale disturbances influence stream temperature. Our results show that the experimental canopy gaps that we evaluated resulted in small temperature increases that were often below the state of Oregon's threshold of a temperature change below $0.3\text{ }^{\circ}\text{C}$. However, our results also demonstrate the importance of site specific factors, which are particularly relevant when considering management that explicitly adds canopy openings in other locations.

Many western states focus management regulations on changes in water temperature, however, in addition to changes, the background (or before treatment) stream temperature is also an important consideration. Oregon, Washington and Idaho water quality standards all include a no detectable change limit of $0.3\text{ }^{\circ}\text{C}$ and Alaska does not permit activities to increase weekly average temperatures by more than $1\text{ }^{\circ}\text{C}$, nor does it allow changes to the amplitude or frequency of normal daily temperature cycles (Alaska Department of Environmental Conservation, 2006, Idaho Department of Environmental Quality, 2006, Washington Department of Ecology, 2003). However, in these

states there are also regulatory thresholds for $T_{7\text{DayMax}}$ that are at or near $16\text{ }^{\circ}\text{C}$ for core cold-water fish rearing habitat, $18\text{ }^{\circ}\text{C}$ for non-core juvenile rearing and migration, and $20\text{ }^{\circ}\text{C}$ for migration of salmon and trout. In this study, the streams never reached $18\text{ }^{\circ}\text{C}$ at any point in any day of the 40 day period, but the $16\text{ }^{\circ}\text{C}$ threshold is pertinent to these systems (Figs. 3 and 4). Three of the streams (McTE, Loon, and Chucksney) are at higher elevations than the other three and never reached $16\text{ }^{\circ}\text{C}$. The other three stream sites are at lower elevations and exceeded the $16\text{ }^{\circ}\text{C}$ threshold, however these streams also exceeded it before the cut and in the reference reaches in both periods (Figs. 3 and 4). The streams at lower elevations have higher discharges, which as discussed earlier, likely can buffer temperature increases from the amount of light due to a canopy gap. However, these sites are the most concerning regarding exceedance of the $16\text{ }^{\circ}\text{C}$ threshold. So although the larger discharge sites in our experiment can buffer these systems from a larger temperature change (relative to the smaller ones in our study), these systems are closer to relevant thresholds (if have not surpassed), so any small change could push them above the threshold regulatory criteria.

Forest management regulations creating riparian buffer zones on the westside of the Cascade range in the Pacific Northwest have resulted in most fish-bearing streams being highly shaded (Kaylor and Warren, 2017). However, when buffers are small or sun angle creates increases in light – even moderate increases in light – temperatures can increase. Studies assessing contemporary management in headwater streams in western Washington have seen temperature responses ranging from 0.2 to $2.4\text{ }^{\circ}\text{C}$ (Pollock et al., 2009, Janisch et al., 2012), however in a study in the Oregon Coast Range no evidence of significant increases in daily maximum temperature or $T_{7\text{DayMax}}$ were found, although this study was at the catchment scale (Bladon et al., 2016). In a large-scale assessment of forest management across the Oregon Coast Range, larger buffer widths, which retained more canopy cover and limited more light, were successful at preventing temperature increases whereas narrower buffers were not (Groom et al., 2011b). These studies illustrate that the amount of light exposure in a stream is important even if a buffer is present, so if light is already elevated and temperature increases are a concern, gaps may not be a viable management option. However, if streams remain shaded with wide buffers, managers may have the option to create complex forest structure using canopy gaps without substantially affecting temperature.

The implications from this work suggest that riparian canopy gaps as a management action are bounded by two primary considerations – stream size and background stream temperature. Larger streams with higher discharges can absorb energy increases with minimal changes in temperature, however, larger streams often have higher background temperatures, so less of an increase is allowable before exceeding thermal thresholds. Baseline temperatures will be important data to consider before implementation as will natural variability within the reach to assess if increases exceed the $0.3\text{ }^{\circ}\text{C}$ criteria for change. If considering larger canopy openings, the size of the cleared areas will also likely drive the magnitude of stream temperature response, but for the range of gap sizes evaluated here, stream size best described the variation in temperature responses amongst sites. Therefore, whether gaps are a result of management or natural processes, the temperature responses and their ecological implications will depend on site specific conditions. If applying riparian canopy gaps as a management practice, site specific variables are required to assess the influence of the treatment on stream temperature. In order for management to restore complex riparian structure, the frequency of treatments should consider the natural occurrence of gaps in old-growth forests of those systems, and should depend on stream size and on background temperature staying below water quality criteria thresholds.

5. Conclusion

While extensive research has assessed impacts of forest management

on stream temperature, experiments that quantify stream temperature responses to restoration alternatives such as canopy gaps are rare. The reach scale temperature increases associated with the experimental gaps in this study were substantially smaller than those seen in larger scale canopy removal treatments or in studies assessing historic timber harvest practices that cut to the stream edge. Although responses were small in magnitude, stream temperature did increase significantly at all six sites due to the gap when considering daily data over the full summer period. Considering maximum summer stream temperatures over a shorter duration of time, in contrast to our expectations, the $T_{7\text{DayMax}}$ temperature response was not related to changes in light or to gap size within the range of gap sized evaluated here. However, stream size was an important determinant of the relative magnitude of stream temperature response in $T_{7\text{DayMax}}$. Overall, our study results suggest that riparian canopy gaps may be a viable a management strategy that can be implemented with minor effects on stream temperature in some systems, but not all. In addition to considering the ecological context and potentially sensitive biota, careful consideration should be given to stream size and background temperatures in planning gap experiments or in assessing potential effects of natural gap formation on stream temperatures.

CRedit authorship contribution statement

Allison Swartz: Conceptualization, Investigation, Formal analysis, Visualization, Data curation, Writing - original draft, Writing - review & editing. **David Roon:** Formal analysis, Writing - review & editing. **Maryanne Reiter:** Funding acquisition, Conceptualization, Writing - review & editing. **Dana Warren:** Conceptualization, Resources, Supervision, Project administration, Funding acquisition, Writing - review & editing.

Declaration of Competing Interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.foreco.2020.118354>.

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