AN ABSTRACT OF THE THESIS OF

Jonah Nicholas for the degree of Master of Science in Sustainable Forest Management presented on November 29, 2022.

Title: Summer Low Flow Response to Timber Harvest and Riparian Treatments in Forested Headwater Streams of Coastal Northern California

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Kevin D. Bladon Catalina Segura

Headwater streams are individually small yet collectively critical waterways that make up the upper most portions of a stream network. These streams are surrounded by ecologically important riparian zones which are influenced by both the stream and the surrounding terrestrial ecosystem. Headwater streams are often closely associated with their surrounding riparian zones which are transitional areas between terrestrial and stream ecosystems, and these zones provide key habitat for aquatic and terrestrial organisms, especially during the dry season and severe weather events. Human activity in these headwater areas can have large impacts on the adjacent stream and downstream areas. To protect the vital ecosystem services riparian areas and headwater streams provide, many regions regulate riparian buffers as part of timber harvests. Many jurisdictions in the western United States have increased riparian buffer protections in recent years, mostly with limited research about the effects of the new implemented riparian buffers. This study assessed the effects of timber harvests and different riparian buffer treatments on two aspects of summer streamflow, total daily streamflow and diel streamflow.

We tested the effects of three riparian buffer treatments on low flow headwater streamflow in coastal Northern California. We selected 2 currently used riparian treatment and 1 previously used riparian treatment and compared these treatments to two reference streams. These treatments were a part of clearcut timber harvests in industrial timberland. We measured the streamflow and weather conditions for one pre-harvest year and two post-harvest years. During the two post-harvest summers, we observed significant summer streamflow increases in all treated streams, ranging from 1.3 to 4.4 times the pre-harvest streamflow. However, the streamflow increases were not driven by riparian treatments but was more related to the amount of the catchment that was harvested. We also observed diel streamflow increases in each treated stream, ranging from 1.1 to 2.5 times the pre-harvest diel streamflow, with the riparian treatment and the amount of catchment harvested both influencing the increase.

Our results indicate that the riparian treatment applied has a limited effect on the summer headwater streamflow in the first few years following harvest, and other factors of the harvest are more impactful. Increasing riparian buffer density did not reduce the effects of timber harvest on summer low flow, and our results show that forest operations on the hillslope above the riparian treatment have large effects on headwater summer low flow. Our results also indicate that riparian vegetation is the driver of summer diel fluctuations in these sites, with the hillslope vegetation having little or no influence on the diel signals. This shows forest managers and forest regulators should not rely on standardized riparian buffers alone to protect water resources in headwater streams, as hillslope runoff processes also influence headwater streams. Localized riparian buffer design and harvest planning may be needed to sufficiently protect headwater streams in some areas.

Summer Low Flow Response to Timber Harvest and Riparian Treatments in Forested Headwater Streams of Coastal Northern California

by
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I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request.

Jonah Nicholas, Author

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CONTRIBUTION OF AUTHORS

Jonah Nicholas: Methodology, Formal analysis, Investigation, Writing – Original Draft, Writing – Review & Editing, Visualization.

Kevin Bladon: Conceptualization, Funding Acquisition, Methodology, Resources, Writing – Review & Editing, Visualization, Supervision, Project Administration.

Catalina Segura: Conceptualization, Funding Acquisition, Methodology, Resources, Writing – Review & Editing, Visualization, Supervision, Project Administration.

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1 Introduction

1.1 Headwater Streams

Headwater streams are individually small, but collectively significant pieces of a stream network that make up the upper-most drainages of a catchment area (Jackson, 2019). They are the smallest streams in a network and can be anywhere from centimeters to a few meters wide. Headwater streams are surrounded by ecologically important riparian zones which are influenced by both the stream and the surrounding terrestrial ecosystem (Anderson et al., 2007; Freeman et al., 2007; Schlosser, 1991). These small streams usually make up the majority of a stream network (70–80% of catchment drainage area) (MacDonald and Coe, 2007; Richardson, 2019; Sidle et al., 2000) and play invaluable roles in regulating nutrients, organic matter, and sediment in stream systems (Lowe and Likens, 2005). Headwater stream functions are under pressure from climate change (Turner et al., 2017), human activity (Chellaiah and Kuglerová, 2021), and wildfire area burned and intensity (Loiselle et al., 2020; Saksa et al., 2020).

Headwater streams are often closely associated with their surrounding riparian zones which are transitional areas between terrestrial and stream ecosystems (Clinton et al., 2010; Cole et al., 2020; Kallenbach et al., 2018). The connection between hillslopes and small streams allows for many interactions influencing water quantity and quality in the stream (Bernhardt et al., 2018; Coats and Jackson, 2020; Kiffney et al., 2003; Leach et al., 2017; Richardson, 2019). In forest ecosystems, these interactions affect stream water quantity through intercepting precipitation and transpiration, reducing the amount of water reaching the stream (Anderson et al., 2007; Bond et al., 2002; Dick et al., 2018). Additionally, the shade and wind protection provided by trees in the riparian zone influence riparian microclimate and the dead material from the surrounding vegetation adds nutrients and increases biological activity in the stream, affecting stream water quality (Anderson et al., 2007; Anderson and Poage, 2014; Moore et al., 2005; Olson et al., 2007). In Mediterranean climates, the effect of nearby vegetation on streams is often greater during the dry summer due to the absence of precipitation inputs and elevated evapotranspiration rates (Moore et al., 2020). However, the absolute influence of the riparian vegetation on streams may be modified by the species composition, vegetation density, canopy closure, and age of (Broadmeadow and Nisbet, 2004; Zhang et al., 2021).

This increase in vegetation effects in the warm season can result in substantial reductions in streamflow or the complete cessation of flow during the dry season (Acuña et al., 2014; Johnson et al., 2009; Lane et al., 2022). As climate change brings more variability to weather conditions and increased disturbance severity and scale disrupts ecosystem processes and habitats, riparian areas have been identified as potential “climate refugia” that provide stable terrestrial and aquatic habitat (Isaak et al., 2016; Krosby et al., 2018; O’Briain et al., 2017). Additionally, climate change driven unpredictable precipitation and weather patterns may increase the risk of headwater stream impermanence (Acuña et al., 2014; Buttle et al., 2019; Milliman et al., 2008). The interaction of changing climate and human forest management practices may lead to dramatic changes to headwater ecosystems. As such, it is increasingly important to improve our understanding of the effects of forest management on headwater stream summer streamflow (Lowe and Likens, 2005).

1.2 Quantifying Streamflow

The presence of sufficient streamflow for aquatic ecosystem processes to occur is essential for freshwater systems, so we looked at two aspects of headwater streamflow: daily streamflow and diel streamflow. Daily streamflow is the total volume of water flowing down a stream in 24 hours, here normalized by catchment area (mm d^{-1}) to ease comparison to other catchments. Diel streamflow is the range of discharge rate values over 24 hours, or the difference between the highest and lowest discharge rates in one day, here discussed in the units of streamflow rate (L s^{-1}) (Bren, 1997; Godwin, 1931). This value is in the absence of precipitation, as storm events water inputs dominate the smaller scale of diel streamflow. Diel streamflow is often referred to by other terms, with diel fluctuation, diel signals, diurnal cycles, and diurnal variations all being used to describe the same phenomenon (Bren, 1997; Burt, 1979; Kirchner et al., 2020; Moore et al., 2011; Schwab et al., 2016). Although historically ignored, diel fluctuations will be increasingly important to understand as our assessments of available water resources increase in detail and accuracy, requiring the inclusion of diel streamflow (Gribovszki et al., 2010). Additionally, methods to estimate riparian evapotranspiration using diel streamflow have been developed and understand the effect of timber harvests on diel streamflow may improve these methods (Gribovszki et al., 2010). the parameters of daily streamflow and diel streamflow characterize the quantity and timing of water in a stream and are useful measures of streamflow volume in headwater systems.

1.3 Hydrologic Effects of Timber Harvests

There is a long history of studying the effects of timber harvest on daily streamflow in the western United States (Brown, 1969; Monteith, 1965; Moring, 1975; Rothacher, 1970). Industrial forest operations disturb vegetation, forest structure, and soils of the surrounding hillslope, impacting runoff processes that drive streamflow in headwater streams (Bent, 2001; Bowling et al., 2000; Clinton, 2011). In general, forest harvesting results in increases in streamflow during the dry season for the first several years after harvest (Bosch and Hewlett, 1982; Moore et al., 2020; Segura et al., 2020). However, longer-term data has shown that summer streamflow decreases likely associated to higher transpiration rates of the younger trees that reestablish after harvest (Moore et al., 2020; Perry and Jones, 2017; Segura et al., 2020).

The retention of vegetated riparian buffers has long been recognized as an effective forest management option to mitigate sediment and nutrient transport from hillslopes to streams and limit stream temperature changes after harvesting (Bladon et al., 2016; Clinton, 2011; Hatten et al., 2018; Lee et al., 2020; Shearer, 2007). To provide the desired functions and protect streams, the majority of current riparian regulations and best management practices prescribe fixed-width buffers (P Lee et al., 2004; Richardson et al., 2012). Width criteria, such as waterbody type, presence of fish, and slope, vary regionally and generally designate a predetermined width of vegetation retention surrounding the stream (P Lee et al., 2004). Due to contention about how best to maximize both economic and ecological values from riparian vegetation, buffer regulations have been modified in many regions often without an assessment of their efficacy. Moreover, few studies have assessed the connection between riparian buffers and stream discharge (Shearer, 2007).

While the drivers of daily streamflow have been the focus of many studies, the collective knowledge of diel streamflow fluctuations in headwater streams is less developed (Gannon et al., 2020). Despite the long recognition of diel fluctuations in streamflow (Godwin, 1931), there is still uncertainty about the drivers of these daily fluctuations in streamflow (Bond et al., 2002; Gannon et al., 2020; Graham et al., 2013; Moore et al., 2011). Diel signals have been related to temperature-viscosity fluctuations, where increased water temperature reduces water viscosity enough to increase hyporheic and subsurface-flow (Schwab et al., 2016). Similarly, diel fluctuations have been related to daily snow melt cycles, where increased snow melt during the day increases streamflow (Gannon et al., 2020; Graham et al., 2013; Moore et al., 2011). A commonly observed driver of diel fluctuations in forested, non-snow dominated catchments has been transpiration cycles driven by riparian or hillslope vegetation (Barnard et al., 2010; Boggs et al., 2015; Bond et al., 2002). At night, streamflow reaches a maximum daily level when transpiration rates are at their lowest, but during the day elevated transpiration rates reduce flow of subsurface water to the stream (Boggs et al., 2015; Gannon et al., 2020; Graham et al., 2013). This pattern is often delayed from the peak evapotranspiration time in the day, as these diel signals must travel through flow paths to streams (Bren, 1997; Gannon et al., 2020; Graham et al., 2013; Wondzell et al., 2007). It also remains uncertain about how far from the stream, daily transpiration can influence streamflow rates. For example, some studies have observed diel signals generated from hillslope vegetation (Barnard et al., 2010; Burt, 1979; Moore et al., 2011), while others have concluded only riparian vegetation controls diel streamflow (Bond et al., 2002; Bren, 1997; Gannon et al., 2020). In general, few studies have assessed the effect of timber harvest or other disturbance on diel streamflow, with most concluding that harvest increases diel fluctuations, unless the riparian vegetation is removed (Bren, 1997).

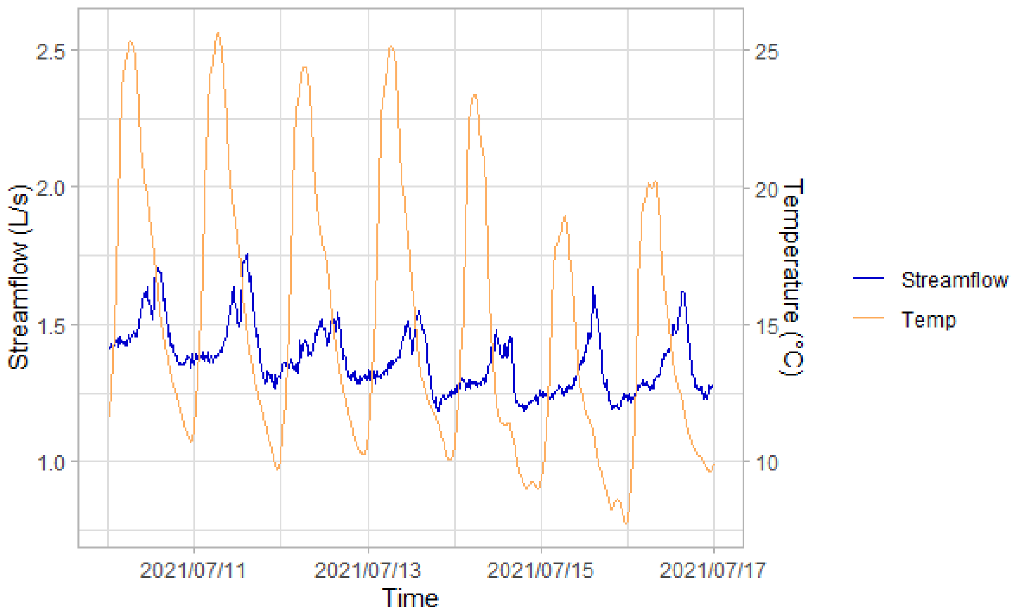

Figure 1. Example of diel streamflow over 7 days. In the absence of precipitation and snow pack, streamflow may decrease during the day and evening as transpiration reduces flow to the stream. Streamflow returns to higher levels during the night and morning. Streamflow and temperature measurements from stream REF1 for one week during July 2021.

1.4 Research Question

This study builds off the results of previous research and further assesses timber harvest impacts and riparian treatments on streamflow by applying three different harvest treatments to headwater streams in coastal northern California and observing changes in streamflow. We implemented three currently and previously used riparian buffer treatments alongside clearcut harvests and compared the results to two reference streams in conifer-dominated, steep topography coastal headwater streams. With the observations from these streams, we looked to answer the question: How does forest harvesting and different riparian buffer treatments affect summer low flow and summer diel fluctuations in the headwater streams of coastal northern California? Using summer streamflow measurements, we sought to quantify and compare the effects of harvest and postulate on the drivers of diel streamflow in these sites.

We hypothesized that all three timber harvests would increase the daily streamflow and the most intensive riparian buffer (most canopy cover removed) would increase the daily streamflow the most, as more vegetation removed decreases transpiration and interception, increasing runoff (Moore et al., 2020, Figure 2). Additionally, we expected the timber harvests to increase the diel streamflow due to increased transpiration rates in riparian buffer vegetation (Boggs et al., 2015; Bren, 1997; Gannon et al., 2020). We also predicted hillslope vegetation would not significantly contribute to diel fluctuations in streamflow in these sites because of subsurface flow contraction in the Mediterranean dry summers, and as such removal of the hillslope vegetation would provide more resources for riparian vegetation, our predicted dominant source of diel streamflow (Figure 2). We also predicted that the riparian treatments with the greatest canopy cover removal would result in lower diel streamflow. While our results did not align completely with our predictions, we did see an overall increase in total daily streamflow and an increase in diel streamflow following the harvests. However, we did not observe the riparian buffer treatment to have a large effect on the daily streamflow or diel streamflow.

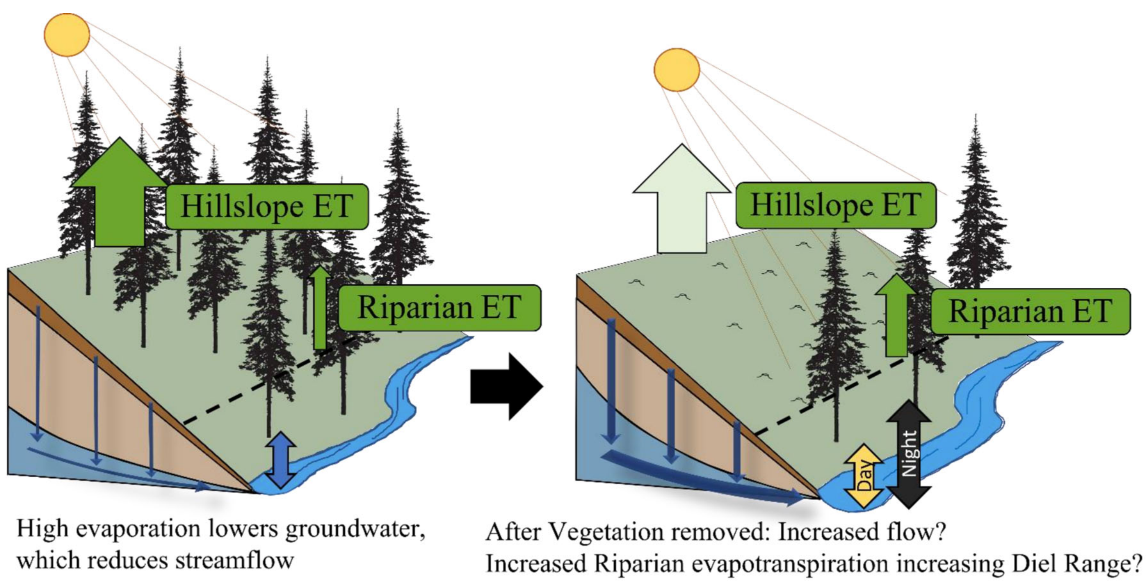

Figure 2. Graphical abstract portraying the expected results of timber harvest and riparian treatment. As hillslope vegetation is removed, transpiration and interception decrease allowing more daily streamflow. As riparian vegetation has access to more sunlight and water, diel signals increase.

2 Methods

2.1 Site Description

Our research was conducted in five small catchments in the Northern Coast Range of California, adjacent to Redwood National Park (41.252°N, 123.972°W). The Northern Coast Range of Northern California has steep topography, is heavily forested, and has a maritime climate greatly influenced by the Pacific Ocean. Mean daily air temperature in this area has historically varied between 23.5 °C and 0 °C with an annual average temperature of 10 °C. Annual precipitation typically exceeds 2,000 mm, with cool, wet winters and warm, dry summers (Davey et al., 2007). Most of the precipitation occurs as rainfall between the months of October and May, with rare inputs of snowfall or development of a snowpack. The proximity of the Pacific Ocean maintains cool temperatures and higher humidity and is associated with frequent dense fog (Burgess and Dawson, 2004; Dawson, 1998; Ewing et al., 2009; Francis et al., 2020). This unique coastal climate extends approximately 80 km inland (Davey et al., 2007)—all five of our study watersheds were located within 20 km of the Pacific Ocean and well within this maritime area. Soils in our study area are well-drained gravelly clay loams of the Coppercreek and Sasquatch series, with depths typically between 70 to 100 cm (Soil Survey Staff, NRCS, 2016). Field observations indicated increasing clay content with soil depth. The uppermost soil layers were composed of well drained organic detritus, likely leading to the presence of soil macropores and pipes that quickly transmit runoff as shallow subsurface stormflow to the stream (Amatya et al., 2016; Keppeler & Brown, 1998). The lithology consists of marine-derived sedimentary and metasedimentary rock of the Franciscan Complex (Woodward et al., 2011). The watersheds have complex topography with many undulations and mean catchment slopes ranging from 18–25 °.

Our study catchments drain part of the Tectah watershed, which is a tributary of the Klamath River basin (Figure 3). Before this study, the catchments were entirely forested and managed for commercial timber. All five catchments historically supported old growth redwood forests that were first harvested in the early 1900s. The forests were reestablished as a mix of mature (43–44 yrs. at the beginning of the study), even-aged stands of Douglas-fir (*Pseudotsuga menziesii*) and coastal redwood

(*Sequoia sempervirens*), with western hemlock (*Tsuga heterophylla*), red alder (*Alnus rubra*), and tanoak (*Notholithocarpus densiflorus*) also common. Prevalent understory species included Pacific madrone (*Arbutus menziesii*), California laurel (*Umbellularia californica*), huckleberry (*Vaccinium ovatum*), salal (*Gaultheria shallon*), and western sword fern (*Polystichum munitum*). The riparian area species composition and stand structure varied between different streams, with red alder dominating three of our study catchments while conifer dominated the other two catchment riparian areas.

Our study sites were headwater streams with steep channels (16–22 %) and small drainage areas (30–40 ha). Streams were primarily step-pool systems with sections of cascade morphology (Montgomery and Buffington, 1997) (Table 1). The stream valleys were V-shaped with narrow riparian areas. The monitored sections of our study streams had active channel widths that were less than two meters. All streams were considered perennial and remained flowing through the entire study period.

Table 1. Topographic characteristics of our five study catchments and streams.

Stream/ Treatment*	Catchment Area	Average Catchment Slope	Mean Stream Gradient**	Stream Length	Active Channel Width
	(ha)	(°)	(%)	(m)	(m)
REF1	40.4	24.1	20	1072	1.7
REF2	30.3	18.2	16	1310	1.4
HCP	27.9	22.6	22	960	1.9
ASP	37.0	22.3	16	1383	1.8
PRE	30.1	23.6	20	1093	1.9

* Described in section 2.2.

** Stream gradient range was obtained from GIS topographical analysis from 10 m resolution USGS National Elevation Dataset.

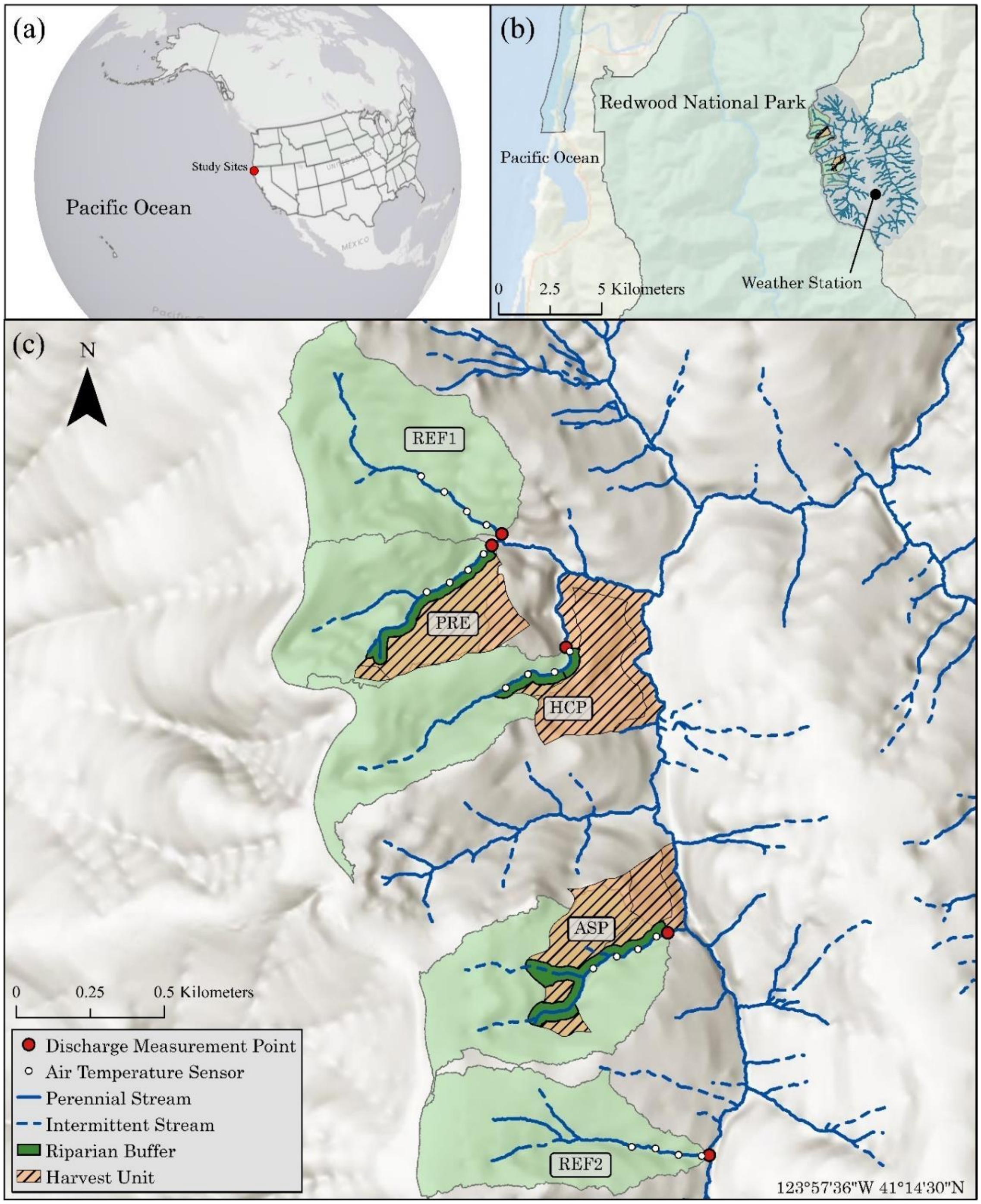

Figure 3. (a) Map of the location of our study catchments in coastal Northern California. (b) Map of our study sites, illustrating their location relative to Redwood National Park and the Pacific Ocean. (c) Map of our study catchments showing the discharge measurements points and the observed extent of perennial and intermittent streams, riparian buffers, and harvest units.

2.2 Treatments

We designed our study to compare the effects of forest harvesting and three different riparian buffer treatments on streamflow. For comparison, we instrumented and monitored two reference catchments (REF1 and REF2), which remained unharvested during the entire study period (Figure 4A). In our other three study catchments, forest harvesting followed current forest practices regulations, which limited the cutblock areas to less than 8.1 ha and constrained the harvesting to one side of the stream. However, we were able to implement different riparian buffer treatments for research purposes. The first riparian buffer treatment we assessed was associated with the current California Anadromous Salmonid Protection (ASP) regulations (Crowfoot and Porter, 2021). The current policy requires the retention of a 30.5 m riparian buffer between harvested areas and perennial streams. Within the ASP buffers, the trees growing in the 9.1 m adjacent to the stream were left untouched as the “core zone”, while minimal harvesting of trees in the outer 21.3 m occurred with the target retention of at least 80 % of the canopy cover (Figure 4B). We also assessed the effectiveness of an approved modification to the current ASP regulations, which was the riparian buffer associated with the Habitat Conservation Plan (HCP) of our industry partner. The HCP riparian buffer also required retention of 30.5 m of vegetation adjacent to the stream after forest harvesting. However, some of the trees in the 9.1 m directly adjacent to the stream were harvested with a target retention of 85 % canopy cover, while the trees in the outer 21.3 m of buffer were harvested with a target retention of 70 % canopy cover (Figure 4C). Finally, we also assessed the effectiveness of the riparian regulations that preceded the current ASP regulations, which were replaced in 2009. These pre-ASP (PRE) riparian buffer regulations also required the retention of a 30.5 m vegetated buffer, but the canopy cover was reduced to 50 % over the whole width of the buffer (Figure 4D).

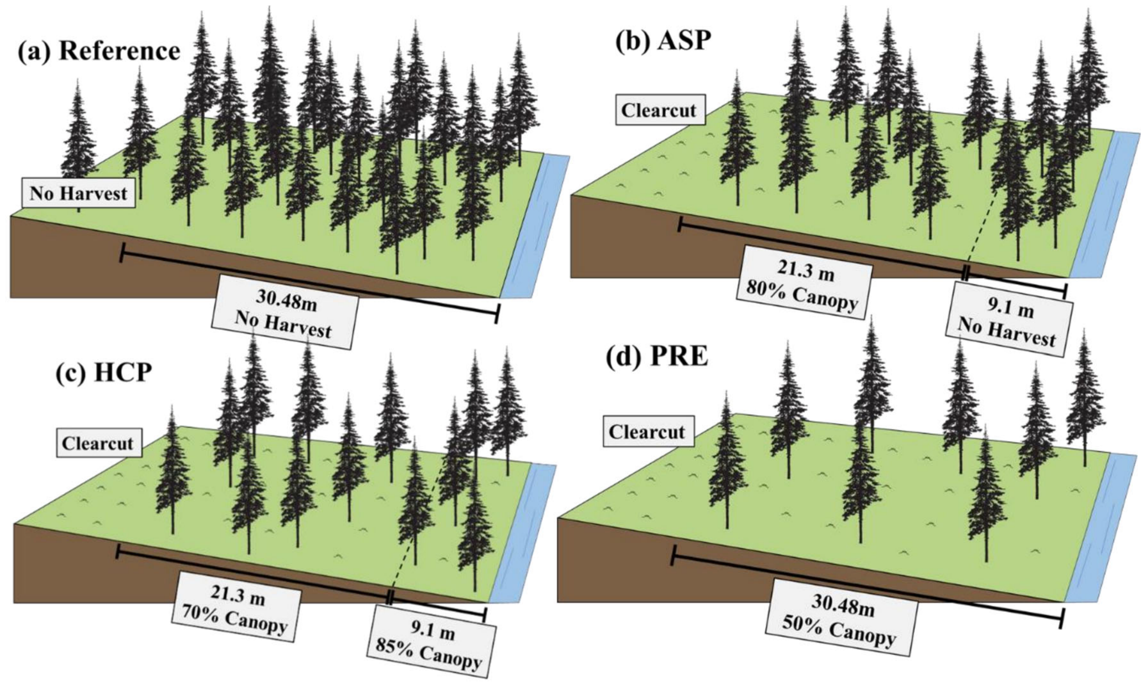

Figure 4. Diagrams of the three riparian buffer treatments examined in our study, including the (a) reference (REF1 and REF2) catchments, (b) current California Anadromous Salmonid Protection (ASP) riparian prescription, (c) modified ASP or approved habitat conservation plan (HCP) riparian prescription, and (d) pre-2009 riparian prescription (PRE). The percent canopy indicated below each prescription is the target percent of canopy cover retained in the riparian area.

Forest harvesting occurred in the study sites from September 14th to December 16th, 2020, primarily using a cable yarding system, with small areas harvested using a ground-based system. During the cable-based harvest, trees on the study sites were felled using chainsaws by ground crews and moved to landings using cable systems. In the riparian buffers, individual trees were marked for harvest before the harvest began and were felled into clearings between trees. Felled trees were then pulled up to landings by the cable system. As part of the normal harvesting procedure, rocked roads were constructed to landing sites and skid trails were constructed where ground-based systems were used. Given that we were assessing an industrial operation, we had to forgo some of the tightly controlled aspects in the study design in favor of operational reality, resulting in usual variability in the harvesting intensity in our study catchments (Table 2).

Table 2. Study site, treatment implemented, and aspect of the harvested sites.

Stream/* Treatment	Catchment Area	Clearcut Area in the Catchment	Percent Catchment Harvested	Area of Riparian Buffer	Percent Riparian Buffer Treated	Harvest Area Aspect
	(ha)	(ha)	(%)	(ha)	(%)	
REF1	40.4	--	0	--	--	--
REF2	30.3	--	0	--	--	--
HCP	27.9	0.9	3.3	1.1	18.4	N
ASP	37.0	7.1	19.1	3.4	47.1	S
PRE	30.1	7.6	25.2	1.9	32.1	NW

* Described in section 2.2

2.3 Field data

We quantified streamflow in each catchment throughout the summer months (June–August) during one summer of the pre-harvest period (2020) and two summers of the post-harvest period (2021–2022). To quantify streamflow, we recorded continuous (every 15 minutes) measurements of stream stage at the outlet of each catchment with pressure transducers (Solinst Levelogger Edge Model 3001, accuracy ± 0.05 %) installed inside stilling wells. All measurements were compensated for barometric pressure (Solinst Barologger Edge model 3001, accuracy ± 0.05 kPa). We quantified instantaneous discharge using salt dilution gauging with a slug injection (Moore, 2005, 2004). We injected a slug of salt solution containing 0.075–0.15 kg of salt and 2 L of stream water 20 meters above our measurement point. The salt solution was determined to increase the background electrical conductivity (EC) approximately 200 %, which was dependent on the streamflow volume at the time of measurement. We quantified EC downstream of the slug injection site at one second intervals for ~20–90 minutes until the conductivity returned to background levels (YSI ProDSS Multiparameter Digital Water Quality Meter; accuracy ± 0.5 %; Figure 5). Following the salt wave measurement, we quantified the calibration constant (k) of the stream water by repeatedly adding increments of salt solution to a known volume of stream water and simultaneously measuring the increased EC with each addition of salt solution. This produced a set of EC values that corresponded to the relative concentrations of salt in the stream water sample (Moore, 2005). The slope of the relationship between concentration and EC provided the estimate of k. We repeated this process after each streamflow measurement because values of k can vary with suspended sediment and mineral content of the stream water at the time of measurement (Hongve, 1987).

With the EC wave and calibration constant measured, we used the total increase in EC compared to the EC background value to calculate the streamflow at that time:

$$Q = \frac{V}{k\Delta t \sum[EC(t) - EC_{bg}]}$$

where Q = streamflow (L s^{-1}), V = volume of salt injected (L), k = calibration constant (cm μS^{-1}), Δt = conductivity measurement interval (s), EC(t) = conductivity measurement ($\mu S\ cm^{-1}$), and EC_{bg} = background conductivity measurement ($\mu S\ cm^{-1}$). This process was repeated approximately 12 times per stream throughout the summer and fall to capture a range of streamflow events during the pre-harvest year (2020) and first post-harvest year (2021) to develop a relationship between the continually measured stage and instantaneous discharge. We used a power function to develop a unique stage-discharge relationship for each stream:

$$Q = C(h + a)^n$$

where Q = streamflow ($m^3\ s^{-1}$), h = recorded stage readings (m), and C, a, n = variable constants. Each relationship was fitted using the minipack.lm package in R (Elzhov et al., 2022) (Appendix A).

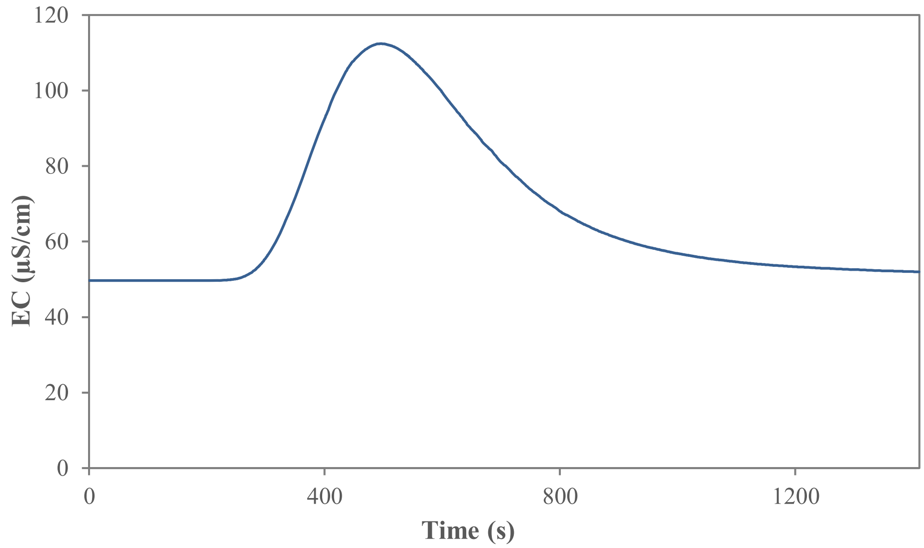

Figure 5. Example of the temporal measurement of electrical conductivity (EC) in a stream as the injected slug of salt passes by the measurement point. We targeted a peak conductivity ~2-times greater than the baseline EC.

To quantify the change in vegetation cover after the timber harvest, we sampled six fixed-area circular mensuration plots (area: 0.026 ha) in each of our study catchments. We spaced the mensuration plots evenly along the length of the riparian buffer, with a spacing of ~58–105 m apart, depending on the length of reach (Figure 6). We collected riparian vegetation data in the summer months during both the pre- and post-harvest periods. Each plot center was located ~10 m upslope from the edge of the stream, attempting to capture both the core and outer sections of the riparian buffers. Within each plot we quantified stand density (trees per unit area), basal area (based on tree diameter at breast height; m^2 ha^{-1}), and canopy closure. Canopy closure was estimated using hemispherical photography and image processing (Glatthorn and Beckschäfer, 2014; Paletto and Tosi, 2009; Zhang et al., 2005). We captured hemispherical photos of the canopy using a Nikon D7100 DSLR camera with a Sigma circular fisheye lens (180° viewing angle, 1:6 magnification) 1 m above the center of each mensuration plot. At each plot we took photos with a range of exposures, centered on the sky exposure measured in a clearing, to ensure we obtained a correctly lighted photo to process. Using the HemiView program, we distinguished a threshold brightness value between sky and canopy and categorized each pixel in the photo as either canopy or sky. We then calculated the percentage of the view above each plot that was canopy cover (Vieglais, 1996). Some photos required manual edits to darken areas of sun glare, reflections, or areas in direct sunlight (Appendix A).

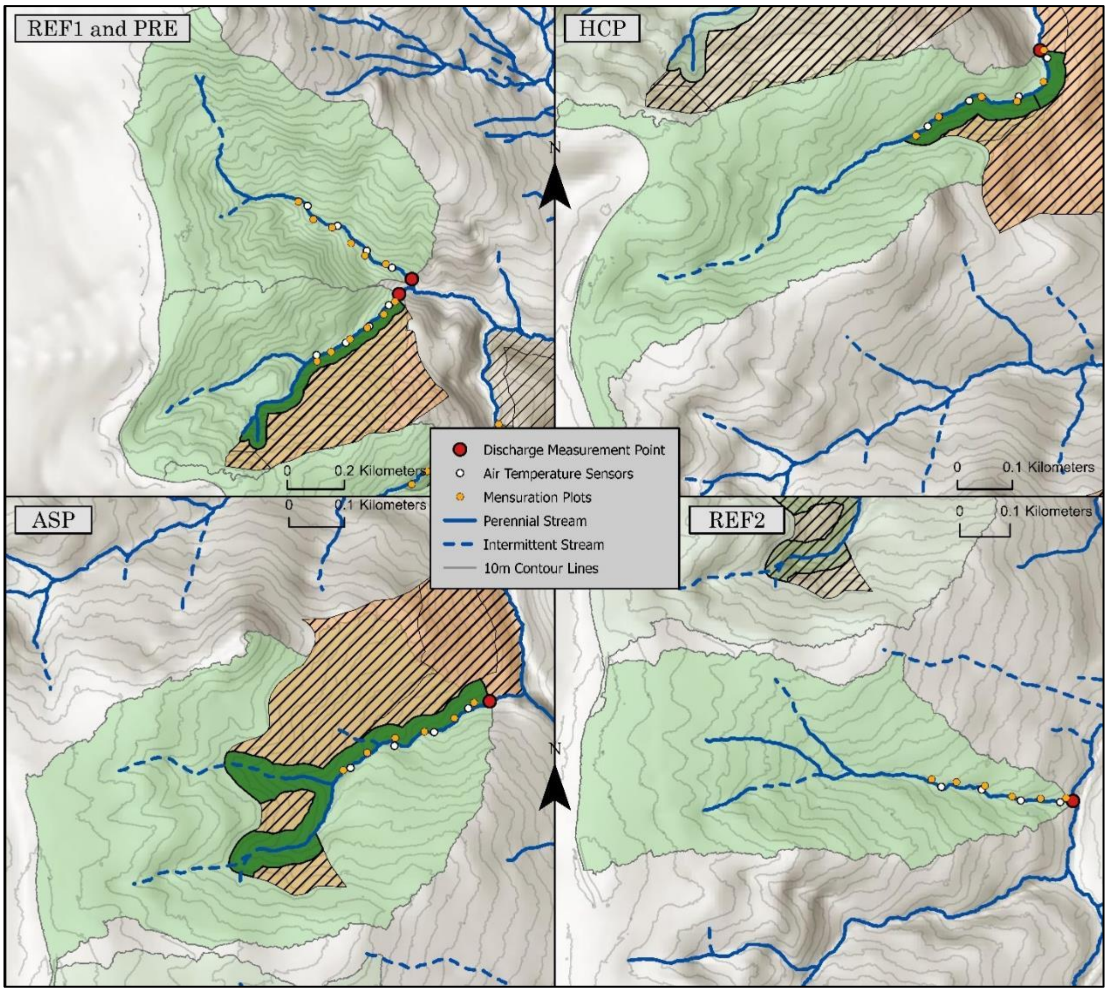

Figure 6. Detailed map of our five study sites, including discharge measurement locations, air temperature thermistor locations used in estimating local potential evapotranspiration, locations of canopy cover measurements, observed extent of perennial and intermittent streams, riparian buffers, and harvest units.

We established a meteorological station (Onset HOBO U30 NRC) approximately two kilometers southeast of our study catchments and at an elevation of 750 m. Over the period of study, the meteorological station provided data on precipitation (tipping bucket S-RGB-M002, 0.2 mm resolution, ± 1 % accuracy), barometric pressure (S-BPB-CM50, ± 3.0 mbar accuracy), relative humidity (S-THB-M008, ± 0.2 °C accuracy), net radiation (solar pyranometer S-LIB-M003, ± 10 W m^{-2} accuracy), and wind speed (S-WSB-M003, ± 1.1 m s^{-1} accuracy). We also quantified local air temperature near each study stream with four thermistors evenly spaced along the riparian corridor and approximately ~1 m above the stream. With some variation between the streams, the first thermistor was placed ~20 m upstream of the outlet with each following thermistor located ~100 m farther up the reach (Figure 6). We used mean local air temperature along with atmospheric pressure, relative humidity, solar radiation, and wind speed from the meteorological station to estimate potential evapotranspiration (PET) with the Penman-Monteith method (Dlouhá et al., 2021; Monteith, 1965).

2.4 Statistical Analyses

We compared pre- and post-harvest climatic conditions, canopy closure, and summer flows (i.e., daily magnitude and diel range) to assess changes due to the timber harvests and riparian buffers. We used a parametric test (t-test) for normally distributed variables and a non-parametric test (Wilcoxon rank-sum test or Wilcoxon signed rank test) for variables that were not normally distributed or had a small sample size (Student, 1908; Wilcoxon, 1945). As such, a t-test was used for assessing changes in the air temperature, solar radiation, relative humidity (RH), and potential evapotranspiration (PET). We used the Wilcoxon rank-sum test for assessing changes in the precipitation, canopy closure, and summer streamflow (both daily total and diel range), and used the Wilcoxon signed rank test to assess the significance of the ratio between post-harvest and pre-harvest streamflow (both daily total and diel range).

We compared both streamflow magnitude and diel streamflow between pre- and post-harvest periods. However, to account for climatic differences between the pre- and post-harvest periods we applied correction factors. These factors were estimated based on observations in the reference sites. We used both reference sites (REF1 and REF2) to correct the first post-harvest summer (2021) but only used the first reference site (REF1) to correct the second post-harvest summer (2022), due to the second post-harvest site being harvested between the two post-harvest summers.

We developed linear mixed-effects models to predict daily streamflow and summer diel streamflow based on climatic observations for the pre- and post-harvest periods independently. The optimal overall model was selected using Akaike information criterion (Akaike, 1974). When modeling daily streamflow, a model with maximum daily air temperature (T_a) as the independent variable and month of the summer (July, August) as covariates was the strongest model (Appendix B):

$$\mu\{Streamflow|Air\ Temperature\} = \beta_0 + \beta_1 * T_a + \beta_2 * I.July + \beta_3 * I.August + \varepsilon$$

where β_0 is the intercept, β_{1-3} are model coefficients, and I.July and I.August are indicator variables representing the month of the summer. Temporal autocorrelation in daily streamflow was accounted for by including a correlation factor in the model using the lme function in the nlme R package (Pinheire et al., 2018).

When modeling diel streamflow, a model with daily PET as the independent variable and month of the summer as a covariate was the strongest model:

$$\mu\{Diel\ Streamflow|PET\} = \beta_0 + \beta_1 * PET + \beta_2 * I.July + \beta_3 * I.August + \varepsilon$$

where β_0 is the intercept, β_{1-3} are model coefficients, and I.July and I.August are indicator variables representing the month of the summer.

The difference between the pre- and post-harvest models was used to calculate a scale factor to correct observed streamflow data in the harvested watersheds in the post-harvest period (Figure 7). Separate scale factors were estimated for each post-harvest study year—2021 and 2022. The effect of the different riparian treatments was assessed based on comparisons of the adjusted daily streamflow diel streamflow distributions, both in terms of absolute values in mm for daily streamflow and L s^{-1} for diel streamflow, and in relative terms using ratios between post-harvest and pre-harvest streamflow ($\frac{Q_{Post}}{Q_{Pre}}$).

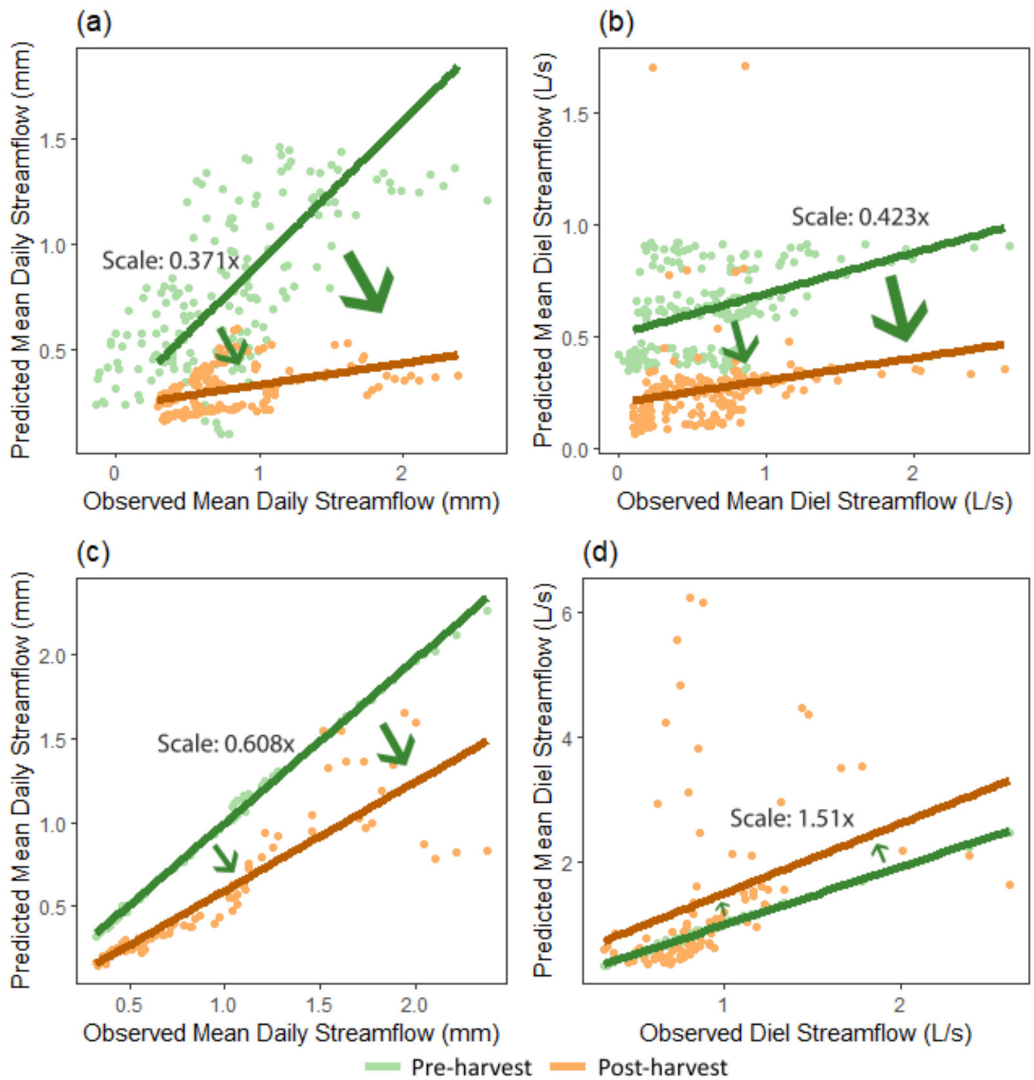

Figure 7. Scale factors used to correct observed streamflow data based on climatic differences. The difference between the slopes of the pre- and post-harvest predicted streamflow in the reference streams is the scale factor. (a) Scale factor adjusting daily streamflow in 2021. (b) Scale factor adjusting diel streamflow in 2021. (c) Scale factor adjusting daily streamflow in 2022. (d) Scale factor adjusting diel streamflow in 2022.

To complement the primary tests, we also assessed the change in the relationship between cumulative streamflow of each of the three harvested catchments and cumulative streamflow in the reference catchments. We created a generalized linear model for the relationship between the cumulative streamflow for each harvested and reference stream. We determined if the relationship changed significantly by including an indicator variable in the model representing the pre- and post-harvest time periods. If this factor was significantly different than 0, then we conclude the relationship changed significantly. Due to the lack of independence in our samples, we could not perform an ANCOVA test to determine a change in slope. This secondary test did not account for differences in climate between the pre- and post-harvest periods. All statistical analyses were conducted using R (R Core Team, 2018), and linear mixed effects models were created using the nlme package and model comparisons were performed using the emmeans package (Lenth et al., 2022; Pinheire et al., 2018).

3 Results

3.1 Study period meteorological observations

During the spring and summer (March–August) of the pre-harvest year (2020) the total precipitation was 415 mm. Approximately 95 % (395 mm) of the precipitation fell during the spring (March–May). Comparatively, during the spring and summer of the first post-harvest year (2021) the total precipitation was only 251 mm, with 80 % (201 mm) falling in the spring. During the second post-harvest year (2022), the total precipitation was 595 mm with 81 % falling during the spring. Precipitation was not significantly greater during the pre-harvest period compared to the first post-harvest year ($U = 15683$, $p < 0.340$). However, during the second post-harvest year the precipitation was significantly greater than the pre-harvest year ($U = 14456$, $p = 0.014$; Figure 8).

The daily mean air temperature during the pre-harvest summer (June–August 2020) was 18.7 ± 5.0 (SD) °C. Comparatively, the daily mean air temperature during the first post-harvest summer was 19.4 ± 4.5 °C and during the second post-harvest summer was 17.8 ± 4.6 °C. We did not find evidence of a difference in daily mean air temperature between the pre-harvest summer and either of the post-harvest summers (2021: $t = 1.20$, $p = 0.16$; 2022: $t = 1.19$, $p = 0.24$). The mean solar radiation during summer 2020 was 260.8 ± 327.8 W m^{-2}, while the mean solar radiation during summer 2021 was 266.6 ± 331.8 W m^{-2} and summer 2022 was 196.9 ± 52.9 W m^{-2}. Statistically, there was no evidence that the solar radiation was different between summer 2020 and summer 2021 ($t = 0.03$, $p = 0.97$); however, there was strong evidence that summer 2022 received less radiation ($t = 10.4$, $p < 0.001$). Relative humidity (RH) also varied between summers. The mean RH during the pre-harvest period was 64.3 ± 16.7 %. The mean RH during the first post-harvest summer was 61.2 ± 14.4 % while the mean RH during the second post-harvest summer was 71.1 ± 16.2 %. We found no evidence of a difference in RH between the pre- and first post-harvest summer ($t = 1.40$, $p = 0.032$), but we did find evidence of a difference in RH between the pre-harvest and second post-harvest summers ($t = 2.82$, $p = 0.005$; Figure 8).

Potential evapotranspiration during the pre-harvest period was 3.3 ± 0.7 mm day^{-1}, compared to a mean daily PET during the first post-harvest period of 3.6 ± 0.7 mm day^{-1} and the second post-harvest period of 3.07 ± 0.79 mm day^{-1}. Statistically, there was no evidence that PET was different between the pre- and first post- harvest years (t = 1.04, p = 0.33); but there was evidence of a difference in PET between the pre-harvest and second post-harvest years (t = 2.15, p < 0.033; Figure 8).

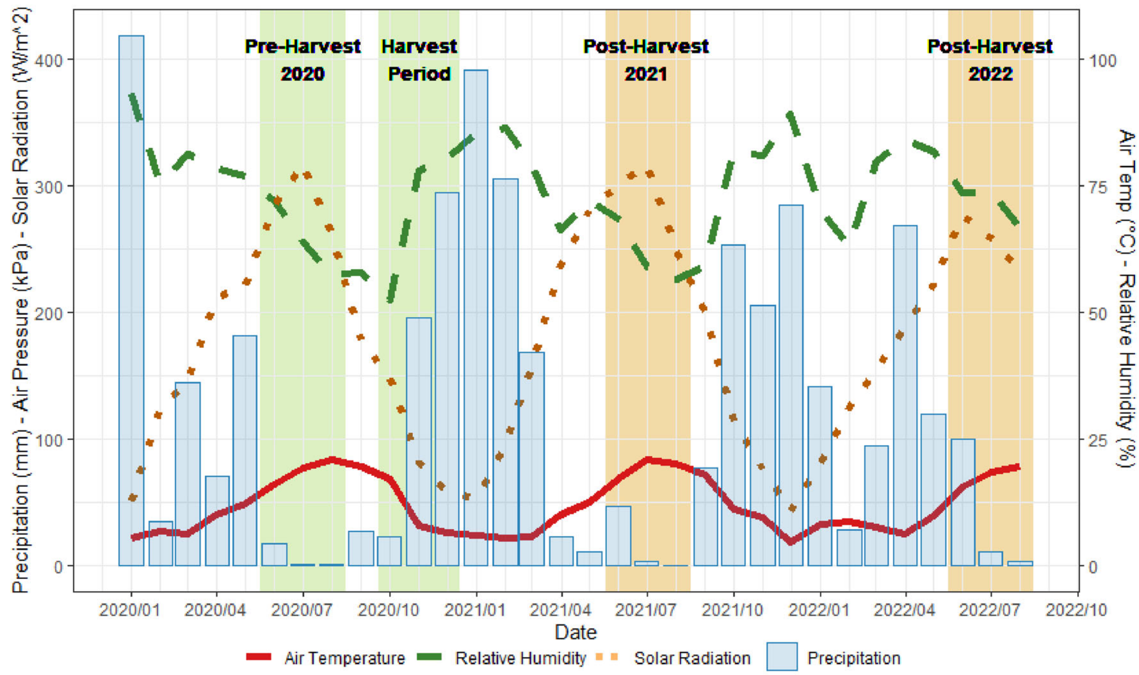

Figure 8. Precipitation, air temperature, solar radiation, and relative humidity between January 2020 and September 2022 at a meteorological station located 2 km from the study area. The green vertical bands indicate the pre-harvest summer and the harvest period, and the orange vertical bands indicate the focal period of our study—the first and second first post-harvest low flow periods.

3.2 Canopy cover

The canopy cover of the riparian areas across all five study catchments exceeded 90 % during the pre-harvest period (Figure 9). Specifically, the median canopy cover [95% CI] along the two reference streams was 97 % [96, 99] in REF1 and 98 % [96, 99] in REF2. Similarly, during the pre-harvest period the median canopy cover was 96 % [95, 97] along the ASP stream, 93 % [91, 98] along the HCP stream, and 93 % [90, 94] along the PRE stream.

During the post-harvest period, the median canopy cover [95% CI] along the two reference streams decreased by 1–3 % to 96 % [95, 97] in REF1 and 95 % [87, 97] in REF2 (Figure 9). As expected, the largest change in median canopy cover after harvesting occurred in the PRE treatment, where the canopy cover decreased by 6 % [0.6, 14.4] ($z = 1.52$, $p = 0.132$). We measured negligible changes in canopy cover in the other two treatments: the second largest decrease was in the ASP stream where the median canopy cover in the riparian buffer declined by 2 % [0.6, 6.75] ($z = 2.09$, $p = 0.026$) and the median canopy cover in the riparian buffer of the HCP site increased slightly by 3 % [0.1, 4.6] ($z = 1.36$, $p = 0.180$; Figure 9).

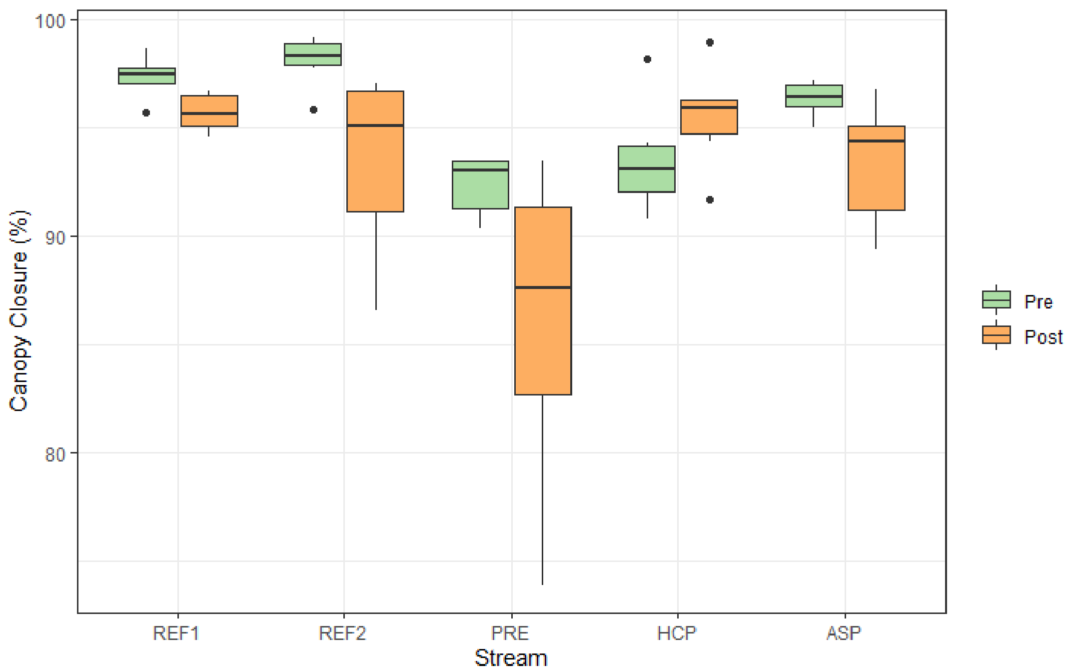

Figure 9. Box plots of canopy closure in the riparian areas of each of the five study streams during the pre-harvest and post-harvest periods. Displayed are the median, interquartile range (IQR) and outliers of canopy cover measurements in the six mensuration plots along each study stream. Any value greater than 1.5-times the IQR from the first or third quartiles is represented by a point.

3.3 Streamflow

To make comparisons between catchments easy, we assessed daily streamflow in units of mm, normalized by drainage area. During the pre-harvest period (2020), the total summer streamflow was 83 mm in REF1 and 53 mm in REF2. Thus, the mean daily streamflow (SD) was 0.9 ± 0.5 mm day^{-1} in REF1 and 0.6 ± 0.2 mm day^{-1} in REF2. Total streamflow during the summer of the pre-harvest period was 59 mm in the ASP stream, 109 mm in the HCP stream, and 32 mm in the PRE stream. Mean daily streamflow of 0.6 ± 0.2 mm day^{-1} in the ASP stream, 1.2 ± 0.6 mm day^{-1} in the HCP stream, and 0.4 ± 0.2 mm day^{-1} in the PRE stream (Figure 10.a).

During the first post-harvest summer (2021), total normalized summer streamflow decreased to 22 mm in REF1 and 35 mm in REF2. Thus, mean daily streamflow was 0.2 ± 0.1 mm day^{-1} in REF1 and 0.4 ± 0.1 mm day^{-1} in REF2. Comparatively, the total summer streamflow increased to 91 mm in the ASP stream, decreased to 53 mm in the HCP stream, and increased to 43 mm in the PRE stream. Thus, the mean daily streamflow was 1.0 ± 0.1 mm day^{-1} in the ASP stream, 0.6 ± 0.1 mm day^{-1} in the HCP stream, and 0.5 ± 0.2 mm day^{-1} in the PRE stream (Figure 10.a). However, these direct comparisons of raw streamflow rates between the pre- and post-harvest periods did not account for the large difference in precipitation that occurred in the study period.

Due to the harvest of stream REF2 before the second post-harvest summer, the observed is not representative of an untreated site, and therefore we will not present the second post-harvest summer (2022) streamflow data for REF2. During the second post-harvest summer (2022), we observed total summer streamflow to be 48.8 mm in REF1, 151.1 mm in ASP, 67.3 mm in HCP, and 83.3 mm in PRE. The corresponding mean daily streamflow (SD) was 0.5 ± 0.4 mm day^{-1} in REF1, 1.6 ± 0.4 mm day^{-1} in ASP, 0.7 ± 0.3 mm day^{-1} in HCP, and 0.9 ± 0.4 mm day^{-1} in PRE (Figure 10.a).

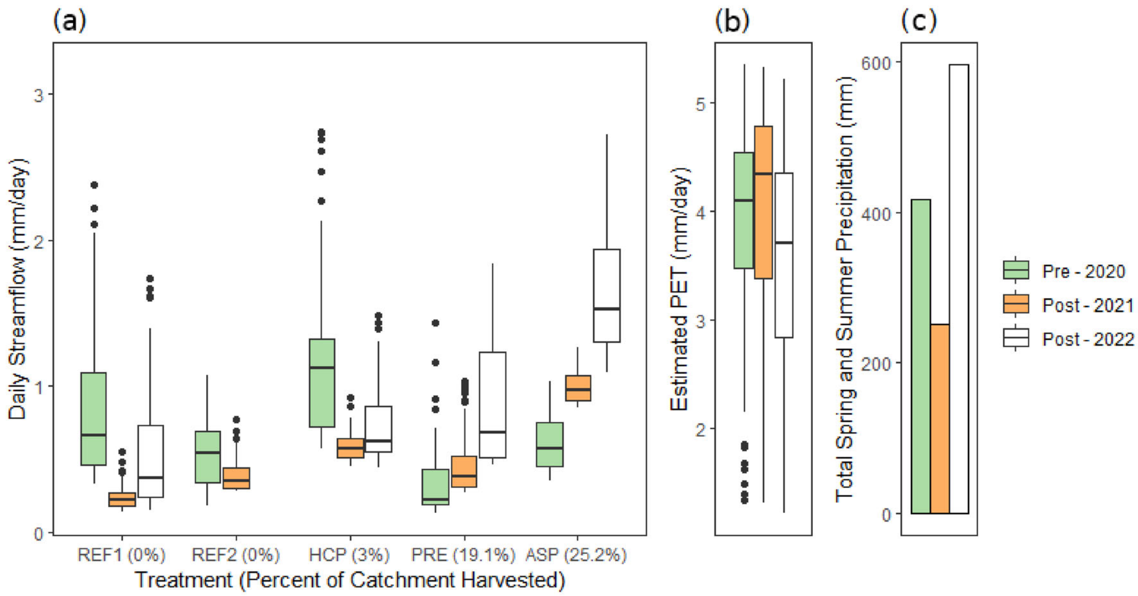

Figure 10. (a) Total daily streamflow in each study stream during the pre-harvest and post-harvest periods over the summer months (June–August). Sites are ordered by percentage of catchment harvested in the treatment (indicated on the x-axis). (b) Potential evapotranspiration (PET) during the pre-harvest and post-harvest periods. (c) Total spring and summer (March–August) precipitation during the pre-harvest and post-harvest periods.

After compensating for differences in weather conditions by applying correction factors (described in section 2.4), we observed a substantial increase in daily streamflow during the low flow season in all three of the harvested streams. The increases appeared to be related to the proportion of catchment area harvested (Figure 11). The greatest increase in median daily streamflow [95 % CI] occurred in the ASP stream, which had the greatest proportion of catchment harvested (25.2%)—we observed a 0.74 mm [0.72, 0.77] or 74 % increase in the first post-harvest summer ($W = 8464$, $p < 0.001$) and an even larger increase in streamflow in the second post-harvest summer of 1.13 mm [1.04, 1.27] or 113 % ($W = 8464$, $p < 0.001$). Comparatively, the second largest increase in median daily streamflow was 0.27 mm [0.24, 0.30] or 54 % during the first post-harvest summer in the PRE stream ($W = 8306$, $p < 0.001$), which had 19.1% of the catchment harvested. We observed a larger increase of 0.52 mm [0.42, 0.75] or 105 % in the second post-harvest summer. The smallest increase in median daily streamflow was 0.20 mm [0.14, 0.24] (41%) in the HCP stream during the first post-harvest summer ($W = 6622$, $p < 0.001$), which had only 3% catchment area harvest. We measured a non-statistically significant increase of 0.074 mm [-0.039, 0.14] or 12% in the second post-harvest year ($W = 4722$, $p = 0.175$; Figure 11.a).

We also found it insightful to look at the ratio of post-harvest to pre-harvest streamflow in both post-harvest years. This allows us to account for the variety in pre-harvest streamflow between the treatment streams. In summer 2021, the median ratio of post-harvest to pre-harvest summer streamflow [95 % CI] in the HCP treatment was 1.6 [1.4, 1.7] ($V > 4278$, $p < 0.001$), while the PRE treatment has a median ratio of 4.1 [3.8, 4.3] ($V > 4278$, $p < 0.001$), and the ASP treatment had a median ratio of 4.4 [4.2, 4.5] ($V > 4278$, $p < 0.001$). We also observed a significant increase in streamflow in each treated stream in the second post-harvest summer. In summer 2022, the median ratio of post-harvest to pre-harvest summer streamflow in the HCP treatment was 1.1 [1.1, 1.6] ($V = 3058$, $p < 0.001$), while the median ratio in the PRE treatment was 4.4 [4.3, 4.6] ($V > 4278$, $p < 0.001$), and the median ratio in the ASP treatment was 4.2 [4.1, 4.3] ($V > 4278$, $p < 0.001$; Figure 11.b)

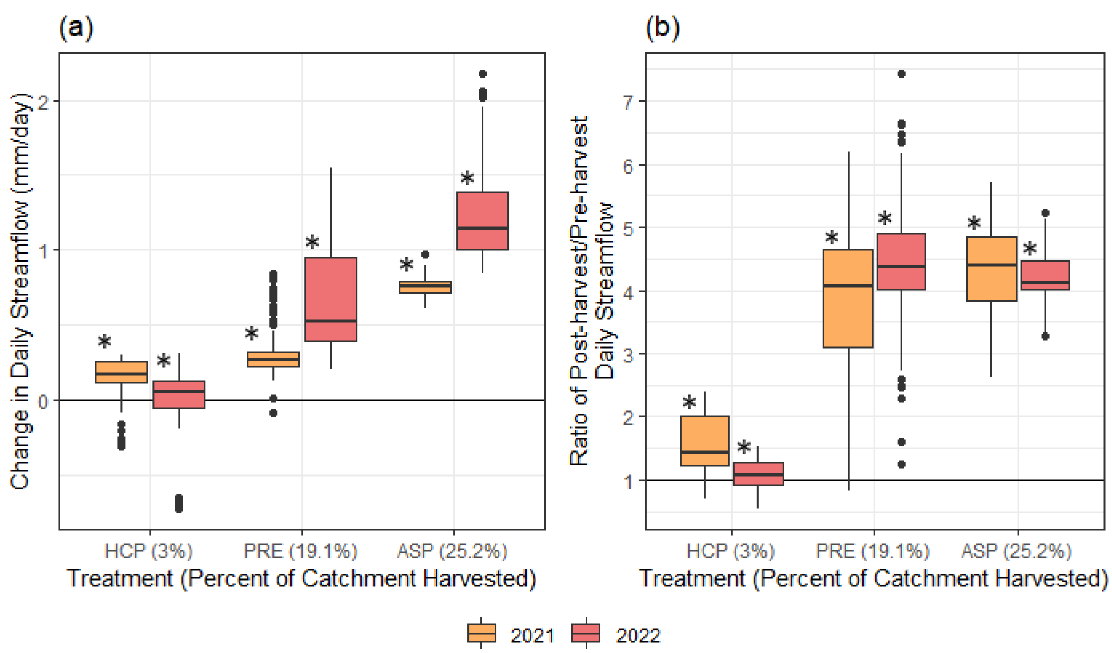

Figure 11. Change in daily streamflow in treated streams. Streamflow was adjusted for different weather effects in each year of the study (see section 2.4). Changes in

both years (2021 and 2022) are relative to the pre-harvest year (2020). Sites are ordered by percentage of catchment harvested in the treatment. Asterisk indicates significance at a 95 % significance level. (a) Magnitude of change in daily streamflow. (b) Ratio of post-harvest to pre-harvest daily streamflow. A ratio of 1 indicates no change from the pre-harvest year.

After quantifying the change in summer daily streamflow in each of the treatments, we also looked at this change in each month. We found a variety of patterns in the resulting streamflow changes. When looking at the absolute change in daily streamflow, the PRE and ASP treatments both resulted in an increase in streamflow that reduced as the summer progressed. This reduction was more extreme in the second post-harvest year (2022), but the overall increase in streamflow was also higher in the second post-harvest summer. The HCP treatment, in contrast, resulted in a larger increase in streamflow as the summer progressed. Another interesting pattern is the reduction of variance in daily streamflow as the summer progressed in all three treatments (Figure 12.a).

It is also useful to look at the ratio of post-harvest to pre-harvest streamflow, as this accounts for the somewhat different pre-harvest streamflow levels in each of the treated streams. In the HCP treatment, the same patterns seen in the absolute change in streamflow are present. In the PRE treatment, we see an undulating pattern as the summer progresses, with the change in streamflow decreasing in July but increasing in August. The ASP treatment resulted in an increase in streamflow change steadily through the first post-harvest summer (2021), but a pattern similar to the PRE treatment in the second-post harvest summer (2022) (Figure 12.b).

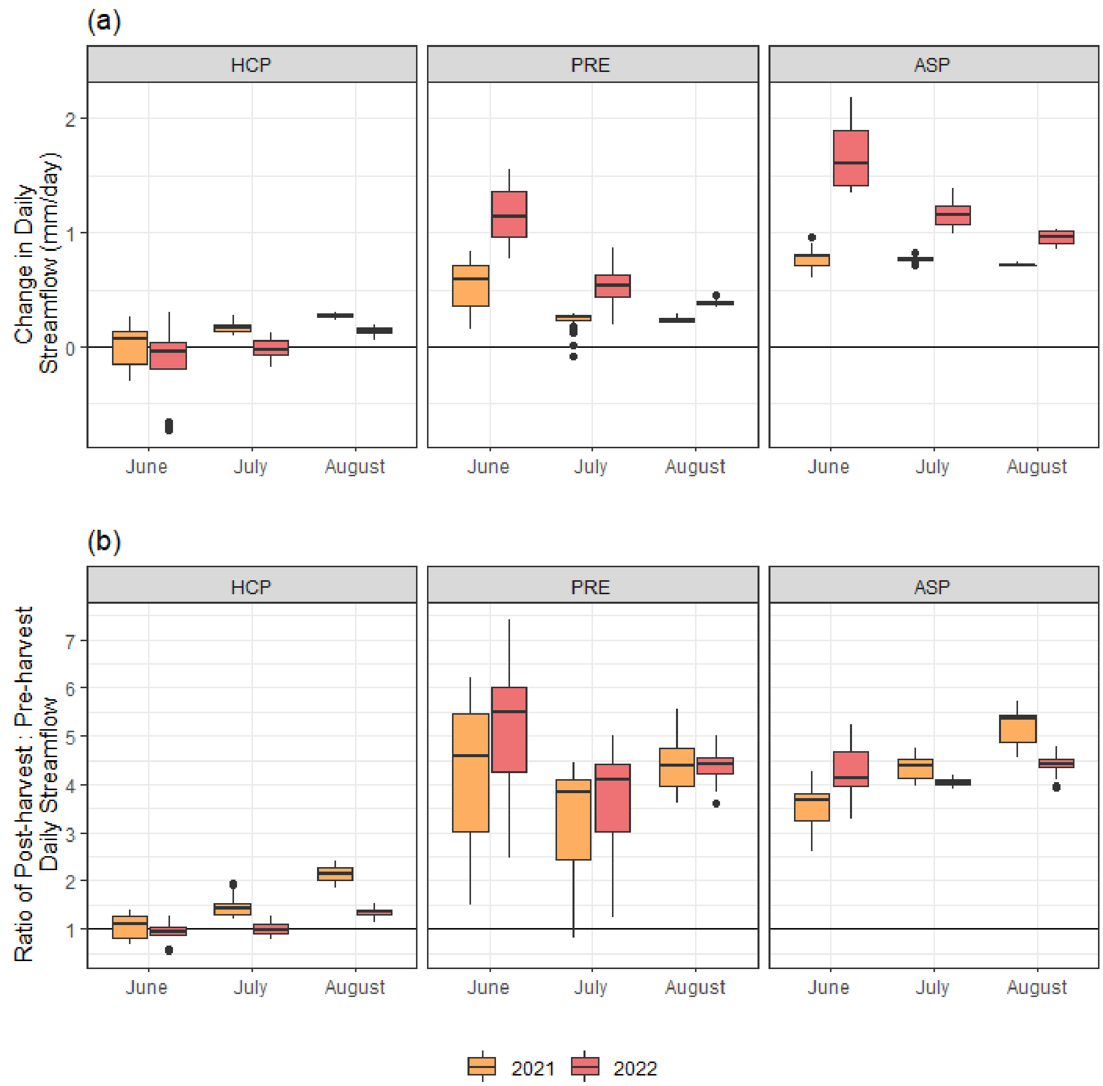

Figure 12. Changes in streamflow separated into months of the summer for each treatment. Diel streamflow is adjusted for climatic conditions. 3% of the HCP catchment was harvested, 19.1% of the PRE catchment was harvested, and 25.2% of the ASP catchment was harvested. (a) Absolute change in daily streamflow from each treatment. (b) Ratio of post-harvest to pre-harvest daily streamflow. A value of 1 indicates no change from the pre-harvest year.

The changing relationship between cumulative summer discharge in the treated and references sites compliments our results from the previous adjusted streamflow comparisons. This analysis is limited, however, as the second reference site (REF2) was harvested in between the first and second post-harvest summers.

Accordingly, we can only compare the treated streams to the first reference site (REF1). In general, we observed a greater change in the relationship between cumulative streamflow of the first reference stream and the treated sites than between the second reference stream and the treated sites. We found the greatest increase in slope in the relation for the ASP stream. In the first post-harvest summer and when compared to the first reference stream (REF1), the slope of the discharge relationship increased by 3.55 [3.47, 3.64] (523 %, $p < 0.001$), and in the second post-harvest summer, the slope increased by 2.39 [2.33, 2.46] (76.3 %, $p < 0.001$). The second largest increase in slope was in the PRE stream, where the first post-harvest summer had an increase of 1.42 [1.40, 1.45] (360%, $p < 0.001$) when compared to REF1 and the slope increased by 1.32 [1.30, 1.34] (75 %, $p < 0.001$) in the second post-harvest summer. The slope increased the least amount in the final treatment stream, HCP, and the increase was 1.19 [1.14, 1.23] (90 %, $p < 0.001$) in the first post-harvest summer and increased by 0.07 [0.04, 0.09] (5 %, $p = 0.006$) in the second post-harvest summer (Figure 13).

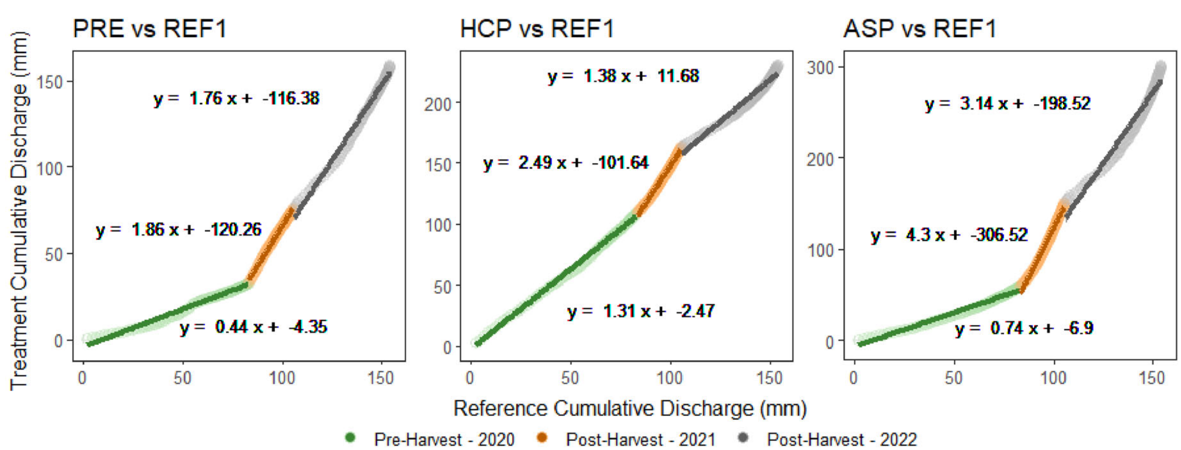

Figure 13. Cumulative summer daily streamflow relationships between the harvested and reference streams during the pre-harvest (green lines), first post-harvest summer (orange lines), and second post-harvest summer (grey lines) periods. As the REF2 stream was disturbed before the second post-harvest summer, no comparison can be made for that period.

3.4 Treatment effects on diel streamflow

The summer diel streamflow varied between 0.2 and 0.9 L s^{-1} in the pre-harvest period. In the two reference streams, REF1 had a mean diel streamflow of 0.90 ± 0.40 (SD) L s^{-1} while REF2 had a mean diel streamflow of 0.33 ± 0.17 L s^{-1}. In the treatment streams, HCP had a mean diel streamflow of 0.53 ± 0.30 L s^{-1}, ASP had a mean diel streamflow of 0.21 ± 0.08 L s^{-1}, and PRE had a pre-harvest mean diel streamflow of 0.35 ± 0.22 L s^{-1} (Figure 14).

During the first year after harvest, we observed a decrease in diel streamflow in the two reference sites and one treated stream, alongside a slight increase in the other two treated streams. In the second post-harvest year, we observed an increase in all streams, both reference and treated. However, these differences do not account for the different climatic conditions between pre- and post-harvest periods (section 3.1). In REF1, the first post-harvest summer mean diel streamflow was 0.30 ± 0.33 (SD) L s^{-1} and the second post-harvest summer diel streamflow was 1.39 ± 1.41 L s^{-1}. In REF2, the first summer mean diel streamflow was 0.24 ± 0.22 L s^{-1}. HCP had a mean diel streamflow of 0.31 ± 0.32 L s^{-1} in the first post-harvest summer and a diel streamflow of 0.81 ± 0.50 L s^{-1} in the second post-harvest summer. ASP had a mean diel streamflow of 0.27 ± 0.17 L s^{-1} in the first post-harvest summer and a diel streamflow of 0.77 ± 0.41 L s^{-1} in the second post-harvest summer, while PRE had a mean diel streamflow of 0.35 ± 0.27 L s^{-1} in the first post-harvest summer and a diel streamflow of 0.87 ± 0.58 L s^{-1} in the second post-harvest summer (Figure 14).

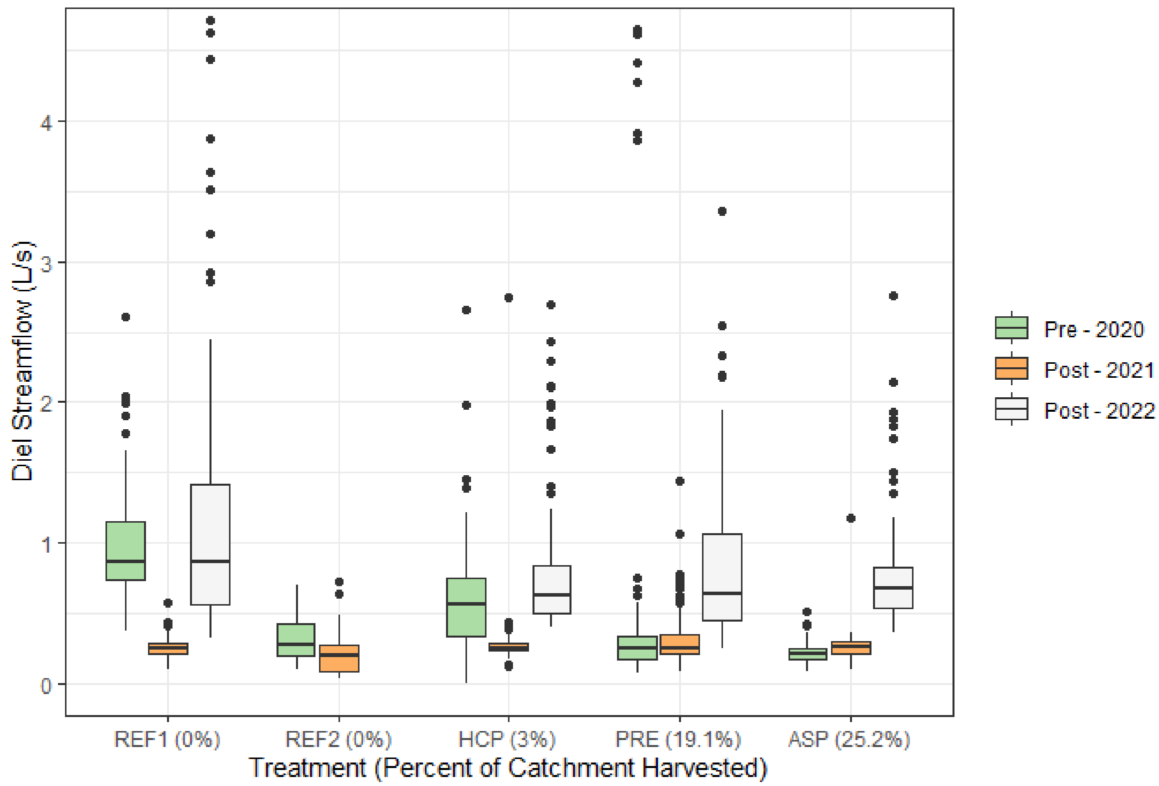

Figure 14. Boxplots of diel streamflow in each study stream during the pre-harvest and post-harvest periods in the summer months (June–August). Sites are ordered by percent of catchment harvested (listed in the x-axis labels). Displayed are the median, interquartile range (IQR) and outliers of canopy cover measurements in the six mensuration plots along each of our five study streams. Any value greater than 1.5 x IQR from the first or third quartiles is represented by a point.

After accounting for differences in climate conditions between the pre- and post-harvest years, we found statistical evidence of differences in diel streamflow after each treatment in at least one summer. The largest increase in median diel streamflow [95% CI] was in the ASP treatment, where the first post-harvest summer had an increase of 0.17 L s^{-1} [0.15, 0.18] (80 %) and the second post-harvest summer had an increase of 0.35 L s^{-1} [0.30, 0.39] (159 %). Both increases were statistically significant (2021: $W = 8334$, $p < 0.001$; 2022: $W = 8181$, $p < 0.001$). The second largest increase in median diel streamflow was detected for the PRE treatment, in which the first post-harvest median diel streamflow was 0.12 L s^{-1} [0.09, 0.15] (52 %) and the second post-harvest summer diel streamflow was 0.17 L s^{-1} [0.09, 0.28]

(74 %). These increases were also statistically significant (2021: $W = 6520$, $p < 0.001$; 2022: $W = 56665$, $p < 0.001$). We observed an increase of 0.04 L s^{-1} [0.015, 0.075] (7.5 %) in the median diel streamflow from the HCP treatment in the first post-harvest summer (W = 5333, p = 0.002), but found no statistical evidence for a change in median diel streamflow in the second post-harvest summer ($W = 3959$, $p = 0.45$) at a 95% confidence level (Figure 15.a).

The ratio of post-harvest to pre-harvest diel streamflow is additionally interesting to see. In summer 2021, the median ratio of post-harvest to pre-harvest diel streamflow [95 % CI] in the HCP treatment was 1.4 [1.2, 1.6] ($V > 4278$, $p < 0.001$), while the PRE treatment has a median ratio of 2.2 [1.8, 3.1] ($V > 4278$, $p < 0.001$), and the ASP treatment had a median ratio of 2.9 [2.7, 3.2] ($V > 4278$, $p < 0.001$). In summer 2022, the median ratio of post-harvest to pre-harvest diel streamflow in the PRE treatment was 1.5 [1.2, 1.9] ($V > 4278$, $p < 0.001$) and the median ratio in the ASP treatment was 2.3 [2.1, 2.6] ($V > 4278$, $p < 0.001$). We did not observe a ratio significantly above 1 in the HCP treatment ($V = 2280$, $p = 0.586$; Figure 15.b)

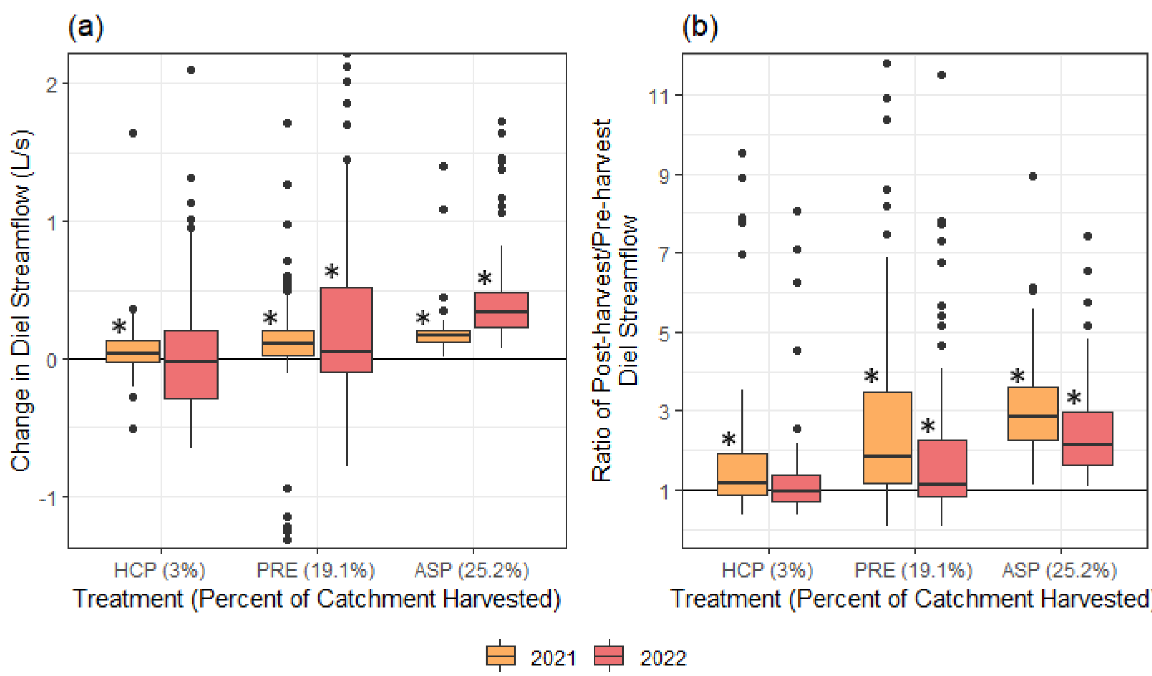

Figure 15. Change of compensated diel streamflow in treated streams following harvest. Data was compensated against weather conditions. Changes in both years

(2021 and 2022) are relative to the pre-harvest year, 2020. Asterisk indicates significance at a 95 % significance level. Sites are ordered by percentage of catchment harvested in the treatment. (a) Magnitude of change in diel streamflow. (b) Ratio of post-harvest to pre-harvest diel streamflow. A ratio of 1 indicates no change from the pre-harvest year.

When the summer diel streamflow is separated into months, a few patterns can be seen. In general, these patterns are similar to those observed in the daily streamflow, but more subdued. When looking at the absolute change in diel streamflow, we saw a steady increase through all months of the summer in the ASP stream, with the largest increase in June. The variance in the change was larger in the second post-harvest summer. The PRE treatment resulted in an increase in diel streamflow in the months of June and August, but a high variance in the month of July and a negative median diel streamflow change in the second post-harvest summer. We observed very little change in diel streamflow in June and July in the first post-harvest summer in the HCP stream but saw an increase in august. We saw a higher variance in diel streamflow across all three months in the second post-harvest year and this variance decreased as the summer progressed. The median diel streamflow varied from increases in June and August to a decrease in July (Figure 16.a).

In the HCP treated stream, the ratio of post-harvest to pre-harvest diel streamflow followed the same pattern as the absolute change in diel streamflow. However, we observed a decrease in the variance of the ratio of post- to pre-harvest diel streamflow in the PRE treated stream as the summer progressed. In July, we saw a streamflow ratio less than 1 in the PRE stream, indicating a median decrease in diel streamflow. We observed an increase in diel streamflow each month of the summer, with a larger increase in the ratio of diel streamflow in the first post-harvest year when compared to the second-post harvest year in the ASP stream. This is the opposite of the absolute increase in diel streamflow, where we saw a consistently higher increase in diel streamflow in the second post-harvest year (Figure 16.b).

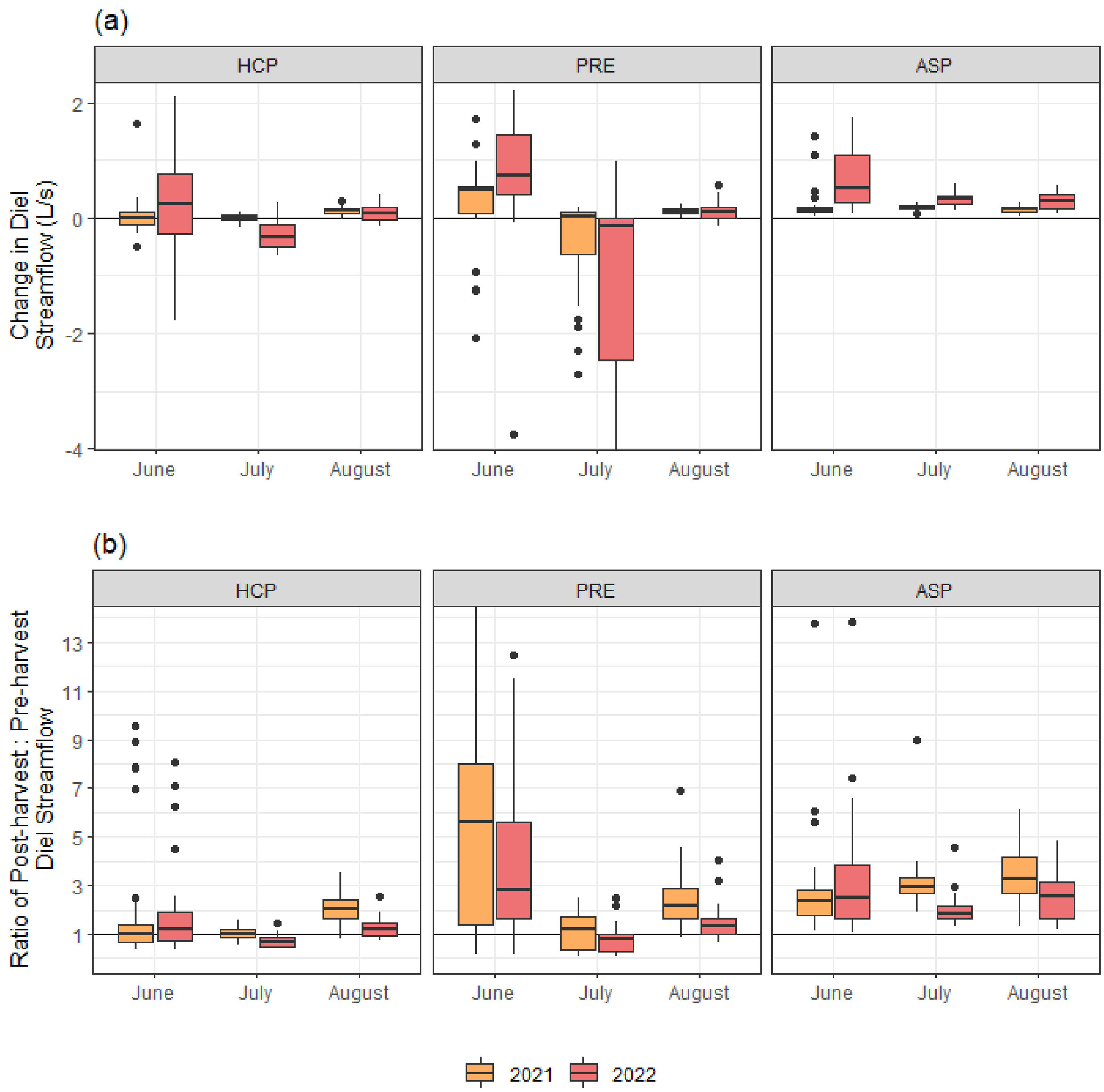

Figure 16. Changes in diel streamflow range separated into months of the summer for each treatment. Diel streamflow is adjusted for climatic conditions. 3% of the HCP catchment was harvested, 19.1% of the PRE catchment was harvested, and 25.2% of the ASP catchment was harvested. (a) Absolute change in diel streamflow from each treatment. (b) Ratio of post-harvest to pre-harvest diel streamflow. A ratio value of 1 indicates no change from the pre-harvest year.

4 Discussion

4.1 Daily Streamflow

Our results indicated a general increase in streamflow following all three harvests and riparian treatments, consistent with a number of previous studies reporting an increase in streamflow directly following harvest (Abdelnour et al., 2011; Harr, 1983; Moore et al., 2020; Rothacher, 1970; Segura et al., 2020). Interestingly, the increase in daily streamflow in each stream did not follow with the amount of riparian canopy cover removed as we predicted. We expected the removal of more vegetation in the riparian zone would correspond to a larger reduction in transpiration and interception, which would cause an greater increase in daily streamflow (Grant et al., 2008; Moore et al., 2020). Counter to our expectations however, the riparian prescription with the largest amount of canopy removed (PRE) resulted in the second largest increase in streamflow (306 %), while the lowest removal of canopy (ASP) resulted in the largest increase in streamflow (339 %). The more moderate riparian treatment (HCP) resulted in a much lower streamflow increase of 41 % (Figure 17). It is important to remember that we only observed two post-harvest years, and we would expect these increases in streamflow to change dramatically after a few years following harvest, as seen in previous research (Gronsdahl et al., 2019; Moore et al., 2020; Perry and Jones, 2017; Segura et al., 2020).

While we did not see the expected streamflow increases correspond to riparian harvest intensity, we did see a close pattern between the percent area harvested in the clearcut harvest and the streamflow increase. Multiple previous studies have found similar results, showing an increase in streamflow with higher percentages of the catchment being harvested. Our results support these conclusions, as the stream with the largest proportion of the catchment harvested (ASP) had the greatest increase in streamflow and the stream with the smallest harvest (HCP) also has the least increase in streamflow. Interestingly, we found a statistically significant increase in summer streamflow during both post-harvest years in our HCP stream despite only 3.3 % of the catchment being harvested, while previous studies have indicated a threshold of noticeable streamflow increase around 20 % of the catchment being harvested (Bosch and Hewlett, 1982; Stednick, 1996). Most previous studies with similar harvest areas observed normalized summer streamflow increases after harvest of between 0.1 mm and 1.1 mm, and our results are within this range (0.74 mm from ASP, 0.27 mm from PRE, and 0.20 mm from HCP in 2021; Stednick, 2008, 1996). However, since we focused on low summer streamflow, this ends up being a larger percentage increase when compared to annual streamflow increase, as most past studies have focused on (Moore et al., 2020). Overall, our results show that the percentage of the catchment harvested has a larger impact on the daily streamflow increase than the intensity of the riparian area treatment.

In precipitation-dominated locations, the consensus cause of summer streamflow increases after harvest is the reduction in evapotranspiration following vegetation removal. The removal of overstory and understory vegetation reduces interception of precipitation (Link et al., 2004; Sollins et al., 1980) and reduces the removal of water from soil moisture storage through transpiration (Bearup et al., 2014; Bond et al., 2002; Maxwell and Condon, 2016), thereby increasing the amount of water that flows through sub-surface paths and reaches the stream (Moore et al., 2020). Our results support this explanation, as the largest timber harvests removed the largest amount of transpiring vegetation (Figure 18). Additionally, we found the largest increase in streamflow after the only timber harvest performed on a south-facing slope. As these sites are located in the northern hemisphere, the increased incident solar radiation on south-facing slopes makes them more susceptible to the effects of processes driven by solar radiation (Tian et al., 2001). As ET rates are partially driven by the amount of solar radiation, we expect south-facing slopes to have higher average ET and therefore a larger increase in streamflow following the removal of vegetation during harvest.

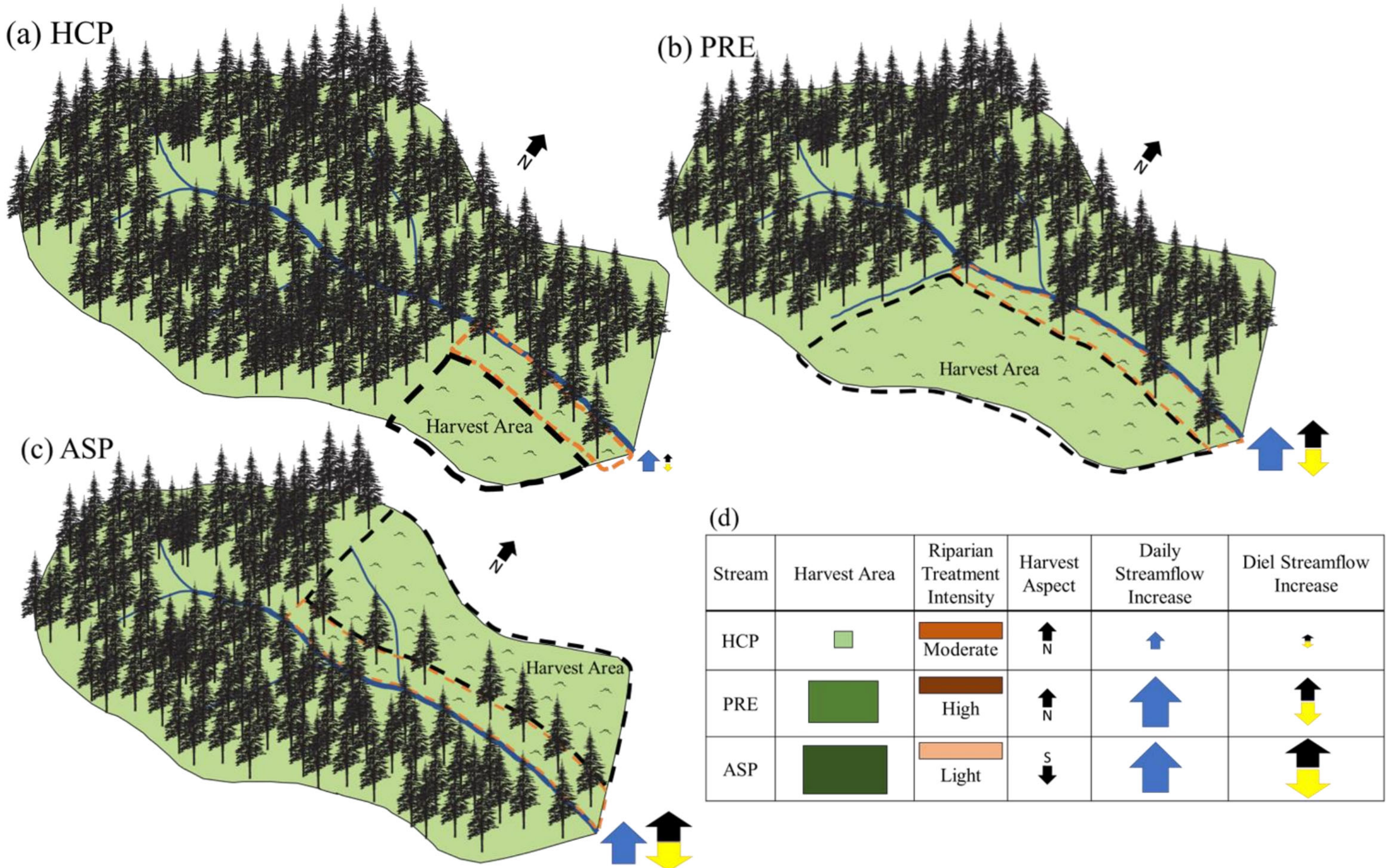

Figure 17. Graphical summary of results. (a) The results of the HCP treatment, where a small harvest area and moderate intensity riparian treatment caused a small increase in both daily streamflow and diel streamflow. (b) The results of the PRE treatment, where a large harvest and high intensity riparian treatment resulted in a large streamflow increase and a moderate increase in diel streamflow. (c) The results of the ASP treatment, where a large harvest area and light intensity riparian treatment resulted in a large increase in both daily streamflow and diel streamflow. The south aspect of the timber harvest also contributes to the large increases. (d) Table summarizing the results of the three treatments.

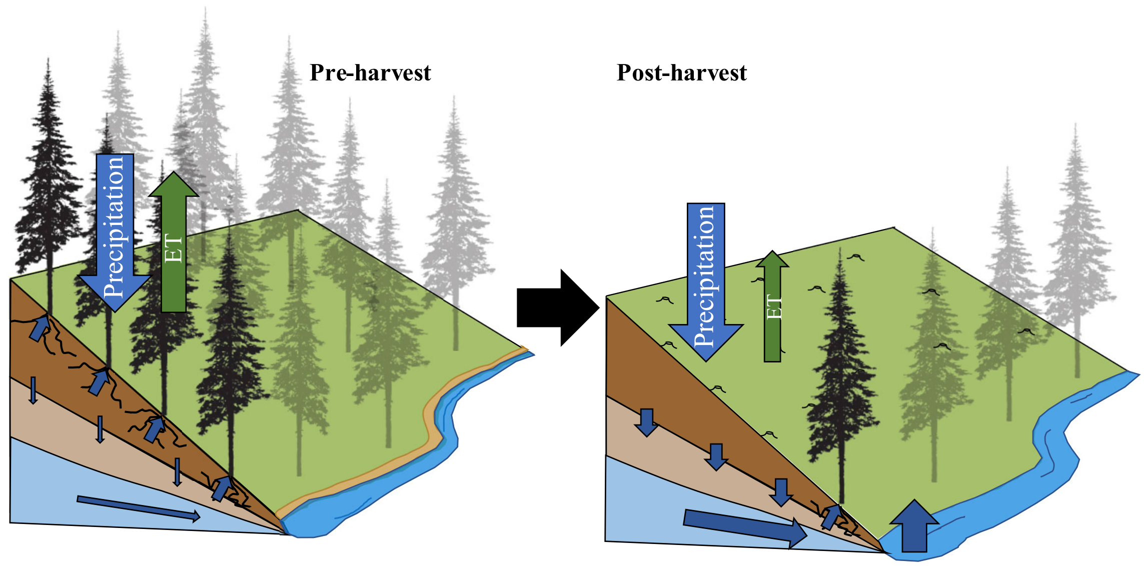

Figure 18. Graphical explanation of the cause of daily streamflow increase. Timber harvests decrease the total evapotranspiration in the catchment, so the same amount of precipitation during the spring and summer results in more water reaching the stream following a timber harvest, increasing streamflow. The aspect of the harvest may also influence the increase in streamflow, as south facing slopes may have higher evapotranspiration rates pre-harvest, resulting in a larger increase following harvest.

The increase in summer streamflow we observed is especially important for headwater streams in this region, as the warm, dry summers can result in reduced habitat area for aquatic and riparian species in the later stages of the summer (Dieterich and Anderson, 2000). An increase in streamflow could potentially bring with it a decrease in average stream temperature, as a larger thermal mass required more energy to heat. However, reduced shade, increases soil temperature, and increased wind velocity from timber harvests may negate or reverse these effects (Bladon et al., 2016; Segura et al., 2020). If stream water flow is increased without other negative impacts, this most vulnerable time for headwater aquatic ecosystems may be improved. Additionally, human population centers are also most vulnerable to water scarcity in the dry season, and increased streamflow may provide more water security to downstream human populations (Griffin and Chang, 1991). This increased water yield may become more valuable as communities face water scarcity challenges from continued development and climate change (Hall, 2003). However, these potential benefits only are realized if the other effects of timber harvests are sufficiently mitigated.

When assessing the change in streamflow found in our results, it is essential to recall the time frame of our study and the scale of the impacts. We measured one year of pre-harvest data and compared it to two years of post-harvest data. This time frame only captures the removal of vegetation and does not capture the regrowth process of understory and overstory vegetation. We were not able to assess how quickly the treated streams rebound from the disturbance and how the streamflow. Previous research has indicated streamflow may decrease after few years from harvesting due to the high transpiration rate of rapidly regrowing trees, especially in replanted stands. (Coble et al., 2020; Perry and Jones, 2017; Segura et al., 2020). A lack of information about the progression of streamflow reductions on buffered catchments makes this difficult to predict (Chellaiah and Kuglerová, 2021; Kuglerová et al., 2014). This pattern may be present in our streams, but our limited study period did not capture this change. When we attempt to expand our observed increases in streamflow to a larger scale catchment, it becomes less clear if these increases are significant. Typical large catchments in the study area are mosaics of harvested, regrowing, and reserve stands, at a variety of ages and growing stages. Even if all harvested streams yield more streamflow for the first few years following harvest, the net water yield of a mix of newly harvested, regrowing, and mature stands will likely dampen any large-scale effect of timber harvests (Moore et al., 2020).

4.2 Diel Streamflow

The summer diel streamflow we observed pre-harvest was similar to other studies in forested headwater catchments (Bond et al., 2002; Kobayashi et al., 1999). Our results showed an increase in diel streamflow range across all treatments following harvest and riparian treatment, although a smaller proportional increase than what was seen in the daily streamflow. Both the PRE and ASP treatments increased diel streamflow range by a similar amount, with the HCP treatment not consistently showing a statistically significant increase. While we observed daily streamflow increase to closely correspond to the amount of the catchment harvested, we observed a more limited, but still present, relationship between the percent of catchment harvested and the diel streamflow. We observed a similar relationship between the intensity of riparian harvest and the increase in diel streamflow, with the least intensive riparian treatment (PRE) resulting in the largest increase in diel streamflow and the most intense riparian treatment resulting in a smaller increase.

The dominant source of diel fluctuations in our sites is likely the withdrawal of water in the daytime through evapotranspiration. This is due to the drivers of other diel fluctuation processes not being present or as powerful in these sites; there was no snowpack in these sites and the relatively small range of daily temperature limits changes in water viscosity and thermal expansion (Caine, 1992; Czikowsky and Fitzjarrald, 2004; Gribovszki et al., 2010; Motoyama et al., 1986; Schwab et al., 2016). Our results indicate that in these sites, the applied timber harvests increase the riparian vegetation transpiration and in turn increase the range of streamflow rates during each day of the summer. Our hypothesized explanation for this effect is that removal of hillslope vegetation increases the sunlight, air temperature, and water availability for near-stream vegetation which increases transpiration (Boggs et al., 2015; Guenther et al., 2012; Figure 19). While hillslope vegetation transpiration is reduced, the vegetation on the hillslope is not connected to the stream and does not contribute to the diel signals (Barnard et al., 2010; Bren, 1997; Moore et al., 2011). Our results agree with other studies in similar sites which showed adjacent riparian vegetation dominates summer diel signals and hillslope vegetation has limited effects (Bond et al., 2002; Bren, 1997; Kobayashi et al., 1999). These results support our hypothesis that timber harvests increase the sunlight and wind on riparian vegetation, increasing the transpiration in the controlling areas for diel streamflow fluctuations (Boggs et al., 2015; Figure 19). We also support our results and hypothesis by observing the largest increase in diel streamflow in the only treated stream on a south-facing slope, with the highest average solar radiation (Tian et al., 2001). Additionally, this explains why increased harvest areas, also increase the diel fluctuations, as a higher percent of the catchment being harvested increased the amount of the riparian area exposed to increased solar radiation and higher air temperatures, increasing transpiration, and thereby increasing diel streamflow range. It is important to acknowledge that other studies have found counter results, with the hillslope vegetation exhibiting dominant control over the diel streamflow fluctuations (Barnard et al., 2010; Moore et al., 2011). It may be that our steep catchments and narrow stream valleys had deeper subsurface water flow paths and limited the effect of hillslope vegetation on diel signals, or that the dry summers disconnect hillslope vegetation from subsurface flow paths, so this may not be consistent across all headwater streams (Tague and Grant, 2004).

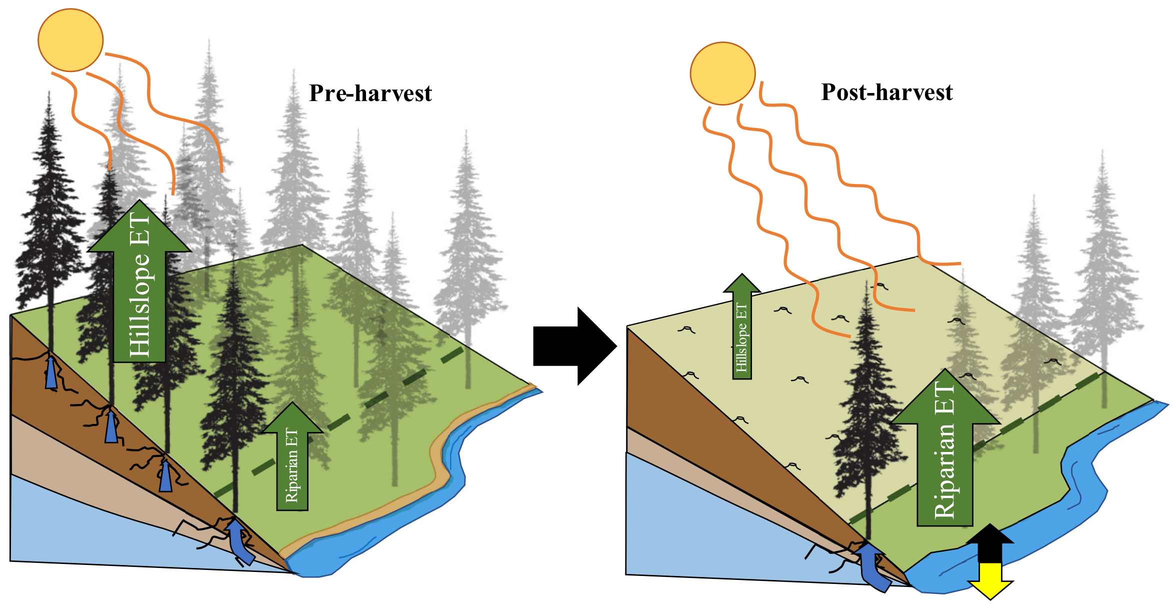

Figure 19. Graphical explanation for the cause of increased diel streamflow following timber harvests. Harvests increase the resource availability of riparian vegetation, which increases transpiration. This increase in riparian transpiration increases the diel streamflow range, and the reduction in hillslope transpiration does not reduce the diel signals as the hillslope vegetation is disconnected from the stream.

It is essential to again consider the time frame and scale of our results. Our limited post-harvest measurement seasons limits our results to two years following harvest. We do not have sufficient information to make conclusion about how this may change as hillslope vegetation regrows or riparian vegetation restores full canopy cover. We would expect the same amount of transpiration to increase the relative amplitude of the diel signals as total daily streamflow decreases from regrowing vegetation, however Bren (1997) found that the amplitude of diel signals increases with increased flow rates, potentially counteracting this effect. When looking at a larger scale, it is unlikely these diel signals carry beyond the headwater stream as the signal becomes mixed with other headwater streams and the signals interfere with each other (Barnard et al., 2010; Graham et al., 2013). We do not expect diel signals to have a significant impact on downstream water quantity or aquatic ecosystem health.

4.3 Potential Confounding Factors

Whether by limitations in study design, logistics of implementing the study, or by random circumstance, there were a few confounding factors that affected our results: treatment application, study period weather patterns, and site differences. As shown in the riparian canopy measurements, we measured only a portion of the canopy removal that was performed at the sites. The foresters and loggers who implemented the riparian treatments as part of the clearcut harvests on the above hillslopes did not remove trees evenly along the length and width of the riparian buffer, but instead marked and removed patches of trees along the uphill edge of the buffer to result in an average canopy cover that fulfilled the treatments. Our measurement method of centered plots along the length of the riparian buffer did not adequately capture the full amount of vegetation removed. This uneven removal of canopy cover could have influenced the effect of the riparian vegetation on the streamflow, as the vegetation could not make best use of available water and sunlight, potentially resulting in lower transpiration that an evenly harvested riparian buffer.

A second potential confounding factor is the weather that occurred during the study period. The pre-harvest year (2020) was an approximately average year in terms of temperature and precipitation, but the first post-harvest year (2021) was a significant drought year (NOAA National Centers for Environmental Information, 2021), with around half of the spring and summer precipitation as the pre-harvest year, and in the second post-harvest year (2022) our study sites received a slightly above average amount of precipitation (CNRFC, 2022). This required us to compensate for the dramatic difference in precipitation, which introduced a confounding factor. Our compensation did not account for potential changes in runoff processes caused by the amount of water in the hillslopes (Singh et al., 2021; Wu et al., 2020). We are unable to say if the streamflow in the treated sites would have followed the same patterns in a post-harvest year with average precipitation.

However, as we observed similar patterns in both post-harvest summers with opposingly low and high precipitation, we expect similar patterns to occur in a similar range of precipitation

A third potential confounding factor is the difference in site topography, tree species composition, and especially aspect. Each site had slightly different slopes and stream gradients but were quite comparable. Streams REF1, REF2, and ASP each had one section of the reach length (~100 m) that was dominated by red alder instead of the usual redwood and Douglas-fir, which may have influenced the water use of the riparian buffers (Francis et al., 2020). The most likely impactful, however, was the direction the harvested stands faced. The harvests at streams HCP and PRE were both on the north-facing slope, while the ASP treatment was applied to the south-facing slope. The aspect of the harvested area, as described earlier, has been seen to be impactful in previous studies and we have evidence for the effects in our results (Tian et al., 2001). Because of the lack of treatment repetition on hillslopes with various aspects, we cannot be sure of the effects of aspect, but our results provide some evidence of the importance of its effects on streamflow.

A fourth confounding factor was the harvesting of site REF2 between the two post-harvest years. For the first post-harvest year, we used both reference streams to quantify the effects of the variable weather conditions, but we were only able to use stream REF1 during the second post-harvest summer. Ensuring the consistency of our reference streams would give us more confidence in our conclusions about the effects of timber harvest and riparian treatment while accounting for climatic conditions.

4.4 Management Implications

By understanding the effects of human forest operations on streamflow, we can implement practices that protect natural resources and habitat while continuing to use needed forest products. Riparian buffers have a long history in the western United States and have been adapted many times over the years for increased stream protection (Clinton, 2011; Cole et al., 2020). Our results show that both riparian areas and hillslope areas affect streamflow, and as such riparian buffers alone cannot prevent all effects from timber harvest on summer low flow. Forest managers and regulators can use such results to help inform comprehensive forest management practices and requirements that best protect the health of riparian and aquatic systems.

If the goal of riparian buffers is to prevent or limit the impact of forest operations on streamflow, then it appears riparian buffers alone do not prevent all the effects of timber harvests on water quantity, at least in summer months in this region. However, if riparian buffers are intended to maintain aquatic and riparian habitat after forest operations, an increase in streamflow may improve habitat, especially in dry summer months. If factors of water quality (temperature, pollutants, dissolved oxygen, nutrients, carried sediments) are adequately protected by riparian buffers, this net increase in streamflow may be a benefit to aquatic health for the first few years after harvest. Some studies have shown sufficient protection from riparian buffers, so this may be the case in some locations and systems (Anderson and Poage, 2014; Clinton, 2011; Cristan et al., 2016; Rykken et al., 2006). Other studies have not found complete protection of water quality from riparian buffers, and in these situations an increase in streamflow may not result in an overall improvement after harvest (Anderson and Poage, 2014; Philip Lee et al., 2004; Marczak et al., 2010; Moring, 1982). Overall, there are mixed results assessing the effectiveness of riparian buffers, and our results show that small differences in riparian buffer design do not appear to have a significant impact on the streamflow in these sites.

Although identified for many decades, it is unclear what the effects of diel streamflow fluctuations on aquatic and riparian ecosystem health are, but we can assume an increase in diel streamflow range would be a detriment to habitat and ecosystem health in most locations, especially during the dry season (Godwin, 1931). Reduction in streamflow during the day temporarily reduces the streamflow and provides the same effects on ecosystem and habitat as a reduction in daily streamflow, but for a limited time. This draw-down in streamflow may reduce the area of aquatic habitat and create separation between pools in step-pool stream systems (Dieterich and Anderson, 2000; Meyer et al., 1999; Montgomery and Buffington, 1997). However, we observed this increase in diel fluctuations at the same time as a general increase in daily streamflow caused by the timber harvest. Increased diel signals likely have a limited effect when occurring at the same time as an increase in streamflow to replace or exceed any daytime reduction caused by increased transpiration rates in riparian vegetation. Additionally, diel signals do not usually carry downstream past confluences, as the diel fluctuations from multiple headwater streams can interfere with each other (Barnard et al., 2010; Graham et al., 2013).

Based on the very small difference in effect we observed from different riparian treatments and the lack of relationship between intensity of riparian treatment and the increase in streamflow, it appears factors other than riparian treatment have a larger effect on streamflow. Our results show that the amount of the catchment harvests has a much larger impact, and previous research has shown that other factors of the timber harvest, including harvested area, aspect, and species composition, may have a larger effect on the effectiveness of riparian buffers than the amount of canopy cover removed (Jyväsjärvi et al., 2020; Marczak et al., 2010; Moore et al., 2020; Stednick, 1996). Given the high degree of variability in environmental factors between individual streams and the seeming importance of other factors besides canopy closure, our results support the need for a more varied and tailored approach to riparian buffer implementation, similar to the approach laid out by Haberstock et al. in their paper on riparian buffer width for effective protection of fish habitat (Goates et al., 2007; Haberstock et al., 2000). Our results show that reducing riparian harvest and increasing buffer density do not necessarily translate into reduced harvest effects on streamflow. Designing riparian buffers around one metric, often canopy cover or width, without consideration of vegetation type and structure, topography, and aspect will likely not be the most optimal method to limit the effects of forest operations on streamflow (Chellaiah and Kuglerová, 2021; Cole et al., 2020; Jyväsjärvi et al., 2020; Marczak et al., 2010; Walter et al., 2009). Developments in stream and riparian area monitoring (Kluber et al., 2009; Marquardt et al., 2012; Olson et al., 2007) and GIS tools (Reeves et al., 2018) have improved the feasibility of site-specific riparian management and region specific planning to tailor timber harvests and riparian treatments to individual streams or at least to regional watersheds (Anderson and Poage, 2014; Kuglerová et al., 2014).

Furthermore, as shown in our problems accurately measuring the amount of canopy cover removed by the riparian treatments, riparian buffer design may want to assume a ‘worst case scenario’ of vegetation removal when prescribing a certain amount of allowed removal. In the timber harvests we observed, vegetation removal in the riparian area was not evenly distributed along the riparian area but clumped in areas of easy access for logging equipment. This uneven pattern of vegetation removal is likely unintended in the riparian buffer design and may reduce the effective width of the riparian buffer and increase the effects of the timber harvest (McDermott et al., 2009). Our observations of typical riparian buffer implementation suggest that riparian buffers should be designed with practical implementation and potential deviations from intended implementation in mind. Confirming regulated riparian buffer implementation is difficult (Claggett et al., 2010), so riparian buffer design should be done carefully to account for unintended implementations.

5 Conclusion

Due to the vital role headwater streams play in ecosystems and the recent and planned increases in riparian buffer treatment protections in the western United States, it is essential that we understand the effects forest harvests have on headwater streams. Land managers need increased knowledge to inform their management plans and regulators need knowledge of harvesting effects to regulate efficient protections for aquatic systems. The objective of this research was to increase understanding of the effects forest harvesting and riparian buffer treatments had on one aspect of aquatic systems: summer streamflow. We assessed these effects by measuring streamflow and weather conditions in three treated stream and two reference streams for one pre-harvest summer and two post-harvest summer located on industrial timberland in the coast range of Northern California. Our results showed that there can be large increases in streamflow in the years directly following a timber harvest, but these increases more closely relate to the amount of the catchment harvested with less effect from the different riparian buffer designs we tested. We also found that diel fluctuations in streamflow are mainly driven by riparian vegetation and multiple drivers of this process are present in these sites, including harvest area and riparian buffer intensity. These results add to past research and support many previously found results while providing some new insight into the processes behind diel streamflow and riparian buffer effects.

As in any study, there were limitation in our study design and logistical constraints that affected our results:

- Our study involved one stream for each treatment. While the chosen streams were very similar in average hillslope, stream gradient, vegetation species composition, elevation, soil type, and topographic position, there are always small differences between sites, and we were not able to observe the below-ground geology, soil depth, and flow paths present at these sites. This limited our ability to confirm our hypotheses for the processes behind the effects we observed. Soil moisture storage or groundwater data may have allowed us to track hillslope evapotranspiration signals through the subsurface water to the streams.

- The length of our study is essential to keep in mind when interpreting our results. Our study observed streamflow for two post-harvest years, and previous research has shown a return to pre-harvest streamflow, and sometimes a decline in streamflow, after a few years following harvest (Gronsdahl et al., 2019; Moore et al., 2020; Perry and Jones, 2017; Segura et al., 2020). We cannot make statements about this transition in these sites due to our limited study period, as the rate of regrowth of hillslope vegetation could dramatically change the time it takes to return to pre-harvest streamflow.
- Our study only observed perennial headwater streams. Many of the smaller streams, including other streams in this area, are not perennial and dry up at some point through the dry season. The permanence of headwater streams and the time at which they may dry up may also affected by timber harvests and they may respond differently to different riparian buffers. While we did not observe a large effect of riparian buffers on streamflow in our study streams, there may be a significant effect of the riparian buffer treatment on streamflow permanence.

There are still many future directions of research surround the topics of this research. In order for land managers and regulator to make informed decisions regarding headwater stream management and riparian buffers, a variety of research projects must add to the depth and breadth of our collective knowledge regarding these topics. There are multiple potential areas of inquiry, including the following:

- This research looked solely at the effects of timber harvests and riparian buffer treatments on streamflow, but this in an incomplete and independently unless measure of overall stream health. Plenty of past research has focused on induvial aspects of low flow stream water quality and health, but future research should focus on collating all aspects of stream health with indicator species to assess effects of timber harvests on stream health as a whole.

- In our study observed the changes in streamflow following timber harvest and riparian buffer treatments, we made some assumptions about subsurface water flow processes and the linkages between transpiration, sap flow, soil moisture, groundwater, and streamflow in these sites. While our assumptions are based on past research (Boggs et al., 2015; Moore et al., 2011; Singh et al., 2021), we still need to develop considerable understanding about subsurface water processes, especially in headwater systems.
- Past research has shown the location of vegetation that drives diel fluctuations can change depending on site. The diel signals in some streams are dominated by both riparian and hillslope vegetation (Barnard et al., 2010; Burt, 1979; Graham et al., 2013; Moore et al., 2011), while our results agree with other studies that have found riparian vegetation alone to control the diel signals (Barnard et al., 2010; Bren, 1997; Moore et al., 2011). Further research may determine what aspects of a stream may decide which vegetation controls the diel fluctuations.
- Our study assessed the effects of a variety of canopy cover defined riparian buffer treatments, but comparisons between other design methods, such as riparian buffers defined by basal area removal, fixed width buffers, or specific tree targeting buffers, may help indicated optimal riparian buffer design and further inform regulators about which protections are most effective while still allowing for resource extraction.
- As alternatives to clear cut harvests are being implemented more frequently, especially in forested areas where resource extraction is not the primary objective, it is important to understand how these alternative types of harvest impact streamflow and how riparian buffers interact differently with these types of harvest.
- While we observed a large increase in daily streamflow following harvest in the headwater catchment, it is not as clear how these increases would travel downstream to larger streams and rivers. Some previous research has assess the result of larger disturbances on streamflow (Moore et al., 2020), but it is still unclear how the combination of many small headwater timber harvests

combine to impact the larger rivers downstream. As many of these systems have been subject to regular harvest through the past century, it may be difficult to find an adequate control system to compare to.

- Much of the historical research assessing timber harvest effects on streamflow and riparian buffer effectiveness have been located in the western North America, although recent research has expanded into other regions (Deutscher and Kupec, 2014; Iroumé et al., 2006; Kuglerová et al., 2014). A common observation from riparian research is that effects from timber harvest are often very site dependent, and expanding research, even replicative research, into new environments and regions will increase our understanding of the processes behind these effects as well as providing more localized knowledge for forest regulators and managers.

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Appendix A: Addendum figures

Figure A.1. HCP treatment hemispherical photos of the canopy used in the quantification of canopy cover. The left photo is of the pre-harvest riparian canopy and the right photo is of the post-harvest canopy.

Figure A.2. PRE treatment hemispherical photos of the canopy used in the quantification of canopy cover. The left photo is of the pre-harvest riparian canopy and the right photo is of the post-harvest canopy.

Figure A.3. Ratio of post- to pre-harvest streamflow in each treated stream each day through the summer. (a) Ratio of post- to pre-harvest total daily streamflow. (b) Ratio of post- to pre-harvest diel streamflow.

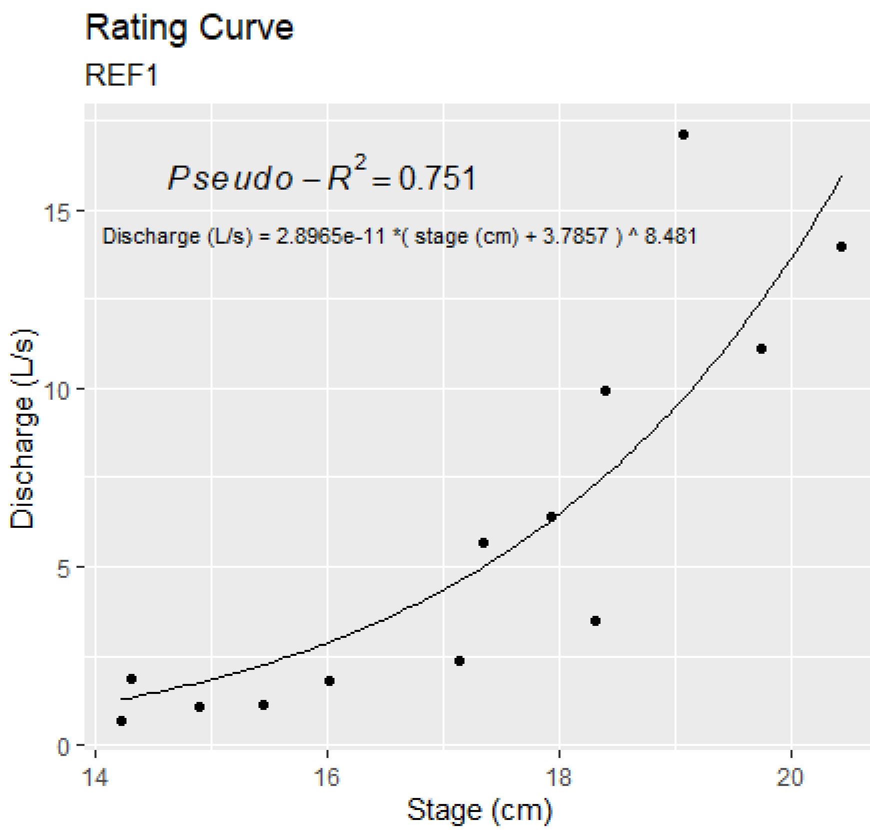

Figure A.4. Rating curve of stream REF1. Discharge plotted against stage allows us to create a regression that we can use to convert continuous stage measurements into discharge estimates.

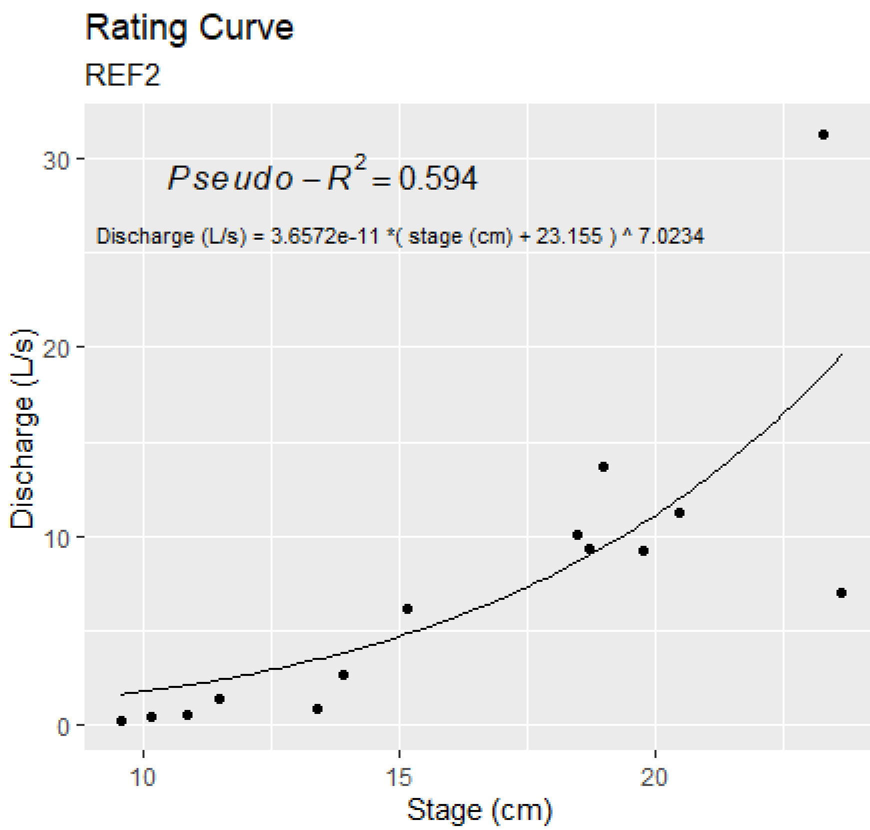

Figure A.5. Rating curve of stream REF2. Discharge plotted against stage allows us to create a regression that we can use to convert continuous stage measurements into discharge estimates.

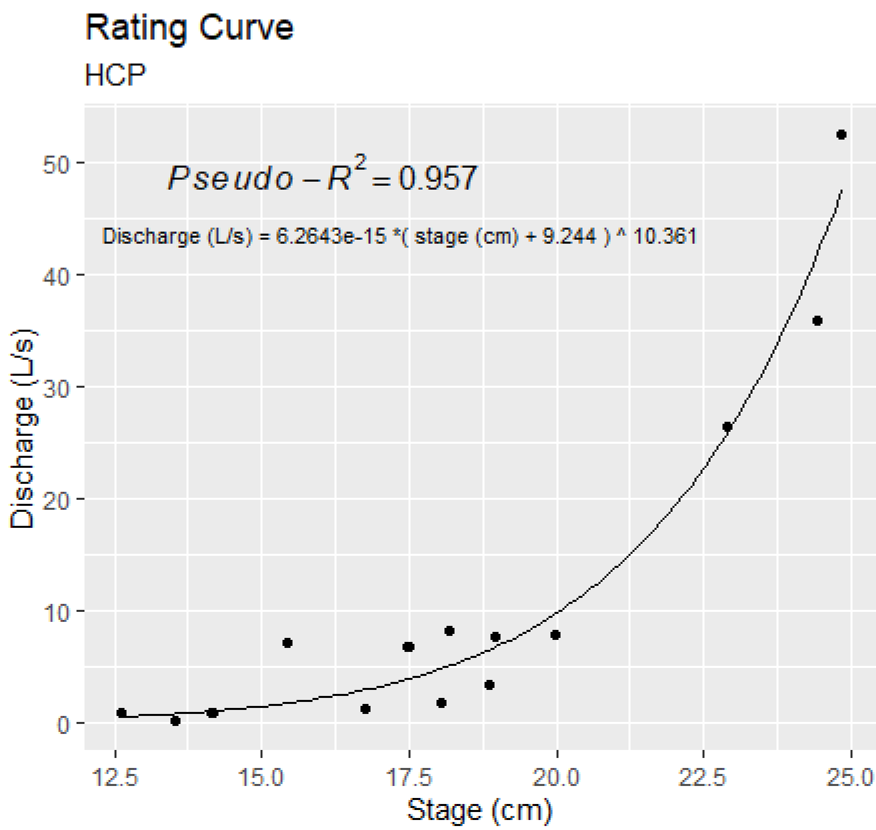

Figure A.6. Rating curve of stream HCP. Discharge plotted against stage allows us to create a regression that we can use to convert continuous stage measurements into discharge estimates.

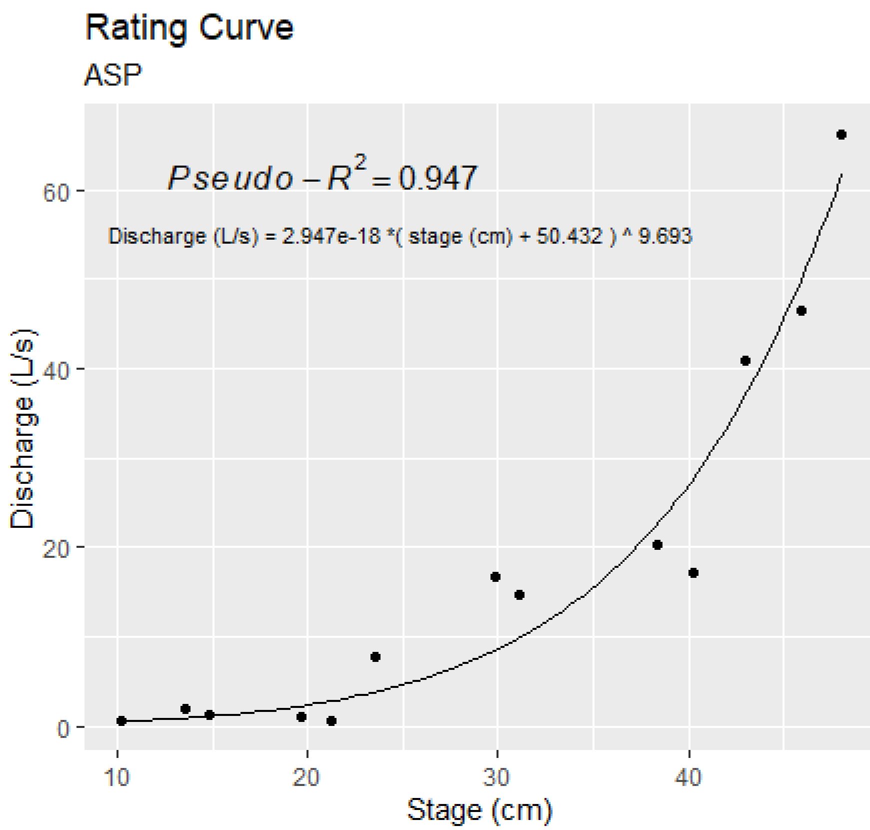

Figure A.7. Rating curve of stream ASP. Discharge plotted against stage allows us to create a regression that we can use to convert continuous stage measurements into discharge estimates.

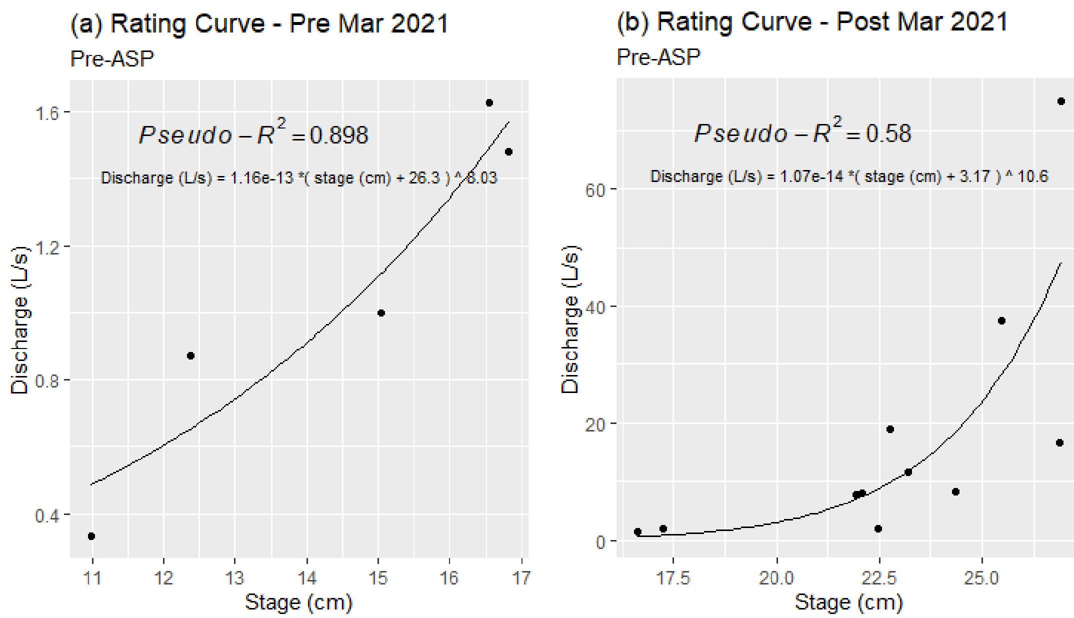

Figure A.8. Rating curves of stream PRE. Discharge plotted against stage allows us to create a regression that we can use to convert continuous stage measurements into discharge estimates. (a) Rating curve for the PRE stream for before March 2021. (b) Rating curve for the PRE stream for after March 2021. Two rating curves were required for stream PRE due to a stream cross section change during March 2021. A unique rating curve was needed for both the pre-March 2021 and post-March 2021 periods. Only 5 discharge measurements were gathered before March 2021, so the rating curve for PRE during that period is very limited.

Appendix B: Alternative models for compensating streamflow data

Daily Streamflow:

When constructing a model for reference daily streamflow, both pre- and post-harvest, we tested a few potential variables in the model, including mean daily air temperature, maximum daily air temperature, daily PET estimate, and daily precipitation. In this case, maximum daily air temperature was the best fitting model. In every model tested, the month of the summer (as an indicator variable) was a very powerful variable.

Used model:

$$\mu\{Streamflow|Max\ Air\ Temp\} = \beta_0 + \beta_1 * T_{a.max} + \beta_2 * I.July + \beta_3 * I.August + \varepsilon$$

AIC = 51.37 R^2 model = 0.669 R^2 Mean Air Temp variable= 0.190

Tested alternative models:

$$\mu\{Streamflow|Mean\ Air\ Temp\} = \beta_0 + \beta_1 * T_{a.mean} + \beta_2 * I.July + \beta_3 * I.August + \varepsilon$$

AIC = 58.20 R^2 model = 0.654 R^2 Mean Air Temp variable= 0.152

$$\mu\{Streamflow|PET\} = \beta_0 + \beta_1 * PET + \beta_2 * I.July + \beta_3 * I.August + \varepsilon$$

AIC = 76.64 R^2 model = 0.610 R^2 PET variable = 0.005

$$\mu\{Streamflow|Precipitaion\} = \beta_0 + \beta_1 * Precip + \beta_2 * I.July + \beta_3 * I.August + \varepsilon$$

AIC = 84.58 R^2 model = 0.594 R^2 Precip variable= 0.005

Diel Streamflow:

When constructing the model for reference diel streamflow, we tested the same potential factors as in the daily streamflow model. In this case, daily PET estimate was the best fitting factor. As with the daily streamflow, the month of the summer was always a very powerful variable.

Used model:

$$\mu\{Diel\ Streamflow|PET\} = \beta_0 + \beta_1 * PET + \beta_2 * I.July + \beta_3 * I.August + \varepsilon$$

AIC = 58.21 R^2 model = 0.482 R^2 PET variable = 0.151

Tested alternative models:

$$\mu\{Diel\ Streamflow|Max\ Air\ Temp\} = \beta_0 + \beta_1 * T_{a.max} + \beta_2 * I.July + \beta_3 * I.August + \varepsilon$$

AIC = 67.06 R^2 model = 0.443 R^2 Max Air Temp variable = 0.087

$$\mu\{Diel\ Streamflow|Mean\ Air\ Temp\} = \beta_0 + \beta_1 * T_{a.mean} + \beta_2 * I.July + \beta_3 * I.August + \varepsilon$$

AIC = 68.24 R^2 model = 0.433 R^2 Mean Air Temp variable = 0.070

$$\mu\{Diel\ Streamflow|Precip\} = \beta_0 + \beta_1 * PET + \beta_2 * I.July + \beta_3 * I.August + \varepsilon$$

AIC = 71.51 R^2 model = 0.406 R^2 Precip variable = 0.128