LOWER YUBA RIVER ACCORD
MONITORING AND EVALUATION PROGRAM

O. mykiss Adult Spawning
Physical Habitat in the Lower Yuba River

APRIL 2014—FINAL REPORT

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Foreword

On March 18, 2008, the State Water Resources Control Board (SWRCB) approved a consensus-based, comprehensive program to protect and enhance approximately 37.1 km of aquatic habitat in the lower Yuba River, extending from Englebright Dam downstream to the river’s confluence with the Feather River near Marysville. This program is known as the Lower Yuba River Accord (Accord).

The lower Yuba River Accord (Accord) consists of a Fisheries Agreement and several other elements. The Fisheries Agreement includes descriptions of the River Management Team (RMT), the River Management Fund (RMF), and the Monitoring and Evaluation Program (M&E Program). The Fisheries Agreement in its entirety can be found on the Accord RMT website (http://www.yubaaccordrmt.com). The RMT Planning Group (hereafter referred to as the RMT) includes representatives of the California Department of Fish and Wildlife (CDFW), National Marine Fisheries Service (NMFS), Pacific Gas and Electric Co. (PG&E), U.S. Fish and Wildlife Service (USFWS), California Department of Water Resources (DWR), Yuba County Water Agency (YCWA), and one representative for the four non-government organizations (Friends of the River, South Yuba River Citizen’s League, The Bay Institute, and Trout Unlimited) that are parties to the Fisheries Agreement. The RMT developed the M&E Program to guide the efficient expenditure of RMF funds to evaluate the biological provisions of the Fisheries Agreement of the Yuba Accord.

Multiple studies were identified by the RMT to address the specific analytics that are necessary to evaluate the performance indicators detailed in the M&E Program. The purpose of this report is to document findings for some of the performance indicators that are identified in the M&E Program.

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Executive Summary

The primary purpose of the M&E Program is to provide the monitoring data necessary to evaluate whether implementation of the Yuba Accord flow schedules are “protective” of the fish and aquatic habitat resources of the lower Yuba River. The M&E Program framework was designed to address two over-arching goals.

- Evaluate whether implementation of the Yuba Accord maintains fish in “good condition” and promotes “viable salmonid populations” in the lower Yuba River.
- Identify and evaluate relationships between flows and water temperatures resulting from implementation of the Yuba Accord, and fish population and aquatic habitat attributes.

Anadromous salmonids in the lower Yuba River include fall-run and phenotypic spring-run Chinook salmon (*Oncorhynchus tshawytscha*), and steelhead (*O. mykiss*). “Steelhead” is the name commonly applied to the anadromous form of the biological species *O. mykiss*. The physical appearance of *O. mykiss* adults and the presence of seasonal runs and year-round residents indicate that both anadromous (steelhead) and resident rainbow trout exist in the lower Yuba River downstream of Englebright Dam. Steelhead exhibit perhaps the most complex suite of life history traits of any species of Pacific salmonid. Members of this species can be anadromous or freshwater residents and, under some circumstances, members of one form can apparently yield offspring of another form (YCWA 2010).

A key component in assessing the viability of the *O. mykiss* population in the lower Yuba River is to characterize and quantify the relationship between physical river conditions and ecological functions for each life stage. This study focused on determining the physical factors that are important for spawning *O. mykiss*. The influences of physical conditions and processes on spawning habitat selection were assessed by developing a predictive physical habitat model based on microhabitat suitability, and by exploring physical-biological linkages at larger scales to gain a comprehensive understanding of *O. mykiss* spawning habitat in the lower Yuba River.

Past work in the lower Yuba River by the RMT provided the comprehensive, spatially-explicit physical and biological datasets and a 2D hydrodynamic model that were fundamental in assessing *O. mykiss* spawning habitat. Using spatially distributed 2D hydrodynamic model results, substrate mapping, and a survey of *O. mykiss* redds from two years of observation, microhabitat suitability criteria were used to create physical habitat models. These physical habitat models were developed at the microhabitat scale based on several representations of the suitability of hydraulic conditions, as well as hydraulic conditions in conjunction with substrate size suitability. One physical microhabitat model (RMT-12) combined the RMT water depth and velocity habitat suitability curves (HSCs), and a new substrate HSC developed by the RMT, with the simulated hydraulic conditions from the lower Yuba River 2D hydrodynamic model and a map of mean substrate diameter. According to the RMT-12 physical habitat model, there is a strong preference for *O. mykiss* spawning in areas characterized by a mean water column velocity of about 1.18-2.25 ft/s, and water depths of 1.25-2.76 ft. The substrate range preferred for *O. mykiss* spawning was within medium gravel/small cobble size class (32-90 mm), with a
strong preference for the narrow range of 50-60 mm mean substrate size, and with slightly lower preference for mean substrate size from 90-200 mm.

The physical microhabitat models were tested for their ability to predict observed spawning locations with the highest accuracy. Several tests (including the Mann-Whitney U test and the Forage Ratio test) were used to quantitatively compare the models at three discharges corresponding to groups of redd observations made at similar discharges. RMT-12 was found to be the best performing physical microhabitat model.

The RMT-12 physical habitat model correctly identified 46-67% of redds from the three redd groups as located within preferred habitat. Of those redds not located within preferred habitat, 18-64% of them were located within 5 feet of preferred microhabitat. This physical habitat model was bioverified at three discharges, and can be used not only to predict areas of high quality microhabitat for spawning, but also to quantify the availability of spawning habitat in the lower Yuba River.

The best physical habitat model (RMT-12) was used to quantify the amount of spawning habitat available (weighted usable area, or WUA) as a function of discharge at three spatial scales: the entire lower Yuba River segment; above and below Daguerre Point Dam; and by geomorphic reach. The WUA-discharge relationships showed that the highest WUA values at all spatial scales were predicted at about 600-700 cfs. There was a significant decrease in \textit{O. mykiss} spawning habitat at discharges above approximately 1,000 cfs.

In assessing \textit{O. mykiss} spawning habitat in the lower Yuba River, this study found that there were other physical variables at larger spatial scales that contributed to explaining spawning habitat utilization. A marked preference was observed for the two most upstream alluvial reaches of the lower Yuba River, Timbuctoo Bend Reach and Parks Bar Reach, accounting for 92% of all redds observed. The amount of high quality microhabitat available in these two preferred reaches was equal to or less than in other reaches, and water temperature throughout the lower Yuba River during the \textit{O. mykiss} spawning season was uniformly cool. Hence, the preference for \textit{O. mykiss} spawning in the two most upstream alluvial reaches in the lower Yuba River may be associated with the instinctive behavioral impulse to migrate to upstream areas for spawning, rather than in response to physical variables.

The preferred morphological units (MUs) for \textit{O. mykiss} spawning changed with increasing discharge, demonstrating that \textit{O. mykiss} shift spawning to different MUs in order to utilize their preferred hydraulic conditions. The preferred MUs changed from riffle transition at 880 cfs, to riffle, riffle transition, and fast glide at 1,000 cfs, and riffle transition, slow glide, and point bar at 1,300 cfs. Of the MUs that were preferred at one of the discharges and not at the others, it was often the water velocity values that were not within the range of high suitability (from the RMT water velocity HSC) and made the MU unsuitable at those discharges.

Size of preferred MUs was also explanatory of spawning habitat utilization. Small riffle transitions (1,000-10,000 ft²) were preferred at 880 cfs, 1,000 cfs, and 1,300 cfs. Both small (1,000-5,000 ft²) and very large (90,000-10,000 ft²) riffles, as well as small (1,000-5,000 ft²) and
large (70,000-80,000 ft²) fast glides were preferred at 1,000 cfs, and large slow glides (60,000-80,000 ft²) were preferred at 1,300 cfs.

The size of patches of high quality microhabitat was another physical variable that helped describe *O. mykiss* spawning habitat utilization. Patches of highest habitat quality and patches of preferred habitat both were evaluated. Various patch sizes were preferred between the redd groups at the three discharges tested, but the 1,000-5,000 ft² patch size was preferred across all three redd groups. Larger patch sizes were preferred when considering preferred habitat rather than just the highest quality habitat, as 5,000-10,000 ft² patches were a preferred size for all three redd groups, in addition to preference for patches from 100,000 ft² through 140,000 ft² amongst the three redd groups.

Downcutting was the only topographic change process for which any *O. mykiss* spawning preference was demonstrated. Processes such as island emergence and sub-avulsion were avoided. In general, processes of fill were avoided, while scour processes were preferred.

The spatial relationships between reds, and between redd location and preferred microhabitat, were also investigated. For the three redd groups, between 55-88 % were located either within areas of preferred microhabitat or within 5 ft of preferred microhabitat. A majority of reds (72 %) also were located within 16 ft of another redd. The observed tendency for proximal spawning suggests that *O. mykiss* spawning site selection may be influenced by social behavior, as well as by the suitability of physical habitat variables.

This study provided a comprehensive assessment of *O. mykiss* spawning physical habitat, from microhabitat quality predictions at the 3-by-3 ft cell size to physical-biological linkages at various spatial scales and analysis units. *O. mykiss* spawning site selection was found to be highly predictable by multivariate hydraulic and geomorphic expressions. The predictive physical habitat model produced by this research provides the means to characterize and identify areas of high quality microhabitat for *O. mykiss* spawning, and habitat availability through the use of habitat-discharge relationships. The amount of high quality habitat is highly dependent upon discharge, and decreases rapidly once flows exceed maximum habitat availability. There is far more preferred microhabitat available in the lower Yuba River under present flow regimes than is used by the current *O. mykiss* population.
Frequently Asked Questions

*Are the locations in the lower Yuba River that adult *O. mykiss* use to create redds merely random occurrences?*

No. Redd locations are not randomly located in the lower Yuba River, but are highly organized and patterned. The nonrandom distribution of redds can be explained by interdependent physical variables across spatial scales. This report reveals those explanations.

*Are the locations in the lower Yuba River that adult *O. mykiss* use to create redds equally distributed down the length of the river?*

No. There is a very skewed distribution of *O. mykiss* redd occurrences towards the upstream reaches of the lower Yuba River. Redd locations preferentially occur 3.85-4.05 times more often than random chance would expect in Timbuctoo Bend Reach, which is the farthest upstream alluvial geomorphic reach. There also is a slight, but statistically significant preference for the Parks Bar Reach (located between Highway 20 Bridge and the confluence with Dry Creek) for redds constructed at about 1,300 cfs.

*I thought anadromous salmonids couldn’t get past Daguerre Point Dam - isn’t that a problem?*

Not for *O. mykiss* adults. Fish passage is not a focus of this report, but multiple fish-monitoring methods (e.g., the Vaki Riverwatcher™ at Daguerre Point Dam, carcass surveys upstream of Daguerre Point Dam, and redd surveys above Daguerre Point Dam) all show that adult *O. mykiss* pass upstream of the dam. In fact, the vast majority of redds were observed upstream of the dam.

*Are the spawning locations equally distributed along the Timbuctoo Bend and Parks Bar reaches?*

No. Redds are highly clustered, with large unoccupied areas in between individual redds.

*That is because spawners always use riffles, right?*

Actually, no. *O. mykiss* spawners used a large variety of fluvial landforms, or morphological units (MUs) for spawning, and the most utilized landforms differed depending on the discharge at which the redds were built. A previous report by Wyrick and Pasternack (2012) documents the abundance, diversity, and distribution of fluvial landforms in the lower Yuba River. *O. mykiss* spawn during the winter, and the high variability in winter flows meant that redds were observed over a wide range of discharges. Redd observations were grouped based on similarity of discharge when they were observed, with three substantial groups observed at flows about 880 cfs, 1,000 cfs, and 1,300 cfs, referred to as Q<sub>low</sub>, Q<sub>mid</sub>, and Q<sub>high</sub>, respectively. The most utilized fluvial landform at Q<sub>low</sub> was riffle transition, while fast glide, riffle, and riffle transition were...
most utilized at Q_{mid}. At Q_{high}, riffle transition and slow glide were the most frequently used landforms for spawning. While the utilization of landforms differed with discharge, the hydraulic conditions within each of the most utilized landforms were fairly consistent, suggesting that *O. mykiss* shift to different landforms as discharge changes in order to utilize the hydraulic conditions that are preferred for spawning. In addition to the type of MU preferred at each discharge tested, the size of preferred MUs also was found to be a factor in spawning site selection by adult *O. mykiss*. At all three discharges tested, spawners generally preferred smaller riffle transitions over a few large ones. There was preference for both large and small riffles and fast glides at Q_{mid}, and large point bars were preferred at Q_{high}. There was no statistically significant avoidance of any MUs at Q_{low}, but slackwater was avoided at Q_{mid} and runs were avoided at Q_{high}.

**Are there any processes of topographic change in the lower Yuba River that impact *O. mykiss* spawning locations?**

** Generally, no.** The analysis was conducted to determine if there was any preference for specific topographic change processes that had influenced the formation and change in landforms from 1999-2008. There was no significant preference or avoidance of any topographic change processes within MUs for *O. mykiss* spawning. In general, areas that underwent scour processes were more utilized than areas that had experienced fill over the previous decade.

**What is the best predictor of the point locations where adult *O. mykiss* choose to spawn in the lower Yuba River?**

**The RMT-12 CHSI physical habitat model.** The physical microhabitat model (RMT-12) combined the RMT water depth and velocity habitat suitability curves (HSCs), and a new substrate HSC developed by the RMT, with the simulated hydraulic conditions from the lower Yuba River 2D hydrodynamic model and a map of mean substrate diameter. The suitable ranges of water depth and velocity values were determined from habitat suitability curves developed from *O. mykiss* spawning observations on the lower Yuba River. According to the RMT-12 physical habitat model, there is a strong preference for *O. mykiss* spawning in areas characterized by a mean water column velocity of about 1.18-2.25 ft/s, and water depths of 1.25-2.76 ft. The substrate range preferred for *O. mykiss* spawning was within medium gravel/small cobble size class (32-90 mm), with a strong preference for the narrow range of 50-60 mm mean substrate size, and with slightly lower preference for mean substrate size from 90-200 mm.

This combination of predictors (RMT-12) met 3 different types of “bioverification” (i.e., combined habitat suitability difference test, Mann-Whitney U test, and Forage Ratio test). Resulting predictions are statistically significant above the 95% confidence level.

**Exactly how well did this predictive model work? How many redds did it correctly locate?**
Well. Compared against the $Q_{\text{low}}$, $Q_{\text{mid}}$, and $Q_{\text{high}}$ redd groups from the 2010 and 2011 redd surveys that represented 72% of the 261 total number of redd observations, the RMT-12 CHSI physical habitat model correctly identified 67%, 67%, and 46% of redds as in preferred microhabitat for each of the redd groups, respectively.

Is 46-67% predictive success “good”?

Yes. Think about it in terms of money. If you made a bet where you won $100 on every redd located in what was predicted to be preferred habitat and lost $100 on every redd located in what was predicted to be avoided or tolerated habitat, then you would net $2,500 over two years. That is a bet that any rational gambler would gladly take.

What about the 33-54% of redds that were not successfully predicted - don’t they matter?

Of course. For the redds that were not located within areas of preferred microhabitat, 12-43% were located within 3 ft of preferred habitat patches, and 18-64% were located within 5 ft, which is reasonably close to the prediction. This accounts for 52-81% of redds located either within preferred habitat or within 3 ft of preferred habitat, and 55-88% located within preferred habitat or within 5 ft of preferred habitat. The spawning activity near areas designated as preferred habitat may still be influenced by the suitable microhabitat conditions, but their location just outside of the preferred areas may result from the social behavior of the fish to cluster around other spawners, or from small errors in the delineation of the habitat quality class patches due to the propagation of uncertainty from the hydrodynamic model results and the substrate mapping efforts into the CHSI model values. This still leaves 12-45% of individual $O.\ mykiss$ spawners whose behavior could not be predicted based on microhabitat suitability of hydraulics and substrate size. These individuals may be selecting spawning locations based on other habitat characteristics beyond the microhabitat scale, or that we are not measuring or are difficult to model. There is also the chance that these individuals are simply selecting spawning sites based on random choices. Either way, the outcome is diverse life histories that may promote diversity and resilience in the population.

Is there a need for spawning habitat restoration projects in the lower Yuba River?

Not at this time. According to the physical habitat model presented in this study, there is an overabundance of preferred microhabitat available to spawning $O.\ mykiss$ in the lower Yuba River. Also, due to the timing of $O.\ mykiss$ spawning during the winter months, there is little competition with other salmonids for the use of suitable spawning habitat. There is plenty of unoccupied preferred habitat that is not used in any particular year, making the lower Yuba River capable of sustaining a much larger $O.\ mykiss$ spawning population than is currently present.
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1 INTRODUCTION

Oncorhynchus mykiss (O. mykiss) is a species of salmonid native to tributaries along the Pacific coasts of North America and Asia. Prior to dam construction, water development and watershed perturbations, the anadromous form of O. mykiss (i.e., steelhead) were distributed throughout the Sacramento and San Joaquin rivers (Busby et al. 1996; McEwan 2001). Steelhead currently occur in the upper Sacramento River and its tributaries, as well as the Stanislaus, Calaveras, Merced, and Tuolumne rivers (NMFS 2009). Naturally spawning populations of steelhead also occur in the Feather, American, Mokelumne, and Yuba rivers, but these populations have had substantial hatchery influence and their ancestries are not clear (Busby et al. 1996).

Historic Central Valley steelhead run sizes are difficult to estimate because of the lack of data, but McEwan (2001) suggested that steelhead run sizes may have approached one to two million adults annually. Due to the lack of sustained monitoring programs for steelhead throughout most of the Central Valley, there is a paucity of reliable data to estimate run sizes of steelhead, particularly wild stocks. Nonetheless, NMFS (2009) reported that over the last 30 years the steelhead populations in the upper Sacramento River have declined substantially. Central Valley O. mykiss were listed as a threatened species under the U.S. Endangered Species Act in 1998 (Good et al. 2005).

There are many factors contributing to the decline of steelhead in the Central Valley, although the single greatest stressor is thought to be the loss of spawning habitat due to the construction of impassable dams (McEwan 2001; Lindley et al. 2006). It is estimated that 80% of their historical spawning and rearing habitat has been lost due to the construction of dams (Lindley et al. 2006). Protection of the remaining spawning habitat is of utmost importance for the protection of the species. It is, therefore, important to understand O. mykiss spawning habitat preferences in order to ensure that these conditions are abundant and available in the sections of rivers that are still accessible.

Physical habitat is defined as a location with measurable, characteristic attributes where organisms perform a designated ecological function, usually during one of their life stages. Abundant and diverse aquatic physical habitats are necessary to sustain a population. Microhabitat features are point-scale measurements of physical habitat attributes that are utilized by organisms to perform an ecological function. These measurable and characteristic attributes stem from the interaction among hydrology, hydraulics, and geomorphology of the river. The most common attributes used to describe microhabitat are hydraulics (i.e., water depth and velocity), substrate size, water temperature, and cover. Today there is a consensus that the most important physical characteristics for Pacific salmonid spawning at the microhabitat scale are water depth, velocity, and substrate size, but more investigation is needed into the spatial patterns of these variables at different spatial scales, as well as the contributory roles of other variables.

This study was conducted to better understand the availability and utilization of physical habitat to determine the physical factors that are important in spawning site selection by O. mykiss in the lower Yuba River. Spawning habitat needs and preferences of O. mykiss are conjectured to be dependent on several physical variables, including water depth and velocity, substrate, and also...
factors of water quality. By studying the availability and utilization of physical habitat characterized by these various variables, a better understanding of the physical habitat conditions that are ecologically significant for spawning can be gained, as well as the ranges of these variables that are most suitable for spawning. The concept of habitat suitability leads to the determination of habitat quality and the characterization of various levels of habitat quality. Habitat suitability curves (HSCs) can be developed that assign different levels of suitability across the range of important physical habitat variables utilized. Developing and applying HSCs assists in determining the range of habitat quality and the abundance of high-quality habitat available in a stream. The combination of HSCs with transect-based hydraulic sampling or with one-dimensional (1D) or two-dimensional (2D) hydraulic models of a stream segment creates a physical habitat model that can provide an estimate of habitat availability on a larger scale (Waddle 2001). A greater understanding and knowledge of the physical habitat conditions suitable for *O. mykiss* spawning provides the opportunity to protect the quality and quantity of physical spawning habitat presently available in streams, and the ability to create these preferred spawning habitat conditions through river restoration or rehabilitation efforts.

1.1 Existing Spawning Habitat Literature

1.1.1 Salmonid Spawning Habitat Characterization

The selection of a spawning site by a salmonid is a complex process that is not fully understood, but there have been many studies that have shown that salmonids select spawning locations largely based on external physical attributes of the aquatic environment at various spatial scales. It is thought that local hydraulic conditions of water depth and velocity may have the largest impact on spawning site selection, but there are many possible controlling variables at the same and larger spatial scales that may affect where adult salmonids choose to spawn. Local hydraulics (Pasternack 2008), substrate (Kondolf and Wolman 1993), and in-gravel dissolved oxygen (Merz and Setka 2004) have been found to be important factors at the point scale. At the scale of 1-10 times the channel width, there are factors such as patterns of in-channel fluvial landforms (Geist and Dauble 1998; Moir et al. 2004; Moir and Pasternack 2008), aquatic vegetation and streamwood (Elkins et al. 2007; Senter and Pasternack 2010), and other physical heterogeneities (Lisle 1983; Wheaton et al. 2004b). At the reach and segment scales of about 100-10,000 channel widths, factors include bed slope and channel type (Buffington et al. 2004; Elkins et al. 2007; Pess et al. 2008; Soulsby et al. 2012), longitudinal water temperature trend (Torgersen et al. 1999), hydraulic geometry, deep scour risk (DeVries 1997), and flow regime, including flow fluctuations (Moir et al. 2006). For all of the above variables, there is also the possibility that spatial pattern (e.g., size, shape, and connectivity) influences spawner preference, but this has not yet been investigated.

1.1.2 *O. mykiss* Spawning Habitat Overview

While there has been much interest and extensive study regarding the variables affecting salmonid spawning habitat selection with a particular emphasis on Chinook salmon (*O. tshawytscha*), there has not been the same abundant, specific research regarding *O. mykiss* spawning habitat selection. Early studies that discussed *O. mykiss* spawning habitat were conducted because of their importance as a game fish, and presented the physical habitat conditions where spawning was observed (e.g., Burner 1951; Briggs 1953). More recent research
has been conducted regarding microhabitat factors that influence *O. mykiss* spawning, including hydraulic (water depth and velocity) conditions, substrate size, temperature, and cover.

### 1.1.2.1 Hydraulic Habitat

Local water depth and velocity are important factors for successful salmonid spawning. Water depth and velocity are interdependent, and areas of greater depth often have associated lower velocities relative to areas of shallower depth. Areas of excessive water depth and lower velocity may facilitate the deposition of fine sediment, making the substrate composition unsuitable for redd building and embryo incubation. Water velocities that are too low may not provide the flushing power to assist the female *O. mykiss* in the movement of gravel during redd building, or provide an adequate flow of well-oxygenated water through the gravel that is needed for embryo survival and development. Excessive water velocity, however, could impede spawning by causing fatigue in the females building redds, and has the possibility of causing scour of redds during embryo development. The movement of spawning gravels during high flows can destroy incubating embryos (Kondolf et al. 1991). This may impact *O. mykiss* spawning sites more frequently than other salmonids, as *O. mykiss* peak spawning occurs from December through April during high winter flows (McEwan 2001). The survival of salmon embryos can be greatly reduced by scour in active spawning habitats (Rhodes et al. 1994; Purser et al. 2009). While it was previously thought that *O. mykiss* and other salmonids tend to avoid actively eroding or recently eroded areas for spawning in order to prevent the loss of the incubating eggs, it does not appear that they actually avoid spawning in areas that are susceptible to scour (Bigelow 2003). A general range of suitable velocities for *O. mykiss* spawning as analyzed by previous studies is 0.30-1.34 m/s (0.98-4.40 ft/s) (Briggs 1953; Smith 1973; Swift 1976; Bovee 1978; USFWS 1996, 1997, 2007). Suitable water depths are reported to range from 0.10-1.0 m (0.33-3.3 ft) (Briggs 1953; Sams and Pearson 1963; Smith 1973; Swift 1976; USFWS 1996, 1997, 2007).

### 1.1.2.2 Stream Substrate

*O. mykiss* and other salmonids have specific requirements for spawning regarding the size of substrate needed for successful spawning and embryo incubation. The sediment size composition of the substrate may be the most important factor in the selection of spawning habitat, and the availability of suitably sized gravels can limit spawning success and the number of fish able to spawn in a stream (Kondolf and Wolman 1993; Buffington et al. 2004). The substrate size suitable for *O. mykiss* spawning depends on the particle size that an individual female has the power to transport by thrashing her tail to create a depression in the channel bed into which the eggs are laid (Burner 1951; Kondolf and Wolman 1993). If the substrate is too large, females will not be capable of moving it to produce the depression. Suitable substrate sizes for *O. mykiss* spawning compiled from several past studies range from 10.4 mm to 152.4 mm (0.41-6.0 inches), which spans gravel and cobble size ranges (Burner 1951; Briggs 1953; Chambers et al. 1954; Chambers et al. 1955; Orcutt et al. 1968; Cederholm and Salo 1979; Shirazi and Seim 1981; Kondolf and Wolman 1993; USFWS 1996, 1997). Limited availability of suitably sized spawning substrate can lead to superimposition of redds, where a female salmonid digs out a new egg pocket within the egg pocket of another existing redd, which can cause mortality of the developing embryos (McNeil 1964; Hayes 1987; Campos and Massa 2009). In the lower Yuba River, the overall mean substrate diameter (*D*<sub>mean</sub>) within the bankfull channel is 97.4 mm (RMT 2013a). On the lower Yuba River salmonids tend to spawn in mean substrate sizes ranging from about 50-150 mm (Figure 1). Most of the substrate in the lower Yuba River is characterized by
average D_{mean} values within this size range, with the exception of the sand/silt areas near the confluence of the Feather River, and the boulder/bedrock regions in the upper sections of Timbuctoo Bend and most of Englebright Dam reaches. However, recent gravel augmentation projects conducted by the Corps have improved spawning substrate in the Englebright Dam Reach (Corps 2013).

Figure 1. Longitudinal distribution of the mean substrate diameter. The box represents the typical range of spawning substrate sizes observed on the lower Yuba River. Source: RMT 2013a.

The amount of fine particles (i.e., sand and silt) in the substrate also has an important impact on *O. mykiss* spawning and successful egg development. The percolation of water through the gravel is vital to supplying well-oxygenated water to the developing embryos, and the permeability of substrate is thought to be an important co-dependent factor in the selection of spawning sites (Burner 1951; Barnhart 1986; McEwan 2001). The deposition of fine sediment can bury and suffocate developing embryos that need adequate dissolved oxygen and water exchange to survive (Barnhart 1986; Chapman 1988). High levels of suspended sediment and deposition of fine sediment decreases the amount and quality of available spawning habitat (Bogan 1993; Bednarek 2001). Conversely, the truncated sediment supply downstream of dams can cause surface armoring of the downstream channel. Armoring occurs when smaller sizes of substrate preferentially flush downstream during moderate flows, leaving behind a coarse surface layer on top of the well-mixed underlying bed sediment. The lack of sediment supply from upstream to replenish the channel bed downstream exacerbates this problem. Decreased sediment supply and armoring of the channel downstream of dams has been shown to reduce available spawning habitat for salmonids (Pess et al. 2008; Czuba et al. 2011).

### 1.1.2.3 Water Temperature

Water temperature is an important physical variable for spawning microhabitat, for both the selection of spawning habitat by the adult salmonid and for subsequent successful incubation and development of embryos. Suitable water temperatures for *O. mykiss* spawning reported in previous studies range from 4 to 18 °C (39.2-64.4 °F) (Bovee 1978; Raleigh et al. 1984;
McEwan 2001), and suitable temperatures for embryo incubation reportedly range from 0-12 °C (32-53.6 °F) (Calhoun 1966; Bovee 1978; Raleigh et al. 1984). The RMT (2013) developed lifestage-specific water temperature index values for salmonids in the lower Yuba River based on information described in Attachment A to RMT (2010), as well as updated information provided in Bratovich et al. (2012). The RMT (2013a; 2013b) utilized 57°F as the upper tolerance water temperature index value for steelhead spawning and embryo incubation. *O. mykiss* spawning occurs in the lower Yuba River primarily during the months of January through April upstream of Daguere Point Dam (RMT 2013a). Temperature was not included in the microhabitat suitability analysis in this study, as water temperatures in the lower Yuba River throughout the *O. mykiss* spawning period are consistently suitable upstream of Daguere Point Dam (RMT 2013a). The sub-thermocline reservoir releases from Englebright Lake and New Bullards Bar Reservoir provide cold water temperatures in the lower Yuba River during the winter and early spring months. Daily water temperature monitoring results from October 2006 through June 2013 at the Smartsville Gage and at Daguere Point Dam indicate that water temperatures remain below 57°F throughout the January through April steelhead spawning period (RMT 2013b; YCWA 2013). Mean daily temperature recorded at the Smartsville Gage below Englebright Dam ranged from 8.28-10.25 °C (46.9-50.5 °F) during the 2010 spawning season, and from 8.4-10.5 °C (47.1-50.9 °F) during the 2011 spawning season (Campos and Massa 2011; Campos and Massa 2012).

### 1.1.2.4 Cover

Cover is provided by many elements, such as overhanging vegetation, submerged vegetation, instream structures such as large woody debris and boulders, deep pools, and surface turbulence, and is a vital component of habitat for *O. mykiss*, particularly the fry and juvenile lifestages (Raleigh et al. 1984; Pasternack 2010). Specific types of cover also may be important for spawning site selection by *O. mykiss*, such as large woody debris, boulders, and other structures that provide refuge from predation and resting zones during spawning activity (Wheaton et al. 2004b; Senter and Pasternack 2010). Given the diverse types of structures that can provide valuable cover for *O. mykiss*, the complexity of the use of cover by spawning *O. mykiss*, and the difficulty of quantifying suitability of the various types of cover for spawning, cover was not explicitly incorporated into the *O. mykiss* spawning habitat suitability modeling for this study. However, the preference of *O. mykiss* to spawn near areas of cover was analyzed and is described in later sections.

### 1.2 Study Goals and Objectives

The overall goal of this study was to better understand and quantify physical habitat preferences of adult spawning *O. mykiss* in the lower Yuba River of north central California. At the microhabitat scale, the objectives were to: (1) determine physical conditions preferred for *O. mykiss* spawning; and (2) develop a predictive 2D physical habitat model capable of quantifying spawning habitat quality and availability in the lower Yuba River.

Habitat preferences also were assessed at larger spatial scales (e.g., morphological unit, reach, and segment (i.e., the entire lower Yuba River) scales) by analyzing spatial patterns of physical-biological linkages associated with spawning, including preferences for larger scale habitat variables and the processes that create and maintain these conditions. The evaluation of *O.
mykiss spawning physical habitat preferences at each of the spatial scales provided a comprehensive understanding of O. mykiss spawning habitat in the lower Yuba River.

Near-census river science is a comprehensive, spatially explicit, process-based paradigm for studying rivers emphasizing the 1-m scale as the basic building block for characterizing geomorphic processes and ecological functions (Pasternack 2011). Near-census data from the lower Yuba River used in this study included a high-resolution topographic map, sub-meter resolution areal color imagery, substrate grain size maps, 1-m resolution 2D hydrodynamic model outputs for a wide range of discharges, and O. mykiss redd surveys conducted throughout the lower Yuba River from the onset of spawning in January through April for two years (2010 and 2011). This report presents the complete near-census data and analyses that were conducted to examine predictive relationships between flows, landforms, and O. mykiss adult spawning physical habitat. The analysis of O. mykiss spawning microhabitat was conducted using pre-existing methods of physical habitat suitability modeling (Leclerc et al., 1995; and others, Elkins, Pasternack et al. 2013) applied to the near-census 2D hydrodynamic model outputs at ~1 m resolution. This spatial detail of point-scale (microhabitat) suitability conditions over the full alluvial extent of the lower Yuba River facilitated the evaluation of statistical distributions and spatial patterns in microhabitat availability.

The evaluation of O. mykiss spawning habitat preferences at larger scales involved 2D spatial analysis of various possible linkages between physical and biological variables at multiple spatial scales. The physical spatial structure of a river is made up of: (1) the landforms at various spatial scales that are discharge-independent, including morphological units and reach characteristics; (2) patches of homogenous physical habitat conditions at a scale of ~0.1-10 channel widths (“mesohabitat”) that change as a function of discharge; and (3) the physical processes that form and maintain these conditions. This study presents the methods and results of analyzing physical spatial structure in the lower Yuba River and physical habitat preferences of spawning O. mykiss at various spatial scales in the lower Yuba River.

With regard to physical microhabitat modeling, the spatial distribution of available habitat, and the influence of larger scale physical parameters on spawning site selection, the specific scientific questions investigated in this study include the following:

- Which of the existing O. mykiss spawning depth and velocity HSCs provides the best predictions of microhabitat suitability, including predictions of preferred habitat and avoided habitat?

- Can the physical microhabitat model be improved by incorporating substrate suitability rather than being based only on hydraulics?

- How does the availability of O. mykiss spawning habitat change with discharge?

- Is there preference for O. mykiss spawning according to physical parameters at larger spatial scales, such as:
  - Substrate size
  - Geomorphic reach
- Morphological unit type
- Type of topographic change process

- Are there spatial patterns in the utilization of spawning habitat by *O. mykiss* in the lower Yuba River, such as:
  - Longitudinal position
  - Size of morphological unit
  - Distance from other reds
  - Distance from the wetted edge

- What can the physical microhabitat model tell us about the spatial distribution and patterns of habitat availability on the lower Yuba River, such as:
  - Size of preferred and highest-quality habitat patch
  - Distance of redd from areas of preferred microhabitat

### 2 BACKGROUND

#### 2.1 Yuba River Watershed

The Yuba River Watershed drains 3,480 km² (~1,350 mi²) of the western slope of the Sierra Nevada and includes portions of Sierra, Placer, Yuba, and Nevada counties. The Yuba River is a tributary of the Feather River, which drains into the Sacramento River. The montane-Mediterranean climate is characterized by cool, wet winters and hot, dry summers (Storer et al. 2004). Almost all precipitation occurs from October through April, and snow pack accumulates through the winter at high elevations. Heavy flooding can occur in the winter when weather systems driven by the Pacific Ocean El Niño Southern Oscillation produce warm rain-on-snow events. Spring runoff is dominated by snowmelt during April-June as air temperatures warm. Dry conditions prevail May through September with occasional convective thunderstorms at high elevations. Annual precipitation ranges from more than 1,500 mm (59.1 in) in the Sierra Nevada to about 500 mm (19.7 in) at Marysville (Curtis et al. 2005).

#### 2.2 Lower Yuba River

##### 2.2.1 Lower Yuba River Physical Geography

The lower Yuba River is defined as the 37.1 km (23.1 mi) river segment beginning at Englebright Dam and ending at the confluence with the Feather River (Figure 2). The lower Yuba River is a regulated wandering gravel/cobble bed (mean diameter \(D_{\text{mean}}\) ~ 100 mm) river that is meandering to straight in pattern (Wyrick and Pasternack 2012). It is characterized by thick mixed coarse sediment alluvial fill and significant variations in channel and valley width. The lower Yuba River experiences a relatively dynamic flood regime with frequent overbank floods and high probability of large floods.
There is a long history of human disturbance in the Yuba River watershed, from which the form and function of the lower Yuba River has been invariably altered. These anthropogenic impacts include hydraulic gold mining, logging, road building, dredger gold mining, gravel mining, dam construction, channelization, partial flow regulation, and flow diversion. The gold rush era of the mid-nineteenth century brought hydraulic gold mining to the region, which transformed the lower Yuba River with the deposition of millions of tons of mining sediment (Gilbert 1917; Curtis et al. 2005; James et al. 2009; Pasternack 2010). Daguerre Point Dam, an 8 m (26 ft) run-of-the-river dam, was built in 1906 to prevent mining sediment from moving further downstream. Levees and training berms flanking the active river corridor were specifically designed in a funnel shape to narrow toward the confluence with the Feather River thereby promoting backwater floodplain hydraulics that trap sediment in the lower Yuba River as much as possible. In 1941, Englebright Dam, an 80 m (262 ft) concrete arch dam with a water storage capacity of 82.6 million m$^3$, was built as an additional barrier for mining sediment, and was later retro-fitted for hydroelectric power generation (Snyder et al. 2004). The construction of Englebright Dam helped to protect the downstream ~100 miles of previously navigable waterways through the lower Yuba, Feather, and Sacramento rivers and the Bay-Delta from further degradation and enable some recovery of the lower Yuba River active corridor. Englebright Dam is an impassable barrier for fish, and marks the upstream extent of habitat available to migrating species.
The present day lower Yuba River is in a unique condition compared to the other tributaries of the Sacramento and San Joaquin Rivers. Rivers such as the Feather, American, and Mokelumne are highly degraded by systemic channelization and channel narrowing, vegetation encroachment, levees, bed armoring, and loss of gravel/cobble sediment supply (Edwards 2001). They also have reservoirs and other hydroelectric generation facilities that dramatically alter the natural flow regime. In sharp contrast, the lower Yuba River has a tremendous surplus of gravel/cobble sediment in the alluvial lowland and a highly active flood regime due to limited storage capacity in the South and Middle Yuba tributaries (Pasternack 2008; Wyrick and Pasternack 2012). This combination, together with the presence of Englebright Dam, has yielded a long-term trend of channel and floodplain incision to recover the original river elevation without exhausting the availability of the gravel/cobble mixture necessary for spawning adult salmonids. Overall, despite a long, complex history of anthropogenic impacts to the watershed and channel, the lower Yuba River is highly dynamic and exhibits a strong interplay between the hydrological regime, valley topography, vegetation, and fluvial landforms (Wyrick and Pasternack 2012).

2.2.2 Lower Yuba River Fish Populations

The lower Yuba River sustains at least 28 fish species downstream of Englebright Dam, of which eight are anadromous species (CDFG 1991; Massa 2004; Massa and McKibbin 2005; Campos and Massa 2010). Key species utilizing the lower Yuba River include Chinook salmon and steelhead. The lower Yuba River is utilized by two principal phenotypic Chinook salmon runs (i.e., spring-run and fall-run). The Central Valley spring-run Chinook salmon ESU (Evolutionarily Significant Unit) was listed as threatened under the federal Endangered Species Act (ESA) during 1999, and NMFS reaffirmed the threatened status of this ESU during 2005. The Central Valley fall-/late fall-run Chinook salmon ESU (a combination of the fall- and late fall-runs as characterized by NMFS) was included on the Species of Concern List under the ESA in 2004 due to concerns about population size and hatchery influence (NMFS 2009). Late fall-run Chinook salmon populations occur primarily in the Sacramento River (CDFG Website 2007 as cited in RMT 2010), although incidental observations of phenotypic late fall-run Chinook salmon have been reported to occur in the lower Yuba River (D. Massa, CDFG, pers. comm. 2009; M. Tucker, NMFS, pers. comm. 2009), which may primarily originate from Coleman National Fish Hatchery on Battle Creek (RMT 2013a). The California Central Valley steelhead Distinct Population Segment (DPS) was listed as threatened under the ESA in 1998. For detailed information on steelhead and spring-run and fall-run Chinook salmon in the lower Yuba River, refer to RMT (2013a).

The Yuba River is thought to have historically supported a large *O. mykiss* population. Although the *O. mykiss* in the lower Yuba River have been widely stated to be one of the largest remaining wild populations in the Central Valley, the RMT (2013a) found that a substantial amount of straying hatchery-origin steelhead also utilize the lower Yuba River. Genetic analyses conducted by NMFS indicate that below-barrier populations of steelhead in the Central Valley, including the lower Yuba River (particularly in consideration of historic plantings and documented straying) likely do not accurately represent the ancestral population genetic structure (RMT 2013a). RMT (2013a) concluded that the current *O. mykiss* population in the lower Yuba River likely does not represent a pure ancestral genome.
The life history characteristics of *O. mykiss* are similar to other Pacific salmonids, with some complex differences regarding their migratory and spawning activity. The plasticity of *O. mykiss* life history strategies within a single population differs from other salmonids, which are primarily anadromous. Another characteristic of *O. mykiss* that differs from other Pacific salmonids is that they are iteroparous, meaning individuals may spawn multiple times in their lifetime, compared to species that are semelparous and die after spawning, such as Chinook salmon. The rates of repeat spawning, however, are generally low and vary greatly among populations (McEwan 2001).

The *O. mykiss* population in the lower Yuba River is composed of both anadromous steelhead and resident rainbow trout, while the progeny of both may exhibit either life history (Zimmerman et al. 2009; Mitchell 2010; RMT 2013a). The physical appearance of *O. mykiss* adults and the presence of seasonal runs and year-round residents indicate that both sea-run (steelhead) and resident rainbow trout exist in the Yuba River downstream of Englebright Dam. *O. mykiss* in the lower Yuba River may be exhibiting a predominately residential life history pattern. Previously, it has been surmised that perhaps the inclination towards *O. mykiss* residency rather than anadromy was associated with cold water temperature conditions provided in the river (RMT 2013a). However, as stated in RMT (2013a), research in the Central Valley does not necessarily support that presumption. For example, Satterthwaite et al. (2010) do not predict that a cool summer with high flow will strongly favor residency - rather, they conclude that the single most important factor in preserving the anadromous life history of *O. mykiss* is survival during the period between emigration to the ocean and returning to spawn. Additionally, NMFS (Lindley et al. 2007) suggest that *O. mykiss* may take up residency in dam tailwaters experiencing hypolimnetic releases due to their phenotypic plasticity, or the fitness of *O. mykiss* using these areas may exceed the fitness of anadromous fish, which would drive an evolutionary (i.e., genetic) change if life history strategy is heritable.

### 2.2.3 Fish Management Actions

As a rare dynamic gravel/cobble-bed river with suitable instream flow and water temperature conditions (RMT 2013a) and no hatchery located directly on the lower Yuba River, the lower Yuba River has become the focus of river management practices for aiding the recovery of regional anadromous fish populations. Previous and ongoing fisheries habitat improvement efforts include implementation of the Yuba Accord flow schedules, which were designed to provide suitable flow and water temperature conditions for anadromous salmonids, construction of the Narrows 2 flow bypass tunnel to provide continuous flow conditions during emergency outages, and physical habitat improvements, such as implementation of the Corp’s Gravel Augmentation Implementation Plan to inject gravel/cobble in the Englebright Dam Reach, and riparian habitat restoration efforts at Hammon Bar in the Parks Bar Reach (YCWA 2013).

The NMFS (2009) Draft Recovery Plan states that “The lower Yuba River, below Englebright Dam, is characterized as having a high potential to support a viable population of steelhead, primarily because: (1) the river supports a persistent population of steelhead and historically supported the largest, naturally reproducing population of steelhead in the Central Valley to support all life stage requirements; (3) the river does not have a hatchery on it; (4) spawning habitat availability does not appear to be limited; and (5) high habitat restoration potential.”
NMFS (2009) further states that “For currently occupied habitats below Englebright Dam, it is unlikely that habitats can be restored to pre-dam conditions, but many of the processes and conditions that are necessary to support a population of steelhead can be improved with improvements to instream flow regimes, water temperatures, and habitat availability. Continued implementation of the Yuba Accord is expected to address these factors and considerably improve conditions in the lower Yuba River.”

2.3 Spatial Analysis Units

Both statistical and spatial analyses rely on identification of experimental units that are then sampled and compared. The smallest point scale in this study from which all spatial analysis units are built up from is the near-census microhabitat scale involving 1 m x 1 m raster cells. Any variable or set of variables may be used to identify and delineate units composed of multiple raster cells. Different sets of units may also be defined at different spatial scales. Therefore, by using several different ways of breaking up the river into different sets of analysis units it is possible to address the various spawning habitat questions identified in the goals of this study. The spatial analysis units used in this study include hierarchical landform scales, topographic change processes, and mesohabitat patches. The aggregation of smaller-scale microhabitat suitability up to these larger spatial scales allowed for the evaluation of spatial patterns in physical habitat suitability and the determination of spawning preference based on larger scale river characteristics, providing a more comprehensive assessment of *O. mykiss* spawning habitat quality and availability in the lower Yuba River.

2.3.1 Hierarchical Landform Scales

The lower Yuba River is delineated at several scales into scientifically meaningful sections, and analysis at these various scales allows for a more comprehensive understanding of river health and habitat suitability. This study relies on river segregation using geomorphic principles.

2.3.1.1 River Segment

The segment scale refers to the entire length of the lower Yuba River from Englebright Dam to the confluence with the Feather River. The area considered for spawning habitat analysis consists of only the wetted channel area as determined at specific discharges of interest.

2.3.1.2 Geomorphic Reaches

The reach scale refers to eight longitudinally distributed, geomorphically distinct reaches in the lower Yuba River (*Figures 2 and 3*). Reaches were delineated based on the longitudinal profile and associated geomorphic variables, as summarized in *Table 1* (Wyrick and Pasternack 2012).
Figure 3. Longitudinal thalweg profile showing reach breaks described in Table 1.
2.3.1.3 **Morphological Units (MUs)**

The lower Yuba River was delineated at the ~1-10 channel width (W) scale by identifying specific classes of contiguous landforms, known as morphological units (MU) (Wyrick and Pasternack 2012). MUs are topographic features in the river that are defined independently of flow. This is in contrast to the concept of mesohabitat, which is habitat delineated at the same 1-10 W scale but is flow dependent. While Wyrick and Pasternack (2012) delineated 31 total MU types within the lower Yuba River, only 12 are relevant to this study.

Those 12 MU types can be categorized based on their general location with respect to the channel (Table 2). The in-channel bed units are those that are continuously wetted down to the baseflow discharge, which in the lower Yuba River is 24.9 m³/s (880 cfs) above Daguerre Point Dam and 15 m³/s (530 cfs) below Daguerre Point Dam due to agricultural diversions at the dam. The 12 in-channel MUs constitute the main lower Yuba River landforms available for *O. mykiss* adult spawning within the range of normal spawning flows. The in-channel bed units include chute, fast glide, pool, riffle, riffle transition, run, slackwater, and slow glide. The in-channel bar units are those that are primarily located within the zone between the baseflow and bankfull wetted area boundaries. The bankfull discharge in the lower Yuba River is estimated to be 141.6 m³/s (5,000 cfs) (Wyrick and Pasternack 2012). The in-channel bar units include lateral bar, medial bar, point bar, and swale. One other relevant MU where steelhead redds were surveyed was the hillside/bedrock MU, which is not physiographically bounded by any particular wetted

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Table 1. Reaches of the lower Yuba River with geomorphic delineations.

<table>
<thead>
<tr>
<th>Reach Name</th>
<th>Min</th>
<th>Mean</th>
<th>Max</th>
<th>Bed Slope (%)</th>
<th>Thalweg Length (ft)</th>
<th>Starting Point Description</th>
<th>Reach Abbreviation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Englebright Dam</td>
<td>316</td>
<td>415</td>
<td>693</td>
<td>0.31</td>
<td>1,259</td>
<td>Englebright Dam</td>
<td>EDR</td>
</tr>
<tr>
<td>Narrows</td>
<td>162</td>
<td>304</td>
<td>596</td>
<td>n/a</td>
<td>2,042</td>
<td>Confluence with Deer Creek</td>
<td>NR</td>
</tr>
<tr>
<td>Timbuctoo Bend</td>
<td>373</td>
<td>589</td>
<td>1866</td>
<td>0.201</td>
<td>6,337</td>
<td>Onset of emergent gravel floodplain upstream of Blue</td>
<td>TBR</td>
</tr>
<tr>
<td>Parks Bar</td>
<td>387</td>
<td>1007</td>
<td>1432</td>
<td>0.188</td>
<td>7,919</td>
<td>Highway 20 Bridge</td>
<td>PB</td>
</tr>
<tr>
<td>Dry Creek</td>
<td>783</td>
<td>987</td>
<td>1552</td>
<td>0.135</td>
<td>3,801</td>
<td>Confluence with Dry Creek</td>
<td>DC</td>
</tr>
<tr>
<td>Daguerre Point Dam</td>
<td>755</td>
<td>1628</td>
<td>2305</td>
<td>0.176</td>
<td>5,639</td>
<td>Daguerre Point Dam</td>
<td>DPD</td>
</tr>
<tr>
<td>Hallwood</td>
<td>573</td>
<td>1175</td>
<td>2394</td>
<td>0.131</td>
<td>8,382</td>
<td>Slope break near Eddie Drive</td>
<td>HW</td>
</tr>
<tr>
<td>Marysville</td>
<td>325</td>
<td>744</td>
<td>1842</td>
<td>0.052</td>
<td>5,334</td>
<td>No evident feature</td>
<td>MV</td>
</tr>
</tbody>
</table>

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*O. mykiss* Adult Spawning

**Physical Habitat in the Lower Yuba River**
area and can exist within the bankfull channel. However, it was not delineated below the baseflow inundation zone.
Table 2. Qualitative descriptions of in-channel morphological units mapped in the lower Yuba River.

<table>
<thead>
<tr>
<th>Type</th>
<th>Unit Name</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chute</td>
<td>Area of high velocity, steep water surface slope, and moderate to high depth located in the channel thalweg. Chutes are often located in a convergent constriction downstream of a riffle as it transitions into a run, forced pool, pool, or glide.</td>
<td></td>
</tr>
<tr>
<td>Fast Glide</td>
<td>Area of moderate velocity and depth and low water surface slope. Commonly occur along periphery of channel and flanking pools. Also exist in straight sections of low bed slope.</td>
<td></td>
</tr>
<tr>
<td>Pool / Forced Pool</td>
<td>Pools are areas of high depth and low velocity, and low water surface slope. The distinction between ‘forced pool’ and ‘pool’ cannot be made automatically within GIS. A ‘forced pool’ is one that is typically along the periphery of the channel and is “over-deepened” from local convective acceleration and scour during floods that is associated with static structures such as wood, boulders, and mostly bedrock outcrops. A ‘pool’ is not formed by a forcing obstruction.</td>
<td></td>
</tr>
<tr>
<td>Run</td>
<td>Area with shallow depths, moderate to high velocities, rough water surface texture, and steep water surface slope. Riffles are associated with the crest and backslope of a transverse bar.</td>
<td></td>
</tr>
<tr>
<td>Slow Glide</td>
<td>Area of low velocity and low to moderate depths and low water surface slope. May be located near water’s edge as a morphological unit along the channel thalweg transitions laterally towards the stream margins.</td>
<td></td>
</tr>
<tr>
<td>Slackwater</td>
<td>Shallow, low-velocity regions of the stream that are typically located in adjacent embayments, side channels, or along channel margins. Velocities are near stagnant during baseflow conditions and rise slower than other bed units’ as stage increases.</td>
<td></td>
</tr>
<tr>
<td>Lateral Bar</td>
<td>Area located at the channel margins at an elevation band between the autumnal low-flow stage and bankfull stage. Lateral bars are orientated parallel to the flow. The feature slopes toward the channel thalweg with an associated increase in both flow depth and velocity when submerged. Sediment size tends to be smaller than in adjacent sections of the channel.</td>
<td></td>
</tr>
<tr>
<td>Medial Bar</td>
<td>Area that is separated from the channel banks at low-flow stages at an elevation band between low-flow and bankfull stages. Can be accreting or eroding.</td>
<td></td>
</tr>
<tr>
<td>Point Bar</td>
<td>Accreting area located on the inside of a meander bend at an elevation band between the low-flow stage and bankfull stage. Point bars are curved and begin where there is clear evidence of point-bar deposition. The feature slopes toward the channel thalweg with an associated increase in both flow depth and velocity when submerged. Sediment size tends to be smaller than in adjacent sections of the channel.</td>
<td></td>
</tr>
<tr>
<td>Swale</td>
<td>A weakly-defined geometric channel or adjacent bench on the floodplain that only conveys flow at stages above low-flow.</td>
<td></td>
</tr>
</tbody>
</table>
2.3.2 Topographic Change Processes (TCPs)
The lower Yuba River was also delineated into regions of specific topographic change processes (TCPs), which are the mechanisms that cause landform deformation. TCPs were delineated in ArcGIS using a quasi-objective decision tree based on the location of scour and fill with respect to the 1999 and 2006-2008 in/out channel regions, as determined from digital elevation models (DEMs) created at these two time periods. The delineations were created without any consideration of the 2D model-predicted hydraulics or magnitudes of scour and fill, but were simply the locations of presence/absence of scour or fill. Regions of “No detectable change” also were delineated as areas in which any topographic variance was too small to be detected within the limits of the DEM differencing method (i.e., the scour/fill volumes in these regions are therefore equal to zero) (Carley et al. 2012). In total, 19 process types were identified and delineated, including nine scour-specific processes, nine fill-specific processes and “no change”.

2.3.3 Cover Elements
There are many types of cover elements that could influence salmonid spawning habitat utilization by providing cover or refugia during spawning activity. Wheaton et al. (2004b) reported a disproportionately high utilization of cover elements for Chinook spawning at one site on the lower Mokelumne River. In a more comprehensive investigation, Senter and Pasternack (2010) reported that cover elements play an important role in Chinook salmon spawning habitat at river sites that only have marginal spawning habitat; at sites with suitable hydraulic and substrate conditions, cover exhibited a limited role, primarily for adult refugia during rest periods. The cover elements mapped in the lower Yuba River that may play a role in O. mykiss spawning behavior include boulders, rip rap, streamwood, wood jams, and individual wood pieces. These elements were mapped within the wetted area boundary for a flow of 4,000 cfs, and boulders (> 3 m in the longest dimension), rip rap, streamwood in the wetted channel (> 3 m in the longest dimension and > 1 m above the substrate), wood jams (≥ 3 m long), and wood pieces (≥ 3 m long and ≥ 10 cm in diameter) were mapped as point, line, or polygon features using a handheld Trimble GeoXT differential GPS unit following the protocol in Pasternack (2010).

2.3.4 Mesohabitat Patches
Mesohabitat patch units are contiguous units of multiple 1 m x 1 m raster cells with homogeneous microhabitat suitability based on the hydraulic and substrate conditions available in the area. Microhabitat is usually considered to be the point-scale physical condition, such that when points are aggregated into sizable patches, they may be termed mesohabitat. The mesohabitat patches were delineated based on the suitability of O. mykiss spawning habitat as determined from the best performing 2D physical microhabitat model.

2.4 Hydrology During Data Collection
The hydrology in the lower Yuba River above Daguerre Point Dam varied significantly during the two years of O. mykiss spawning seasons (i.e., January 1, 2010 through April 30, 2011) (Figure 4). January through April of 2010 was a relatively dry winter with comparatively low flows throughout the winter - the average discharge over the period was 39.0 m³/s (1,379 cfs). There were a few in-channel peak events during January and April. The lower Yuba River
experienced much higher flows during the 2011 *O. mykiss* spawning season compared to the same period in 2010, with an average discharge over the time period of 140.5 m$^3$/s (4,963 cfs). There were two notable, sustained overbank floods (~3-4.5 times bankfull discharge) during the 2011 spawning season on March 16 (peak 15-minute discharge=22,708 cfs) and April 21 (peak 15-minute discharge=14,458 cfs).

Figure 4. Smartsville gage hydrograph during the period of the *O. mykiss* spawning surveys. The dashed line denotes the bankfull discharge of 5,000 cfs.

3 METHODS

The following methods were employed to conduct a comprehensive analysis of *O. mykiss* spawning physical habitat in the lower Yuba River. Overall, a predictive 2D physical habitat model based on microhabitat suitability, local hydraulics, and substrate size was developed and tested, and then this predictive tool was used to evaluate spawning habitat preferences for a diversity of spatial analysis units in an effort to fully describe the factors affecting spawning site selection by *O. mykiss* and provide an assessment of *O. mykiss* spawning habitat quality and availability in the lower Yuba River. All data in the study were collected or generated in English units consistent with regulatory requirements and then converted to SI units for this article, hence the appearance of some unusual values in SI units (e.g. 0.9144-m represents a 3-ft raster cell size).

3.1 Biological Observation Data

A redd is the depression in riverbed substrate created by an adult *O. mykiss* into which eggs are laid. Redds are identifiable by the downstream mound of gravel called the tailspill that results
Redd surveys of the lower Yuba River were conducted during the winter spawning seasons of 2010 and 2011 (Campos and Massa 2009). Surveys covered the whole lower Yuba River, except for the Narrows reach, which contains Class III-IV rapids that make the reach too dangerous and difficult to survey. The geographic location and a series of attributes were recorded using a handheld Trimble GeoXT differential GPS unit at each redd, including the date of observation, visual assessment of the undisturbed substrate upstream of the redd, water depth and mean column water velocity measured 0.5 ft upstream of the redd, and habitat type (i.e., pool, riffle, run, or glide) (Campos and Massa 2009; Campos and Massa 2010b). The discharge at which each redd was surveyed was obtained from the flow records of the Yuba River Smartsville and Marysville gages through the California Department of Water Resources’ (CDWR) online CDEC. The physical dimensions of each redd were also measured in the field to determine the size of *O. mykiss* reds in the lower Yuba River, including the length and width of the redd pot, as well as the length and two measurements of the width of the tailspill (Figure 5). The redd pot is the area at the head from which substrate is excavated by the activity of the female *O. mykiss*, creating a depression in the river bed into which the eggs are deposited and fertilized. The tailspill is the area downstream of the pot where gravels are deposited after they have been mobilized by the gravel digging action of the female with her tail. An estimate of the average size of an *O. mykiss* redd in the lower Yuba River was calculated by taking the overall average length of a redd (average pot length plus average tailspill length), multiplied by the average pot width or tailspill width, whichever was larger.
3.1.1 Redd groups
Variable winter flows during the *O. mykiss* spawning season resulted in a wide range of discharges at which *O. mykiss* redds were observed (Figure 6). *O. mykiss* redd observations were made over an order of magnitude range of within-channel discharges, from 18.97 to 117.79 m³/s (670 to 4,160 cfs) (Campos and Massa 2011; Campos and Massa 2012). While the 2D model could be run at each individual observed discharge to determine the hydraulic conditions throughout the lower Yuba River, it is not possible to validate whether the hydraulic conditions utilized for spawning at that location and discharge are indicative of high-quality habitat or preferred conditions unless there is a substantial sample size of *O. mykiss* redds with which to compare.
Of the 261 *O. mykiss* redd observations, there were three reasonably narrow ranges of discharges that provided large enough sample sizes to enable bioverification and other analyses. For each group, 2D hydraulics were represented by a single modeled discharge (Table 3). There was a group of 43 redds observed at a constant flow of 24.24-24.27 m$^3$/s (856-857 cfs), a group of 54 redds observed at flows between 27.18-29.73 m$^3$/s (960-1,020 cfs), and a third group of 94 redds observed at flows between 35.68-37.09 m$^3$/s (1,260-1,310 cfs). Channel hydraulics and physical habitat conditions associated with each of these *O. mykiss* spawning groups were represented by a single modeled flow, which was selected from an available set of pre-existing simulations that included some whose discharges were close to the observed discharge values: 24.9 m$^3$/s (880 cfs), 28.3 m$^3$/s (1,000 cfs), and 36.8 m$^3$/s (1,300 cfs), respectively. These three groups and their associated discharge are referred to as $Q_{\text{low}}$, $Q_{\text{mid}}$, and $Q_{\text{high}}$. The thee redd groups account for 72\% of all redd observations, while the other 28\% of redds were observed at dissimilar discharges at which there was not a large enough sample size with which to conduct any meaningful microhabitat analysis and were therefore omitted.

Error analysis was conducted to verify that the simulated hydraulics for the selected discharges accurately represented the hydraulics at the actual redd-building flows. Depth and velocity values at each of the redd locations in each of the three groups were identified from the 2D models whose discharges were the closest to observed (one higher and one lower). Then the average of the local depth and velocity values at each redd at the higher and lower modeled discharges were used to linearly interpolate the average depth and velocity values for the group at the average discharge of the group. For example, the average of the depth and velocity values at the 43 redds from the 800 cfs and 880 cfs modeled results were used to interpolate the average depth and velocity at 856.1 cfs, which was the average observed discharge of the 43 redds in the $Q_{\text{low}}$ group (Figure 7). For the $Q_{\text{mid}}$ group, the average depth and velocity modeled values from the 54 redds at 1,000 cfs and 1,300 cfs were used to interpolate the average modeled depth and

![Figure 6. Distribution of redd observations with discharge](image)

Of the 261 *O. mykiss* redd observations, there were three reasonably narrow ranges of discharges that provided large enough sample sizes to enable bioverification and other analyses. For each group, 2D hydraulics were represented by a single modeled discharge (Table 3). There was a group of 43 redds observed at a constant flow of 24.24-24.27 m$^3$/s (856-857 cfs), a group of 54 redds observed at flows between 27.18-29.73 m$^3$/s (960-1,020 cfs), and a third group of 94 redds observed at flows between 35.68-37.09 m$^3$/s (1,260-1,310 cfs). Channel hydraulics and physical habitat conditions associated with each of these *O. mykiss* spawning groups were represented by a single modeled flow, which was selected from an available set of pre-existing simulations that included some whose discharges were close to the observed discharge values: 24.9 m$^3$/s (880 cfs), 28.3 m$^3$/s (1,000 cfs), and 36.8 m$^3$/s (1,300 cfs), respectively. These three groups and their associated discharge are referred to as $Q_{\text{low}}$, $Q_{\text{mid}}$, and $Q_{\text{high}}$. The thee redd groups account for 72\% of all redd observations, while the other 28\% of redds were observed at dissimilar discharges at which there was not a large enough sample size with which to conduct any meaningful microhabitat analysis and were therefore omitted.

Error analysis was conducted to verify that the simulated hydraulics for the selected discharges accurately represented the hydraulics at the actual redd-building flows. Depth and velocity values at each of the redd locations in each of the three groups were identified from the 2D models whose discharges were the closest to observed (one higher and one lower). Then the average of the local depth and velocity values at each redd at the higher and lower modeled discharges were used to linearly interpolate the average depth and velocity values for the group at the average discharge of the group. For example, the average of the depth and velocity values at the 43 redds from the 800 cfs and 880 cfs modeled results were used to interpolate the average depth and velocity at 856.1 cfs, which was the average observed discharge of the 43 redds in the $Q_{\text{low}}$ group (Figure 7). For the $Q_{\text{mid}}$ group, the average depth and velocity modeled values from the 54 redds at 1,000 cfs and 1,300 cfs were used to interpolate the average modeled depth and

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O. mykiss Adult Spawning

Physical Habitat in the Lower Yuba River

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velocity at the averaged observed discharge of 1,005.1 cfs. For the $Q_{\text{high}}$ group, the average depth and velocity values at the 94 redds from the modeled 1,000 cfs and 1,300 cfs flows were used to interpolate the average depth and velocity values at the average observed discharge of 1,272.3 cfs. Percent difference was then calculated as the difference between the interpolated average depth and velocity at the average spawning flow for each group and the average depth and velocity at the nearest modeled discharge that was used to represent the hydraulic conditions for each group. Percent differences of 0.35-1.48 % in average depth and 0.15-1.90 % in average velocity were found between the 2D model runs used and the modeled conditions at the average of the observed redd building flows (Table 3).

Figure 7. Example illustration of the error analysis concept and calculations

Overall, the low percent differences in average depth and velocity show that hydraulic conditions for each redd group are well represented by the single modeled discharge nearest the average observed discharge from the group. The model results from the single modeled discharge chosen to represent each of the three redd groups are sufficiently similar to the observed conditions and accurately represent the hydraulic conditions at each of the respective discharges. Along with the physical location of the redd observations in each redd group, the hydraulic model results were used to assess the predictive capability of each of the models of physical microhabitat suitability across the three discharges assigned to the redd groups.

Table 3. Observed and modeled flows for *O. mykiss* redd groups

<table>
<thead>
<tr>
<th>Redd group</th>
<th># reds</th>
<th>Discharge of 2D model run used</th>
<th>Range of observed discharges in the group [Average discharge]</th>
<th>% Diff in Discharge</th>
<th>% Diff in Depth</th>
<th>% Diff in Velocity</th>
</tr>
</thead>
<tbody>
<tr>
<td>$Q_{\text{low}}$</td>
<td>43</td>
<td>880 cfs</td>
<td>856-857 cfs [856.1 cfs]</td>
<td>2.71</td>
<td>1.48</td>
<td>1.90</td>
</tr>
<tr>
<td>$Q_{\text{mid}}$</td>
<td>54</td>
<td>1,000 cfs</td>
<td>960-1,050 cfs [1,005.1 cfs]</td>
<td>0.51</td>
<td>0.35</td>
<td>0.15</td>
</tr>
<tr>
<td>$Q_{\text{high}}$</td>
<td>94</td>
<td>1,300 cfs</td>
<td>1,260-1,310 cfs [1,272.3 cfs]</td>
<td>2.13</td>
<td>0.97</td>
<td>1.27</td>
</tr>
</tbody>
</table>

### 3.2 Lower Yuba River Digital Elevation Model

River corridor topography and bathymetry were collected for the creation of a high-resolution digital elevation model (DEM) using a combination of ground-based, boat-based, and remote sensing (i.e., Light Detection and Ranging, or LiDAR) methods in accordance with a pre-designed protocol (Pasternack 2009). A detailed presentation of the methods for topographic and
bathymetric mapping may be found in (Pasternack et al. 2013). Final base Earth DEM products include a triangulated irregular network (TIN) DEM using points that met the QA/QC standards in the project as well as a 3 ft resolution raster.

3.3 Lower Yuba River 2D Hydrodynamic Modeling and Validation

The Surface-water Modeling System v.10.0 (Aqua veo, LLC, Provo, UT) and Sedimentation and River Hydraulics (SRH-2D, v. 2.1) were used to produce 2D near-census hydrodynamic models of the lower Yuba River according to the procedures of Pasternack (2011). The models span the ~37 km of the lower Yuba River, from Englebright Dam to the confluence with the Feather River, except for a small section in the Narrows canyon. The models provide outputs of depth and velocity with 1.0-1.5 m resolution. A brief summary is provided in this section, while a detailed presentation of the methods for producing 2D hydrodynamic models for the lower Yuba River may be found in Barker (2011), Abu-Aly (2012), and Pasternack et al. (2013).

Given the length of the lower Yuba River, the segment was divided into five model domains (three upstream of Daguerre Point Dam and two downstream of it). Also, to reduce the dry area in each simulation to save computational time, different computational meshes were made for the lowest flows (0 - 36.8 m$^3$/s, 0 - 1,300 cfs), intermediate flows (36.8 - 212.4 m$^3$/s, 1,300 cfs - 7,500 cfs), and higher flows (> 212.4 m$^3$/s, 7,500 cfs). No simulations for flows greater than 212.4 m$^3$/s (7,500 cfs) are reported in this study. For the three upstream domains the internodal spacing of the computational mesh was ~0.91 m (3 ft) for the baseflows and intermediate flows, while for the two downstream ones the internodal spacing was ~1.5 m (5 ft) for those same flow ranges.

Input discharge for the uppermost three domains was obtained from the USGS gaging stations on the Yuba River at Smartsville below Englebright Dam (#11418000) plus the small influx from Deer Creek at Mooney Flat Road (#11418500). There is flow diverted for agriculture midway down the lower Yuba River at Daguerre Point Dam, so for the two domains below the dam the discharge was obtained from the USGS gaging station downstream of these diversions on the Yuba near Marysville, CA (#11421000). The corresponding water surface elevations at the exit of the model domains for these discharges was obtained by creating stage-discharge rating curves using automated water-level loggers and/or direct observation with an RTK GPS or total station (viewable in Pasternack et al. 2013). One domain ended at the Marysville USGS gage, so the USGS rating curve was used in that case.

Boundary roughness was primarily addressed during the mapping effort by creating a highly detailed DEM, but unresolved roughness was addressed by using a global Manning’s roughness value (n). The use of a constant value for unvegetated substrate was justified on the lower Yuba River, because the New Years 2006 flood with a peak flow of ~3,087 m$^3$/s (~19.5 times larger than bankfull discharge) was observed to erase much of the MU-scale spatial variability in the bed surface grain size distribution. This particular phenomenon occurs on the lower Yuba River because the underlying valley fill is composed of relatively homogeneous hydraulic mining sediment deposited primarily between 1850 and 1940 prior to the construction of Englebright Dam (Pasternack 2008). Past site-scale 2D model studies on the lower Yuba River used an n value of 0.043 for the unvegetated gravel/cobble riverbed (Moir and Pasternack 2008; Sawyer et al. 2010). In this segment-scale analysis, an evaluation of observed and modeled water surface
elevations found that an n value of 0.04 was best for the three downstream domains, an n value of 0.032 for the bedrock canyon below Englebright Dam, and an n value of 0.03 for the upstream-most alluvial domain, Timbuctoo Bend (Pasternack et al. 2013). Based on LiDAR mapping, the area of vegetation within the model domain for this study was < 4 % and likely consisted of overhanging canopy, so it was not explicitly accounted for in boundary roughness. Extensive analysis of roughness in site-scale 2D models on the lower Yuba River previously found no systematic change in Manning’s n as a function of discharge, including at two sites tested over a range of 0.1 to 22 times bankfull discharge (Sawyer et al. 2010). The reason is that as water rises, the water column makes contact with new types of roughness elements such as boulder clusters, bedrock outcrops, vegetation, and valley width variations. As a result, high roughness is maintained with increasing discharge despite drowning out of some features. Thus, the constant roughness values reported above were used for all discharges and the suitability of this choice was carefully tested in model validation procedures using independent data from multiple discharges.

Extensive model validation was performed for unvegetated model simulations for an order of magnitude of flow ranges (~14 to 170 m³/s) (Barker 2011). Mass conservation between specified input flow and computed output flows was within 1 %. As an example of water surface elevation performance relative to the river’s mean substrate size of ~100 mm (Wyrick and Pasternack 2012), the mean signed vertical deviation for 197 observations at 24.92 m³/s was -1.8 mm. For unsigned deviations, 27 % were within 3.1 cm vertical, 49 % of deviations within 7.62 cm, 70 % within 15.25 cm, and 94 % within 30.5 cm. From cross-sectional surveys yielding 199 observations, predicted versus observed depths yielded a coefficient of determination (r²) of 0.66. Using Lagrangian tracking of an RTK GPS on a floating kayak, surface velocity magnitude was measured at 5,780 locations, yielding a predicted versus observed r² of 0.787, which is significantly higher than commonly reported (Figure 8). Median unsigned velocity magnitude error was 16 %, which is less than commonly reported. Using Lagrangian tracking of an RTK GPS on a floating kayak, velocity direction was also tested at those 5,780 points, yielding a predicted versus observed r² of 0.796 (Figure 8). This parameter is not commonly tested, but likely should be for 2D models. Median direction error was 4 %, with 61 % of deviations within 5° and 86 % of deviations within 10°. Further details on the model validation are reported in Barker (2011). Overall, the lower Yuba River 2D model met or exceeded all common standards of 2D model performance.
Output from the 2D model was used to create detailed raster maps with values of water depth and mean column velocity at every point on a 0.914 m (3 ft) grid of the lower Yuba River. Individual depth and velocity maps were created at each modeled discharge value, ranging from very low flow to massive flood flows. The use of 2D hydraulic results allowed the analysis of the spatial patterns of habitat quality. The depth and velocity model outputs at 24.9 m$^3$/s (880 cfs), 28.3 m$^3$/s (1,000 cfs), and 36.8 m$^3$/s (1,300 cfs) were used in the physical model development and bioverification tests of the models when coupled with the data from the $Q_{\text{low}}$, $Q_{\text{mid}}$, and $Q_{\text{high}}$ redd groups, as described in later sections.

### 3.4 Substrate Mapping

A comprehensive mapping effort of the substrate of the lower Yuba River was conducted in 2010 because of the importance of substrate size as a factor in spawning site selection of *O. mykiss* and other salmonids. A visual classification system was used to conduct the substrate survey of the bankfull channel. A key feature of the RMT’s substrate classification is that it does not blindly apply the generic Wentworth scale (log base 2), but instead uses classes that make sense for the observational conditions expected in consideration of the typical gravel/cobble-bed particle size distributions and the associated anadromous salmonid substrate suitability requirements (Table 4) (Pasternack et al. 2013). The substrate classification system used for the lower Yuba River substrate survey designated the following substrate class sizes that represent biologically relevant categories for salmonid habitat utilization (Jackson et al. 2013; Pasternack et al. 2013):
Table 4. Substrate classification that links statistical properties of lower Yuba River bed material grain size distributions and physical habitat suitability.

<table>
<thead>
<tr>
<th>Class</th>
<th>Particle Size Range (mm)</th>
<th>Habitat suitability</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bedrock</td>
<td>No alluvium</td>
<td>Periphyton only</td>
</tr>
<tr>
<td>Boulder Field</td>
<td>D&gt;256</td>
<td>Chinook salmon and steelhead trout fry, parr, and smolt cover and foraging</td>
</tr>
<tr>
<td>Large Cobble</td>
<td>128&lt;D&lt;256</td>
<td>Chinook salmon and steelhead trout fry and parr cover and foraging</td>
</tr>
<tr>
<td>Cobble</td>
<td>90&lt;D&lt;128</td>
<td>Chinook salmon spawning, embryo incubation, and fry cover</td>
</tr>
<tr>
<td>Medium Gravel/Small Cobble</td>
<td>32&lt;D&lt;90</td>
<td>Chinook salmon and steelhead trout spawning, embryo incubation, and fry cover</td>
</tr>
<tr>
<td>Fine Gravel</td>
<td>2&lt;D&lt;32</td>
<td>Steelhead trout spawning and embryo incubation</td>
</tr>
<tr>
<td>Sand</td>
<td>0.0625&lt;D&lt;2</td>
<td>Submerged aquatic vegetation (SAV)</td>
</tr>
<tr>
<td>Silt/Clay</td>
<td>D&lt;0.0625</td>
<td>Submerged aquatic vegetation (SAV)</td>
</tr>
</tbody>
</table>

The field crew surveyed the river bed to identify areas of similar substrate composition and used the visual classification approach to determine what percentage of the area was composed of each class size (Pasternack 2010). The boundary of each fairly homogenous area was mapped with a handheld Trimble GeoXT differential GPS. The substrate mapping was conducted for the bankfull channel in summer and fall 2010, with deep water substrate mapping performed using a video camera system in fall 2011 (Pasternack 2010).

A raster of weighted mean substrate diameter for the lower Yuba River was calculated based on the substrate map of the river. The substrate map consisted of polygons that represented areas of similar substrate composition. Each polygon was characterized by the percentage of the area represented by each substrate class (i.e., percentage bedrock, percentage boulder, etc.). The median substrate size of each substrate visual class was used with the percentage of bed area in each visual class to calculate mean diameter at all points on the river (Pasternack et al. 2013). The overall weighted mean grain size for the lower Yuba River is 97.4 mm, which is in the cobble size class. The mean grain size raster shows a downstream decrease in particle size, as is typical for an alluvial river (Figure 9). Further information and some segment and reach scale analysis of substrate data are presented in Wyrick and Pasternack (2012) and Jackson et al. (2013). The raster of Dmean was used as the basis for determining substrate suitability of spawning habitat.
3.5 Habitat Suitability Curves and Physical Microhabitat Model Development

3.5.1 Habitat Suitability Curves
To characterize habitat quality and ultimately quantify available spawning habitat, it is necessary to predict the physical conditions that make habitat suitable for *O. mykiss* spawning. There are two major assumptions inherent to habitat suitability assessment and physical habitat modeling that are important to address. First, that organisms select habitat (Lack 1933) and have preferences for physical habitat conditions seemingly based on the ability of sites to provide the organism with the best chance to maximize reproductive success (Morrison et al. 1992; Stalnaker et al. 1995). Second, that fish select habitat they utilize in proportion to their preferences for specific physical habitat characteristics (Stalnaker 1979; Knapp and Preisler 1999), meaning conditions that are more or less preferred can be inferred to represent various levels of habitat quality. Fish disproportionately utilize the most suitable conditions available, but will utilize less suitable conditions, and their preference for use decreases with decreasing suitability of the physical habitat conditions (Stalnaker 1979).

Given these assumptions, it is possible to create a link between physical river conditions and ecological functions. Habitat suitability assessment was conducted in this study using HSCs for

### Figure 9. 2010-2011 Lower Yuba River segment-scale substrate map showing the weighted mean grain size in the river binned according to the RMT’s substrate classification scheme.
the physical variables that are important for spawning habitat. HSCs describe habitat suitability as a function of these physical habitat variables. Habitat suitability values range from 0 (non-habitat) to 1 (highly suitable habitat) across the range of values for a physical habitat variable, such as depth, velocity, or substrate size. Assigning a value of suitability across the range of values demonstrates the understanding that while a large range of values for physical variables may be used for spawning, certain values are utilized disproportionately more often and are more suitable for *O. mykiss* spawning. By combining suitability values from several physical habitat variables, the overall quality of spawning habitat can be quantified at a location, along with distribution of the varying levels of habitat quality available throughout the river.

Habitat suitability curves have been used extensively to simulate physical habitat availability for use in instream flow assessments, such as with the popular Physical Habitat Simulation (PHABSIM) software package (Bovee et al. 1998). HSCs are the foundation of physical habitat simulation and habitat availability calculations, making it important that the curves accurately represent the physical habitat requirements of spawning *O. mykiss*. There are several types of HSCs that can be developed and used to determine habitat quality in a stream and determine the amount of suitable habitat available (Bovee 1986). HSCs are often developed based on direct observation of habitat utilization, however the curves can also be developed based on a theoretical ideal range of habitat conditions or through a consensus approach of referencing and altering other suitability curves. Theoretical HSCs are based on information other than field observations, such as life history studies in the literature or from the professional judgment of biologists. While theoretical HSCs require very few resources to develop and are generally based on the professional opinion of biologists with years of experience, their accuracy has been questioned because they cannot be supported by quantifiable evidence and different biologists strongly disagree with each other on the specifics of each HSC, making the expert-based process unrepeatable and opaque. Some scientists assert that theoretical curves are as valid as observation-based curves, and some theoretical HSCs developed by fisheries biologists were found to be essentially identical to observation-based curves developed independently (Baldridge 1981; Bovee et al. 1998). However, the concern remains that theoretical HSCs may be highly subjective and should be avoided if there is the possibility to develop suitability curves that are more defensible.

Observation-based HSCs are developed using field observations of the conditions used by an organism while preforming an ecological function, and they are generally assumed to be more accurate than theoretical HSCs. The values of suitability throughout a range of values of a given physical habitat variable are based on frequency analysis of the microhabitat conditions utilized in the stream. While the use of actual observational data makes these curves more defensible than theoretical curves, there also may be a bias inherent in observation-based HSCs regarding the availability of microhabitat. The field observations represent the microhabitat conditions that were seen to be utilized in the field and do not necessarily represent a species’ preferred physical habitat if there is low availability of those preferred conditions (Bovee 1986). There have been some efforts to compensate for this potential bias by correcting suitability values by the relative availability of the conditions in the stream (Bovee et al. 1998). These “preference” HSCs should, in theory, be more accurate than observation-based curves because they take into account not only utilization of specific physical habitat conditions, but also the availability of those conditions in the stream. However, the alteration of observation-based HSCs based on habitat
availability can drastically alter the curves developed based on actual observations and may introduce additional error and bias to the HSCs (Payne and Allen 2009), so preference HSCs are not recommended for use in PHABSIM applications (Bovee 1996). The habitat availability data is still valuable to collect in the field, because even if it is not built into the HSCs, the data can be used after the models of habitat availability are made to corroborate the estimates of habitat quality and demonstrate the selectivity of the various levels of habitat quality by the organisms.

The possible dependence of observation-based curves on the availability of the various habitat conditions may make these HSCs site-specific and generally not transferable for use in other streams. Even so, many studies have used existing HSCs developed for other streams in instream flow assessments. For *O. mykiss* spawning, one of the most cited sets of suitability curves is from Bovee (1978), which developed curves for depth, velocity, temperature, and substrate size from data compiled from various sources. The HSCs in Bovee (1978) have been utilized widely for *O. mykiss* physical habitat suitability and instream flow studies because they are provided with the PHABSIM modeling package, but the use of these standard curves has been found to introduce significant bias to the instream flow results for streams of different sizes and may not adequately represent microhabitat selection in all streams (e.g. Annear and Conder 1984; Vondracek and Longanecker 1993). PHABSIM users are encouraged to develop site-specific suitability curves, but often this is impractical due to the amount of time, money, and resources required in developing HSCs. A transferability test was developed to test the applicability of HSCs to different streams for which they were not developed (Thomas and Bovee 1993), but there has been debate as to the adequacy of this test (Freeman et al. 1999; Williams et al. 1999). While there is inherent uncertainty in developing biologically meaningful HSCs to estimate available physical habitat, it is generally understood that the most accurate and defensible approach is to use site-specific HSCs developed from observations in the study area.

Another complicating factor is the potential dependence of microhabitat selection on discharge, particularly when HSCs are developed from data collected at a single discharge (Vondracek and Longanecker 1993). It is possible that depth, velocity, and substrate size preferences vary depending on the discharge experienced by the individual fish and the factors may be interdependent on one another (Beecher et al. 1995; Jager and Pert 1997; Williams 1997), but the changes in utilized conditions may also be a result of changing availability of those conditions as discharge increases (Beecher et al. 1997). In general, there seems to be a need for further validation of HSCs across a range of discharge values in order to determine whether the suitability of various microhabitat conditions changes with discharge. There should be additional verification to ensure that the HSCs accurately represent the physical habitat conditions selected at various flows.

There are many issues inherent in HSCs and habitat suitability modeling that are still being debated. Most studies pick a set of curves and use them to determine instream flow requirements, without any indication of the accuracy of the curves they chose. Even if HSCs are developed for a specific stream, often times investigations have not been conducted whether the conditions considered suitable are actually conditions preferred for spawning, or if they represent only the conditions that were available in the specific stream. The issues regarding the impact of habitat availability on the development of HSCs, transferability of curves between streams, as well as...
the question of the independence of microhabitat selection and discharge are still concerns in using HSCs to model habitat quality and availability.

This study tests several sets of HSCs, both observation-based water depth and velocity HSCs developed from data collected in the lower Yuba River, a set of theoretical-based depth and velocity HSCs developed during a complex negotiation process in a dam relicensing on another river in the Central Valley, as well as many theoretical-based HSCs for the suitability of substrate size specifically for the lower Yuba River. The comparison of results from these various types of HSCs provides a test of which curves are most applicable in the lower Yuba River, and addresses the question of applicability of theoretical HSCs and transferability of HSCs for modeling physical habitat in a river for which they were not developed.

The question of habitat availability bias and preferred habitat conditions is also addressed in this study. High resolution, near-census data provides the ability to very accurately quantify habitat availability, making it possible to quantify the selectivity, or preference and avoidance, of certain habitat conditions relative to the availability of those specific conditions in the lower Yuba River. The ability to accurately quantify habitat availability by moving away from a sampling approach to near-census data availability, and the testing of the selectivity for certain habitat conditions, helps to remove the potential bias of low availability of preferred microhabitat conditions.

Another innovative aspect of this study is the modeling of physical habitat with HSCs across a range of in-channel discharges. The same sets of HSCs were tested at three different discharges, which assists in answering the question of discharge-dependent selection of microhabitat conditions.

3.5.2 Hydraulic HSCs

HSCs that define the suitability of habitat for spawning based on local hydraulic conditions of water depth and velocity are referred to as hydraulic HSCs. Three sets of hydraulic HSCs, consisting of separate water depth and velocity HSCs, were tested for their predictive ability - The U.S. Fish & Wildlife Service curves (USFWS), the Tuolumne River Technical Advisory Committee curves (TRTAC), and the Yuba River Management Team curves (RMT) (Figures 10 and 11). These three sets of hydraulic HSCs were used because they were developed either based on data from the lower Yuba River specifically (USFWS and RMT), or from a negotiation process amongst fisheries experts and stakeholders for another river in the Central Valley with a similarly sized drainage basin, the Tuolumne River (TRTAC). The Bovee (1978) hydraulic HSCs were not tested in this study, even though they are widely applied in many instream flow studies, due to the availability of hydraulic HSCs developed specifically for the lower Yuba River and another comparable basin that are more applicable than the Bovee (1978) HSCs for O. mykiss spawning in the lower Yuba River. The Bovee (1978) HSCs generally define much smaller ranges of suitable depths and velocities than the three sets of HSCs tested in this study, and are included as dashed gray lines in Figures 10 and 11 for comparison.

The USFWS curves were developed from O. mykiss spawning observations in the lower Yuba River using the methods detailed in Gard (2010). The USFWS velocity HSC shows increasing habitat suitability with increasing mean column velocity, with peak suitability at ~ 0.80-0.90 m/s
(2.6-2.9 ft/s) and suitability decreasing gradually to a value of zero at 2.11 m/s (6.93 ft/s). The highest habitat suitability occurs at a higher velocity than the TRTAC and RMT curves. The USFWS velocity HSC also assigns at least a moderate level of habitat suitability to a larger range of velocity values than the other two HSCs as well. Similarly, the USFWS depth HSC assigns peak habitat suitability at a greater value of depth than the other two HSCs, with the highest suitability assigned to water depths of 2.14-6.07 m (7-19.90 ft), then decreasing to a suitability of zero at 30.48 m (100 ft) of depth.

The TRTAC curves were developed through a relicensing stakeholder negotiation process for the Tuolumne River (Stillwater Sciences 2013). The TRTAC velocity HSC has the largest range of highest suitability velocity values of the three HSCs, with a suitability value of 1 assigned to velocities of 0.34-0.79 m/s (1.1-2.6 ft/s). The TRTAC velocity HSC reaches a suitability of zero at 1.34 m/s (4.4 ft/s), which is significantly lower than the maximum suitable velocity from the USFWS curve and more comparable to the maximum suitable velocity from the RMT curve. The TRTAC depth HSC assigns the largest range of depths as highly suitable of the three curves, with maximum suitability assigned to all water depths greater than 0.305 m.

The Yuba River Management Team (RMT) curves were developed from *O. mykiss* spawning observations in the lower Yuba River using a non-parametric tolerance limits approach. The development of the Yuba RMT curves was based on the measured values from 242 depth observations and 236 velocity observations at *O. mykiss* redds from the 2010 and 2011 redd surveys. Several velocity measurements were excluded due to meter error, resulting in the discrepancy between depth and velocity observations used to develop the RMT HSCs. The RMT HSCs are the most restrictive of the three sets of hydraulic HSCs considered in this study by assigning the smallest ranges of water depth and mean column velocity that are considered suitable for *O. mykiss* spawning. The RMT velocity HSC has the smallest range of highest suitability velocity values from 0.36-0.69 m/s (1.18-2.25 ft/s) and also the smallest range of velocity values considered at all suitable for spawning *O. mykiss*, with a maximum suitable velocity of 1.11 m/s (3.63 ft/s). The RMT depth HSC is much more restrictive than the other two curves, with maximum suitability assigned to depths of 0.38-0.84 m (1.25-2.76 ft) and a maximum suitable depth 1.30 m (4.27 ft).
3.5.3 Substrate HSCs

In addition to the hydraulic attributes of water depth and velocity, substrate size is another important physical variable used to characterize microhabitat suitability for _O. mykiss_ spawning. The sediment size composition of the substrate may be the most important factor in the selection of spawning habitat, and the availability of suitably-sized gravels can limit the spawning success and the number of fish able to spawn in the stream (Buffington et al. 2004; Kondolf and Wolman 1993). Kondolf and Wolman (1993) found that _O. mykiss_ prefer to spawn in small gravel to
cobble-sized substrate, with a median diameter ranging from 10.4 to 46.0 mm (0.41-1.81 inches), and also suggested that salmonids can spawn in substrates with a median diameter up to 10% of fish length. Given that observations of fork length of *O. mykiss* using the Vaki Riverwatcher system (described below) in the fish ladders at Daguerre Point Dam yield a range of fish sizes from 180-600 mm (7.1 to 23.6 inches), this 10% benchmark produces estimates of the suitable bed material size of ~18-60 mm (Massa et al. 2012).

A suite of substrate HSCs were developed to test the performance of several HSCs with varying relationships between mean substrate diameter and habitat suitability for *O. mykiss* spawning. While existing USFWS, TRTAC, and RMT hydraulic HSCs were chosen for consideration, the existing substrate HSCs were not usable, because each source requires a unique type of substrate data that is different from what was collected by the RMT using the near-census framework (section 3.4). Instead, new substrate HSCs were developed and tested based on the substrate data collected by the RMT. In the study by Gard (2010), the HSC for each variable was developed based on the assumption that each variable is independent of the others. Thus, each HSC is a univariate function and can be used in isolation as well as in combination with other HSCs. The same is true for the RMT and TRTAC HSCs- they are all univariate functions. Despite calls in the literature for co-dependent functions for depth, velocity, and substrate (e.g. Moir and Pasternack 2010), there is often insufficient data quantity to produce those complex multivariate functions. Thus, this study treated each HSC using the same assumption as the studies that produced them- each variable’s HSC is treated as an independent variable that can be used with other univariate functions.

There were eight different substrate HSCs tested. Some were developed based on observations of substrate size used for spawning by *O. mykiss* in the lower Yuba River from the redd surveys. Some curves were more general representations of the typical substrate size that *O. mykiss* tend to utilize for spawning based on information from the literature and from fisheries biologists on the RMT. The substrate HSCs fall into three distinct categories of curves - binomial curves, statistical function curves, and hybrid curves. The binomial curves act as “on/off” switches; there is a range of suitable values for mean diameter where the suitability is equal to 1, and outside of that suitable range the suitability is equal to 0. Substrate HSCs 6, 7, 9, and 10 are binomial substrate curves (Figure 12). For the statistical function curves, habitat suitability varies from 0 to 1 throughout the range of substrate sizes based on the frequency distribution of Dmean size utilized at the observed redd locations as determined from the Dmean raster. Substrate HSCs 2 and 8 are statistical function substrate HSCs (Figure 13). The hybrid curves are either step functions like substrate HSC 11, where a range of Dmean values is highly suitable and another range is assigned an intermediate suitability value, or a compound curve like substrate HSC 12, which has a range of Dmean values assigned the highest suitability value, and the suitability decreases linearly to a Dmean value beyond which the habitat suitability is zero (Figure 14). HSCs were turned into equations that define the value of suitability across the range of applicable values for each physical habitat variable.

Substrate HSC 2 (S2) was developed from the mean diameter value at each of the observed redd locations extracted from the substrate Dmean map of the lower Yuba River. The distribution of mean diameter values at each of the 261 redds was classified by the visual classification size
ranges that were used when the substrate survey was conducted. The suitability assigned to each bin is proportional to the frequency of redd observations within each interval of mean diameter.

Substrate HSC 6 (S6) assigned a suitability of 1 for areas where mean diameter is between 2 mm and 90 mm. Suitability is 0 for mean diameter values that fall outside of this range.

Substrate HSC 7 (S7) assigned a suitability of 1 for areas where mean diameter is between 32 mm and 90 mm. Suitability is 0 for mean diameter values that fall outside of this range.

The development of substrate HSC 8 (S8) was very similar to the development of S2, except that the value of mean diameter at all redd locations was classified into 10 mm bins rather than the ranges defined in the visual classification system. The suitability of each 10 mm bin is proportional to the frequency of redd observations whose $D_{\text{mean}}$ value fell within that interval.

Substrate HSC 9 (S9) assigned a suitability of 1 for areas where mean diameter is between 32 mm and 200 mm. Suitability is 0 for mean diameter values that fall outside of this range.

Substrate HSC 10 (S10) is another binomial suitability curve, where a suitability of 1 is assigned for the range of $D_{\text{mean}}$ from 32-150 mm, chosen so as to be intermediate to curves S7 and S9. All values of $D_{\text{mean}}$ outside of this range have a suitability of 0.

Substrate HSC 11 (S11) assigned a suitability of 1 to the $D_{\text{mean}}$ range from 32-100 mm, then suitability was decreased linearly from a suitability of 1 at 100 mm to a suitability of 0 at 200 mm.

Substrate HSC 12 (S12) assigned a suitability value of 1 for the range of $D_{\text{mean}}$ from 32-90 mm and an intermediate suitability of 0.4 for the $D_{\text{mean}}$ range of 90-200 mm. All values of $D_{\text{mean}}$ outside of these ranges have a suitability of 0.
Figure 12. Binomial substrate HSCs

Figure 13. Statistical function substrate HSCs
**3.5.4 Habitat Suitability Indices**

A habitat suitability index (HSI) is a value from 0 to 1 that results from applying an HSC to the value of a single physical variable at a single point in the river. To create a map of HSI using ArcGIS, the equation for an individual HSC is applied to the raster for a single physical variable using the raster calculator. It is also possible to mathematically combine the HSI values for multiple individual physical variables to get an HSI value that is representative of the suitability of several physical habitat variables at a single point.

HSI rasters for individual variables or combinations of variables may be classified into habitat suitability bins that assign a conceptual meaning to a range of HSI values. Because HSI bins are ordered from lowest to highest values over the range of 0 to 1, they are interpreted to equate with various levels of habitat quality and ecological functionality of the physical habitat based on expert judgment. Areas where HSI is exactly 0 are non-habitat by definition. The nomenclature for HSI bins depends on the number of HSI bins used and the numerical range of each bin. A significant benefit of grouping HSI values into bins and only analyzing categorical occurrence within a bin rather than analyzing raw HSI values is that categorical analysis greatly reduces the uncertainty in HSI analysis resulting from propagation of error through all prior steps in the 2D physical habitat modeling workflow. Topographic error is usually the largest single source of error in 2D physical habitat modeling, but other sources of error come from flow inputs measurement inaccuracy (i.e., discharge is only measured to within ~5-10% in most field data and water surface elevation fluctuations increase with increasing velocity and discharge, so they can be off by 1 to 10 cm from the 15-minute averaged value), turbulence closure simplifications, boundary roughness parameterization, and the structural limits of 2D fluid mechanics approximations when the channel is in fact sloped, meandering, and bounded by a porous bed. All of these sources of error propagate through complex nonlinear equations, including the HSCs, causing error in HSI values. A claim that an HSI value is exactly 0.343 would be extremely difficult to defend. Meanwhile, substantiating a claim that an HSI value is between 0.2 and 0.4 is more defensible and could be done using a procedure such as that used in Section 3.1.1.

Any delineation of HSI intervals can be used to define habitat quality classes. Leclerc et al. (1995) delineated arbitrary HSI bins of 0-0.1, 0.1-0.4, 0.4-0.7, and 0.7-1 to represent unacceptable, low acceptability, medium-quality habitat, and high-quality habitat. These HSI
bins were adopted by Pasternack et al. (2004) and Elkins et al. (2007). Pasternack (2008) first used the smaller HSI intervals of 0.2, with the highest HSI values from 0.6-1 grouped into a single class of high-habitat quality on the basis of expert judgment. While subjectively defined, both HSI bin schematics have successful been used to bioverify physical habitat models for Chinook spawning in the Central Valley of California (Pasternack et al. 2004; Elkins et al. 2007; Pasternack 2008). In the absence of additional information, HSI bins were delineated with HSI intervals of 0.2 to define the habitat quality classes for the study of *O. mykiss* spawning habitat (Table 5). HSI results for the lower Yuba River were grouped into habitat quality classes with the reclassify raster tool in ArcMap based on the thresholds of the HSI bins. Mapping HSI results according to habitat quality class allowed for the analysis of the spatial distribution and patterns of habitat quality and availability in the river.

**Table 5. Habitat Suitability Index delineations and Habitat Quality Class descriptions.**

<table>
<thead>
<tr>
<th>Habitat Suitability Bin</th>
<th>Habitat Quality Class</th>
</tr>
</thead>
<tbody>
<tr>
<td>HSI = 0</td>
<td>Non-habitat</td>
</tr>
<tr>
<td>0&lt;HSI&lt;0.2</td>
<td>Poor quality habitat</td>
</tr>
<tr>
<td>0.2&lt;HSI&lt;0.4</td>
<td>Low quality habitat</td>
</tr>
<tr>
<td>0.4&lt;HSI&lt;0.6</td>
<td>Medium habitat quality</td>
</tr>
<tr>
<td>0.6&lt;HSI&lt;0.8</td>
<td>High quality habitat</td>
</tr>
<tr>
<td>0.8&lt;HSI&lt;1.0</td>
<td>Highest quality habitat</td>
</tr>
</tbody>
</table>

### 3.5.4.1 Hydraulic Habitat Suitability Index

The hydraulic habitat suitability index (HHSI) is calculated by combining the depth HSI and velocity HSI. HHSI provides a value of habitat suitability based on hydraulic conditions at each point on the river without considering substrate or any other variable. Using the depth and velocity rasters created from the 2D model output at the discharge of interest, depth habitat suitability index (DHSI) and velocity habitat suitability index (VHSI) rasters were created by using piece-wise functions that define suitability across the range of depth or velocity values for each HSC and the raster calculator tool in ArcMap. Equation 1 shows the way the DHSI and VHSI rasters were combined by taking the geometric mean to create the hydraulic habitat suitability index (HHSI) rasters.

\[
HHSI = (DHSI \times VHSI)^{1/2}
\]

An HHSI raster is a grid of suitability values based on hydraulic conditions at every point of the lower Yuba River for each of the three sets of hydraulic HSCs at each of the three modeled discharges corresponding to the redd groups, \( Q_{\text{low}} \) (24.9 \( \text{m}^3/\text{s} \), 880 cfs), \( Q_{\text{mid}} \) (28.3 \( \text{m}^3/\text{s} \), 1,000 cfs), and \( Q_{\text{high}} \) (36.8 \( \text{m}^3/\text{s} \), 1,300 cfs). The HHSI rasters for each pair of HSCs (i.e., USFWS, TRTAC, and RMT) at each flow rate were used to compare against utilization observations for the three redd groups to determine the best performing set of hydraulic HSCs using the bioverification metrics explained in Section 3.6.
3.5.4.2  **Substrate Habitat Suitability Index**

The substrate HSCs were turned into equations that define the value of suitability for each curve over the range of mean diameter values in the lower Yuba River. Using the raster calculator in ArcMap, substrate HSC equations and the $D_{\text{mean}}$ raster were used to create substrate habitat suitability index (SHSI) rasters for each of the eight substrate HSCs.

3.5.4.3  **Combined Habitat Suitability Index**

The combined habitat suitability index (CHSI) integrates the individual hydraulic HSIs with one of the substrate HSI. CHSI is calculated differently for the different categories of substrate HSCs. For the binomial curves (S6, S7, S9, S10), CHSI was calculated using the raster calculator in ArcMap by multiplying the value of HHSI by the value of SHSI (i.e., 0 or 1) at each point in the river, as in Eq. 2.

$$CHSI = HHSI \times SHSI$$  \[2\]

This method maintained the value of HHSI at all points where the mean diameter was within the suitable range, and assigned a suitability value of 0 to all areas with mean diameter outside of the suitable range.

The method to calculate CHSI for the statistical function curves is different, since the substrate suitability value for these curves can be any value within the range of 0 to 1, rather than equal to 0 or to 1. For the complex function curves, the values of DHSI, VHSI, and SHSI are combined equally into the CHSI value by taking the geometric mean, as shown in Eq. 3.

$$CHSI = (DHSI \times VHSI \times SHSI)^{1/3}$$  \[3\]

This method of calculating CHSI was used for CHSI methods incorporating S2, S8, and S11, as it is the best calculation method available when suitability values range from 0 to 1.

Calculating CHSI with the S12 curve was slightly different, since S12 has a range of $D_{\text{mean}}$ values where suitability is 1, a range of values where suitability is 0.4, and the remainder of the range of substrate values is assigned a suitability of 0. For the range where suitability is 1, the approach used for the binomial curves was applied to calculate CHSI from the product of HHSI and SHSI, resulting in the CHSI value being the same as the HHSI value in these areas (CHSI=HHSI*1). For the areas of the river that fell into the range with an intermediate SHSI of 0.4, CHSI was calculated by taking the geometric mean of VHSI, DHSI, and SHSI, as in Eq. 4.

$$CHSI = (DHSI \times VHSI \times 0.4)^{1/3}$$  \[4\]

Twenty-four CHSI rasters result from the three sets of hydraulic HSCs, combined with each of the eight substrate HSCs applied to the hydraulic and substrate data from the lower Yuba River. The CHSI rasters at each of the three discharges of interest were compared to determine the best performing combined set of hydraulic and substrate HSCs using the bioverification metrics explained in Section 3.6.
3.5.4.4 Physical Habitat Modeling

A major benefit of using HSCs to quantify indices of habitat suitability is the ability to combine the suitability assessments with model results or field observations to create physical habitat models. These physical habitat models provide an estimate of the quality and availability of habitat at the river reach or segment scale. Quantifying the availability of suitable habitat is essential to ensuring that adequate spawning habitat is available in a stream and in determining where and how to conduct restoration efforts.

The traditional approach of habitat suitability modeling to determine habitat quality and availability in a stream is based on transect-based habitat simulation, such as the often used PHABSIM and others. Physical habitat models that are transect-based depend on measurements of physical habitat attributes made only at transects within “representative” reaches of the stream and the HSCs are applied to the data collected at the transects. The habitat suitability results at a given transect are applied to the areas between measured transects, which results in the habitat availability of the entire stream being only an estimate based on the physical habitat conditions present at the transects. The reliance on transects and reach-averaged estimates of habitat availability creates significant limitations, including that transect-based physical habitat models cannot assess the availability of suitable habitat at a scale relevant to individual fish (Leclerc et al. 1994), the number of transects used in habitat suitability modeling may affect the discharge-habitat relationships developed and the accuracy of instream flow recommendations (Gard 2005), and the spatial variability and distribution of suitable physical habitat cannot be analyzed.

Habitat suitability modeling for salmonids and other fish has progressed rapidly in recent years with advancements in remote sensing, 2D modeling, and spatial and 3D tools in GIS (Pasternack and Senter 2011). In this study, the use of 2D hydrodynamic model results with near-census substrate data allows the representation of hydraulic and substrate conditions in the river at a much finer resolution and on a larger scale compared to traditional transect-based sampling or the use of 1D model results. The combination of these high-resolution hydraulic and substrate datasets with HSCs improves upon the traditional methods of habitat suitability modeling and habitat availability assessment by: (1) providing the capacity to quantify habitat quality at every cell within the model domain; (2) more accurate representations of areas of complex flow such as eddies and areas of recirculation (Ghanem et al. 1996); (3) analysis of the spatial patterns and distribution of physical habitat quality and availability; and (4) presumably more accurate determination of the quantity of suitable habitat available. The spatial distribution and patterns of available habitat can be studied to investigate the influence of larger scale river processes on the development of preferred microhabitat. The development of 2D physical habitat models and the ability to map habitat quality provides a novel predictive capability in determining where *O. mykiss* will spawn and the physical habitat conditions preferred for spawning, which could have implications in future instream flow negotiations and in evaluating river rehabilitation efforts. The coupling of 2D modeling with habitat suitability has been a useful tool in comparing restoration designs and monitoring restoration projects (Brown and Pasternack 2009; Elkins et al. 2007; Wheaton et al. 2004a), and is a valuable method of evaluating the *O. mykiss* spawning habitat quality and quantity available in the lower Yuba River.

In this study, the HHSI and CHSI rasters resulting from the application of hydraulic and substrate HSCs to near-census data for depth, velocity, and substrate size provided a
representation of habitat suitability at each 0.914 m by 0.914 m (3 ft by 3 ft) raster cell of the lower Yuba River. The HHSI and CHSI values in the raster cells within the wetted area were interpreted to represent the various levels of habitat quality available according to the habitat quality classes defined in Table 5. The HHSI and CHSI rasters are therefore regarded as spatially-explicit physical habitat models, from which the utilization and availability of spawning habitat can be quantified.

3.6 Physical Habitat Model Bioverification

2D physical habitat models provided predictions of spawning habitat quality (i.e., HSI values and bin classification) resulting from the application of HSCs to 2D hydrodynamic model results (with and without substrate data). These predictions of spawning habitat quality were then tested against real observations using a method called bioverification, which tested each model’s capacity to accurately characterize habitat quality by determining its ability to predict utilization of high-quality habitat for spawning as well as the preference and avoidance for utilization of the various levels of habitat quality. In general, a bioverified model should predict higher quality spawning habitat in areas where spawning was observed compared to areas that were not utilized for spawning. Predictions of spawning habitat quality made by each physical habitat model were tested using the biological observations of spawning in the lower Yuba River to determine which model best predicts high-quality habitat in the locations utilized for spawning and lower quality habitat in areas not utilized for spawning. The three HHSI physical habitat models and twenty-four CHSI physical habitat models were tested using the bioverification metrics. This approach assumes that *O. mykiss* adult spawners seek and utilize locations with preferred physical attributes.

Bioverification of physical habitat models was conducted using the biological observation data from the lower Yuba River, with the three redd groups providing a test of each model at three different discharges. The use of these bioverification techniques allowed for independent testing of HSCs and provided the means by which to determine the set of HSCs that provided reasonable models and the best model of *O. mykiss* physical spawning habitat. A bioverified physical habitat model is capable of providing reliable predictions of habitat quality in the lower Yuba River and is useful in applying to management problems. The best bioverified physical habitat model was used in further analyses of *O. mykiss* spawning habitat utilization and spatial patterns in the lower Yuba River. Three metrics of bioverification were employed to test and compare the predictive performance of the physical habitat models.

3.6.1 HSI Difference Test

A simple performance indicator for a physical habitat model is the ability to show a strong differentiation between the habitat quality of utilized and non-utilized (i.e., available) spawning locations. The locations of redds in the three groups, Qlow, Qmid, and Qhigh make up three datasets of utilized locations, with n=43, 54, and 94, respectively. The ETGeowizards toolbox in ArcGIS 10.1 was used to create random point datasets of the same size as the three utilized datasets. The points in these datasets were randomly distributed within the wetted area of the corresponding discharge and were used as a random selection of non-utilized locations. The HSI values at the utilized locations and at the non-utilized points were extracted from the physical habitat models of HHSI or CHSI using the Interpolate Shape tool in ArcMap. HSI difference equals the mean...
HSI at utilized points minus the mean HSI at non-utilized points; it was calculated for each of the HHSI and CHSI physical habitat models.

The HSI difference test is a simple test of bioverification. If the HSCs and resulting physical habitat models are correct in predicting the conditions suitable for *O. mykiss* spawning, the utilized points should be located in areas of high habitat quality. To pass the HSI difference test, the mean HSI of the utilized points must be higher than the mean HSI of the set of non-utilized points. The best performing physical habitat model for this test was that which had the greatest HSI difference. Note that this test has no statistical significance taken alone, which is why it is deemed a simple test. The HSI difference test was conducted on the three HHSI models, as well as the three CHSI models resulting from the hydraulic HSCs combined with the best performing substrate HSC.

### 3.6.2 Mann-Whitney U Test

The Mann-Whitney U Test is a non-parametric statistical test to compare the distributions of two independent samples using rank sums, specifically by testing whether one distribution is stochastically greater than the other (Mann and Whitney 1947). The test is used here to evaluate the statistical significance of the differentiation in habitat quality between utilized and non-utilized locations among the different physical habitat models. HSI values at the utilized and non-utilized locations were extracted from each physical habitat model, and then the HSI values were ranked from smallest to largest. The Mann-Whitney U test statistic was calculated for each of the three redd groups according to standard equations.

The Mann-Whitney U test was performed with a significance level of 5 % (2-sided). Tests with a *p*-value < 0.05 had a statistically significant difference between the median HSI for utilized and non-utilized locations for the given physical habitat model. Tests with a *p*-value > 0.05 failed bioverification, because the difference between the medians of the two samples is not statistically significant at the 5 % significance level. The physical habitat models that had a statistically significant Mann-Whitney U test result (*p*-value < 0.05) were then used to calculate the median HSI values of the utilized and non-utilized samples. Physical habitat models that had higher median HSI at utilized points compared to the non-utilized points were considered bioverified according to the Mann-Whitney U test, since the physical habitat model predicted higher habitat quality at the locations used for spawning over the sample of locations not used for spawning with statistical significance. The Mann-Whitney U test was conducted on the three HHSI models as well as the three CHSI models resulting from the hydraulic HSCs combined with the best performing substrate HSC.

### 3.6.3 Electivity and the Forage Ratio

The most restrictive test of bioverification used for the *O. mykiss* spawning analysis, as well as the most widely used in this study, was a test of electivity using the forage ratio (FR). The FR was originally defined to indicate an organism’s preference or avoidance for a certain type of prey, calculated as the ratio of the relative abundance of the prey item in the organism’s stomach to the availability of that prey item in the environment (Hess and Swartz 1940; Ivlev 1961; Savage 1931; Shorygin 1939). The FR has been adapted to indicate preference for a resource or domain in other applications and has been widely used as a metric of electivity in ecology to
determine resource and habitat selection behavior (e.g., Johnson 1980; Kobayashi et al. 2008; Williams 1938).

3.6.3.1  **Forage Ratio Concept**

The FR provides a metric that assesses an organism’s behavior and selection of habitat relative to no selectivity. In this study, the FR was used to indicate preference or avoidance of habitat used by *O. mykiss* for spawning. The FR is the ratio of percent utilization (%U_i) to percent available area (%A_i), where “i” indicates the specific class or domain being evaluated (e.g., habitat quality class, MU, reach, etc.), as in Eq. 7. Percent utilization is calculated as the number of redds observed within a designated domain divided by the total number of redds (Eq. 5). Percent available area is calculated as the wetted area that is predicted to be within a specific domain divided by the total wetted area at the discharge of interest (Eq. 6).

\[
\%U_i = 100 \times \frac{\text{redds}_i}{\text{total redds}} \quad [5]
\]

\[
\%A_i = 100 \times \frac{\text{bed area}_i}{\text{total area}} \quad [6]
\]

\[
FR = \frac{\%U_i}{\%A_i} \quad [7]
\]

An FR equal to 1 represents uniform distribution of redds—the percent occurrence is exactly proportional to the percent available area in that domain. Given random chance of the occurrence of redds in a designated domain (FR=1), there is no disproportionate use of the domain, either positive or negative, which would indicate preference or avoidance of the area in the domain of interest. An FR value > 1 indicates preference for the designated domain, as it is being utilized in a greater proportion than it is available in the landscape, while an FR value < 1 indicates avoidance of the domain, as it is being utilized is in a smaller proportion than it is available in the landscape. However, the odds of FR values exactly equaling 1 are very low. For values that are higher or lower than 1, FR values indicate the percent deviation from random chance. For example, an FR of 1.5 indicates that the percent occurrence is 50% greater than would be expected at random and a value of 0.5 indicates that the percent occurrence is 50% less than what would be expected at random.

One of the criticisms of the FR is that it is asymmetrical about the random distribution FR=1 and, in theory, values of FR can range from 0 to infinity. This can cause issues in the event of rare occurrences and domains with very low available area, as the FR would show a very strong preference for the domain even though the high FR value is only a result of the very small percent available area in the denominator. A constraint was imposed on the FR analysis for this study in order to mitigate this issue. Domains that do not have at least 1% of the total available area were excluded from the electivity analysis. This constraint effectively limits the FR to a maximum value of 100, which would occur under the unlikely case that 100% of occurrences were located in only 1% of the total available area. Excluding these very small percent area domains prevented extremely high FR values that did not represent a true preference for that habitat quality class or domain, but were only the result of very low percent available area.
Another criticism of using the FR test to determine habitat preference and avoidance is the dependence on percent available area in each domain, which can be difficult to accurately quantify. It is especially difficult to quantify available area in the case of transect-based habitat suitability models that depend on habitat availability measured at transects that are assumed to represent the availability of habitat in the reaches between transects. This issue is diminished when habitat availability can be accurately quantified using the near-census approach.

3.6.3.2 Lower Yuba River Forage Ratio Analysis

Electivity analysis using the FR test was performed on the three HHSI and twenty-four CHSI physical habitat model results for each of the three redd groups. The habitat quality classes designated in Table 5 were the domains used in bioverification of the HHSI and CHSI physical habitat models. Percent redd occurrence was calculated by extracting the value of HHSI or CHSI from the raster at each of the observed redd locations and determining how many redds occurred within each habitat quality class. Percent available area in each habitat quality class was calculated by reclassifying the raster in ArcMap into the habitat quality classes, which provided a count of cells that are within each habitat quality class. The count of cells in a habitat quality class divided by the total count of cells in the wetted area at a specific modeled discharge yielded the percent available area values.

The FR test was also conducted to determine preference and avoidance for larger scale river characteristics, such as river reach, MU, or area of topographic change process. The components of the FR were calculated in the same way as for the HHSI or CHSI results, where percent utilization was calculated using the count of redds observed in each domain divided by the total count of redds, and percent area was calculated as the wetted area within each domain over the total wetted area at the discharge of interest.

3.6.3.3 Bootstrapping Test

Most studies that use the FR as a measure of electivity index define preference as having FR > 1, and avoidance as having FR < 1 (e.g., Lechowicz 1982; Deudero and Morales-Nin 2001; Estep et al. 2011). However, the difference from FR=1 is important in order to determine whether certain habitat quality classes were significantly preferred or avoided. For small datasets, the FR value must be further from 1 in order to indicate statistically significant preference or avoidance, because the fewer data points are used, the greater the possibility of a random occurrence of FR deviating strongly from 1.

Statistical bootstrapping is a method for assigning a measure of accuracy to sample estimates (Efron and Tibshirani 1993). When data of a given size are used to calculate any test metric, such as physical habitat models and redd locations used to calculate the FR, bootstrapping can be used to quantify the statistical confidence limits to show that the data are non-random. To do this, many random datasets of the same sample size as the real dataset are created and used to compute the test metric. With enough random datasets, the statistical distribution of the test metric values for the random datasets should be a normal distribution, and a mean and standard deviation can be calculated. The statistical confidence limits are then given by multiples of the standard deviation calculated for the normal distribution, with two standard deviations above and below the mean value yielding the upper and lower 95 % confidence limits, respectively. After determining the upper and lower statistical confidence limits for the 95 % confidence level, an
FR value that is sufficiently greater than or less than the random FR=1 can be considered nonrandom at the 95 % confidence level.

Considering the small sample sizes in this study, ranging from n=43 (for an individual redd group) to n=261 (for the total number of observations), the statistical bootstrapping test was used to determine the 95 % confidence threshold values of FR that indicate statistically significant differences from the mean of 1 for each of the FR tests conducted. Any designated domain with an FR higher than the upper 95 % confidence threshold (i.e., FR=1+2\(\sigma\), where \(\sigma\) is the standard deviation in FR from the statistical bootstrapping test) was considered preferred habitat, and any domain with an FR value falling below the lower 95 % confidence threshold (i.e., FR=1-2\(\sigma\)) was considered avoided habitat. Any domains with an FR value within the upper and lower 95 % confidence interval thresholds was considered tolerated habitat.

For the statistical bootstrapping test for this study, 10 datasets of randomly distributed points within the wetted area of interest were created using an ETGeowizards tool in ArcMap 10.1 and used to calculate the standard deviation in the FR values of each domain. For the HHSI and CHSI physical habitat models, sets of randomly distributed points were created that were the same sample size as the number of observed O. mykiss redds in each redd group. For each of the 10 random datasets, the HSI values at each of the random points of the 10 datasets from the best performing HHSI and CHSI physical habitat models were used to calculate the FR for each habitat quality class. The mean FR value and the standard deviation of the FR values were calculated for each habitat quality class from the 10 random datasets at each of the three discharges. The 95 % confidence interval thresholds were taken as the FR values two standard deviations away from the mean FR value of each habitat quality class. The weighted averages of the 95 % confidence interval threshold FR values were calculated based on the percent area in each habitat quality class.

The bootstrapping test was also conducted to determine preference and avoidance thresholds to use in the FR test based on utilization of reach, substrate size, MUs, and TCPs. For these bootstrapping tests, FR was calculated for each domain for the 10 random datasets and the weighted average of the 95 % confidence limit FR values for each domain was calculated based on the percent area in the domain at the discharge of interest. The preference and avoidance thresholds for each discharge were used to interpret the results from each of the FR tests on the three redd groups or for the group of all redds combined, as applicable to each test. The preference and avoidance thresholds applied to each of the FR tests for the various spatial scales and physical parameters are presented in the corresponding Results section for each test.

3.6.3.4 FR Test Bioverification Performance Indicators

The FR test of a physical habitat model was required to meet two performance indicators in order for the model to be bioverified and therefore considered a successful model of physical microhabitat for O. mykiss spawning in the lower Yuba River.

Performance Indicator 1. One or more designated domains must be preferred (FR greater than the upper 95 % confidence limit) and one or more domains avoided (FR lower than the lower 95 % confidence limit). A physical habitat model must predict areas of both preference and avoidance, otherwise it provides only a trivial prediction. With a model that predicts both areas
of preference and areas of avoidance, a more precise and risky prediction is made as to where *O. mykiss* will spawn in the lower Yuba River. **Figure 15** illustrates this idea, where the top image shows a model that provides a trivial prediction that the fish will spawn anywhere on the river because the whole area is designated as preferred habitat, while the bottom image represents a riskier model that provides a more precise and meaningful prediction of specific areas that are expected to be preferred for spawning and certain areas that are designated as avoided habitat and are not expected to be utilized for spawning.

**Figure 15.** A physical habitat model that indicates areas of both preference and avoidance provides a more meaningful prediction of *O. mykiss* spawning habitat.

**Performance Indicator 2.** For domains such as habitat quality classes that have a logical order inherent to them, the FR test results must respect that order. In physical habitat modeling, the basic idea is that high habitat suitability indicates high habitat quality. Therefore, for a HHSI or CHSI physical model to be valid, the FR values must be higher for domains of high HSI and high habitat quality, and lower for domains of low HSI and low habitat quality. More specifically, the magnitude of FR should decrease with decreasing habitat quality. If the FR test of a physical habitat model determined from a set of HSCs shows preference for redd occurrence in areas of low HSI and avoidance for areas of high HSI, the HSCs do not successfully delineate the suitable habitat conditions for *O. mykiss* spawning.

The top image in **Figure 16** illustrates a scenario in which there is a preference for areas of low HSI, represented through the disproportionate occurrence of redds (represented as black dots) in areas of low HSI (red rectangles), the HSCs are violated, because what was predicted as highly suitable habitat was not preferred habitat for *O. mykiss* spawning and the model is not bioverified. The bottom image in **Figure 16** illustrates a scenario in which the bioverified physical habitat model is consistent with the HSCs, because the majority of redd occurrences are within areas of high HSI and therefore high-quality habitat.
Figure 16. The top image shows a model that violates the predictions from the HSCs since areas of low quality habitat are preferred, and the bottom image shows a model that is consistent with the predictions from the HSCs since areas of high-quality habitat are preferred.

Figure 17 illustrates the second performance indicator of bioverification, in that FR values should generally decrease as habitat quality decreases, or at least decrease in selectivity category from preference to tolerance to avoidance with decreasing habitat quality. Models in which the FR value increases for lower habitat quality classes (such as the red lines in Figure 17) fail the second performance indicator of bioverification based on the FR test. The green line in Figure 17 represents the trend in the FR test results of a physical habitat model with consistently decreasing FR values with decreasing habitat quality, and is therefore a bioverified model.
Figure 17. The FR values of a bioverified physical habitat model should decrease with decreasing habitat quality (green). The red lines indicate trends in the FR values across habitat quality classes that do not meet the second bioverification performance indicator.

If a model failed performance indicator 2, further scrutiny was used to determine why it failed using the decision tree in Figure 18. The models were considered on a case-by-case basis to determine if the failure of performance indicator 2 was significant enough for the model to fail bioverification completely. For example, if the explicit values of FR did not continuously decrease with decreasing habitat quality, but the trend was from preference for the highest habitat quality classes, to tolerance for the moderate habitat quality classes, then to avoidance for the lowest habitat quality classes, the model may still be considered bioverified. A model is considered to have failed this performance indicator if the FR for a higher habitat quality class fell into the tolerance range and the FR value for a lower habitat quality class came back up into the preferred range. There must be a continuous decrease from preference to tolerance to avoidance as habitat quality decreases in order for a model to have met the second performance indicator of bioverification. If two contiguous habitat quality classes had results within the same preference/tolerance/avoidance category and the FR value of the lower class was significantly higher than the FR value of the higher class, however, the model was again considered to have failed performance indicator 2. On the other hand, if the values of several contiguous habitat quality classes wavered around a similar value, the model could be considered to have met performance indicator 2 if the FR values did not differ substantially. The FR values were considered to be reasonably similar if the FR value of a lower habitat quality class was within 50% of the FR value of a higher quality class, and the model was still considered to have met bioverification performance indicator 2.
A model may have technically failed the second indicator of bioverification if there were small fluctuations between the FR values of tolerated classes, but this does not signify the model is worthless, rather, that the distinction between habitat quality classes may not represent meaningful differences in habitat quality. The classification system used to delineate the habitat quality classes in this study is strict in assigning a different class for every 0.2 interval of habitat suitability. The difference in quality between adjacent intervals of 0.2 HSI may not actually be meaningful physically or biologically to the fish. In that case, it may be beneficial in future work to define the classification system differently, and assess the FR values based on three habitat quality classes created by the aggregation of avoided, tolerated, and preferred classes. If the FR values after redefinition of the habitat class intervals do not continuously decrease with decreasing habitat quality from preferred to tolerated to avoided, the model undoubtedly fails bioverification.

It is possible for more than one physical habitat model to be bioverified according to the FR test. When choosing between several bioverified models, the model that provided the highest FR
value in the preferred domains was considered the best performing model. The model with the highest FR value in preferred domains is most accurately modeling spawning habitat preference, and it is more valuable to model the habitat that *O. mykiss* prefer to use for spawning rather than modeling where they do not spawn. The best bioverified HHSI model was therefore the model that performed the best in predicting preferred spawning habitat based on hydraulics, and the best bioverified CHSI model performed the best in predicting preferred spawning habitat based on overall microhabitat, including hydraulic conditions and substrate size.

### 3.7 Habitat-Discharge Relationship

After determining the best physical habitat model based on the bioverification tests, the most successful CHSI model was used to simulate habitat suitability at a range of discharges, and the relationship between discharge and the quantity of available spawning habitat was determined. The relationship between habitat availability and discharge is a valuable tool in determining the instream flows needed to provide adequate habitat at crucial times in the life history of the species of interest. This can be especially valuable downstream of dams where flow schedules can be optimized to meet critical habitat needs throughout the year for sensitive life stages of federally listed species. Note that because this study uses a near-census approach spanning the whole alluvial river valley with 1-m raster cells, the numerical value for habitat availability is a genuine areal extent, in contrast to past statistically based studies that yielded an index that was not a direct accounting of habitat availability.

#### 3.7.1 Weighted Usable Area

Weighted usable area (WUA) is a metric that represents the abundance of physical habitat available at a specific discharge based on statistical sampling (Bovee et al. 1998; Payne 2003). It is the traditional metric used in instream flow studies to indicate the amount of available physical habitat as a function of discharge and to determine the optimum instream flow that provides the greatest amount of available habitat. The WUA value of a specific section or cell of the river is defined as the wetted area weighted by the habitat suitability of the physical conditions in that section or cell. The combined WUA value at a given discharge is then the sum of the individually weighted suitability of cells within the wetted area. The units of WUA equate it with an area of available habitat. Some argue that WUA should be treated as an index of the relative abundance of physical habitat rather than a specified quantity of available habitat (Williamson et al. 1993), because with traditional transect-based physical habitat modeling, WUA was calculated from a statistical sampling of transects within the river rather than from the habitat suitability of the entire wetted area. In this study, WUA may be considered in terms of the actual area of available habitat at each discharge due to the more accurate quantification of available habitat from the spatially-explicit physical habitat models of the entire lower Yuba River.

WUA values were calculated as the sum of the habitat suitability values from each raster cell within the wetted area as determined from the best performing CHSI physical habitat model, multiplied by the area of a raster cell (0.914 m x 0.914 m), as in Equation 8.

\[
WUA = \sum_{i=1}^{n} CHSI_i \times A_i
\]

where \(n=\)number of rasters cells in the wetted area at a specific discharge, \(CHSI_i\) is the index of habitat suitability of cell \(i\) from the best CHSI physical habitat model, and \(A_i=\)the area of a raster
cell. As discharge increases in the lower Yuba River, more area with woody vegetative cover becomes inundated. Areas with vegetation greater than 0.61 m (2 ft) tall are excluded from the calculation of available spawning habitat, as *O. mykiss* cannot spawn in areas with well-established vegetation. At a discharge of 28.3 m$^3$/s (1,000 cfs), more than 4% of the wetted area has woody vegetative cover, so the vegetated areas were excluded from the WUA calculations at discharges greater than or equal to 28.3 m$^3$/s (1,000 cfs). For flows less than 28.3 m$^3$/s (1,000 cfs), the percent wetted area that is vegetated is insignificant and therefore the exclusion of vegetated areas from the WUA calculations was not conducted.

After determining the WUA value across a range of discharges, these values were plotted against discharge to produce a WUA-discharge curve that illustrates a functional relationship between discharge and physical microhabitat availability (Bovee et al. 1998). WUA was calculated at discharges ranging from 8.5-141.6 m$^3$/s (300-5,000 cfs) from the results of the best performing CHSI physical habitat model. Each WUA curve illustrates a discharge at which the availability of spawning habitat is highest, and the WUA curves can be used to advise decisions regarding the flows that will maximize *O. mykiss* spawning habitat availability during their spawning season. WUA-discharge relationships were calculated for three spatial scales: the entire lower Yuba River segment, above and below Daguerre Point Dam, and by individual geomorphic reach of the lower Yuba River. The evaluation of WUA-discharge relationships at these three spatial scales facilitates a more complete understanding of the availability of spawning habitat relative to discharge throughout the length of the lower Yuba River.

### 3.8 Physical-Biological Linkages

Microhabitat conditions are fundamentally important in the selection of spawning location by *O. mykiss*, but prior research has shown that there may be factors at larger spatial scales and within different spatial analysis units that also influence salmonid spawning site selection. Regions stratified by different spatially explicit physical parameters were evaluated for possible linkages or influences of these physical parameters on spawning site selection. Physical-biological linkages that may impact *O. mykiss* spawning were identified at various spatial scales using a number of spatial analysis units.

Some physical-biological linkages were evaluated based on the spatial distribution or patterning of the location of redds in relation to the physical parameter. Examples of this include the longitudinal distribution of *O. mykiss* redds in the lower Yuba River and the proximity of redds to areas of high-quality habitat or to the river bank.

Other physical-biological linkages were investigated using the FR test to evaluate *O. mykiss* preference or avoidance for spawning in areas of certain physical or geomorphic conditions. While the FR test has so far been described for use in determining preference and avoidance by habitat quality class (i.e., HSI bins), it can also be calculated for domains other than habitat quality class, such as geomorphic reach or MU. The FR test was conducted by determining the percent of redds observed in each domain and the percent available area represented by each domain to determine if there was preference for spawning habitat utilization by *O. mykiss* at the scale of the physical parameter of interest. The FR test was used to assess the linkages between the biological process of spawning and other physical habitat parameters at larger spatial scales than the microhabitat scale, providing an indication of how larger-scale geomorphic, hydraulic,
or physical processes may influence spawning site selection. Analysis of the importance of various physical parameters was conducted using the FR test, including substrate at the segment scale, geomorphic reach, MU, size of preferred MU, area of topographic change process, and size of high-quality habitat patch.

3.8.1 Two-way Stratification

For all of the physical-biological linkages, it is possible that many of the physical variables are interdependent. A two-way stratification process was conducted on some of these analyses to further evaluate any combined influence of two physical parameters on spawning site selection. The redd observations were stratified by two of the physical parameters of interest, for example determining the amount of high-quality habitat as determined from the best CHSI physical habitat model available within each geomorphic reach. Other examples of two-way stratification analyses tested in this study include determining the availability of high-quality microhabitat within the MUs preferred at each discharge as well as evaluating the preference or avoidance of specific TCPs within the MUs that were preferred at each discharge. These two-way stratification tests were conducted primarily using the Spatial Join tool in ArcMap to determine the specific domain from each of the two physical parameters of interest that the redds were observed in. These attributes of the redds were interpreted either by considering the percentage of high-quality microhabitat within the physical parameter of interest, or by conducting an FR test on the combined effect of the two physical parameters. The two-way stratification FR tests were conducted starting with the preferred domains of a physical parameter as determined from a previous FR test, and determining if any of the domains from another physical parameter were preferred or avoided within the preferred areas from the first physical parameter, using the percentage of redds observed in both domains and the percentage of wetted area that was determined to belong to both domains.

4 RESULTS

4.1 Biological Observations

There were a total of 261 redd observations made during two *O. mykiss* spawning seasons: 223 observations from January through April of 2010 and 38 observations from January through April of 2011. Results and analyses of the annual *O. mykiss* redd surveys are available in Campos and Massa (2011) and Campos and Massa (2012), including the abundance of spawning observations, the temporal distribution of redd observations throughout the spawning season, size of *O. mykiss* redds, and some initial spatial analyses of *O. mykiss* redd locations. The average size of an *O. mykiss* redd across both years of redd surveys was ~1 m$^2$, with average sizes of 0.96 m$^2$ and 1.05 m$^2$ from redd observations in 2010 and 2011, respectively (data available in Campos and Massa 2011; Campos and Massa 2012). Estimates of the average redd size were calculated by taking the average length of redds (average pot length plus average tailspill length), multiplied by the average pot width or tailspill width, whichever was larger.

Infrared-imaging technology has been used to monitor fish passage at Daguerre Point Dam in the lower Yuba River since 2003 using Vaki Riverwatcher systems. The Vaki Riverwatcher infrared systems produced by Vaki Aquaculture Systems Ltd., of Iceland, provided a tool for monitoring fish passage year-round. The Vaki Riverwatcher system records both silhouettes and electronic images of each fish passage event in both of the Daguerre Point Dam fish ladders. By capturing
silhouettes and images, fish passage can be accurately monitored even under turbid conditions. Vaki Riverwatcher systems, as well as videographic systems, are located at both the north and south ladders of Daguerre Point Dam. The Vaki Riverwatcher and videographic systems in the fish ladders at Daguerre Point Dam provides a quantitative measure of the number of individuals migrating above the dam.

The number of *O. mykiss* passing beyond Daguerre Point Dam in each of the two spawning season was compared to the number of redd observations made during each season. Most of the *O. mykiss* redd observations were above the dam, meaning the spawners were either residential individuals in the upper reaches of the lower Yuba River or were anadromous individuals that had to migrate through the fish ladders at Daguerre Point Dam in order to spawn in the upper lower Yuba River reaches. Several issues render comparison of the abundance of *O. mykiss* with the number of spawning observations problematic. Due to issues with the functionality of the Vaki Riverwatcher system, the system was not operational during much of the 2009/2010 *O. mykiss* upstream migration season. A total estimate of only 32 *O. mykiss* passed Daguerre Point Dam during the period extending from August 1, 2009 through July 31, 2010, while there were 223 redd observations in that year (January-April, 2010). During the 2010/2011 upstream migration season, the Vaki Riverwatcher systems were operational for a much larger fraction of the time, and 457 *O. mykiss* were estimated to have passed upstream through Daguerre Point Dam during the period of August 1, 2010 through July 31, 2011. During the 2010/2011 spawning season, however, high flows and resulting high turbidity and poor visibility conditions prevented the survey crew from conducting consistent weekly surveys during the January through April *O. mykiss* spawning period.

### 4.2 Physical Habitat Model Bioverification

#### 4.2.1 Hydraulic Habitat Suitability Analysis

The results of the bioverification tests on the three HHSI physical microhabitat models are presented below for each of the three discharges tested: $Q_{\text{low}}$, $Q_{\text{mid}}$, and $Q_{\text{high}}$.

#### 4.2.1.1 HHSI Difference Test

HHSI difference testing found one model to partially fail bioverification, and one model which bioverified completely (Table 6). The USFWS physical habitat model passed the HHSI difference test at $Q_{\text{low}}$ and $Q_{\text{mid}}$, but it failed bioverification at $Q_{\text{high}}$, because the mean HHSI of utilized points was less than the mean HHSI of non-utilized points. This means that the USFWS model did not predict that *O. mykiss* were expected to spawn in areas of high-quality habitat, since the average habitat quality of random, available points was actually higher than the average habitat quality at the locations utilized for spawning. The USFWS model also had the smallest difference between mean HHSI of utilized points and mean HHSI of non-utilized points, making it the worst performing model of the three according to this bioverification test. The HHSI difference test is a basic test that is not a very strenuous measure of model performance, so when a model fails this test it is really performing poorly at characterizing physical habitat quality.

The TRTAC model was bioverified according to the HHSI difference test, because of a positive difference between the average HHSI of utilized points and the average HHSI of non-utilized points across all three discharges tested. The RMT physical habitat model was bioverified according to the HHSI difference test and was the best performing model, because it resulted in
the largest positive HHSI difference of the three HHSI physical habitat models at all three discharges tested.

Table 6. Results of the HHSI difference test. HHSI difference values highlighted in red and blue represent the worst and best performing model at each discharge, respectively.

<table>
<thead>
<tr>
<th>Metric</th>
<th>FWS</th>
<th>TRTAC</th>
<th>RMT</th>
</tr>
</thead>
<tbody>
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<td>$Q_{low}$</td>
<td>$Q_{low}$</td>
</tr>
<tr>
<td>Non-utilized*</td>
<td>0.409</td>
<td>0.726</td>
<td>0.350</td>
</tr>
<tr>
<td>Utilized*</td>
<td>0.439</td>
<td>0.896</td>
<td>0.743</td>
</tr>
<tr>
<td>HHSI Diff</td>
<td><strong>0.030</strong></td>
<td>0.169</td>
<td><strong>0.393</strong></td>
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<table>
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<td>$Q_{mid}$</td>
<td>$Q_{mid}$</td>
</tr>
<tr>
<td>Non-utilized*</td>
<td>0.476</td>
<td>0.719</td>
<td>0.337</td>
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<tr>
<td>Utilized*</td>
<td>0.574</td>
<td>0.941</td>
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<tr>
<td>HHSI Diff</td>
<td><strong>0.099</strong></td>
<td>0.221</td>
<td><strong>0.412</strong></td>
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<table>
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<td>$Q_{high}$</td>
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<td>Non-utilized*</td>
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<td>0.277</td>
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<tr>
<td>Utilized*</td>
<td>0.410</td>
<td>0.836</td>
<td>0.572</td>
</tr>
<tr>
<td>HHSI Diff</td>
<td><strong>-0.099</strong></td>
<td>0.198</td>
<td><strong>0.296</strong></td>
</tr>
</tbody>
</table>

*Mean values

4.2.1.2 Mann-Whitney U Test

Results of the Mann-Whitney U test of the three HHSI physical microhabitat models demonstrate that only the RMT HHSI model was bioverified according to this metric (Table 7). The $p$-value was statistically significant across all three discharges tested for the RMT HHSI model. The median HHSI values at utilized and non-utilized locations from the RMT model at $Q_{low}$ were 0.81 and 0.12; the distributions in the two groups differed significantly (Mann-Whitney $U=486$, $n_1=n_2=43$, $p<0.05$ two-tailed). At $Q_{mid}$, the median HHSI at utilized and non-utilized locations were 0.86 and 0.22, and differed significantly (Mann-Whitney $U=572$, $n_1=n_2=54$, $p<0.05$ two-tailed). The difference between the median HHSI values of the utilized and non-utilized samples was also statistically significant at $Q_{high}$, with median values of and 0.62 and 0.09, respectively (Mann-Whitney $U=2,490$, $n_1=n_2=94$, $p<0.05$ two-tailed). The fact that there are statistically significant differences between the HHSI at locations utilized for spawning and the HHSI at random, available points shows that the RMT HHSI model indicates a strong differentiation in quality between utilized and non-utilized habitat, and that the RMT model predicts there to be high-quality habitat in the locations that *O. mykiss* were observed to utilize for spawning.
The USFWS model failed bioverification according to the Mann-Whitney U test, because there was not a statistically significant difference between the HHSI values at utilized and non-utilized points at $Q_{\text{low}}$ and $Q_{\text{mid}}$, as evidenced by the high $p$-values. $Q_{\text{high}}$ was the only discharge where there was a statistically significant difference between the median HHSI values of the utilized and non-utilized samples, which were 0.46 and 0.55, respectively (Mann-Whitney $U=3,623$, $n_1=n_2=94$, $p < 0.05$ two-tailed). However, even though there was a statistically significant difference for $Q_{\text{high}}$, the direction of the difference was opposite of that expected, with the median HHSI of non-utilized points greater than the median HHSI of utilized points, so the model fails bioverification at $Q_{\text{high}}$ as well. The TRTAC model was also not bioverified, because there were not statistically significant differences between the median HHSI at locations utilized for spawning and the median HHSI at randomly chosen non-utilized locations at any of the discharges tested.

Table 7. Results of the HHSI Mann-Whitney U test. The test was performed at a 5% significance level. Significant results are indicated by a $p$-value < 0.05, highlighted in gray.

<table>
<thead>
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<th>RMT</th>
</tr>
</thead>
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<td>43</td>
<td>43</td>
</tr>
<tr>
<td>$n_2$</td>
<td>43</td>
<td>43</td>
<td>43</td>
</tr>
<tr>
<td>U</td>
<td>864</td>
<td>887</td>
<td>486</td>
</tr>
<tr>
<td>p</td>
<td>0.601</td>
<td>0.746</td>
<td>&lt;0.05</td>
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<tbody>
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<td>$Q_{\text{mid}}$</td>
<td>$Q_{\text{mid}}$</td>
</tr>
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<td>54</td>
<td>54</td>
</tr>
<tr>
<td>$n_2$</td>
<td>54</td>
<td>54</td>
<td>54</td>
</tr>
<tr>
<td>U</td>
<td>1252</td>
<td>1253</td>
<td>572</td>
</tr>
<tr>
<td>p</td>
<td>0.204</td>
<td>0.208</td>
<td>&lt;0.05</td>
</tr>
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</table>

<table>
<thead>
<tr>
<th>Metric</th>
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<th>TRTAC</th>
<th>RMT</th>
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</thead>
<tbody>
<tr>
<td>Dataset</td>
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<td>$Q_{\text{high}}$</td>
<td>$Q_{\text{high}}$</td>
</tr>
<tr>
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<td>94</td>
<td>94</td>
<td>94</td>
</tr>
<tr>
<td>$n_2$</td>
<td>94</td>
<td>94</td>
<td>94</td>
</tr>
<tr>
<td>U</td>
<td>3623</td>
<td>3714</td>
<td>2490</td>
</tr>
<tr>
<td>p</td>
<td>&lt;0.05</td>
<td>0.059</td>
<td>&lt;0.05</td>
</tr>
</tbody>
</table>

$n_1=n_2=$ sample size of utilized and non-utilized points

$U=$ Mann-Whitney U test statistic

4.2.1.3 HHSI FR Test

Before presenting the FR values for real lower Yuba River data it is important to lay out the stochastic surrogate results that were the basis for determining if the real FR values were
The results of the HHSI FR bootstrapping test show that given the relatively small number of observations of *O. mykiss* in each flow group, there were fairly wide ranges of FR values that can be obtained from purely random numbers (Table 8). A positive outcome was that all three flow groups showed very similar FR values for the 95% confidence limits, despite varying in point numbers. These thresholds were used as the preference and avoidance thresholds by which to interpret the FR test results at each of the three discharges. In comparison, the FR values for the 95% confidence limits for Chinook salmon on the lower Yuba River (Pasternack et al. 2013) were ~0.93 and ~1.07. The reason the confidence limits are so much tighter for Chinook salmon is that there were 3,000 observations, instead of ~40-100 for *O. mykiss*, with the relatively low number of *O. mykiss* observations strongly influencing the relatively larger confidence bands.

<table>
<thead>
<tr>
<th>Threshold</th>
<th>Q_{low}</th>
<th>Q_{mid}</th>
<th>Q_{high}</th>
</tr>
</thead>
<tbody>
<tr>
<td>Avoidance</td>
<td>0.49</td>
<td>0.50</td>
<td>0.58</td>
</tr>
<tr>
<td>Preference</td>
<td>1.51</td>
<td>1.50</td>
<td>1.42</td>
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</table>

FR test results for the three HHSI models at Q_{low} show that only the RMT HHSI model met both performance indicators for bioverification (Figure 19). At Q_{low}, the RMT HHSI model met the first bioverification performance indicator by predicting both preferred and avoided habitat quality classes. RMT HHSI indicated preference for the three highest habitat quality classes (0.4-0.6, 0.6-0.8, and 0.8-1), tolerance for the habitat quality class 0.2-0.4, and avoidance for the two lowest classes. The RMT model provided a balanced predictive capability by having several habitat quality classes falling into the avoided, tolerated, and preferred categories, as well as having increasing FR values with the increasing habitat quality represented by each class. The FR value of the 0.4-0.6 class (FR=1.68) is negligibly higher than that of the 0.6-0.8 class (FR=1.66), so that does not constitute a failure of the RMT model. The FR value of 2.69 in the highest habitat quality class is well above the upper 95% confidence level and indicates a highly statistically significant preference for that class.

At Q_{low}, the USFWS HHSI model met the first bioverification performance indicator by predicting both preferred and avoided habitat quality classes. The USFWS model failed the second performance indicator, because of a demonstrated preference for the 0.4-0.6 habitat quality class, but the higher habitat quality classes of 0.6-0.8 and 0.8-1 were avoided.

The TRTAC model failed the first bioverification performance indicator, as none of the habitat quality classes were preferred.
**Figure 19.** FR test results for HHSI models at Q\textsubscript{low} (24.9 m\textsuperscript{3}/s, 880 cfs).

**Figure 20** shows a spatial comparison of the HHSI models at a cluster of redds from the Q\textsubscript{low} redd group. The top panel shows the USFWS HHSI model, where redds do not tend to fall in to highest habitat quality areas (represented by the dark blue). In fact, there are several redds that occurred in the red areas of very low habitat quality. The TRTAC model shown in the middle panel of **Figure 20** performed better, as many redds were located in the dark blue areas of high-quality habitat. The TRTAC HSCs for depth and velocity are very broad and assign high suitability to a wide range of values, and once combined into HHSI, over 60% of the wetted area is in the highest habitat quality bin at Q\textsubscript{low} (24.9 m\textsuperscript{3}/s, 880 cfs). The FR value for the highest habitat quality class is lowered by the high percentage of available area in the TRTAC model, preventing the 0.8-1 habitat quality class from being a preferred domain. The bottom panel shows the RMT HHSI model results, and it is clear that many of the redds are located in areas of medium-high to high-quality habitat. The cluster of redds on the far right-hand side of the image especially demonstrates the successful performance of the RMT HHSI model, as the cluster of redds seems to match very nicely to the dark blue area predicted to be high-quality habitat by the RMT model.
Figure 20. A comparison of the mapped HHSI results for the three HHSI methods at a cluster of redds from the $Q_{low}$ group. The black dots represent redd observations.

Similar trends were present in the FR test results among the three HHSI models at $Q_{mid}$ (Figure 21). The USFWS and RMT models met the first performance indicator of bioverification, while the TRTAC model again failed this criterion by failing to demonstrate preference for any of the habitat quality classes. The USFWS model failed the second bioverification performance indicator, because the FR value of 1.93 in the 0.6-0.8 class indicates preference, while the FR value of 0.86 for the highest habitat quality class drops that class below the preference range into the tolerated range. The RMT model met the second performance indicator for bioverification and showed a balanced predictive model, with the two lowest habitat quality classes avoided, the two middle classes tolerated, and the two highest habitat quality classes preferred, with the highest FR value corresponding to the highest habitat quality class.
A comparison of the HHSI model results at $Q_{\text{mid}}$ (28.3 m$^3$/s, 1,000 cfs) is shown in Figure 22 for a cluster of redds from the $Q_{\text{mid}}$ group. While most of the redds in the cluster occur in areas of high-quality habitat according to the USFWS HHSI model, there are large areas of high-quality habitat that are not utilized, causing the FR values for the higher habitat quality classes to be low and the model to not bioverify according to the FR test. As for the TRTAC HHSI results shown in the middle panel, most of the redds are within areas of high-quality habitat, but again the amount of wetted area designated as high-quality habitat is so large that the FR test did not show a preference for the higher habitat quality classes, causing the TRTAC HHSI model to fail bioverification. The RMT HHSI results again show a successful model, where the predicted areas of high-quality habitat correspond well with the locations of redd observations.

It is remarkable that the high-quality habitat predicted by the USFWS model almost completely differs from that predicted by the RMT model - a large swath that the USFWS model identifies as high-quality habitat is deemed non-habitat by the RMT model. This difference highlights that the two models are extremely different. Coincidentally, the primary area of observed spawning at the site occurred in the one place where both models agreed that spawning would happen. What makes the models differ in their ultimate bioverification success is that the RMT model aggressively excludes a large area as non-habitat and is able to precisely hone in on where the spawning occurred at the site. The USFWS model lacks that predictive specificity.
Figure 22. A comparison of the mapped HHSI results for the three HHSI methods at a cluster of redds from the Q<sub>mid</sub> group.

At Q<sub>high</sub> (36.8 m<sup>3</sup>/s, 1,300 cfs), the same patterns emerged amongst the three HHSI models (Figure 23). The TRTAC HHSI method failed the first indicator of bioverification by failing to have an FR value in any habitat quality classes above the preference threshold. The USFWS model met the first bioverification indicator, but failed the second indicator because of the decrease in FR from the 0.4-0.6 class to the 0.8-1 class, because preference was shown for the moderate quality habitat class and not for the higher habitat quality classes. The RMT model met both performance indicators for bioverification, and provided a high FR value of 2.60 for the highest habitat quality class.
Figure 23. FR test results for HHSI models at Q_{high} (36.8 m$^3$/s, 1,300 cfs).

Figure 24 shows an example cluster of redds from the Q_{high} redd group and a comparison of the three HHSI models at 36.8 m$^3$/s (1,300 cfs). The areas of high-quality habitat predicted by the USFWS HHSI model do not correspond well with the location of observed redds, which agrees with the failure of the USFWS model to bioverify using the FR test. The TRTAC HHSI results show that most of the redds occurred within areas of high-quality habitat, however the majority of the river is assigned as high-quality habitat. The high percent area of high habitat quality causes the FR values to be low, therefore no preference was shown for the high habitat quality classes and the TRTAC model was not bioverified. The RMT HHSI results in Figure 24 show that the areas of high-quality habitat align well with the locations of redd observations, demonstrating that the RMT HHSI model again provides good predictions of habitat quality and utilization. Most of the redds in the cluster are located in areas of the highest habitat quality predicted by the RMT model, with very few redds occurring in areas of habitat quality that are not preferred (HHSI<0.6) under the RMT model.
At all three modeled discharges, the RMT HHSI performed the best in the bioverification test based on HHSI FR. The RMT model was the only one that passed both of the performance indicators for bioverification, and it did so for the three discharges tested. The RMT model predicted classes of preference, tolerance, and avoidance, and the redd observations showed a preference for the predicted highest habitat quality classes and avoidance for the lowest quality classes. Of the three HHSI representations, the RMT model also had the highest FR value in each of the preferred quality classes and the lowest FR value in each of the avoided habitat classes.

4.2.2 Combined Habitat Suitability Analysis

CHSI physical habitat models were created for the various methods of CHSI—three HHSI models each combined with the eight substrate HSCs to produce 24 individual physical habitat models based on hydraulic conditions and substrate size. Each CHSI model is named by the HHSI model followed by the substrate method that the HHSI results were combined with (e.g., USFWS-2 is the USFWS HHSI results combined with the S2 substrate HSI).

4.2.2.1 CHSI FR Test

The results of the bootstrapping test as conducted on the RMT-12 CHSI results were similar to those for the HHSI FR test, except the thresholds for $Q_{\text{high}}$ had a narrower range of uncertainty (Table 9). The corresponding preference and avoidance thresholds were applied to the results from each of the three discharges in order to interpret the FR test results. After calculating the FR values for each habitat quality class for all of the CHSI methods, the performance indicators for
bioverification for the FR test were applied to determine which CHSI methods provided successful models of physical microhabitat for *O. mykiss* spawning.

**Table 9. 95 % confidence interval thresholds based on CHSI for the 3 redd groups**

<table>
<thead>
<tr>
<th>Threshold</th>
<th>Q_{low}</th>
<th>Q_{mid}</th>
<th>Q_{high}</th>
</tr>
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<tbody>
<tr>
<td>Avoidance</td>
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<tr>
<td>Preference</td>
<td>1.49</td>
<td>1.53</td>
<td>1.4</td>
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</table>

The results of the FR test of the CHSI models were similar to those for the HHSI models. At Q_{low} (24.9 m³/s, 880 cfs), all of the 24 CHSI models met the first performance indicator of bioverification, because they have at least one habitat quality class that is avoided and at least one class that is preferred.

All of the USFWS CHSI models failed the second bioverification performance indicator, as none of them demonstrated preference for the highest habitat quality class, while they all showed preference for one or more lower habitat quality classes (Figure 25). All of the TRTAC CHSI models failed the second performance indicator because there was not a consistent decrease from preference to tolerance to avoidance with decreasing habitat quality (Figure 26). Of the RMT CHSI models at Q_{low}, two of the RMT models (RMT-2 and RMT-8) failed to meet the second bioverification performance indicator, while six of the RMT models (RMT-6, RMT-7, RMT-9, RMT-10, RMT-11, and RMT-12) met the second performance indicator and were bioverified at Q_{low} (Figure 27). While the explicit values of FR did not consistently decrease with decreasing habitat quality for all of the bioverified RMT models (see RMT-6, 7, 11, or 12 in Table 10), there was no instance where the descending selectivity categories of preference, tolerance, and avoidance were violated, so the second bioverification performance indicator was still met.

**Figure 25. FR test results for USFWS CHSI models at Q_{low}.**
For the FR test at $Q_{\text{mid}}$ (28.3 m$^3$/s, 1,000 cfs), all of the 24 CHSI models met the first performance indicator of bioverification. All of the USFWS CHSI models except USFWS-12 failed the second performance indicator by showing preference for one of more of the lower habitat quality classes and tolerance or avoidance for a class of higher habitat quality (Figure 28).

Figure 26. FR test results for TRTAC CHSI models at $Q_{\text{low}}$.

Figure 27. FR test results for RMT CHSI models at $Q_{\text{low}}$. 
All of the TRTRAC CHSI models failed the second indicator of bioverification for failing to demonstrate a consistent decrease in FR with decreasing habitat quality (Figure 29). Three of the RMT models (RMT-2, RMT-8, and RMT-10) failed the second performance indicator of bioverification at $Q_{\text{mid}}$, while the remaining five RMT CHSI models (RMT-6, RMT-7, RMT-9, RMT-11, and RMT-12), met the second performance indicator and were bioverified at $Q_{\text{mid}}$ (Figure 30).

Figure 28. FR test results for USFWS CHSI models at $Q_{\text{mid}}$.

Figure 29. FR test results for TRTAC CHSI models at $Q_{\text{mid}}$. 
At $Q_{\text{high}}$ (36.8 m$^3$/s, 1,300 cfs), there were fewer CHSI models that met the first bioverification performance indicator than at the other two discharges tested. USFWS-2 failed the first performance indicator by failing to predict preference for any of the habitat quality classes (Figure 31). The remaining USFWS CHSI models met the first performance indicator for bioverification, but all USFWS models failed the second indicator because there was tolerance or avoidance shown for high habitat quality classes and preference for moderate to low habitat quality classes.

TRTRAC-9 failed to meet the first performance indicator, while TRTAC-2, TRTAC-6, TRTAC-7, TRTAC-10, TRTAC-11, and TRTAC-12 failed to meet the second performance indicator of bioverification (Figure 32). TRTRAC-8 was the only TRTAC CHSI model to meet both performance indicators and the model was bioverified based on the FR test at $Q_{\text{high}}$. RMT-7 failed by the first performance indicator, while RMT-2, RMT-6, RMT-10, and RMT-11 failed by the second performance indicator (Figure 33). The models RMT-8, RMT-9, and RMT-12 each met both of the performance indicators and were bioverified models based on the FR test at $Q_{\text{high}}$. 

Figure 30. FR test results for RMT CHSI models at $Q_{\text{mid}}$. 

![Graph showing FR test results for RMT CHSI models at Q=1,000 cfs.](image)
Figure 31. FR test results for USFWS CHSI models at $Q_{\text{high}}$.

Figure 32. FR test results for TRTAC CHSI models at $Q_{\text{high}}$. 
Table 10 shows the explicit FR values resulting from the FR test for each of the 24 CHSI models at the three discharges tested. The CHSI model names highlighted in yellow are those that failed the first performance indicator of bioverification, because the models failed to demonstrate preference for one or more habitat quality classes, and avoidance for one or more of the low habitat quality classes. The model names highlighted in red are those that failed the second performance indicator of bioverification because they did not show a consistent pattern from preference to tolerance to avoidance with decreasing habitat quality (see the decision tree in Figure 18). The CHSI models highlighted in green are those that met both performance indicators of bioverification at the respective discharge.
<table>
<thead>
<tr>
<th>Qlow</th>
<th>Yellow=CHSI models that failed the first performance indicator of bioverification;</th>
<th>Red=CHSI models that failed the second performance indicator of bioverification;</th>
<th>Green=CHSI models that met both bioverification performance indicators and were bioverified at the specific discharge</th>
</tr>
</thead>
<tbody>
<tr>
<td>Qmid</td>
<td><strong>Yellow=CHSI models that failed the first performance indicator of bioverification:</strong></td>
<td><strong>Red=CHSI models that failed the second performance indicator of bioverification:</strong></td>
<td><strong>Green=CHSI models that met both bioverification performance indicators and were bioverified at the specific discharge</strong></td>
</tr>
<tr>
<td>Qhigh</td>
<td>The models highlighted in green in the bottom row are those CHSI models that were bioverified for all three discharges tested. $Q_{low}$, $Q_{mid}$ and $Q_{high}$</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
In order for a CHSI model to bioverify overall, the model must have been bioverified at each of the three individual flow rates tested. Any model that failed bioverification at $Q_{low}$, $Q_{mid}$, or $Q_{high}$ was not considered a bioverified model overall. The bottom row of Table 10 shows whether the CHSI model was bioverified across all three discharges, with the green cells indicating fully bioverified models. RMT-9 and RMT-12 were the only two CHSI physical habitat models to bioverify at the three discharges tested. In order to more easily compare the bioverified models, Figures 34 to 36 show the FR results for only the two RMT CHSI models that bioverified, as well as the bioverified RMT HHSI model FR results, in order to easily show the differences in FR between the HHSI model and the CHSI models that incorporated substrate suitability.

![Figure 34. FR test results for the bioverified RMT HHSI model based on hydraulic suitability, and bioverified RMT-9 and RMT-12 CHSI models based on hydraulic and substrate suitability at $Q_{low}$.](image-url)
There are not significant differences in FR test results at the three discharges tested between the RMT HHSI model and the two bioverified RMT CHSI models (RMT-9 and RMT-12). The pattern of increasing FR values from avoided, to tolerated, to preferred habitat with increasing habitat quality is consistent. Differences between the models in defining a habitat quality class as
avoided, tolerated, or preferred are rare, such as the 0.4-0.6 class at $Q_{\text{low}}$, where RMT-12 defined tolerance while RMT HHSI and RMT-9 indicated preference, and in the 0.4-0.6 class at $Q_{\text{mid}}$, where RMT-9 indicated preference, while the other two models showed tolerance of this habitat quality class. In general, the FR values for each habitat quality class across the three discharges were very similar for the three physical habitat models, but the RMT-12 model produced the highest FR values in the highest habitat quality class of 0.8-1, and the highest or very similar FR values to the other two models in the other preferred habitat quality class of 0.6-0.8, making RMT-12 the most successful physical habitat model.

RMT-9 and RMT-12 were the only CHSI models that bioverified across all three discharges tested, and these two models were compared to determine which provided the most accurate and precise predictions of $O.\ mykiss$ spawning habitat. Comparisons of the RMT-9 and RMT-12 CHSI models with the RMT HHSI model are shown in Figures 37 through 39, with an example of the mapped physical habitat model results at a cluster from each of the redd groups. The difference between RMT-9 and RMT-12 is slight - the S9 substrate HSC assigns a suitability value of 1 to substrate sizes ranging from 32-200 mm, while S12 assigns the highest suitability of 1 to substrates sizes of 32-90 mm, and a lower suitability of 0.4 to substrate sizes of 90-200 mm. The areas of the channel designated in the 0.6-0.8 and 0.8-1 habitat quality bins were smaller for RMT-12 than for the RMT-9 CHSI or RMT HHSI models, resulting in smaller percent area values and correspondingly higher FR values for these classes. The generally higher FR values with the RMT-12 model for the two highest habitat quality classes show a stronger preference compared to the two other bioverified models. The RMT-12 CHSI model is a more precise model because it consistently showed preference for the habitat quality classes of 0.6-0.8 and 0.8-1 for the three discharges, while the RMT-9 model showed preference for all three classes between 0.4 and 1 for $Q_{\text{low}}$ and $Q_{\text{mid}}$, meaning a larger area of the lower Yuba River with relatively lower combined habitat suitability was classified as preferred habitat. RMT-12 also consistently had the higher FR value in the highest habitat quality class. While the FR values of RMT-12 for the preferred 0.6-0.8 class are not higher than those of RMT-9 at $Q_{\text{low}}$ and $Q_{\text{high}}$, the fact that the FR values for the highest habitat quality class of 0.8-1 are consistently higher for RMT-12 makes it the better physical habitat model. The FR test results on the CHSI models showed that adding in the measure of substrate suitability improved the prediction of the RMT model, as the RMT-12 CHSI model showed stronger preference for the preferred classes than the RMT HHSI model alone. Considering all of the metrics for bioverification, RMT-12 has proven to be the most accurate physical microhabitat model for $O.\ mykiss$ spawning in the lower Yuba River.
Figure 37. A comparison of the mapped CHSI results for the RMT HHSI, RMT-9 CHSI, and RMT-12 CHSI models at a cluster of redds from the $Q_{\text{low}}$ group. The black dots represent redd observations.
Figure 38. A comparison of the mapped CHSI results for the RMT HHSI, RMT-9 CHSI, and RMT-12 CHSI models at a cluster of redds from the $Q_{\text{mid}}$ group. The black dots represent redd observations.
Figure 39. A comparison of the mapped CHSI results for the RMT HHSI, RMT-9 CHSI, and RMT-12 CHSI models at a cluster of redds from the \( Q_{\text{high}} \) group. The black dots represent redd observations.

### 4.2.2.2 CHSI12 Comparison of USFWS, TRTAC, and RMT

The results of the FR test on all of the CHSI models showed that RMT-12 was bioverified and provided the best physical habitat model of microhabitat for *O. mykiss* spawning based on hydraulics and substrate size. While the CHSI FR test results showed that USFWS-12 and TRTAC-12 were not bioverified, the two simpler metrics of bioverification (CHSI difference test and the Mann-Whitney U test) were used to compare the three hydraulic models with the S12 substrate HSC and confirm this result. Comparing USFWS-12, TRTAC-12, and RMT-12 was, in essence, comparing the effect of the S12 substrate HSC. This analysis was done to demonstrate whether substrate could compensate for the inadequate hydraulic HSCs of the USFWS and TRTAC models. In other words, if the S12 HSC could “fix” the USFWS and TRTAC hydraulic HSI representations that failed, or if the hydraulic conditions were the most important factor in determining the quality of microhabitat. The CHSI difference test and the Mann-Whitney U test were used to compare the three S12 CHSI models to determine if adding in the measure of substrate could improve the performance of the USFWS or TRTAC curves to compete with the best performing RMT-12 physical habitat model.

#### CHSI Difference Test

According to the simple performance indicator of the CHSI difference test, the TRTAC-12 and RMT-12 CHSI models were bioverified at the three discharges tested by having a positive difference between the average CHSI of utilized points and the average CHSI of non-utilized points (Table 11). USFWS-12 failed the CHSI difference test, as it consistently had the smallest
differences between utilized and non-utilized points, and the difference at \( Q_{\text{high}} \) (36.8 m\(^3\)/s, 1,300 cfs) was negative, meaning the average CHSI of non-utilized points was higher than the average CHSI of utilized points. The USFWS-12 CHSI model was therefore not bioverified by the CHSI difference test because of the failure at \( Q_{\text{high}} \). TRTAC-12 passed the CHSI difference test since the model showed positive differences between the average CHSI of utilized and non-utilized points across all three discharges. The best performing CHSI model was RMT-12, as it showed larger differences between the average CHSI of utilized and non-utilized points than the other two CHSI microhabitat models.

### Table 11. Results of the CHSI difference test. CHSI difference values highlighted in red and blue represent the worst and best performing model at each discharge, respectively.

<table>
<thead>
<tr>
<th>Metric</th>
<th>FWS-12</th>
<th>TRTAC-12</th>
<th>RMT-12</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dataset</td>
<td>( Q_{\text{low}} )</td>
<td>( Q_{\text{low}} )</td>
<td>( Q_{\text{low}} )</td>
</tr>
<tr>
<td>Non-utilized*</td>
<td>0.362</td>
<td>0.557</td>
<td>0.299</td>
</tr>
<tr>
<td>Utilized*</td>
<td>0.436</td>
<td>0.835</td>
<td>0.698</td>
</tr>
<tr>
<td>CHSI difference</td>
<td>0.074</td>
<td>0.278</td>
<td>0.399</td>
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</table>

<table>
<thead>
<tr>
<th>Metric</th>
<th>FWS-12</th>
<th>TRTAC-12</th>
<th>RMT-12</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dataset</td>
<td>( Q_{\text{mid}} )</td>
<td>( Q_{\text{mid}} )</td>
<td>( Q_{\text{mid}} )</td>
</tr>
<tr>
<td>Non-utilized*</td>
<td>0.395</td>
<td>0.551</td>
<td>0.277</td>
</tr>
<tr>
<td>Utilized*</td>
<td>0.558</td>
<td>0.875</td>
<td>0.704</td>
</tr>
<tr>
<td>CHSI difference</td>
<td>0.163</td>
<td>0.324</td>
<td>0.428</td>
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</table>

<table>
<thead>
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<th>TRTAC-12</th>
<th>RMT-12</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dataset</td>
<td>( Q_{\text{high}} )</td>
<td>( Q_{\text{high}} )</td>
<td>( Q_{\text{high}} )</td>
</tr>
<tr>
<td>Non-utilized*</td>
<td>0.398</td>
<td>0.474</td>
<td>0.250</td>
</tr>
<tr>
<td>Utilized*</td>
<td>0.382</td>
<td>0.688</td>
<td>0.483</td>
</tr>
<tr>
<td>CHSI difference</td>
<td>-0.017</td>
<td>0.214</td>
<td>0.233</td>
</tr>
</tbody>
</table>

*Mean values

**Mann-Whitney U Test**

The Mann-Whitney U test results of USFWS-12, TRTAC-12, and RMT-12 again show that USFWS-12 is the worst performing model and RMT-12 is the best performing model of the three (Table 12). There are not statistically significant differences between the median CHSI values of the utilized and non-utilized samples at \( Q_{\text{low}} \) and \( Q_{\text{high}} \) with the USFWS-12 model. The median CHSI of the utilized and non-utilized samples from the USFWS-12 model at \( Q_{\text{mid}} \) were 0.63 and 0.41; the distributions in the two groups differed significantly (Mann-Whitney \( U=996, n_1=n_2=54, p < 0.05 \) two-tailed).

The TRTAC-12 and RMT-12 models passed the Mann-Whitney U test, since the results were statistically significant at all three discharges. For the TRTAC-12 model, the median CHSI values from the utilized and non-utilized samples at \( Q_{\text{low}} \) were 0.98 and 0.73; the distributions in the two groups differed significantly (Mann-Whitney \( U=661, n_1=n_2=43, p < 0.05 \) two-tailed).
Q\textsubscript{mid}, the median CHSI at utilized and non-utilized locations were 1.00 and 0.66 (Mann-Whitney \(U=894, n_1=n_2=54, p < 0.05\) two-tailed). At Q\textsubscript{high}, the median CHSI at utilized and non-utilized locations were 0.81 and 0.62 (Mann-Whitney \(U=3,375, n_1=n_2=94, p < 0.05\) two-tailed). For the RMT-12 model, the median CHSI values at utilized and non-utilized locations at Q\textsubscript{low} were 0.74 and 0.14; the distributions in the two groups differed significantly (Mann-Whitney \(U=426, n_1=n_2=43, p < 0.05\) two-tailed). At Q\textsubscript{mid}, the median CHSI at utilized and non-utilized locations were 0.74 and 0.11 (Mann-Whitney \(U=512, n_1=n_2=54, p < 0.05\) two-tailed). The difference between the median HHSI values of the utilized and non-utilized samples was also statistically significant at Q\textsubscript{high}, with median values of 0.51 and 0.04, respectively (Mann-Whitney \(U=3,203, n_1=n_2=94, p < 0.05\) two-tailed).

While the TRTAC-12 CHSI model passed the Mann-Whitney U test, it failed the more stringent FR test, so RMT-12 remained the most successful physical habitat model for \textit{O. mykiss} spawning.
Table 12. Results of the CHSI Mann-Whitney U test. The test was performed at a 5% significance level. Significant results are indicated by a p-value < 0.05, highlighted in gray.

<table>
<thead>
<tr>
<th>Metric</th>
<th>FWS-12</th>
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<th>RMT-12</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dataset Q_{low}</td>
<td>Q_{low}</td>
<td>Q_{low}</td>
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</tr>
<tr>
<td>n_{1}</td>
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<td>43</td>
<td>43</td>
</tr>
<tr>
<td>n_{2}</td>
<td>43</td>
<td>43</td>
<td>43</td>
</tr>
<tr>
<td>U</td>
<td>766</td>
<td>661</td>
<td>426</td>
</tr>
<tr>
<td>p</td>
<td>0.171</td>
<td>&lt;0.05</td>
<td>&lt;0.05</td>
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</tbody>
</table>

<table>
<thead>
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<th>Metric</th>
<th>FWS-12</th>
<th>TRTAC-12</th>
<th>RMT-12</th>
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<tbody>
<tr>
<td>Dataset Q_{mid}</td>
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<td>Q_{mid}</td>
<td></td>
</tr>
<tr>
<td>n_{1}</td>
<td>54</td>
<td>54</td>
<td>54</td>
</tr>
<tr>
<td>n_{2}</td>
<td>54</td>
<td>54</td>
<td>54</td>
</tr>
<tr>
<td>U</td>
<td>996</td>
<td>894</td>
<td>512</td>
</tr>
<tr>
<td>p</td>
<td>&lt;0.05</td>
<td>&lt;0.05</td>
<td>&lt;0.05</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Metric</th>
<th>FWS-12</th>
<th>TRTAC-12</th>
<th>RMT-12</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dataset Q_{high}</td>
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<tr>
<td>n_{1}</td>
<td>94</td>
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</tr>
<tr>
<td>n_{2}</td>
<td>94</td>
<td>94</td>
<td>94</td>
</tr>
<tr>
<td>U</td>
<td>4048</td>
<td>3375</td>
<td>3203</td>
</tr>
<tr>
<td>p</td>
<td>0.321</td>
<td>&lt;0.05</td>
<td>&lt;0.05</td>
</tr>
</tbody>
</table>

n_{1}=n_{2}=sample size of utilized and non-utilized points
U=Mann-Whitney U test statistic

In general, adding in the metric of substrate suitability improved upon the predictions based on hydraulics for physical habitat models that were already successful, namely the RMT model. For the USFWS and TRTAC models, adding in substrate suitability could not fix the fact that the models failed to successfully predict habitat utilization based on the suitability of hydraulic variables. This indicates that while substrate is helpful in improving upon the predictions of good physical habitat models, substrate size seems to be of secondary importance to the hydraulic conditions in spawning site selection.

4.3 Habitat-Discharge Relationships

The best physical microhabitat model, RMT-12, was used to compute WUA-discharge relationships for the entire lower Yuba River segment, sections above and below Daguerre Point Dam, and by individual geomorphic reach.

The WUA-discharge curve for the entire lower Yuba River segment is shown in Figure 40. The maximum WUA value was calculated at 17.61 m³/s (622 cfs). Above 24.9 m³/s (880 cfs), WUA
quickly decreased and stabilized to a fairly constant value for flows of 84.95 m$^3$/s (3,000 cfs) and higher.

![Graph of WUA vs. Discharge](image)

**Figure 40.** Segment-scale WUA discharge relationship for spawning *O. mykiss.*

The WUA-discharge curves in **Figure 41** for sections above and below Daguerre Point Dam demonstrated that the same discharge of 17.61 m$^3$/s (622 cfs) provided the peak WUA value in both sections. This is the same discharge at which the peak WUA value occurred on the segment scale. The WUA again decreased rapidly with increasing discharge, with the WUA values of the section below Daguerre Point Dam dropping off more rapidly than the section of the lower Yuba River above Daguerre Point Dam.
The WUA-discharge relationships calculated by individual geomorphic reach are shown in Figure 42. The peak WUA value for each reach occurred around generally the same discharge, with slight variations. The peak WUA value for the Marysville Reach occurred at the lowest discharge of all the reaches at 15 m$^3$/s (530 cfs). The peak WUA value for the Parks Bar Reach occurred at 17 m$^3$/s (600 cfs). The peak WUA for the Hallwood Reach occurred at 17.6 m$^3$/s (622 cfs), which is the same discharge that produced the peak WUA value as for the lower Yuba River segment and for above and below Daguerre Point Dam WUA relationships. The peak WUA for Daguerre Point Dam and Timbuctoo Bend reaches occurred at 19.8 m$^3$/s (700 cfs), and in the Dry Creek Reach, the peak WUA occurred at 24.9 m$^3$/s (880 cfs). The WUA-discharge relationship for the Englebright Dam Reach is barely visible in Figure 42, because the lack of suitably-sized substrate in the Englebright Dam Reach results in virtually no habitat available there at any discharge. Most of the WUA values for the individual reaches dropped off rapidly around 28.3 m$^3$/s (1,000 cfs), and reached fairly constant and low WUA values by around 57 m$^3$/s (2,000 cfs).

While the highest relative habitat availability is in the Hallwood Reach, it was not a reach that was frequently utilized for spawning, with only 6 of 261 (2.3 %) redd observations made in the reach. Hallwood is also the second most downstream alluvial geomorphic reach. Timbuctoo Bend and Parks Bar reaches are two of the most upstream alluvial reaches and were the most utilized reaches for spawning, with 63.0 % and 27.2 % of redd observations, respectively.

The Parks Bar Reach had the next most available habitat throughout most of the range of in-channel discharges. The Timbuctoo Bend Reach has lower relative habitat availability than most of the other reaches, except for the Marysville and Englebright Dam reaches. The discharges with the peak WUA values for Timbuctoo Bend and Parks Bar were 19.8 m$^3$/s (700 cfs) and 17
m³/s (600 cfs), respectively. The reasons for this discrepancy of higher spawning rates in upstream reaches of relatively lower habitat availability are explored in Sections 4.4.1 and 4.4.2 of this report.

4.4 Physical-Biological Linkage Analysis

Bioverification found that the RMT-12 physical habitat model performs well in predicting spawning habitat utilization based on microhabitat characteristics. The percent of redd observations found to occur in areas of preferred microhabitat under the RMT-12 model was 46 and 67 % for the three datasets tested, which shows fairly good predictive capability of the microhabitat model. However, that leaves a significant amount of O. mykiss adults that utilized areas of tolerated or avoided habitat to spawn. A greater understanding of spawning habitat preferences may be obtained by considering preferences for larger scale river characteristics and various physical-biological linkages to determine the impact of these parameters on O. mykiss spawning site utilization, in addition to the preferred microhabitat conditions. Additional analyses were conducted to evaluate the dependence of spawning site selection on variables or processes at a larger spatial scale. The results of several analyses of spawning habitat utilization and preference using various spatial scales and spatial analysis units are presented, including the segment, reach, and MU scales, areas of topographic change processes, mesohabitat patches, and spatial relationships between redd observations.

4.4.1 Lower Yuba River Segment Scale Analyses

At the segment scale, influence on O. mykiss spawning site selection beyond microhabitat attributes was investigated by assessing the longitudinal distribution of redds in the lower Yuba River and also by assessing the preference for mean substrate size at the location of all redds observed in the lower Yuba River segment.
4.4.1.1  Longitudinal Distribution of Redds

The simple question of whether spawning activity by *O. mykiss* is randomly distributed along the lower Yuba River or non-randomly organized is answered by examining the longitudinal distribution of redd observations. The probability distribution function of all 261 redds along the length of the lower Yuba River discretized into 6 m (20 ft) intervals shows utilization is clearly skewed towards the upstream end of the alluvial valley of the lower Yuba River, with most of the redd observations occurring in the Timbuctoo Bend Reach (Figure 43, see Table 1 for reach abbreviations).

![Figure 43. Longitudinal distribution of reds along the lower Yuba River.](image)

The cumulative distribution function was then calculated to mathematically determine if the reds are randomly distributed in the lower Yuba River. If the distribution is unorganized throughout the segment, there is equal probability for spawning to occur at any point along the length of the river and the cumulative distribution function is linear, represented by the dashed line in Figure 44. If the distribution of reds is non-random or is organized in some way, the cumulative distribution function is curved. Any section of the river over which the distribution is locally random is identified by the cumulative distribution function being parallel to the dashed line.

The cumulative distribution function of all 261 redd observations shown in blue is strongly curved, with very few occurrences in the downstream reaches and a sharp increase of observations further upstream in the lower Yuba River (Figure 44). While Chinook spawning habitat utilization in the lower Yuba River was found to be highly influenced by water temperature, this cannot explain the longitudinal trend in *O. mykiss* spawning observations. Cooler water temperatures in the lower Yuba River were found to extend longitudinally throughout the fall spawning season, and new Chinook redd observations appeared to be associated with the extension of thermally suitable areas downstream (RMT 2013a). Water temperature cannot explain the trend in the distribution of *O. mykiss* spawning in the upstream
reaches, because they spawn in the winter when the lower Yuba River has uniformly low water temperatures that are suitable for spawning. The longitudinal distribution of *O. mykiss* spawning sites was highly non-random, and further analyses were conducted to investigate this upstream trend (see Section 4.4.2.2), as well as other factors that may influence redd locations.

![Cumulative distribution function of redds along the lower Yuba River](image)

**Figure 44.** Cumulative distribution function of redds along the lower Yuba River

### 4.4.1.2 FR Test by Substrate Size

The FR test was conducted on the 261 redd locations to evaluate substrate size preference and avoidance for spawning at the segment scale using the substrate size utilized at each redd location as extracted from the map of mean substrate diameter of the lower Yuba River. The wetted area at 113.3 m$^3$/s (4,000 cfs) was used for calculations, because this was the highest discharge at which an *O. mykiss* redd was observed, and is therefore the discharge at which all redds are within the wetted area. Substrate size data consisted of D$_{mean}$ values ranging from 0.31-384 mm (see Section 3.4). These data were analyzed with the FR test using two different size classification schemes – first by the same classification scheme used in the original substrate survey, and second by narrow, uniform intervals of 10 mm.

The results of the bootstrapping test using all redds and the visual classification bins for substrate size provided preference and avoidance thresholds of FR=1.19 and FR=0.81. The relatively narrow band for non-statistically significant differences from FR=1 is due to the larger sample size and the fact that there are effectively only 5 substrate bins for the random points to be spread across, because the percent available area in the 0-0.0625 mm and 0.0625-2 mm bins is less than 1 %, so the likelihood of random datasets differing from FR=1 is less.

**Figure 45** shows the FR test results with the visual classification bins used to conduct the field substrate survey. The 32-90 mm visual classification bin for D$_{mean}$ was the only size class which was preferred with 95 % confidence, with an FR value of 1.47. The classification bins of 2-32
mm and 90-128 mm were avoided, and the 128-256 mm bin was tolerated for spawning utilization. The remaining three visual classification bins were not utilized for spawning in the lower Yuba River, because no redds were observed in these substrate size bins.

![Figure 45](image)

**Figure 45.** FR test results for substrate mean diameter ($D_{\text{mean}}$) at redd locations by visual substrate class.

The mean substrate diameter values at each of the redds were then classified into 10 mm bins for the range from 30-90 mm, which was the only substrate class for which a preference was shown using the first classification scheme (Figure 45). The FR test was conducted again to determine if there was preference for a specific substrate size within the more narrow, uniform intervals of 10 mm (Figure 46). The bootstrapping test of all 261 redds with 10 mm substrate size bins produced 95% confidence limits of 1.53 for the preference threshold and 0.47 for the avoidance threshold, which are used to interpret the FR tests results shown in Figure 46. The 10 mm bins are much more specific than the substrate classification system used to conduct the substrate survey in the field, where sizes from 32 mm to 90 mm were assigned to the same group of medium gravel/small cobble. A strong preference was shown for the 50-60 mm mean diameter size, with an FR value of 4.55. There was no preference for the other 10 mm bins, and the 30-40 mm and 40-50 mm bins were avoided.
This analysis of preference based on mean diameter substrate size at the redd locations was used to guide the development of most of the substrate HSCs.

### 4.4.2 Geomorphic Reach Scale Analyses

At the reach scale, influence on spawning site selection was evaluated with an FR test by individual geomorphic reach and by assessing the amount of high-quality *O. mykiss* spawning habitat (CHSI > 0.6) available in each reach.

#### 4.4.2.1 FR Test by Geomorphic Reach

The FR test for reach preference was conducted on the three redd groups, *Q*$_{low}$ (n=43), *Q*$_{mid}$ (n=54), and *Q*$_{high}$ (n=94). In order to analyze preference for spawning in a specific reach, the percent of redds observed in each reach was calculated for each redd group, as well as the percent wetted area represented in each reach at the relevant modeled discharges for *Q*$_{low}$ (24.9 m$^3$/s, 880 cfs), *Q*$_{mid}$ (28.3 m$^3$/s, 1,000 cfs), and *Q*$_{high}$ (36.8 m$^3$/s, 1,300 cfs).

The results from the bootstrapping test by reach are shown in **Table 13**, and the most conservative values of 0.33 and 1.67 for the avoidance and preference thresholds were used to interpret the FR test results for all three redd groups.

**Table 13.** 95 % confidence thresholds by reach for the 3 redd groups.

<table>
<thead>
<tr>
<th>Threshold</th>
<th><em>Q</em>$_{low}$</th>
<th><em>Q</em>$_{mid}$</th>
<th><em>Q</em>$_{high}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Avoidance</td>
<td>0.33</td>
<td>0.42</td>
<td>0.5</td>
</tr>
<tr>
<td>Preference</td>
<td>1.67</td>
<td>1.58</td>
<td>1.5</td>
</tr>
</tbody>
</table>
Figure 47 displays the results of the FR test by reach, with the most downstream reach, Marysville Reach, on the left, to the most upstream reach, Englebright Dam Reach, on the right. The Timbuctoo Bend Reach is the only reach where the FR demonstrates a strong preference by all three of the redd groups. FR values for Timbuctoo Bend ranged from 3.85-4.05 amongst the redd groups, indicating that Timbuctoo Bend was utilized for spawning about 4 times more than would be expected with random selection of a spawning site. A slight preference was shown for Parks Bar Reach at Q_{high} (FR=1.71), while it was tolerated for the other discharges tested. The remaining reaches of the lower Yuba River (Marysville, Hallwood, Daguerre Point Dam, Dry Creek, and Englebright Dam) were avoided at all three discharges. A summary of the FR test results and hydraulic and geomorphic characteristics of each reach at the three discharges are presented in Table 14. The vast majority of *O. mykiss* redds were observed in two reaches of the lower Yuba River, with 65-70% observed in the Timbuctoo Bend Reach and 26-34% in Parks Bar Reach across the three redd groups (Table 14).
Two-way Stratification: Reach and microhabitat availability

The two-way stratification technique was used to determine the amount of suitable microhabitat available in each reach, as higher percentages of high-quality habitat available could possibly help explain the preference for certain reaches over others. Using the RMT-12 CHSI model, the percent of wetted area in each habitat quality class was plotted by reach for each of the three discharges tested (Figures 48-50). The percent of area in the Timbuctoo Bend Reach that is preferred microhabitat did not change drastically with discharge, with 14.3-18 % of the wetted area in the 0.6-0.8 and 0.8-1 habitat quality classes combined at each of the three discharges. Comparing the preferred Timbuctoo Bend Reach to other non-preferred reaches, there is substantially less high-quality habitat available in Timbuctoo Bend. Timbuctoo Bend has only 3.4 % of its wetted area that is of the highest habitat quality. About half of the wetted area of Timbuctoo Bend is non-habitat and it has more area that is in the lowest habitat quality bin than several other reaches, except for the Marysville and Englebright Dam reaches, which are the
most downstream and upstream reaches, respectively. The results for the Parks Bar Reach are comparable at $Q_{\text{high}}$ to the Timbuctoo Bend Reach results, because it has very little percent area in the preferred habitat quality bins and a substantial portion of its area that is designated as non-habitat.

![Graph showing habitat quality by reach at $Q_{\text{low}}$.](image1)

Figure 48. Amount of available habitat by reach with the RMT-12 CHSI model at $Q_{\text{low}}$.

![Graph showing habitat quality by reach at $Q_{\text{mid}}$.](image2)

Figure 49. Amount of available habitat by reach with the RMT-12 CHSI model at $Q_{\text{mid}}$. 
Microhabitat availability by reach does not provide an explanation for the preference exhibited by *O. mykiss* to spawn in Timbuctoo Bend, as there is far less high-quality habitat present there than in other reaches. Therefore, there may be additional factors influencing *O. mykiss* spawning site selection in the Timbuctoo Bend Reach besides the amount of high-quality microhabitat available, as determined by hydraulics and substrate. The Timbuctoo Bend Reach is one of the most upstream alluvial reaches of the lower Yuba River, and therefore may be preferred due to its upstream extent in the lower Yuba River. In the two years of redd surveys, there were no reds observed upstream of the Narrows reach in the Englebright Dam reach. While it is not possible to determine the amount of microhabitat available in the Narrows, there is no suitable habitat available in the Englebright reach, as 99.9% of the wetted area of the reach is in the lowest habitat quality class and is designated as non-habitat by the bioverified RMT model. This can primarily be attributed to the fact that Englebright Dam Reach is dominated by bedrock and large boulders, which is not suitable spawning substrate for *O. mykiss* to build redds in. Therefore, adult *O. mykiss* that migrate to the Englebright Dam Reach may then move back downstream until they arrive at the first area of suitable spawning habitat, which would be the Narrows Reach (which has not been surveyed for reds), followed by the Timbuctoo Bend Reach, which may be one factor in a relatively high preference for spawning in the Timbuctoo Bend Reach, despite a relatively small percentage of the reach providing suitable habitat as identified by the bioverified RMT model.

**4.4.3 Morphological Unit Analyses**

At the MU scale, MU characteristics’ influence on *O. mykiss* spawning site selection was evaluated by FR tests by MU, microhabitat availability within MUs, and size of preferred MUs.
The FR test was conducted for each of the three redds groups by MU to determine any potential preference for spawning in specific MUs. The results of the bootstrapping test by MU are shown in Table 15. The most conservative 95 % confidence interval thresholds from the $Q_{\text{low}}$ redd group were used to interpret the FR test results for each of the three redd groups, where FR > 1.73 indicate preference and FR < 0.27 indicate avoidance of the MU.

**Table 15. 95 % Confidence Interval Limits for MU FR Analysis**

<table>
<thead>
<tr>
<th>Threshold</th>
<th>$Q_{\text{low}}$</th>
<th>$Q_{\text{mid}}$</th>
<th>$Q_{\text{high}}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Avoidance</td>
<td>0.27</td>
<td>0.34</td>
<td>0.38</td>
</tr>
<tr>
<td>Preference</td>
<td>1.73</td>
<td>1.66</td>
<td>1.62</td>
</tr>
</tbody>
</table>

The preferred and avoided MUs changed with each of the three discharges tested (Figures 51-53). At $Q_{\text{low}}$, the only preferred MU was riffle transition. There were no MUs that were utilized for spawning at $Q_{\text{low}}$ that fell into the range of avoidance, and all other utilized MUs were tolerated at this discharge. At $Q_{\text{mid}}$, fast glide, riffle, and riffle transition were preferred, while slackwater was avoided. At $Q_{\text{high}}$, the preferred MUs were riffle transition, slow glide, and point bar, while run was the only avoided MU. The lateral bar, point bar, and swale MUs were not wetted at $Q_{\text{low}}$ and therefore were not available for spawning. The fact that these MUs were utilized at the higher discharges and within the tolerated range according to the FR test indicates that these units may provide good spawning habitat based on the hydraulic conditions available, but they may not be wetted for a sufficient duration during the spawning season to be utilized extensively.

![Figure 51. FR test by MU at $Q_{\text{low}}$.](image-url)
Table 16 shows key hydraulic and geomorphic variables extracted at the MU scale. The mean depth, velocity, and substrate $D_{\text{mean}}$ were extracted for MUs that were preferred and avoided at any one of the three discharges tested. MUs that were preferred at one discharge and not at another were included in Table 16 to determine if there are meaningful differences in the hydraulics of the MU at discharges where the MU was preferred, compared to discharges at
which it was not preferred. MUs highlighted in blue were preferred, those highlighted in red were avoided, and those without highlighting were tolerated.

Mean depth, mean velocity, and D_{mean} substrate size were correlated with percent of redds observed within each MU type and with FR values for each MU. These correlations were conducted for each of the three redd groups, as well as for the three groups combined. The only significant correlations found were with the Q_{mid} redd group, between FR and mean depth, and FR and mean velocity (Figures 54 and 55). The next section further analyzes the hydraulic and substrate conditions available in preferred MUs at each of the three discharges of interest.

Table 16. Spawning, hydraulic, and geomorphic metrics at the MU scale.

<table>
<thead>
<tr>
<th>MU</th>
<th>% of redds in MU</th>
<th>% of area in MU</th>
<th>Forage Ratio</th>
<th>Mean depth (m)</th>
<th>Mean velocity (m/s)</th>
<th>Substrate (mm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Q_{low}</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>fast glide</td>
<td>23.26</td>
<td>13.70</td>
<td>1.7</td>
<td>1.06</td>
<td>0.53</td>
<td>90.7</td>
</tr>
<tr>
<td>run</td>
<td>11.63</td>
<td>8.33</td>
<td>1.4</td>
<td>1.14</td>
<td>0.78</td>
<td>115.68</td>
</tr>
<tr>
<td>riffle</td>
<td>13.95</td>
<td>12.69</td>
<td>1.1</td>
<td>0.51</td>
<td>1.00</td>
<td>117.89</td>
</tr>
<tr>
<td>riffle transition</td>
<td>30.23</td>
<td>14.77</td>
<td>2.0</td>
<td>0.51</td>
<td>0.54</td>
<td>94.0</td>
</tr>
<tr>
<td>slackwater</td>
<td>6.98</td>
<td>15.71</td>
<td>0.4</td>
<td>0.43</td>
<td>0.10</td>
<td>91.5</td>
</tr>
<tr>
<td>slow glide</td>
<td>13.95</td>
<td>11.52</td>
<td>1.2</td>
<td>0.63</td>
<td>0.30</td>
<td>93.9</td>
</tr>
<tr>
<td>point bar</td>
<td>0.00</td>
<td>0.60</td>
<td>0.0</td>
<td>n/a</td>
<td>n/a</td>
<td>n/a</td>
</tr>
<tr>
<td>Q_{mid}</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>fast glide</td>
<td>25.93</td>
<td>13.42</td>
<td>1.9</td>
<td>1.12</td>
<td>0.57</td>
<td>90.69</td>
</tr>
<tr>
<td>run</td>
<td>12.96</td>
<td>8.16</td>
<td>1.6</td>
<td>1.19</td>
<td>0.83</td>
<td>115.68</td>
</tr>
<tr>
<td>riffle</td>
<td>22.22</td>
<td>12.43</td>
<td>1.8</td>
<td>0.56</td>
<td>1.04</td>
<td>117.89</td>
</tr>
<tr>
<td>riffle transition</td>
<td>25.93</td>
<td>14.47</td>
<td>1.8</td>
<td>0.56</td>
<td>0.58</td>
<td>94.03</td>
</tr>
<tr>
<td>slackwater</td>
<td>1.85</td>
<td>15.39</td>
<td>0.1</td>
<td>0.47</td>
<td>0.12</td>
<td>91.62</td>
</tr>
<tr>
<td>slow glide</td>
<td>7.41</td>
<td>11.28</td>
<td>0.7</td>
<td>0.68</td>
<td>0.34</td>
<td>93.86</td>
</tr>
<tr>
<td>point bar</td>
<td>0.00</td>
<td>0.92</td>
<td>0.0</td>
<td>0.06</td>
<td>0.16</td>
<td>94.72</td>
</tr>
<tr>
<td>Q_{high}</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>fast glide</td>
<td>8.51</td>
<td>12.76</td>
<td>0.7</td>
<td>1.24</td>
<td>0.65</td>
<td>90.69</td>
</tr>
<tr>
<td>run</td>
<td>2.13</td>
<td>7.77</td>
<td>0.3</td>
<td>1.32</td>
<td>0.93</td>
<td>115.68</td>
</tr>
<tr>
<td>riffle</td>
<td>9.57</td>
<td>11.83</td>
<td>0.8</td>
<td>0.67</td>
<td>1.14</td>
<td>117.89</td>
</tr>
<tr>
<td>riffle transition</td>
<td>25.53</td>
<td>13.77</td>
<td>1.9</td>
<td>0.68</td>
<td>0.67</td>
<td>94.03</td>
</tr>
<tr>
<td>slackwater</td>
<td>7.45</td>
<td>14.65</td>
<td>0.5</td>
<td>0.57</td>
<td>0.18</td>
<td>91.63</td>
</tr>
<tr>
<td>slow glide</td>
<td>35.11</td>
<td>10.73</td>
<td>3.3</td>
<td>0.80</td>
<td>0.42</td>
<td>93.86</td>
</tr>
<tr>
<td>point bar</td>
<td>3.19</td>
<td>1.85</td>
<td>1.7</td>
<td>0.11</td>
<td>0.21</td>
<td>100.22</td>
</tr>
</tbody>
</table>

*MUs highlighted in blue and red were preferred and avoided, respectively
Figure 54. MU scale correlation between mean depth (m) and FR value for the $Q_{\text{mid}}$ redd group.

Figure 55. MU scale correlation between mean velocity (m/s) and FR value for the $Q_{\text{mid}}$ redd group.

4.4.3.1 Two-way stratification: MU and microhabitat availability
There was little continuity in preferred MUs across the three discharges tested, possibly indicating that *O. mykiss* do not select spawning sites based on MU type, but rather by the physical habitat conditions available in the MU at the discharge experienced during spawning. It is valuable to investigate the hydraulic and substrate conditions available in the preferred MUs at each discharge to determine if they align with the suitable microhabitat conditions as defined by
the bioverified hydraulic and substrate HSCs. Table 17 presents the range of depth, velocity, and substrate $D_{\text{mean}}$ values that define suitable habitat based on the hydraulic RMT and S12 HSCs, including the lower limit of any habitat suitability (HSI>0), the lower limit of optimal suitability (HSI=1), the upper limit of optimal suitability, and the upper limit of any habitat suitability. The hydraulic and substrate conditions available in the MUs included in Table 16 at each discharge were then compared to the suitable hydraulic and substrate conditions from the bioverified RMT HSCs to see if the MU preferences were consistent with the preference for microhabitat conditions, indicating that the fish are shifting between MUs in order to utilize the preferred microhabitat. In addition to comparing the mean depth, velocity, and substrate values in each preferred MU at the three discharges, an analysis was also conducted to determine the percentage of MU area that was determined to be preferred microhabitat according to the RMT-12 CHSI physical habitat model, which includes areas with CHSI of 0.6 or greater. The availability of high-quality microhabitat within MUs is helpful in determining whether MUs preferred at one discharge had a greater percentage of high-quality microhabitat than at a discharge at which the MU was not preferred.

Table 17. Depth, velocity, and $D_{\text{mean}}$ ranges for suitable habitat from the RMT hydraulic HSCs and S12 substrate HSC.

<table>
<thead>
<tr>
<th></th>
<th>Low end of suitability range (HSI=0)</th>
<th>Lower value for ideal suitability (HSI=1)</th>
<th>Higher value for ideal suitability (HSI=1)</th>
<th>High end of suitability range (HSI=0)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Depth (m)</td>
<td>0.08</td>
<td>0.38</td>
<td>0.84</td>
<td>1.3</td>
</tr>
<tr>
<td>Velocity (m/s)</td>
<td>0.009</td>
<td>0.36</td>
<td>0.69</td>
<td>1.11</td>
</tr>
<tr>
<td>Mean substrate (mm)</td>
<td>32</td>
<td>32</td>
<td>90</td>
<td>200</td>
</tr>
</tbody>
</table>

Riffle transition was the only MU to be preferred at all three discharges tested. The mean values for depth and velocity for the riffle transition areas at each of the three discharges fell within the range of values with the optimal suitability from the RMT HSCs. The mean substrate size for riffle transition areas fell into the range of substrate sizes assigned a suitability value of 0.4 from the S12 substrate HSC. This supports the earlier impression that *O. mykiss* seem to select spawning locations based primarily on the hydraulic conditions present, with substrate size being of secondary importance. The majority of the area of riffle transition in the lower Yuba River was designated as preferred habitat from the RMT-12 model, with 76% at $Q_{\text{low}}$ and $Q_{\text{mid}}$ and 63% at $Q_{\text{high}}$.

Fast glide was preferred at $Q_{\text{mid}}$, but not at $Q_{\text{low}}$ or $Q_{\text{high}}$. Across all three discharges, the mean depth in fast glide is higher than the range of values that are assigned optimal suitability by the RMT depth HSC, but the mean velocity values fell into the highest suitability range from the velocity HSC. The mean substrate size at all three discharges fell within the range of moderate suitability on the S12 HSC. About 21% of fast glide areas were designated as preferred microhabitat using the RMT-12 model at $Q_{\text{mid}}$, which is a relatively low percentage for an MU that was preferred at this discharge. There was more available preferred habitat at $Q_{\text{low}}$ at 29%, and the FR value for fast glides at $Q_{\text{low}}$ was on the verge of preference, at 1.7. There was
significantly less preferred microhabitat in fast glides at $Q_{\text{high}}$, with only 10% of the area with CHSI of 0.6 or greater.

Riffle was another MU that was only preferred at $Q_{\text{mid}}$. In this instance, the mean depth value across all three discharges fell within the optimal suitability range from the RMT depth HSC, while the mean velocity values were faster than the ideal range defined by the RMT velocity HSC. The mean velocities in riffle for $Q_{\text{low}}$ and $Q_{\text{mid}}$ were still within the range of velocity values that were assigned suitability greater than zero. The mean velocity in riffles at $Q_{\text{high}}$ was higher than the maximum velocity assigned any habitat suitability by the RMT velocity HSC and therefore had a suitability of zero. The mean substrate values were not within the ideal $D_{\text{mean}}$ range from the S12 HSC, but they were within the range assigned a moderate suitability of 0.4. At $Q_{\text{mid}}$, 22% of riffle areas were defined as preferred microhabitat. A larger fraction of riffle areas were preferred habitat at $Q_{\text{low}}$, at 28%. Only 9% of riffles were preferred microhabitat at $Q_{\text{high}}$, most likely due to the elevated velocities present in riffles at this discharge.

Slackwater was one of the only MUs to be avoided, but was avoided only at $Q_{\text{mid}}$. While the mean depth value for slackwater at $Q_{\text{mid}}$ fell within the range of the highest depth suitability of the RMT depth HSC, the mean velocity value of 0.12 m/s was substantially lower than the ideal velocity range from the RMT velocity HSC of 0.36-0.69 m/s. The substrate $D_{\text{mean}}$ value of 91.62 mm was within the range of moderate suitability from the S12 SHSC. There was very little available preferred microhabitat in slackwater at all three discharges, with 3% at $Q_{\text{low}}$, 6% at $Q_{\text{mid}}$, and 14% at $Q_{\text{high}}$.

Run was the only other MU to be avoided at one of the three discharges, and it was avoided at $Q_{\text{high}}$ with an FR value of 0.3, and tolerated at $Q_{\text{low}}$ and $Q_{\text{mid}}$. The mean depth values at all three discharges were greater than the suitable range defined by the RMT HSC, however the mean depth in runs at $Q_{\text{high}}$ of 1.32 m was substantially higher than what would be considered suitable for *O. mykiss* spawning according to the RMT HSC. The mean velocity values at the three discharges were greater than the range of optimal suitability, but were within the range of some habitat suitability as determined from the RMT HSC. The average substrate $D_{\text{mean}}$ values in runs were within the range of moderate suitability from the S12 HSC. While the percentage of preferred microhabitat in runs was relatively low at all three discharges (21% at $Q_{\text{low}}$ and 12% at $Q_{\text{mid}}$), it was lowest at $Q_{\text{high}}$ where the MU was avoided, with only 1.3% of runs designated as preferred microhabitat at $Q_{\text{high}}$.

Slow glide was a preferred MU at $Q_{\text{high}}$, but not at the other two discharges. The mean depth in slow glide across all three discharges fell within the ideal range for high suitability. Only the mean velocity at $Q_{\text{high}}$ fell within the ideal range from the RMT velocity HSC. The mean velocity in slow glides at $Q_{\text{low}}$ and $Q_{\text{mid}}$ was lower than the range defined as having the highest suitability, indicating the velocity conditions in slow glide areas at $Q_{\text{low}}$ and $Q_{\text{mid}}$ were less suitable for *O. mykiss* spawning. The mean substrate values across all three discharges again fell within the moderate suitability range based on the S12 substrate HSC. Of the three discharges, there was the least amount of preferred microhabitat available in slow glides at $Q_{\text{low}}$ at 33%, compared to 39% and 40% of slow glides defined as preferred microhabitat at $Q_{\text{mid}}$ and $Q_{\text{high}}$, respectively.
Point bar was an MU that was not utilized at Q\text{low} or Q\text{mid} for spawning, as point bars were not often wetted at these discharges and therefore represented less than 1% of the wetted area of the lower Yuba River at the respective discharge. Interestingly, point bar was a preferred MU at Q\text{high}, even with only 7% of point bars designated as preferred microhabitat from the RMT-12 model. The mean depth and velocity values in point bars at Q\text{high} were below the range of optimal suitability, but were within the range of values where some suitability was defined by the RMT HSCs. The average D\text{mean} value in point bars fell within the range of 0.4 habitat suitability from the S12 HSC. Point bar seems to be a good example of the greater importance of suitable hydraulic conditions available within an MU at the discharge experienced by *O. mykiss* during spawning, rather than an explicit preference for specific MUs regardless of discharge.

As discharge increased, *O. mykiss* shifted to different MUs within the lower Yuba River in order to utilize preferred hydraulic conditions. In several instances, if an MU was preferred at one discharge, the mean velocity value in the MU at the other discharges would be generally slower than the velocity range of highest suitability, while the depths fell within the high suitability range. These results may support the assumption that, while depth is very important in spawning site selection, velocity is a driving force and must be within a specific ideal range in order for *O. mykiss* to successfully spawn there.

4.4.3.2 FR Test by Size of MU

Each MU type in the lower Yuba River is present in a wide range of sizes. Therefore MU size may influence *O. mykiss* spawning site selection. Of the MUs that were preferred at each discharge, the FR test was conducted to determine if *O. mykiss* exhibited significant preference or avoidance based on the areal size of the MU. The FR results based on size of MU are presented below, with the x-axis labels representing the upper bound of the MU area bin (Figures 56-58). The MUs that are included in the FR test for size preference and avoidance are only those that were preferred at the discharge of interest. The most conservative 95% confidence levels calculated using the bootstrapping test by MU were used to indicate preference and avoidance for this analysis, with a preference threshold of 1.73 and an avoidance threshold of 0.27.

At Q\text{low}, riffle transition was the only preferred MU, and preference was shown for riffle transitions of 93-465 m$^2$ (1,000-5,000 ft$^2$) and 465-929 m$^2$ (5,000-10,000 ft$^2$). Tolerance was shown for two other patch sizes, 0-93 m$^2$ (0-1,000 ft$^2$) and 1,858-2,787 m$^2$ (20,000-30,000 ft$^2$). Riffle transitions of 929-1,858 m$^2$, as well as all riffle transitions greater than 2,787 m$^2$, were avoided at Q\text{low}.  

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*O. mykiss* Adult Spawning

Physical Habitat in the Lower Yuba River

April 2014
Fast glide, riffle, and riffle transition were preferred MUs at $Q_{\text{mid}}$. Large fast glides of 6,503-7,432 m$^2$ (70,000-80,000 ft$^2$) and small fast glides of 93-465 m$^2$ (1,000-5,000 ft$^2$) were preferred sizes at $Q_{\text{mid}}$. In fact, the 93-465 m$^2$ (1,000-5,000 ft$^2$) size bin was preferred for all three of the preferred MUs at $Q_{\text{mid}}$. For riffles, there was also preference shown for the very large riffles of 8,361-9,290 m$^2$ (90,000-100,000 ft$^2$), and for riffle transitions of 465-929 m$^2$ (5,000-10,000 ft$^2$). All three of the preferred MUs at $Q_{\text{mid}}$ were avoided in the smallest patch size of 0-93 m$^2$ (0-1,000 ft$^2$), as well as most of the larger patch sizes above 465 m$^2$ (5000 ft$^2$).
Riffle transition, slow glide, and point bar were the preferred MUs at $Q_{\text{high}}$. Only very large slow glides were preferred, with the $5,574-6,503 \text{ m}^2$ ($60,000-70,000 \text{ ft}^2$) and $6,503-7,432 \text{ m}^2$ ($70,000-80,000 \text{ ft}^2$) sizes exhibiting preference to be utilized for *O. mykiss* spawning. All other size classes of slow glides were avoided, except for $93-465 \text{ m}^2$ ($1,000-5,000 \text{ ft}^2$), which was tolerated. There is again preference shown for small riffle transitions of $93-465 \text{ m}^2$ ($1,000-5,000 \text{ ft}^2$) at $Q_{\text{high}}$. Many sizes of riffle transitions were avoided, with several other patch sizes tolerated for spawning use. The only size point bar that was utilized was $465-929 \text{ m}^2$ ($5,000-10,000 \text{ ft}^2$) and was a preferred size patch. All other sizes of point bars were avoided and not utilized for spawning.

Across the three discharges tested, relatively small riffle transitions were consistently preferred. Of the MUs that were preferred at only one of the three discharges, both small and very large riffles, as well as small and fairly large fast glides, were preferred at $Q_{\text{mid}}$, and large slow glides were preferred at $Q_{\text{high}}$.

### 4.4.4 Topographic Change Process Analyses

At the scale of areas delineated by the various topographic change processes (TCP) occurring in the lower Yuba River, the influence of these processes on spawning site selection was evaluated by FR tests by area of topographic change and by topographic change process within preferred MUs.

#### 4.4.4.1 FR Test by TCP

The FR test on TCP was conducted for the three redd groups individually (*Figure 59*), as well as for all 261 observed redds (*Figure 60*). The wetted area at $113.3 \text{ m}^3/\text{s}$ ($4,000 \text{ cfs}$) was used for FR calculations for the group of all redds, as it is the discharge at which all redds are within the

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*O. mykiss* Adult Spawning

Physical Habitat in the Lower Yuba River

April 2014
wetted area. The results of the bootstrapping test by TCP for the three redd groups are shown in Table 18. The most conservative preference and avoidance thresholds from the \( Q_{low} \) redd group were used to interpret the results of the TCP FR test, yielding a preference threshold of 1.64 and an avoidance threshold of 0.36.

### Table 18. 95 % confidence intervals for the 3 redd groups by TCP FR test

<table>
<thead>
<tr>
<th>Threshold</th>
<th>( Q_{low} )</th>
<th>( Q_{mid} )</th>
<th>( Q_{high} )</th>
</tr>
</thead>
<tbody>
<tr>
<td>Avoidance</td>
<td>0.36</td>
<td>0.45</td>
<td>0.52</td>
</tr>
<tr>
<td>Preference</td>
<td>1.64</td>
<td>1.55</td>
<td>1.48</td>
</tr>
</tbody>
</table>

Across all four groups of redds, the only process for which preference was shown was downcutting. Downcutting was preferred in the \( Q_{high} \) redd group as well as the group of all redds, but was not preferred in the \( Q_{low} \) or \( Q_{mid} \) groups. Amongst the \( Q_{low} \), \( Q_{mid} \), and \( Q_{high} \) redd groups, redd observations occurred in only four TCPs: downcutting, noncohesive bank migration, in-channel fill, and areas of no detectable change. In-channel fill was the only utilized TCP that was avoided for spawning. Areas of noncohesive bank migration and no detectable change were tolerated for *O. mykiss* spawning.

![FR test results for TCP for the three redd groups](image)

**Figure 59. FR test results for TCP for the three redd groups**

When considering the group of all observed redds together, redd observations were made in eight TCPs. Downcutting was the only process which was preferred by *O. mykiss* spawners. Most of the processes, including avulsion, noncohesive bank migration, no detectable change, in-channel fill, and bar emergence, were tolerated. Island emergence and sub-avulsion were avoided. In general, the processes that were avoided were fill processes, while the only preferred process was a process of scour.
4.4.4.2 Two-way Stratification: FR Test on TCP by MU

The FR test was applied to a two-way stratification of TCP and MU to determine whether, within preferred MUs, there was preference or avoidance of specific TCPs. The MUs tested were fast glide, riffle, riffle transition, and slow glide, and the TCPs that were utilized for spawning within these MUs were downcutting, no detectable change, in-channel fill, and noncohesive bank migration.

There were no solid conclusions gained by this analysis, primarily because the percent area of each TCP within each MU was very small. The percent area of a specific TCP within a specific MU often represented less than 1% of the area attributed to the MU in the entire lower Yuba River, which falls under the constraint placed on the FR test to assign an FR value of 0 to domains representing less than 1% of the available area. Another limitation of this analysis was the very small sample sizes resulting from segmenting the redd groups by MU and by TCP. There were rarely more than 6-9 redd observations that occurred in a specific TCP-MU combination area, which is a very small sample size from which to draw conclusions about the combined influence of TCP and MU on *O. mykiss* spawning in the lower Yuba River. The combination of very small percent areas with very small sample sizes for percent redd occurrence resulted in FR values that were either 0 due to the minimum area constraint, or were very large (FR~10-50), again due to small percent areas that were generally around 1-3% of the whole area of an MU in the lower Yuba River. Overall, the two-way analysis of TCP and MU did not produce meaningful results regarding the influence of these parameters on *O. mykiss* spawning habitat.

4.4.5 Cover Element Analysis

The results of the analysis of the influence of cover elements on *O. mykiss* spawning habitat utilization did not show any relationship between redd location and proximity to cover. Of the
five types of cover considered—boulders, rip rap, streamwood, wood jams, and wood pieces—0-4.21 \% of the 261 redd observations were made within 10 m (32.8 ft) of the cover element. The vast majority of redd observations were located too far away from cover elements for them to have influenced the selection of spawning habitat by \textit{O. mykiss} in the lower Yuba River.

### 4.4.6 Mesohabitat Patch Analysis

In addition to microhabitat quality determined by the RMT-12 CHSI physical habitat model, the influence of mesohabitat patch size on \textit{O. mykiss} spawning site selection was evaluated using the FR test on the size of patches of homogeneous physical microhabitat conditions.

#### 4.4.6.1 FR Test by Mesohabitat Patch Size

The FR test to determine the influence of the areal size of a mesohabitat patches was done for the three redd groups, Q\textsubscript{low}, Q\textsubscript{mid}, and Q\textsubscript{high}. The test was conducted both on patches of only the highest habitat quality (CHSI>0.8) and on patches of preferred microhabitat (CHSI>0.6) from the RMT-12 physical habitat model. The most conservative preference and avoidance thresholds as calculated by the bootstrapping test on CHSI were used to interpret the mesohabitat patch size FR test results, with 1.53 as the preference threshold and 0.47 as the avoidance threshold.

Across the three redd groups, there was preference for a wide range of patch sizes of the highest quality microhabitat (Figure 61). For all three discharges tested, there was a preference for patches of 93-465 m$^2$ (1,000-5,000 ft$^2$). At Q\textsubscript{low}, there was also preference for sizes of 1,858-2,787 m$^2$ (20,000-30,000 ft$^2$) and 2,787-3,716 m$^2$ (30,000-40,000 ft$^2$). Preference was also shown at Q\textsubscript{mid} for patches of 1,858-2,787 m$^2$ (20,000-30,000 ft$^2$). In addition to the smaller size of 93-465 m$^2$ (1,000-5,000 ft$^2$), there was also preference for patches of 929-1,858 m$^2$ (10,000-20,000 ft$^2$) and patches of 4,645-5,574 m$^2$ (50,000-60,000 ft$^2$) at Q\textsubscript{high}. There was avoidance shown for a variety of patch sizes at each of the three discharges, due to a lack of redd observations in those patch sizes.

![Figure 61. FR test for highest quality microhabitat patch size](image-url)
The results for preference of mesohabitat patch size were quite different when considering patches of preferred microhabitat (0.6<CHSI<1) rather than patches of just the highest quality microhabitat, shown in Figure 62. Preference was shown for patches of 465-929 m² (5,000-10,000 ft²) for all three discharges. At Q_low, the only other preferred patch size was much larger at 12,077-13,006 m² (130,000-140,000 ft²). At Q_mid, preference was also shown for two other patch sizes that were quite large at 11,148-12,077 m² (120,000-130,000 ft²) and 12,077-13,006 m² (130,000-140,000 ft²). There was preference for a few various patch sizes at Q_high in addition to patches of 465-929 m² (5,000-10,000 ft²), including 93-465 m² (1,000-5,000 ft²), 2,787-3,716 m² (30,000-40,000 ft²), and 9,290-10,219 m² (100,000-110,000 ft²).

In general, the preferred patches of preferred microhabitat were of significantly larger sizes than the preferred patches of highest habitat quality. All of the preferred sizes based on patches of preferred microhabitat were less than 5,574 m² (60,000 ft²), while the preferred sizes of highest microhabitat quality patches ranged from 465 m² (5,000 ft²) to 13,006 m² (140,000 ft²).

4.4.7 Spatial Relationships of *O. mykiss* Redd Observations

The spatial relationships between *O. mykiss* reds were investigated to evaluate the influence of interactions with other *O. mykiss* individuals on spawning behavior and the propensity to spawn in clusters. The general location of spawning observations within the channel was also investigated relative to the water’s edge. The spatial relationships that were considered in this report were the distance of *O. mykiss* reds from areas of preferred microhabitat conditions, from other nearby reds, and from the wetted edge of the river at the discharge of interest.
4.4.7.1 Distance from Preferred Habitat

Using the RMT-12 physical habitat model, 46 and 67% of redds occurred in areas of preferred microhabitat (0.6 < CHSI < 1) for the three independent datasets. While a large fraction of redds were successfully predicted to be in areas of preferred microhabitat, this did leave a significant percentage of redd observations that were not located in preferred microhabitat. A test was conducted to determine the percentage of redd observations that were located within certain small distances of areas of preferred microhabitat, as defined from the RMT-12 physical habitat model. Results of this analysis show that a significant number of redds are located on the fringe, just outside model-predicted preferred habitat (Table 19). With a buffer of 0.91 m (3 ft), 52, 78, and 81% of redds were observed either within preferred habitat or within 0.91m (3 ft) of preferred microhabitat for the three datasets. With a buffer of 1.52 m (5 ft), these percentages increased to 55, 81, and 88% of redds located within preferred habitat or within 1.52 m (5 ft) of preferred microhabitat.

Table 19. Summary of results for distance to preferred habitat

<table>
<thead>
<tr>
<th>Redd Group</th>
<th># redds</th>
<th>% in preferred microhabitat</th>
<th>% within 3 ft of preferred microhabitat</th>
<th>% within 5 ft of preferred microhabitat</th>
<th>% beyond 5 ft of preferred microhabitat</th>
</tr>
</thead>
<tbody>
<tr>
<td>Q_low</td>
<td>43</td>
<td>67%</td>
<td>81%</td>
<td>88%</td>
<td>12%</td>
</tr>
<tr>
<td>Q_mid</td>
<td>54</td>
<td>67%</td>
<td>78%</td>
<td>81%</td>
<td>19%</td>
</tr>
<tr>
<td>Q_high</td>
<td>94</td>
<td>46%</td>
<td>52%</td>
<td>55%</td>
<td>45%</td>
</tr>
</tbody>
</table>

This analysis shows that for the 33-54% of redds that were not located in areas of preferred microhabitat, 18, 44, and 64% of these redds were located very near areas of preferred habitat. This could be indicative of *O. mykiss* adults near high-quality habitat areas, but not within the areas sharply delineated by the model as high-quality habitat, or the spawning adults could be influenced by a group mentality to spawn near other spawning individuals.

4.4.7.2 Distance from Nearest Redd

To test the propensity of individuals to spawn near other spawners of the same species, an analysis was conducted to determine the influence of the distance from one redd to its nearest neighboring redd. *O. mykiss* redds observations were separated into two groups based on the spawning season in which they occurred, with 223 redds observed in the first spawning season from January through April of 2010, and 38 redds observed in the second spawning season from January through April 2011. The redd observations were separated by year, as only redds within the same spawning season could possibly influence the location of redd building by other *O. mykiss* females. The percentage of redds within a specified distance of another redd was calculated for a range of distances from < 1 m to > 1,000 m (< 3.2 ft to > 3,280 ft) from the nearest neighboring redd (Figure 63).

A relatively small percentage of redds were located within 1 m (3.2 ft) of another redd, with 11% of 2010 observations and 0% of 2011 observations within 1 m (3.2 ft). Redd superimposition occurs when late spawning female individuals dig redds on top of existing redds where the eggs are still developing, and can result from limited availability of suitable spawning gravels (Hayes
The average size of observed *O. mykiss* redds in the lower Yuba River was ~1 m² (0.96 m², 10.3 ft² in 2010 and 1.05 m², 11.3 ft² in 2011), so any redds built within 1 m (3.2 ft) of another redd have the potential to be superimposed and possibly cause egg mortality.

The majority of redd observations in both years occurred between 1 and 5 m (3.2 and 16.4 ft) of another redd, with 65 % and 50 % of redd observations in 2010 and 2011, respectively. This proximity could indicate a tendency of *O. mykiss* females to cluster and dig a redd near the redds of other spawners, but a distance of 5 m (16.4 ft) is large enough to prevent the deleterious effects of superimposition from spawning too close together.

![Figure 63. Percent of a. 2010 redd observations and b. 2011 redd observations within a specified distance of the nearest neighboring redd.](image)

The influence of proximity to other redds most likely does not impact spawning locations at distances greater than 5 m (16.4 ft), as individual fish are likely not influenced by spawning activity of other individuals at this distance. Ten percent of redds observed in 2010 and 18 % of redds observed in 2011 were located within a distance of 5 to 10 m (16.4 to 32.8 ft) of another redd. The same percentage of redds in 2010 and 2011 were located between 10 and 50 m (32.8 to 164.0 ft) of another redd, with 5 % of observations from each year. Similar percentages of observations were also measured between 50 m and 500 m (164.0 to 1,640 ft) of another redd, with 6 % in 2010 and 8 % in 2011. The remainder of redds were at a distance of > 500 m (> 1,640 ft) from the nearest redd, with 2 % of 2010 observations within 1,000 m (3,280 ft), 19 % of observations in 2011 within 1,000 m (3,280 ft) and 3 % of observations more than 1,000 m (3,280 ft) from their nearest neighboring redd. Considering the substantially smaller sample size of 2011 observations (38 redds in 2011 vs. 223 redds in 2010), it is reasonable that more of the
2011 observations are located at further distances from each other and are more spread out throughout the lower Yuba River.

### 4.4.7.3 Distance from Wetted Edge

In order to further investigate the spatial patterning of spawning habitat utilization at the micro-scale, the FR test was conducted on the location of redds relative to the wetted edge. The average wetted width of the channel and the average distance to the center of the channel at $Q_{\text{low}}$, $Q_{\text{mid}}$, and $Q_{\text{high}}$ is presented in Table 20. For the three discharges tested, the center of the channel is about 30 m from the wetted edge on average.

**Table 20. Average channel width and center of channel at $Q_{\text{low}}$, $Q_{\text{mid}}$, and $Q_{\text{high}}$.**

<table>
<thead>
<tr>
<th>Discharge</th>
<th>Average wetted width of the channel (m)</th>
<th>Average distance to the center of the channel (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>$Q_{\text{low}}$ (24.9 m$^3$/s)</td>
<td>61.44</td>
<td>30.72</td>
</tr>
<tr>
<td>$Q_{\text{mid}}$ (28.3 m$^3$/s)</td>
<td>62.75</td>
<td>31.38</td>
</tr>
<tr>
<td>$Q_{\text{high}}$ (36.8 m$^3$/s)</td>
<td>66</td>
<td>33</td>
</tr>
</tbody>
</table>

The results of the bootstrapping test as conducted on the distance from the wetted edge are shown in Table 21. The 95% confidence intervals were applied as the preference and avoidance thresholds to interpret the results of the FR tests for the three redd groups based on the distance of the redds to the water’s edge at each respective discharge.

**Table 21. 95% confidence intervals for the 3 redd groups by distance from wetted edge FR test**

<table>
<thead>
<tr>
<th>Threshold</th>
<th>$Q_{\text{low}}$</th>
<th>$Q_{\text{mid}}$</th>
<th>$Q_{\text{high}}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Avoidance</td>
<td>0.40</td>
<td>0.29</td>
<td>0.49</td>
</tr>
<tr>
<td>Preference</td>
<td>1.60</td>
<td>1.71</td>
<td>1.51</td>
</tr>
</tbody>
</table>

The results of the FR test show that, while the vast majority of redds were located from 1 to 15 m from the wetted edge, there was little statistical preference shown for any of these distances from the edge of the water (Figures 64-66). At $Q_{\text{low}}$ (24.9 m$^3$/s, 880 cfs), there was no preference shown for any of the distances from the wetted edge. There were redd observations from the $Q_{\text{low}}$ redd group at distances from 1-5, 5-10, 10-15, and 15-20 m from the wetted edge, but the FR values for each of these intervals were between 0.4 and 1.6, indicating tolerance of these intervals for *O. mykiss* spawning. Most of the redds in the $Q_{\text{low}}$ group were located closer to the wetted edge than to the center of the channel, as all but 4 of 43 redds were located within 15 m of the wetted edge, about half the distance to the center of the channel, which was 30.72 m from the water’s edge on average.
Redds in the Q_{mid} redd group were observed at distances from 1-40 m from the wetted edge. There was an avoidance for spawning very near the wetted edge, with no redds observed within 1 m of the edge and tolerance of the area 1-5 m from the wetted edge, with 10 % (5 of 54) of redds observed between 1-5 m from the edge, giving a FR value of 0.46 for the interval. There was no preference shown for any of the distance intervals within the Q_{mid} group given the high preference interval of 1.71. With an average distance to the center of the channel of 31.38 m at Q_{mid} (28.3 m³/s, 1,000 cfs), 75.9 % of redds were located within 15 m of the wetted edge, about halfway into the middle of the channel on average. There were very few redd observations from the Q_{mid} redd group that were further than 20 m from the bank, with 1 redd each occurring in the intervals of 20-30 m and 30-40 m from the wetted edge.
There were only a few redd observations in the lower Yuba River of the three red groups within 1 m of the wetted edge, and they were all in the $Q_{high}$ redd group. The area directly adjacent to the wetted edge up to 1 m towards the center of the channel had 3.19% of the redd observations (3 of 94) from the $Q_{high}$ group, and tolerance was shown for the interval. There were also redd observations at distances from 1-50 m from the wetted edge. There was a single redd observation that was within the interval 40-50 m from the wetted bank, however, only 0.63% of the wetted area at $Q_{high}$ is 40-50 m from the wetted edge, so the FR is defined as zero due to the minimum percent area constraint applied to FR test results in this study. The vast majority of redds in the $Q_{high}$ redd group were observed from 1-20 m from the wetted edge (94% of observations in the $Q_{high}$ group). There was a tolerance shown for spawning for distances of 1-5 m from the wetted edge, as well as for distances of 10-15 m (FR=1.45), 15-20 m (FR=0.62), and 20-30 m (FR=0.51). A preference was shown within the $Q_{high}$ redd group for spawning at a distance of 5-10 m from the wetted edge. Given the average distance to the center of the channel at $Q_{high}$ was 33 m, there was preference shown to spawn much closer to the wetted edge than to the center of the channel, with 84% of $Q_{high}$ redd observations located within 15 m of the wetted edge.

Figure 65. FR test results for redd location relative to distance from the wetted edge for the $Q_{mid}$ redd group.
Across the three redd groups, the only preferred distance interval was 5-10 m from the wetted edge and the vast majority of redds within each of the groups were located within half of the distance to the center of the channel, indicating a tendency to spawn nearer to the wetted edge. The average CHSI from the RMT-12 physical habitat model was determined for the same ranges of distance from the wetted edge that were used for the FR test to determine if the microhabitat conditions are of higher quality in the areas that were more utilized by spawning *O. mykiss*. The 5-10 m range at *Q*<sub>high</sub> had a low average CHSI of 0.29, which was lower than several of the other distance ranges at that discharge. At all three discharges, the 0-1 m range had the lowest average CHSI values around 0.03. The areas from 1 m to 15 m from the wetted edge, which contained 76-91% of the redd observations, had relatively low average CHSI values ranging from 0.24 to 0.32. The areas with the highest average CHSI values were the furthest from the wetted edge (40-50 m) for each of the discharges tested. The quality of microhabitat conditions cannot explain the higher utilization of areas closer to the wetted edge. Like other stratified analyses, it is difficult to know if distance from the wetted edge is the governing variable, or if it in turn is a proxy for a more direct variable.

### 5 DISCUSSION AND CONCLUSIONS

This research was based on the innovative approach of near-census river science to evaluate *O. mykiss* spawning at various spatial scales, ranging from the microhabitat scale to the river segment scale. This provided the opportunity to develop a comprehensive understanding of *O. mykiss* spawning habitat utilization and preferences in the lower Yuba River, as well as the spatial patterns in spawning habitat availability and use. There was strong stratification of *O. mykiss* redd occurrence for all represented types of physical habitat. Overall, *O. mykiss* spawning behavior was predictable and required a holistic blend of hydraulic and geomorphic representations to explain.
5.1 Physical Habitat Model Bioverification

This study of *O. mykiss* spawning habitat used a traditional approach of physical habitat modeling through the assessment of habitat suitability with HSCs, but applied this approach to near-census datasets and 2D hydrodynamic model outputs. The resulting physical habitat model is spatially-explicit, high resolution, and predictive of habitat quality and quantity across a range of discharges up to bankfull. Near-census physical habitat models make it possible to accurately quantify the amount of spawning habitat available in the lower Yuba River without the experimental design sacrifices and trade-offs required of a sampling methodology, which is beneficial in developing habitat-discharge relationships.

The RMT hydraulic HSCs provided the best physical habitat of the three HHSI models tested based on hydraulic conditions. The hydraulic conditions that are most suitable according to the RMT HSC for mean column velocities range from 0.36-0.69 m/s (1.18-2.25 ft/s), and depths of 0.38-0.84 m (1.25-2.76 ft). The RMT velocity HSC has a moderate range of highly suitable velocities, while the USFWS HSC defined a much more narrow range and TRTAC defined a slightly broader range of highly suitable velocities. The RMT depth HSC varied greatly from the other two depth HSCs, which assigned a high suitability to depths much greater than the RMT curve. In general, the TRTAC hydraulic HSCs assigned the highly suitable ranges of depth and velocity too broadly, resulting in larger areas of the river categorized as high-quality habitat and a physical habitat model that was not sufficiently restrictive to predict preference for *O. mykiss* spawning habitat utilization. The USFWS depth and velocity HSCs were skewed to assign high suitability to greater depths and velocities than the successfully bioverified RMT model. The USFWS HSCs incorrectly defined the suitability of slightly shallower depths and slightly slower velocities, causing the model to not perform well and ultimately fail bioverification tests.

The RMT hydraulic HSCs and the S12 substrate HSC combined to create the most successful CHSI physical habitat model. The S12 HSC assigned the highest suitability to the substrates of medium gravel/small cobble with mean diameter sizes of 32-90 mm, and a moderate suitability value of 0.4 to cobble-sized substrate with mean diameters of 90-200 mm. The RMT-9 and RMT-12 models were both bioverified, however the RMT-12 performed slightly better and provided a more restrictive and precise model. The difference between these two models was small, but meaningful. The S9 curve assigned high suitability to all substrate with a mean diameter of 32-200 mm, while S12 restricted the high suitability range to 32-90 mm, with a lower suitability assigned to sizes of 90-200 mm. This lower suitability for the cobble class size better represented the substrate preferences of spawning *O. mykiss* for smaller mean diameter substrate.

Incorporating the metric of substrate suitability to the physical habitat model improved upon the model based on hydraulics alone, as evidenced by the generally higher preference FR values resulting from the RMT-12 CHSI model over the RMT HHSI model. However, the improvement was relatively small. Substrate suitability seems to be of secondary importance to hydraulics on the lower Yuba River, since adding the metric of substrate suitability could not fundamentally improve the USFWS and TRTAC HHSI models to a point where they would be considered acceptable physical habitat models. Hydraulic conditions are of primary importance in determining the habitat *O. mykiss* will utilize and prefer for spawning, and it appears that they
will tolerate suboptimal substrate in order to utilize areas of optimal depth and velocity. Water depth and velocity are fundamentally important factors in predicting where *O. mykiss* will spawn, and through the MU analysis it was determined that water velocity may be the single most important factor, because velocity values must be within an ideal range for *O. mykiss* to successfully spawn, regardless of depth.

The testing of several types of HSCs was a novel aspect of this study, and ultimately the utilization-based hydraulic HSCs that were site-specific to the lower Yuba River provided the best physical habitat model, along with a substrate HSC that was theoretically-based. The physical habitat models that were developed from these HSCs were found to bioverify at three discharges, showing that for the lower Yuba River the microhabitat conditions that are preferred for *O. mykiss* spawning are consistent throughout the range of discharges tested in this study. The predictions were slightly less successful at 1,300 cfs compared to the two lower discharges, so it would be beneficial to be able to test the physical habitat models against spawning observations across a wider range of discharges than was possible with the data used in this study to determine if the hydraulic and substrate conditions preferred by *O. mykiss* spawners are not discharge-dependent.

One of the most innovative methodological aspects of this study was the application of statistical bootstrapping tests to provide a measure of statistical significance to the FR test results. The addition of the bootstrapping test in conjunction with the FR test provides a 95 % confidence interval around FR=1 for the preferences and avoidances interpreted from the test, rather than depending on the simple FR >1 for preferred domains and <1 for avoided domains. This added level of statistical significance means the FR test results for even a relatively small dataset can provide meaningful outcomes in regards to understanding the physical habitat preferences of spawning *O. mykiss*.

This study provided a comprehensive and quantitative understanding of the physical habitat conditions that are preferred for *O. mykiss* spawning. The RMT-12 physical habitat model predicted 67 %, 67 %, and 46 % of redd observations from the Q_low, Q_mid, and Q_high groups as being located within areas of preferred microhabitat conditions based on water depth, velocity, and mean substrate diameter. This leaves 33-54 % of individual *O. mykiss* spawners whose spawning location within ~1m could not be predicted accurately based on microhabitat suitability of hydraulics and substrate size. For those redds that were not located in areas of high-quality microhabitat, many of them were located within 0.91-1.52 m (3-5 ft) of high-quality habitat. With the addition of a 1.52 m buffer around areas of high-quality microhabitat, 55-88 % of redds across the three redd groups were accounted for as being located within or very near high-quality habitat. The spawning activity near areas designated as preferred habitat may still be influenced by the suitable microhabitat conditions, but their location just outside of the preferred areas may result from the social behavior of the fish to cluster around other spawners, or from small errors in the delineation of the habitat quality class patches due to the propagation of uncertainty from the hydrodynamic model results and the substrate mapping efforts into the CHSI model values. Model performance at the meter scale was also potentially impacted by localized topographic changes during the time between river mapping of Timbuctoo Bend—where most adults spawned—in 2006 and the time span of the spawning surveys in 2010-2011. It may also have been impacted by winter flow fluctuations in the week between surveys. For the 12-45
% of adults defying model predictions, these individuals may be selecting spawning locations based on other habitat characteristics beyond the microhabitat scale, or that we are not measuring or are difficult to model. There is also the chance that these individuals are simply selecting spawning sites based on random choices. Either way, the outcome is diverse life histories that may promote diversity and resilience in the population. Because the lower Yuba River has millions of ft² of diverse and dynamic in-channel bed areas, there is plenty of opportunity for individuals to chart their own course, but most importantly there is plenty of unoccupied preferred habitat that is not used in any particular year. In other words, the river has an overabundance of both preferred habitat and diverse features to provide for the full range of *O. mykiss* spawning behavior.

### 5.2 Habitat-Discharge Relationship

There was significant agreement amongst the WUA-discharge relationships calculated at the three spatial scales, with discharges of 15-20 m³/s (530-700 cfs) providing the highest amount of available spawning habitat. The amount of available habitat rapidly decreased after reaching the peak availability, with discharges greater than 20 m³/s (700 cfs) providing drastically reduced availability of spawning habitat. For discharges of about 85 m³/s (3,000 cfs) and greater, the WUA curves stabilized to a minimum amount of relative habitat availability. There is no lack of high-quality spawning habitat for *O. mykiss* in the lower Yuba River, with millions of square feet of preferred habitat—the river currently sustains a population of several hundred *O. mykiss* spawners each year, but has abundant spawning habitat capable of sustaining thousands of spawners.

The WUA-discharge relationships calculated for *O. mykiss* spawning provide a tool to use for the optimization of flows during the spawning season to provide the most habitat availability of the highest quality, though it is evident that all flows provide far more habitat than the current population size can use.

In the lower Yuba River, the optimal flows to provide the most habitat for spawning *O. mykiss* and those for spawning Chinook salmon are very similar, with a peak discharge from the Chinook salmon spawning WUA-discharge relationship at about 17 m³/s (600 cfs) (Pasternack et al. 2013).

### 5.3 Physical-Biological Linkages

The investigation of the influence of larger scale physical and geomorphic conditions and the spatial relationships of redds on *O. mykiss* spawning habitat utilization revealed interesting linkages in spawning habitat preferences beyond point-scale microhabitat quality.

At the segment scale, it was shown that there is non-random organization of redd locations and a tendency for spawning in the more upstream section of the lower Yuba River. The preference for substrate size was within the range of 32-90 mm mean diameter, which was classified as the medium gravel/small cobble class. In investigating the 32-90 mm size preference by smaller size intervals of 10 mm, there was a strong preference for substrate of 50-60 mm mean diameter. This preference matched the range of substrate size that was expected to be used for *O. mykiss* spawning when developing the substrate visual classification system, and was also in agreement
Physical Habitat in the Lower Yuba River

with the estimate of suitable bed material of 20-76 mm resulting from the 10% of fish body length estimate presented in Kondolf and Wolman (1993).

At the reach scale, there was a marked preference for the two most upstream alluvial reaches of the lower Yuba River (out of 8 total reaches), with 92% of all observed redds located in Timbuctoo Bend and Parks Bar. It is interesting, however, that there is not a large amount of high-quality microhabitat available in these two reaches, with only 2-10% of the wetted area designated as the highest microhabitat quality. This finding matches the general understanding of the nature of salmonids to migrate upstream to spawn, and shows that they may forgo reaches with overly abundant high-quality habitat for the instinct to move further upstream. Whereas longitudinal Chinook spawning locations are highly dependent on water temperature as cool water extends downstream from Englebright Dam throughout their fall spawning period, *O. mykiss* spawning site selection is not limited by water temperature because of their winter spawning period.

The influence of MU, size of MU, TCP, and the combined influence of TCP and MU was investigated and yielded some interesting results. There has long been a perception that *O. mykiss* and other salmonids prefer to spawn primarily in shallow, swift water such as the conditions present in riffles (Briggs 1953 and others). The preferred MUs for *O. mykiss* spawning, however, varied substantially with discharge. Riffle transition was the only MU preferred across discharges of 24.9 - 36.8 m³/s (880 - 1,300 cfs), and riffles were only preferred at Qₘᵢₙ (28.3 m³/s, 1,000 cfs). However, while the preferred MUs changed with discharge, the hydraulic conditions available at each discharge within the preferred MUs were well aligned with the hydraulic conditions assigned high suitability by the RMT hydraulic HSC that were found to be preferred microhabitat. The different MU preference with discharge may demonstrate that *O. mykiss* shift spawning to different MUs in order to utilize their preferred hydraulic conditions. As for the preferred size of MU, relatively smaller riffle transitions of 93-929 m² (1,000-10,000 ft²) were preferred across all three discharges tested compared to the preferred sizes of other MUs. The size preferences for other MUs were more varied, with both small and large riffles and fast glides preferred at Qₘᵢₙ, and very large slow glides preferred at Qₜₕᵢₙ.

The only TCP for which preference was shown was downcutting, which indicated areas of the channel that had undergone vertical scour between 1999 and 2009. This refutes the previous assumption that, evolutionarily, *O. mykiss* and other salmonids should avoid actively eroding or scouring areas in order to better insure the survival of incubating eggs. While Bigelow (2003) found that salmon do not appear to avoid building redds in areas susceptible to scour, this study showed that *O. mykiss* actually prefer areas that had experienced scour over the previous decade. In fact, the TCPs that were most utilized by *O. mykiss* for spawning were areas of scour, while the only TCPs that were utilized and also avoided were those that had experienced fill processes. This finding was similar to the preferences of Chinook spawners in the lower Yuba River, who strongly preferred areas subjected to scour (Pasternack et al. 2013).

There were interesting results regarding the influence on spawning site selection of various spatial relationships at the meso-scale, beyond actual microhabitat quality discussed above. While there was significant variation in the preferred patch size of highest microhabitat quality
and of preferred microhabitat across the three discharges, there was some consistency. Patches of highest quality habitat of 93-465 m$^2$ (1,000-5,000 ft$^2$) were preferred at the three discharges, while slightly larger patches of preferred microhabitat of 465-929 m$^2$ (5,000-10,000 ft$^2$) were consistently preferred with increasing discharge. This seems logical, considering the areas of highest habitat quality were the most restricted and generally smaller, so the preferred patches were most likely in the same areas as the preferred highest quality patches, just enlarged to encompass the entire surrounding area of preferred habitat, with CHSI ranging from 0.6-1.

The analysis of the distance from one redd to its nearest neighboring redd revealed that clustering of *O. mykiss* spawning activity was common, with 50-65 % of redds in the two years of observations located within 1-5 m of another redd. While there seems to be a tendency of individuals to spawn near other spawners, the fish do not seem to be limited by available spawning gravel or hydraulic habitat, otherwise they would be observed to spawn even closer together and perhaps superimpose their redds if they were not encountering abundant levels of suitable habitat. There was also a tendency for redds to be located nearer to the wetted edge of the river than to the center of the channel, with 75-90 % of redd observations amongst the three redd groups occurring within half the distance to the center of the channel. The proximity to the river bank could not be explained by the quality of microhabitat available in those areas. While it is unknown if the distance from the wetted edge is a governing variable in spawning site selection, this analysis does provide an interesting spatial understanding of *O. mykiss* spawning sites nonetheless.

### 5.4 Management Implications

The physical habitat models developed in this study are predictive models that are very beneficial as river management tools. The RMT-12 models of microhabitat quality were created for a wide range of discharges from less than baseflow to bankfull discharges in the lower Yuba River (300-5,000 cfs), and were used to determine the discharge that provides the most high-quality microhabitat for *O. mykiss* spawning. Physical habitat models of this type can be developed for any species and lifestage of interest, and can be combined to develop flow regimes that optimize the discharge in regulated rivers to maximize the availability of suitable physical habitat conditions for threatened or endangered species throughout the year. The near-census, spatially-explicit physical habitat models developed in this study can make the optimization of the instream flows for several species and lifestages of interest an even more useful tool for river managers.

The bioverified physical habitat models for *O. mykiss* spawning in the lower Yuba River provide a means by which to assess the effectiveness of the Yuba Accord flow schedules in providing spawning habitat for threatened *O. mykiss* and keeping the population in good condition. This study determined that there are millions of ft$^2$ of preferred microhabitat available to spawning *O. mykiss*, and the lower Yuba River is capable of sustaining a much larger *O. mykiss* population than is currently present.
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