# Headwater stream temperature: interpreting response after logging, with and without riparian buffers, Washington, USA.

Jack E. Janisch[a], Steve M. Wondzell[b], and William J. Ehinger[a]

[a] Environmental Assessment Program, Washington Department of Ecology
Mailstop 47710, Olympia, Washington 98554–7710, USA
Email: jack.janisch@ecy.wa.gov
Email: william.ehinger@ecy.wa.gov

[b] Pacific Northwest Research Station, U.S.D.A. Forest Service, Olympia Forestry Sciences Laboratory,
3625 93rd Avenue S.W., Olympia, Washington 98501, USA
Email: swondzell@fs.fed.us

Corresponding Author: Jack E. Janisch, jack.janisch@ecy.wa.gov
Tel.: +1 360 407 6649
Fax: +1 360 407 6700

**Abstract**

We examined stream temperature response to forest harvest in small (< 9 ha) forested headwater catchments in western Washington, USA over a seven year period (2002-2008). These streams have very low discharge in late summer ($\bar{x} \approx 0.3$ L $s^{-1}$) and many become spatially intermittent. We used a before-after, control-impacted (BACI) study design to contrast the effect of clearcut logging with two riparian buffer designs, a continuous buffer and a patch buffer. We focused on maximum daily temperature throughout July and August, expecting to see large temperature increases in the clearcut streams ($n$ = 5), much smaller increases in the continuously buffered streams ($n$ = 6), with the patch-buffered streams ($n$ = 5) intermediate. Statistical analyses indicated that all treatments resulted in significant ($\alpha$ = 0.05) increases in stream temperature. In the first year after logging, daily maximum temperatures during July and August increased in clearcut catchments by an average of 1.5 $^{o}$C (range 0.2 to 3.6 $^{o}$C), in patch-buffered catchments by 0.6 $^{o}$C (range -0.1 to 1.2 $^{o}$C), and in continuously-buffered catchments by 1.1 $^{o}$C (range 0.0 to 2.8 $^{o}$C). Temperature responses were highly variable within treatments and, contrary to our expectations, stream temperature increases were small and did not follow expected trends among the treatment types. We conducted further analyses in an attempt to identify variables controlling the magnitude of post-harvest treatment responses. These analyses showed that the amount of canopy cover retained in the riparian buffer was not a strong explanatory variable. Instead, spatially intermittent streams with short surface-flowing extent above the monitoring station and usually characterized by coarse-textured streambed sediment tended to be thermally unresponsive. In contrast, streams with longer surface-flowing extent above the monitoring station and streams with substantial stream-adjacent wetlands, both of which were usually characterized by fine-textured streambed sediment, were thermally responsive. Overall, the area of surface water exposed to the ambient environment seemed to best explain our aggregate results. Results from our study suggest that very small headwater streams may be fundamentally different than many larger streams because factors other than shade from the overstory tree canopy can have sufficient influence on stream energy budgets to strongly moderate stream temperatures even following complete removal of the overstory canopy.

**Keywords**: headwater streams, stream temperature, forests, logging, riparian buffers, Pacific

Northwest

## 1. Introduction

Salmon stocks are at significant risk of extinction throughout the Pacific Northwestern United States (Nehlsen et al., 1991). Much remaining spawning and rearing habitat available for salmonids in the Pacific Northwest is concentrated in forested areas subject to logging. Therefore, much attention has focused on how logging and related land-use practices affect salmonid habitat and water quality. Consequently, states have established forest practices rules to minimize logging impacts on forest streams. For example, in Washington State, forest practices rules require retention of riparian buffers along fish-bearing streams to protect streams from temperature increases or loading of fine sediment following logging, and to provide continued sources of large wood to maintain high quality stream habitat for salmonids. Headwater streams (typically $1^{st}$-order, < 1.3 m bankfull width, and < 500 m long) currently receive little protection from potential logging impacts because they are too small, too steep, or too spatially intermittent during summer low flows to support fish.

Headwater streams can influence fish-bearing streams lower in the network in many ways. First, headwater streams export organic and inorganic materials and can subsidize food webs in larger, downstream receiving waters (Freeman et al., 2007; Wipfli et al., 2007) and contribute to processes creating high-quality fish habitat (Reeves et al., 1995, 2003). Second, high-gradient, 1st-order channels and non-channelized headwall seeps can support amphibians (Davic and Welsh, 2004), many species of which are in decline (Kiesecker et al., 2001). Third, cumulative thermal and sediment loading from logged headwater catchments may affect downstream water quality (Beschta and Taylor, 1988; Hostetler, 1991; Poole and Berman, 2001; Alexander et al., 2007).

The direct effects of logging on stream temperatures have mostly been studied on larger streams that were not spatially intermittent during annual low flow. These studies suggest that the sensitivity of streams to temperature increases following logging is related to channel width and discharge (where discharge is, in turn, a function of width, depth, and flow velocity) and to both aspect and elevation (Beschta et al., 1987; Poole and Berman, 2001; Isaak and Hubert, 2001; Moore et al., 2005a). Given that headwater streams on commercial forest land in western Washington are small and shallow, and generally occur at relatively low elevations, the available literature suggests that maximum daily water temperatures during late-summer low-flow periods would be highly sensitive to loss of shade following forest harvests that remove the riparian forest canopy.

Some attributes of small headwater streams, however, contradict these expectations. For example, many headwater streams are spatially intermittent during late-summer low-flow periods. These streams are thus dominated by subsurface flows, and exchange of surface water with the subsurface (hyporheic exchange) could limit heating during the day and cooling at night (Johnson, 2004; Wondzell, 2006). Also, understory vegetation may effectively shade very small streams after removal of the riparian forest canopy and could significantly moderate water temperatures, even if air temperatures in the riparian zone increased following logging (Johnson, 2004). Similarly, vegetative debris (branches with leaves or needles) left after logging might cover small headwater streams and could provide effective shade immediately after logging (Jackson et al., 2001). Finally, headwater reaches, by definition, are locations of groundwater discharge, either from accumulated upslope soil water or deeper groundwater sources. Decreased evapotranspiration after logging could increase inputs of cold groundwater to headwater streams which would also buffer streams from temperature increases.

This study focuses on very small headwater streams in catchments ranging in size from 2 to 9 ha and at the limit of perennial flow. Headwater streams constitute much of the total stream length in any stream network. Consequently, management decisions addressing land-use activities near headwater stream have the potential to influence large areas of land. Management issues related to these streams are important to both state and Federal governments, among others. Thus a large-scale experimental study of forest harvest effects on small headwater streams was undertaken as a collaborative effort among the Washington State Departments of Ecology and Natural Resources and the USDA Forest Service's Pacific Northwest Research Station. The study was conducted on state-owned lands where forest practices rules do not require riparian buffers be retained along non-fish bearing streams – thus allowing the variety of treatments examined in this study.

This study specifically compared stream temperature responses to three different logging treatments. We examined the effect of clearcut logging to see if thermal responses were similar to those previously documented in studies of larger streams. We  contrasted the effect of clearcut logging with two riparian buffer designs – a continuous buffer and a patch buffer – to see if retention of trees in buffer strips along headwater channels would substantially mitigate thermal responses, and to see if thermal responses were sensitive to the design of the riparian buffer. Finally, we examined correlations between post-logging temperature changes and a variety of catchment characteristics to identify those factors that could control thermal responsivity of headwater streams to forest harvest. We focused on maximum daily temperature during the low-flow period in late summer when we expected to see the largest thermal responses. We expected to see large temperature increases in the clearcut streams, small and non-significant increases in the continuously-buffered streams, with the patch-buffered streams intermediate.

## 2. Methods

### *2.1. Study Site Description*

Study sites were located in the temperate forests of western Washington and ranged in elevation from ~ 10 to 400 m. Study catchments were located in two areas (Fig. 1) which spanned a precipitation gradient. The Willapa Hills area, approximately 25 km from the Pacific Ocean, received ~ 210 cm (SD = 40) of precipitation per year (source: COOP station # 456914, Raymond, WA; period of record: 1980 – 2010). The Capitol Forest area, approximately 75 km from the Pacific Ocean, received ~ 130 cm (SD = 8) of precipitation (source: COOP station # 456114, Olympia, WA; period of record: 1949 – 2010) (WRCC, 2010). In both areas, ~ 90% of precipitation fell between October and April. Conversely, summers were dry and typically little precipitation fell during July and August. Annual precipitation during the study ranged from approximately -20% to +10% of long-term averages. Bedrock lithology differed between the two areas. Marine sediments, mixed with some basalts, predominated in the Willapa Hills area whereas basalts of the Crescent Formation predominated at Capitol Forest (Washington Division of Geology and Earth Resources, 2005).

This study had a sample size of 30 catchments, of which two were 2$^{nd}$-order streams and the remainder were 1$^{st}$-order. The valley floors were usually no more than a few meters wide, and in many places, the bankfull channel occupied the full width of the valley floor. Catchment area ranged in size from 1.9 to 8.5 ha and was near the areal limit necessary to sustain perennial flow throughout the year. Discharge in these catchments averaged 0.3 L $s^{-1}$ in July and August, both before and after logging (Alex Foster, pers. comm., USDA Forest Service, Olympia, WA). Many of the streams in our study catchments become spatially intermittent in late summer.

Eight catchments were originally designated as reference catchments and 22 catchments were designated for treatments. However, two of the reference catchments and five of the treated catchments did not provide usable data because they either went dry at the monitoring stations or were dry along the full length of the treated portion of the catchment above the monitoring stations. A sixth treated catchment experienced a data logger malfunction. Thus only six reference and 16 treated catchments provided temperature data usable in our analyses.

Upland forests in the study catchments were dominated by Douglas-fir (*Pseudotsuga menziesii* (Mirbel) Franco) and western hemlock (*Tsuga heterophylla* (Raf.) Sarg.). Within each catchment, the trees were generally even-aged, but tree ages among catchments ranged from 60 to 110 years (Wilk et al., 2010). Conifers in all catchments were approximately 40 m tall (Jeff Ricklefs, pers. comm., WA DNR, Olympia, WA) and the forest canopy was closed, providing dense shade throughout the catchment before logging. Red alder (*Alnus rubra* Bong) was the dominant hardwood species, and was more common in riparian areas.

### *2.2. Study Design*

The study catchments were grouped into "clusters" of three to five catchments that were located close together (Table 1). Each cluster included a reference catchment, and several treatment catchments. Temperature was monitored using a before-after-control-impact (BACI) approach. The pre-logging calibration period lasted 1–2 summers and stream temperature was monitored for two or more summers after logging. Because of the large number of catchments, the logging treatments occurred over an extended period of time, with forest harvest on the first cluster of catchments beginning in September 2003 and the last cluster of catchments harvested in July 2005. All catchments within a cluster were harvested in the same year.

Logging methods were typical of those currently in use in western Washington. Logging roads were constructed prior to logging. Roads were located in upslope or ridge-top locations and only in one catchment did a newly built road intersect a stream channel (near the head of the stream). To protect the headwater channels, the logging prescription required that logging equipment would not be operated closer than 10 m from the stream bank, falling and limbing would be directed away from channels, and logs would not yarded through or across the stream channel. Despite these prescriptions, in a few places, logging equipment did impact stream channels and logging slash (limbs and needles from logged trees) was left in stream channels in some of the clearcut and patch-buffered catchments. Also, streams in the headwater catchments studied here were confluent to larger, fish-bearing streams. The Washington Forest Practices Act requires unharvested buffers along fish-bearing streams. These ranged in width from 50 to 85 m at our study sites and the lower portion of each headwater stream flowed through these buffers. To prevent confusion with the harvest treatments applied in this study, we use the terminology of the Washington State Forest Practices Act and refer to these wider buffers along fish-bearing streams as riparian management zones (RMZ; Fig. 2).

Three forest harvest treatments were examined in this study – continuous buffers, patch buffers, and clearcut harvest (Fig. 2). In all three treatments, the upland portions of the catchments were clearcut harvested so that these treatments differed only in the way the riparian zone was harvested. For continuous buffers, the riparian forest in a 10- to 15-m-wide zone on each side of the stream channel was left unharvested along the full length of the headwater stream. For patch buffers, portions of the riparian forest approximately 50 −110 m long were retained in distinct patches along some portions of the headwater stream channel, with the remaining riparian area clearcut harvested. The patch buffers spanned the full width of the floodplain and extended well away from the stream. Their location and size followed Washington Department of Natural Resources guidelines to protect areas sensitive to disturbance. Because this was an operational study, we did not specify a standard treatment design for either the size or location of patch buffers within a catchment. Consequently, there is substantial variation among the patch treatments. In no case, however, was the full length of a stream channel fully contained within a patch. In clearcut treatments overstory trees were harvested from the catchment, including the entire riparian zone. Prescriptions could not be randomly assigned within clusters. Rather, prescriptions were applied as regulatory constraints and boundaries of the timber-sales allowed (Table 1). This, combined with the uneven number of catchments within each cluster, prevented a perfectly balanced and nested experimental design.

### *2.3. Channel and Catchment Attributes*

The full length of each channel was surveyed with a clinometer and sub-divided into segments wherever longitudinal gradients changed by more than 5%, or where changes occurred in valley-floor confinement. Confinement, calculated as the ratio of the floodplain width to the bankfull channel width, was categorized as confined (≤ 2), moderately confined (2 to 4), and unconfined (≥ 4). Width of the 100 year floodplain was estimated by doubling the depth of the ordinary high water mark, then moving perpendicular to the channel to intersect the bank at this height. Length, gradient, and aspect were recorded for each segment, and within each segment the surface sediment of the streambed was categorized as fine-textured (dominant particle size < 2.5 mm including all clays, silts, and sands) or coarse-textured (dominant particle size > 2.5 mm and including fine gravels, cobbles, and larger particles). Streambed texture was determined from a visual evaluation of the streambed of the active channel within each stream segment. The full length of each stream channel was surveyed two to three times between late June and early October of each year, recording the proportion of the length of each channel segment with surface-flowing water. Using these data, we estimated length of continuously wetted channel above the monitoring station in each catchment on the date of each survey and averaged lengths across survey dates to calculate the average wetted stream length. Surface flow lengths averaged 76.6 m (SE = 20.8) in the calibration year, and > 80% of average yearly changes in flow length during the post-logging period (relative to the calibration year for a given stream) were < ± 10 m. Range of flow lengths for the two study areas the first year after logging were similar. We then calculated the segment length weighted average channel gradient and aspect, and also determined substrate categories, over the wetted stream length above each monitoring station.

The stream-adjacent wetland areas in each headwater catchment were measured in early summer of 2004. We recorded the area of all wetlands that were contiguous with the bankfull channel and showed a visible surface-water connection to the channel. Potential wetlands were first identified using simplified wetland identification and delineation methods (US Army Corps of Engineers, 1987; USDA, 2003; USDA, 2005; USDA, undated) and then further evaluated on the basis of hydrology, soil chroma and texture, and the presence of obligate or facultative wetland vegetation (Janisch et al., 2011). Areas meeting all wetland criteria were delineated and their locations recorded with GPS. Subsequently, the area of each wetland was estimated from a GIS layer built from our field data. Total wetland area was summed along the length of the wetted stream channel above each monitoring station.

Riparian canopy density was quantified twice, once in 2003 prior to logging and again in the first summer after logging. Riparian overstory was photographed using a Nikon 900 CoolPix digital camera with a Nikon FC-E8 fish-eye converter lens[1]. The camera was centered over the channel, at a height of approximately 1.2 m. In contrast to many other studies, relatively few photographs were taken and these were widely spaced. One photo was taken at the temperature sampling location near the bottom of the catchment and another was taken at the head of the channel, at the point of channel formation. Other photos were taken between these two locations, typically spaced 40 to 80 m apart. As a result, each stream is characterized by only two to five photographs. We estimated the percentage of sky blocked by riparian canopy vegetation or by surrounding ridges for the entire $360^{o}$ view above a level horizon within each photograph (Hemiview Canopy Analysis software, v. 2.1, 1999). Hereafter we refer to this as canopy + topographic density (CTD), which is analogous to canopy density of Kelly and Kruger (2005) but includes topography. CTD was summarized in two ways for each catchment. The $CTD_{total}$ was averaged from all photos along the full length of the channel within the catchment. The $CTD_{fe}$ was averaged for a subset of photos along the wetted stream length above each monitoring station.

*2.4. Water Temperature*

Stream temperature was monitored low in the catchments, close to the RMZ boundary (Figure 2). Washington Department of Ecology staff monitored six of the eight clusters using Onset StowAway Tidbit data loggers (accuracy ± 0.2 °C; resolution 0.16 °C) programmed to record every 30 minutes (Table 1). Stream temperature loggers in these catchments were shaded with large pieces of tree bark. At the remaining two clusters, water temperature was monitored by the Pacific Northwest Research Station staff using Maxim Thermochron iButton data loggers (accuracy ± 1.0 °C; resolution 0.5 °C) shaded inside 10-cm long plastic pipe and held to the streambed with large rocks. The iButton data loggers were programmed to record hourly. Late summer discharge was very low in all the catchments and stream water was usually less than 3 cm deep at our monitoring sites. Consequently, temperature loggers were placed in areas with the greatest flow velocity and the deepest water, and even these locations required frequent maintenance to ensure data loggers remained submerged. Once

[1] The use of trade or firm names in this publication is for reader information and does not imply endorsement by the US Department of Agriculture of any product or service.

locations were established, stream temperature loggers were kept in the same locations for the remainder of the study.

We employed a rigorous quality assessment and quality control protocol to identify erroneous temperature data using a post-deployment accuracy check and field notes for the six clusters using Tidbit data loggers. The temperature calibration of the Tidbit data loggers was checked in both an ice bath and a warm water bath. Departures from factory specifications triggered a data review to identify and exclude erroneous data. Malfunctioning data loggers were returned to the manufacturer for data retrieval and these data were then reviewed for usability. We also used field notes and temperature plots to identify periods when the stream was dry or when data loggers were exposed to air.. Data from the affected time periods for these loggers was excluded from analysis.

Headwater catchments in the two clusters where iButton data loggers were used to collect temperature data were all adjacent to each other. Because of the close proximity of the catchments, temperature data were compared among the catchments to identify any time periods when temperature trends among catchments were dramatically different, or periods when temperature data loggers malfunctioned. No obviously erroneous data were found so the full data records were used in the analysis.

### *2.5. Statistical Analysis*

We analyzed post-treatment changes in July through August daily maximum temperatures. Treatment catchments were paired with reference catchments within each cluster.  However, two of the eight reference catchments dried completely by late summer of the calibration year. In these cases, we conducted our analyses by substituting the nearest reference catchment from the closest cluster within the Willapa Hills or Capitol Forest study areas. Our analyses followed the methods developed by Watson et al. (2001) and Gomi et al. (2006).

We developed regression relationships between temperatures measured in the treatment ($T_{predicted}$) and corresponding reference ($T_{ref}$) catchments of the general form:

(1) $$T_{predicted} = \beta_0 + \beta_1 T_{ref} + \beta_2 \sin(2\pi j/t) + \beta_3 \cos(2\pi j/t) + \epsilon,$$

where $j$ = day of year, $t$ = 365.25 (number of days per year), and $\beta_0 - \beta_3$ are regression coefficients. Sine and cosine terms at a daily time step were included to model seasonality and retained even if not significant. The error term, $\epsilon$ (i. e., residuals), was later modeled by an autoregressive generalized least squares (GLS) procedure to isolate the random error and temperature response components (see below). This procedure was repeated for both water and air temperature.

Regression equations were used to predict expected daily maximum temperatures of treated catchments during July and August of each year during the post-logging period. Predicted daily maximum temperatures were subtracted from observed daily maximum temperature to calculate the change in stream temperature resulting from the logging treatment (i.e., $T_{obs} - T_{pred}$), hereafter referred to as the temperature response. When significant positive auto-correlation was detected, an adjustment was applied using coefficients from an iterative auto-regression / GLS procedure (SAS v. 9.2, SAS Institute Inc., Cary, NC, USA) of the general form:

(2) $$\hat{u}_t = (y_t - \hat{y}_t) - \hat{w}_1 (y_{t-1} - \hat{y}_{t-1}) - \hat{w}_2 (y_{t-2} - \hat{y}_{t-2}) - \ldots \hat{w}_k (y_{t-k} - \hat{y}_{t-k}),$$

where the quantity $\hat{u}$ *is* defined as the estimated daily random disturbance following Gomi et al. (2006), $y$ is observed temperature, and $\hat{y}$ is predicted temperature on day $t$. Lag $i$ autocorrelation coefficients, $\hat{w}$, were estimated by the GLS procedure. Significant positive residual autocorrelation was observed in approximately half of the watersheds (Lag 0, $n$ = 9; Lag 1, $n$ = 6; Lag 2, $n$ = 1).

We used the same autocorrelation analysis to examine pairs of reference catchments to evaluate the assumption of temperature stationarity implicit in BACI study designs, i.e., that the relation between catchments did not change from the calibration year versus the post-treatment years. This analysis compared reference catchments where temperatures were monitored with iButtons with catchments where temperatures were measured with Tidbit temperature loggers. The different loggers have different accuracy and resolution and may affect the slope and intercepts of regression lines fit to the data with subsequent effect on the confidence intervals measured in this analysis. The largest standard deviation (SD) among all reference pairs calculated during the calibration-year was used as a guide to meaningful stream temperature change. We followed Gomi et al. (2006), calculating a 95% confidence interval for the daily random disturbance as 0.00 ± 1.96*SD of the single largest SD of all pairwise comparisons among reference catchments in the calibration year. If the daily random disturbances of the pairs of reference catchments exceed the 95% prediction interval in the post-calibration years the assumption of stationarity would be violated, calling into question the results observed in our treated catchments. Similarly, if the daily random disturbances in the treated catchments exceed the 95% prediction interval in the post-treatment years, there is likely to be a significant treatment response for that catchment.

We used ANOVA to test for an overall treatment response. However, this analysis was complicated by a small sample size, an unbalanced sampling design, and non-random assignment of treatments. We had a large number of treated and reference catchments included in a complex study design where individual catchments were grouped into clusters, but the resulting statistical blocks had small sample sizes, and were unbalanced because not all treatments were replicated within every cluster and some clusters had duplicates of some treatments (e.g., the Split Rue cluster included one reference, one continuously buffered, and two clearcut catchments, but did not include a patch-buffered catchment). Also, the time series data were discontinuous because we focused on only the months of July and August over a three-year period. Finally, this was an “operational study” in that the harvest treatments were applied just like any other commercial forest harvest currently practiced in the State of Washington on state-owned lands. Consequently, treatments could not always be applied randomly (i.e., reference catchments were usually located so as to minimize the amount of road construction). Given these issues, fitting a statistical model to the data proved problematic, with many reasonable models failing to converge to a solution. We report results from a simple ANOVA model that had the lowest AIC of all the models examined. That model only included fixed effects for treatment, years since treatment, and day of year, accounted for repeated measurements across days within each catchment, and used an autoregressive term to account for Lag 1 autocorrelation in the data (Proc MIXED, SAS v. 9.2, SAS Institute Inc., Cary, NC, USA).

We conducted a correlation analysis between the post-logging change in temperature and the descriptive variables on a subset of catchments to examine possible factors that might control post-logging thermal responses. Descriptive variables were only measured in the catchments monitored by Washington Department of Ecology, so this analysis was only conducted on that subset of six clusters (Table 1) which include a total of 15 treated catchments. However, of these, five went dry during the summer and a data logger malfunctioned at a sixth catchment so data from only nine catchments were available for this portion of the analyses. Due to small sample size all correlations are reported as uncorrected coefficients. Also, this correlation analysis was only conducted for the first year after logging because the greatest stream temperature response was expected immediately after logging. Specifically, we examined relationships between the descriptive variables (elevation, catchment area, aspect, gradient, surface flow, CTD, depth, and wetland area) and the post-treatment change in stream water temperatures ($T_{obs} – T_{pred}$). For variables showing significant correlations, we used regression analysis to further examine their relation to post-harvest changes in stream temperature. Streambed sediment texture was categorized as either fine or coarse, so this variable could not be used in the correlation analysis. However, we performed separate regression analyses for the data from catchments with fine- vs. coarse-textured streambeds.

## 3. Results

### *3.1. Treatment Responses*

#### *3.1.1. Canopy and topographic density (CTD)*

The $CTD_{total}$ averaged 94% over the stream channels before logging and did not differ significantly between reference and treatment catchments. The $CTD_{total}$ in the reference catchments ($\bar{x}$ = 95.0%, SE = 0.4) did not change substantially after logging ($\bar{x}$ = 93.5%, SE = 0.3). In contrast, $CTD_{total}$ decreased in all of the treated catchments after logging. The $CTD_{total}$ over the stream channels in the clearcut catchments ($\bar{x}$ = 53%, SE = 7.4) and in the patch-buffered treatments ($\bar{x}$ = 76%, SE = 5.1) were both significantly lower than in the reference catchments. The $CTD_{total}$ over the stream channels in the continuously buffered treatments ($\bar{x}$ = 86%, SE = 1.7) was not significantly different from the reference catchments.

#### *3.1.2. Stream temperature*

Our study was relatively unique in that we had multiple reference catchments in reasonably close proximity which allowed us to test the assumption of stationarity that is implicit in all BACI designs. We compared temperature changes ($T_{obs} – T_{pred}$) between pairs of reference catchments in the post-logging period. Because there is no a priori way to specify which reference catchment will serve as the dependent variable and which will serve as the independent variable in these paired comparisons, each regression analysis for pairs of reference catchments was conducted twice. For example, we first used the ROTT reference to predict the expected temperature in the TAGS reference catchment, and then used the TAGS reference to predict the expected temperature in the ROTT reference catchment. Consequently, the overall mean calculated from all possible reference pairs is very close to 0.0 °C. However, one comparison will result in a positive temperature change and the other will result in a negative temperature change. Averaging these shows that the mean temperature change for the reference catchments in the post-logging period ranged from -0.41 to 0.47 $^{o}$C in the first post-logging year and were smaller in subsequent years. Similarly, the SDs of the changes in stream temperature, averaged over all reference pairs, were largest in the first post-calibration year, and even in that year, most of the daily random disturbances fell well within the 95% confidence interval calculated from the calibration year (Fig. 3), suggesting that the assumption of stationarity was met in our analyses. These results set practical bounds on the magnitude of temperature changes that can reliably indicate a treatment response in our BACI-designed study.

Our overall test for post-treatment temperature changes suggested that treatments ($p$ = 0.0019), the number of years post-treatment ($p$ = 0.0090), and the day of the year ($p$ = 0.0007) were all significant main effects explaining the observed change in temperature. The statistical model fit to the data showed that the relation between treatments and stream temperature responses was somewhat complex, varying across years and with date within a year (Fig. 4). In general, temperature changes were greatest in the clearcut catchments, smallest in the patch buffered catchments and intermediate in the continuously buffered catchments (Fig. 4; Table 2). The statistical model also suggested that the temperature changes for all treatments were largest in the first post-treatment year and declined in each subsequent year (Table 2). Further, temperature changes were largest in early July and decreased over the sampling period to a minimum in late August.

The statistical model showed that temperature changes in clearcut treatments remained significantly greater than zero ($\alpha$ > 0.05) in all three post-treatment years (Fig. 4). For continuously buffered catchments, temperature changes were significantly greater than zero ($\alpha$ > 0.05) in the first two post-treatment years. In the third post-treatment year, the magnitude of the temperature change estimated from the statistical model was not significantly different from zero after Julian day 228 (~ 15

August). For the patch-buffered treatments, temperature changes were significantly greater than zero ($\alpha > 0.05$) in the first post-treatment year. In the second and third post-treatment years, the magnitude of the temperature change in the patch-buffered catchments estimated from the statistical model was only significantly different from zero in the early summer – becoming insignificant on day 231 in year 2 (~ 19 August) and on day 202 in year 3 (~ 20 July) (Fig. 4). There was one notable outlier among the patch-buffered catchments where stream temperature increased by approximately 4 °C in year 3 following a debris flow that scoured the channel to bedrock. The data from this site for this year were excluded from the analysis.

There was high variability among catchments within each treatment group. The largest change in maximum daily temperature averaged over July and August in the first year after logging was 3.6 °C in one clearcut catchment, 2.8 °C in one continuously-buffered catchment, but only 1.2 °C in one patch-buffered catchment (Fig. 3). In contrast, one or more catchments within each treatment group showed little or no change in the average maximum daily temperature in the first year after logging ($\bar{x}$ = 0.2 °C, - 0.02 °C, - 0.05 °C for minimum temperature changes observed in clearcut, continuous, and patch treatments, respectively).

*3.1.3. Correlations between temperature responses and stream and catchment variables*

Some landscape variables were significantly correlated with post-logging change in stream temperature (Table 3). Wetland area (0.96, $p < 0.01$) and length of surface flow (0.67, $p = 0.05$) were both strongly correlated with post-logging temperature change. Aspect was also significant (0.80, $p = 0.01$) but, surprisingly, streams with the greatest temperature increases had northerly aspects. Also surprisingly, CTD was only weakly (and non-significantly) correlated with the post-logging changes in stream temperature. Analyzing just the thermally responsive catchments showed that only two variables, wetland area (0.96, $p < 0.01$) and wetted stream length (0.81, $p = 0.05$), were highly correlated to post-logging temperature changes. Regression analyses of these variables showed that streams with coarse-textured substrates responded quite differently from streams with fine-textured substrates (Fig. 5). Coarse-textured streams all had wetted stream lengths of 85 to 90 meters and showed no post-logging increase in temperature. In contrast, fine-textured streams of similar length showed post-logging temperature increases of approximately 1.0 °C (Fig. 5a). Coarse-textured streams also lacked riparian wetlands (Fig. 5b).

## 4. Discussion

Stream temperature generally increased after logging, which followed our expectations based on the results of many other studies of larger streams (Moore et al., 2005b). Further, the temperature increases were largest in the clearcut treatments and smaller in the buffered treatments which would be consistent with many other studies that have found riparian buffers to be effective at limiting temperature increases following forest harvest (Brown and Krygier, 1970; Castelle and Johnson, 2000). Interpreting the temperature response of headwater streams to logging is not always straightforward, however. Temperatures of small streams can vary spatially and show mixed warming and cooling patterns, even when well shaded (Dent et al., 2008). Hypothesized sources of variation in small stream temperature include interaction with groundwater (Dent et al, 2008) and the influence of stream surface area and hyporheic exchange (Pollock et al., 2009). Pollock et al. (2009) in particular stressed that factors in addition to the condition of riparian canopy may affect stream temperature.

Still, several results from this study proved contrary to our expectations. First, stream temperature changes after logging were relatively small. The average daily maximum temperature in the clearcut catchments increased by only 1.5 °C in the first year after logging (Table 2), and the greatest temperature increase observed in a single catchment was only 3.6 °C (Fig. 3a). Compare these results to those of Gomi et al. (2006), who found that post-harvest temperature increases in clearcut catchments ranged from 2 to 8 °C. Second, the magnitude of temperature increases after logging in the buffered treatments did not follow the trend expected given the changes in canopy density (CTD) resulting from the harvest treatments. We expected that the temperature changes would be largest in the clearcut catchments, smallest in the continuously-buffered catchments and intermediate in the patch-buffered catchments. The observed responses were quite different. The magnitude of response in the continuously buffered catchments was larger than that observed in the patch-buffered catchments in all three post-treatment years. Further, the patch-buffered catchments had the smallest post-logging change in temperature (Table 2). Third, correlations between post-logging stream temperature changes and CTD (Table 3) were not significant. Finally, post-logging temperature changes were highly variable within treatment groups. Some catchments in both the clearcut and continuously-buffered treatments had large post-logging increases in temperature while other catchments showed little if any response (Fig. 3a).

The unexpected results combined with the high variability among catchments within treatment groups begged the question: What factors were controlling the thermal responsivity of streams after forest harvest removed all or part of the overstory canopy? Of the variables measured (Table 3), three showed high and statistically significant correlations: aspect, length of wetted channel, and riparian wetland area. This analysis suggested that aspect was significant and that streams with the greatest post-logging temperature increases tended to have northerly aspects, contrary to the predictions from solar loading models (Bartholow, 2000; Anderson et al., 2004).

The correlations with the length of wetted channel and with wetland area suggest that stream temperature after logging increased in direct proportion to the area of exposed water surface area and saturated soils upstream of monitoring stations. Length of continuously wetted stream channel above the stream-temperature monitoring stations ranged from as little as 34 m to a maximum of 203 m (Fig. 5a). There was a similarly wide range in wetland area among the catchments, from catchments entirely lacking riparian wetlands to catchments with more than 150 $m^2$ of wetlands (Fig. 5b). We cannot easily isolate the effects of the length of wetted channel from the effects of wetland area. Certainly, the surface area of some streams (~ 20 $m^2$) is at the low end of observed range in wetland areas so that a single tiny wetland can double the surface area of a 1$^{st}$-order stream. However, we do not know how well connected the riparian wetlands are to the stream. Few of the observed wetlands originated from obvious side-slope seeps (Janisch et al., 2011). Instead, most of the observed wetlands were located in the valley floor and intersected by the stream channel. In any case, both factors appear to have strong effects on the sensitivity of tiny headwater streams to forest harvest.

The area of exposed surface water above the stream temperature monitoring station may explain the lack of post-logging temperature increases observed in the patch-buffered catchments because this group of catchments had the smallest wetland areas of all the treated catchments. Similarly, the area of exposed surface water may also explain why the two warmest streams had northerly aspects. These two streams had the largest wetland areas (~ 150 $m^2$) and among the longest lengths of continuously wetted stream channel observed among catchments in our study.

Additionally, our results imply substrate may be an important determinant of thermal responsivity. In general, thermally unresponsive streams occurred on coarse-textured substrates, whereas thermally responsive streams occurred on fine-textured substrates (Figs. 5a & b). This result seems reasonable given the potential influence of stream–groundwater interactions on stream temperature (Brown, 1969; Johnson, 2004; Moore and Wondzell, 2005), that is, surface sediment textures provide a loose index of likely saturated hydraulic conductivity ($K$) and therefore the likelihood of hyporheic exchange. Hyporheic exchange (i.e., the flow of stream water into the streambed, through the sub-surface, and subsequently returning to the stream) may substantially buffer stream temperature (Moore et al., 2005b). Streambeds composed of fine-textured sediment likely have low $K$ which would limit hyporheic exchange and thus low potential for stream−groundwater interactions to buffer stream heating.

We suspect that fine substrates would promote formation of tiny riparian wetlands, thus increasing the surface area of channel-associated water exposed to heating processes. Conversely, coarse-textured sediments would allow greater sub-surface flow rates so that streams on coarse substrates would be more likely to be intermittent and thus have shorter lengths of continuously-wetted stream channel above our temperature monitoring stations.

The correlation analyses reported here are not sufficient to prove a causal relationship. They are consistent with the expected behavior of streams, but these variables may be correlated with other factors that actually control post-logging thermal response. For example, many studies have shown that the loss of transpiring tree canopies to forest harvest reduces transpirational water losses resulting in greater low-flow discharge. Discharge is one of the factors known to influence stream heating. Similarly, groundwater inputs, especially if they are located near the temperature monitoring station, could significantly reduce post-logging increases in daily maximum temperatures in late summer. We cannot discount this possibility. However, stream discharges were extremely low and streams were very shallow so that groundwater influences would likely be small.

A number of other factors might also help account for the small temperature increases observed in this study. For example, we noted that logging slash accumulation varied substantially among catchments, and along channels within a single catchment, from none to ~ 1 m deep accumulations of limbs and foliage. This slash was largely confined to clearcut streams where logging occurred along the full length of the stream channel. Thus, logging slash may have shaded the stream channel in the first years after logging (Jackson et al., 2001), especially in clearcut streams where residual slash cover was greatest. However, we did not quantify the amount of slash and do not know the degree to which logging slash might have shaded the stream. Windthrow often confounds experimental efforts to examine the effect of riparian buffers in mitigating stream temperature increases after logging. We did not quantify windthrow, but our observations showed that windthrow occurred primarily in catchments in the Willapa Hills study area in the second year after logging and later. The amount of windthrow increased gradually until a severe windstorm in 2007 when widespread and extensive windthrow occurred in our buffered treatments. Much of our analysis focuses on the temperature responses in the first year after logging, before substantial windthrow had occurred.

We measured canopy + topographic density (CTD) in the full $360^{o}$ view above the stream showing in our fish-eye photographs. We did not measure canopy density along the sun's path, therefore we do not have a direct measure of shade. Also, the camera was located 1.2 m above the stream surface and thus could not measure shade from logging slash or low-growing vegetation. Further, because we include topographic features that block the view to a level horizon, CTD is quite large in the clearcut treatments. If we had measured canopy density along the solar path from the immediate stream surface we may have observed a much stronger correlation between post-logging increases in stream temperature and canopy density. Still, our measure of CTD is directly related to shade, and while not significant, it showed weak negative correlation to post-logging stream temperature changes – that is, as canopy density decreased, the magnitude of temperature response increased.

Overall, headwater stream temperatures in late summer increased after logging, but warming patterns were complex and not simply related to riparian canopy retention treatments. All the headwater streams studied were shallow with very low discharge in late summer, but only some were thermally responsive to logging. We suggest that several factors determine the thermal responsivity of headwater streams. Especially important is surface area of the stream and associated wetlands. Substrate texture also appears to be important, likely because it indicates strength of stream-groundwater interactions which can substantially buffer the thermal regimes of small streams. In total, we conclude that headwater streams differ from larger, spatially continuous, temporally perennial, fish-bearing streams. Confirming the results we observed, developing management tools to predict the thermal responsivity of headwater streams, and developing management practices sensitive to these differences, will require substantial additional work.

**Acknowledgements**

This research was funded by grants from the Washington Department of Natural Resources. Additional funding was provided by the USDA Forest Service Pacific Northwest Research Station and the Washington Department of Ecology. The temperature analysis was part of the Riparian Ecosystem Management Study, an integrated research project conducted by the Washington Department of Natural Resources, the USDA Forest Service's Pacific Northwest Research Station, and the Washington Department of Ecology. We thank Pat Cunningham from the Pacific Northwest Research Station for statistical assistance. Shannon Cleason, Christopher Clinton, Brian Engeness, Stephanie Estrella, Alex Foster, Tiffany Foster, Jeremy Graham, Nicholas Grant, Chad Hill, Kevin Kennedy, Jordan Martinez, Jeremiah McMahan, Charlotte Milling, Brenda Nipp, Christen Noble, Tanya Roberts, Matias Rudback, Crystal Vancho, Troy Warnick, and Elizabeth Werner assisted with data collection and management. Steve Barrett, Washington Department of Ecology, wrote code to summarize daily temperatures. Comments of two anonymous referees further improved earlier drafts.

**References**

Alexander, R.B., Boyer, E.W., Smith, R.A., Schwarz, G.E., Moore, R.B., 2007. The role of headwater streams in downstream water quality. J. Am. Water Resour. Assoc. 43, 41–59. doi: 10.1111/j.1752-1688.2007.00005.x.

Anderson, R.J., Bledsoe, B.P., Hession, W.C., 2004. Width of streams and rivers in response to vegetation, bank material, and other factors. J. Am. Water Resour. Assoc. 40, 1159–1172.

Bartholow, J.M., 2000. Estimating cumulative effects of clearcutting on stream temperatures. Rivers **7**, 284–297.

Beschta, R.L., Taylor, R.L., 1988. Stream temperature increases and land use in a forested Oregon watershed. Water Resour. Bull. 24, 19–25.

Beschta, R.L., Bilby, R.E., Brown, G.W., Holtby, L.B., Hofstra, T.D., 1987. Stream temperature and aquatic habitat: Fisheries and forestry interactions, in: Salo, E.O., Cundy, T.W. (Eds.), Streamside Management: Forestry and Fishery Interactions. University of Washington, Institute of Forest Resources Contribution No. 57, Seattle, WA, pp. 191–232.

Brown, G.W., 1969. Predicting temperatures of small streams. Water Resour. Res. 5, 68–75. doi: 10.1029/WR005i001p00068.

Brown, G.W., Krygier, J.T., 1970. Effects of clear-cutting on stream temperature. Water Resour. Res. 6, 1133–1139. doi: 10.1029/WR006i004p01133.

Castelle, A.J., Johnson, A.W., 2000. Riparian vegetation effectiveness. Tech. Bull. No. 799. National Council for Air and Stream Improvement, Research Triangle Park, N.C.

Davic, R.D., Welsh, H.H., Jr., 2004. On the ecological roles of salamanders. Annu. Rev. Ecol. Evol. Syst. 35, 405–434. doi: 10.1146/annurev.ecolsys.35.112202.130116.

Dent, L., Vick, D., Abraham, K., Schoenholtz, S., Johnson, S., 2008. Summer temperature patterns in headwater streams of the Oregon Coast Range. J. Am. Water Resour. Assoc. 44, 803–813. doi: 10.1111/j.1752-1688.2008.00204.x.

Freeman, M.C., Pringle, C.M., Jackson, C.R., 2007. Hydrologic connectivity and the contribution of stream headwaters to ecological integrity at regional scales. J. Am. Water Resour. Assoc. 43, 5–14. doi: 10.1111/j.1752-1688.2007.00002.x.

Gomi, T., Moore, R.D., Dhakal, A.S., 2006. Headwater stream temperature response to clear-cut harvesting with different riparian treatments, coastal British Columbia, Canada. Water Resour. Res. 42, W08437. doi: 10.1029/2005WR004162.

Hemiview Canopy Analysis®, v. 2.1 SR1. 1999. Delta–T Devices, Ltd. United Kingdom.

Hostetler, S.W., 1991. Analysis and modeling of long-term stream temperatures on the Steamboat Creek basin, Oregon: Implications for land use and fish habitat. Water Resour. Bull. 27, 637–647. doi: 10.1111/j.1752-1688.1991.tb01465.x.

Isaak, D.J., Hubert, W.A., 2001. A hypothesis about factors that affect maximum summer stream temperatures across montane landscapes. J. Am. Water Resour. Assoc. 37, 351–366.

Jackson, C.R., Sturm, C.A., Ward, J.M., 2001. Timber harvest impacts on small headwater stream channels in the Coast Ranges of Washington. J. Am. Water Resour. Assoc. 37, 1533–1549. doi: 10.1111/j.1752-1688.2001.tb03658.x.

Janisch, J.E., Foster, A.D., Ehinger, W.J., 2011. Characteristics of small headwater wetlands in second-growth forests of Washington, USA. For. Ecol. Manage. 261, 1265–1274.

Johnson, S.L., 2004. Factors influencing stream temperatures in small streams: substrate effects and a shading experiment. Can. J. Fish. Aquat. Sci. 61, 913–923.

Kelley, C.E., Krueger, W.C., 2005. Canopy cover and shade determinations in riparian zones. J. Am. Water Resour. Assoc. 41, 37–46.

Kiesecker, J.M., Blaustein, A.R., Belden, L.K., 2001. Complex causes of amphibian population declines. Nature 410, 681–684.

Moore, R.D., Wondzell, S.M., 2005. Physical hydrology and the effects of forest harvesting in the Pacific Northwest: a review. J. Am. Water Resour. Assoc. 41, 763–784.

Moore, R.D., Spittlehouse, D.L., Story, A., 2005a. Riparian microclimate and stream temperature response to forest harvesting: a review. J. Am. Water Resour. Assoc. 41, 813–834.

Moore, R.D., Sutherland, P., Gomi, T., Dhakal, A.S., 2005b. Thermal regime of a headwater stream within a clear-cut, coastal British Columbia, Canada. Hydrol. Processes 19, 2591–2608. doi: 10.1002/hyp.5733.

Nehlsen, W., Williams, J.E., Lichatowich, J.A., 1991. Pacific salmon at the crossroads: Stocks at risk from California, Oregon, Idaho and Washington. Fisheries 16, 4–21.

Pollock, M.M., Beechie, T.J., Liermann, M., Bigley, R.E., 2009. Stream temperature relationships to forest harvest in Western Washington. J. Am. Water Resour. Assoc. 45, 141–156.

Poole, G.C., Berman, C.H., 2001. An ecological perspective on in-stream temperature: natural heat dynamics and mechanisms of human-caused thermal degradation. Environ. Manage. 27, 787–802. doi: 10.1007/s002670010188.

Reeves, G.H., Benda, L.E., Burnett, K.M., Bisson, P.A., Sedell, J.R., 1995. A disturbance-based ecosystem approach to maintaining and restoring freshwater habitats of evolutionarily significant units of anadromous salmonids in the Pacific Northwest. Am. Fish. Soc. Symp. 17, 334–349.

Reeves, G.H., Burnett, K.M., McGarry, E.V., 2003. Sources of large wood in the main stem of a fourth-order watershed in coastal Oregon. Can. J. For. Res. 33, 1363–1370. doi: 10.1139/X03-095.

US Army Corps of Engineers, 1987. Wetlands Research Program Technical Report Y–87–1. US Army Corps of Engineers Waterways Experiment Station. Vicksburg, MS. Revised 1997, Washington State wetlands identification manual. Washington Department of Ecology Publication No. 96–94.

US Department of Agriculture, Undated. Western wetland flora: Field office guide to plant species. USDA Soil Conservation Service, West National Technical Center. Portland, Oregon. Jamestown, ND: Northern Prairie Wildlife Research Center Online. http://www.npwrc.usgs.gov/index.htm (Version 16JUL1997).

US Department of Agriculture, 2005. Soil Survey Geographic Data Base (SSURGO). Natural Resources Conservation Service.

US Department of Agriculture, 2003. Field indicators of hydric soils in the United States, version 5.01. Hurt, G.W., Whited, P. M., Pringle, R.F. (Eds.). USDA Natural Resources Conservation Service in cooperation with the National Technical Committee for Hydric Soils, Fort Worth, TX.

Washington Division of Geology and Earth Sciences, 2005. Digital 1:100,000-scale geology of Washington State. Washington Department of Natural Resources, Olympia, WA.

Watson, F., Vertessy, R., McMahon, R.T., Rhodes, B., Watson, I., 2001. Improved methods to assess water yield changes from paired-catchment studies: application to the Maroondah catchments. For. Ecol. Manage. 143, 189–204.

Western Regional Climate Center [WRCC]. 2010. Historical climate summaries, Western United States: weather stations, 1949–2010. Reno, NV. U.S. A. http://www.wrcc.dri.edu

Wilk, R.J., Raphael, M.G., Nations, C.S., Ricklefs, J.D., 2010. Initial response of small ground-dwelling mammals to forest alternative buffers along headwater streams in the Washington Coast Range, USA. For. Ecol. Manage. 260, 1567–1578.

Wipfli, M.S., Richardson, J.S., Naiman, R.J., 2007. Ecological linkages between headwaters and downstream ecosystems: transport of organic matter, invertebrates, and wood down headwater channels. J. Am. Water Resour. Assoc. 43, 72–85. doi: 10.1111/j.1752-1688.2007.00007.x.

Wondzell, S.M., 2006. Effect of morphology and discharge on hyporheic exchange flows in two small streams in the Cascade Mountains of Oregon, USA. Hydrol. Processes 20, 267–287. doi: 10.1002/hyp.5902.

**List of Figures**

continuously-wetted channel above the stream temperature monitoring stations.