RESPONSE OF COASTAL STREAM HABITAT AND JUVENILE STEELHEAD TO
THE HONEYDEW FIRE IN HUMBOLDT COUNTY, CALIFORNIA

By

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ABSTRACT

Response of coastal stream habitat and juvenile steelhead to the Honeydew Fire, Humboldt County, California

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Fire plays an important role in shaping the Pacific Northwest landscape, including riparian areas. Riparian areas provide important habitat for many species of the family Salmonidae, including steelhead (*Oncorhynchus mykiss*). Over the last century, fire suppression has resulted in forests becoming denser and more prone to intense and catastrophic wildfires. Global climate change is expected to increase the frequency and severity of wildfires. With the general decline of salmonids throughout the Pacific Northwest, understanding the role that wildfire plays in fish ecology is of increased importance. To determine the effects of wildfire on steelhead, two burned streams (Kinsey Creek and Big Flat Creek) and one unburned stream (Spanish Creek), located in the King Range National Conservation Area, California were chosen. Each study stream had one to two years of pre-fire data. The study streams were monitored during the summer and early fall months over a three-year period post-fire to determine potential changes in fish habitat (pool depth, water temperature, large woody debris abundance and volume) or fish response (abundance, condition, mean size, and density).
A lack of extensive pre-fire data prevented this study from reaching any conclusions about fire effects. However, I found a variety of apparent responses to fire. These apparent responses may be due to fire or could be a result of natural variation; I did not have the dataset needed to distinguish between them. There was some evidence of a post-fire reduction in pool depth in Big Flat Creek, particularly the first year post-fire, but not in Kinsey Creek. The maximum depth of pools in Spanish Creek varied little among years. Daily mean and maximum water temperature was lower the first year post-fire than it was pre-fire in both Kinsey and Big Flat creeks, both the burned streams. Daily mean and maximum water temperature was higher than it was pre-fire the second and third years’ post-fire in Kinsey Creek while remaining lower the third year post-fire in Big Flat Creek. The observed increase in the mean of the daily mean and maximum temperature in Kinsey Creek was less than 1°C. Post-fire temperatures in the study streams were in the preferred range of juvenile steelhead. Number of large woody debris pieces per 100 m increased post-fire in Big Flat Creek and volume of large woody debris per 100 m was greater three years post-fire than it was pre-fire. However, there was only one year of pre-fire LWD data in Big Flat Creek and LWD in Kinsey Creek did not appear to respond to fire, which both prevent the conclusion of a fire effect. In addition, LWD volume per 100 m in Spanish Creek was greater in the post-fire period than in the pre-fire period. Juvenile steelhead abundance in Big Flat Creek was significantly greater three years after fire (12,157 ± 2,854) than the year of the fire (7,461 ± 359), but the average post-fire abundance (7,343) was very similar to the pre-fire abundance (7,461). Like in Big Flat Creek, the highest observed abundance of juvenile steelhead in Kinsey
Creek was three years post-fire (1932 ± 284) but it was not significantly different than the pre-fire value (1509 ± 342). Abundance of juvenile steelhead in Spanish Creek varied from 730 ± 160 to 3887 ± 695. Fire appeared to negatively affect the summer body condition of age 0+ juvenile steelhead in Big Flat Creek the first year post-fire. Fire did not affect summer body condition of juvenile steelhead in Kinsey Creek. Summer body condition varied little among years in Spanish Creek. Fire appeared to have a positive effect on fork length of juvenile steelhead; average fork length of 0+ and 1+ juvenile steelhead increased significantly post-fire in the burned streams but not in the unburned stream. However, when the pre- and post-fire mean fork length of age 0+ and age 1+ juvenile steelhead in the study streams are compared in a balanced dataset, it appears that fire had a positive effect on the mean fork length of age 0+ juvenile steelhead but did not affect the mean fork length of age 1+ juvenile steelhead. Conclusive evidence of fire effects in this study were not possible due to the limited extent of pre-fire data but overall, juvenile steelhead were able to persist if not thrive after fire disturbance.
ACKNOWLEDGEMENTS

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Appendix D. Fork length distribution of juvenile steelhead in Big Flat Creek, King Range National Conservation Area, California from 2003-2006.
INTRODUCTION

Fire is a natural part of the Pacific Northwest’s landscape (Agee 1993, Whitlock et al. 2003). Historically, frequent low-intensity fire was one of the main agents for creating forest spatial heterogeneity, which is one of the key attributes of a healthy forest (Reeves et al. 1995, Wills and Stuart 1994). Healthy forests are one of the main indicators of healthy watersheds, and healthy watersheds support healthy native fish populations (Franklin et al. 2000). However, one of the primary goals of forest management over the last century has been fire suppression to protect private property and the economic value of trees. Fire suppression has fundamentally altered many forests in the Pacific Northwest (Hessburg et al. 2005), contributing to the buildup of unnaturally high fuel loads (Covington and Moore 1994, Everett et al. 1997). High fuel loads, when combined with global warming, leads to predictions of increased frequency, magnitude, and severity of forest fires (Hessburg and Agee 2003, Westerling et al. 2006). More intense and severe forest fires have the potential to negatively impact important populations of native fishes, including native salmonids (Rieman et al. 2003). Severe fires, often occurring in forests subject to fire suppression (Kauffman 1990), can both cause direct mortality (Gresswell 1999) and greater long-term indirect effects through changes in riparian vegetation. With the general decline of salmonid species throughout the Pacific Northwest (Nehlsen et al. 1991, Gresh 2000), understanding the role that fire plays in aquatic ecosystems is of increased importance. Land managers have to decide
whether or not to suppress fires burning in watersheds containing listed species (Rieman et al. 2000).

In early September 2003, lightning strikes ignited the Honeydew Fire in the King Range National Conservation Area (KRNCA) in Humboldt County, California. The fire lasted for 42 days and burned approximately 51 km² (roughly 20% the KRNCA). Fire damage occurred along the western side of the KRNCA (Figure 1), including thirteen small watersheds that provide habitat for steelhead (Oncorhynchus mykiss; BLM, 2004). Steelhead populations in the study streams are currently federally listed as threatened (NMFS 2000). Despite their declining abundance throughout California (Busby et al. 1996), steelhead exhibit a complex variety of life history strategies that increase the probability of persistence during periods of environmental disturbance (Gresswell 1999). Steelhead are adapted to survive in small, steep, and highly dynamic coastal watersheds (Engle 2005), readily colonizing newly opened habitat (Howell 2006). Recolonization of streams by salmonids can occur in less than two years (Howell 2006, Novak and White 1989, Rieman et al. 1997).

Immediate effects of fire on salmonid habitat include elevated water temperature, changes in stream pH, and elevated metals and nutrients (Gresswell 1999, Minshall & Brock 1991, Rieman et al., 1997). Fire can cause elevated water temperature due to the reduction of riparian vegetation (Gresswell 1999) and aggradation from excessive sediment due to post-fire erosion (May & Lee 2004, Dunham et al. 2007, Johnson and Jones 2000).
Figure 1: Map of the King Range National Conservation Area, California with the three study streams shown.
Elevated water temperatures can alter fish abundance, species diversity, duration of egg incubation, and offspring survival (Gresswell 1999). Changes in stream pH often occur with ash-fall and may be the reason for distress and mortality of salmonids during fire (Cushing and Olson 1963). The levels of trace elements (e.g. aluminum, iron, lead, and zinc) in ash extracts can be unsafe for aquatic organisms. (Woodward 1989). Elevated concentrations of nitrogen and phosphorus resulting from ash-fall and sediment delivery have been observed in streams following fire (Brass et al. 1996, Robinson and Minshall 1996, Spencer et al. 2003). Ammonia introduced to stream water by smoke may also cause fish mortality (Minshall et al. 1997).

Long-term effects of fire include increased erosion and altered large woody debris (LWD) recruitment (Wondzell and King 2003). Fire can accelerate erosional processes by increasing soil-water content and decreasing root strength of riparian and upland vegetation (Wondzell and King 2003). Erosion is greatest during storms soon after a fire that has consumed the majority of vegetation and duff in the drainage (Gresswell 1999). Erosion delivers sediment to streams, which can alter stream morphology by reducing pool frequency, increasing channel width, decreasing size of bed material, and increasing turbidity (Miller et al. 2003). Excessive turbidity negatively affects steelhead egg-to-fry survival and juvenile steelhead feeding efficiency (Shaw and Richardson 2001, Sigler et al. 1984). High suspended sediment concentrations have been observed to cause direct mortality of juvenile and adult salmonids (Bozek and Young 1994). Excessive sediment in streams may decrease the area of suitable spawning gravels, reducing spawning
success (Brown 1989). Fire can also cause an increase in annual water yield (e.g.,
increased summer flows and/or peak flows) due to decreased infiltration from fire-
induced water repellency and decreased evapotranspiration from loss of forest vegetation
(Beschta 1990, Rieman and Clayton 1997, Gresswell 1999). Increased peak flows can
cause bank erosion and landslides (Wondzell and King 2003) as well as localized
removal of riparian vegetation (Dwire and Kauffman 2003). Post-fire-related flooding
can kill or displace large numbers of fish and negatively alter stream habitat (Novak and
White 1989, Burton 2005). Erosional processes can also deliver an excessive abundance
of LWD, blocking fish passage, covering important spawning sites, and damaging habitat
(Gresswell 1999).

Although many fire-related impacts adversely affect fish, fire may also be
beneficial by increasing stream productivity, LWD abundance, and habitat heterogeneity.
Short-term increases in productivity can occur in streams where primary production is
limited by low water temperatures and heavy shading (Gresswell 1999). Erosional events
(including debris flows and landslides) can transport large boulders and LWD to streams,
which increase physical structure and habitat complexity (Wondzell and King 2003).
Large woody debris may persist in stream channels through the period of forest re-
establishment, influencing stream morphology and fish habitat for decades (Gresswell
1999). An increase in LWD recruitment usually increases the rate of pool formation and
habitat structure (Fausch and Northcote 1992), improving over-winter survival of
juvenile steelhead (Solazzi et al. 2000, Roni and Quinn 2001). Large woody debris and

This study benefitted from the availability of pre-fire data that provided a rare opportunity to assess the response of juvenile steelhead and coastal stream habitat to fire-related disturbance. Pre-fire juvenile steelhead sampling (length, weight, and distribution) was conducted by the California Cooperative Fish and Wildlife Research Unit (CCFWRU) in Big Flat Creek (2000 and 2003), Kinsey Creek (2000) and Spanish Creek (1999 and 2000). Few fire-effects studies have had pre- and-post fire data, and very few have focused on anadromous salmonids (Gresswell 1999). To understand the effects of wildfire on steelhead and stream habitat in the KRNCA, this study had two objectives: 1) collect juvenile steelhead abundance, density, fork length and weight to understand steelhead dynamics within the period of fire recovery and 2) describe observed habitat including pool depth, LWD (abundance, volume, and number of jams) and water temperature within the period of fire recovery.
MATERIALS AND METHODS

Study Area

The KRNCA is located in southwestern Humboldt County in Northern California and is managed by the Bureau of Land Management (BLM) to maintain its primitive character, being one of the last wild and undeveloped coastal landscapes in the United States (Figure 1; BLM 2004). The KRNCA is managed primarily for recreation, wildlife habitat, sustainable use of natural resources, and maintenance of healthy watersheds for anadromous fish (BLM 2004). In the fall of 2006, 63% of the KRNCA was designated as wilderness, conferring an additional degree of protection to the area containing the study streams, including the prohibition of the use of natural resources. The KRNCA comprises an area of 243 km² and 56 km of coastline, extending from Sinkyone Wilderness State Park north to the mouth of the Mattole River. Elevation ranges from sea level to 1200 m. The area surrounding the King Range has some of the highest rates of crustal deformation, surface uplift, and seismic activity in North America, attributed to high rates of geologic uplift from the tectonic forces of the nearby Mendocino Triple Junction (Davenport et al. 2002). Geologically, the King Range lies within the Coastal belt of the Franciscan Complex geologic formation, within the King Peak unit of the King Range terrane (McLaughlin et al. 2000). Due to its high level of geologic activity, the King Range terrane is relatively weak, easily weathered, and inherently susceptible to landslides, erosion, and mass wasting (Davenport et al. 2002).
The upper reaches of King Range streams have predominantly steep slopes, making for a quick transition from the riparian forest to one dominated by Douglas fir (*Pseudotsuga menziesii*). The riparian zone is comprised of big leaf maple (*Acer macrophyllum*), bay laurel (*Umbellularia californica*), willow (*Salix sp.*) and red alder (*Alnus rubra*). Red alder tends to colonize disturbed riparian areas prior to other deciduous trees. Douglas fir forests with subdominant species including sugar pine (*Pinus lambertiana*), tanoak (*Lithocarpus densiflora*), canyon live oak (*Quercus chrysolepis*) and madrone (*Arbutus menziesii*) predominate in the upland areas (BLM 2004). The King Range receives between 254-508 cm of precipitation a year with most of it occurring as rain between October and April; it is one of the wettest places in California (Figure 2) (BLM 2004).
Figure 2: The amount of precipitation (cm) by water year from 1999 to 2006 measured at the Honeydew gauge (Humboldt County, California). The Honeydew gauge is the closest rain gauge to the study streams.
Study Streams

Three streams on the west slope of the King Range were selected for this study, based on having been sampled prior to the Honeydew Fire: Spanish Creek, Kinsey Creek, and Big Flat Creek (Figure 1). All three of the study streams contained populations of steelhead prior to the fire. The Honeydew Fire did not burn in Spanish Creek’s watershed, burned approximately half of Kinsey Creek’s watershed, and burned the majority of Big Flat Creek’s watershed. Spanish Creek is a 2nd-order stream located 11 km south of Punta Gorda, with a drainage area of 4.6 km² (Table 1). The riparian zone in the lower section of Spanish Creek is presently in a state of recovery from the major flood of 1964, exhibiting a large alluvial fan at its mouth (Engle 2005). Kinsey Creek is a 2nd-order stream located 13.5 km south of Punta Gorda, with a drainage area of 3.8 km² (Table 1). Kinsey Creek exhibits a dense riparian zone extending nearly to the intertidal Pacific and is the steepest of the study streams. Big Flat Creek is a 3rd-order stream located 13.5 km north of Shelter Cove, with a drainage area of 16.3 km² (Table 1). Big Flat Creek transports a relatively large amount of bed load originating from large headwater landslides, forming a large alluvial fan at its mouth (BLM 2004). Post-fire habitat and fish sampling were conducted in the summers (June-September) of 2004 - 2006. Sampling extended from the intertidal zone to the estimated end of anadromy in each study stream.
Table 1: Characteristics of Spanish, Kinsey, and Big Flat creeks located within the King Range National Conservation Area, California.

<table>
<thead>
<tr>
<th>Creek</th>
<th>Reach Length (m)</th>
<th>Stream Order</th>
<th>Drainage Area (km²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Spanish</td>
<td>1345</td>
<td>2</td>
<td>4.7</td>
</tr>
<tr>
<td>Kinsey</td>
<td>747</td>
<td>2</td>
<td>3.8</td>
</tr>
<tr>
<td>Big Flat</td>
<td>3195</td>
<td>3</td>
<td>16.3</td>
</tr>
</tbody>
</table>
Habitat typing was performed to characterize the amount of each habitat type in the study streams and to select habitat units for electrofishing. Habitat unit surveys were conducted during summer base flow using a modified form of Level II Habitat Survey and Inventory Protocol (Flosi and Reynolds 1994). A stadia rod was used to measure wetted stream width and an associated average depth at 1/3 and 2/3 of the unit length, along with maximum depth and pool-tail depth. Habitat units were characterized as runs, riffles, pools, and cascades to be consistent with past KRNCA investigations (Engle 2005, Duffy 2007, personal communication). For consistency, the same observer was used to identify channel units throughout the study when possible. The same observer was used all three years’ post-fire in Big Flat Creek but not in Spanish and Kinsey creeks. Habitat surface area was calculated by averaging the unit wetted width measurements and multiplying by the unit length. In 2005, above-average seasonal rainfall forced us to re-measure wetted stream width of habitat units selected for electrofishing in order to accurately calculate surface area before fish sampling.

Large Woody Debris

Large woody debris surveys were conducted in the three study streams using the same reach lengths as for habitat typing, from the mouth to estimated end of anadromy. Large woody debris was defined as a log at least 2-m-long and 10-cm-wide at mid-point
located in the bankfull channel. Pre-fire LWD surveys were conducted in each study stream (Spanish Creek in 1999, Kinsey Creek in 2000, and Big Flat Creek in 2003). Each piece was identified to species and its length and diameter at mid-point were visually estimated to the nearest 0.1 m. After a randomly selected starting piece of LWD, every tenth piece was visually estimated, and then the true length and mid-point diameter were measured with a stadia rod and/or a measuring tape. To correct for differences between actual and estimated individual LWD volume, and to then calculate the total LWD volume in a study stream, the equations for estimating total habitat area using visual estimation methods developed by Hankin and Reeves (1988) were used.

Additional variables recorded included decay class, root wad presence, input mechanism, grouping, evidence of burning, and pool type. Input mechanism was defined as the method by which the LWD piece arrived in the stream (mass movement, bank undercut, wind throw, or undetermined). Decay class was adapted from Maser and Trappe (1984) (Table 2). Pools formed by LWD were characterized as lateral scour pool, plunge pool, dammed pool, or backwater pool. Volume and density were calculated for each streams study reach. Length, width, and height of LWD jams were estimated.

Yearly precipitation and flow events were examined to determine if there were any major storm events that could contribute large woody debris to the study streams during the pre- and post-fire periods.
Table 2: The five-class system used to classify the level of decay or large woody debris in the study streams. The decay classes were adapted from Maser and Trappe (1984).

<table>
<thead>
<tr>
<th>Decay Class</th>
<th>Bark</th>
<th>Twigs</th>
<th>Texture</th>
<th>Shape</th>
<th>Wood Color</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Intact</td>
<td>Present</td>
<td>Intact</td>
<td>Round</td>
<td>Original color</td>
</tr>
<tr>
<td>2</td>
<td>Intact</td>
<td>Absent</td>
<td>Intact</td>
<td>Round</td>
<td>Original color</td>
</tr>
<tr>
<td>3</td>
<td>Trace</td>
<td>Absent</td>
<td>Smooth; some surface abrasions</td>
<td>Round</td>
<td>Original color; darkening Dark</td>
</tr>
<tr>
<td>4</td>
<td>Absent</td>
<td>Absent</td>
<td>Abrasions; some holes and openings</td>
<td>Round to oval</td>
<td>Dark</td>
</tr>
<tr>
<td>5</td>
<td>Absent</td>
<td>Absent</td>
<td>Vesicular; many holes and openings</td>
<td>Irregular</td>
<td>Dark</td>
</tr>
</tbody>
</table>
The closest rain gauge to the study streams is in Honeydew, so rainfall measured at that gauge was used as a surrogate for rainfall in the study streams’ watersheds. The closest gaged stream to the study streams is the Mattole River so the streamflow data from the USGS 11469000 Mattole River near Petrolia gage was used as a surrogate for flow events in the study streams. The National Weather Service provided data on windstorms during the study period.

**Water Temperature**

Water temperature data loggers were placed in two locations (upper and lower) in each study stream. Water temperature data loggers (Hobo Water Temp Pro, Onset Computer Corp., Pocasset, MA) were used to monitor temperature during the warmest months of the year (June 20-September 20). Data loggers were placed in study streams by attachment to a “peanut-shaped” rock using plastic coated metal cable with clamps. Data loggers were programmed to record temperature at 30-minute intervals. Pre-fire water temperature was available for Spanish Creek in 1999 and 2000, Kinsey Creek in 2000 and Big Flat Creek in 2000, and 2003. Post-fire water temperature was available for Spanish in 2004 (restricted to July 21st to September 20th for 2004) to 2006, Kinsey in 2004 to 2006, and Big Flat in 2004 and 2006.
Juvenile Steelhead Abundance Estimates

Juvenile steelhead abundances were estimated for each year and study stream that such data was available. Total juvenile steelhead abundance in each study reach was estimated using the multistage sampling design of Hankin (1984). The habitat typing (previously described) was completed first, and then fish sampling was performed in selected habitat units. Habitat units were selected to be sampled using a stratified systematic random sampling design where independent random starts occurred for each stratum in each study stream in each year. Stratified habitats included pools, runs, riffles, and cascades. To efficiently sample the study area, a sub-sample of each stratum was selected for sampling to determine steelhead abundance (25% of pools and runs and 15% of riffles and cascades). Selected channel units were sampled for fish abundance using multiple-pass depletion electrofishing between mid-August and early October. Block nets (6-mm mesh) were placed at the top and bottom of each channel unit to prevent emigration or immigration during electrofishing. Electrofishing removal passes of timed equal-effort were made using a backpack electrofishing unit (Model 12, Smith-Root Inc., Vancouver, Washington). Two electrofishing passes were conducted in each sampling unit. If the number of fish captured during the second pass was greater than 25% of the number of fish caught during the first pass, a third pass was done. All field methods involving vertebrate animals were approved by the Humboldt State University Institutional Animal Care and Use Committee (Protocol No. 02/03.F.07-A).
Total juvenile steelhead abundance in all habitat types in each study reach was estimated by using the formulas described in Brakensiek (2002) for calculating abundance in riffles. Abundance data for Big Flat Creek were collected in 2000 but were for a different reach length and so could not be used in comparing pre- and post-fire abundance. Pre-fire abundance data were available for Spanish Creek in 1999 and 2000, Kinsey Creek in 2000 and Big Flat Creek in 2003.

Fish in half of the sampled units were weighed to the nearest 0.01 g (Scout Pro SP 402, Ohaus Inc., Parsippany, NJ) and measured to the nearest mm fork length. Juvenile steelhead were classified into age class based on fork length; 0+ (<110 mm) and 1+ (≥ 110) mm for Spanish and Big Flat creeks and 0+ (<90 mm) and 1+ (≥ 90 mm) for Kinsey Creek. The fork length breaks for age classes were determined through scale aging, examination of fork length frequency histograms, and age data from Engle (2005). Pre-fire juvenile steelhead fork length and weight data were available for Spanish Creek in 1999 and 2000, Kinsey Creek in 2000 and Big Flat Creek in 2000 and 2003.

Data Analysis

Due to a limited amount of pre-fire data, I could not use the before-after-control-impact (BACI) statistical design to analyze the data. Instead, I examined post-fire trends through time, focusing on the changes in differences between burned and unburned study streams and using pre-fire data where I could. Many of the statistical analyses performed
had little power to detect a fire effect due to limited pre-fire data. All statistical analyses were conducted using R statistical software (R Development Core Team, 2011) except for the mixed effects model which was analyzed using SAS software, Version 9.2 of the SAS System for Windows. Copyright © 2008 SAS Institute Inc. SAS and all other SAS Institute Inc. product or service names are registered trademarks or trademarks of SAS Institute Inc., Cary, NC, USA.

Analysis of habitat data focused on pool depth. It was hypothesized that pools, and particularly the maximum depth of pools, would be the most susceptible to changes in depth due to post-fire sediment inputs. To determine if fire had an effect on maximum depth of pools; maximum pool depth with distance upstream was plotted for each year that data was available for each of the study streams. These plots were then examined to see if there were changes in the years before and after fire in the burned streams (Kinsey and Big Flat) and whether there were similar changes in the unburned creek (Spanish). For each year and study stream, the slope of the regression line between maximum depth and distance upstream was calculated, along with the 95% confidence interval for the slope. The slope estimates were then compared to see if there were changes post-fire in the burned streams and whether the un-burned stream showed a similar trend. Ideally, residual pool depth would have been used in this analysis but residual pool depth data were not collected pre-fire. Habitat data were collected when study streams were typically at summer baseflow, but during some years flow was likely above baseflow due
to late spring storms. This was the case in 2005 for example, so depth observations from 2005 are likely biased high.

Large woody debris data were not analyzed statistically. There was only one year of pre-fire LWD data for each of the study streams, and additionally, after examining the data, there were no trends apparent that warranted formal analysis. The large woody debris data are summarized graphically. Comparison is made pre- and post-fire between number of LWD pieces, density of LWD pieces, LWD volume, density of LWD volume, and LWD size class for the three study streams.

Water temperature analysis was only performed on the upper sites for the three streams because they provided the most complete dataset. Water temperature probes were placed in lower locations in some streams in only some years, and temperature probes were not recovered from the lower locations for some streams and years. In addition, the lower temperature probe locations were less likely to show a fire effect, particularly for Spanish and Big Flat creeks, because the lower parts of Spanish and Big Flat creeks are alluvial flats which have limited riparian vegetation.

To determine whether fire had an effect on daily mean and maximum water temperature in streams that experienced fire, linear models were used to compare the relationship in water temperature (June 20th to September 20th) before and after fire between a burned stream (Big Flat or Kinsey) and an unburned stream (Spanish) with “Year” or “Burn” as additional variables. The five a priori models tested for each
temperature metric were: 1) $T_{burned\ stream} = T_{unburned\ stream} + \text{Burn} + T_{unburned\ stream}:\text{Burn}$, 2) $T_{burned\ stream} = T_{unburned\ stream} + \text{Burn}$, 3) $T_{burned\ stream} = T_{unburned\ stream} + \text{Year} + T_{unburned\ stream}:\text{Year}$, 4) $T_{burned\ stream} = T_{unburned\ stream} + \text{Year}$, 5) $T_{burned\ stream} = T_{unburned\ stream}$. The factor “Burn” denoted whether the impact period was pre- or post-fire. The factor “Year” was included to determine if post-fire relationships should just be considered between-year variation or just related to fire. The model best describing the data was chosen based on Akaike’s Information Criterion (AIC). Models that were within two AIC units of each other were considered competing models. If fire had an effect on water temperature then the relationship in water temperature in a burned compared to an unburned stream would change after fire. A fire effect on water temperature would be reflected in the best model containing the variable “Burn”. Using the unburned stream in the model helped to control for within-year variation in water temperature and allowed detection of any fire effects. It was hypothesized that fire would increase water temperature in the burned streams by reducing riparian vegetation and consequently increasing solar radiation. If fire increased water temperature in a burned stream (Kinsey and Big Flat), then the estimated slope parameter of the linear relationship between the burned stream and Spanish Creek would increase post-fire compared to pre-fire. The estimated slope parameter, with 95% confidence intervals, for the relationship between the burned streams and Spanish Creek were calculated for each year data were available. The yearly slopes were then compared to determine if there was a fire effect that was not captured when pre- and post-fire years were pooled. Pre-fire data were only available in Spanish,
Kinsey, and Big Flat creeks for one year (2000). This limited pre-fire data reduces the confidence in the results due to a lack of information on between-year variation before the fire between the burned and unburned streams. The modeling analysis has limited power to detect a burn effect on water temperature because of the limited extent of the data, especially for Big Flat Creek, because it only has one year of pre-fire and two years of post-fire data.

In addition to the modeling approach, potential changes in water temperature in burned streams post-fire were analyzed by calculating the difference in daily mean, daily maximum, and daily range (maximum – minimum) in water temperature from June 20th to September 20th (except for 2004 which was between July 21st and September 20th) between a burned stream (Kinsey and Big Flat) and the unburned stream (Spanish) and then comparing the difference between years pre- and post-fire. The mean difference in daily mean water temperature was calculated for each year data were available and compared to see if there were differences between pre- and post-fire years. The same technique was used to see if fire had an effect on daily maximum water temperature and daily water temperature range (maximum – minimum).

Analysis of juvenile steelhead abundance and density was hampered by limited pre-fire data. To determine if there were significant differences in juvenile steelhead abundance pre- and post-fire, comparison of 95% confidence intervals was used because for the two burned streams there was only one year of pre-fire data. Comparison of
juvenile steelhead density pre- and post-fire was done graphically because there was only one year of pre-fire density data for only one of the burned study streams.

To analyze the summer body condition of juvenile steelhead, the estimated slope and 95% confidence interval of the regression line between log mass and log fork length was calculated for each stream and year that data were available. Estimated slopes were then compared for the pre- and post-fire years for the burned and unburned streams to determine if there was a fire effect. Juvenile steelhead that had a larger summer body condition would have a steeper length-weight regression slope. Juvenile steelhead all start as fry at nearly the same size. It was hypothesized that the effects of fire could impact the stream conditions in burned streams by increasing productivity through increased light from loss of riparian vegetation and increased nutrient input, leading to different growth rates among the streams in burned and unburned locations. So one would expect the y-intercept to be about the same and the regression slopes to vary when comparing length-weight relationships for different time periods, if fire affected growth rates.

To determine whether fire had an effect on average fork length of age 0+ and age 1+ juvenile steelhead a mixed effects model was fit with fixed effects of “Burn” (burned or not burned) and “Fire” (pre or post) and the random effects of “Creek” nested within “Burn” and “Year” by “Creek” nested within “Burn”. The random effect of “Creek” nested within “Burn” was included to account for potential correlation in fish sizes for each creek among years, and the random effect of “Year” by “Creek” nested within
“Burn” is included because juvenile steelhead fork lengths measured from the same creek within a year are correlated with each other. Both random effects are nested within the “Burn” factor because a creek can only occur in either the burned or non-burned levels. The degrees of freedom for the mixed effects models were determined using the Satterthwaite approximation. It was hypothesized that fire might not increase juvenile steelhead summer body condition but that it might increase average fork length of age 0+ and age 1+ steelhead. Juvenile steelhead might not weigh more for a given size but they may be longer post-fire. It was hypothesized, that like summer body condition, average fork length of age 0+ and age 1+ juvenile steelhead might increase post-fire as a response to increased light and nutrient inputs into the burned streams from the fire. Mean fork length of juvenile steelhead was plotted against mean density of juvenile steelhead for each year stream data were available to see if there was evidence of density dependence of fork length. Density dependence of juvenile steelhead fork length in the study streams is important in interpreting the potential results of the pre- and post-fire fork length analysis.
RESULTS

Habitat

Types of habitat available were generally consistent with the relatively high gradient of all three streams. Riffles were the most common habitat by surface area, comprising from 59-80% of habitats in Spanish Creek, 45-62% of habitats in Kinsey Creek and 32-64% of habitats in Big Flat Creek (Appendix A). Over all years and streams, runs were the second most common habitats, followed by cascades and pools. Surface area of habitats was generally consistent with number of habitats (Appendix A). Spanish Creek exhibited the shallowest maximum depths (0.22-0.62 m) and Big Flat Creek the deepest (0.31-0.96 m) (Appendix A). The number of pools all three years’ post-fire in Big Flat Creek was lower than the pre-fire value; however, the total surface area of pools three years post-fire was greater than the pre-fire value (Appendix A).

The longitudinal distribution of pools and their apparent maximum depth varied among streams and years. There were no major trends apparent in maximum depth with distance upstream in Spanish Creek (Figure 3). The pools in 2004 were located further downstream in Spanish Creek compared to the other years (Figure 3). The estimated slopes of the regression lines of maximum depth with distance upstream in Spanish Creek did not differ significantly among years (Figure 4). In Kinsey Creek, there were also no major trends apparent in maximum depth with distance upstream (Figure 5).
Figure 3: Maximum depth (m) of pools in Spanish Creek with distance upstream (m) from 1999, 2000, and 2004 to 2006.
Figure 4: The estimated slopes of the regression lines for maximum pool depth with distance upstream in Spanish Creek for 1999, 2000, and 2004 to 2006. The error bars are 95% confidence intervals.
Figure 5: Maximum depth (m) of pools in Kinsey Creek with distance upstream (m) from 2004 to 2006.
The estimated slopes of the regression lines of maximum depth with distance upstream in Kinsey Creek did not differ significantly among years (Figure 6). Pools were distributed similarly in Kinsey Creek in 2004 and 2006, while in 2005 pools were found only in the upper half of Kinsey Creek (Figure 5). In Big Flat Creek, the deepest pools were found the furthest upstream in the pre-fire year (Figure 7). It appears that some of the deepest pools were lost or filled in with sediment during the post-fire years (Figure 7). However, it also appears that some new pools that were relatively deep were formed in the lower part of Big Flat Creek in 2006 (Figure 7). There was a post-fire trend of the deepest pools not being in the upper section of Big Flat Creek and instead distributed more uniformly (Figure 8). The estimated slope of the regression line of maximum depth with distance upstream in Big Flat Creek in 2005 and 2006 differed significantly from the pre-fire value (Figure 8). Both 2004 and 2006 were relatively wet years, with the highest peak streamflow during 1999 through 2006 occurring in 2006 (Figures 9, 10) and the highest annual precipitation occurring in 2004 (Figure 2). These wet years combined with the major fire disturbance may have contributed to the observed changes in pool depth and longitudinal distribution in Big Flat Creek.

**Large Woody Debris**

The abundance of LWD pieces per 100 m was highly variable year to year in Big Flat Creek and was less variable in Spanish and Kinsey creeks. The abundance of LWD per 100 m in Spanish Creek (unburned) varied little between the pre- and post-fire time
Figure 6: The estimated slopes of the regression lines for maximum pool depth with distance upstream in Kinsey Creek for 2004 to 2006. The error bars are 95% confidence intervals.
Figure 7: The maximum depth (m) of pools in Big Flat Creek with distance upstream (m) from 2003 to 2006.
Figure 8: The estimated slopes of the regression lines for maximum depth with distance upstream in Big Flat Creek from 2003 to 2006. The error bars are 95% confidence intervals.
Figure 9: Daily mean streamflow (cfs) at the USGS Mattole River near Petrolia gage for the pre-fire water years 1999, 2000, and 2003.
Figure 10: Daily mean streamflow (cfs) for the USGS Mattole River near Petrolia gage for the post-fire water years 2004 – 2006.
periods (Table 3). In Kinsey Creek (half burned), number of LWD pieces per 100 m did not differ between pre- and post-fire time periods (Table 3).

In Big Flat Creek, (majority burned), number of LWD pieces per 100 m increased post-fire relative to pre-fire (Table 3). With only one year of pre-fire data it is impossible to say that the fire increased the number of LWD pieces per 100 m in Big Flat Creek, but it appears that fire may have played a role. If there was some major event besides the fire that caused increased LWD input into the streams then it would have been apparent in Spanish Creek. However, the number of LWD pieces per 100 m in Spanish Creek varied little between the pre- and post-fire periods, giving some weight to the fire having potentially increased the number of LWD pieces per 100 m in Big Flat Creek. In Big Flat Creek post-fire there were 13 to 36 (11-16%) pieces of LWD that were observed to have been burned. In Kinsey Creek post-fire there were only 0-2 (0-8%) pieces of LWD that were observed to have been burned.

The pattern of LWD volume per 100 m was not consistent among years in the three study streams. The volume of LWD per 100 m in Spanish Creek (unburned) was greater in the post-fire period than in the pre-fire period (Table 3). In Kinsey Creek, LWD volume per 100 m increased the first year post-fire, compared to the pre-fire volume, but then returned close to the pre-fire value during the second and third years post-fire (Table 3). The volume of LWD per 100 m in Big Flat Creek during the first two years post-fire were lower than the pre-fire value but then increased in the third year post-fire to become larger than the pre-fire value (Table 3). Few LWD jams were
Table 3: The volume of LWD (m³) per 100 m, number of LWD pieces per 100 m, and number of LWD jams in the study reaches of Spanish, Kinsey, and Big Flat creeks, King Range National Conservation Area, California before and after fire. Spanish Creek was not burned, while Kinsey and Big Flat creeks were burned.

<table>
<thead>
<tr>
<th>Year</th>
<th>Fire</th>
<th>Spanish Creek</th>
<th></th>
<th>Kinsey Creek</th>
<th></th>
<th>Big Flat Creek</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Vol. (m³)/100 m</td>
<td>Pieces/100 m</td>
<td>Number of Jams</td>
<td>Vol. (m³)/100 m</td>
<td>Pieces/100 m</td>
<td>Number of Jams</td>
</tr>
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<td>Pre</td>
<td>13.0</td>
<td>10.7</td>
<td>0</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2000</td>
<td>Pre</td>
<td></td>
<td></td>
<td></td>
<td>3.3</td>
<td>4.4</td>
<td>0</td>
</tr>
<tr>
<td>2001</td>
<td>Pre</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>Pre</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2003</td>
<td>Pre</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2004</td>
<td>Post</td>
<td>22.2</td>
<td>9.6</td>
<td>2</td>
<td>14.2</td>
<td>3.1</td>
<td>2</td>
</tr>
<tr>
<td>2005</td>
<td>Post</td>
<td>58.0</td>
<td>9.9</td>
<td>3</td>
<td>3.3</td>
<td>3.0</td>
<td>0</td>
</tr>
<tr>
<td>2006</td>
<td>Post</td>
<td>16.4</td>
<td>11.2</td>
<td>0</td>
<td>2.6</td>
<td>4.4</td>
<td>0</td>
</tr>
</tbody>
</table>
encountered during either pre- or post-fire surveys except in Big Flat Creek, where 18 LWD jams were found in 2006 (Table 3). These 18 LWD jams accounted for 95% of the total volume of LWD found in Big Flat Creek in 2006. In all three streams, the presence of LWD jams appears as a stair-step pattern in cumulative LWD relative to distance upstream (Figures 11-13).

The number of LWD pieces in each decay class differed somewhat among streams. In Spanish Creek, distribution of LWD among decay classes was similar most years except for in 2005 (Figure 14). In 2005 there were a relatively high number of LWD pieces in decay classes 2 and 3, which suggest that some newer pieces of LWD entered the study reach of Spanish Creek (Figure 14). In Kinsey Creek, the number of LWD pieces in each decay class was similar among years (Figure 15). In Big Flat Creek, the largest number of LWD pieces in decay classes 1, 2, and 3 were observed in 2006 (Figure 16). This suggests that there were new inputs of LWD in 2006, which is consistent with the observation of a large increase in LWD volume per 100 m in 2006.
Figure 11: Cumulative LWD volume (m³) with distance upstream (m) in Spanish Creek during 1999 and 2004 to 2006.
Figure 12: Cumulative LWD volume (m³) with distance upstream in Kinsey Creek from 2000, and 2004 to 2006.
Figure 13: Cumulative LWD volume (m³) with distance upstream (m) in Big Flat Creek from 2003 to 2006.
Figure 14: The number of LWD pieces observed in each decay class in Spanish Creek from 1999 and 2004 to 2006.
Figure 15: The number of LWD pieces observed in each decay class in Kinsey Creek from 2000 and 2004 to 2006.
Figure 16: The number of LWD pieces observed in each decay class in Big Flat Creek from 2003 to 2006.
**Water Temperature**

Daily mean water temperature in the study streams between June 20 and September 20 was always in the favorable range for juvenile steelhead rearing, including post-fire in the burned streams (Figures 17, 19, 21). Daily maximum water temperature in the study streams never reached stressful levels for juvenile steelhead (Figures 18, 20, 22). The highest observed water temperature in the study streams was 19.1 °C in Big Flat Creek.

Fire may have had an effect on daily mean and daily maximum water temperature in Big Flat and Kinsey creeks when individual years were compared but not when years were pooled into pre- and post-fire in the model. The observed changes in water temperature may be partly attributable to the accuracy of the temperature loggers. The best fitting models for Big Flat Creek mean and maximum water temperature (June 20 – September 20) were the models that included Spanish Creek interacting with year (Tables 4, 5). If fire was a significant predictor in determining the relationship between water temperature in Big Flat Creek and Spanish Creek then the best fitting model would have included “Burn” in it. However, the variable “Burn” was not included in the best fitting models. All models including the variable “Burn” were separated from the best fitting models for both mean and maximum temperature by more than 100 AIC units (Tables 4, 5). The variable “Year” contributing to the best models is evidence of year-to-year variation in the relationships between burned and unburned streams, with some of this
Figure 17: Daily mean water temperature (°C) in upper Spanish Creek from June 20\textsuperscript{th} to September 20\textsuperscript{th} (except for 2004, which is from July 21 to September 20).
Figure 18: Daily maximum water temperature (°C) in upper Spanish Creek from June 20th to September 20th (except for 2004, which is from July 21 to September 20).
Figure 19: Daily mean water temperature (°C) in upper Kinsey Creek from June 20th to September 20th.
Figure 20: Daily maximum water temperature (°C) in upper Kinsey Creek from June 20th to September 20th.
Figure 21: The daily mean water temperature in upper Big Flat Creek from June 20th to September 20th.
Figure 22: The daily maximum water temperature (°C) in upper Big Flat Creek from June 20th to September 20th.
<table>
<thead>
<tr>
<th>Model</th>
<th>AIC</th>
<th>Δ AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Big Flat Mean Temp =Spanish Mean Temp+Year+Spanish Mean Temp:Year</td>
<td>37.62691</td>
<td>0</td>
</tr>
<tr>
<td>Big Flat Mean Temp =Spanish Mean Temp+Year</td>
<td>62.64224</td>
<td>25.01533</td>
</tr>
<tr>
<td>Big Flat Mean Temp =Spanish Mean Temp+Burn</td>
<td>141.8534</td>
<td>104.2265</td>
</tr>
<tr>
<td>Big Flat Mean Temp =Spanish Mean Temp+Burn+Spanish Mean Temp:Burn</td>
<td>143.7011</td>
<td>106.0742</td>
</tr>
<tr>
<td>Big Flat Mean Temp =Spanish Mean Temp</td>
<td>190.3764</td>
<td>152.7495</td>
</tr>
</tbody>
</table>
Table 5: The AIC values for the Big Flat Creek maximum water temperature model.

<table>
<thead>
<tr>
<th>Model</th>
<th>AIC</th>
<th>Δ AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Big Flat Max Temp = Spanish Max Temp + Year +</td>
<td>288.2517</td>
<td>0</td>
</tr>
<tr>
<td>Spanish Max Temp: Year</td>
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<td></td>
</tr>
<tr>
<td>Big Flat Max Temp = Spanish Max Temp + Year</td>
<td>326.8791</td>
<td>38.6274</td>
</tr>
<tr>
<td>Big Flat Max Temp = Spanish Max Temp + Burn</td>
<td>470.4579</td>
<td>182.2062</td>
</tr>
<tr>
<td>Big Flat Max Temp = Spanish Max Temp + Burn +</td>
<td>476.3118</td>
<td>188.0601</td>
</tr>
<tr>
<td>Spanish Max Temp: Burn</td>
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<td></td>
</tr>
<tr>
<td>Big Flat Max Temp = Spanish Max Temp</td>
<td>511.105</td>
<td>222.8533</td>
</tr>
</tbody>
</table>


variation most likely attributable to individual variation among temperature loggers. The variable “Year” contributing to the best models was not unexpected however, because “Burn” has little power due to there being only one pre-burn observation. However, when the slope of the linear relationship between Big Flat Creek and Spanish Creek for daily mean and maximum water temperature was compared by year it appeared that fire may have had an effect on mean water temperature (Figure 23). The slope of the linear relationship for daily mean water temperature between Big Flat Creek and Spanish Creek was significantly different the third year post-fire (2006) when compared to the pre-fire value (2000) (Figure 23). This difference could be a fire effect or a result of natural variation, but there is not enough data to differentiate between the two. The slope of the linear relationship for daily maximum water temperature between Big Flat Creek and Spanish Creek was not significantly different for the two years’ (2004, 2006) post-fire when compared to the pre-fire value (Figure 23). When the mean of the difference in daily mean temperature was compared between Big Flat Creek and Spanish Creek, it appeared that Big Flat Creek was cooler relative to Spanish for the two post-fire years (Figure 24; Table 6). This same pattern was evident when comparing the difference in daily maximum water temperature and daily water temperature range (maximum – minimum temperature) between Big Flat and Spanish creeks (Figures 25, 26; Table 6), except that the difference in daily water temperature range was greater three years post-fire than it was pre-fire (Table 6).
Figure 23: The estimated coefficient for the slope parameter for each year in the best-fitting linear model for the Big Flat mean and maximum temperature models. The slope parameter is the linear relationship between Big Flat and Spanish Creek water temperature. The error bars are 95% confidence intervals.
Figure 24: The difference in daily mean water temperature (°C) from June 20 to September 20 (except 2004 is from July 21 to September 20) between Big Flat Creek and Spanish Creek during 2000, 2004, and 2006. Spanish Creek was not burned and Big Flat Creek burned.
Figure 25: The difference in daily maximum water temperature (°C) from June 20 to September 20 (except 2004 is from July 21 to September 20) between Big Flat Creek and Spanish Creek during 2000, 2004, and 2006. Spanish Creek was not burned and Big Flat Creek burned.
Figure 26: The difference in daily temperature range (maximum-minimum) (ºC) from June 20 to September 20 (except for 2004, which is from July 21 to September 20) between Big Flat Creek and Spanish Creek during 2000, 2004, and 2006.
Table 6: The mean of the difference in the daily mean water temperature (°C) (June 20 to September 20*), daily maximum water temperature and daily water temperature range (maximum – minimum) between Kinsey Creek and Big Flat Creek with Spanish Creek. *Except for 2004, which is from July 21 to September 20).

<table>
<thead>
<tr>
<th>Year</th>
<th>Mean water temperature (°C)</th>
<th>Maximum water temperature (°C)</th>
<th>Water temperature range (°C)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Kinsey-Spanish</td>
<td>Big Flat-Spanish</td>
<td>Kinsey-Spanish</td>
</tr>
<tr>
<td>2000</td>
<td>-0.07</td>
<td>0.91</td>
<td>-0.26</td>
</tr>
<tr>
<td>2004</td>
<td>-0.13</td>
<td>0.40</td>
<td>-0.89</td>
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<td>2006</td>
<td>0.30</td>
<td>0.74</td>
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</tbody>
</table>
In Kinsey Creek, the best fitting model for daily mean and maximum water temperature (June 20 – September 20) was the model that contained Spanish Creek by year (Tables 7, 8). Fire was not a significant predictor in determining the relationship between water temperature in Kinsey Creek and Spanish Creek. None of the models including the variable “Burn” were considered alternatives to the best-fitting models. However, when the slope of the linear relationship between Kinsey Creek and Spanish Creek daily mean and daily maximum water temperature were compared by year, it appeared that fire did have an effect, particularly for maximum temperature (Figure 27). The estimated slope for maximum temperature was smaller the first year post-fire compared to the pre-fire value, then increased to greater than the pre-fire value for the second and third years post-fire (Figure 27). The estimated slope for mean temperature exhibited a similar but less pronounced trend. When the mean of the difference in daily mean and daily maximum water temperature differences between Kinsey and Spanish were compared by years, the results were similar to the results from the estimated slopes (Figures 28, 29). It appeared that daily mean and maximum water temperature in Kinsey were cooler relative to Spanish the first year post-fire and is then warmer relative to Spanish the second and third years’ post-fire (Figures 28, 29; Table 6). The result was the same for the mean difference in daily temperature range (maximum – minimum) between Kinsey and Spanish creeks; the difference in temperature range was smaller the first year post-fire and larger the second and third years’ post-fire (Figure 30; Table 6).
Table 7: The AIC values for the Kinsey Creek mean water temperature model.

<table>
<thead>
<tr>
<th>Model</th>
<th>AIC</th>
<th>Δ AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Kinsey Mean Temp = Spanish Mean Temp + Year + Spanish Mean Temp: Year</td>
<td>-520.18</td>
<td>0</td>
</tr>
<tr>
<td>Kinsey Mean Temp = Spanish Mean Temp + Year</td>
<td>-508.21</td>
<td>11.97</td>
</tr>
<tr>
<td>Kinsey Mean Temp = Spanish Mean Temp + Burn</td>
<td>-154.20</td>
<td>365.98</td>
</tr>
<tr>
<td>Kinsey Mean Temp = Spanish Mean Temp + Burn + Spanish Mean Temp: Burn</td>
<td>-143.87</td>
<td>376.30</td>
</tr>
<tr>
<td>Kinsey Mean Temp = Spanish Mean Temp</td>
<td>-50.95</td>
<td>469.23</td>
</tr>
</tbody>
</table>
Table 8: The AIC values for the Kinsey Creek maximum water temperature model.

<table>
<thead>
<tr>
<th>Model</th>
<th>AIC</th>
<th>Δ AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Kinsey Max Temp =Spanish Max Temp+Year+Spanish Max Temp:Year</td>
<td>-98.57</td>
<td>0</td>
</tr>
<tr>
<td>Kinsey Max Temp =Spanish Max Temp+Year</td>
<td>3.12</td>
<td>101.69</td>
</tr>
<tr>
<td>Kinsey Max Temp =Spanish Max Temp+Burn</td>
<td>502.98</td>
<td>601.55</td>
</tr>
<tr>
<td>Kinsey Max Temp =Spanish Max Temp+Burn+ Spanish Max Temp:Burn</td>
<td>525.66</td>
<td>624.22</td>
</tr>
<tr>
<td>Kinsey Max Temp =Spanish Max Temp</td>
<td>582.11</td>
<td>680.68</td>
</tr>
</tbody>
</table>
Figure 27: The estimated coefficient for the slope parameter for each year in the best-fitting linear model for the Kinsey mean and maximum temperature models. The error bars are 95% confidence intervals.
Figure 28: The difference in daily mean water temperature (°C) from June 20th to September 20th (except for 2004, which is from July 21st to September 20th) between Kinsey Creek and Spanish Creek during 2000, and 2004 to 2006. Spanish Creek was not burned and Kinsey Creek was burned.
Figure 29: The difference in daily maximum water temperature (°C) from June 20th to September 20th (except for 2004, which is from July 21st to September 20th) between Kinsey Creek and Spanish Creek during 2000, and 2004 to 2006. Spanish Creek was not burned and Kinsey Creek was burned.
Figure 30: The difference in daily water temperature range (maximum-minimum) (°C) from June 20th to September 20th (except for 2004, which is from July 21st to September 20th) between Kinsey Creek and Spanish Creek during 2000, and 2004 to 2006. Spanish Creek was not burned and Kinsey Creek was burned.
Steelhead Abundance and Density

Estimated juvenile steelhead abundance varied among years in all streams. In Spanish Creek (unburned), pre-fire abundance varied five-fold; while post-fire abundance varied two-fold (Figure 31). In Kinsey Creek and Big Flat Creek, the single pre-fire abundance estimate for each stream was within the range of post-fire abundance estimates. Highest estimated abundance of juvenile steelhead during the period sampled in both Kinsey Creek and Big Flat Creek (burned streams) was in 2006, three years after the fire (Figure 31). Although this pattern was not observed in Spanish Creek (unburned), limited pre-fire data precluded statistical testing of data.

Steelhead Age Composition, Summer Body Condition, and Length

Age 0+ steelhead predominated the populations in all three streams studied during most years. Among years age 0+ steelhead comprised 73-88% of the sample population in Spanish Creek, 2%-82% of the sample population in Kinsey Creek and 61%-91% of the sample population in Big Flat Creek (Figure 32). Age 1+ steelhead made up a higher proportion of the population in Big Flat Creek during all three post-fire years sampled compared to the pre-fire years (Figure 32).

When the estimated mass-fork length regression slope is compared among years in each stream, it appears that there was a fire effect in Big Flat Creek (Figures 33, 34).
Figure 31: Juvenile steelhead abundance in Spanish, Kinsey, and Big Flat creeks from 1999 to 2006. Kinsey and Big Flat creeks were burned. Spanish Creek was not burned. The error bars are 95% confidence intervals.
Figure 32: The number of juvenile steelhead captured in Spanish, Kinsey, and Big Flat creeks by year. The percentage of the juvenile steelhead population comprised of age 0+ fish is labeled for each year.
Figure 33: Juvenile steelhead log mass-log fork length regression lines for Spanish, Kinsey, and Big Flat creeks for all years that data was collected.
Figure 34: The estimated juvenile steelhead mass-fork length regression slopes for all years fork length and weight data were collected in Spanish, Kinsey, and Big Flat creeks. Error bars are 95% confidence intervals.
There were no significant differences among years in the mass-fork length regression slope in Spanish and Kinsey creeks (Figures 33, 34). There were significant differences among years in Big Flat Creek (Figure 34). The mass-fork length regression slope in Big Flat Creek was significantly different in a negative direction the first and second years post-fire compared to the pre-fire slopes (Figure 34). In Big Flat Creek, it appears that the mass-fork length regression slope was recovering towards the pre-fire values, with the slope increasing during the second and third years’ post-fire after the initial decrease the first year post-fire (Figure 34). It also does not appear that density dependence is playing a role in the observed decrease in summer body condition (slope) post-fire in Big Flat Creek. Lowest observed density of juvenile steelhead in Big Flat Creek occurred in 2004, while the highest observed density occurred in 2006 (Figure 35) when the mass-fork length regression slope was not significantly different than the pre-fire slope in 2003 (Figure 34). However, age 1+ juvenile steelhead made up a higher proportion of the population in 2004. To determine whether the higher proportion of age 1+ juvenile steelhead may be biasing the mass-fork length regression slope, the data for Big Flat Creek was subset into age 0+ and age 1+ fish and then the mass-fork length regression slopes for each age class for each year were calculated (Figure 36). The estimated mass-fork length regression slopes with 95% confidence intervals were calculated for each age class for each year in Big Flat Creek. When the mass-fork length regression slopes are compared among years by age class in Big Flat Creek it appears that fire had an effect on the summer body condition of the age 0+ age class in Big Flat Creek but not on the age 1+ age class.
Figure 35: The density (number per m²) of juvenile steelhead trout in the study reaches of Spanish, Kinsey, and Big Flat creeks, King Range National Conservation Area, California in the years that data was collected between 1999 and 2006. Spanish Creek was not burned; Kinsey and Big Flat creeks were burned.
Figure 36: Juvenile steelhead log mass-log fork length regression lines for age 0+ and age 1+ age classes in Big Flat Creek for all years that data was collected.
The estimated mass-fork length regression slope for age 0+ juvenile steelhead in Big Flat Creek in 2004 was significantly different in a negative direction compared to the pre-fire estimated slopes (Figure 37). It appears that there was a temporary decrease in the summer body condition of age 0+ juvenile steelhead in Big Flat Creek the first year post-fire. There were no significant differences among years for the estimated mass-fork length regression slopes for the age 1+ age class in Big Flat Creek (Figure 38).

The mixed-effects models for juvenile steelhead fork length provided evidence that fork length changed post-fire in the burned streams. There was strong evidence that fork lengths of age 0+ juvenile steelhead were longer post-fire in the burned streams (P=0.0048), while there was no evidence that fork length of age 0+ juvenile steelhead were longer post-fire in the unburned stream (P=0.5312) (Tables 9, 10; Figure 39). There was moderate evidence that fork length of age 1+ juvenile steelhead were longer post-fire in the burned streams (P=0.0365), while there was no evidence that fork length of age 1+ juvenile steelhead were longer post-fire in the unburned stream (P=0.4466) (Tables 9, 10; Figure 39). Overall all streams and years, juvenile steelhead fork length decreased with increasing density (Figure 40). Regression of log(FL) against log(density) yielded a significant relationship (P = 0.0006) with an adjusted R^2 of 0.68. However, when the fork length data are compared with the data balanced so that only years that have data for all three study streams are compared, it appears that there was a fire effect on fork length.
Figure 37: The estimated juvenile steelhead mass-fork length regression slopes for the age 0+ age class in Big Flat Creek for all years that fork length and weight data were collected. Error bars are 95% confidence intervals.
Figure 38: The estimated juvenile steelhead mass-fork length regression slopes for the age 1+ age class in Big Flat Creek for all years that fork length and weight data were collected. Error bars are 95% confidence intervals.
Figure 39: Average fork lengths (mm) of 0+ and 1+ juvenile steelhead trout in Spanish, Kinsey, and Big Flat creeks. Error bars are 95% confidence intervals.
Table 9: The fixed-effects estimates for the mixed-effects models for fork length of age 0+ and age 1+ juvenile steelhead. SE = standard error of estimate, DF = degrees of freedom.

<table>
<thead>
<tr>
<th>Effect</th>
<th>Fire</th>
<th>Burn</th>
<th>Estimate</th>
<th>SE</th>
<th>DF</th>
<th>t-value</th>
<th>P-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Age 0+</td>
<td>Intercept</td>
<td>82.43</td>
<td>11.89</td>
<td>1.05</td>
<td>6.93</td>
<td>0.084</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Burn</td>
<td>Burned</td>
<td>-15.54</td>
<td>14.59</td>
<td>1.06</td>
<td>-1.06</td>
<td>0.472</td>
</tr>
<tr>
<td></td>
<td>Fire</td>
<td>Post</td>
<td>-2.23</td>
<td>3.41</td>
<td>8.2</td>
<td>-0.65</td>
<td>0.531</td>
</tr>
<tr>
<td></td>
<td>Fire*Burn</td>
<td>Post</td>
<td>Burned</td>
<td>12.80</td>
<td>4.32</td>
<td>7.87</td>
<td>2.97</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Effect</th>
<th>Fire</th>
<th>Burn</th>
<th>Estimate</th>
<th>SE</th>
<th>DF</th>
<th>t-value</th>
<th>P-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Age 1+</td>
<td>Intercept</td>
<td>139.52</td>
<td>8.71</td>
<td>1.59</td>
<td>16.02</td>
<td>0.010</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Burn</td>
<td>Burned</td>
<td>-17.79</td>
<td>10.88</td>
<td>1.71</td>
<td>-1.63</td>
<td>0.264</td>
</tr>
<tr>
<td></td>
<td>Fire</td>
<td>Post</td>
<td>5.65</td>
<td>7.06</td>
<td>8.02</td>
<td>0.80</td>
<td>0.447</td>
</tr>
<tr>
<td></td>
<td>Fire*Burn</td>
<td>Post</td>
<td>Burned</td>
<td>8.10</td>
<td>8.83</td>
<td>7.54</td>
<td>0.92</td>
</tr>
</tbody>
</table>
Table 10: Contrasts from the mixed-effects models for the fork length of Age 0+ and Age 1+ juvenile steelhead trout in burned and unburned streams during the pre- and post-fire time periods. DF = degrees of freedom.

<table>
<thead>
<tr>
<th></th>
<th>Age 0+</th>
<th></th>
<th></th>
<th>Age 1+</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Estimate</td>
<td>Standard Error</td>
<td>DF</td>
<td>t-value</td>
<td>P-value</td>
</tr>
<tr>
<td>NoPre-YesPost</td>
<td>2.23</td>
<td>3.41</td>
<td>8.20</td>
<td>0.65</td>
<td>0.5312</td>
</tr>
<tr>
<td>YesPre-YesPost</td>
<td>-10.58</td>
<td>2.65</td>
<td>7.35</td>
<td>-3.99</td>
<td>0.0048</td>
</tr>
</tbody>
</table>
Figure 40: The log of the mean fork length (mm) of juvenile steelhead vs. the log of the density of juvenile steelhead (number per m²) in Spanish, Kinsey, and Big Flat creeks.
compared to the pre-fire year 2000 (Table 11). The mean fork length of age 0+ juvenile steelhead increased during the post-fire period in Kinsey and Big Flat creeks, the burned streams, but not in Spanish Creek, the unburned stream when compared to the pre-fire year 2000 (Table 11).
Table 11: Post-fire (2004-2006) mean fork length (mm) of age 0+ and age 1+ juvenile steelhead in Spanish, Kinsey, and Big Flat creeks compared to their pre-fire (2000) mean fork length (mm). SE = standard error.

<table>
<thead>
<tr>
<th>Creek</th>
<th>Post-fire Mean Fork Length (mm)</th>
<th>SE</th>
<th>Pre-fire Mean Fork Length (mm)(2000)</th>
<th>SE</th>
<th>Difference (Post-Pre)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Age 0+</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Spanish</td>
<td>81</td>
<td>0.58</td>
<td>78</td>
<td>0.79</td>
<td>3</td>
</tr>
<tr>
<td>Kinsey</td>
<td>69</td>
<td>0.55</td>
<td>60</td>
<td>1.10</td>
<td>9</td>
</tr>
<tr>
<td>Big Flat</td>
<td>85</td>
<td>0.36</td>
<td>73</td>
<td>0.60</td>
<td>12</td>
</tr>
<tr>
<td></td>
<td>Age 1+</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Spanish</td>
<td>144</td>
<td>2.67</td>
<td>129</td>
<td>3.71</td>
<td>15</td>
</tr>
<tr>
<td>Kinsey</td>
<td>127</td>
<td>2.58</td>
<td>116</td>
<td>2.06</td>
<td>11</td>
</tr>
<tr>
<td>Big Flat</td>
<td>141</td>
<td>1.31</td>
<td>123</td>
<td>1.75</td>
<td>18</td>
</tr>
</tbody>
</table>
DISCUSSION

Changes in some steelhead metrics and habitat metrics were observed between pre- and post-fire periods in both burned streams. However, not all changes could be attributed to fire. Many of the analyses had little power to detect a fire effect due to limited pre-fire data. Limited pre-fire data also inhibited interpretation of patterns in pre- and post-fire yearly data. Abundance of juvenile steelhead in both Big Flat Creek and Kinsey Creek was greater three years post-fire than it was pre-fire. However, with only one year of pre-fire data it cannot be concluded these post-fire increases were due to the fire.

The mixed-effects model provided evidence that mean fork length of age 0+ and age 1+ juvenile steelhead increased significantly in both burned streams post-fire but not in the unburned stream. However, when the post-fire mean fork length of age 0+ and age 1+ juvenile steelhead were compared in the study streams using a balanced dataset, it appeared that fire had a positive effect on mean fork length of age 0+ juvenile steelhead and did not affect the mean fork length of age 1+ juvenile steelhead. Mean fork length among steelhead in all streams was significantly negatively related to density. While fire could have contributed to changes in fork length by altering productivity or the dates of fry emergence from gravel, there was minimal riparian burning from the Honeydew Fire in Kinsey and Big Flat creeks. Thus it appears that density, and not increased productivity from greater solar radiation or the introduction of nutrients, was the primary
 mechanism influencing fork length. In either case, steelhead smolts that enter the marine environment at a larger size have higher smolt-to-adult survival (Ward et al. 1989, Bond 2006), which could result in a higher adult return rate to the burned streams depending on the overall smolt production for a given year. Post-fire increase in fish size may persist for many years after fire. In Idaho, rainbow trout in burned streams had significantly longer mean total length compared to unburned streams 12-14 years post-fire (Koetsier et al. 2007).

It appears that the Honeydew Fire may have negatively affected the summer body condition of age 0+ juvenile steelhead in Big Flat Creek the first year post-fire, while age 1+ juvenile steelhead were unaffected. The first year after the fire was a wet year, making post-fire erosion events highly likely in the Big Flat Creek watershed, especially since the majority of this watershed burned. This post-fire erosion in Big Flat Creek could have contributed to prolonged high turbidity in the winter and spring, thereby reducing the feeding efficiency of juvenile steelhead. However, with only the age 0+ class being affected and not the age 1+ age class, there is a possibility that the decrease in summer body condition the first year post-fire in the age 0+ age class was a result of natural variability and not a fire effect. More pre- and post-fire data would provide more confidence that this is a true fire effect. It is possible that age 0+ juvenile steelhead could be more impacted by higher turbidity than age 1+ juvenile steelhead. Mechanisms related to the fire other than turbidity could also influence the observed summer body condition of age 0+ juvenile steelhead the first year post-fire. Although summer body
condition of age 0+ juvenile steelhead appeared to be reduced in Big Flat Creek initially after the fire, it appeared to recover the following year in which it was not significantly different than in the pre-fire year. Summer body condition of juvenile steelhead in Kinsey Creek did not show a response to fire, which could be due to it having been less disturbed by the Honeydew Fire than Big Flat Creek. In Idaho, body condition of rainbow trout did not differ between burned and unburned streams 12-14 years post-fire (Koetsier et al. 2007).

Proportion of age 1+ steelhead that comprised the juvenile steelhead population was greater post-fire than it was pre-fire in Big Flat Creek. This may have been the result of carrying capacity for age 1+ juvenile steelhead increasing or increasing survival from age 0+ to age 1+. The increase in the proportion of age 1+ steelhead was most pronounced in Big Flat Creek, the most completely burned stream, suggesting that steelhead may have some resilience to fire. Age 0+ steelhead were also found in abundance in the summer after the fire which suggests that the fire did not significantly affect spawning success. Similar to other studies of juvenile salmonid abundance in burned areas (Howell 2006, Sestrich et al. 2011, Burton 2005), abundance of juvenile steelhead in Big Flat (2004) and Kinsey Creek decreased initially after the fire in 2004 and 2005, then increased in 2006. Natural disturbance in relatively pristine systems is believed to increase long-term productivity compared to systems in which disturbance is suppressed or anthropogenically altered (Reeves et al. 1995). Similar post-fire comparison studies have found increased juvenile salmonid densities in streams
following fire (Howell 2006, Burton 2005; Gresswell 1999). Novak and White (1989) found that within two years after a fire, numbers and biomass of rainbow trout increased by 55% and 51% in a Montana stream. Rieman et al. (1997) found that fish densities three years post fire were greater in burned reaches than in similar reaches that did not burn in an Idaho stream. Burton (2005) found that post-fire floods increased fish productivity in Idaho streams.

In Kinsey Creek in 2005 there was a nearly missing age 0+ steelhead age class (1 out of 61 sampled). Adult steelhead may have had very limited access to the stream or spawning success was very low. There was no evidence that this nearly missing age class was due to the fire.

The abundance of juvenile steelhead in Big Flat Creek in 2004 is anomalous compared to the pattern of juvenile steelhead abundance in the other study streams during the post-fire years. If the abundance of juvenile steelhead in Big Flat Creek had conformed to the pattern of Spanish and Kinsey creeks then the abundance in 2004 would have been greater than the abundance in 2005 and similar to the abundance in 2006. Big Flat Creek shows a pattern of increasing abundance during the post-fire years while Spanish and Kinsey creeks have up-down-up patterns, suggestive of a more neutral trend. It is possible that the fire may be responsible for the juvenile steelhead abundance in 2004 in Big Flat Creek being out of step with the pattern in the other study streams.
There were also some observed changes in fish habitat in burned study streams post-fire. Abundance per 100 m, volume per 100 m, and jams of LWD increased in the third year post-fire in Big Flat Creek and the first year post-fire in Kinsey Creek compared to their pre-fire values. However, LWD volume per 100 m and number of pieces per 100 m in Kinsey Creek returned to near pre-fire values in subsequent years. In all study streams, LWD jams contributed a lot to the overall volume of LWD observed. The large increase in LWD volume per 100 m, particularly through several large jams, in Big Flat Creek in 2006 may be partly attributed to strong winter storms creating high peak flows and a wind storm with hurricane force winds that occurred that winter, in combination with Big Flat Creek having a majority of its watershed burned. These events in 2006 could have had a greater effect on LWD in Big Flat Creek than in Kinsey Creek because Big Flat was more disturbed by the fire. Big Flat Creek would be expected to have more variability in wood volume over time due to its larger size and hence larger stream power. The observed changes in LWD volume post-fire in Big Flat Creek could be due to natural variability and unrelated to the fire. Total pool surface area increased in Big Flat Creek in 2006 (Table 2), coincidental with an increase in LWD pieces and jams. Juvenile steelhead density is greatest in pools in Big Flat Creek (pers. obs.), so the creation of pool habitat would be beneficial for the steelhead in Big Flat Creek. The presence of LWD in streams has been shown to be important in creating and maintaining high quality habitat for juvenile anadromous salmonids in the Pacific Northwest (Beechie and Sibley 1997, Fausch and Northcote 1992, Bilby and Ward 1991, Rosenfeld et al. 2000). In Idaho, in-stream habitat conditions were improved by post-fire
floods which exported fine sediments and imported gravel, cobble, woody debris, and nutrients (Burton 2005). Chen et al. (2005) found the highest LWD volumes in streams that had burned 40 years before the study period (compared to previously harvested and old growth streams). Much of the large woody debris in the study streams is Douglas-fir, which being a conifer persists as LWD for a long time before rotting (Hyatt and Naiman 2001).

In Idaho, post-fire floods exported almost all of the instream LWD out of small study streams into higher order channels, causing the loss of important salmonid habitat (Burton 2005). The loss of major amounts of LWD due to post-fire floods did not happen in this study. Post-fire LWD volume in Kinsey Creek dropped the second and third years post-fire but loss of LWD during these same years did not occur in Spanish and Big Flat creeks suggesting that this loss of LWD was not due to a flood.

Increased volume of large woody debris in Big Flat Creek may help create more pools but there was some evidence that post-fire erosion reduced the depth of pools in Big Flat Creek, particularly the first year post-fire. However, there was no evidence that fire reduced the depth of pools in Kinsey Creek. Pools in Kinsey Creek may have not shown a response to the fire due to the lower level of fire disturbance in Kinsey Creek’s watershed. The first year post-fire (2004) was a wet year, which increases the probability of post-fire erosion events which would then contribute sediment to pools. The deepest observed pools in Big Flat Creek all occurred in the upper section of Big Flat Creek the year pre-fire. However, some relatively deep pools were formed in the lower part of Big
Flat Creek the third year post-fire (2006). During this study, highest peak flows in the Mattole River, the nearest gaged river, occurred in 2006. High flows combined with the fire could have contributed to channel changes in Big Flat Creek and the formation of new pools in the lower part of the stream in 2006. The channel changes in Big Flat Creek could also be normal variation and unrelated to fire.

Daily mean and maximum water temperature in Kinsey Creek appeared to be higher the second and third years post-fire after decreasing the first year post-fire, while in Big Flat Creek water temperature was lower during the first and third years’ post-fire. The apparent decrease in water temperature post-fire in Big Flat Creek is anomalous. Since the majority of the Big Flat Creek watershed burned it was expected that the greater degree of disturbance in this stream would lead to a greater increase in water temperature than in the less severely burned watershed. The opposite was the case, with Big Flat Creek remaining cooler the third year post-fire while water temperature in Kinsey Creek increased. This could be a result of the minimal riparian burning in either creek, which would limit an increase in solar radiation which is the mechanism for post-fire increases in water temperature. The observed water temperature patterns could also be normal variation; with more pre-fire data it would be possible to say with more certainty that the changes were a fire effect. The accuracy of the temperature loggers used was around 0.5°C. The observed changes in difference in water temperature between a burned stream and the control stream were less than 1.6°C, which could make some of the observed changes possibly related to temperature logger bias. There are a
couple of ways that temperature loggers could vary that have implications for the
reliability of the results. The temperature loggers could vary in their offset, how much
warmer or cooler they read the water temperature compared to each other, over all
temperatures. The offset of the temperature loggers could also change with temperature
so that the offset is not constant. If the offset of the temperature loggers is constant at all
temperatures then the analysis that I used would be valid. However, if the offset of the
temperature loggers changes with temperature then my analysis is compromised. It is not
known what kind of variability the loggers used in this study had. Multiple loggers were
not placed at each site to allow the nature of the variability of the loggers to be examined.
With the nature of the variability of the temperature loggers unknown, the reliability of
the water temperature results are questionable. In addition, the water temperature results
for the two burned streams did not respond similarly relative to the unburned stream,
which further suggests that the observed water temperature results may be an artifact of
the temperature loggers.

The observed increase in the mean of the daily maximum water temperature (June
20 to September 20) in Kinsey Creek was less than 1°C. This is in contrast to other
studies that found that one of the major effects of fire was an increase in maximum water
temperature greater than 1°C (Dunham et al. 2007, Sestrich et al. 2011). Water
temperatures in the upper locations of the study streams were in the favorable range for
juvenile steelhead during all years data were collected, even post-fire in the burned
streams. It is suspected that water temperature increases were minimal in this study for
two reasons. First, all three study streams are heavily influenced by marine conditions. Second, the riparian zones of the study streams were only partially burned; riparian zones most likely kept the streams well shaded and served as fire buffers. Due to minimal anthropogenic impacts, the riparian zones in the study streams were in good condition, having well developed canopies and relatively moist conditions, which may have prevented extensive burning. Riparian zones in other areas of the Pacific Northwest have had limited amounts of burning compared to the adjacent uplands and so have acted as fire buffers (Pettit and Naiman 2007). Riparian zones in Douglas-fir forests in Washington State have been shown to burn with less frequency than adjacent side-slope forests (Everett et al. 2003). Riparian reserves in Douglas-fir forests in the Klamath Mountains have median fire return intervals approximately double those of nearby upland sites (Skinner 2003).

Global climate change is projected to increase the frequency of wildfires, and there is evidence that this is already occurring (Westerling et al. 2006). Understanding how more intense and severe wildfires in the future could impact native fish populations, especially those listed under the ESA, and how to start managing forests to minimize this threat, is an important topic (Mote et al. 2003, Rieman and Clayton 1997). A lack of extensive pre-fire data in this study prevents any conclusions about fire’s effect on juvenile steelhead, but the results of this study suggest that wildfire may have a minimal impact on fish populations in relatively undisturbed coastal watersheds. Future studies would benefit from having more extensive pre-fire data, which would enable them to
present conclusive results about fire effects. In relatively undisturbed areas, wildfire suppression may not be the top priority, and the disturbance due to the wildfire may be important for forest and fish health (Rieman et al. 2000). This study also suggests that using prescribed fires that minimally impact riparian areas could be an important tool in improving forest health and making them more resilient to the projected impacts of climate change, while still having little effect on native fishes.
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Brakensiek, K.E. 2002. Abundance and survival rates of juvenile coho salmon (Oncorhynchus kisutch) in Prairie Creek, Redwood National Park. Master’s thesis. Humboldt State University, Arcata, California.


Engle, R.O. 2005. Distribution and summer survival of juvenile steelhead in two streams within the King Range National Conservation Area, California. Master’s thesis. Humboldt State University, Arcata, California.


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PERSONAL COMMUNICATIONS

Duffy, W.G. 2007. Leader, United States Geological Survey, California Cooperative Fish and Wildlife Research Unit, Humboldt State University, 1 Harpst St. Arcata, California 95521.
APPENDICES
Appendix A. Summary of habitat characteristics in Spanish, Kinsey, and Big Flat creeks, King Range National Conservation Area, California. Data from 1999 and 2000 are modified from data collected by Engle (2005). Maximum depth in Spanish Creek in 1999 was only collected in pools. SA = surface area. SD = standard deviation.

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Appendix D. Fork length distribution of juvenile steelhead in Big Flat Creek, King Range National Conservation Area, California from 2003-2006.