# Evaluation of the 2008–2010 Sailor Bar Gravel Placements on the Lower American River, California

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***2010–2011 Data Report***

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**Water Forum**

**&**

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## EXECUTIVE SUMMARY

In September 2010, the United States Bureau of Reclamation, in coordination with the United States Fish and Wildlife Service and the Sacramento Water Forum, continued salmonid habitat improvement in the lower American River (LAR), California, by strategically placing ~11,688 cubic yards (~16,200 tons) of gravel and cobble at Sailor Bar to enhance spawning habitat for Chinook salmon and steelhead trout. This project followed an estimated 7,000 cubic yard placement adjacent to the Nimbus Fish Hatchery in fall 2009, ~1.3 miles (2.09 km) upstream, and another 5,000 cubic yard placement just upstream of that site completed in October 2008. For the 2010 project, placed material was derived from a floodplain source adjacent to the augmentation site (north side of river). Roughly 7,720 cubic yards of material, composed of 8–178 mm gravel sizes, with 95% of material from 8–78 mm ($D_{50}$~30 mm) was placed as spawning material. Additionally, the 2010 augmentation site contained a constructed cobble island (8–178 mm; 95% from 8–125 mm; $D_{50}$~73 mm) and "scallops" in the substrate designed to add habitat heterogeneity to the main channel and rearing habitat for juvenile Chinook salmon and steelhead trout. Further, an additional ~5,500 tons of cleaned cobble was placed downstream of the 2010 augmentation site. The specific purpose of this placement was to divert flow into an adjacent, perched side-channel, thereby preventing the de-watering of salmonid redds in a historically important spawning and rearing area during low-flow conditions. For this reason, the side-channel adjacent to the 2010 gravel placement is treated as an augmentation site throughout this report.

We evaluated the ongoing effects of this project on several physical and biological parameters through July 2011. Mean daily flow ranged from 753–31,425 $ft^3 \bullet s^{-1}$ (21–890 $m^3 \bullet s^{-1}$) for the study period (1 May 2008 through 31 July 2011). The relatively high flows associated with monitoring after mid-December 2010 affected fish behavior and the ability to monitor that behavior. Regardless, important information was gleaned from the sampling period. Key results were as follows:

- Peak flows in an example "wet" year (1999) largely corresponded to increased Chinook salmon rearing habitat requirements from 1 January–1 March. In contrast, peak flows in an example "critical" year (1994) largely occurred after peak rearing habitat requirements (i.e., after 1 March). When flow-inundation curves were applied to flow data from example "critical" and "wet" years, total additional inundated areas ranged from 0–3,289 $m^2$ (0–2,533 $m^2$ floodplain adjacent to the 2008 and 2009 sites and 0–756 $m^2$ side-channel island adjacent to the 2010 site) in the "critical" year and 0–29,384 $m^2$ (0–21,593 $m^2$ floodplain adjacent to the 2008 and 2009 sites and 0–7,755 $m^2$ side-channel island adjacent to the 2010 site) in the "wet" year. Assuming an average HSI value of 0.50, "wet" (1999) outmigration parameters, and the AFRP adult production target (39,840,000 juveniles), these values represent an average of <1.0% (range = 0–2.2%) of the total daily habitat requirements in a "critical" flow year (1994 data) and an average of 2.4% (range = 0–26.8%) of the total daily habitat requirements in a "wet" flow year (1999 data).

- In all, 530 Chinook salmon redds were observed in the LAR for the 2010–2011 spawning season. This was ~24% of the past seven-year average. A total of 175 redds were observed at the 2008 augmentation site (~33%), 34 redds were observed at the 2009 augmentation site (~6%), 0 redds were observed at the 2010 augmentation site (0%), and

42 redds were observed in the side-channel adjacent to the 2010 augmentation site (~8%). This indicates a significant increase in utilization of the 2008 site, 2009 site, and the side-channel adjacent to the 2010 site compared to pre-enhancement conditions, but a significant decrease in utilization of the 2010 site.

- In all, 92 steelhead trout redds were observed in the LAR for the 2010–2011 spawning season. This was ~63% of the past 10-year average. A total of 37 redds were observed at the 2008 augmentation site (~40%), 0 redds were observed at the 2009 augmentation site (0%), 1 redd was observed at the 2010 augmentation site (~1%), and 9 redds were observed in the side-channel adjacent to the 2010 augmentation site (~10%). This indicates a significant increase in utilization of the 2008 site compared to pre-enhancement conditions, but not the 2009 site, 2010 site, or the side-channel adjacent to the 2010 site.

- Average depths selected for spawning were 0.50 m (range = 0.18–0.85 m) for Chinook salmon and 0.69 m (range = 0.11–1.70 m) for steelhead trout, whereas average velocities selected for spawning were 0.77 $m \cdot s^{-1}$ (range = 0.19–2.00 $m \cdot s^{-1}$) for Chinook salmon and 0.87 $m \cdot s^{-1}$ (range = 0.32–2.00 $m \cdot s^{-1}$) for steelhead trout. The average depth selected by spawning Chinook salmon was significantly shallower than the average depth selected by spawning steelhead trout. However, this significant difference may be related to both sampling location and gear biases. Chinook salmon were only sampled within augmentation sites with a limited range of depths using a 1.0 m top-setting rod, whereas steelhead trout were sampled throughout the entire LAR using a 2.0 m top-setting rod. No significant difference was observed for average velocities.

- Average substrate sizes selected for spawning were 30.0 mm (range = 12.5–99.8 mm) $D_{50}$ and 60.1 mm (range = 27.4–148.0 mm) $D_{85}$ for Chinook salmon and 28.8 mm (range = 24.5–56.1 mm) $D_{50}$ and 58.1 mm (range = 53.2–106.4 mm) $D_{85}$ for steelhead trout. Peak HSI values ranged from 10–50 mm $D_{50}$ and 40–90 mm $D_{85}$ for Chinook salmon and 10–40 mm $D_{50}$ and 40–70 mm $D_{85}$ for steelhead trout. No significant differences were observed when average substrate sizes were compared between species. The average $D_{50}$ substrate size selected by Chinook salmon was 1.25-fold larger than the $D_{50}$ of the 2008 site, 1.17-fold smaller than that of the 2009 site, and equal to that of the 2010 site, whereas the average $D_{50}$ substrate size selected by steelhead trout was 1.20-fold larger than the $D_{50}$ of the 2008 site, 1.22-fold smaller than that of the 2009 site, and 1.04-fold smaller than that of the 2010 site. The range of substrate sizes selected by Chinook salmon was ~2.3 to 2.8-fold ($D_{85}$ to $D_{50}$) wider than the range of substrate sizes selected by steelhead trout; possibly due to a wider range of fork lengths (FLs) observed for Chinook salmon than steelhead trout.

- Average maximum movable substrate sizes were 83.3 mm (range = 25.0–125.0 mm) for Chinook salmon and 66.5 mm (range = 40.0–90.0 mm) for steelhead trout. Peak HSI values ranged from 40–90 mm ($D_{50max}$ = 85.8 mm and $D_{85max}$ = 94.6 mm) for Chinook salmon and 50–80 mm ($D_{50max}$ = 65.0 mm and $D_{85max}$ = 71 mm) for steelhead trout. Average maximum movable substrate sizes calculated for Chinook salmon were significantly larger than average maximum movable substrate sizes calculated for steelhead trout. When cumulative maximum movable substrate size distributions for

Chinook salmon and steelhead trout were compared to the $D_{95}$s of the 2008–2010 sites, 0.36% of Chinook salmon in the LAR would be prevented from spawning at the 2008 site, 99.86% would be prevented from spawning at the 2009 site, and 28.30% would be prevented from spawning at the 2010 site, whereas 3.02% of steelhead trout in the LAR would be prevented from spawning at the 2008 site, 100.00% would be prevented from spawning at the 2009 site, and 95.88% would be prevented from spawning at the 2010 site. Substrate sizes selected by Chinook salmon were ~1.4 to 2.8-fold ($D_{85}$ to $D_{50}$) smaller than maximum movable substrate sizes, whereas substrate sizes selected by steelhead trout were ~1.1 to 2.3-fold ($D_{85}$ to $D_{50}$) smaller than maximum movable substrate sizes. These data suggest that bed material size may play a significant role in relative spawning use of the three enhancement sites by both Chinook salmon and steelhead trout.

- Average estimated lengths for fish observed on redds were 0.81 m (range = 0.55–1.00 m) for Chinook salmon, 0.69 m (range = 0.60–0.85 m) for steelhead trout, and 0.76 m (range = 0.55–1.00 m) for total salmonids. Corresponding $D_{50}$s averaged 34.2 mm (range = 19.1–56.2 mm) for Chinook salmon, 29.8 mm (range = 24.5–42.2 mm) for steelhead trout, and 32.1 mm (19.1–56.2 mm) for total salmonids, whereas $D_{85}$s averaged 65.4 mm (range = 51.2–88.9 mm) for Chinook salmon, 57.9 mm (range = 53.2–72.2mm) for steelhead trout, and 61.86 mm (range = 51.2–88.9 mm) for total salmonids. In general, a wider range of steelhead trout sizes were observed spawning in "BELOW" average substrates when compared to "ABOVE" average substrates, whereas the observed size ranges for Chinook salmon and total salmonids were similar for each substrate category.

- Chinook salmon redd lengths averaged 1.20 m (range = 0.35–4.00 m), whereas redd widths averaged 0.94 m (range = 0.30–3.50 m) and tail lengths averaged 2.14 m (range = 0.60–4.70 m). Substrate size did not have a significant affect on these characteristics.

- Steelhead trout redd lengths averaged 0.90 m (range = 0.25–2.50 m), whereas redd widths averaged 0.91 m (range = 0.20–1.60 m), redd depths averaged 0.30 m (range = 0.10–0.80 m), tail lengths averaged 1.72 m (range = 0.20–5.00 m), and average tail widths averaged 0.76 m (range = 0.20–1.68 m). Substrate size had a significant affect on tail length, but not other measured characteristics. Redd characteristics were generally smaller in areas where substrates were relatively large, which suggests that substrate size may alter redd construction capabilities within enhancement sites. Increased sample size may improve statistical comparisons in future assessments.

- High flows and limited habitat complexity in the LAR complicated assessments of spawning location selection in relation to cover. However, general observations indicated Brush was the most common cover type associated with Chinook salmon and steelhead trout redds, followed by Riparian Grass, Trees, Overhanging Vegetation, and Large Woody Material. No redds were associated with Small Woody Material, Cattails, or Scalloped Banks. In general, utilization (%) rates for all cover types were low, and ranged from 0.0%–6.0% for Chinook salmon and 0.0%–8.2% and for steelhead trout. In all, 271 redds (~90%) were not associated with any cover type. Increased sample size should improve this assessment in future studies.

- Salmonid redds were most frequently associated with Main Channel units (212 Chinook salmon and 38 steelhead trout), followed by Side-Channel units (39 Chinook salmon and 10 steelhead trout), Floodplain units (19 Chinook salmon and 4 steelhead trout), Island units (13 Chinook salmon and 4 steelhead trout), and Main Channel/Floodplain units (5 Chinook salmon and 0 steelhead trout). Utilization rates ranged from 67.9%–84.1% for Main Channel units, 14.2%–17.9% for Side-Channel units, 6.7%–7.5% for Floodplain units, 4.9%–7.1% for Island units, and 0.0%–2.0% for Main Channel/Floodplain units.

- Juvenile Chinook salmon and steelhead trout were abundant in the study area, with a total of 9,222 Chinook salmon and 10,285 steelhead trout observed. Fry were the dominant life stage observed, followed by parr and smolts. Brush was the most common cover type associated with juvenile salmonid observations. However, Riparian Grass was also important. Juvenile steelhead trout were more abundant in shallower, higher velocity habitats, whereas Chinook salmon fry and parr were more abundant in deeper, lower velocity habitats associated with cover. Chinook salmon smolts were more abundant in deeper, higher velocity habitats. In general, depths and velocities in large portions of the main channel were unsuitable for juveniles during most of the 2011 sampling season. Therefore, observations during subsequent seasons will be required to effectively evaluate juvenile habitat associations within the restoration area.

- Benthic macroinvertebrates, including key juvenile salmonid prey items, began colonizing floodplains adjacent to gravel augmentation areas within days of inundation. Overall floodplain densities peaked at 8.9 weeks (62 days) inundation (average = 8,238 individuals/$m^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation. Average floodplain densities at 8.9 weeks inundation were significantly higher than average control densities. Floodplain biomass peaked at 10.0 weeks inundation (average = 1.89 g/$m^2$). Average floodplain biomass was significantly lower than average control biomass over the entire monitoring period. Floodplain densities were significantly greater at upstream (2008 and 2009) sites than downstream (2010) sites. However, biomass was not significantly different.

- Overall benthic macroinvertebrate species richness in newly-inundated floodplain habitats peaked at 8.9 weeks inundation (average = 7.0 families), and then fell but remained relatively high at 10.0 weeks inundation (average = 6.9 families). Average floodplain richness at all inundation durations was significantly lower than average control site richness. Simpson's Diversity Index values for floodplain sites showed a "bowl-shaped" curve, with localized peaks at 1.3 and 10.0 weeks inundation and a valley at 5.6 weeks inundation. Average floodplain diversity did not surpass average control diversity at any time during the sampling period (~1–10 weeks). Floodplain species richness was significantly greater at downstream (2010) sites than upstream (2008 and 2009) sites. Similarly, floodplain Simpson's Diversity Index values were generally greater at downstream (2010) sites than upstream (2008 and 2009) sites.

- For key salmonid prey items, the baetid mayfly (family Baetidae) and chironomid midges (family Chironomidae) peaked at 8.9 weeks inundation. At that time, floodplain densities and biomass surpassed control densities and biomass. The hydropsychid caddisfly (family Hydropsychidae) showed slower colonization rates and never surpassed control samples. Total Baetidae, Chironomidae, and Hydropsychidae (BCH) density and biomass in treatment samples were driven by relatively fast-colonizing chironomid midges, which suggests that most juvenile salmonid food production benefits from short-duration inundation events are derived from one benthic macroinvertebrate group; Chironomidae.

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## INTRODUCTION

Substrate recruitment and related ecosystem processes have been interrupted on regulated rivers in California's Central Valley, thereby resulting in a general reduction in both: (1) the quantity and quality of gravel substrates appropriate for anadromous salmonid spawning; and (2) ecosystem processes important for early life-history stages of anadromous salmonids (i.e., juvenile rearing). River regulation, and the resulting combination of channel incision, bed and bank erosion, and coarsening of bed material, have lowered streambed elevations, altered depths and velocities over once-productive spawning beds, reduced the magnitude and frequency of floodplain inundation important for juvenile rearing, and decreased overall habitat heterogeneity, hyporheic water quality, and macroinvertebrate production (Kondolf 1997; Pasternack et al. 2004; Merz and Chan 2004). The cumulative impacts to anadromous salmonid habitat have modified conditions suitable for successful spawning, incubation, and juvenile rearing, thereby exacerbating long-term downward population trends, and threatening the future viability of historically abundant Central Valley stocks (Yoshiyama et al. 1998).

In response to continued declines in anadromous salmonid stocks, Public Law 102-575 was passed by Congress in 1992, and under Title 34, established the Central Valley Project Improvement Act (CVPIA 1992). With the goal of protecting, restoring, and enhancing fish, wildlife, and associated habitats in the Central Valley, this legislation granted authority to the U.S. Bureau of Reclamation (Reclamation) and the U.S. Fish and Wildlife Service (USFWS) to co-lead anadromous fish restoration efforts for the U.S. Department of Interior (Interior). The resulting CVPIA Fisheries Program was directed to make 'all reasonable efforts' to double natural production of anadromous fish [Section 3406(b)(1)]. Section 3406(b)(1)(A) gives "…first priority to measures which protect and restore natural channel and riparian habitat values through habitat restoration actions…" (CVPIA 1992). Additionally, to compensate for actions that have reduced the availability of spawning and rearing habitat, CVPIA Section 3406(b)(13) authorizes and directs Reclamation and USFWS, along with other Federal and State agencies, to create a program to continue the restoration and replenishment of spawning gravel in Central Valley rivers; including the American River below Nimbus Dam.

In 2008, under CVPIA Section 3406(b)(13), a multi-year gravel augmentation project was implemented on the lower American River (LAR) by Reclamation and USFWS, in partnership with the Water Forum, as part of the LAR habitat enhancement program. The overall vision for the habitat enhancement program is to restore ecosystem processes by rehabilitating and enhancing critical channel, floodplain, and riparian habitats for juvenile and adult anadromous salmonids, thereby promoting the recovery of healthy and diverse Chinook salmon *(Oncorhynchus tshawytscha)* and steelhead trout *(O. mykiss)* populations. This vision fits into the framework of salmonid population recovery on the LAR and is aligned with the (b)(13) programmatic goal of "…protecting, restoring, and enhancing fish spawning and rearing habitat to increase fish production and encourage ecosystem function." Additionally, this vision is considered in the context of historic land use and current water management constraints to ensure maximum benefits can be derived from management activities. Primary management goals fit into the framework of the CVPIA, meet the recommendation to use adaptive management in planning, design, and implementation, and include:

1. *Increasing the availability, quantity, and quality of spawning gravel and rearing habitat for American River Chinook salmon and steelhead trout;*

2. *Restoring, enhancing, or maintaining natural ecosystem processes whenever possible; and,*

3. *Determining project effectiveness with an efficient and scientifically-robust monitoring program.*

Gravel augmentation is a widely accepted technique for restoring anadromous salmonid spawning habitats throughout the Central Valley (Wheaton et al. 2004a; Wheaton et al. 2004b). However, both the physical and biological effects of restoration projects are influenced by a suite of intermediate mechanisms and external factors related to hydrodynamics, geomorphology, and ecology (Downs and Kondolf 2002). Therefore, the overall effects of restoration projects on river ecosystems and specified life stages of target species, and secondary influences on non-target organisms, are highly variable both within and among systems. Comprehensive monitoring to evaluate project effectiveness provides an important measure of project performance and success (AMF 2004; CBDP 2005; CVPIA 2008).

In this report, we provide 2010–2011 effectiveness monitoring results for the spawning gravel placements carried out at Sailor Bar on the LAR, California, during the summers of 2008–2010 (three enhancement sites). Field and laboratory activities were designed to assess project effectiveness via overall ecosystem function in an indicator-species-centered context (Wheaton et al. 2004b). Physical and biological factors used to measure project effectiveness included:

*Physical*

*(1) Area of floodplain inundation adjacent to the 2008–2010 augmentation sites at different flows to evaluate effects of different flow management strategies on timing, quantity, and quality of available juvenile salmonid rearing habitat at Sailor Bar.*

*Biological*

*(1) Use of the 2008–2010 augmentation sites and unenhanced sites by spawning Chinook salmon and steelhead trout to evaluate how the enhancement project changed salmonid spawning location preferences throughout the LAR;*

*(2) Use of the 2008–2010 augmentation sites and areas adjacent to the 2008–2010 augmentation sites by spawning Chinook salmon and steelhead trout to evaluate salmonid spawning habitat preferences (including depth, velocity, substrate, cover, and channel unit type) at Sailor Bar;*

*(3) Use of areas adjacent to the 2008–2010 augmentation sites by juvenile Chinook salmon and steelhead trout to evaluate juvenile salmonid rearing habitat preferences (including depth, velocity, substrate, and cover) at Sailor Bar; and*

*(4) Benthic production within the main channel and floodplain areas adjacent to the 2008–2010 augmentation sites to evaluate how the benthic macroinvertebrate community responded to different durations of floodplain inundation at Sailor Bar.*

All methods and results are presented in the form of individual, hypothesis-driven studies designed to explicitly test specific ideas related to each of the physical and biological factors listed above. The discussion addresses the findings of each study in a combined approach designed to examine all results in the context of overall project effectiveness at restoring anadromous salmonid spawning and rearing habitats and overall ecosystem function.

## STUDY AREA

The American River drains a roughly triangular watershed of ~1,900 mi$^2$ (4,921 km$^2$), with elevations ranging from 10,400 ft (3,170 m) at the headwaters to ~25 ft (7.62 m) at the confluence with the Sacramento River. During the late-1800s, the American River was inundated with gravel and smaller sediments as hydraulic gold mining in the Sierra foothills washed huge amounts of debris downstream. In the upper part of the LAR between Folsom and Fair Oaks, bucket line dredging for gold, sometimes at depths >100 ft (30.48 m), was conducted throughout the first half of the twentieth century (Oakland Museum of California 2009).

Two large dams regulate flow levels in the LAR. Folsom Dam is located ~30 mi (48.28 km) upstream from the confluence with the Sacramento River, and creates a 975,000 acre-foot reservoir that supports multiple recreational and commercial uses. Nimbus Dam is located at river mile (RM) 23 (river kilometer 37.02), marks the upstream limit of the LAR, and acts as a regulating facility for Folsom hydropower while diverting a small amount of water into the Folsom-South Canal (Lower American River Task Force 2002). Since the installation of Folsom Dam, the LAR has incised through the accumulated hydraulic mining debris to its earlier bed elevation and is now eroding laterally. Gravel and sediment from the upstream watershed are captured in Folsom Lake and are unable to replenish downstream spawning beds. Since the early 1960s, the net loss of gravel in the system has been ~57,500 yd$^3$ (~44,000 m$^3$ or 92,000 tons) per year, with most loss occurring during events with flows exceeding 100,000 ft$^3$•s$^{-1}$ (2,832 m$^3$•s$^{-1}$) (Fairman 2007). Nimbus Dam serves as the upstream limit to migration for anadromous fish species, and blocks access to ~70% of the historic spawning habitat for Chinook salmon and the entire historic spawning habitat for steelhead trout.

The main hydrologic effect of the two large dams on the LAR has been to dampen variance in winter runoff and to store snowmelt for release during the summer irrigation season. Annual river discharge averages ~3,750 ft$^3$•s$^{-1}$ (2,717,000 acre-feet per year), but has varied from 730–7,900 ft$^3$•s$^{-1}$ (21–224 m$^3$•s$^{-1}$). Runoff comes from winter rains at lower elevations and from spring snowmelt at higher elevations; very high flows all result from winter storms. "Natural" mean monthly flows rise to a peak in May and typically drop to low levels from August–October. Folsom Reservoir is relatively small compared to the mean annual flow. However, peak flow reductions during wet years have been moderate. Geomorphically effective flows still occur with some regularity. Mean daily flow ranged from 753–31,425 ft$^3$•s$^{-1}$ (21–890 m$^3$•s$^{-1}$) for the period 1 May 2008 through 31 July 2011 (Figure 1).

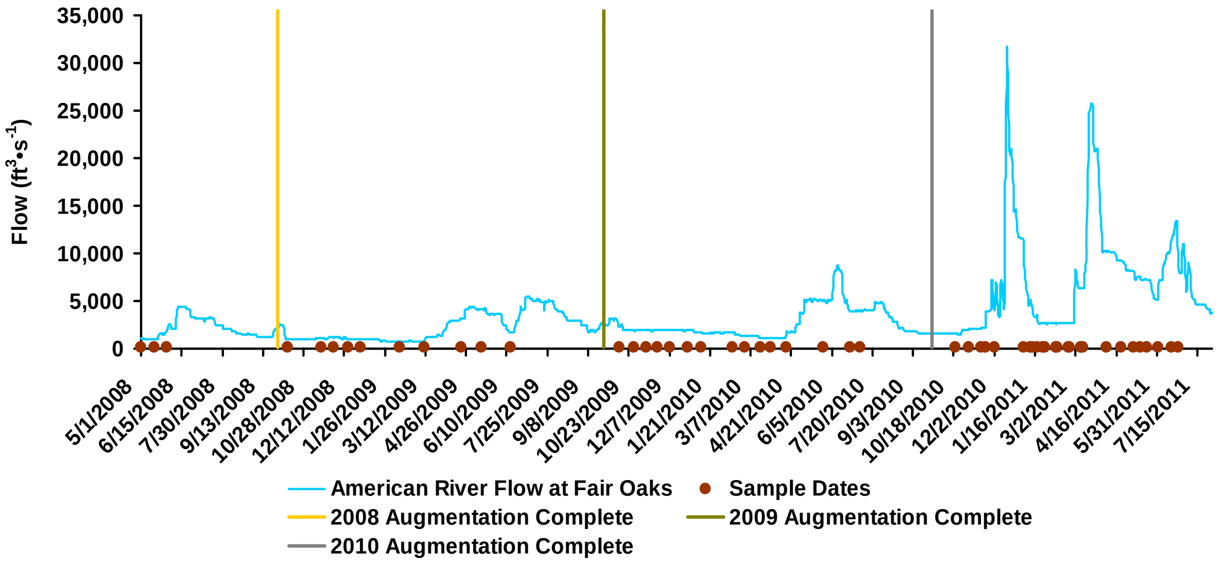


***Figure 1****. Lower American River flow at Fair Oaks from 1 May 2008 through 31 July 2011 (Data Source – California Department of Water Resources, California Data Exchange Center [CDEC]).*

The Sailor Bar project area is located in the LAR reach immediately below Nimbus Dam. This reach is characterized by increased slope, a mainly gravel bed, long pools separated by riffles, and an average wetted width of 275 ft (83.8 m) at 1,000 $ft^3 \bullet s^{-1}$ (28 $m^3 \bullet s^{-1}$). Limited availability of suitable spawning gravel and rearing areas within the LAR, including this reach, have reduced habitat for anadromous salmonids. However, the LAR and Nimbus Hatchery are still productive for both Central Valley fall-run Chinook salmon and steelhead trout. The LAR also provides productive habitat for at least one species of lamprey, the Pacific lamprey (*Lampetra tridentata*), the focus of a USFWS conservation initiative (USFWS 2008).

From 1967–1995, fall-run Chinook salmon escapement to the American River averaged 32,000 (Snider and Reavis 1996), ~16% of the average escapement for the entire Sacramento River system (Moyle 2002). During the early 1990s, fish diverted into Nimbus Hatchery made up 9%–59% of that total (Yoshiyama et al. 2001). Steelhead trout escapement during 2003–2007 was estimated at between 1,310 and 2,672 fish, with an estimated 343–504 naturally spawning adults (Hannon and Deason 2008). Since 1955, the number of steelhead trout entering Nimbus Hatchery has ranged from under 1,000 to ~6,000 (Water Forum 2005).

## METHODS

### *Overview of Gravel Placement*

A 2D hydrodynamic model was used to design gravel placements to maximize salmonid spawning use following the methods of Pasternack et al. (2004). In September 2008[1], ~7,000 tons (~6,350 metric tons) of cleaned gravel was placed in the American River upstream of Sailor Bar in an area across the river from Nimbus Hatchery (Figure 2). The placed material was derived from a rock quarry source on the Yuba River, and was comprised of gravel sizes from 6–

[1] See CFS 2009 report for further information (Sacramento Water Forum Grant No. G14000200)

102 mm, with 95% of the placed material from 6–51 mm ($D_{50}$~24 mm). In September 2009[2], the 2008 augmentation site was extended immediately downstream with an additional ~10,500 tons of cleaned gravel (Figure 2). The placed material was derived from an American River floodplain source at Mississippi Bar, and was comprised of gravel sizes from 7–188 mm, with 95% of the placed material from 7–112 mm ($D_{50}$~35 mm). Additionally, the 2009 augmentation site contained discrete piles of larger (22–280 mm; 95% from 22–234 mm; $D_{50}$~120 mm) and smaller (15–127 mm; 95% from 15–86 mm; $D_{50}$~39 mm) gravel designed to enhance overall site complexity (see Wheaton et al. 2004c) and test the effects of substrate size, depth, and velocity on the overall density and biomass of macroinvertebrates, and the density and biomass of macroinvertebrate taxa preferred by juvenile Chinook salmon and steelhead trout. In September 2010, an additional ~10,700 tons of cleaned gravel was placed ~1.3 mi (2.09 km) downstream of the 2008 and 2009 augmentation sites (Figure 2). The placed material was derived from an American River floodplain source adjacent to the 2010 augmentation site, and was comprised of gravel sizes from 8–178 mm, with 95% of the placed material from 8–78 mm ($D_{50}$~30 mm; Figure 3). Additionally, the 2010 augmentation site contained a constructed cobble island (8–178 mm; 95% from 8–125 mm; $D_{50}$~73 mm) and "scallops" in the substrate designed to add habitat heterogeneity to the main channel and rearing habitat for juvenile Chinook salmon and steelhead trout (Figures 3 and 4). Further, an additional ~5,500 tons of cleaned cobble was placed downstream of the 2010 augmentation site. The specific purpose of this placement was to divert flow into an adjacent, perched side-channel, thereby preventing the de-watering of salmonid redds in a historically important spawning and rearing area during low-flow conditions. For this reason, the side-channel adjacent to the 2010 gravel placement is treated as an augmentation site throughout this report. These actions may also benefit other habitats as gravels mobilize from upstream placements and collect in downstream areas (Figure 2).

[2] See CFS 2010 report for further information (Sacramento Water Forum Grant No. G14000200)

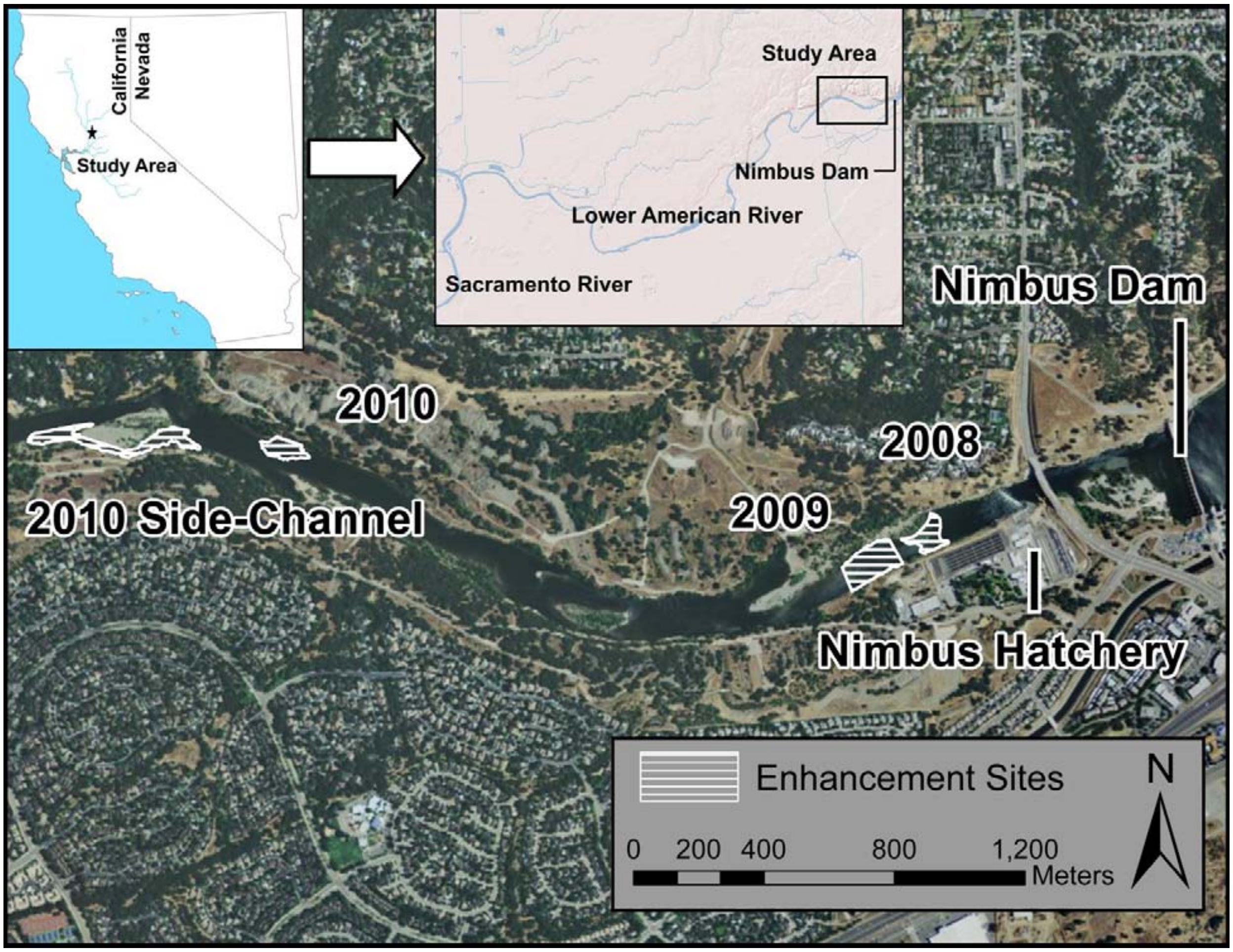


***Figure 2***. *Locations of the 2008–2010 gravel augmentation (i.e., enhancement) sites at Sailor Bar on the lower American River. The 2010 gravel placement was designed to maintain inundation of the side-channel immediately downstream and adjacent to the site. This area is treated as an augmentation site throughout the report (i.e., 2010 side-channel).*

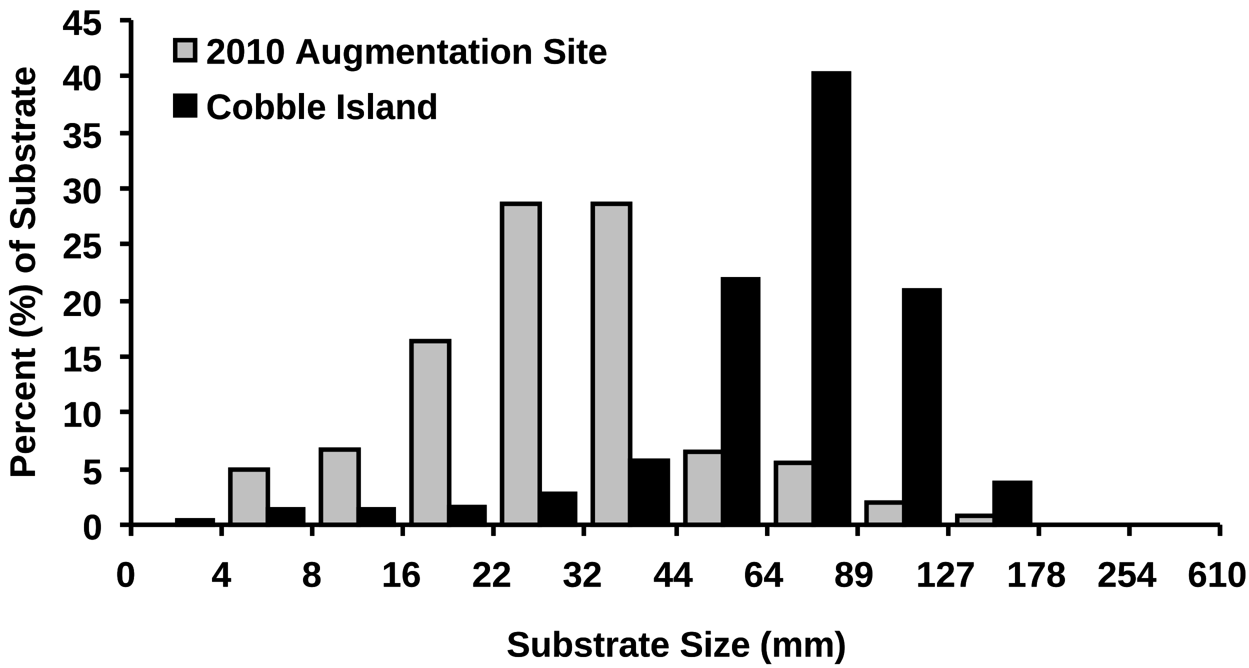


***Figure 3**. Pebble count data from the 2010 augmentation site ($D_{50}$~30 mm) and the cobble island ($D_{50}$~73 mm) added to the 2010 augmentation site.*

### *General Monitoring Approach*

The overall effects of restoration projects on river ecosystems and target species are highly variable both within and among systems, and comprehensive monitoring to evaluate project effectiveness provides an important measure of project performance and success (AMF 2004; CBDP 2005; CVPIA 2008). Therefore, we conducted site-specific effectiveness monitoring to evaluate changes in habitat conditions for Chinook salmon and steelhead trout after the 2008–2010 gravel placements at Sailor Bar on the LAR (due to the difficulty in visually distinguishing steelhead trout from rainbow trout, we group them as one designation throughout this report; steelhead trout). Site-specific effectiveness monitoring tracks physical conditions and biological responses to restoration actions necessary to enhance spawning and rearing habitat for anadromous salmonids, is hypothesis driven, and seeks to answer the question: "Was the project effective at meeting restoration objectives?" Physical conditions important to monitoring typically include parameters related to hydrology and water quality. Biological responses important to monitoring typically include indices related to fish use and abundance, macroinvertebrate production, fish foraging success, diet composition, and potential growth.

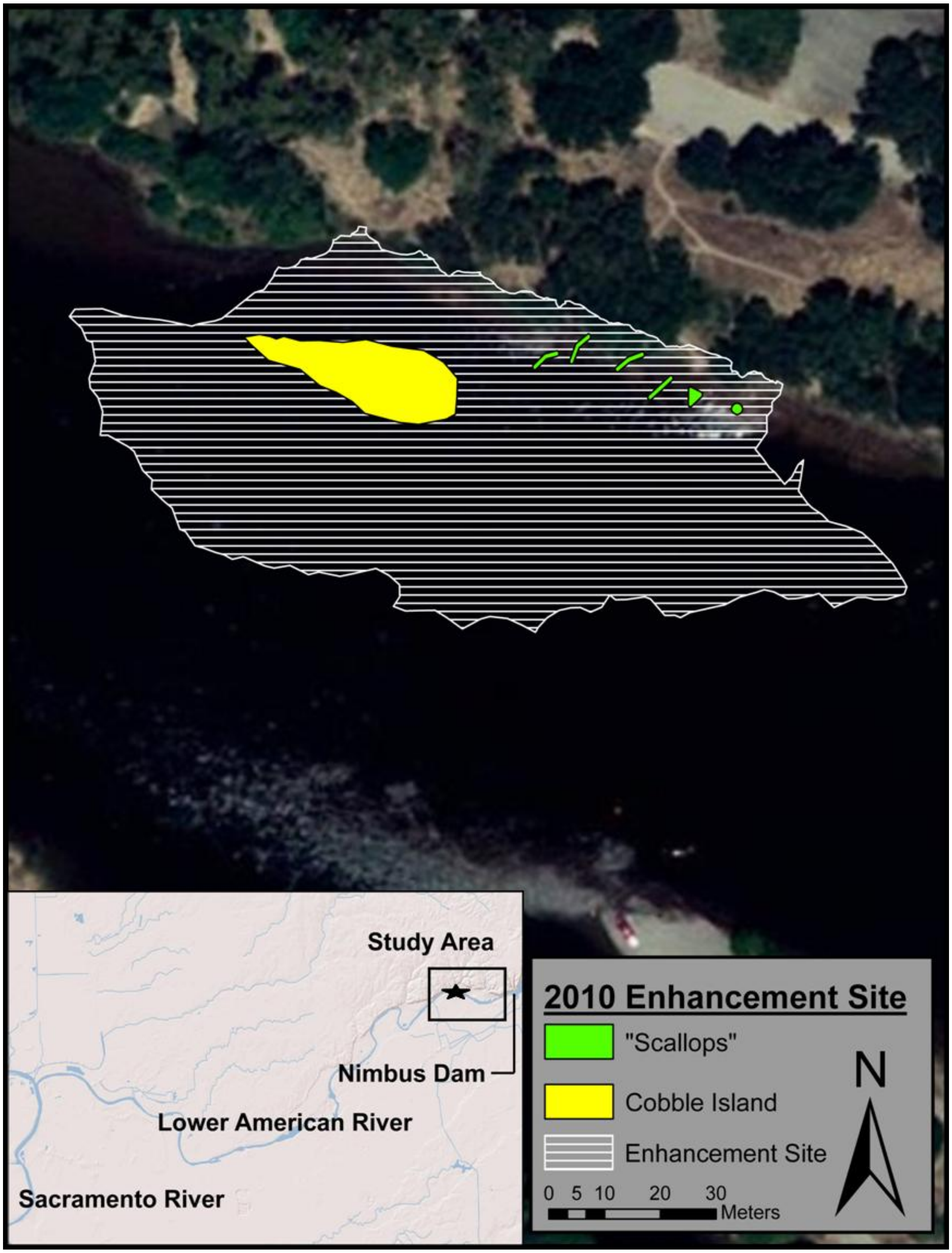


***Figure 4****. Locations of the cobble island ($D_{50}$~73 mm) and "scallops" in the substrate added to the 2010 augmentation (i.e., enhancement) site.*

We employed a Before-After, Control-Impact (BACI) design structure to test differences between unenhanced and enhanced sites (i.e., Control-Impact) by subjecting these areas to replicated measurements over time (i.e., Before-After). We also employed GIS-based habitat mapping to evaluate spawning and rearing habitat preferences of adult and juvenile Chinook salmon and steelhead trout, and to evaluate differences in benthic production between the main channel and adjacent floodplains. To evaluate physical conditions, we used current measurements of water's edge to assess floodplain inundation adjacent to the 2008–2010 augmentation sites after gravel placement, and developed flow-inundation curves. To document biological responses, we measured use of the 2008–2010 augmentation sites and unenhanced sites by spawning Chinook salmon and steelhead trout, compared adult and juvenile salmonid spawning and rearing habitat use to habitat availability, and compared benthic production within the main channel to floodplain areas adjacent to the 2008–2010 augmentation sites.

### *Floodplain Inundation*

In many Central Valley rivers, both spawning and rearing habitat are important factors limiting anadromous salmonid production (Williams 2006). However, the vast majority of gravel augmentation projects are associated only with spawning habitat improvements (Kondolf et al. 2008). This could create a problem if augmentation negatively affects salmonid rearing habitat or if sufficient rearing habitat is not available to support increased juvenile production from augmentation sites. Therefore, expanding our understanding of how gravel augmentation influences both spawning and rearing habitat is an important management goal.

The 2008–2010 gravel placements at Sailor Bar on the LAR effectively filled a highly incised river channel and raised streambed elevation at augmentation sites. This created a situation where damming of river flow by gravel "plugs" forced water onto adjacent floodplains at lower flows than would have occurred under incised conditions. In the case of the 2010 augmentation site, a gravel "plug" also forced water onto and around a natural gravel island and into an adjacent side-channel. In this study, we use current water's edge measurements to develop flow-inundation curves. The goal of this study was to evaluate the effects of different flow management strategies on the quantity and quality of available juvenile rearing habitat at Sailor Bar. Specifically, we tested the following hypothesis:

> $1H_0$: *There is no difference in available juvenile rearing habitat adjacent to the 2008–2010 augmentation sites when quantitatively and qualitatively comparing areas of floodplain inundation under different flow management strategies.*

*Methods* – Differentially corrected GPS water's edge measurements were taken at a variety of flows throughout the spring 2011 sampling season (Table 2; Figures 5 and 6), and average daily flows available for each water's edge measurement (CDEC) were used to develop flow-inundation curves for current conditions (i.e., post-enhancement). Flow-inundation curves were developed for the floodplain adjacent to the 2008 and 2009 augmentation sites and the island adjacent to the 2010 augmentation site both individually and combined. All curves were developed using overlay and area calculation procedures in ArcMap. Differentially corrected GPS coordinates were collected by walking the water's edge using a Trimble GeoXT unit with GPS track capabilities. Qualitative data (i.e., data related to the quality of inundated areas for juvenile salmonid rearing) were collected during GIS-based habitat mapping (see below).

*Analysis* – A total of six water's edge measurements (see above) taken at flows ranging from 2,729 $ft^3 \bullet s^{-1}$ (77 $m^3 \bullet s^{-1}$; 23 February 2011) to 9,449 $ft^3 \bullet s^{-1}$ (268 $m^3 \bullet s^{-1}$; 4 January 2011) were used to develop current (i.e., post-2008–2010 gravel placement) flow-inundation curves (Figure 7). We assumed a minimum flow (i.e., lowest flow available) of 2,729 $ft^3 \bullet s^{-1}$ (77 $m^3 \bullet s^{-1}$), and developed flow-inundation curves for additional inundated areas above this minimum flow level (i.e., flows above 2,729 $ft^3 \bullet s^{-1}$). Curves were developed in a stepwise fashion and were assumed to be linear (i.e., linear relationships developed for each step in flow). Inundated areas were assumed to be "capped" at the largest observed values when flows exceeded flows for which water's edge measurements were available (i.e., capped at inundated areas present at 9,449 $ft^3 \bullet s^{-1}$ on 4 January 2011). Similarly, inundated areas were assumed to be "capped" at the smallest observed values when flows did not reach flows for which water's edge measurements were available (i.e., capped at inundated areas present at 2,729 $ft^3 \bullet s^{-1}$ on 23 February 2011).

Flow-inundation curves developed for current conditions were used to calculate inundated areas ($m^2$) available for use as juvenile salmonid rearing habitat on a daily basis for observed flow data from example "critical" (1994) and "wet" (1999) water years (CDEC). Inundated areas available for use as juvenile salmonid rearing habitat for example water years were then compared to daily average habitat requirements ($m^2$) for juvenile Chinook salmon, which were calculated from smoothed weekly average outmigration timing distributions and fork length (FL) values obtained from 1994 and 1999 rotary screw trap (RST) monitoring on the LAR (Snider and Reavis 1996; Snider et al. 2002) and the average FL – territory size relationship for salmonids in general developed by Grant and Kramer (1990):

$$TS = \frac{\left(\frac{FL}{10}\right)^{2.61}}{10^{2.83}} \tag{1}$$

where *TS* = territory size ($m^2$) and *FL* = fork length (mm). An iterative, locally-weighted least-squares (i.e., LOWESS) method (tension = 0.05) was used to smooth all weekly outmigration timing and FL data. Expanded juvenile production estimates were only available for 1999 (12,566,322 fish; Snider et al. 2002). However, adult escapement estimates were available for both 1999 (50,457 fish; Snider and Titus 2002) and 1994 (31,027 fish; Snider and Reavis 1996). Therefore, we used the ratio of expanded juvenile production to adult escapement in 1999 to estimate expanded juvenile production in 1994 (7,727,278 fish). Additionally, we used weekly outmigration timing data, FL data, and the ratio of expanded juvenile production to adult escapement in 1999 to estimate expanded juvenile production (39,840,000 fish) and daily habitat requirements for the USFWS Anadromous Fish Restoration Program (AFRP) production target for fall-run Chinook salmon in the LAR (160,000 adults; AFRP). Daily estimates of required habitat for juveniles produced by the AFRP adult production target were then compared to inundated areas available for use as juvenile rearing habitat using methods identical to those used for 1994 and 1999 RST data.

We assume average habitat suitability (i.e., HSI = 0.50) results in average territory size for fish of a given FL, whereas above (i.e., HSI > 0.50) or below (i.e., HSI < 0.50) average habitat suitability results in a proportional increase or decrease in average territory size for fish of a given FL. We use an average HSI value of 0.50 as a starting value, and then provide example calculations using both a 50% increase and 50% decrease in average HSI value to provide fishery managers with a general idea of how daily average rearing habitat requirements for juvenile Chinook salmon change with habitat quality. These rearing habitat requirements are then compared to available inundated areas adjacent to the 2008–2010 augmentation sites. Comparisons are made for both "critical" and "wet" years using outmigration data from 1994 and 1999, and simulated outmigration data for the AFRP adult production target (see above). We provide a simple example of how rearing habitat requirements change through time (1 January–31 July) based on the number and average size of juvenile Chinook salmon in the LAR, and how flow management strategies (i.e., critical or wet water years) indirectly influence the potential carrying capacity of the LAR to produce juvenile Chinook salmon. This analysis is not intended to be comprehensive; it provides a simple framework for addressing production capacity, flow management, and restoration actions in a defensible, quantitative manner which can be improved upon in the future using additional habitat mapping, 2D modeling, and detailed habitat suitability relationships.

***Table 1***. *Sample dates, flows, and survey types conducted at Sailor Bar, lower American River, California, 2008–2011.*

| Week Post-2008 Enhancement | Week Post-2009 Enhancement | Week Post-2010 Enhancement | Sample Date | Flow (ft$^3$•s$^{-1}$) | Surveys Conducted |
|---|---|---|---|---|---|
| Pre | Pre | Pre | 01-May-08 | 1,045 | Snorkel; Macroinvertebrates |
| Pre | Pre | Pre | 16-May-08 | 1,002 | Snorkel; Macroinvertebrates |
| Pre | Pre | Pre | 29-May-08 | 1,999 | Snorkel; Macroinvertebrates |
| 0 | Pre | Pre | 29-Sep-08* | 2,138 | None |
| 2 | Pre | Pre | 10-Oct-08 | 1,004 | Spawning; Snorkel; Macroinvertebrates |
| 7 | Pre | Pre | 17-Nov-08 | 1,063 | Spawning; Snorkel; Macroinvertebrates |
| 9 | Pre | Pre | 01-Dec-08 | 1,144 | Spawning; Snorkel; Macroinvertebrates |
| 11 | Pre | Pre | 17-Dec-08 | 990 | Spawning; Snorkel; Macroinvertebrates; Gravel Mobility |
| 13 | Pre | Pre | 30-Dec-08 | 996 | Spawning; Snorkel; Macroinvertebrates |
| 19 | Pre | Pre | 12-Feb-09 | 784 | Snorkel; Macroinvertebrates |
| 23 | Pre | Pre | 11-Mar-09 | 773 | Snorkel; Macroinvertebrates |
| 29 | Pre | Pre | 21-Apr-09 | 3,157 | Snorkel; Macroinvertebrates |
| 32 | Pre | Pre | 13-May-09 | 4,196 | Snorkel; Macroinvertebrates; Gravel Mobility |
| 37 | Pre | Pre | 15-Jun-09 | 1,764 | Gravel Mobility |
| 52 | 0 | Pre | 25-Sep-09* | 2,537 | None |
| 54 | 3 | Pre | 13-Oct-09 | 2,332 | Spawning |
| 56 | 5 | Pre | 29-Oct-09 | 1,922 | Spawning |
| 58 | 7 | Pre | 12-Nov-09 | 1,977 | Spawning |
| 60 | 9 | Pre | 24-Nov-09 | 1,958 | Spawning; Macroinvertebrates |
| 62 | 11 | Pre | 08-Dec-09 | 1,973 | Spawning; Macroinvertebrates |
| 65 | 13 | Pre | 28-Dec-09 | 1,938 | Spawning; Macroinvertebrates |
| 67 | 15 | Pre | 11-Jan-10 | 1,654 | Spawning |
| 72 | 20 | Pre | 15-Feb-10 | 1,668 | Spawning; Snorkel |
| 74 | 22 | Pre | 01-Mar-10 | 1,395 | Spawning |
| 76 | 25 | Pre | 18-Mar-10 | 1,142 | Spawning; Snorkel |
| 78 | 26 | Pre | 29-Mar-10 | 1,140 | Spawning |
| 81 | 29 | Pre | 16-Apr-10 | 1,798 | Snorkel |
| 86 | 35 | Pre | 27-May-10 | 5,096 | Snorkel |
| 91 | 39 | Pre | 25-Jun-10 | 3,914 | Snorkel; Gravel Mobility |
| 92 | 41 | Pre | 07-Jul-10 | 3,964 | Gravel Mobility |
| 104 | 52 | 0 | 24-Sep-10* | 1,594 | None |
| 107 | 56 | 4 | 20-Oct-10 | 1,552 | Spawning; Habitat |
| 109 | 58 | 6 | 04-Nov-10 | 1,952 | Spawning; Habitat |
| 111 | 60 | 8 | 18-Nov-10 | 2,118 | Spawning; Habitat |

| | | | | | |
|---|---|---|---|---|---|
| 112 | 60 | 8 | 22-Nov-10 | 2,141 | Macroinvertebrates; Habitat |
| 113 | 62 | 10 | 02-Dec-10 | 3,999 | Spawning; Macroinvertebrates |
| 118 | 67 | 15 | 04-Jan-11 | 9,449 | Spawning; Macroinvertebrates |
| 119 | 68 | 16 | 11-Jan-11 | 5,110 | Spawning |
| 120 | 68 | 16 | 14-Jan-11 | 3,705 | Spawning; Macroinvertebrates |
| 120 | 69 | 17 | 19-Jan-11 | 3,305 | Macroinvertebrates |
| 121 | 70 | 18 | 25-Jan-11 | 2,741 | Spawning; Macroinvertebrates |
| 121 | 70 | 18 | 27-Jan-11 | 2,718 | Spawning |
| 123 | 72 | 20 | 08-Feb-11 | 2,606 | Spawning |
| 123 | 72 | 20 | 09-Feb-11 | 2,569 | Spawning |
| 123 | 72 | 20 | 10-Feb-11 | 2,654 | Spawning; Snorkel; Macroinvertebrates; Habitat |
| 125 | 74 | 22 | 22-Feb-11 | 2,730 | Spawning |
| 125 | 74 | 22 | 23-Feb-11 | 2,729 | Spawning; Snorkel |
| 127 | 76 | 24 | 08-Mar-11 | 6,346 | Spawning |
| 127 | 76 | 24 | 09-Mar-11 | 6,361 | Spawning |
| 127 | 76 | 24 | 10-Mar-11 | 6,401 | Snorkel; Macroinvertebrates |
| 131 | 80 | 28 | 06-Apr-11 | 10,177 | Snorkel; Macroinvertebrates |
| 133 | 82 | 30 | 21-Apr-11 | 9,277 | Snorkel; Macroinvertebrates |
| 135 | 84 | 32 | 05-May-11 | 8,136 | Snorkel |
| 136 | 85 | 33 | 12-May-11 | 7,486 | Macroinvertebrates |
| 138 | 86 | 34 | 20-May-11 | 7,256 | Snorkel; Macroinvertebrates |
| 139 | 88 | 36 | 01-Jun-11 | 5,143 | Snorkel |
| 141 | 90 | 38 | 16-Jun-11 | 11,523 | Snorkel |
| 143 | 91 | 39 | 24-Jun-11 | 9,400 | Habitat |

**Gravel placement completion date*

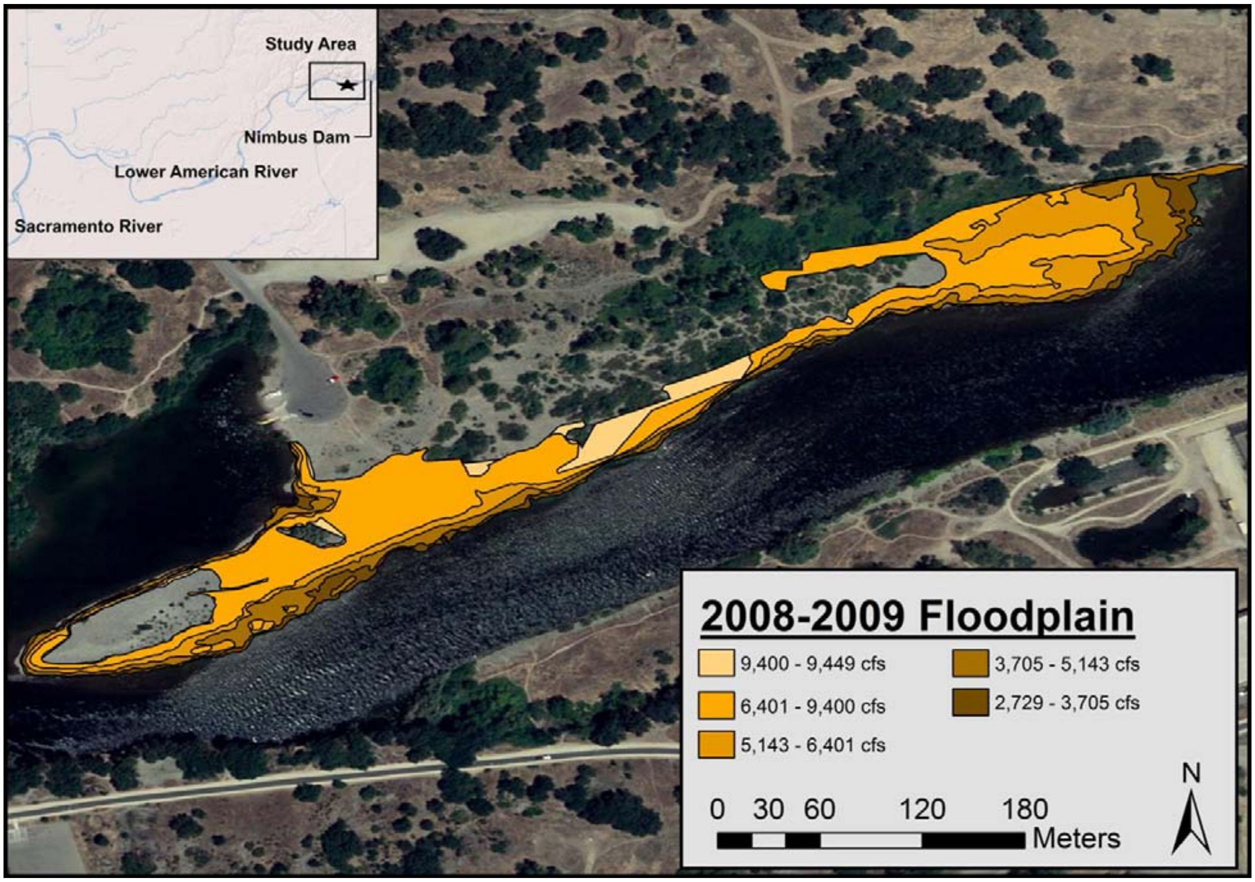


***Figure 5**. Current (post-2008–2010 gravel placement) areas of inundation for the floodplain adjacent to the 2008–2009 augmentation sites. Flows ranged from 2,729 $ft^3{\bullet}s^{-1}$ (23 February 2011) to 9,449 $ft^3{\bullet}s^{-1}$ (4 January 2011). Color scale goes from dark to light (low flow to high flow). As flows increase, darker areas inundate first, followed by lighter areas. Visible areas represent exposed (i.e., non-inundated) floodplain.*

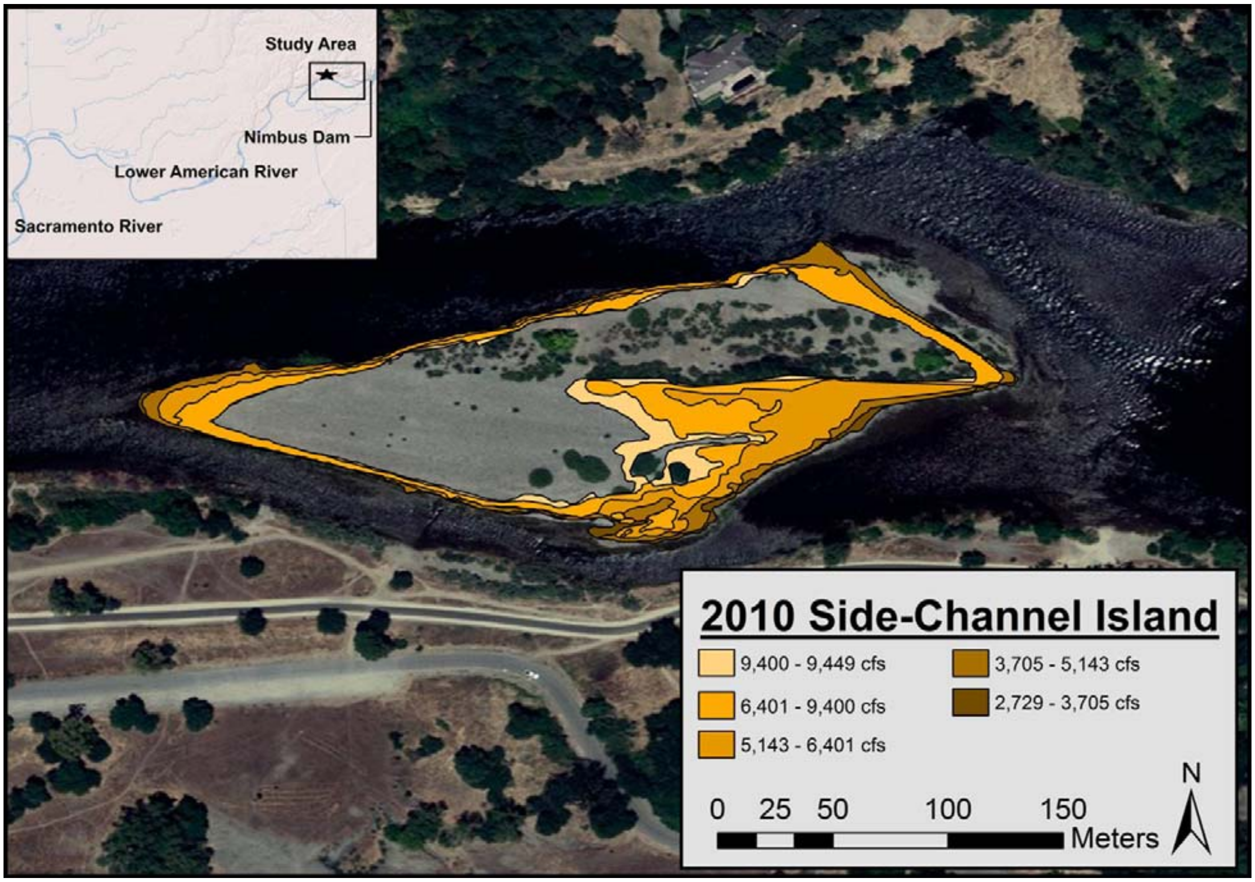


***Figure 6**. Current (post-2008–2010 gravel placement) areas of inundation for the side-channel island adjacent to the 2010 augmentation site. Flows ranged from 2,729 $ft^3 \cdot s^{-1}$ (23 February 2011) to 9,449 $ft^3 \cdot s^{-1}$ (4 January 2011). Color scale goes from dark to light (low flow to high flow). As flows increase, darker areas inundate first, followed by lighter areas. Visible areas represent exposed (i.e., non-inundated) floodplain.*

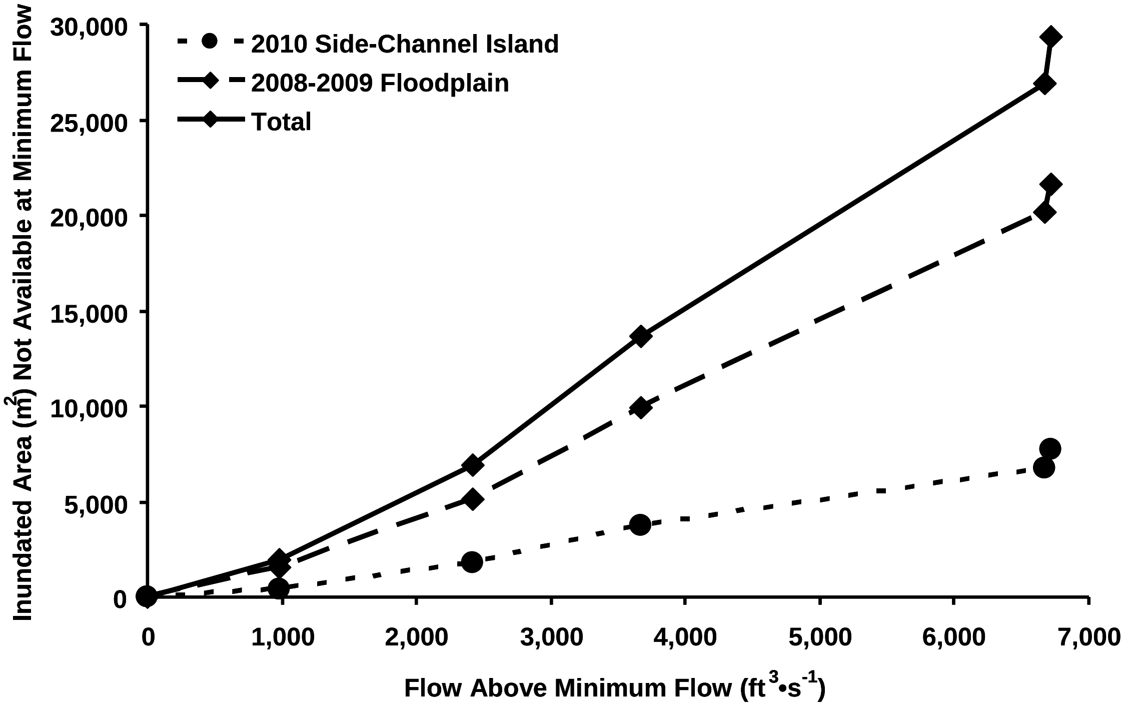


***Figure 7**. Current (post-2008–2010 gravel placement) flow-inundation curves developed for the floodplain adjacent to the 2008 and 2009 augmentation sites and the island adjacent to the 2010 augmentation site both separately and combined (Total). Flows ranged from 2,729 $ft^3 \cdot s^{-1}$ (23 February 2011) to 9,449 $ft^3 \cdot s^{-1}$ (4 January 2011). We assumed a minimum flow of 2,729 $ft^3 \cdot s^{-1}$ (23 February 2011) for all comparisons.*

### *Spawning Habitat Use*

Chinook salmon and steelhead trout spawning distribution can vary widely at individual sites among years. Therefore, an important component of effectiveness monitoring is assessing fish habitat use in relation to the effects of placed gravel. In this study, we compare use of the 2008–2010 augmentation sites and unenhanced sites by spawning Chinook salmon and steelhead trout to evaluate how the 2008–2010 gravel placements changed salmonid spawning location preferences throughout the LAR. Specifically, we tested the following hypothesis:

> *$1H_0$: There is no significant difference in the proportion of Chinook salmon and steelhead trout utilizing augmentation sites when comparing pre-enhancement utilization rates to post-enhancement utilization rates.*

*Methods* – We conducted Chinook salmon and steelhead trout spawning surveys at the 2008–2010 augmentation sites on a roughly bi-weekly basis from 20 October 2010 through 9 March 2011 (Table 2). Differentially corrected GPS coordinates were collected for individual redds using a Trimble GeoXT unit, and Chinook salmon and steelhead trout counts were summed for each sample date. Coordinates for individual redds were input into ArcMap and used to display the spatial extent of spawning throughout the project reach on a monthly basis.

In addition to these surveys, Reclamation has been conducting annual Chinook salmon redd surveys on the LAR since 2004–2005, and annual steelhead trout redd surveys since 2001–2002.

For Chinook salmon, Reclamation uses three flights per year to collect aerial photographs of the reach from the Business-80 Bridge to Nimbus Dam (~29 km). Flights typically occur in early-November, mid- to late-November, and early-December. Photographs are used to estimate redd abundance by reach. For steelhead trout, Reclamation uses boat, snorkel, and walking surveys to locate and GPS redds. Surveys typically occur during spring (January–April), and cover the entire LAR from Nimbus Dam downstream to Paradise Beach (~29 km). Redd abundance is typically reported by reach.

*Analysis* – To evaluate the effects of gravel placement on Chinook salmon and steelhead trout spawning use, we performed a chi-square analysis:

$$X^2 = \sum \frac{(X_a - X_b)^2}{X_b} \qquad (2)$$

Where $X_b$ = redds at enhancement sites before enhancement/total river redds before enhancement, and $X_a$ = redds at enhancement sites after enhancement/total river redds after enhancement (Zar 1999). This analysis allowed us to use data collected prior to and during the 2008–2009 through 2010–2011 spawning seasons to explicitly test whether spawning habitat use after augmentation differed from what would have been expected before augmentation. Separate chi-square analyses were conducted for each augmentation site (i.e., 2008–2010 augmentation sites) using species-specific (i.e., Chinook salmon and steelhead trout) data. Additionally, a qualitative analysis of steelhead trout redd distribution was conducted for the entire length of the LAR, whereby each RM was assigned an average pre-enhancement utilization rate which was compared to post-enhancement utilization rates.

### *Spawning Depth, Velocity, Substrate Size, Redd Characteristics, and Habitat Preferences*

Chinook salmon and steelhead trout spawning distribution can vary widely at individual sites among years. Depth, velocity, substrate, cover, and the discrete channel unit types that control the availability and interactions of these habitat features all influence the distribution of spawning salmonids. Therefore, an important component of effectiveness monitoring is assessing fish habitat use in relation to habitat availability (i.e., habitat preferences), and the resulting effects of spawning location on the spawning process and reproductive success. The goals of this study were to use GIS-based habitat mapping and data collected concurrently with Chinook salmon and steelhead trout redd observations to: (1) develop detailed HSI curves for Chinook salmon and steelhead trout spawning depth and velocity; (2) develop detailed HSI curves for Chinook salmon and steelhead trout spawning substrate size; (3) develop detailed HSI curves for Chinook salmon and steelhead trout maximum movable substrate size; (4) compare spawning substrate size preferences to maximum movable substrate sizes; (5) evaluate the relative suitability of the 2008–2010 gravel placements at Sailor Bar on the LAR in terms of both spawning substrate size preferences and maximum movable substrate sizes; (6) evaluate the effects of substrate size on the size of spawning fish and redd characteristics (i.e., pocket depth, size, and tailspill); and (7) evaluate the relative importance of cover features (i.e., overhanging vegetation, brush, and riparian grass) and discrete channel unit types (i.e., main channel, side-channel, and island) on spawning habitat use. Specifically, we tested the following hypotheses:

*$1H_0$: There is no relationship between Chinook salmon and steelhead trout redd locations and depth; additionally, there is no significant difference between average depths at Chinook salmon and steelhead trout redd locations.*

*$2H_0$: There is no relationship between Chinook salmon and steelhead trout redd locations and velocity; additionally, there is no significant difference between average velocities at Chinook salmon and steelhead trout redd locations.*

*$3H_0$: There is no relationship between Chinook salmon and steelhead trout redd locations and substrate size; additionally, there is no significant difference between average substrate sizes at Chinook salmon and steelhead trout redd locations.*

*$4H_0$: There is no relationship between Chinook salmon and steelhead trout populations spawning in the LAR and maximum movable substrate size; additionally, there is no significant difference between average maximum movable substrate sizes calculated for Chinook salmon and steelhead trout.*

*$5H_0$: There is no significant difference between average substrate sizes at Chinook salmon and steelhead trout redd locations and average maximum movable substrate sizes calculated for Chinook salmon and steelhead trout populations spawning in the LAR.*

*$6H_0$: There is no significant relationship between substrate size at Chinook salmon and steelhead trout redd locations and Chinook salmon and steelhead trout size.*

*$7H_0$: There is no significant relationship between substrate size at Chinook salmon and steelhead trout redd locations and Chinook salmon and steelhead trout redd characteristics (i.e., pocket depth, size, and tailspill).*

We attempted to test four additional hypotheses. However, limited observations or limited data on habitat availability prevented any specific statistical tests:

*$8H_0$: There is no significant difference between average Chinook salmon and steelhead trout spawning substrate size preferences and substrate sizes used for the 2008–2010 gravel augmentation sites at Sailor Bar on the LAR.*

*$9H_0$: There is no significant difference between average maximum movable substrate sizes calculated for Chinook salmon and steelhead trout and substrate sizes used for the 2008–2010 gravel augmentation sites at Sailor Bar on the LAR.*

*$10H_0$: There is no significant difference between the numbers or density of salmonid redds within habitats affected by defined cover features (i.e., overhanging vegetation, brush, and riparian grass).*

*$11H_0$: There is no significant difference between the numbers or density of salmonid redds within habitats affected by defined channel unit features (i.e., main channel, side-channel, and island).*

*Methods (spawning surveys)* – We conducted Chinook salmon and steelhead trout spawning surveys at the 2008–2010 augmentation sites on a roughly bi-weekly basis from 20 October 2010 through 9 March 2011 (Table 2). Differentially corrected GPS coordinates were collected for individual redds using a Trimble GeoXT unit. Additional data collected for each redd observation included: (1) species; (2) redd width (m); (3) redd length (m); (4) tailspill length (m); (5) tailspill velocity ($m \bullet s^{-1}$); (6) pocket depth (m); (7) length(s) of any fish present on the redd; (8) nose velocity ($m \bullet s^{-1}$); (9) nose depth (m); and (10) redd age (i.e., new, new fish on, and old obscure). A Trimble data dictionary developed specifically for spawning surveys was used to log all additional data at the time GPS coordinates were collected for individual redds. Coordinates and accompanying data for individual redds were input into ArcMap and used to display the spatial extent of spawning throughout the project reach and develop .shp files containing the location and attributes of each redd observation.

In addition to these surveys, Reclamation has been conducting annual steelhead trout redd surveys since 2001–2002. For steelhead trout redd surveys, Reclamation uses boat, snorkel, and walking surveys to locate and GPS redds. Surveys typically occur during spring (January–April), and cover the entire LAR from Nimbus Dam downstream to Paradise Beach (~29 km). Additional data collected for each redd observation typically includes: (1) depth (m) surrounding the redd; (2) length(s) of any fish present on the redd; (3) velocity ($m \bullet s^{-1}$) taken at the nose of the redd; (4) pot (i.e., pocket) length (m); (5) pot width (m); (6) pot depth (m); (7) tailspill length (m); (8) tailspill width (m) measured at two different locations; (9) distance to cover (m); (10) background substrate diameter (mm); and (11) redd age (i.e., new, new fish on, and old obscure). A Trimble data dictionary developed specifically for steelhead trout redd surveys is typically used to log all additional data at the time GPS coordinates are collected for individual redds. Coordinates and accompanying data for individual redds are input into ArcMap and used to display the spatial extent of spawning throughout the project reach and develop .shp files containing the location and attributes of each redd observation.

*Methods (GIS-based habitat mapping)* – To map cover features (i.e., overhanging vegetation, brush, and riparian grass), we conducted GIS-based cover surveys at the 2008–2010 augmentation sites and areas adjacent to the 2008–2010 augmentation sites on six separate dates during the 2010–2011 monitoring season (Table 2). A Trimble GeoXT unit was used to collect polyline or point data for discrete patches of cover. Polylines were used to outline large patches of cover, whereas points were collected for small patches of cover. Field notebooks were used to keep detailed notes on each cover feature collected. Notes were tied to each cover feature based on the feature type (i.e., polyline or point) and the numbered order in which each feature was collected (i.e., 1, 2, or 3). Notes varied based on who was collecting cover data, but generally included descriptions of: (1) type (i.e., large woody material, small woody material, overhanging vegetation, terrestrial grass, terrestrial brush, and riparian vegetation); (2) species (i.e., willow or alder); (3) size (i.e., overhanging vegetation ≤1.0 m from the water surface, <10 cm diameter small woody material, or ≥10 cm diameter large woody material); and (4) whether or not cover features were submerged. Additional notes taken for point features included descriptions of size and orientation used to develop polygons for later mapping (see below). Prior to analysis, field notes were used to assign all cover features one or a combination of multiple cover categories from a list of seven total categories which covered the range of cover types observed (Table 3). Cover surveys were conducted within the wetted channel and areas of adjacent floodplain which we believed could become wetted channel throughout the course of the monitoring season.

***Table 2**. Cover categories and descriptions used for GIS-based habitat mapping at Sailor Bar, lower American River, California. Cover features were assigned one or a combination of multiple categories.*

| Cover Type | Description |
|---|---|
| Brush | Living woody material <10 cm diameter (generally blackberries, willows, and alders with instream and overhead cover components) |
| Trees | Living woody material ≥10 cm diameter (generally willows, alders, and larger riparian tree species with instream and overhead cover components) |
| Small Woody Material (SWM) | Non-living woody material <10 cm diameter (classic SWM with instream cover component) |
| Large Woody Material (LWM) | Non-living woody material ≥10 cm diameter (classic LWM with instream cover component) |
| Riparian Grass | Living or non-living riparian grass, sedge, and reed species (generally with instream cover component) |
| Cattail | Classic stand of cattails (generally with instream and overhead cover components) |
| Overhanging Vegetation | Living or non-living woody material overhanging water ≤1.0 m from surface (generally large trees with overhead cover component) |

To map substrate features (i.e., patches of homogenous substrate), GIS-based substrate surveys were conducted concurrent with cover surveys (see above). A Trimble GeoXT unit was used to collect polyline features outlining discrete substrate patches. Standardized field data sheets were then used to collect pebble count data from within each outlined patch. Pebble counts were conducted using methods similar to those of Merz et al. (2006), whereby substrate samples (i.e., individual pebble or piece of gravel) are collected every ~0.3 m along transects running the length of each outlined patch, and a pre-cut template is used to measure substrate size and categorize each sample into 12 size classes (Table 4). Size classes are based on the largest slot (i.e., round hole with specified diameter) through which an individual substrate sample will not pass (Merz et al. 2006). The number of substrate samples measured within each outlined patch varied with patch size. However, ≥25 samples were measured within all patches and ≥100 samples were measured within larger patches. Pebble count data sheets were tied to each substrate feature using similar methods to those used for cover features (see above). Substrate surveys were conducted within the wetted channel and areas of adjacent floodplain which we believed could become wetted channel throughout the course of the monitoring season.

***Table 3**. Substrate size categories and descriptions used for GIS-based habitat mapping at Sailor Bar, lower American River, California (Adapted from Bunte and Abt 2001).*

| Substrate Size Class (mm) | Description |
|---|---|
| <4 | Very fine gravel, sand, and silt |
| 8 | Fine gravel |
| 16 | Medium gravel |
| 22 | Coarse gravel |
| 32 | |
| 44 | Very coarse gravel |
| 64 | |
| 89 | Small cobble |
| 127 | Medium cobble |
| 178 | Large cobble |
| 254 | Boulder |
| 610 | Bedrock |

Due to high flows throughout the 2010–2011 monitoring season, substrate surveys could not be conducted as described above for all substrate patches. Therefore, we employed underwater videography to complete substrate surveys for patches in which water depths were too deep for traditional methods (see above). First, a jet-boat and Trimble GeoXT unit were used to collect polyline features outlining discrete substrate patches. Second, a jet boat equipped with a Lowrance LMS 520c sonar unit with GPS track capabilities, downrigger, SplashCam underwater video camera, DVR video recorder, and a GPS overlay unit was used to conduct video substrate surveys along transects running the length of each outlined patch. Third, pebble counts were conducted for each video transect, whereby the diameters of the first 10 substrate samples (i.e., individual pebble or piece of gravel) along a line transecting still shots taken at ≥10 evenly spaced locations throughout each video transect were measured and categorized based on the size classes described above (Table 4). The GPS overlay unit allowed us to display the precise location of each video still shot in the upper third of the frame to ensure that each still shot was located within outlined substrate patches. Two lasers attached to the underwater video camera and spaced at a known distance allowed us to use the ratio of each measured substrate sample to the measured known distance to calculate all substrate sample diameters (Figure 8). These methods allowed us to successfully merge pebble count data collected using underwater videography with pebble count data collected using more traditional methods (see above).


***Figure 8**. Example still shot taken during substrate surveys using underwater videography. Location of the still shot is displayed in the upper third of the frame. Lasers spaced at 6 inches (15.24 cm; circled in red) allowed us to use the ratio of each measured substrate sample to the measured known distance to calculate all substrate sample diameters. A horizontal transect line would be placed across the middle of each still shot, and the diameters of the first 10 substrate samples (i.e., individual pebble or piece of gravel) along the line would be measured.*

To develop GIS-based habitat maps, polyline and point features for both cover and substrate surveys were converted to .shp files and imported into ArcMap. ArcMap was then used to develop separate polygon .shp files for both cover and substrate. For cover, polylines outlining discrete patches were used for tracing and hand-digitizing polygons, whereas points and accompanying descriptions of size and orientation were used for buffering points to a specified distance or hand-digitizing polygons. Cover categories from the list of seven total categories (see above) were assigned to each final cover polygon in the attribute table of the cover feature .shp file (Figures 9 and 10). For substrate, polylines outlining discrete patches were used for tracing and hand-digitizing polygons. Pebble counts from each substrate size category (see above) were assigned to each final substrate polygon in the attribute table of the substrate feature .shp file (Figures 9 and 10). No further data summarization was needed for the cover feature .shp file. However, substrate $D_{15}$s, $D_{50}$s, $D_{85}$s, and $D_{95}$s were calculated for each polygon in the substrate feature .shp file, along with both the relative (i.e., individual category) and cumulative (i.e., up to and including individual category) percent (%) of substrate in each size class. Additionally, each polygon in the substrate feature .shp file was assigned one of six channel unit categories based on channel unit types observed within the survey area: (1) Floodplain; (2) Island; (3) Main Channel; (4) Main Channel/Floodplain; (5) Scallops; and (6) Side-Channel.

Prior to analysis, overlay procedures in ArcMap were used to identify and assign the closest cover type to each redd observation based on a 5.06 m search radius for Chinook salmon (i.e., closest cover type within 5.06 m; based on mid-point of 1.0 sec. burst speed ranges provided by Mills et al. 2003 for Columbia River Chinook salmon) and a 6.13 m search radius for steelhead trout (i.e., closest cover type within 6.13 m; based on mid-point of 1.0 sec. burst speed ranges provided by Bell 1973). Similar overlay procedures were used to assign pebble counts to each redd observation based on exact location (i.e., point of observation). Channel unit types were assigned to each redd observation based on all channel unit types within the species-specific search radius (i.e., multiple channel unit types assigned to observations; see above).

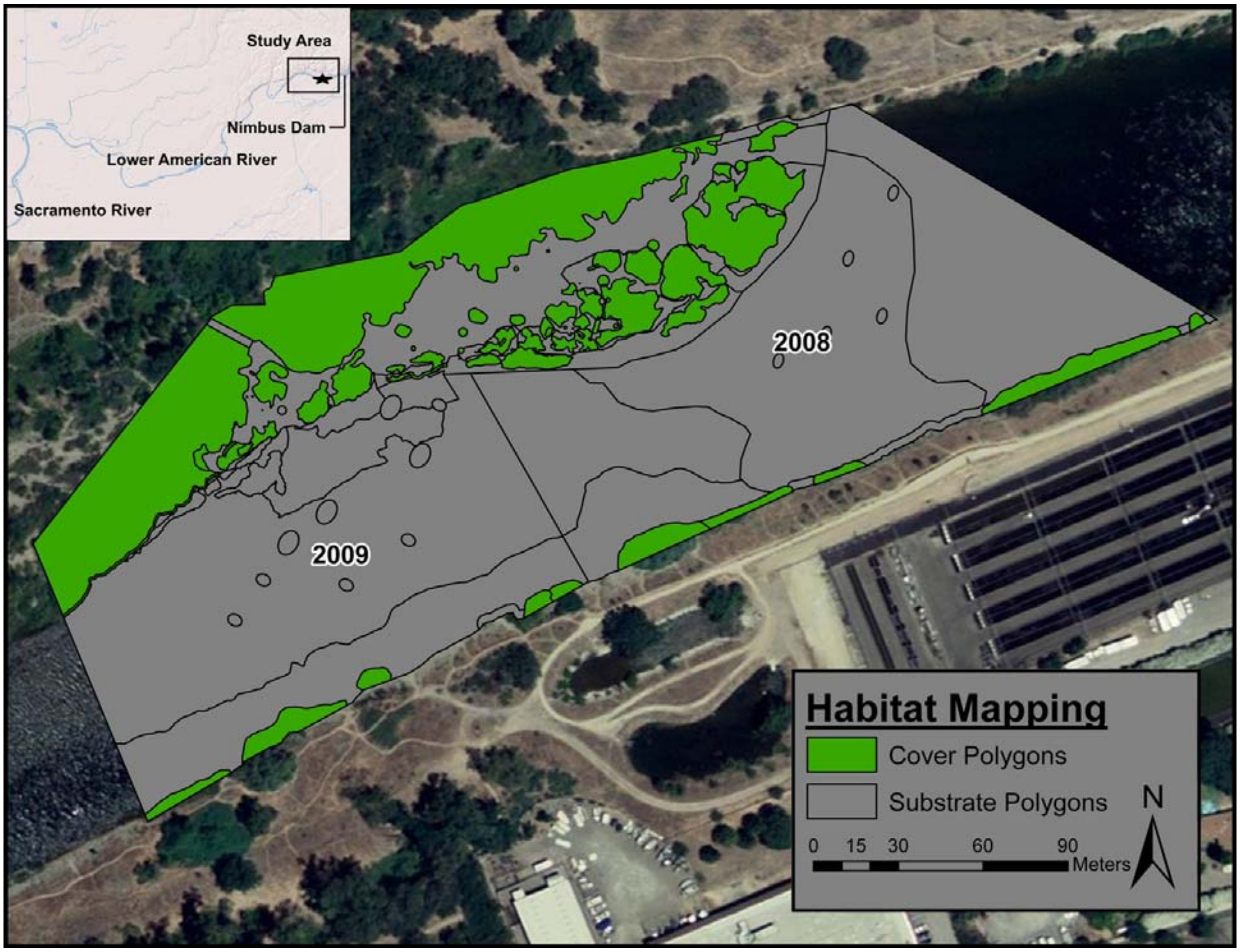


***Figure 9****. GIS-based habitat map developed for the 2008–2009 augmentation sites and floodplain adjacent to the 2008–2009 augmentation sites.*

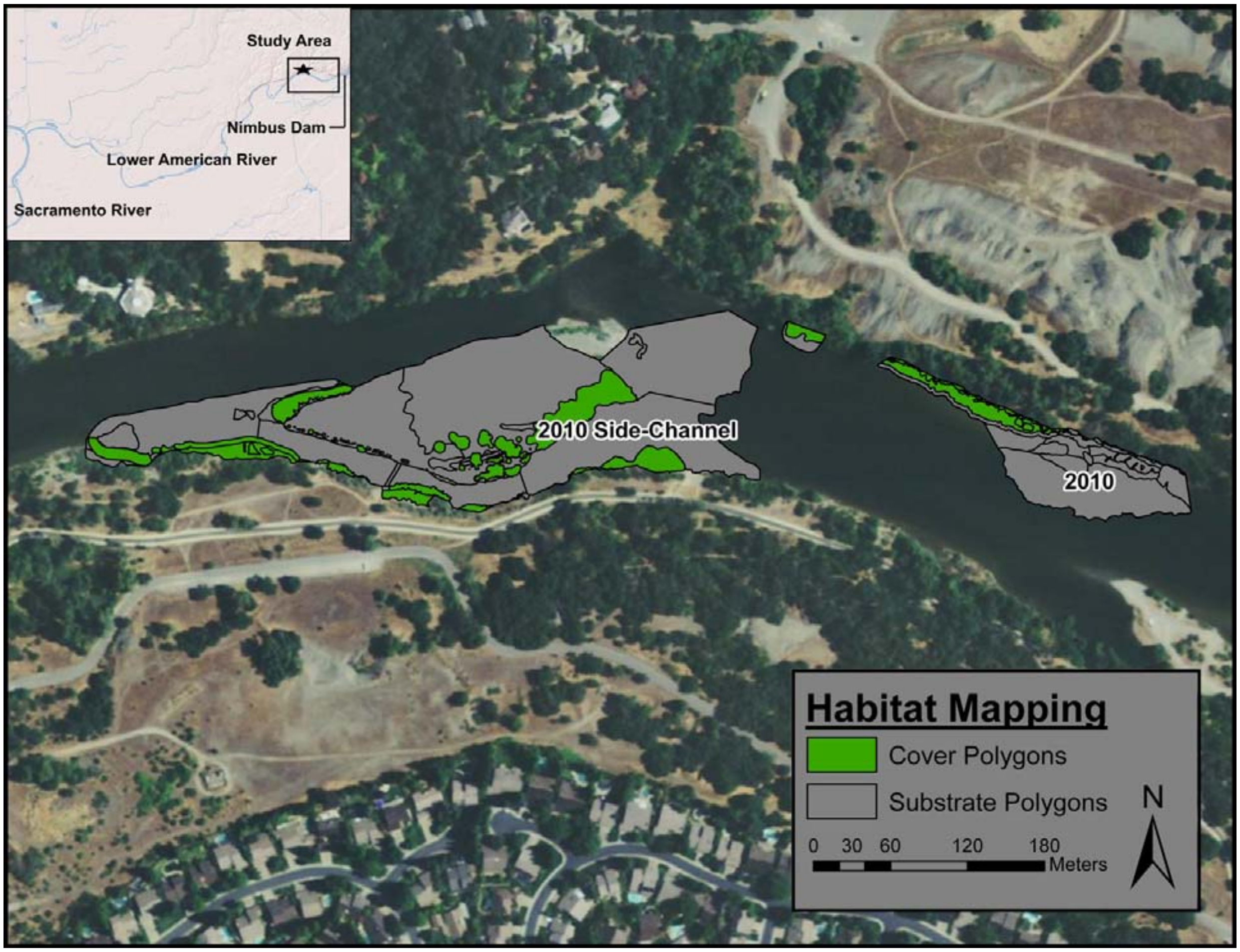


***Figure 10**. GIS-based habitat map developed for the 2010 augmentation site and side-channel island adjacent to the 2010 augmentation site. Cover features on the side-channel island adjacent to the 2010 augmentation site were only partially mapped.*

*Methods and Analysis (depth and velocity)* – For Chinook salmon, a total of 75 redd observations (all from 2010–2011 monitoring) were used to calculate depth and velocity preferences. Nose depth (m) was only available for 16 of the 75 observations. However, pocket depth was available for all 75 observations. Therefore, to develop HSI curves for Chinook salmon spawning depth, we used pocket depth after subtracting the average difference between pocket depth and nose depth for redd observations for which both pocket depth and nose depth were available (16 observations). Nose velocity ($m \cdot s^{-1}$) was taken at 0.6 depth below the water surface when depth was ≤0.8 m and at 0.2 and 0.8 depth below the water surface when depth was >0.8 m. At least one nose velocity was available for all 75 observations. However, two nose velocities were available for 36 of the 75 observations. Therefore, to develop HSI curves for Chinook salmon spawning velocity, we used the average of all available velocities for each redd observation. All depths and velocities were collected during the course of normal bi-weekly spawning surveys (see above).

For steelhead trout, a total of 42 redd observations (all from 2010–2011 monitoring) were used to calculate depth and velocity preferences. Depth (m) surrounding each redd was available for all 42 observations, and was used to develop HSI curves for steelhead trout spawning depth.

Velocity (m•s$^{-1}$) was taken using methods similar to those used for Chinook salmon redds (see above). At least one velocity was available for all 42 observations. However, two velocities were available for 32 of the 42 observations. Therefore, to develop HSI curves for steelhead trout spawning velocity, we used the average of all available velocities for each redd observation. All depths and velocities were collected during the course of normal spawning surveys conducted by Reclamation (see above).

All redd observations were assigned to depth and velocity categories based on 0.1 m increments (range = 0.0 to >2.0 m). An iterative, locally-weighted least-squares (i.e., LOWESS) method (tension = 0.25) was then used to smooth the values assigned to each category. Smoothed values for each category were then used to calculate depth and velocity HSI values by scaling values for each category to the overall maximum value. Additionally, an analysis of variance (ANOVA) with a Tukey test (Zar 1999) was used to test for differences in average depth and velocity preferences among species (i.e., Chinook salmon and steelhead trout). A separate ANOVA was used for each comparison (i.e., depth and velocity). Further, we performed a chi-square analysis:

$$X^2 = \sum \frac{(X_a - X_b)^2}{X_b} \tag{3}$$

Where $X_b$= the expected number of Chinook salmon or steelhead trout redds above or below the average depth or velocity calculated from pooled data (i.e., Chinook salmon and steelhead trout combined) assuming equal distribution above and below the average/total redds above or below the average depth or velocity calculated from pooled data, and $X_a$ = the observed number of Chinook salmon or steelhead trout redds above or below the average depth or velocity calculated from pooled data/total redds above or below the average depth or velocity calculated from pooled data (Zar 1999). Separate chi-square analyses were conducted for each comparison (i.e., depth and velocity). These analyses allowed us to explicitly test whether spawning depth and velocity preferences differed among species, and determine which species selected deeper or shallower depths and faster or slower velocities.

*Methods and Analysis (substrate size)* – For Chinook salmon, a total of 252 redd observations (all from 2010–2011 monitoring) were used to calculate substrate size preferences, whereas only 49 redd observations (all from 2010–2011 monitoring) were used to calculate substrate size preferences for steelhead trout. All redd observations were assigned pebble counts based on location and ArcMap overlay procedures (see above), and pebble count $D_{50}$s and $D_{85}$s were used to assign all redd observations to substrate size categories based on 10 mm increments (range = 0.0 to >150.0 mm). An iterative, locally-weighted least-squares (i.e., LOWESS) method (tension = 0.25) was then used to smooth the values assigned to each category. Smoothed values for each category were then used to calculate substrate size HSI values by scaling values for each category to the overall maximum value. Additionally, an ANOVA with a Tukey test (Zar 1999) was used to test for differences in average substrate size preferences among species (i.e., Chinook salmon and steelhead trout). A separate ANOVA was used for each comparison (i.e., $D_{50}$ and $D_{85}$). Further, we performed a chi-square analysis:

$$X^2 = \sum \frac{(X_a - X_b)^2}{X_b} \tag{4}$$

Where $X_b$ = the expected number of Chinook salmon or steelhead trout redds above or below the average $D_{50}$ and $D_{85}$ calculated from pooled data (i.e., Chinook salmon and steelhead trout combined) assuming equal distribution above and below the average/total redds above or below the average $D_{50}$ and $D_{85}$ calculated from pooled data, and $X_a$ = the observed number of Chinook salmon or steelhead trout redds above or below the average $D_{50}$ and $D_{85}$ calculated from pooled data/total redds above or below the average $D_{50}$ and $D_{85}$ calculated from pooled data (Zar 1999). Separate chi-square analyses were conducted for each comparison (i.e., $D_{50}$ and $D_{85}$). These analyses allowed us to explicitly test whether spawning $D_{50}$ and $D_{85}$ preferences differed among species, and determine which species selected larger or smaller substrate sizes. Substrate size preferences were also qualitatively compared to gravel placed at the 2008–2010 augmentation sites by comparing placed gravel $D_{50}$s to average $D_{50}$s for Chinook salmon and steelhead trout redd observations.

Kondolf (2000) describes a method for determining appropriate spawning substrate size for a salmonid population, whereby the maximum movable substrate size for spawning females is defined as 10% of the average female length. Therefore, we assumed that maximum movable substrate size for an individual fish was 10% of the individual's FL (i.e., maximum movable substrate size = 0.10 × FL), and used historic (2004–2005 through 2007–2008 for Chinook salmon and 2003–2004 through 2006–2007 for steelhead trout) FL data provided by Reclamation and California Department of Fish and Game (CDFG) to calculate population-level (i.e., all Chinook salmon and steelhead trout – males and females) maximum movable substrate size preferences for both Chinook salmon and steelhead trout. For Chinook salmon, a total of 10,411 FL observations were available for analysis, whereas only 364 FL observations were available for steelhead trout. All observations were converted to appropriate units of measure and multiplied by a factor of 0.10 to obtain 10% FL maximum movable substrate sizes. Maximum movable substrate sizes were then assigned to substrate size categories identical to those used for substrate size preferences (see above). An iterative, locally-weighted least-squares (i.e., LOWESS) method (tension = 0.25) was then used to smooth the values assigned to each category. Smoothed values for each category were then used to calculate maximum movable substrate size HSI values by scaling values for each category to the overall maximum value.

An ANOVA with a Tukey test (Zar 1999) was used to test for differences in average maximum movable substrate size preferences among species (i.e., Chinook salmon and steelhead trout), and to test for species-specific differences in average maximum movable substrate size preferences and both average $D_{50}$ and $D_{85}$ calculated from redd observations. A separate ANOVA was used for each comparison (i.e., one to compare Chinook salmon to steelhead trout maximum movable substrate sizes and four to compare species-specific maximum movable substrate sizes to species-specific $D_{50}$ and $D_{85}$). Additionally, we performed a chi-square analysis:

$$X^2 = \sum \frac{(X_a - X_b)^2}{X_b} \tag{5}$$

Where: (1) $X_b$ = the expected number of Chinook salmon or steelhead trout above or below the average maximum movable substrate size calculated from pooled data (i.e., Chinook salmon and steelhead trout combined) assuming equal distribution above and below the average/total fish above or below the average maximum movable substrate size calculated from pooled data, and $X_a$ = the observed number of Chinook salmon or steelhead trout above or below the average maximum movable substrate size calculated from pooled data/total fish above or below the average maximum movable substrate size calculated from pooled data; (2) $X_b$= the expected number of Chinook salmon (i.e., maximum movable substrate size or $D_{50}$ and $D_{85}$) above or below the average substrate size calculated from pooled data (i.e., $D_{50}$ and $D_{85}$ and maximum movable substrate size data combined for Chinook salmon only) assuming equal distribution above and below the average/total Chinook salmon above or below the average substrate size calculated from pooled data, and $X_a$ = the observed number of Chinook salmon (i.e., maximum movable substrate size or $D_{50}$ and $D_{85}$) above or below the average substrate size calculated from pooled data/total Chinook salmon above or below the average substrate size calculated from pooled data; and (3) $X_b$= the expected number of steelhead trout (i.e., maximum movable substrate size or $D_{50}$ and $D_{85}$) above or below the average substrate size calculated from pooled data (i.e., $D_{50}$ and $D_{85}$ and maximum movable substrate size data combined for steelhead trout only) assuming equal distribution above and below the average/total steelhead trout above or below the average substrate size calculated from pooled data, and $X_a$ = the observed number of steelhead trout (i.e., maximum movable substrate size or $D_{50}$ and $D_{85}$) above or below the average substrate size calculated from pooled data/total steelhead trout above or below the average substrate size calculated from pooled data (Zar 1999). Separate chi-square analyses were conducted for each comparison (i.e., one to compare Chinook salmon to steelhead trout maximum movable substrate sizes and four to compare species-specific maximum movable substrate sizes to species-specific $D_{50}$ and $D_{85}$). These analyses allowed us to explicitly test whether: (1) maximum movable substrate size preferences differed among species, and determine which species selected larger or smaller substrate sizes (i.e., would be limited by larger or smaller substrate sizes); (2) Chinook salmon maximum movable substrate size preferences differed from selected $D_{50}$ and $D_{85}$, and determine whether maximum movable substrate sizes were larger or smaller than selected $D_{50}$ and $D_{85}$ (i.e., compare maximum movable substrate sizes to substrate sizes at redd observations); and (3) steelhead trout maximum movable substrate size preferences differed from selected $D_{50}$ and $D_{85}$, and determine whether maximum movable substrate sizes were larger or smaller than selected $D_{50}$ and $D_{85}$ (i.e., compare maximum movable substrate sizes to substrate sizes at redd observations). Maximum movable substrate size preferences were also qualitatively compared to gravel placed at the 2008–2010 augmentation sites by comparing placed gravel $D_{50}$s and $D_{85}$s to average maximum movable substrate sizes for Chinook salmon and steelhead trout. Further, we developed cumulative maximum movable substrate size distributions for both Chinook salmon and steelhead trout, which were then compared to placed gravel $D_{95}$s to calculate a proportion of each population that would be excluded from spawning at each of the 2008–2010 augmentation sites; assuming maximum movable substrate size must be larger than placed gravel $D_{95}$s for successful spawning.

Because maximum movable substrate size is correlated to fish length (Kondolf 2000), average substrate sizes may largely determine the size range of salmonids which can spawn within discrete substrate patches. Smaller substrates may be accessible to spawning fish of all sizes, whereas larger substrates may be accessible to larger fish and inaccessible to smaller fish. Therefore, we used FL data collected concurrently with redd observations (see above) to test the effects of substrate size on the size of spawning Chinook salmon, steelhead trout, and total salmonids (i.e., data for Chinook salmon and steelhead trout combined). For Chinook salmon, a total of 35 FL observations were available for analysis, whereas 31 FL observations were available for steelhead trout, thereby resulting in a total of 66 FL observations available for total salmonids.

Initially, all FL values were plotted against substrate size data (i.e., $D_{50}$ and $D_{85}$) to determine the appropriate method of analysis. Substrate sizes were highly categorized (i.e., most spawning occurred in a few substrate patches, thereby resulting in highly categorized substrate data). Additionally, FL observations appeared to be uncorrelated or only weakly correlated to substrate size, with only one or two observations driving relationships at larger substrate sizes. Therefore, we used $D_{50}$s and $D_{85}$s associated with FL observations to calculate average substrate sizes (both $D_{50}$ and $D_{85}$) used by Chinook salmon, steelhead trout, and total salmonids. Average substrate sizes were then used to partition FL observations into "ABOVE" and "BELOW" average categories based on substrate size, and an ANOVA with a Tukey test (Zar 1999) was used to test for species-specific (i.e., Chinook salmon and steelhead trout) and combined (i.e., total salmonids) differences in average FLs among categories (i.e., "ABOVE" and "BELOW" average substrate size for both $D_{50}$ and $D_{85}$). A separate ANOVA was used for each comparison (i.e., two each for Chinook salmon, steelhead trout, and total salmonids). Following ANOVA tests, average FLs were calculated for Chinook salmon, steelhead trout, and total salmonids. Average FLs were then used to partition FL observations into "ABOVE" and "BELOW" average categories using methods identical to those used for substrate size. This resulted in one partition of FL observations based on substrate size and one partition of FL observations based on FL values, with a total of four combinations of both substrate size categories and FL categories for each FL observation: (1) Substrate "ABOVE" and FL "ABOVE"; (2) Substrate "ABOVE" and FL "BELOW"; (3) Substrate "BELOW" and FL "ABOVE"; and (4) Substrate "BELOW" and FL "BELOW". All FL observations were assigned to each of the four categories as appropriate (see above), and observed values in each of the four combination categories were tested for significant deviations from expected values using a chi-square analysis:

$$X^2 = \sum \frac{(X_a - X_b)^2}{X_b} \tag{6}$$

Where $X_b$ = the expected number of species-specific or combined FL observations in each category calculated from pooled data assuming all FL observations are equally distributed/total FL observations in each category, and $X_a$ = the observed number of species-specific or combined FL observations in each category/total FL observations in each category (Zar 1999). Separate chi-square analyses were conducted for Chinook salmon $D_{50}$ and $D_{85}$, steelhead trout $D_{50}$ and $D_{85}$, and total salmonid $D_{50}$ and $D_{85}$ (i.e., two chi-squares each for three salmonid categories, thereby resulting in six total chi-squares). These analyses allowed us to explicitly test whether the average size of spawning salmonids differed among substrate size categories, and determine whether larger fish were associated with larger substrate sizes and *vice versa*. For chi-square analyses, we assumed that significantly greater than expected proportions of "ABOVE" average FL fish in "ABOVE" average substrate and "BELOW" average FL fish in "BELOW" average substrate indicated a significant relationship between substrate size and fish size. Conversely, we assumed that no significant differences from expected proportions indicated no significant relationship between substrate size and fish size.

*Methods and analysis (redd characteristics)* – Because maximum movable substrate size is correlated to fish length (Kondolf 2000), average substrate sizes may largely determine the size range of salmonids which can spawn within discrete substrate patches. Smaller substrates may be accessible to spawning fish of all sizes, whereas larger substrates may be accessible to larger fish and inaccessible to smaller fish (see above). However, in systems which are highly spawning substrate limited (i.e., the LAR), substrate size may not limit spawning use. In these systems, spawning salmonids may be forced to utilize substrates which are larger than optimal (i.e., near or exceeding 10% FL maximum movable substrate size). In this situation, substrate size may influence redd characteristics (i.e., length, width, and tailspill) by limiting redd size, thereby leading to sub-optimal conditions for embryo incubation and development, which could lead to limited reproductive success (Kondolf 2000). Therefore, we used data collected concurrently with redd observations (see above) to test the effects of substrate size on: (1) redd length (Chinook salmon, steelhead trout, and total salmonids); (2) redd width (Chinook salmon, steelhead trout, and total salmonids); (3) redd depth (i.e., pocket depth; steelhead trout); (4) tail length (Chinook salmon, steelhead trout, and total salmonids); and (5) tail width (steelhead trout). For Chinook salmon, a total of 76 observations were available for redd length, whereas 75 observations were available for both redd width and tail length. For steelhead trout, a total of 42 observations were available for redd length, redd width, tail length, and tail width, whereas 36 observations were available for redd depth. For total salmonids, a total of 118 observations were available for redd length, whereas 117 observations were available for both redd width and tail length. In general, both Chinook salmon and steelhead trout redd survey methods (see above) included one measurement for redd length, redd width, redd depth, and tail length, whereas two measurements were included for tail width. Therefore, single measurements were used for analysis whenever possible, and average measurements were used for analysis when required (i.e., tail width).

Initially, values for all redd characteristics were plotted against substrate size data (i.e., $D_{50}$ and $D_{85}$) to determine the appropriate method of analysis. Substrate sizes were highly categorized (i.e., most spawning occurred in a few substrate patches, thereby resulting in highly categorized substrate data). For Chinook salmon, most redd characteristics appeared to be uncorrelated or only weakly correlated to substrate size, with only one or two observations driving relationships at larger substrate sizes. For steelhead trout, most redd characteristics appeared to be correlated to substrate size. However, only one or two observations appeared to drive relationships at larger substrate sizes. Relationships for total salmonids were similar to those for Chinook salmon. Therefore, we used $D_{50}$ and $D_{85}$ values associated with redd characteristics to calculate average substrate sizes (both $D_{50}$ and $D_{85}$) used by Chinook salmon, steelhead trout, and total salmonids. Average substrate sizes were then used to partition values for redd characteristics into "ABOVE" and "BELOW" average categories based on substrate size, and an ANOVA with a Tukey test (Zar 1999) was used to test for species-specific (i.e., Chinook salmon and steelhead trout) and combined (i.e., total salmonids) differences in average values for redd characteristics among categories (i.e., "ABOVE" and "BELOW" average substrate size for both $D_{50}$ and $D_{85}$ values). A separate ANOVA was used for each comparison (i.e., six for Chinook salmon, 10 for steelhead trout, and six for total salmonids – i.e., three, five, and three comparisons for both $D_{50}$ and $D_{85}$ values). Following ANOVA tests, average values for each redd characteristic were calculated for Chinook salmon, steelhead trout, and total salmonids. Average values were then used to partition redd characteristics into "ABOVE" and "BELOW" average categories using methods identical to those used for substrate size. This resulted in one partition of redd characteristic observations based on substrate size and one partition of redd characteristic observations based on values for each redd characteristic, with a total of four combinations of both substrate size categories and redd characteristic categories for each redd characteristic observation: (1) Substrate "ABOVE" and Redd Characteristic "ABOVE"; (2) Substrate "ABOVE" and Redd Characteristic "BELOW"; (3) Substrate "BELOW" and Redd Characteristic "ABOVE"; and (4) Substrate "BELOW" and Redd Characteristic "BELOW". All redd characteristic observations were assigned to each of the four categories as appropriate (see above), and observed values in each of the four combination categories were tested for significant deviations from expected values using a chi-square analysis:

$$X^2 = \sum \frac{(X_a - X_b)^2}{X_b} \tag{7}$$

Where $X_b$= the expected number of species-specific or combined redd characteristic observations in each category calculated from pooled data assuming all redd characteristic observations are equally distributed/total redd characteristic observations in each category, and $X_a$ = the observed number of species-specific or combined redd characteristic observations in each category/total redd characteristic observations in each category (Zar 1999). Separate chi-square analyses were conducted for Chinook salmon $D_{50}$ and $D_{85}$, steelhead trout $D_{50}$ and $D_{85}$, and total salmonid $D_{50}$ and $D_{85}$ (i.e., two chi-squares each for three Chinook salmon redd categories, five steelhead trout redd categories, and three total salmonid redd categories, thereby resulting in 22 total chi-squares). These analyses allowed us to explicitly test whether the average redd characteristics of spawning salmonids differed among substrate size categories, and determine whether smaller redds (measured using the above categories) were associated with larger substrate sizes and *vice versa*. For chi-square analyses, we assumed that significantly greater than expected proportions of "BELOW" average redd characteristics in "ABOVE" average substrate and "ABOVE" average redd characteristics in "BELOW" average substrate indicated a significant relationship between substrate size and redd size. Conversely, we assumed that no significant differences from expected proportions indicated no significant relationship between substrate size and redd size.

*Methods and analysis (habitat preferences)* – Limited data on the availability of cover features and discrete channel unit types, along with concern over interactions with confounding factors such as depth, velocity, and substrate, prevented any specific statistical tests designed to evaluate the influence of cover features and discrete channel unit types on salmonid spawning habitat preferences. Therefore, we provide a qualitative evaluation of habitat utilization in terms of both the number of redds (No. Redds) and utilization (%) rates associated with cover features and discrete channel unit types within the overall area outlined during GIS-based habitat mapping (see above). Overlay procedures in ArcMap were used to identify and assign the closest cover type to redd observations (see above) based on a 5.06 m search radius for Chinook salmon (i.e., closest cover type within 5.06 m; based on mid-point of 1.0 sec. burst speed ranges provided by Mills et al. 2003 for Columbia River Chinook salmon) and a 6.13 m search radius for steelhead trout (i.e., closest cover type within 6.13 m; based on mid-point of 1.0 sec. burst speed ranges provided by Bell 1973; see above), whereas channel unit types were assigned to each redd observation based on all channel unit types within the species-specific search radius (i.e., multiple channel unit types assigned to observations; see above). We used the largest redd data set available from 2010–2011 monitoring for all calculations (i.e., data set included all redd observations; including redds for which additional associated data were not collected). In total, 252 Chinook salmon and 49 steelhead trout redd observations were available for analysis. Both species-specific (i.e., Chinook salmon and steelhead trout) and combined (i.e., total salmonids) redd data are presented.

### *Juvenile Salmonid Habitat Preferences*

Depth, velocity, substrate, cover, and the discrete channel unit types that control the availability and interactions of these habitat features all influence the distribution and relative density of juvenile salmonids (Rosenfeld et al. 2000). Therefore, an important component of effectiveness monitoring is assessing fish habitat use in relation to habitat availability (i.e., habitat preferences), and the resulting effects of available habitat area and quality on juvenile density and carrying capacity within discrete stream reaches. The goals of this study were to use GIS-based habitat mapping and data collected during juvenile snorkel surveys to evaluate the relative importance of: (1) depth and velocity on rearing habitat use; (2) substrate size on rearing habitat use; and (3) cover features (i.e., overhanging vegetation, brush, and riparian grass) and discrete channel unit types (i.e., main channel, side-channel, and island) on rearing habitat use. Specifically, we tested the following hypotheses:

*1$H_0$:* *There is no relationship between juvenile Chinook salmon and steelhead trout snorkel observations and depth; additionally, there is no significant difference between average depths at juvenile Chinook salmon and steelhead trout snorkel observations.*

*2$H_0$:* *There is no relationship between juvenile Chinook salmon and steelhead trout snorkel observations and velocity; additionally, there is no significant difference between average velocities at juvenile Chinook salmon and steelhead trout snorkel observations.*

*3$H_0$:* *There is no relationship between juvenile Chinook salmon and steelhead trout snorkel observations and substrate size; additionally, there is no significant difference between average substrate sizes at juvenile Chinook salmon and steelhead trout snorkel observations.*

*4$H_0$:* *There is no significant difference between the numbers or density of juvenile Chinook salmon or steelhead trout observed within habitats affected by defined cover features (i.e., overhanging vegetation, brush, and riparian grass).*

*5$H_0$:* *There is no significant difference between the numbers or density of juvenile Chinook salmon or steelhead trout observed within habitats affected by defined channel unit features (i.e., main channel, side-channel, and island).*

*Methods (snorkel surveys)* – We established nine sample transects adjacent to the 2008–2010 augmentation sites for replicated snorkel surveys targeting juvenile Chinook salmon and steelhead trout (Figures 11 and 12). Sample transects were established along bank margins adjacent to the: (1) lower half of the 2008 site (2008 T1 – one transect); (2) upper half and middle of the 2009 site (2009 T1 and T2 – two transects); (3) scallops added to the 2010 site

(2010 T3 – one transect); (4) island added to the 2010 site (2010 T2 – one transect); (5) floodplain below the 2010 site (2010 T1 – one transect); and (6) middle and lower half of the island below the 2010 site (2010SC T1, T2, and T3 – three transects). The exact location of each sample transect was allowed to vary based on flow and water's edge, thereby allowing replicated measurements within discrete areas over time while still allowing a variety of depths, velocities, substrates, cover features, and channel unit types to be sampled. Snorkel surveys were conducted on a roughly bi-weekly basis from 10 February 2011 through 16 June 2011 (Table 2).

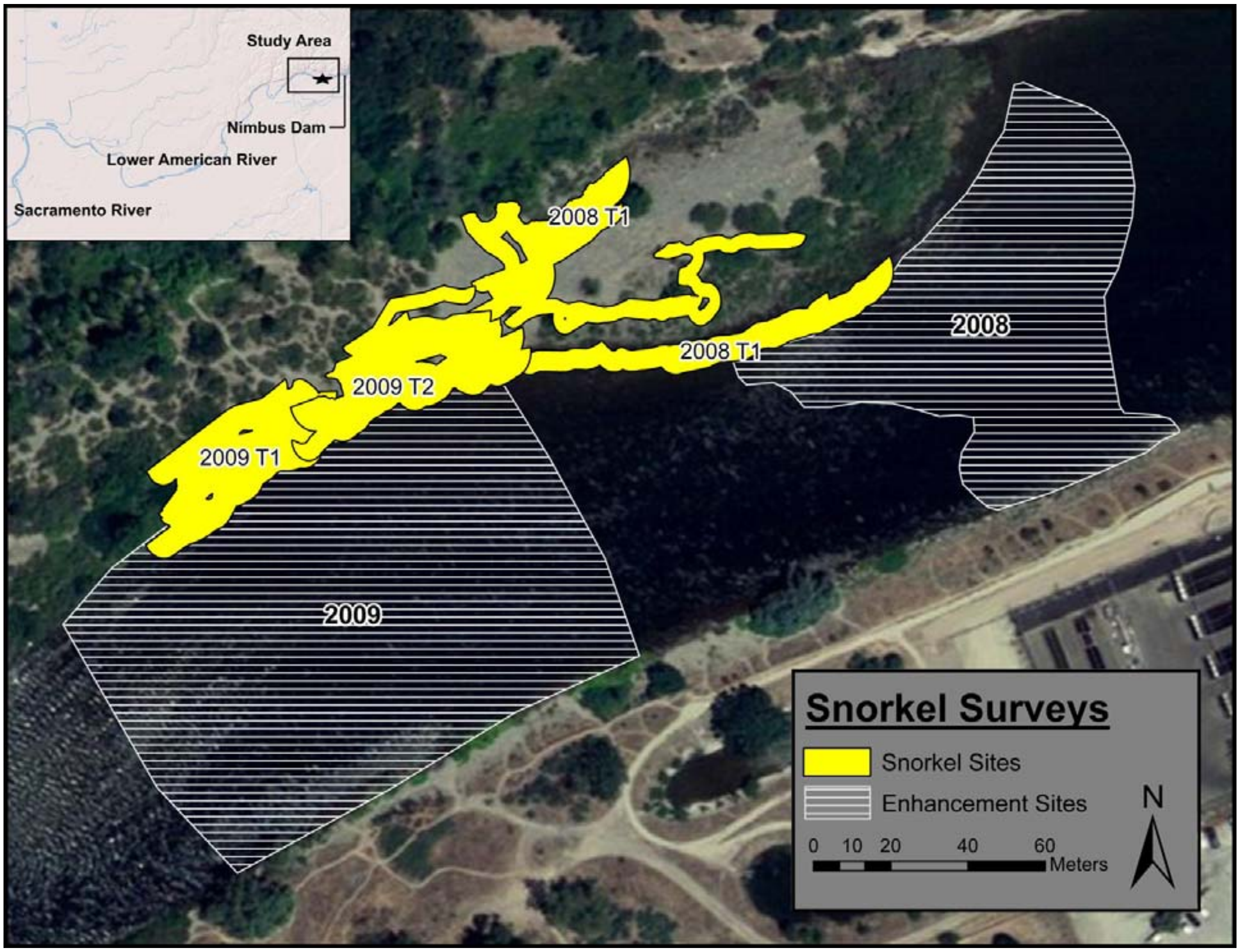


***Figure 11**. Approximate locations of snorkel survey transects (i.e., snorkel sites) adjacent to the 2008–2009 augmentation sites at Sailor Bar on the lower American River. The exact location of each transect varied based on flow and location of water's edge (see above).*

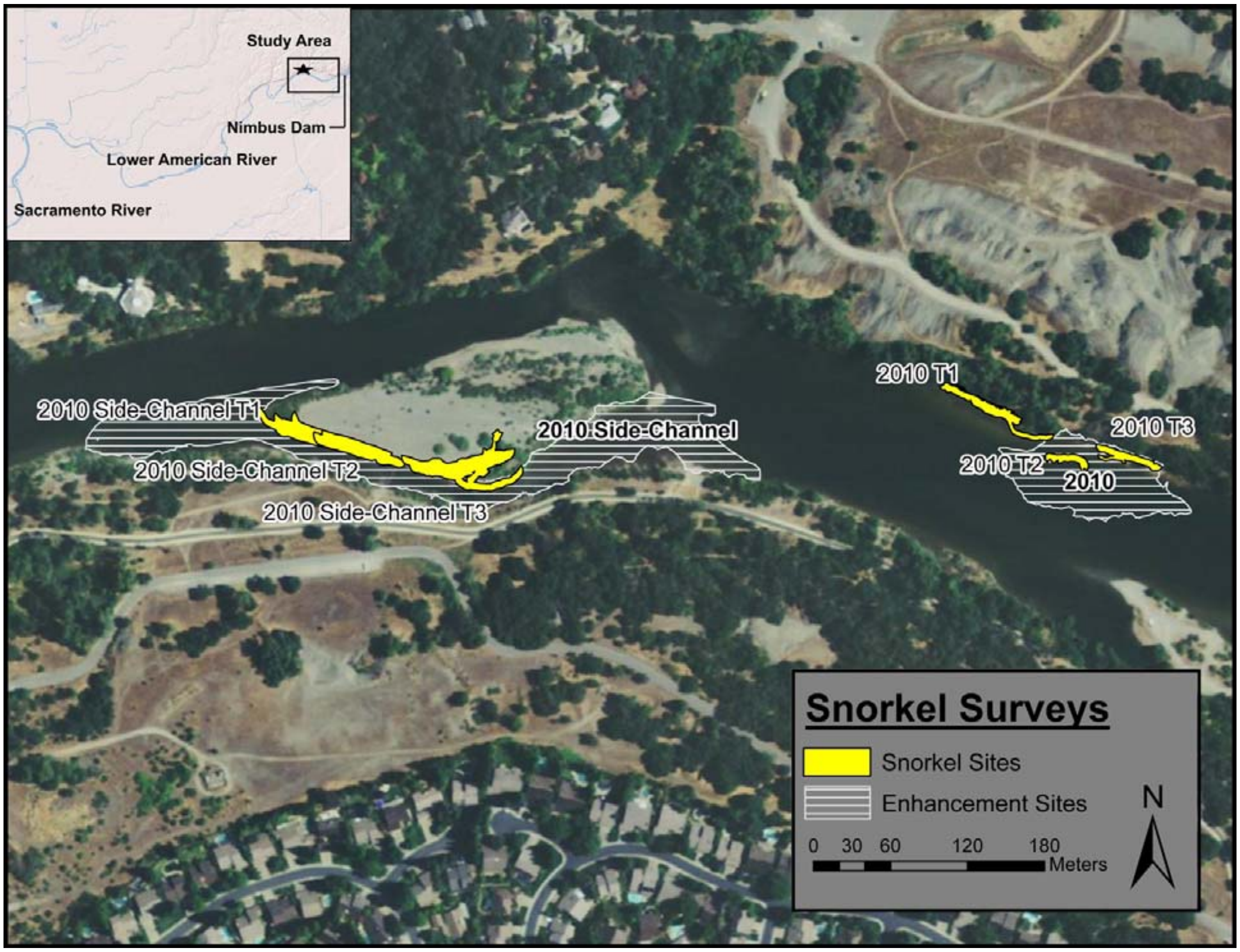


***Figure 12**. Approximate locations of snorkel survey transects (i.e., snorkel sites) adjacent to the 2010 augmentation site and side-channel island adjacent to the 2010 augmentation site at Sailor Bar on the lower American River. The exact location of each transect varied based on flow and location of water's edge (see above).*

Snorkeling methods were consistent with those used by other studies (Edmundson et al 1968; Hankin and Reeves 1988; Jackson 1992; McCain 1992; Dolloff et al. 1996; Cavallo et al. 2003), whereby sample transects were snorkeled by two divers moving upstream adjacent to each other, and numbers of fish for each observation were recorded by species (i.e., Chinook salmon or steelhead trout) and size class (i.e., 0–50, 50–80, 80–100, 100–120, 120–150, or >150 mm). However, a few minor modifications were added. First, the area occupied by an individual or group of fish (i.e., <0.5, 0.5–1.0, 1.0–1.5, 1.5–2.0, 2.0–2.5, 2.5–3.0, 3.0–3.5, 3.5–4.0, 4.0–4.5, or 4.5–5.0 $m^2$) was recorded for each observation. Second, divers used numbered, color-coded weights (i.e., 1, 2, or 3 and bank vs. channel) to mark all observations. Third, differentially corrected GPS coordinates were collected for each marked observation using a Trimble GeoXT unit. Additional data collected for each observation included: (1) site (i.e., 2008, 2009, 2010, and 2010SC); (2) transect number (i.e., 1, 2, or 3); (3) location (i.e., bank or channel); (4) observation number (i.e., 1, 2, or 3); (5) depth (m); and (6) velocity ($m{\bullet}s^{-1}$). A Trimble data dictionary developed specifically for snorkel surveys was used to log all additional data at the time GPS coordinates were collected for individual observations. Coordinates and accompanying data for individual observations were input into ArcMap and used to develop .shp files containing the location and attributes of each observation.

*Methods (GIS-based habitat mapping)* – GIS-based habitat mapping methods used for juvenile salmonid analyses were identical to those used for spawning salmonid analyses (see above). However, overlay procedures in ArcMap used to identify and assign cover types, pebble counts, and channel unit types to each observation were slightly different. To determine utilized habitat, point .shp features associated with individual observations were buffered based on occupied areas (see above) to create polygon .shp features for each observation. Occupied areas were assumed to be circles surrounding the exact location of the GPS-logged point. Therefore, all point .shp features were buffed to a distance equal to the square root of occupied area divided by $\pi$ (i.e., appropriate radius) to create polygon .shp features with areas identical to occupied areas. Overlay procedures in ArcMap were then used to assign cover types, pebble counts, and channel unit types to each polygon .shp feature. The overlay procedures effectively partitioned each observation polygon .shp feature into multiple polygons containing discrete combinations of cover types, pebble counts, and channel unit types. When divided by the total area of each original polygon feature, areas associated with each newly created polygon feature represented the relative area of discrete habitat combinations utilized by juvenile salmonids (for individual observations).

*Analysis* –To analyze relationships between Chinook salmon and steelhead trout abundance and habitat selectivity, we conducted a canonical correspondence analysis (CCA). Canonical correspondence analysis is a direct gradient multivariate analysis that simultaneously ordinates species and sample scores along axes of environmental variation (Leps and Smilauer 2003). There can be significant ontogenetic shifts in habitat associations of juvenile salmonids as swimming performance and predation risks change, so both Chinook salmon and steelhead trout counts were separated into size class categories (described above) prior to analysis. Additionally, a correlation analysis on the habitat-by-observation matrix was used to identify potential sources of multicolinearlity. When the correlation coefficient between two variables was $\geq 0.70$, one of the variables was arbitrarily removed from the matrix. The final habitat-by-observation matrix included: (1) depth; (2) velocity; (3) $D_{50}$; (4) Floodplain; (5) Main Channel; (6) Brush; (7) Riparian Grass; (8) Large Woody Material; and (9) Small Woody Material.

In addition to the CCA (see above), we also used a generalized linear mixed model with a Poisson error structure (Bolker et al. 2008) to test for habitat associations with total abundance of both Chinook salmon and steelhead trout. Separate models were developed for each species, whereby the response variable was the count of Chinook salmon or steelhead trout for each observation and the predictor variables were the habitat characteristics included in the CCA (see above). An observation-level random effect was included to account for over-dispersion in fish observations, whereas sample date was included as a random effect to account for variability in time.

### *Macroinvertebrate Community*

Due to altered sediment and flow regimes, many regulated Central Valley streams have become disconnected from historically important floodplains and secondary channels (Wheaton et al 2004a; Florsheim and Mount 2002; Neary et al. 2001). As a result, restoration projects targeting salmonid populations in these streams are increasingly focused on reconnecting floodplains and secondary channels to restore historic ecosystem function and meet juvenile rearing habitat requirements. However, for these restoration projects to be successful, flow regimes must be established in a meaningful framework designed to meet the habitat requirements of a diverse array of salmonid life-history strategies. At a minimum, flow regimes must be established so that newly reconnected habitats: (1) become inundated when target life stages are present and able to utilize the additional habitat; (2) remain inundated for an appropriate duration to allow form, function (i.e., processes), and biological responses (i.e., primary and secondary production) to fully develop and confer some benefit to target life stages; and (3) do not limit utilization by target life stages due to unsuitable physical habitat conditions (i.e., temperature and sediment loading). In general, inundation timing and the potential limiting effects of physical habitat conditions on target life stages are well understood. However, very little information is available on inundation duration required to allow form, function, and biological responses to fully develop and confer some benefit to target life stages. Therefore, many projects designed to restore historic ecosystem function and meet juvenile rearing habitat requirements are implemented without any consideration of flow regimes required to actually accomplish project goals. In this study, we monitor biological responses to inundation duration by tracking benthic macroinvertebrate production within the main channel and floodplain areas adjacent to the 2008–2010 augmentation sites. This study is intended to provide baseline data related to benthic macroinvertebrate community responses to different durations of floodplain inundation, and to evaluate community responses in terms potential food production for juvenile salmonids. Specifically, we tested the following hypotheses:

*$1H_0$:* *There is no significant difference in the overall density or biomass of benthic macroinvertebrates between main channel sample sites and floodplain sample sites adjacent to the 2008–2010 augmentation sites when floodplain sample sites are sampled at a variety of inundation durations.*

*$2H_0$:* *There is no significant difference in the density or biomass of key benthic macroinvertebrate prey items (i.e., Baetidae, Hydropsychidae, and Chironomidae) for juvenile Chinook salmon and steelhead trout between main channel sample sites and floodplain sample sites adjacent to the 2008–2010 augmentation sites when floodplain sample sites are sampled at a variety of inundation durations.*

*Methods* – We utilized existing salmonid diet data from stomach content studies conducted on the lower American (Merz and Vanicek 1996) and Mokelumne (Merz 2001; Merz 2002) rivers to select the three benthic macroinvertebrate prey items most preferred by juvenile Chinook salmon and steelhead trout; including the families Baetidae, Chironomidae, and Hydropsychidae (BCH) (Merz and Vanicek 1996; Merz 2001; Merz 2002). Following identification of preferred prey items, we collected benthic macroinvertebrate samples within the main channel and floodplain areas adjacent to the 2008–2010 augmentation sites (Figures 13 and 14) on 12 separate sample dates during the 2010–2011 monitoring season (Table 2). Sample dates were determined by flow, and were selected to correspond to progressively increasing durations of floodplain inundation ranging from ~1–10 weeks. The sampling scheme included a series of two paired sampling periods (6 samples each) designed to assess floodplain colonization by benthic macroinvertebrates during both fall and spring monitoring. Because spring samples were more highly correlated to the juvenile rearing period, only spring samples were used for final analyses.

Initially, low flow (2,141–2,654 $ft^3 \bullet s^{-1}$) main channel samples (i.e., control samples) were collected. These main channel samples were followed by high flow (2,741–10,177 $ft^3 \bullet s^{-1}$) floodplain samples (i.e., treatment samples). To ensure that floodplain samples were collected at appropriate inundation durations, flows were continuously monitored throughout both sampling periods (Figure 15), and all samples were collected within ~1.0 m of water's edge.

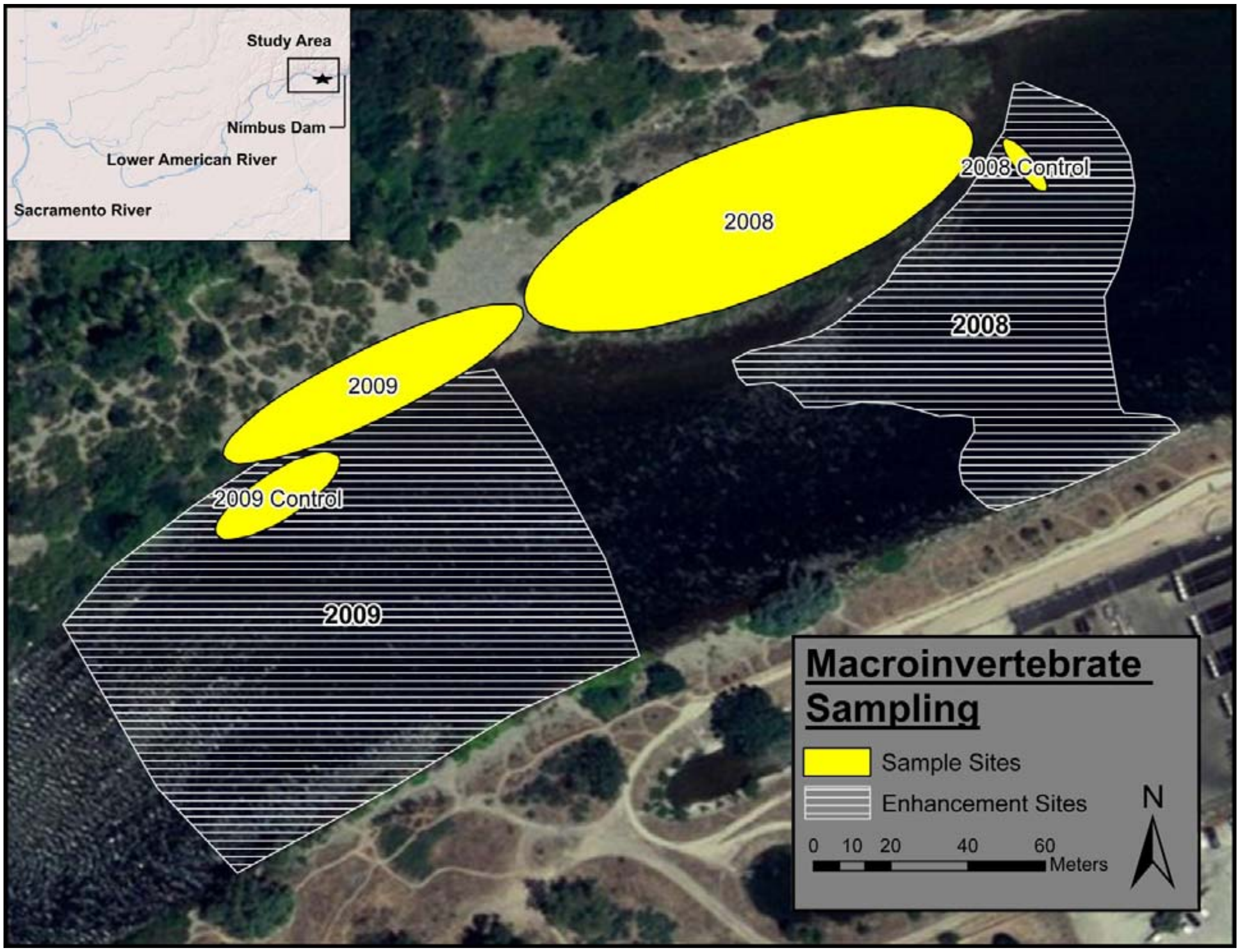


***Figure 13**. Approximate locations of main channel (i.e., control) and floodplain (i.e., treatment) benthic macroinvertebrate sample sites adjacent to the 2008–2009 augmentation sites at Sailor Bar on the lower American River. Floodplain sample sites varied based on flow and location of water's edge (see above).*

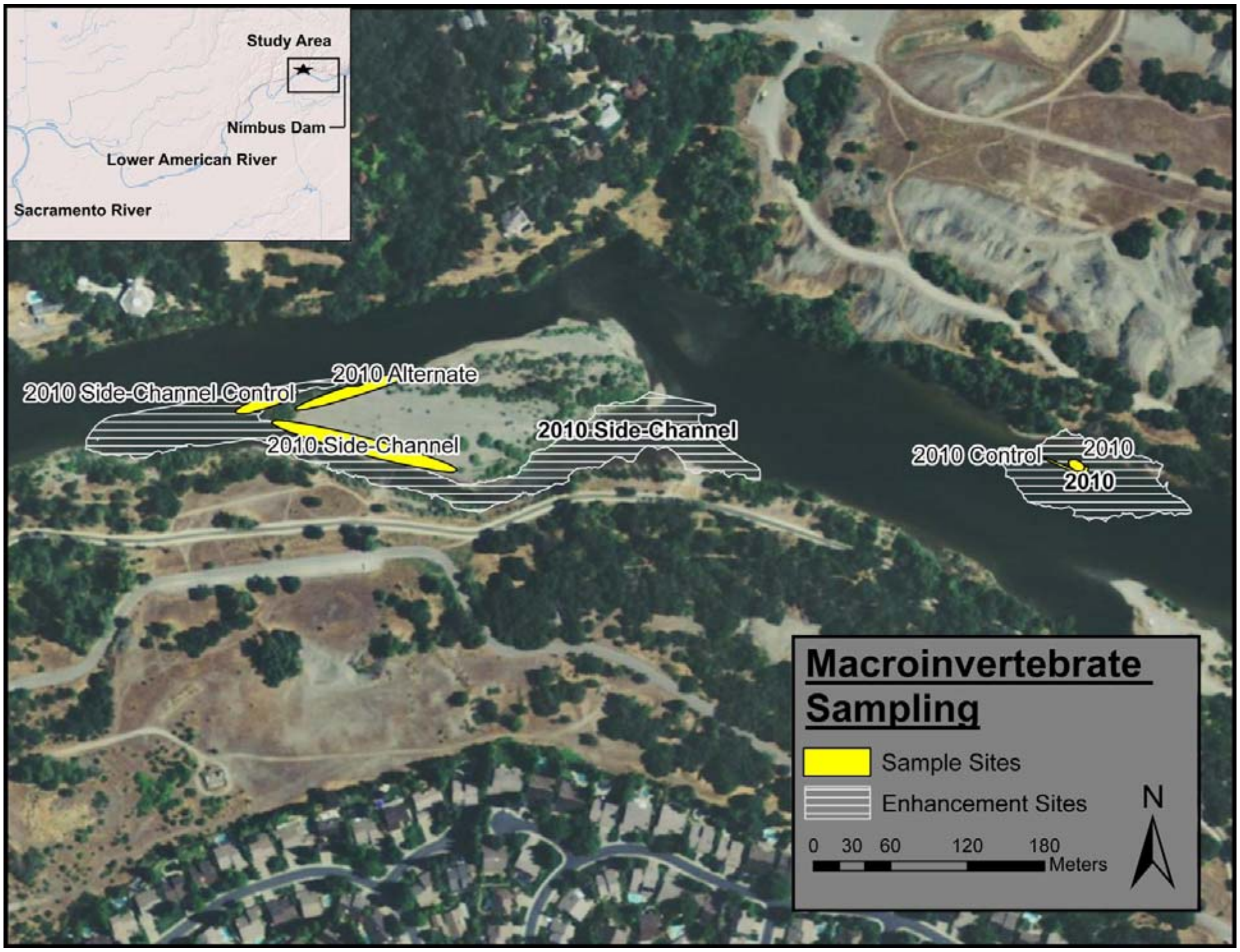


***Figure 14***. *Approximate locations of main channel (i.e., control) and floodplain (i.e., treatment) benthic macroinvertebrate sample sites adjacent to the 2010 augmentation site and side-channel island adjacent to the 2010 augmentation site at Sailor Bar on the lower American River. Floodplain sample sites varied based on flow and location of water's edge (see above).*

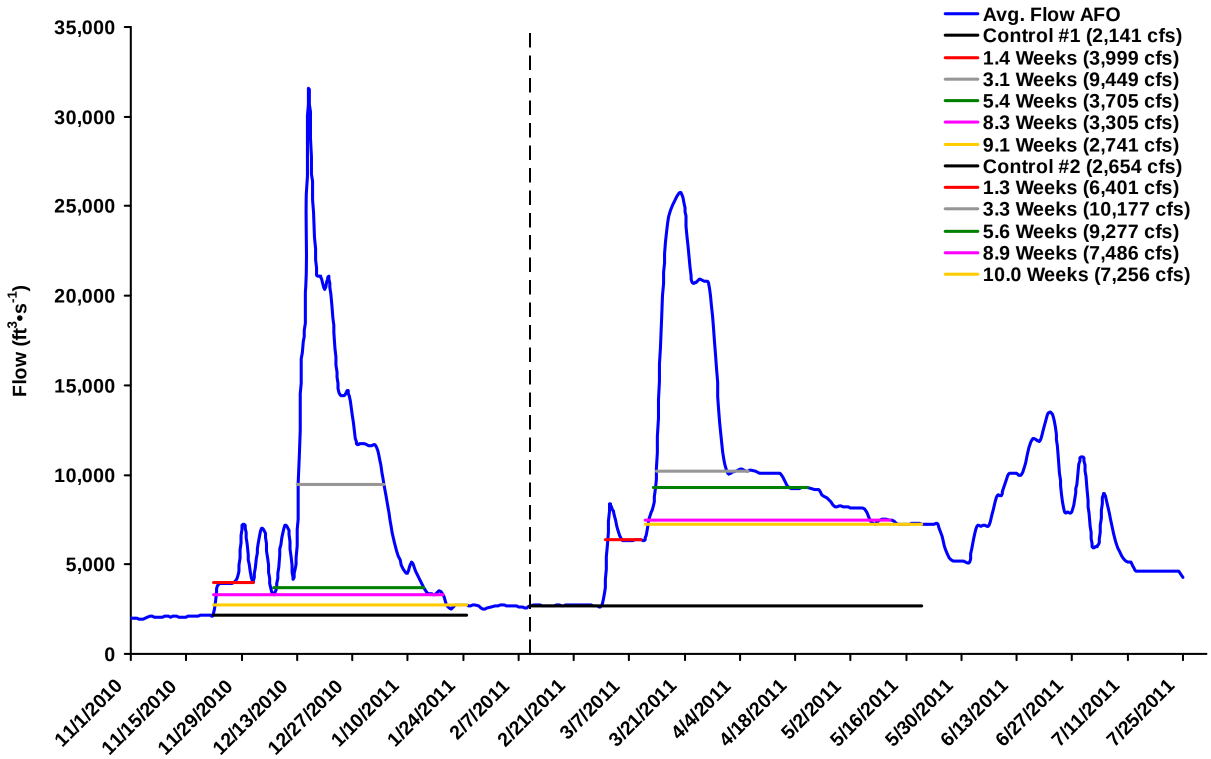


***Figure 15***. *Lower American River flow at Fair Oaks (AFO; CDEC) and conceptual model of benthic macroinvertebrate sampling scheme. Dashed vertical line separates paired samples collected during fall and spring monitoring.*

Benthic macroinvertebrates were collected at four randomly selected locations within each sample site (see above) using a stainless steel Hess sampler (Wildco, Inc; 330 mm inside diameter; 400 mm height; 363 μm mesh with attached 368 μm dolphin bucket). Samples were collected by scouring the substrate with a scrub brush within the 0.086 m$^2$ open bottom of the sampler to a depth of ~15 cm. Individual samples were placed in 500 mL Nalgene bottles containing ~70–95% ethyl alcohol, transported to the laboratory, and hand sorted using a 60x dissecting scope. Organisms were identified to species when possible; otherwise, they were categorized to the lowest taxonomic group possible (typically family). Individual organisms were grouped by type, further categorized by size class (i.e., <2, 2–7, 8–13, 14–20, and >20 mm) and life stage (i.e., larva/nymph, pupa, and adult), and enumerated for each type-size-life-stage combination. Dry biomass values for individual species were used for different life stage and size class combinations. These values were derived from those collected in previous studies on the lower American and Mokelumne rivers (see Merz 1994; Merz and Chan 2004; Washburn and Merz 2008), whereby dry biomass of the organisms was determined by oven-drying samples of each taxonomic group (i.e., order or family) in a representative life stage and size class at 70°C for 24 hours or to constant weight (<24 hours), and then weighing the sample to the nearest 0.0001 g. Densities for life stage and size class combinations were multiplied by individual weight values to develop measures of biomass for each discrete category.

*Analysis* – Differences in the benthic macroinvertebrate community were evaluated by comparing differences in the overall density (number/m$^2$) and biomass (g/m$^2$) of macroinvertebrates, and the density (number/m$^2$) and biomass (g/m$^2$) of preferred macroinvertebrate prey items (i.e., Baetidae, Chironomidae, and Hydropsychidae) between treatment and control samples. First, an ANOVA with a Tukey test (Zar 1999) was used for all comparisons. Comparisons were made on both a control-treatment-basis (i.e., data from all control and treatment samples pooled) and on an inundation-duration-basis (i.e., data from all control samples pooled and data from all treatment samples pooled based on inundation duration). Additionally, comparisons were made on a pooled-upstream-downstream-basis (i.e., data for upstream [2008–2009] and downstream [2010] sample sites pooled and treated together and data for upstream and downstream sample sites treated separately). Comparisons for preferred prey items were first conducted for a combined BCH index of density and biomass (i.e., Baetidae, Chironomidae, and Hydropsychidae grouped together), and then for separate BCH indices of density and biomass (i.e., Baetidae, Chironomidae, and Hydropsychidae treated separately). Second, we performed multiple chi-square analyses:

$$X^2 = \sum \frac{(X_a - X_b)^2}{X_b} \qquad (8)$$

Where $X_b$ = group density or biomass (i.e., BCH combined or treated separately) in control samples/density or biomass for all macroinvertebrates combined in control samples, and $X_a$ = group density or biomass (i.e., BCH combined or treated separately) in treatment samples/density or biomass for all macroinvertebrates combined in treatment samples (Zar 1999). Separate chi-square analyses were used to test for overall treatment effects (i.e., all treatment samples pooled) and peak treatment effects (i.e., only treatment samples taken at peak density or biomass). Additionally, separate chi-square analyses were used to test for location effects (i.e., data from upstream and downstream sample sites pooled and treated together and data for upstream and downstream sample sites treated separately). These analyses allowed us to test for significant differences in average density or biomass between control and treatment samples, significant differences in the relative proportions (density and biomass) of preferred macroinvertebrate prey items between control and treatment samples, and assess differences in upstream (2008 and 2009) and downstream (2010 and 2010 side-channel) sample sites.

## RESULTS

### *Floodplain Inundation*

Flows stayed relatively constant from 1 January–31 July and rarely exceeded minimum flow in the “critical” year, but were highly variable from 1 January–31 July and consistently exceeded minimum flow in the “wet” year (Figure 16). Peak flows in the “wet” year largely corresponded to increased rearing habitat requirements from 1 January–1 March (Figures 16 and 19). In contrast, peak flows in the “critical” year largely occurred after peak rearing habitat requirements (i.e., after 1 March; Figures 16 and 18). When flow-inundation curves were applied to flow data from example “critical” and “wet” years, total additional inundated areas ranged from 0–3,289 $m^2$ (0–2,533 $m^2$ floodplain adjacent to the 2008 and 2009 sites and 0–756 $m^2$ side-channel island adjacent to the 2010 site) in the “critical” year and 0–29,384 $m^2$ (0–21,593 $m^2$ floodplain adjacent to the 2008 and 2009 sites and 0–7,755 $m^2$ side-channel island adjacent to the 2010 site) in the “wet” year (Figure 17). Assuming an average HSI value of 0.50, these values represent an average of <1.0% (range = 0.0–19.5%) of the total daily habitat requirements in the “critical” year (7,727,278 juveniles; Figure 22) and an average of 7.5% (range = 0.0–85.0%) of the total daily habitat requirements in the "wet" year (12,566,322 juveniles; Figure 23). Similarly, assuming an average HSI value of 0.50, "wet" (1999) outmigration parameters, and the AFRP adult production target (39,840,000 juveniles), these values represent an average of <1.0% (range = 0–2.2%) of the total daily habitat requirements in a "critical" flow year (1994 data; Figure 20) and an average of 2.4% (range = 0–26.8%) of the total daily habitat requirements in a "wet" flow year (1999 data; Figure 21). Increasing or decreasing HSI values by 50% substantially altered total juvenile rearing habitat requirements. However, visual inspection of plots suggested that the relative proportion of total juvenile rearing habitat requirements potentially accounted for by areas adjacent to the 2008–2010 augmentation sites remained largely unchanged (i.e., areas adjacent to the 2008–2010 augmentation sites were relatively small in proportion to required habitat regardless of changes in habitat quality; Figures 18-21).

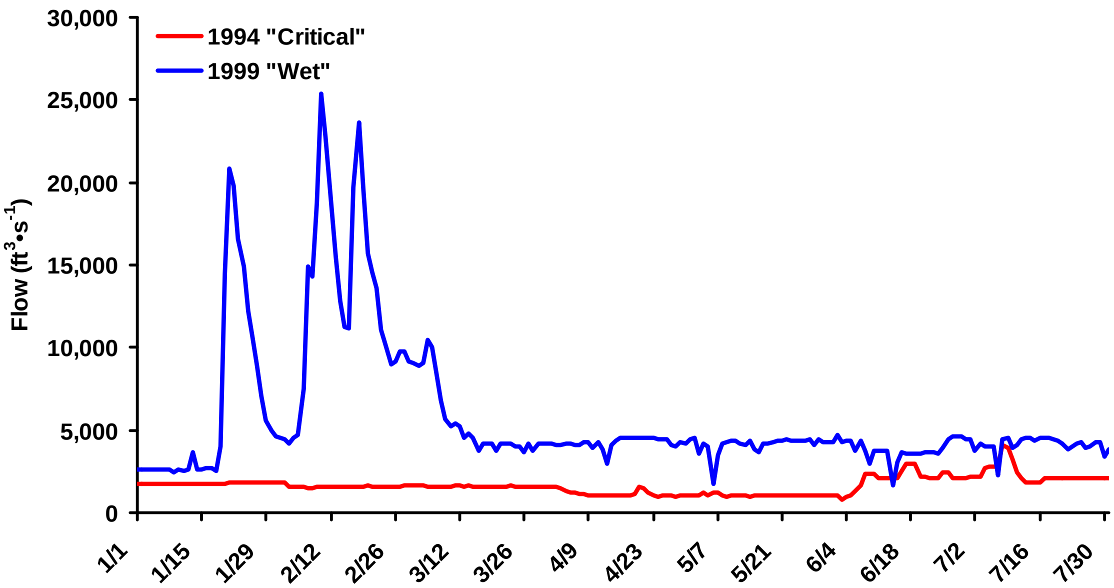


***Figure 16****. Lower American River flow at Fair Oaks from 1 January–31 July during "critical" (1994) and "wet" (1999) years (CDEC).*

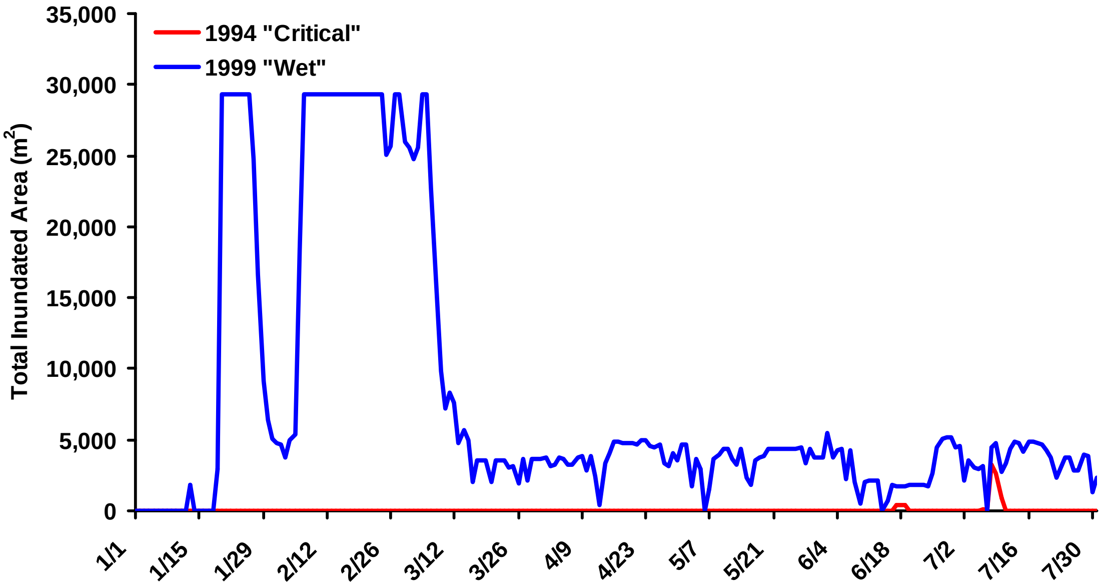


***Figure 17**. Total additional inundated areas adjacent to the 2008–2010 augmentation sites calculated using flow-inundation curves (see above) and lower American River flow at Fair Oaks from 1 January–31 July during "critical" (1994) and "wet" (1999) years (see above).*

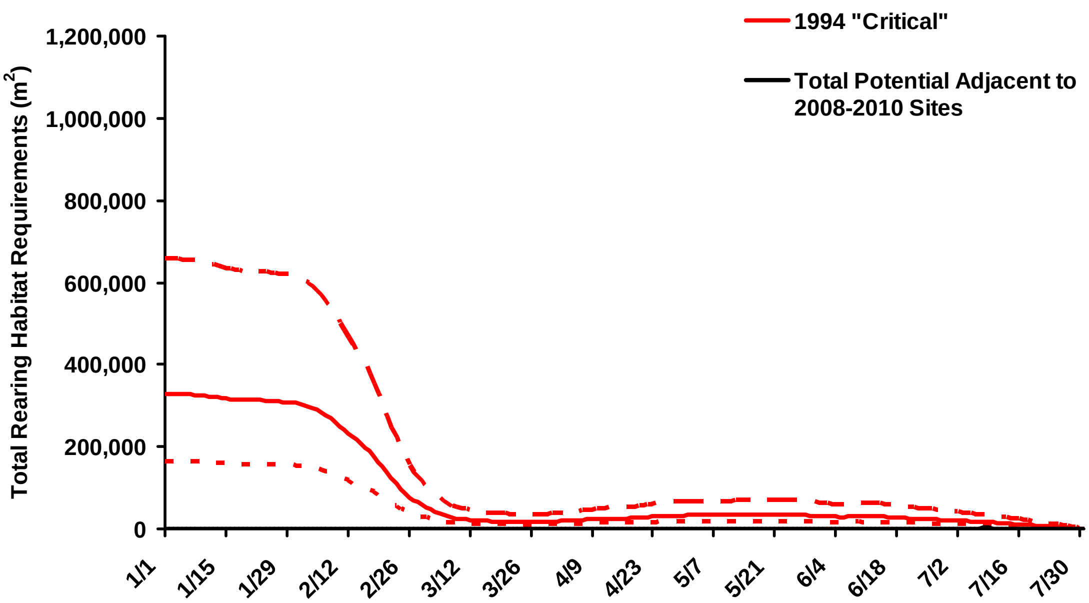


***Figure 18**. Total rearing habitat requirements and total potential rearing habitat adjacent to the 2008–2010 augmentation sites for the example "critical" (1994) year during 1 January–31 July. Solid line represents habitat requirements based on an average HSI value (0.50). Dashed lines represent habitat requirements based on 50% increase or decrease in average HSI value. Potential habitat adjacent to 2008–2010 augmentation sites is largely dwarfed by required habitat. Required habitat is based on 1994 outmigration data.*

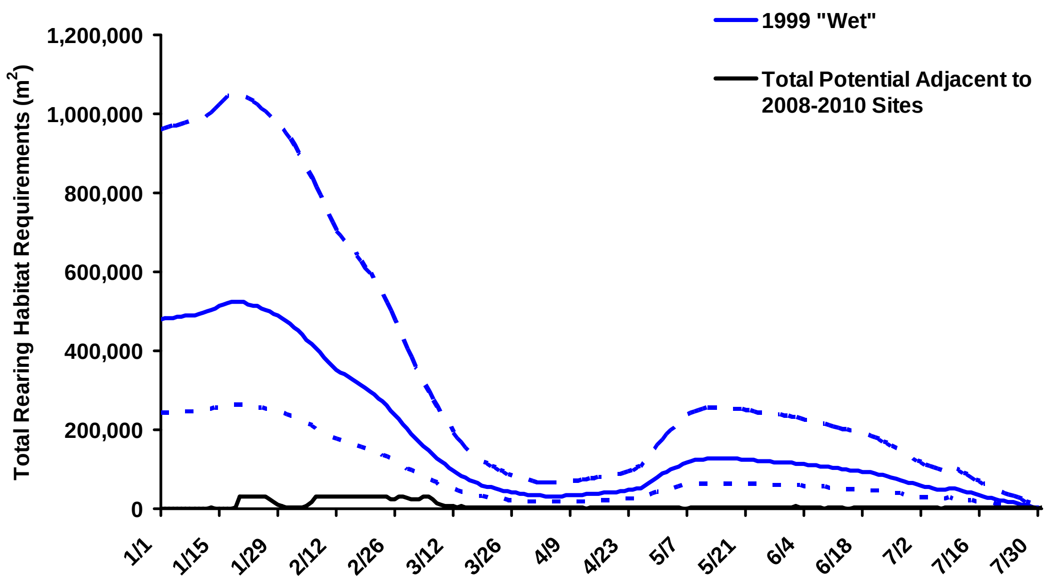


***Figure 19***. *Total rearing habitat requirements and total potential rearing habitat adjacent to the 2008–2010 augmentation sites for the example "wet" (1999) year during 1 January–31 July. Solid line represents habitat requirements based on an average HSI value of 0.50. Dashed lines represent habitat requirements based on a 50% increase or decrease in average HSI value. Potential habitat adjacent to the 2008–2010 augmentation sites is largely dwarfed by required habitat. Required habitat is based on 1999 outmigration data.*

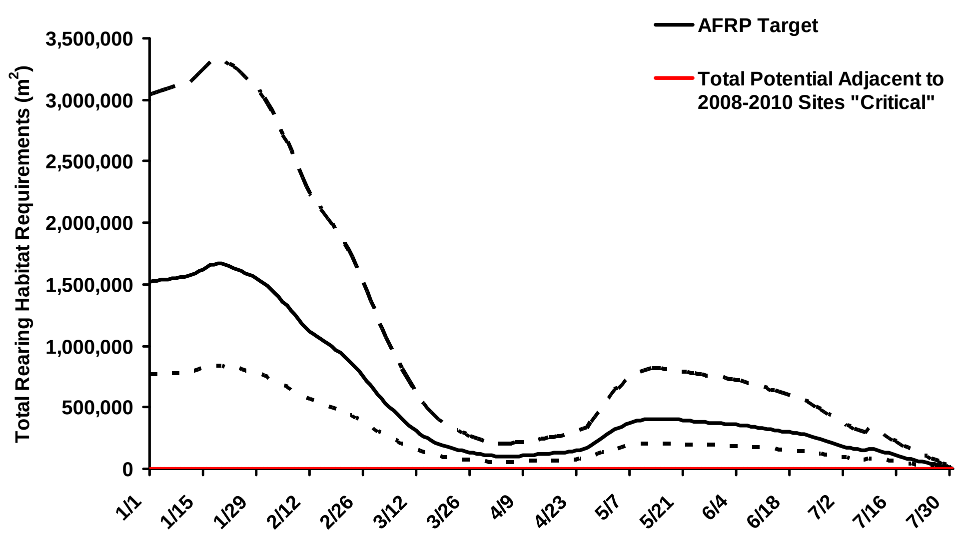


***Figure 20***. *Total rearing habitat requirements and potential adjacent to 2008–2010 augmentation sites for the example "critical" (1994) water year during 1 January–31 July. Solid line represents habitat requirements based on average HSI value (0.50). Dashed lines represent habitat requirements based on 50% increase or decrease in average HSI value. Potential habitat adjacent to 2008–2010 augmentation sites is largely dwarfed by required habitat. Required habitat based on 1999 outmigration parameters applied to predicted number of juveniles produced by AFRP adult production target for fall-run Chinook salmon.*

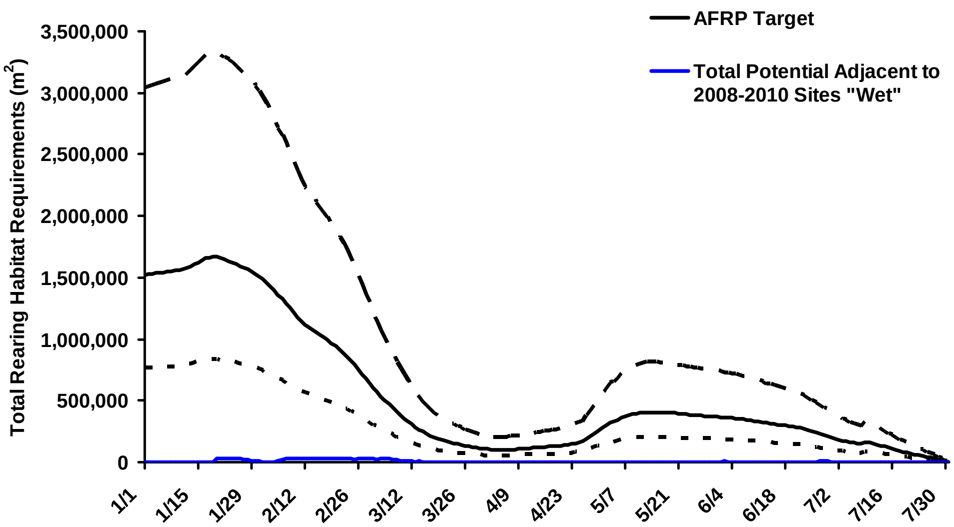


***Figure 21****. Total rearing habitat requirements and total potential rearing habitat adjacent to the 2008–2010 augmentation sites for the example "wet" (1999) water year during 1 January–31 July. Solid line represents habitat requirements based on an average HSI value of 0.50. Dashed lines represent habitat requirements based on a 50% increase or decrease in average HSI value. Potential habitat adjacent to the 2008–2010 augmentation sites is largely dwarfed by required habitat. Required habitat is based on 1999 outmigration parameters applied to the predicted number of juveniles produced by the AFRP adult production target for fall-run Chinook salmon.*

## *Spawning Habitat Use*

### ***Chinook Salmon***

Total annual river redds ranged from 316–5,309 (average = 2,181; 2004–2005 through 2010–2011; Figure 22). Prior to enhancement, the number of redds constructed ranged from 0–13 (average = 3; 2004–2005 through 2007–2008) at the 2008 site, 0–86 (average = 29; 2004–2005 through 2008–2009) at the 2009 site, 0–69 (average = 29; 2004–2005 through 2009–2010) at the 2010 site, and 0–220 (average = 74; 2004–2005 through 2009–2010) in the side-channel adjacent to the 2010 site (Figure 23). Maximum pre-enhancement utilization (%) rates for all four sites were <5% (Figure 24). In 2010–2011, 175 redds were observed at the 2008 augmentation site, 34 redds were observed at the 2009 augmentation site, 0 redds were observed at the 2010 augmentation site, and 42 redds were observed in the side-channel adjacent to the 2010 augmentation site (Figures 25 and 26). The 2008 augmentation site accounted for ~33% of total annual river redds, the 2009 augmentation site accounted for ~6% of total annual river redds, the 2010 augmentation site accounted for 0% of total annual river redds, and the side-channel adjacent to the 2010 augmentation site accounted for ~8% of total annual river redds (Figure 27). Based on pre- and post-enhancement data for all available years (i.e., 2004–2005 through 2010–2011), this indicates a significant increase in utilization of the 2008 site ($\chi^2$ = 1,494.867; $df$ = 1, 15,633; $p$<0.0001), 2009 site ($\chi^2$ = 287.569; $df$ = 1, 15,518; $p$<0.0001), and the side-channel adjacent to the 2010 site ($\chi^2$ = 26.439; $df$ = 1, 15,754; $p$<0.0001) compared to pre-enhancement conditions, but a significant decrease in utilization of the 2010 site ($\chi^2$ = 12.295; $df$ = 1, 15,442; $p$=0.0005). New redd construction at the augmentation sites was bi-modal with peaks in November and January (Figure 27). In general, redds were distributed throughout the 2008 site, confined to river margins and discrete areas within the 2009 site, and distributed across the width of the river at the head of the side-channel adjacent to the 2010 site (Figures 25 and 26).

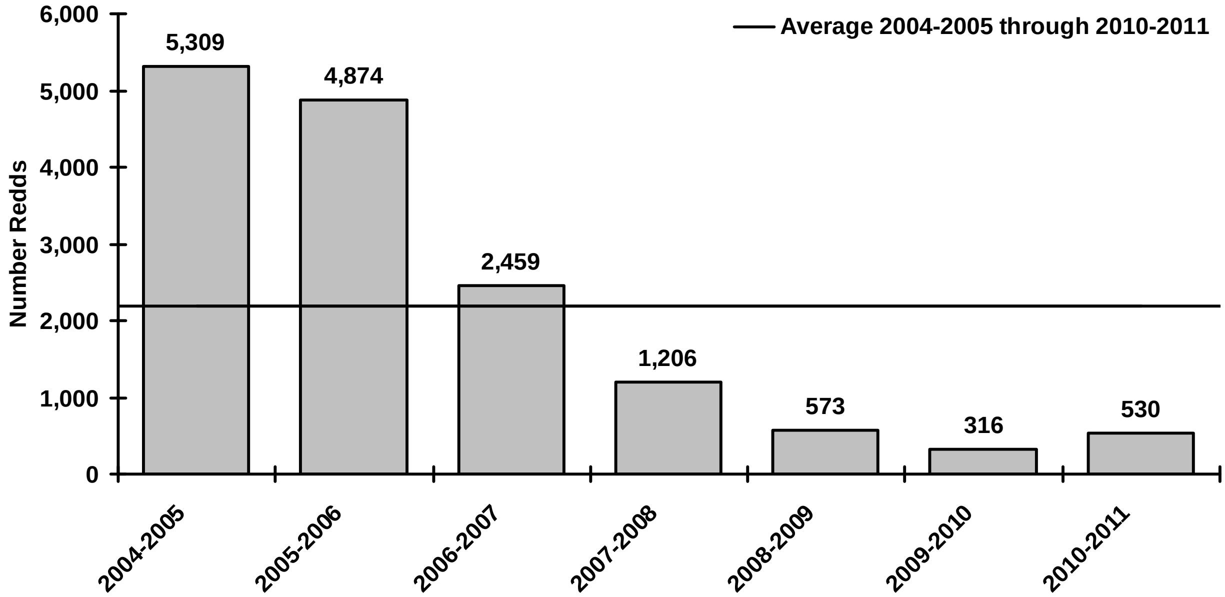


***Figure 22****. Total annual lower American River Chinook salmon redds during the 2004–2005 through 2010–2011 spawning seasons. Solid line represents seven-year average.*

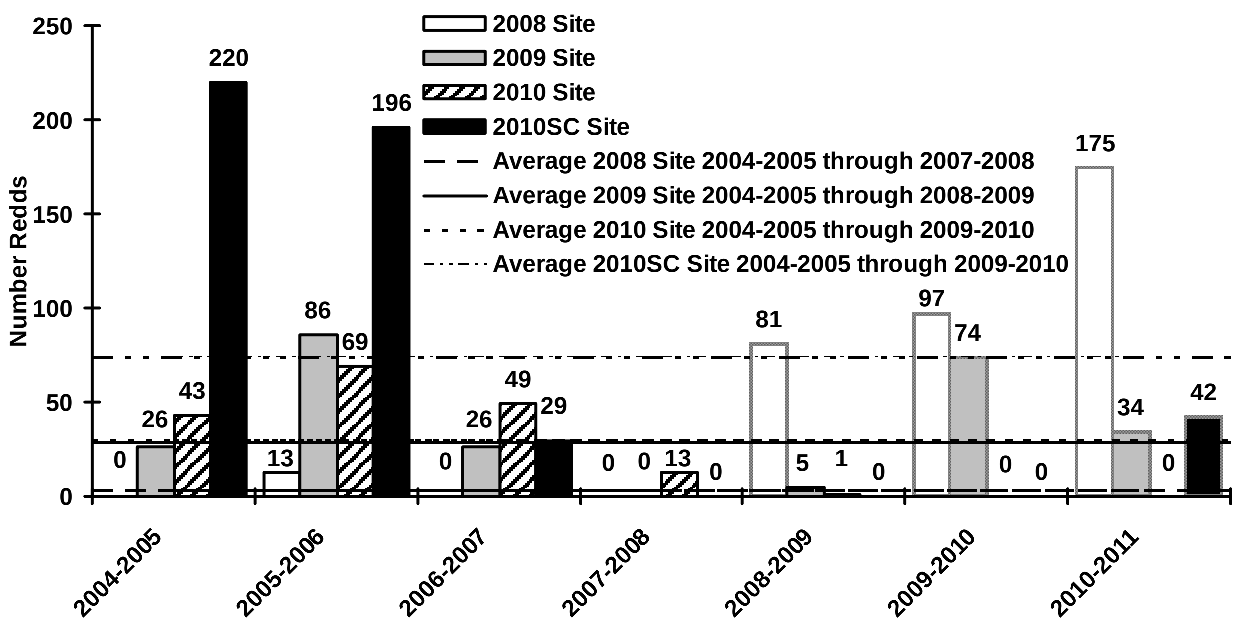


***Figure 23****. Augmentation site redds for Chinook salmon spawning during the 2004–2005 through 2010–2011 spawning seasons. Bars with a dark outline represent pre-enhancement conditions and bars with a light outline represent post-enhancement conditions. Solid and dashed lines represent pre-enhancement averages.*

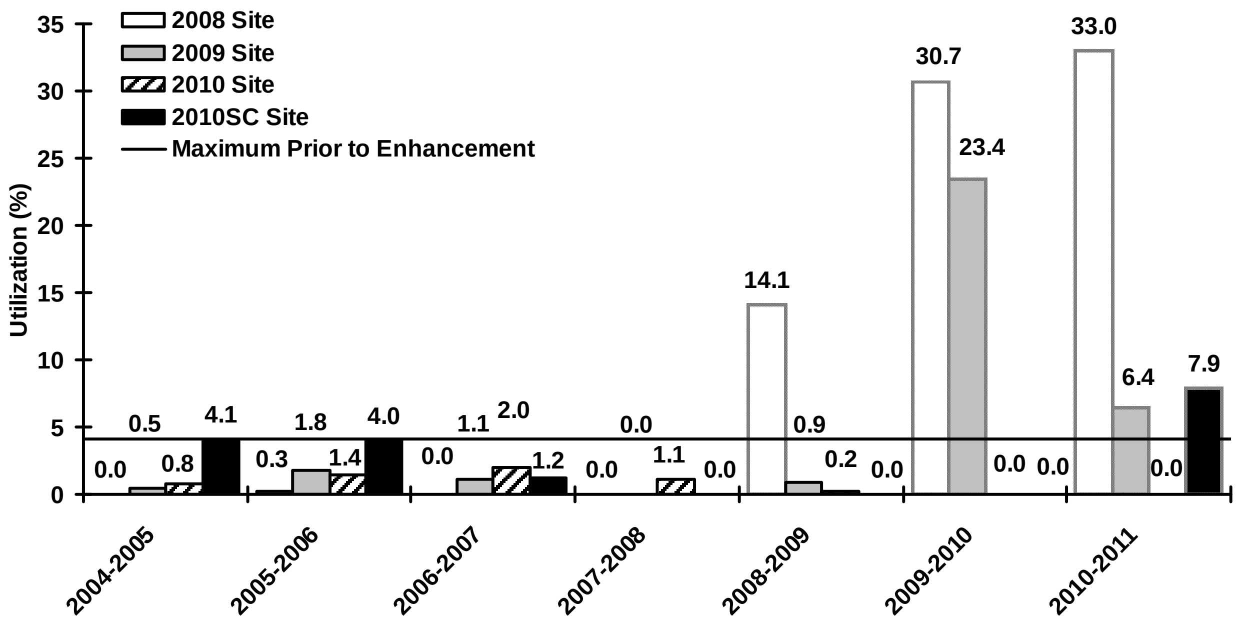


***Figure 24***. *Augmentation site utilization (%) rates for Chinook salmon spawning during the 2004–2005 through 2010–2011 spawning seasons. Bars with a dark outline represent pre-enhancement conditions and bars with a light outline represent post-enhancement conditions. Solid line represents pre-enhancement maximum of all sites.*

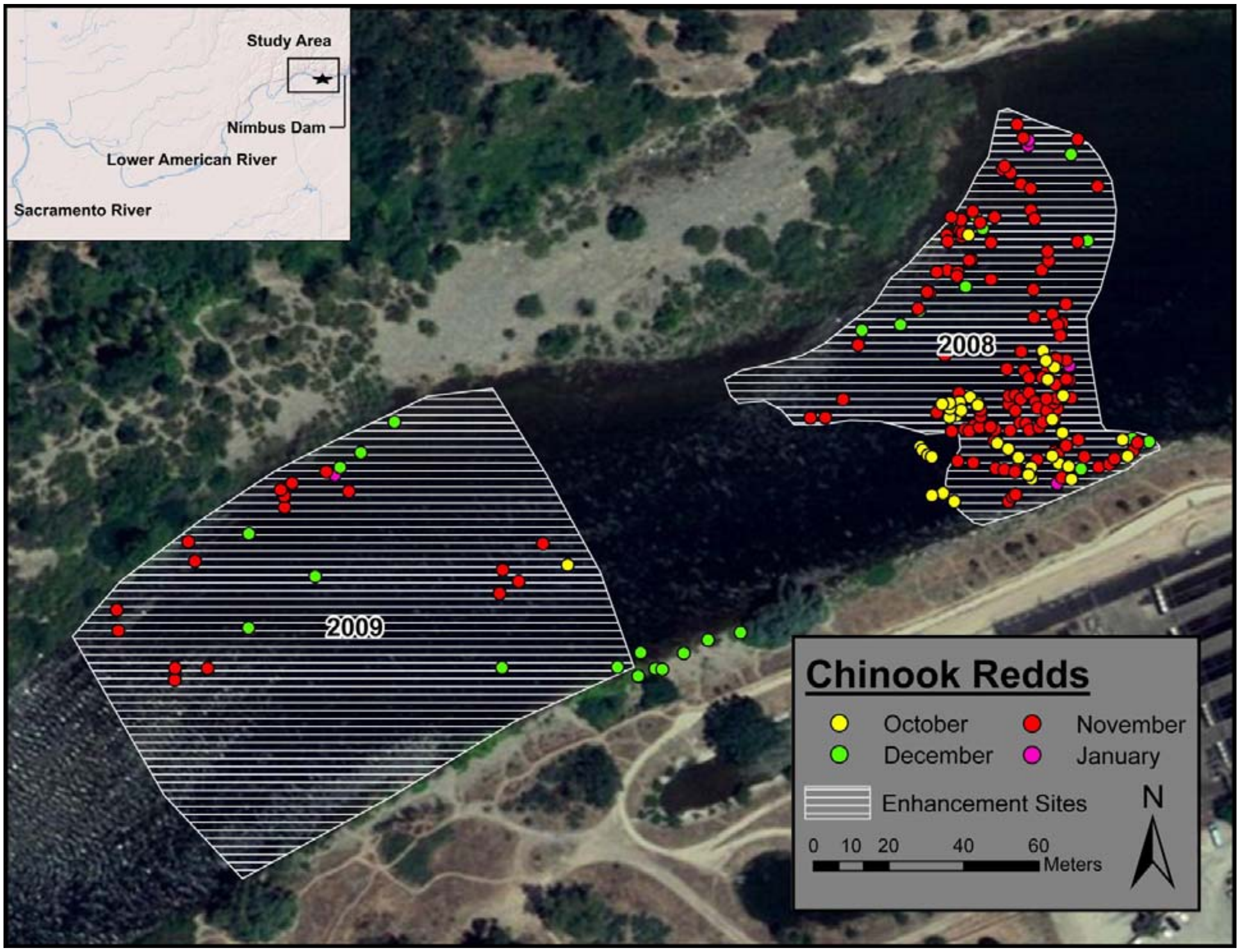


***Figure 25**. Locations of Chinook salmon redds during the 2010–2011 spawning season. Redd locations are color-coded based on month of observation. Only observations from the 2008 and 2009 augmentation sites are provided. Photo is for reference purposes only and does not indicate true flow during the spawning period.*

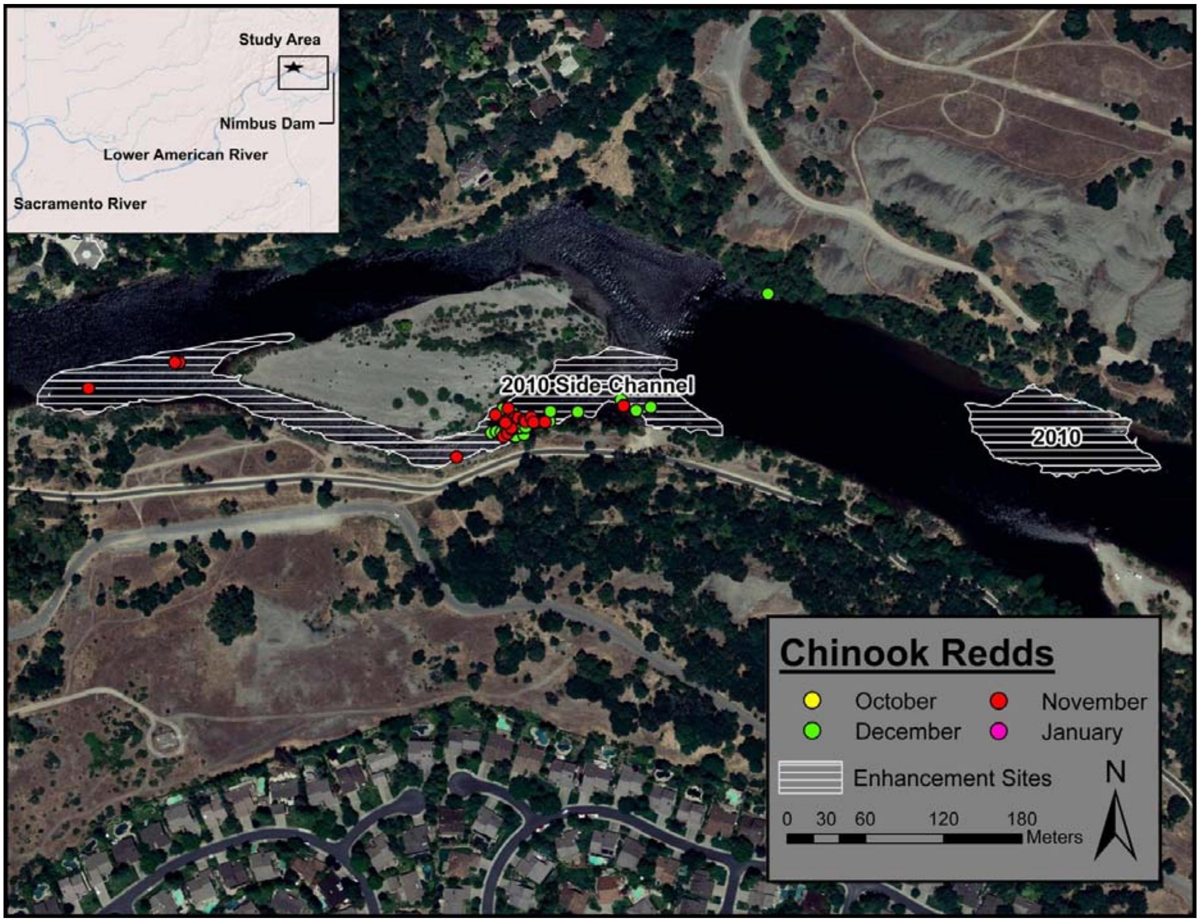


***Figure 26***. *Locations of Chinook salmon redds during the 2010–2011 spawning season. Redd locations are color-coded based on month of observation. Only observations from the 2010 augmentation site and the side-channel adjacent to the 2010 augmentation site are provided.*

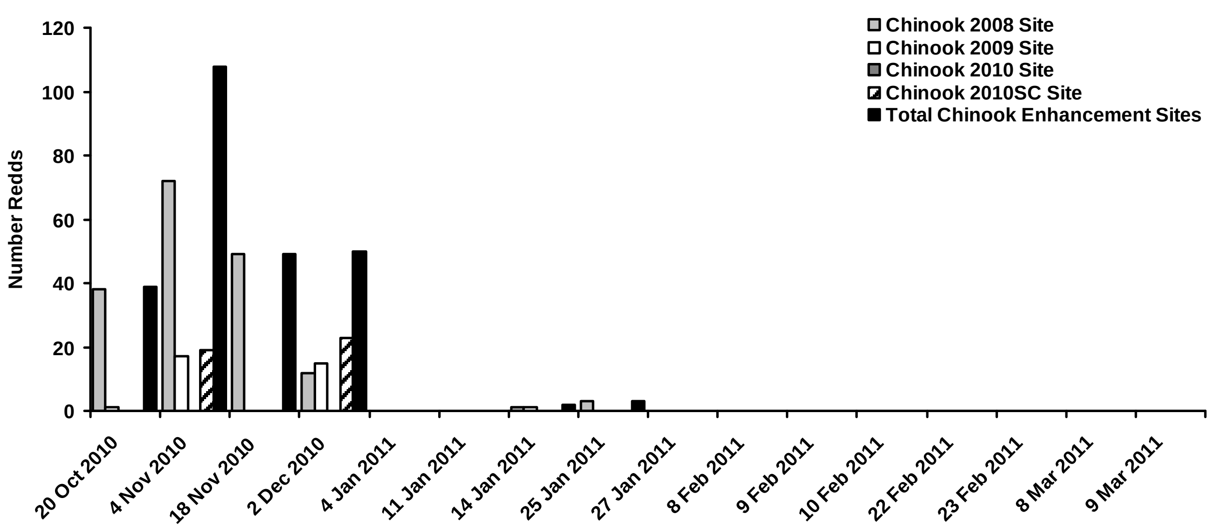


***Figure 27***. *Number of Chinook salmon redd observations at augmentation (i.e., enhancement) sites surveyed during the 2010–2011 spawning season.*

### *Steelhead Trout*

Total annual river redds ranged from 79–215 (average = 146; 2001–2002 through 2010–2011; Figure 28). Prior to enhancement, the number of redds constructed ranged from 0–2 (average = 0.8; 2001–2002 through 2007–2008) at the 2008 site, 0–15 (average = 7.2; 2001–2002 through 2008–2009) at the 2009 site, 0–1 (average = 0.4; 2001–2002 through 2009–2010) at the 2010 site, and 0–37 (average = 18.7; 2001–2002 through 2010–2011) in the side-channel adjacent to the 2010 site (Figure 29). Maximum pre-enhancement utilization (%) rates for all four sites were <24% (Figure 30). In 2010–2011, 37 redds were observed at the 2008 augmentation site, 0 redds were observed at the 2009 augmentation site, 1 redd was observed at the 2010 augmentation site, and 9 redds were observed in the side-channel adjacent to the 2010 augmentation site (Figures 31 and 32). The 2008 augmentation site accounted for ~40% of total annual river redds, the 2009 augmentation site accounted for 0% of total annual river redds, the 2010 augmentation site accounted for ~1% of total annual river redds, and the side-channel adjacent to the 2010 augmentation site accounted for ~10% of total annual river redds (Figure 38). Based on pre- and post-enhancement data for all available years (i.e., 2001–2002 through 2010–2011), this indicates a significant increase in utilization of the 2008 site ($X^2 = 307.918$; $df = 1, 1{,}299$; $p < 0.0001$) compared to pre-enhancement conditions, but not the 2009 site ($X^2 = 0.229$; $df = 1, 1{,}216$; $p = 0.6320$), 2010 site ($X^2 = 1.069$; $df = 1, 1{,}171$; $p = 0.3011$), or the side-channel adjacent to the 2010 site ($X^2 = 0.389$; $df = 1, 1{,}307$; $p = 0.5326$). New redd construction at the augmentation sites gradually increased to a peak in February and then decreased sharply (Figure 33). In general, redds were distributed across the width of the river at the head of the 2008 site and the middle of the side-channel adjacent to the 2010 site (Figures 31 and 32). The single redd at the 2010 site was located in the side-channel between the river margin and the placed cobble island (Figure 32). Pre-enhancement steelhead trout redd observations were distributed throughout RMs 5–23 and slightly skewed towards the upstream end of the LAR, whereas post-enhancement steelhead trout redd observations were distributed throughout RMs 6–22 and highly skewed towards the upstream end of the LAR (Figure 34).

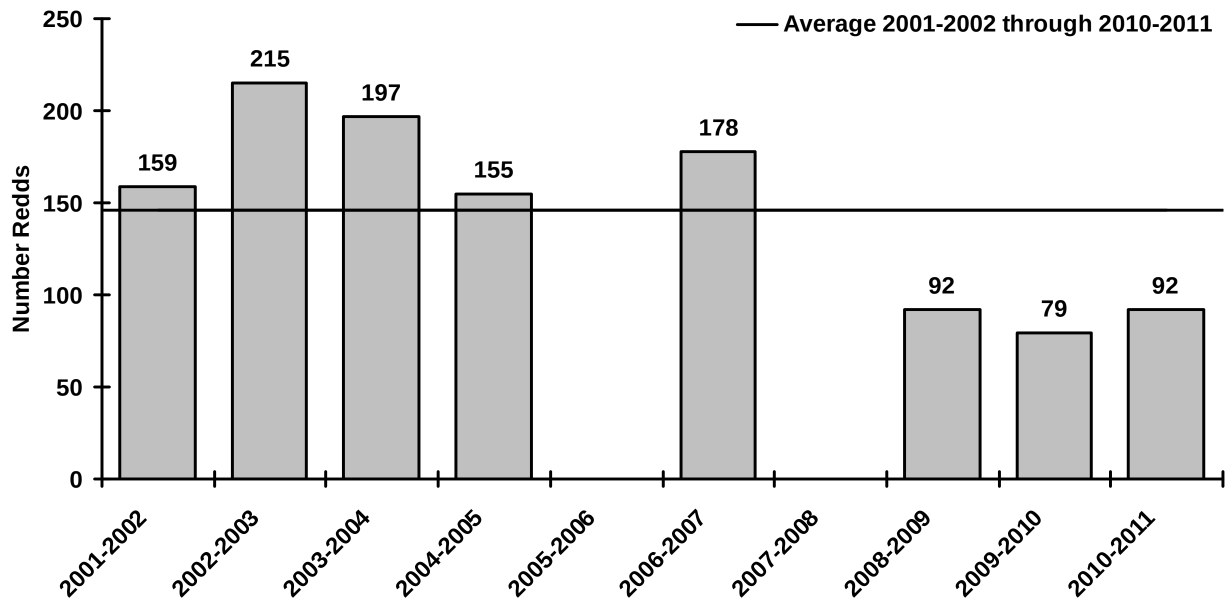


***Figure 28**. Total annual river redds for steelhead trout spawning during the 2001–2002 through 2010–2011 spawning seasons. Solid line represents average during time period. Data were unavailable for the 2005–2006 and 2007–2008 spawning seasons.*

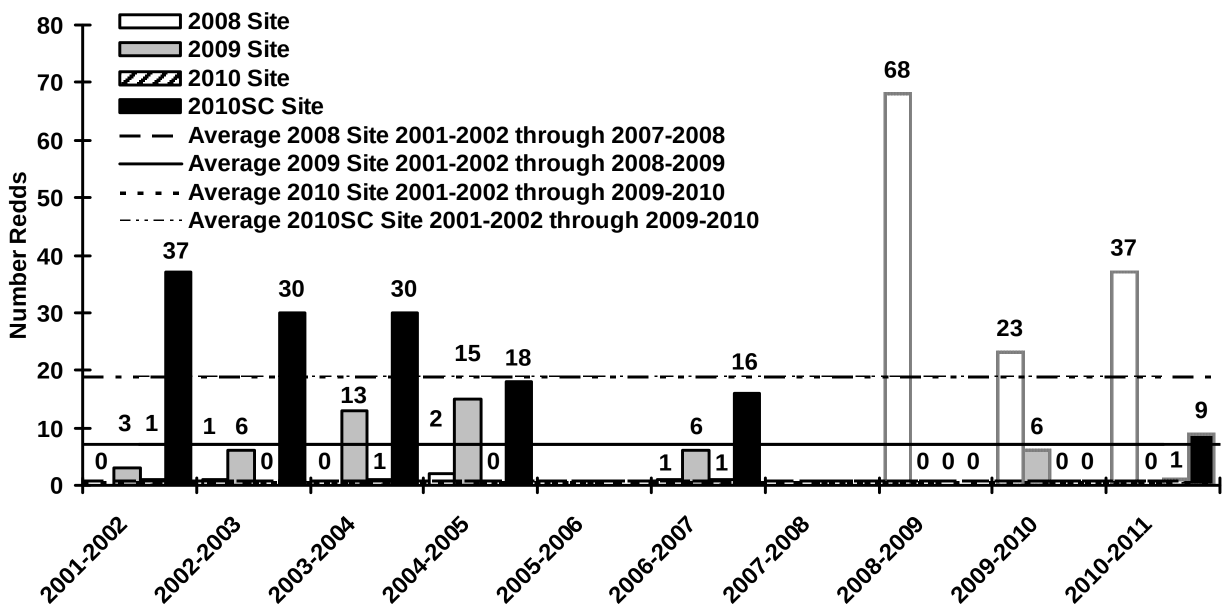


***Figure 29**. Augmentation site redds for steelhead trout spawning during the 2001–2002 through 2010–2011 spawning seasons. Bars with a dark outline represent pre-enhancement conditions and bars with a light outline represent post-enhancement conditions. Solid and dashed lines represent pre-enhancement averages. Data were unavailable for the 2005–2006 and 2007–2008 spawning seasons.*

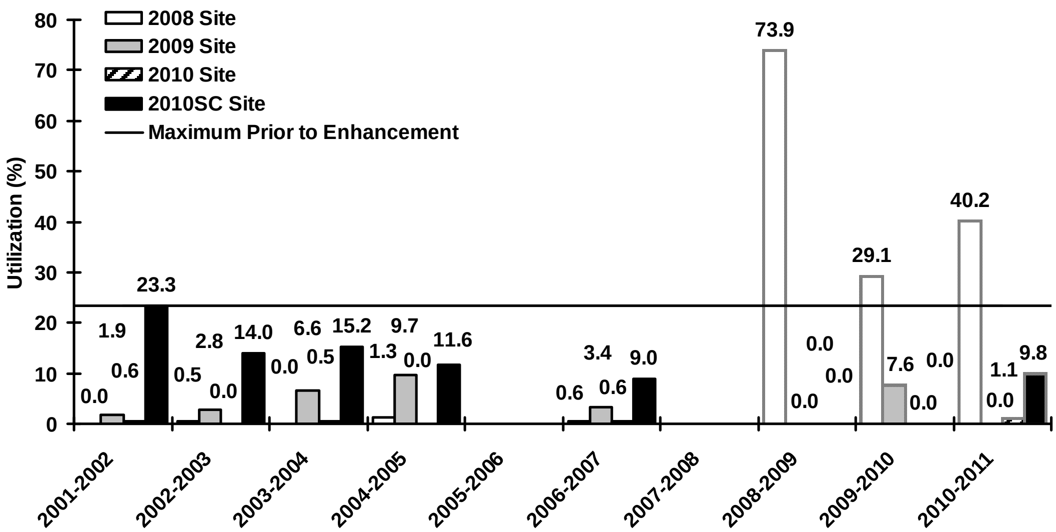


***Figure 30**. Augmentation site utilization (%) rates for steelhead trout spawning during the 2001–2002 through 2010–2011 spawning seasons. Bars with a dark outline represent pre-enhancement conditions and bars with a light outline represent post-enhancement conditions. Solid line represents pre-enhancement maximum of all sites. Data were unavailable for the 2005–2006 and 2007–2008 spawning seasons.*

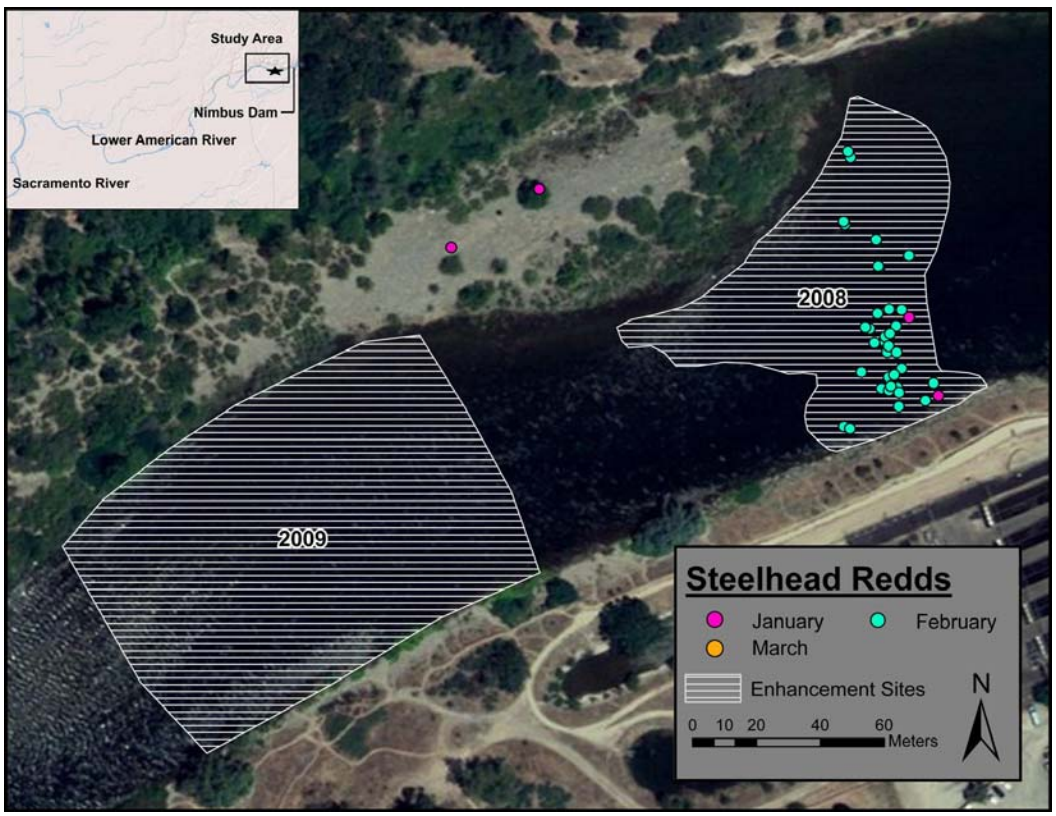


***Figure 31**. Locations of steelhead trout redds during the 2010–2011 spawning season. Redd locations are color-coded based on month of observation. Only observations from the 2008 and 2009 augmentation sites are provided. Photo is for reference purposes only and does not indicate true flow during the spawning period.*

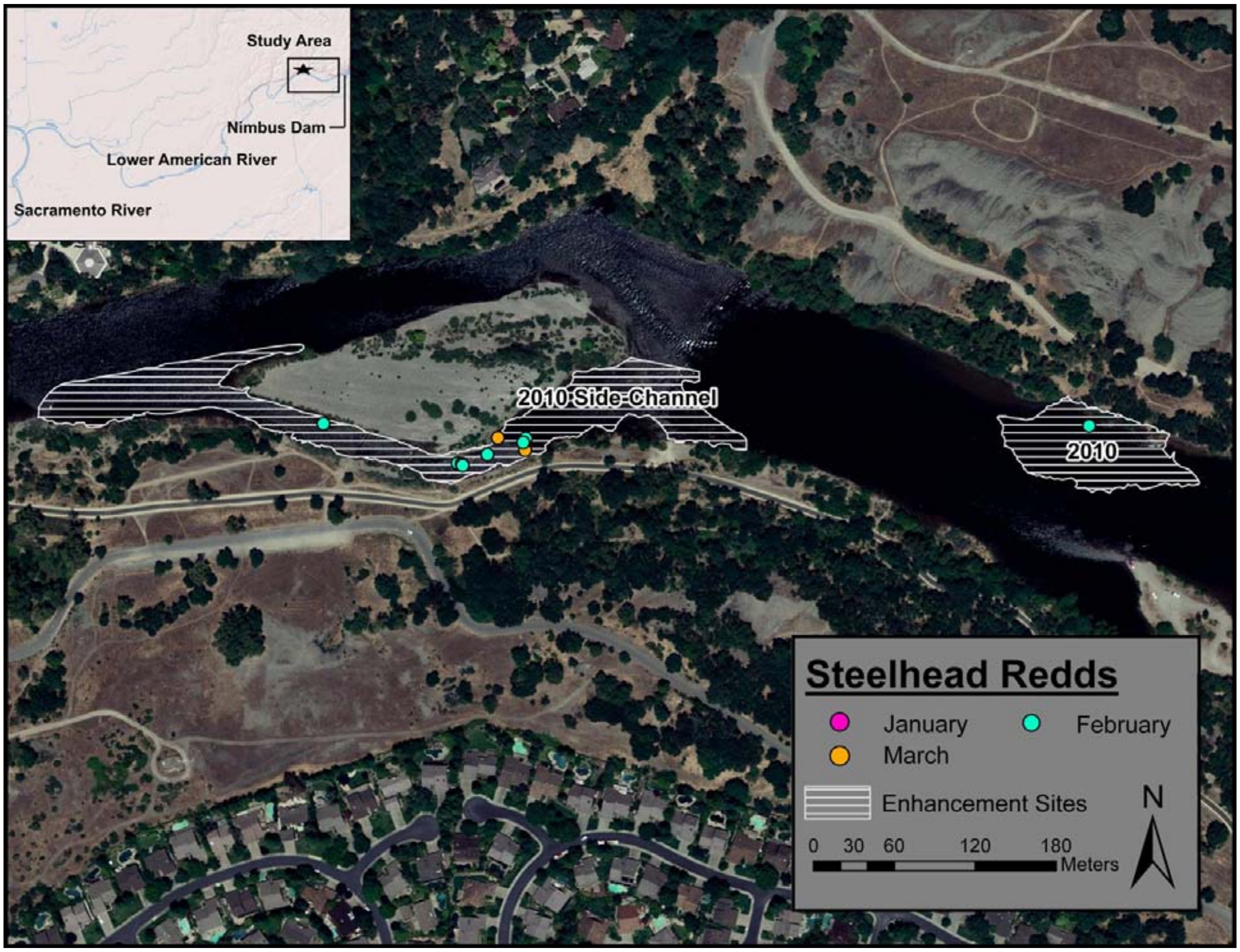


***Figure 32****. Locations of steelhead trout redds during the 2010–2011 spawning season. Redd locations are color-coded based on month of observation. Only observations from the 2010 augmentation site and the side-channel adjacent to the 2010 augmentation site are provided.*

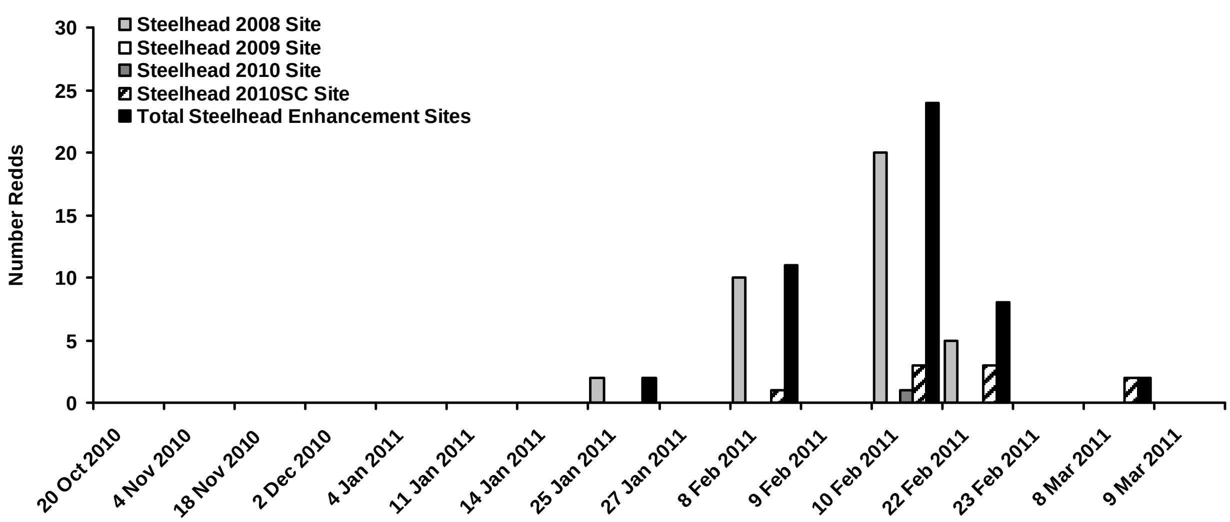


***Figure 33****. Number of steelhead trout redd observations at augmentation (i.e., enhancement) sites surveyed during the 2010–2011 spawning season.*

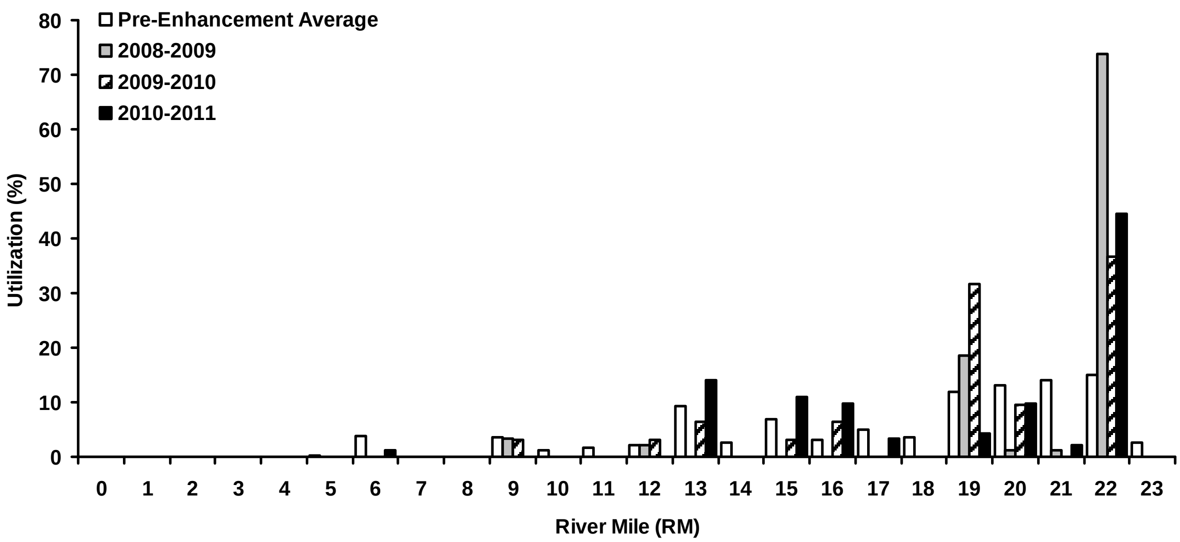


***Figure 34**. Utilization (%) rates calculated for each river mile (RM) based on steelhead trout redd observations for both pre- and post-enhancement conditions.*

## *Spawning Depth, Velocity, Substrate Size, Redd Characteristics, and Habitat Preferences*

### ***Depth and Velocity***

Depths selected by spawning Chinook salmon ranged from 0.18–0.85 m (average = 0.50 m) with peak HSI values from 0.2–0.8 m, whereas depths selected by spawning steelhead trout ranged from 0.11–1.70 m (average = 0.69 m) with peak HSI values from 0.1–1.2 m (Figure 35). Velocities selected by spawning Chinook salmon ranged from 0.19–2.00 m•s$^{-1}$ (average = 0.77 m•s$^{-1}$) with peak HSI values from 0.3–1.2 m•s$^{-1}$, whereas velocities selected by spawning steelhead trout ranged from 0.32–2.00 m•s$^{-1}$ (average = 0.87 m•s$^{-1}$) with peak HSI values from 0.3–1.4 m•s$^{-1}$ (Figure 35). Based on ANOVA results, the average depth selected by spawning Chinook salmon was significantly shallower than the average depth selected by spawning steelhead trout ($F = 15.023$; $df = 1, 115$; $p = 0.0002$). However, this significant difference may be related to both sampling location and gear biases. Chinook salmon were only sampled within augmentation sites with a limited range of depths using a 1.0 m top-setting rod, whereas steelhead trout were sampled throughout the entire LAR using a 2.0 m top-setting rod. No significant difference was observed for average velocities ($F = 3.041$; $df = 1, 115$; $p = 0.0839$; Figure 36). Similarly, chi-square analyses indicated a significantly greater proportion of Chinook salmon redds in below average depths when compared to expected values, with a significantly greater proportion of steelhead trout redds in above average depths ($\chi^2 = 6.577$; $df = 1, 117$; $p = 0.0103$). No significant difference was observed for velocities ($\chi^2 = 2.821$; $df = 1, 117$; $p = 0.0930$). The range of depths selected by Chinook salmon was ~2.4-fold narrower than the range of depths selected by steelhead trout, whereas velocity ranges were approximately equal in width (Chinook salmon ~1.1-fold wider than steelhead trout).

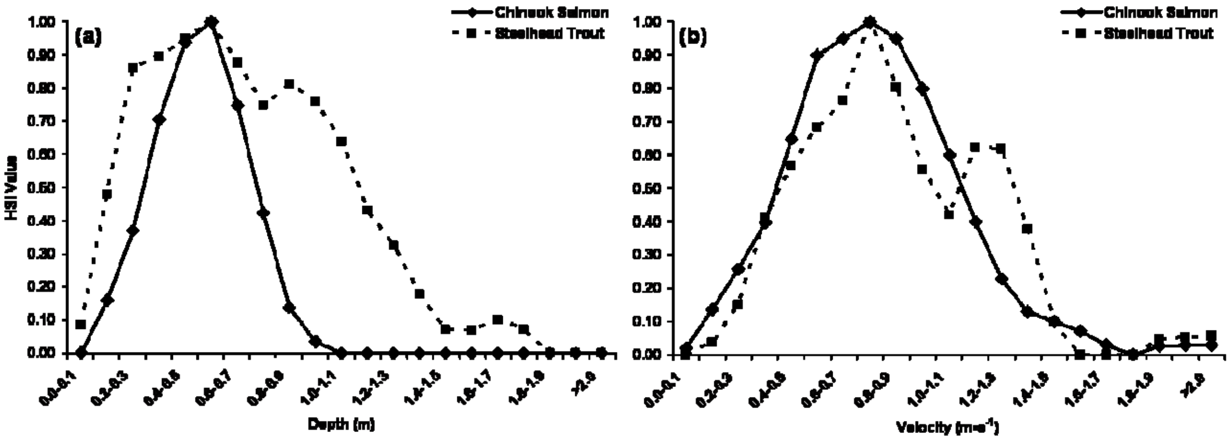


***Figure 35***. *Depth (a) and velocity (b) categories and corresponding HSI values for Chinook salmon and steelhead trout redd observations.*

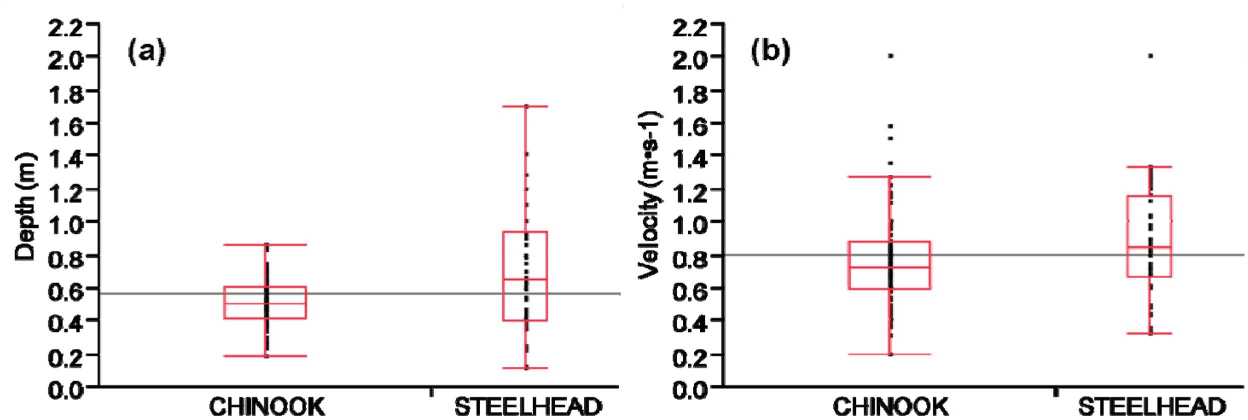


***Figure 36***. *Depths (a) and velocities (b) for Chinook salmon and steelhead trout redd observations. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

### *Substrate Size*

Substrate sizes selected by spawning Chinook salmon ranged from 12.5–99.8 mm $D_{50}$ (average = 30.0 mm) and 27.4–148.0 mm $D_{85}$ (average = 60.1 mm), with peak HSI values from 10–50 mm $D_{50}$ and 40–90 mm $D_{85}$ (Figure 37). Substrate sizes selected by spawning steelhead trout ranged from 24.5–56.1 mm $D_{50}$ (average = 28.8 mm) and 53.2–106.4 mm $D_{85}$ (average = 58.1 mm), with peak HSI values from 10–40 mm $D_{50}$ and 40–70 mm $D_{85}$ (Figure 37). Based on ANOVA results, no significant differences were observed for average $D_{50}$ ($F = 0.579$; $df = 1, 299$; $p = 0.4474$) or $D_{85}$ ($F = 1.020$; $df = 1, 299$; $p = 0.3134$) substrate sizes selected by spawning Chinook salmon and steelhead trout (Figure 38). Similarly, chi-square analyses indicated no significant differences in the proportion of Chinook salmon or steelhead trout redds in substrates with above or below average $D_{50}$s ($\chi^2 = 1.006$; $df = 1, 301$; $p = 0.3160$) or $D_{85}$s ($\chi^2 = 0.490$; $df = 1, 301$; $p = 0.4840$) when compared to expected values. The average $D_{50}$ substrate size selected by Chinook salmon was 1.25-fold larger than the $D_{50}$ of the 2008 site, 1.17-fold smaller than that of the 2009 site, and equal to that of the 2010 site, whereas the average $D_{50}$ substrate size selected by steelhead trout was 1.20-fold larger than the $D_{50}$ of the 2008 site, 1.22-fold smaller than that of the 2009 site, and 1.04-fold smaller than that of the 2010 site. The range of substrate sizes selected by Chinook salmon was ~2.3 to 2.8-fold ($D_{85}$ to $D_{50}$) wider than the range of substrate sizes selected by steelhead trout; possibly due to a wider range of fork lengths (FLs) observed for Chinook salmon than steelhead trout.

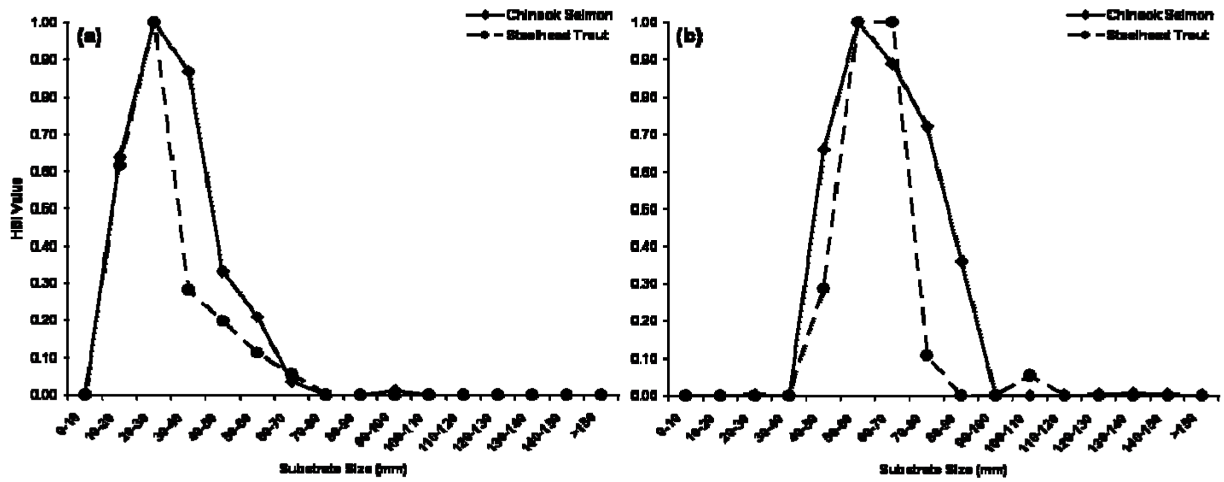


***Figure 37***. *Substrate categories and corresponding $D_{50}$ (a) and $D_{85}$ (b) HSI values for Chinook salmon and steelhead trout redd observations.*

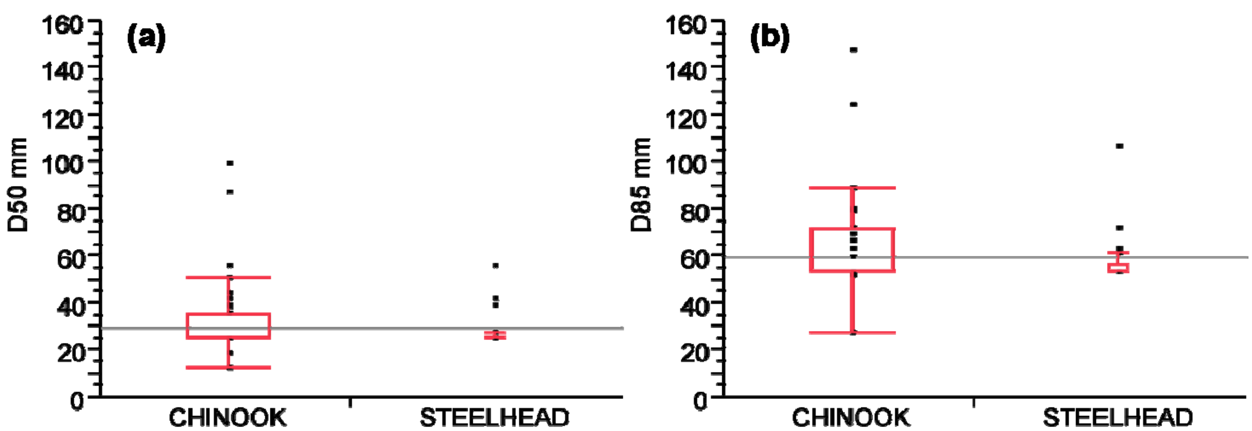


***Figure 38***. *Substrate $D_{50}$ (a) and $D_{85}$ (b) values for Chinook salmon and steelhead trout redd observations. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

Maximum movable substrate sizes calculated for spawning Chinook salmon ranged from 25.0–125.0 mm (average = 83.3 mm) with peak HSI values from 40–90 mm ($D_{50max}$ = 85.8 mm and $D_{85max}$ = 94.6 mm). Maximum movable substrate sizes calculated for spawning steelhead trout ranged from 40.0–90.0 mm (average = 66.5 mm) with peak HSI values from 50–80 mm ($D_{50max}$ = 65.0 mm and $D_{85max}$ = 71 mm; Figure 39). Based on ANOVA results, average maximum movable substrate sizes calculated for spawning Chinook salmon were significantly larger than average maximum movable substrate sizes calculated for spawning steelhead trout ($F = 859.576$; $df = 1, 10{,}773$; $p < 0.0001$; Figure 40). Similarly, chi-square analyses indicated a significantly greater proportion of Chinook salmon with above average calculated maximum movable substrate sizes when compared to expected values, whereas a significantly greater proportion of steelhead trout had below average calculated maximum movable substrate sizes ( $\chi^2$ = 545.758; $df$ = 1, 10,775; $p$ <0.0001). Average maximum movable substrate sizes calculated for Chinook salmon and steelhead trout were all >1.90-fold larger than the $D_{50}$ values of the 2008–2010 sites. However, when compared to the $D_{95}$s of the 2008–2010 sites (i.e., more appropriate index that may indicate potential spawning limitations), the average maximum movable substrate size calculated for Chinook salmon was 1.63-fold larger than the 2008 site, 1.34-fold smaller than the 2009 site, and 1.07-fold larger than the 2010 site, whereas the average maximum movable substrate size calculated for steelhead trout was 1.30-fold larger than the 2008 site, 1.68-fold smaller than the 2009 site, and 1.17-fold smaller than the 2010 site. When cumulative maximum movable substrate size distributions for Chinook salmon and steelhead trout were compared to the $D_{95}$s of the 2008–2010 sites, assuming maximum movable substrate size must be larger than $D_{95}$s for successful spawning, 0.36% of Chinook salmon in the LAR would be prevented from spawning at the 2008 site, 99.86% would be prevented from spawning at the 2009 site, and 28.30% would be prevented from spawning at the 2010 site, whereas 3.02% of steelhead trout in the LAR would be prevented from spawning at the 2008 site, 100.00% would be prevented from spawning at the 2009 site, and 95.88% would be prevented from spawning at the 2010 site (Figure 41). The range of maximum movable substrate sizes calculated for Chinook salmon was 2.0-fold wider than the range of maximum movable substrate sizes calculated for steelhead trout.

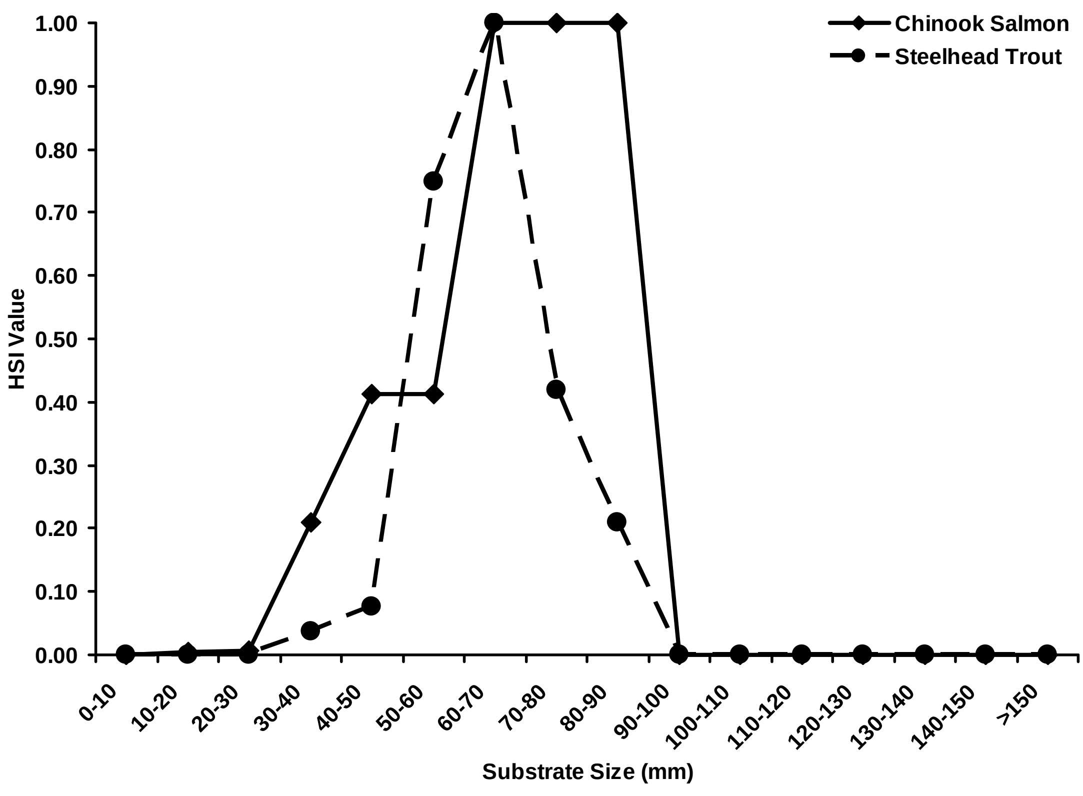


***Figure 39***. *Substrate categories and corresponding maximum movable substrate size (Kondolf 2000) HSI values for Chinook salmon and steelhead trout.*

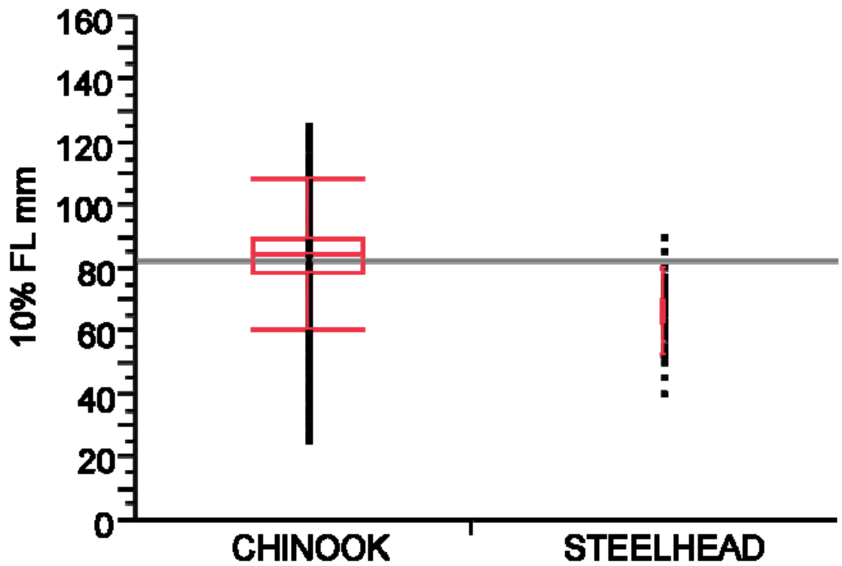


***Figure 40***. *Maximum movable substrate sizes (i.e., 10% FL) calculated for spawning Chinook salmon and steelhead trout. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

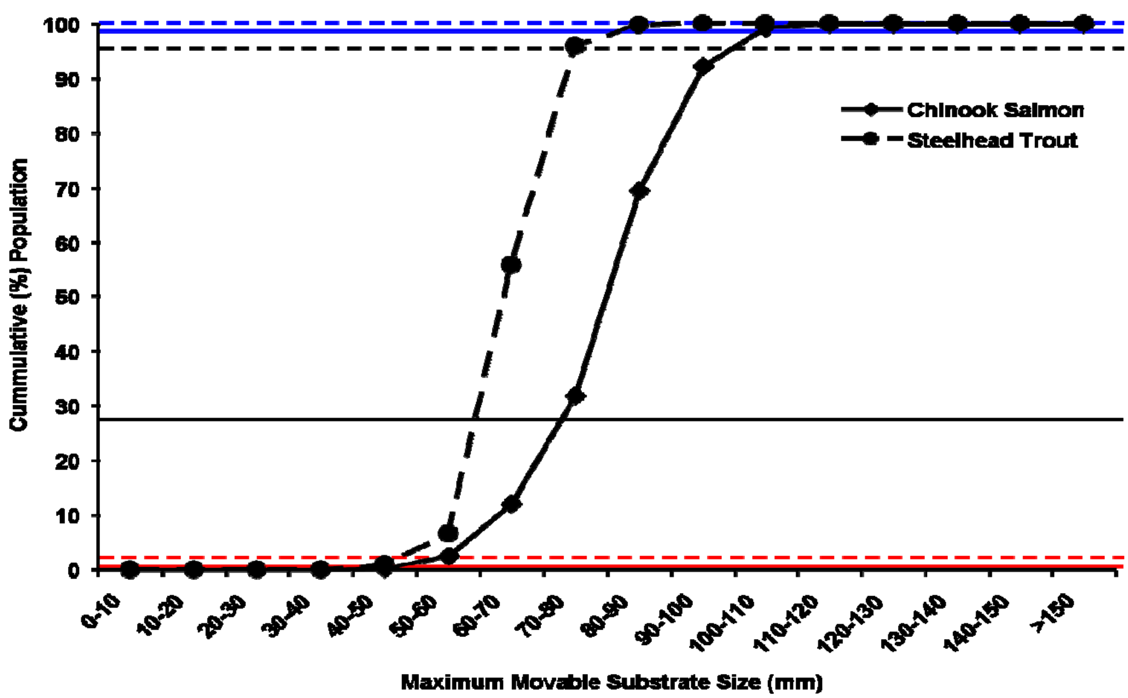


***Figure 41***. *Cumulative maximum movable substrate size distributions for Chinook salmon and steelhead trout in the lower American River and corresponding percent (%) of population prevented from spawning in the 2008–2010 enhancement sites based on $D_{95}$s. Percent (%) of population below horizontal lines would be prevented from spawning at each site based on site $D_{95}$s. Solid horizontal lines represent Chinook salmon, whereas dashed horizontal lines represent steelhead trout. Red horizontal lines correspond to the 2008 site, blue horizontal lines correspond to the 2009 site, and black horizontal lines correspond to the 2010 site. Because values for percent (%) of population prevented from spawning were calculated prior to categorization of substrate sizes, horizontal lines may not be aligned exactly with cumulative distribution curves.*

When ANOVAs were used to compare substrate sizes selected by spawning Chinook salmon to what was expected based on maximum movable substrate size calculations, average substrate sizes selected were significantly smaller than expected for both average $D_{50}$ ($F = 5{,}918.147$; $df = 1, 10{,}661$; $p < 0.0001$) and $D_{85}$ ($F = 1{,}104.506$; $df = 1, 10{,}661$; $p < 0.0001$; Figure 42). Similarly, chi-square analyses indicated a significantly greater proportion of maximum movable substrate sizes above average values when compared to overall expected values, with significantly greater proportions of selected $D_{50}$ ($\chi^2 = 426.157$; $df = 1, 10{,}663$; $p < 0.0001$) and $D_{85}$ ($\chi^2 = 317.969$; $df = 1, 10{,}663$; $p < 0.0001$) substrate sizes below average values. Based on average values, selected substrate sizes were ~1.4 to 2.8-fold ($D_{85}$ to $D_{50}$) smaller than maximum movable substrate sizes. When ANOVAs were used to compare substrate sizes selected by spawning steelhead trout to what was expected based on maximum movable substrate size calculations, average substrate sizes selected were significantly smaller than expected for both average $D_{50}$ ($F = 1{,}302.284$; $df = 1, 411$; $p < 0.0001$) and $D_{85}$ ($F = 54.921$; $df = 1, 411$; $p < 0.0001$; Figure 42). Similarly, chi-square analyses indicated a significantly greater proportion of maximum movable substrate sizes above average values when compared to overall expected values, with significantly greater proportions of selected $D_{50}$ ($\chi^2 = 121.295$; $df = 1, 413$; $p < 0.0001$) and $D_{85}$ ($\chi^2 = 24.702$; $df = 1, 413$; $p < 0.0001$) substrate sizes below average values. Based on average values, selected substrate sizes were ~1.1 to 2.3-fold ($D_{85}$ to $D_{50}$) smaller than maximum movable substrate sizes.

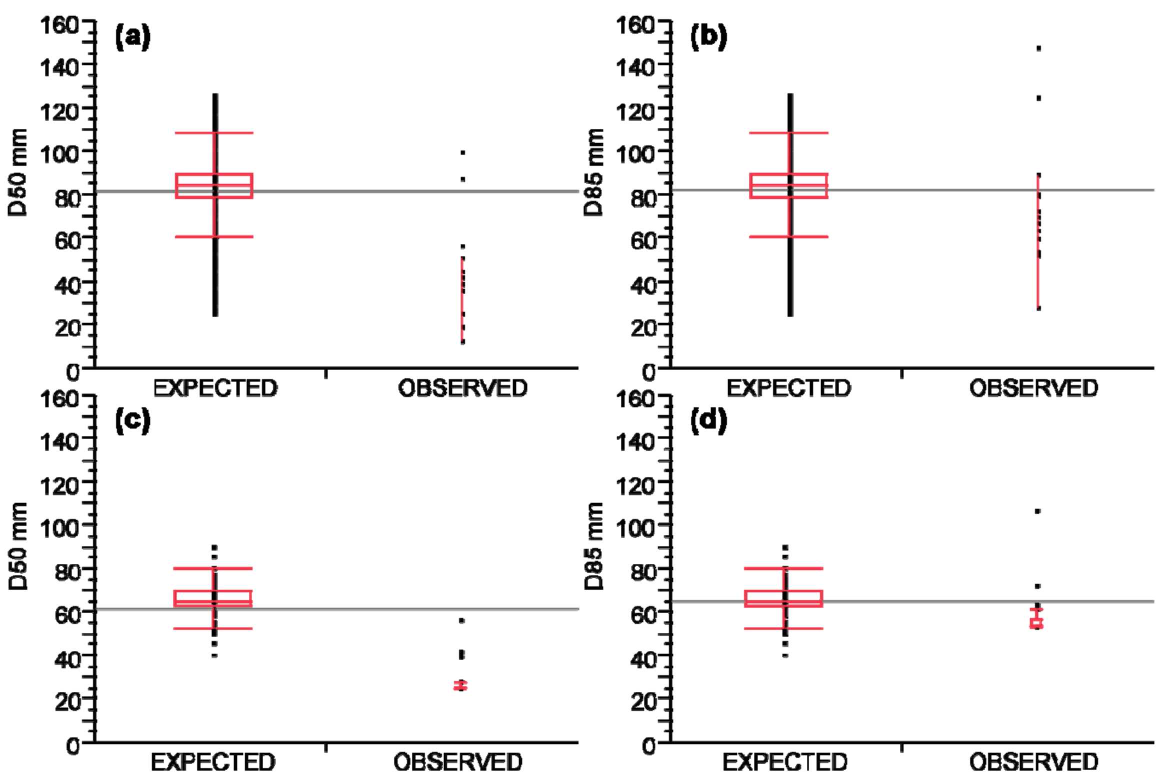


***Figure 42***. *Calculated maximum movable substrate sizes (expected) and observed $D_{50}$s (a and c) and $D_{85}$s (b and d) for spawning Chinook salmon (upper panels) and steelhead trout (lower panels). Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean. Figure compares observed $D_{50}$s and $D_{85}$s to raw 10% FL values.*

Lengths for fish observed on redds ranged from 0.55–1.00 m (average = 0.81 m) for Chinook salmon, 0.60–0.85 m (average = 0.69 m) for steelhead trout, and 0.55–1.00 m (average = 0.76 m) for total salmonids. Corresponding $D_{50}$s ranged from 19.1–56.2 mm (average = 34.2 mm) for Chinook salmon, 24.5–42.2 mm (average = 29.8 mm) for steelhead trout, and 19.1–56.2 mm (average = 32.1 mm) for total salmonids, whereas $D_{85}$s ranged from 51.2–88.9 mm (average = 65.4 mm) for Chinook salmon, 53.2–72.2mm (average = 57.9 mm) for steelhead trout, and 51.2–88.9 mm (average = 61.86 mm) for total salmonids. When ANOVAs were used to compare average lengths for fish observed on redds in substrates with "ABOVE" average $D_{50}$s and $D_{85}$s to fish observed on redds in substrates with "BELOW" average $D_{50}$s and $D_{85}$s, no significant differences were observed for Chinook salmon $D_{50}$ ($F = 0.133$; $df = 1, 33$; $p = 0.7179$) or $D_{85}$ ($F = 0.344$; $df = 1, 33$; $p = 0.5616$), steelhead trout $D_{50}$ ($F = 3.885$; $df = 1, 29$; $p = 0.0583$) or $D_{85}$ ($F = 3.507$; $df = 1, 29$; $p = 0.0712$), or total salmonid $D_{50}$ ($F = 0.175$; $df = 1, 64$; $p = 0.6769$) or $D_{85}$ ($F = 0.265$; $df = 1, 64$; $p = 0.6082$; Figure 43). Similarly, when chi-square analyses were used to compare the expected proportions of fish in each of the four "ABOVE" and "BELOW" combination categories (see above) to observed proportions of fish in each of the four combination categories, no significant differences were observed for Chinook salmon $D_{50}$ ($X^2 = 0.698$; $df = 1, 35$; $p = 0.4035$) or $D_{85}$ ($X^2 = 1.400$; $df = 1, 35$; $p = 0.2367$), steelhead trout $D_{50}$ ($X^2 = 1.314$; $df = 1, 31$; $p = 0.2517$) or $D_{85}$ ($X^2 = 0.606$; $df = 1, 31$; $p = 0.4364$), or total salmonid $D_{50}$ ($X^2 = 0.216$; $df = 1, 66$; $p = 0.6421$) or $D_{85}$ ($X^2 = 0.420$; $df = 1, 66$; $p = 0.5168$). In general, a wider range of steelhead trout sizes were observed spawning in "BELOW" average substrates when compared to "ABOVE" average substrates, whereas the observed size ranges for Chinook salmon and total salmonids were similar for each substrate category.

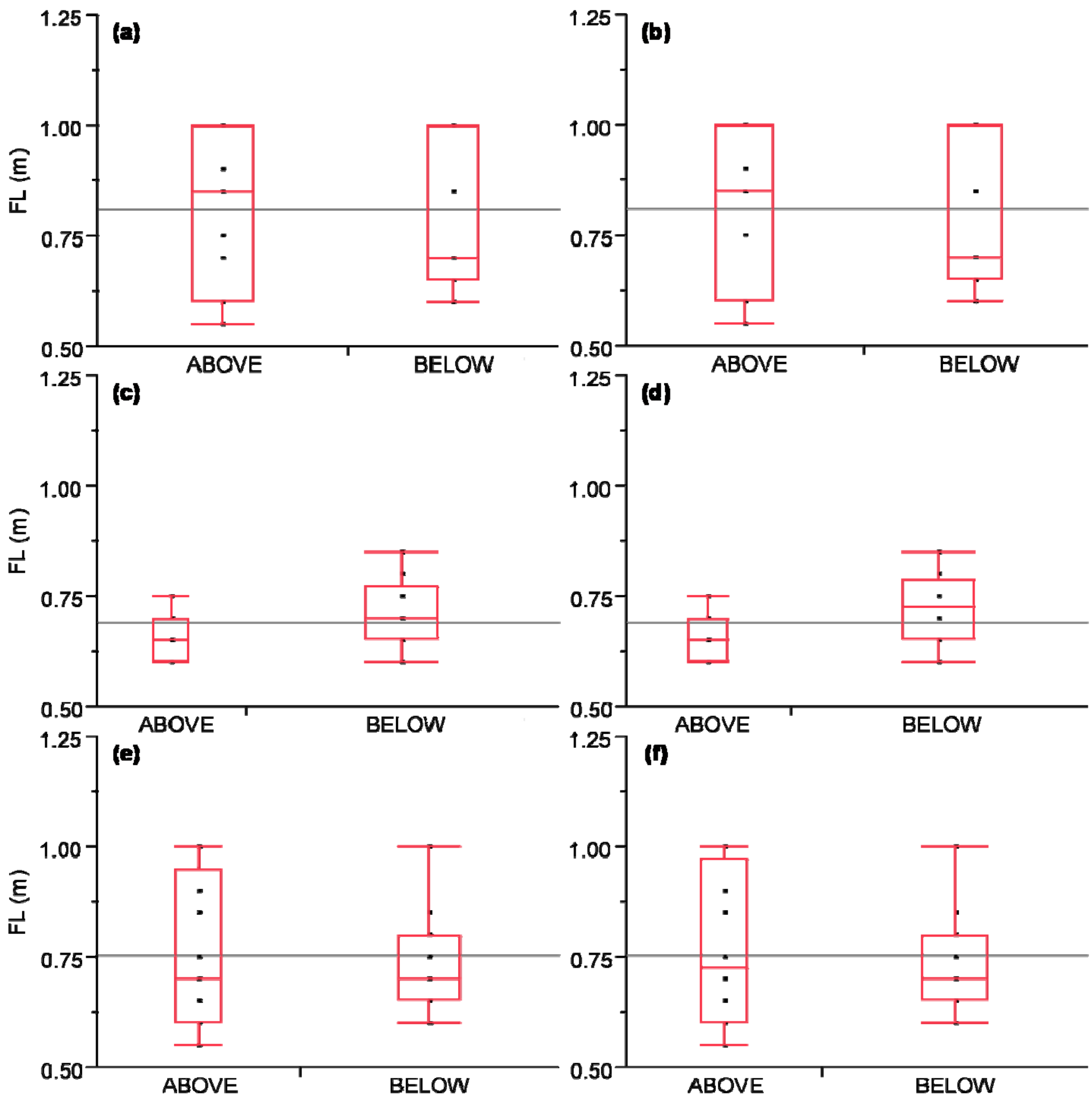


***Figure 43****.* *Fork lengths (FLs) for spawning Chinook salmon (a and b), steelhead trout (c and d), and total salmonids (e and f) associated with redd observations in "above" and "below" average $D_{50}$ (a, c, and e) and $D_{85}$ (b, d, and f) substrates. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean. Data represent $D_{50}$s and $D_{85}$s associated with redd observations for which fish observations were available. In general, a wider range of steelhead trout sizes were observed spawning in "BELOW" average substrates when compared to "ABOVE" average substrates, whereas the observed size ranges for Chinook salmon and total salmonids were similar for each substrate category.*

### *Redd Characteristics*

Chinook salmon redd lengths ranged from 0.35–4.00 m (average = 1.20 m), whereas redd widths ranged from 0.30–3.50 m (average = 0.94 m) and tail lengths ranged from 0.60–4.70 m (average = 2.14 m). Corresponding $D_{50}$s ranged from 19.1–99.8 mm (average = 36.1 mm), whereas $D_{85}$s ranged from 51.2–148.0 mm (average = 67.3 mm). When ANOVAs were used to compare average redd characteristics for redd observations located in substrates with above average $D_{50}$s and $D_{85}$s to substrates with below average $D_{50}$s and $D_{85}$s, no significant differences were observed for redd length and $D_{50}$ ($F = 1.843$; $df = 1, 74$; $p = 0.1787$) or $D_{85}$ ($F = 1.2366$; $df = 1, 74$; $p = 0.2697$), redd width and $D_{50}$ ($F = 0.014$; $df = 1, 73$; $p = 0.9070$) or $D_{85}$ ($F = 0.599$; $df = 1, 73$; $p = 0.4416$), or tail length and $D_{50}$ ($F = 0.398$; $df = 1, 73$; $p = 0.5299$) or $D_{85}$ ($F = 0.009$; $df = 1, 73$; $p = 0.9256$; Figure 44). Similarly, when chi-square analyses were used to compare the expected proportions of fish in each of the four "ABOVE" and "BELOW" combination categories (see above) to observed proportions of fish in each of the four combination categories, no significant differences were observed for redd length and $D_{50}$ ($X^2 = 2.086$; $df = 1, 76$; $p = 0.1487$) or $D_{85}$ ($X^2 = 1.248$; $df = 1, 76$; $p = 0.2640$), redd width and $D_{50}$ ($X^2 = 0.428$; $df = 1, 75$; $p = 0.5129$) or $D_{85}$ ($X^2 = 0.027$; $df = 1, 75$; $p = 0.8700$), or tail length and $D_{50}$ ($X^2 = 1.270$; $df = 1, 75$; $p = 0.2597$) or $D_{85}$ ($X^2 = 0.609$; $df = 1, 75$; $p = 0.4352$). When compared to redd observations in substrates with below average $D_{50}$s and $D_{85}$s, the range of measured values for redd observations in substrates with above average $D_{50}$s and $D_{85}$s was similar (<1.2-fold wider – $D_{50}$ and $D_{85}$) for redd length, ~1.3-fold wider ($D_{50}$ and $D_{85}$) for redd width, and similar (~1.1-fold narrower – $D_{50}$ and $D_{85}$) for tail length.

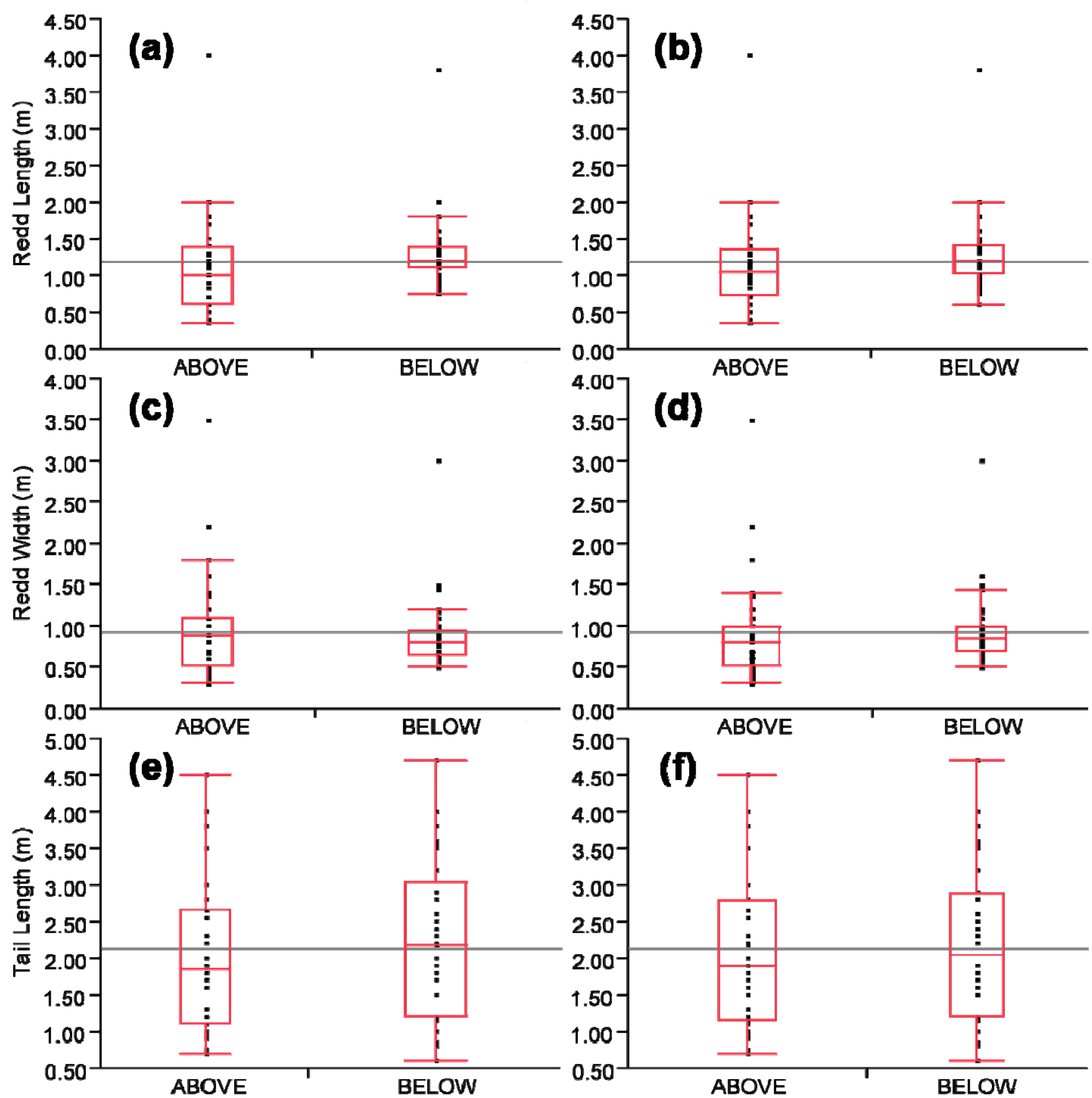


***Figure 44**. Redd lengths (a and b), redd widths (c and d), and tail lengths (e and f) for Chinook salmon redd observations with "ABOVE" and "BELOW" average $D_{50}$s (a, c, and e) and $D_{85}$s (b, d, and f). Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean. Data represent $D_{50}$s and $D_{85}$s associated with redd observations for which redd characteristics were available.*

Steelhead trout redd lengths ranged from 0.25–2.50 m (average = 0.90 m), whereas redd widths ranged from 0.20–1.60 m (average = 0.91 m), redd depths ranged from 0.10–0.80 m (average = 0.30 m), tail lengths ranged from 0.20–5.00 m (average = 1.72 m), and average tail widths ranged from 0.20–1.68 m (average = 0.76 m). Corresponding $D_{50}$s ranged from 24.5–56.1 mm (average = 30.0 mm), whereas $D_{85}$s ranged from 53.2–106.4 mm (average = 59.2 mm). When ANOVAs were used to compare average redd characteristics for redd observations located in substrates with above average $D_{50}$s and $D_{85}$s to substrates with below average $D_{50}$s and $D_{85}$s, no significant differences were observed for redd length and $D_{50}$ ($F = 2.156$; $df = 1, 40$; $p = 0.1498$) or $D_{85}$ ($F = 1.531$; $df = 1, 40$; $p = 0.2232$), redd width and $D_{50}$ ($F = 0.213$; $df = 1, 40$; $p = 0.6468$) or $D_{85}$ ($F = 0.009$; $df = 1, 40$; $p = 0.9256$), redd depth and $D_{50}$ ($F = 0.282$; $df = 1, 34$; $p = 0.5988$) or $D_{85}$ ($F = 0.482$; $df = 1, 34$; $p = 0.4922$), or tail width and $D_{50}$ ($F = 2.981$; $df = 1, 40$; $p = 0.0920$) or $D_{85}$ ($F = 1.667$; $df = 1, 40$; $p = 0.2040$). In contrast, significant differences were observed for tail length and both $D_{50}$ ($F = 4.594$; $df = 1, 40$; $p = 0.0382$) and $D_{85}$ ($F = 4.608$; $df = 1, 40$; $p = 0.0379$; Figure 45). Similarly, when chi-square analyses were used to compare the expected proportions of fish in each of the four "ABOVE" and "BELOW" combination categories (see above) to observed proportions of fish in each of the four combination categories, no significant differences were observed for redd length and $D_{50}$ ($\chi^2 = 2.840$; $df = 1, 42$; $p = 0.0919$) or $D_{85}$ ($\chi^2 = 1.732$; $df = 1, 42$; $p = 0.1881$), redd width and $D_{50}$ ($\chi^2 = 0.361$; $df = 1, 42$; $p = 0.5480$) or $D_{85}$ ($\chi^2 = 0.032$; $df = 1, 42$; $p = 0.8584$), redd depth and $D_{50}$ ($\chi^2 = 0.090$; $df = 1, 36$; $p = 0.7642$) or $D_{85}$ ($\chi^2 = 0.590$; $df = 1, 36$; $p = 0.4423$), tail length and $D_{50}$ ($\chi^2 = 3.694$; $df = 1, 42$; $p = 0.0546$), or tail width and $D_{50}$ ($\chi^2 = 1.883$; $df = 1, 42$; $p = 0.1699$) or $D_{85}$ ($\chi^2 = 0.826$; $df = 1, 42$; $p = 0.3634$). In contrast, significant differences were observed for tail length and $D_{85}$ ($\chi^2 = 4.591$; $df = 1, 42$; $p = 0.0321$). When compared to redd observations in substrates with below average $D_{50}$s and $D_{85}$s, the range of measured values for redd observations in substrates with above average $D_{50}$s and $D_{85}$s was ~2.4-fold narrower ($D_{50}$ and $D_{85}$) for redd length, similar (<1.2-fold narrower – $D_{50}$ and $D_{85}$) for redd width, equal (i.e., 1.0-fold variation) for redd depth, ~3.0-fold narrower ($D_{50}$ and $D_{85}$) for tail length, and ~1.5 to 1.8-fold narrower ($D_{85}$ to $D_{50}$) for tail width.

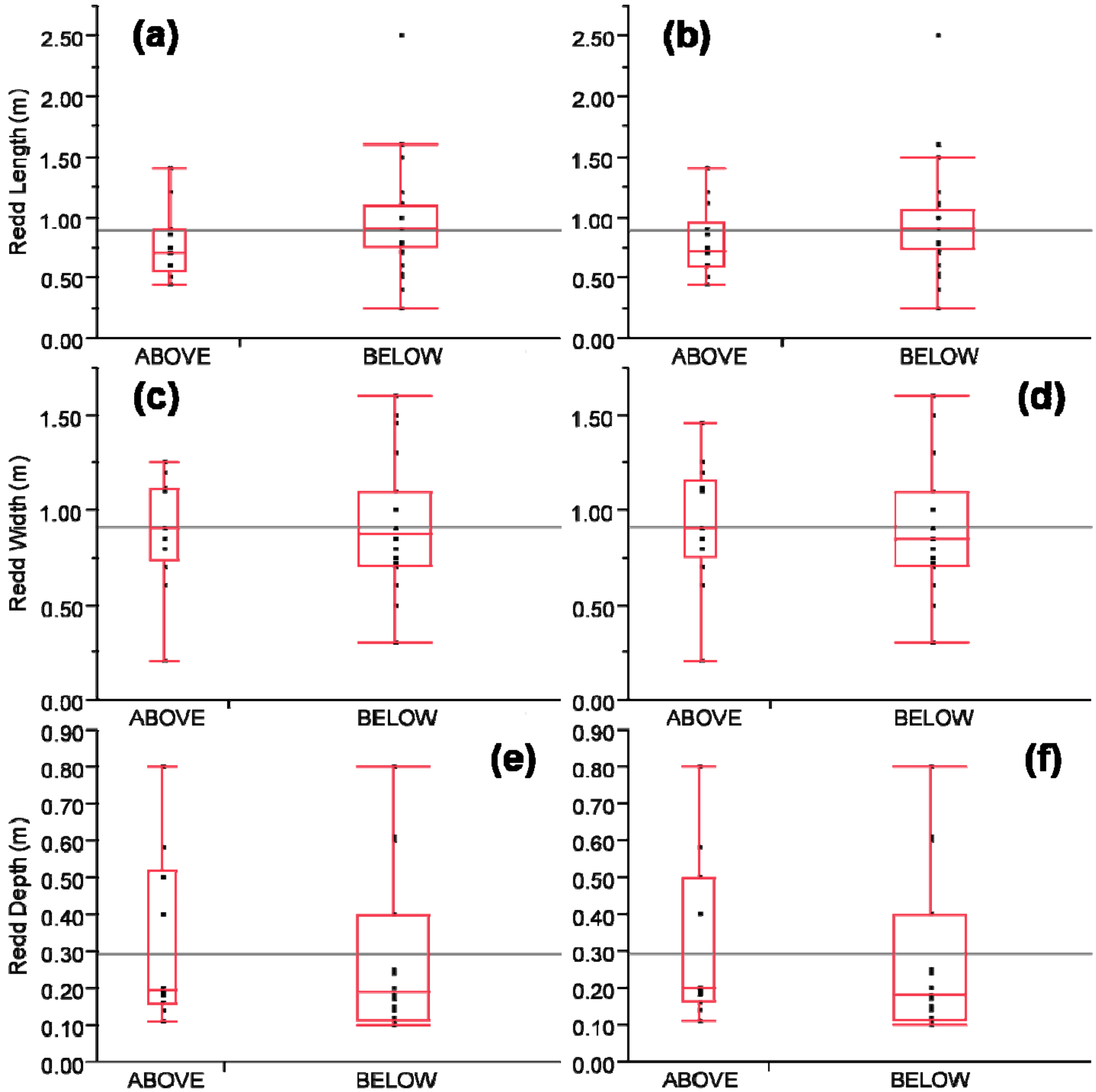


***Figure 45**. Redd lengths (a and b), redd widths (c and d), redd depths (i.e., pocket depths; e and f), tail lengths (g and h), and average tail widths (i and j) for steelhead trout redd observations with "ABOVE" and "BELOW" average $D_{50}$s (a, c, e, g, and i) and $D_{85}$s (b, d, f, h, and j). Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean. Data represent $D_{50}$s and $D_{85}$s associated with redd observations for which redd characteristics were available.*

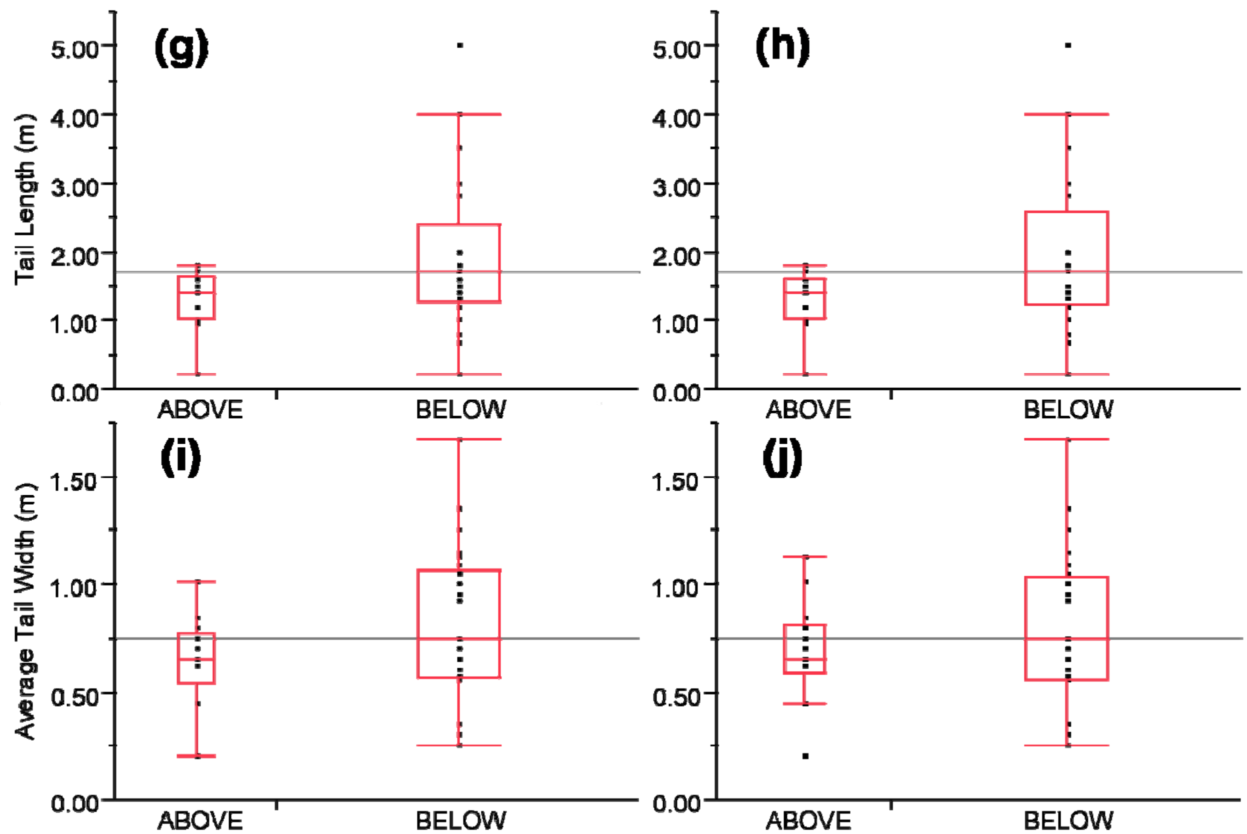


***Figure 45*** *(continued). Redd lengths (a and b), redd widths (c and d), redd depths (i.e., pocket depths; e and f), tail lengths (g and h), and average tail widths (i and j) for steelhead trout redd observations with "ABOVE" and "BELOW" average $D_{50}$s (a, c, e, g, and i) and $D_{85}$s (b, d, f, h, and j). Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean. Data represent $D_{50}$s and $D_{85}$s associated with redd observations for which redd characteristics were available.*

Total salmonid redd lengths ranged from 0.25–4.00 m (average = 1.09 m), whereas redd widths ranged from 0.20–3.50 m (average = 0.93 m), and tail lengths ranged from 0.20–5.00 m (average = 1.99 m). Corresponding $D_{50}$s ranged from 19.1–99.8 mm (average = 34.0 mm), whereas $D_{85}$s ranged from 51.2–148.0 mm (average = 64.5 mm). When ANOVAs were used to compare average redd characteristics for redd observations located in substrates with above average $D_{50}$s and $D_{85}$s to substrates with below average $D_{50}$s and $D_{85}$s, no significant differences were observed for redd length and $D_{50}$ ($F$ = 1.092; $df$ = 1, 115; $p$ = 0.2983) or $D_{85}$ ($F$ = 0.0690; $df$ = 1, 115; $p$ = 0.7932), redd width and $D_{50}$ ($F$ = 0.387; $df$ = 1, 114; $p$ = 0.5353) or $D_{85}$ ($F$ = 0.407; $df$ = 1, 114; $p$ = 0.5250), or tail length and $D_{50}$ ($F$ = 0.315; $df$ = 1, 114; $p$ = 0.5760) or $D_{85}$ ($F$ = 0.1297; $df$ = 1, 114; $p$ = 0.7194; Figure 46). Similarly, when chi-square analyses were used to compare the expected proportions of fish in each of the four "ABOVE" and "BELOW" combination categories (see above) to observed proportions of fish in each of the four combination categories, no significant differences were observed for redd length and $D_{50}$ ($\chi^2$ = 1.741; $df$ = 1, 118; $p$ = 0.1870) or $D_{85}$ ($\chi^2$ = 0.198; $df$ = 1, 118; $p$ = 0.6563), redd width and $D_{50}$ ($\chi^2$ = 0.287; $df$ = 1, 117; $p$ = 0.5921) or $D_{85}$ ($\chi^2$ = 0.003; $df$ = 1, 117; $p$ = 0.9598), or tail length and $D_{50}$ ($\chi^2$ = 0.969; $df$ = 1, 117; $p$ = 0.3248) or $D_{85}$ ($\chi^2$ = 0.068; $df$ = 1, 117; $p$ = 0.7937). When compared to redd observations in substrates with below average $D_{50}$s and $D_{85}$s, the range of measured values for redd observations in substrates with above average $D_{50}$s and $D_{85}$s was equal (i.e., 1.0-fold variation) for redd length, similar (<1.2-fold wider – $D_{50}$ and $D_{85}$) for redd width, and similar (~1.1-fold narrower – $D_{50}$ and $D_{85}$) for tail length.

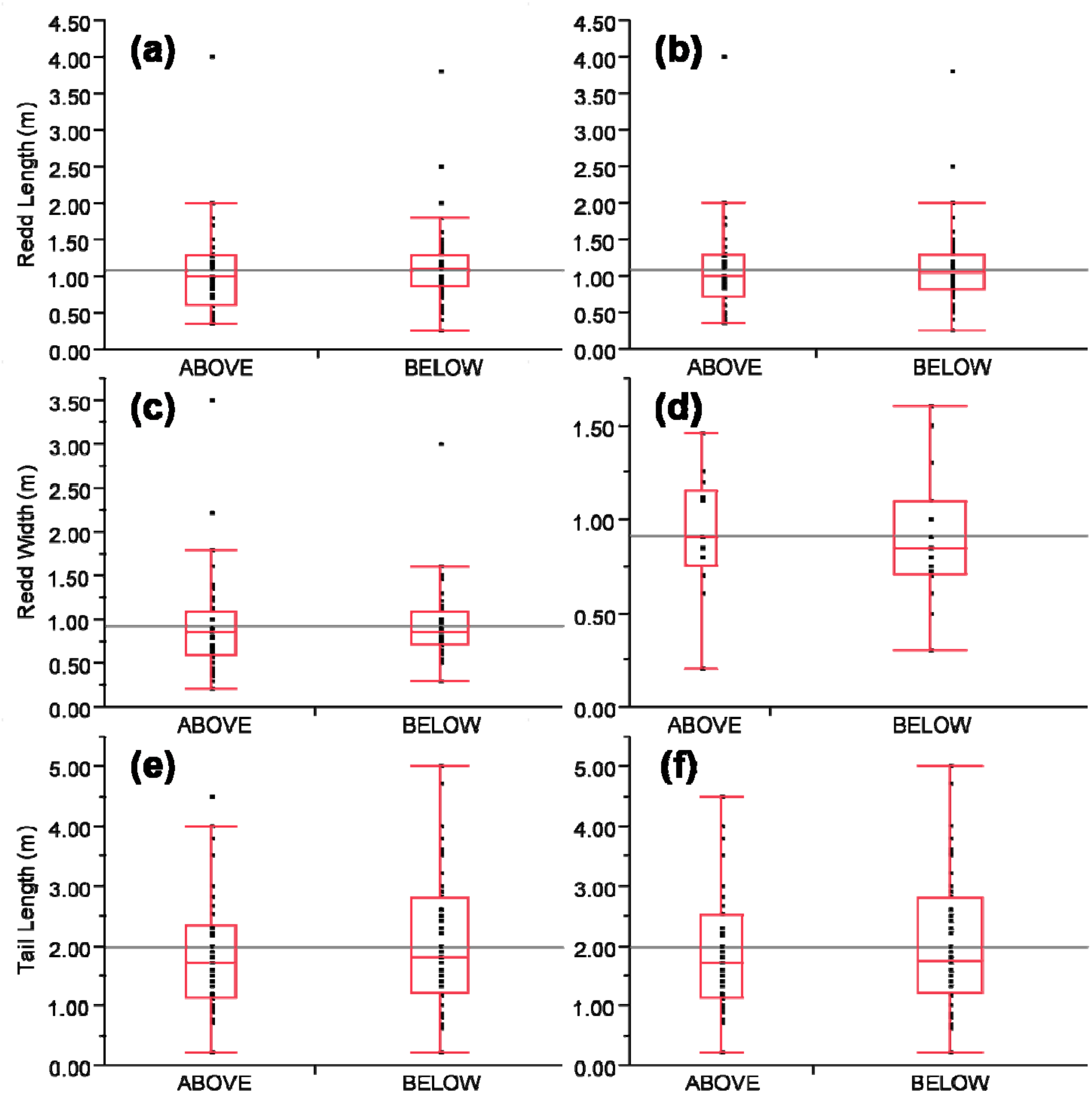


***Figure 46**. Redd lengths (a and b), redd widths (c and d), and tail lengths (e and f) for total salmonid redd observations with "ABOVE" and "BELOW" average $D_{50}$s (a, c, and e) and $D_{85}$s (b, d, and f). Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean. Data represent $D_{50}$s and $D_{85}$s associated with redd observations for which redd characteristics were available.*

### *Habitat Preferences*

The number of redd observations associated with cover features was 25 for Chinook salmon, five for steelhead trout, and 30 for total salmonids. Redds were most frequently associated with individual cover types or a combination of cover types including Brush (15 Chinook salmon, 4 steelhead trout, and 19 total salmonids), followed by Riparian Grass (9 Chinook salmon, 0 steelhead trout, and 9 total salmonids), Trees (6 Chinook salmon, 0 steelhead trout, and 6 total salmonids), Overhanging Vegetation (2 Chinook salmon, 0 steelhead trout, and 2 total salmonids), and Large Woody Material (0 Chinook salmon, 1 steelhead trout, and 1 total salmonids). No redds were associated with Small Woody Material or Cattail cover types (total number of redds may equal more than 30 due to multiple cover types assigned to single cover feature polygons; see above). A total of 271 redds were not associated with any cover type (Figure 47). In general, utilization (%) rates for all cover types were low, and ranged from 0.0–6.0% for Chinook salmon, 0.0–8.2% for steelhead trout, and 0.0–6.3% for total salmonids (Figure 48; total utilization rates may equal more than 100% due to multiple cover types assigned to single cover feature polygons; see above). Approximately 90% of Chinook salmon, steelhead trout, and total salmonids did not utilize cover of any type.

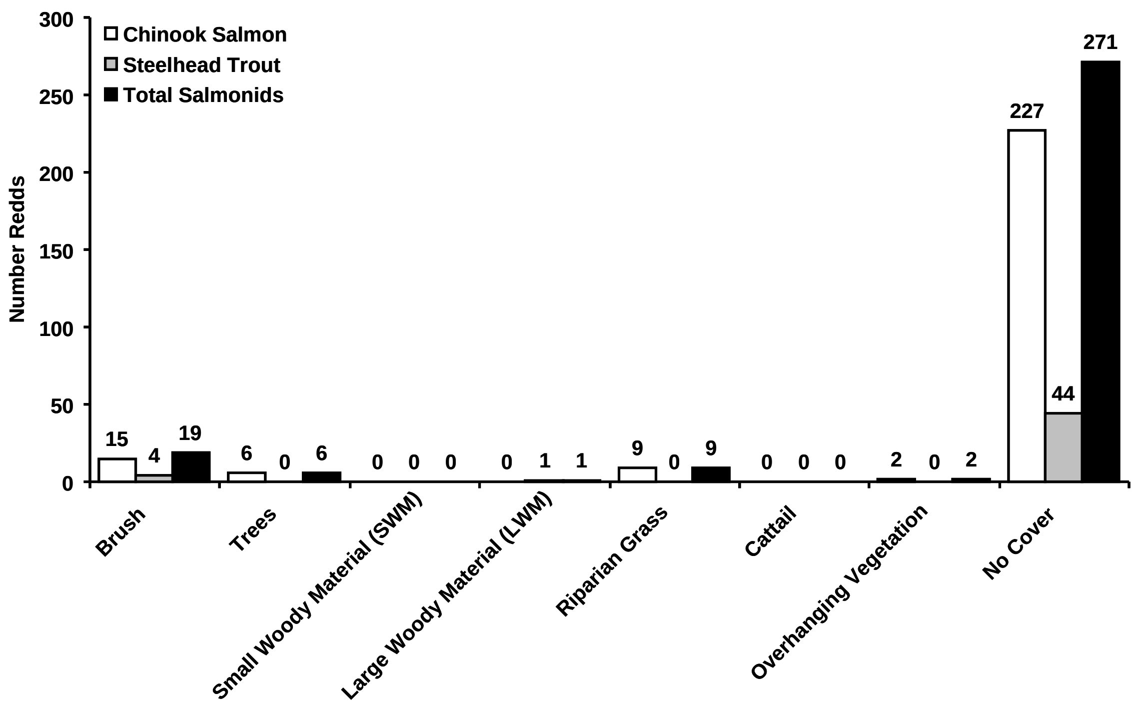


***Figure 47****. Number of Chinook salmon, steelhead trout, and total salmonid redds associated with individual cover types or a combination of cover types including individual cover types used for GIS-based habitat mapping. No Cover is provided for comparison purposes. The 2008–2010 augmentation sites were primarily associated with the No Cover category.*

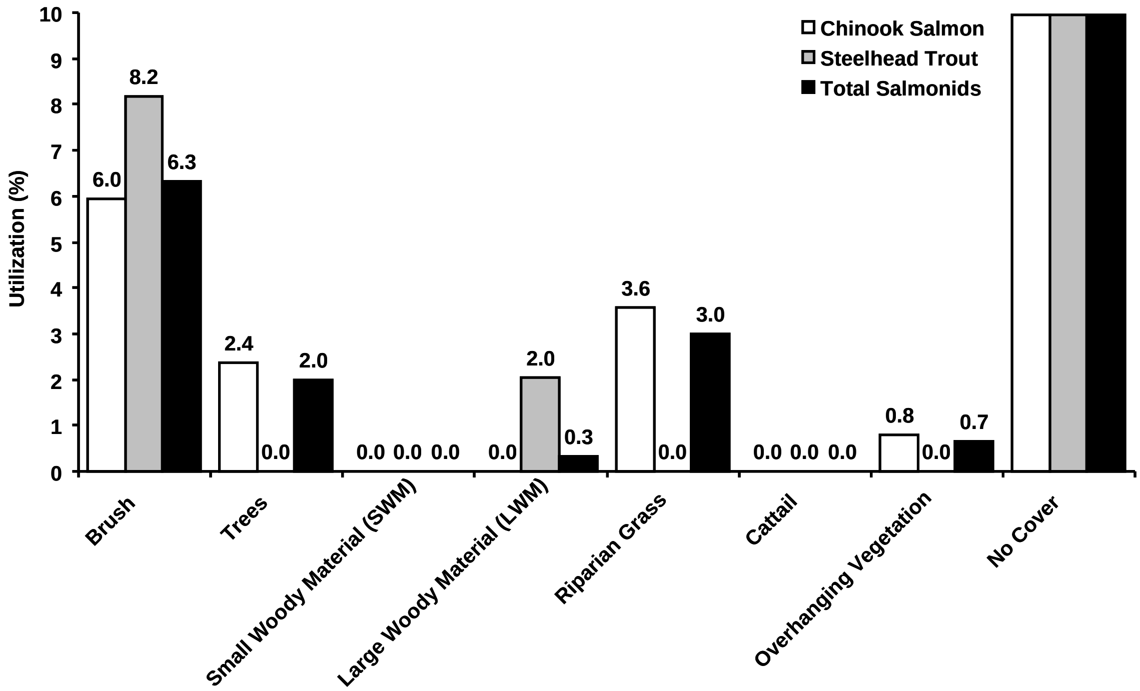


***Figure 48**. Utilization (%) rates for Chinook salmon, steelhead trout, and total salmonids associated with individual cover types or a combination of cover types including individual cover types used for GIS-based habitat mapping. No Cover is provided for comparison purposes. Rates for No Cover were ~90% for all categories. The 2008–2010 augmentation sites were primarily associated with the No Cover category. Figure is truncated at 10% utilization to display Utilization (%) rates associated with each cover type.*

Salmonid redds were most frequently associated with Main Channel units (212 Chinook salmon, 38 steelhead trout, and 250 total salmonids), followed by Side-Channel units (39 Chinook salmon, 10 steelhead trout, and 49 total salmonids), Floodplain units (19 Chinook salmon, 4 steelhead trout, and 23 total salmonids), Island units (13 Chinook salmon, 4 steelhead trout, and 17 total salmonids), and Main Channel/Floodplain units (5 Chinook salmon, 0 steelhead trout, and 5 total salmonids). No redds were associated with Scallops (Figure 49; total number of redds may equal more than 301 due to multiple channel unit types assigned to single redd observations; see above). Utilization (%) rates ranged from 67.9–84.1% for Main Channel units, 14.2–17.9% for Side-Channel units, 6.7–7.5% for Floodplain units, 4.9–7.1% for Island units, and 0.0–2.0% for Main Channel/Floodplain units (Figure 50; total utilization rates may equal more than 100% due to multiple channel unit types assigned to singe redd observations; see above).

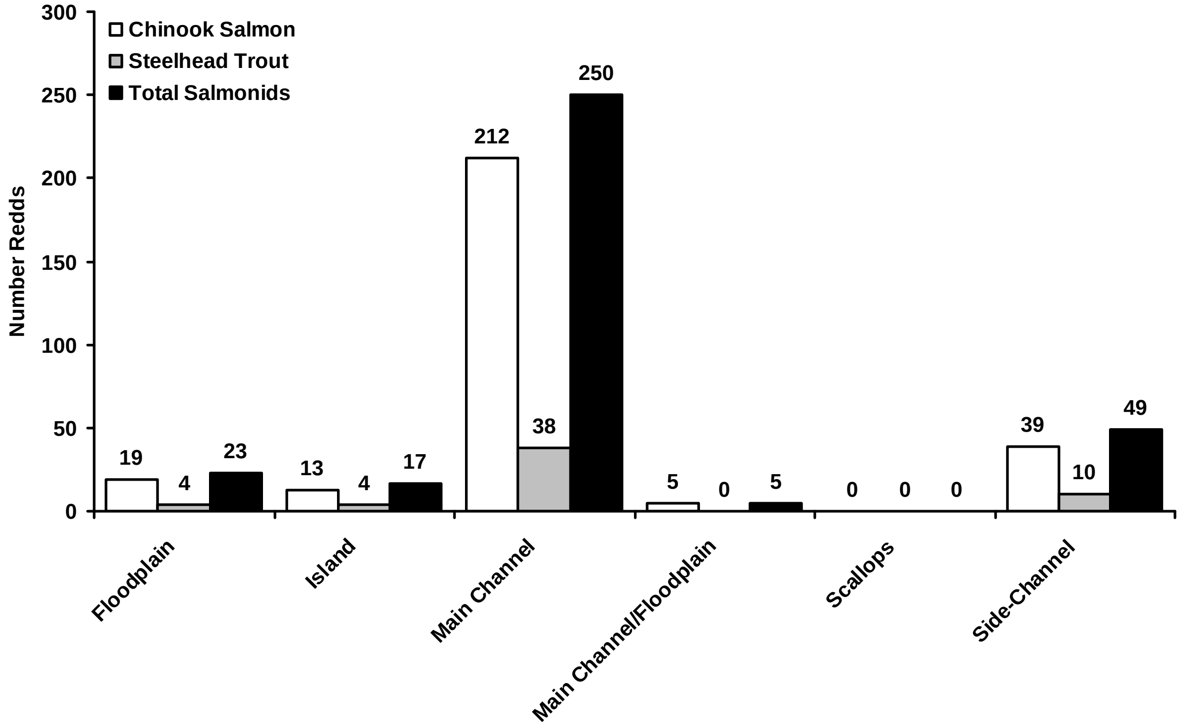


***Figure 49***. *Number of Chinook salmon, steelhead trout, and total salmonid redds associated with individual channel unit types or a combination of channel unit types including individual channel unit types used for GIS-based habitat mapping. The 2008–2010 augmentation sites were located primarily in Main Channel units.*

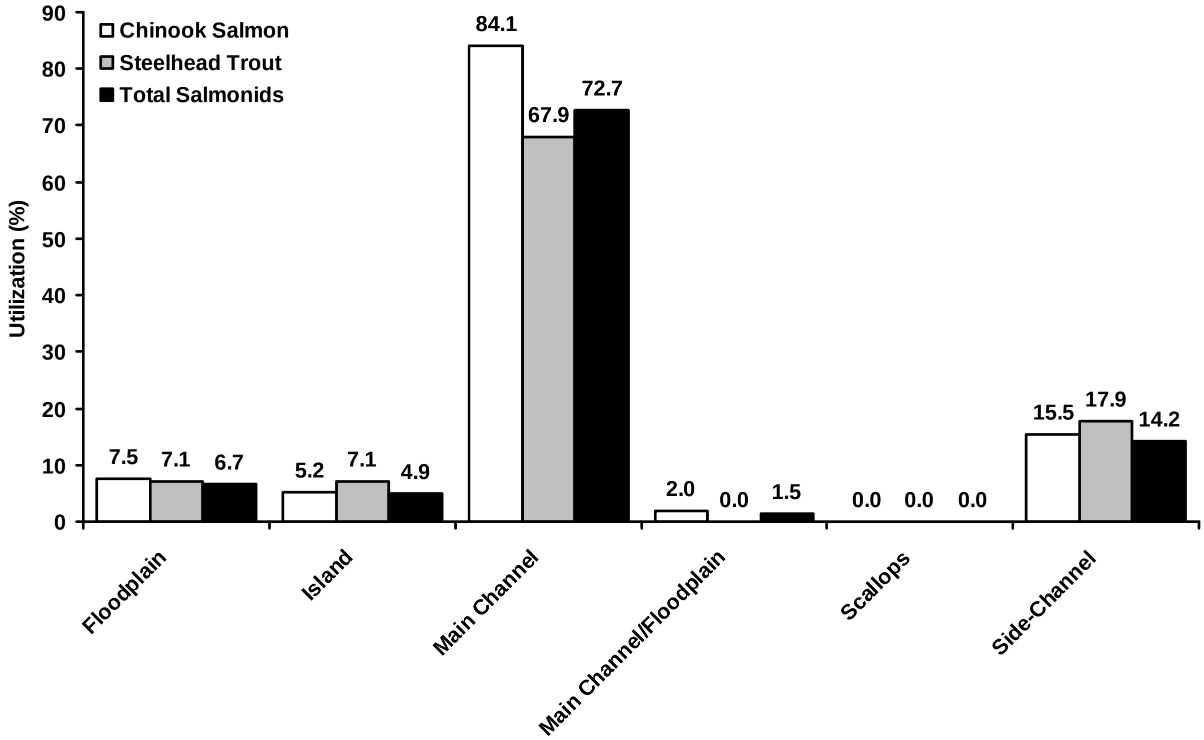


***Figure 50**. Utilization (%) rates for Chinook salmon, steelhead trout, and total salmonids associated with individual channel unit types or a combination of channel unit types including individual channel unit types used for GIS-based habitat mapping. The 2008–2010 augmentation sites were located primarily in Main Channel units.*

### *Juvenile Salmonid Habitat Preferences*

#### ***General***

In all, 9,222 Chinook salmon and 10,285 steelhead trout were observed during snorkel surveys conducted at Sailor Bar. The majority of Chinook salmon observations were fry and parr, which accounted for 44% and 45% of total observations, respectively. Small size classes made up the majority of observations at all sites. However, larger fish (smolts) made up a greater proportion of observations at the 2010 side-channel site (Figure 51). The majority of steelhead trout observations were fry (81%), which dominated at all sites (Figure 51). Chinook salmon were most abundant in and adjacent to the 2009 augmentation site, whereas steelhead trout were most abundant in and adjacent to the 2010 augmentation site. Chinook salmon were more abundant than steelhead trout at the 2008, 2009, and 2010 side-channel sites, whereas steelhead trout were more abundant than Chinook salmon at the 2010 site (Figure 52). The most common cover type associated with fish observations was Brush, which was present 47% of the time, Riparian Grass was present 14% of the time, and Large and Small Woody Material were relatively rare, with frequencies of 1.5% and 3%, respectively (Figure 53). Four channel unit types were recorded, including: (1) Floodplain; (2) Side-Channel; (3) Island; and (4) Main Channel. However, the Side-Channel and Island channel unit types were both exclusively associated with one site, with Island strongly correlated to Side-Channel. Therefore, Floodplain and Main Channel were the only categories evaluated. Floodplain habitat was present 30% of the time, whereas Main Channel habitat was present 3% of the time (Figure 53).

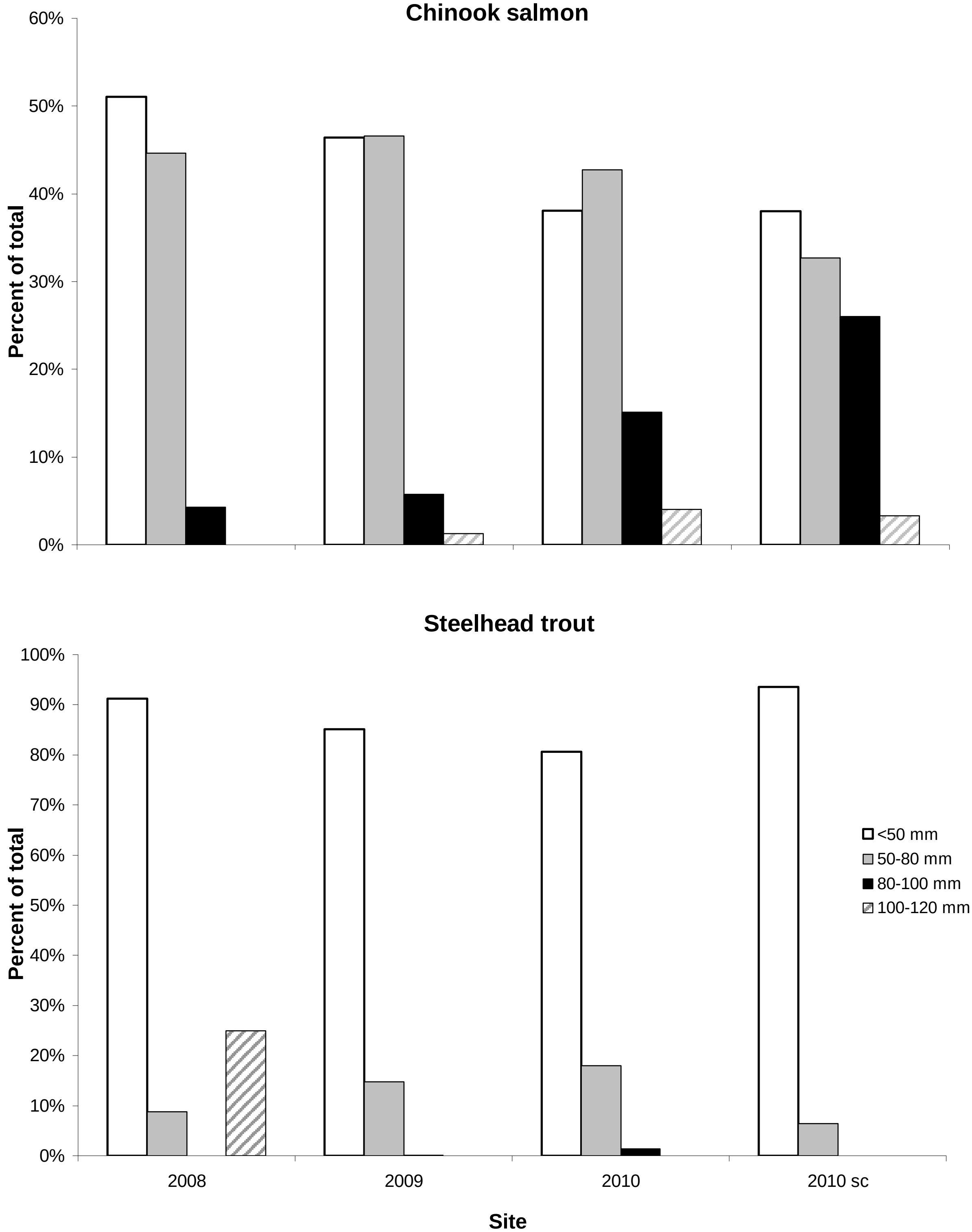


***Figure 51***. *Size class distributions of Chinook salmon (upper panel) and steelhead trout (lower panel) observed in and adjacent to the four restoration sites. Note the change in scale between panels.*

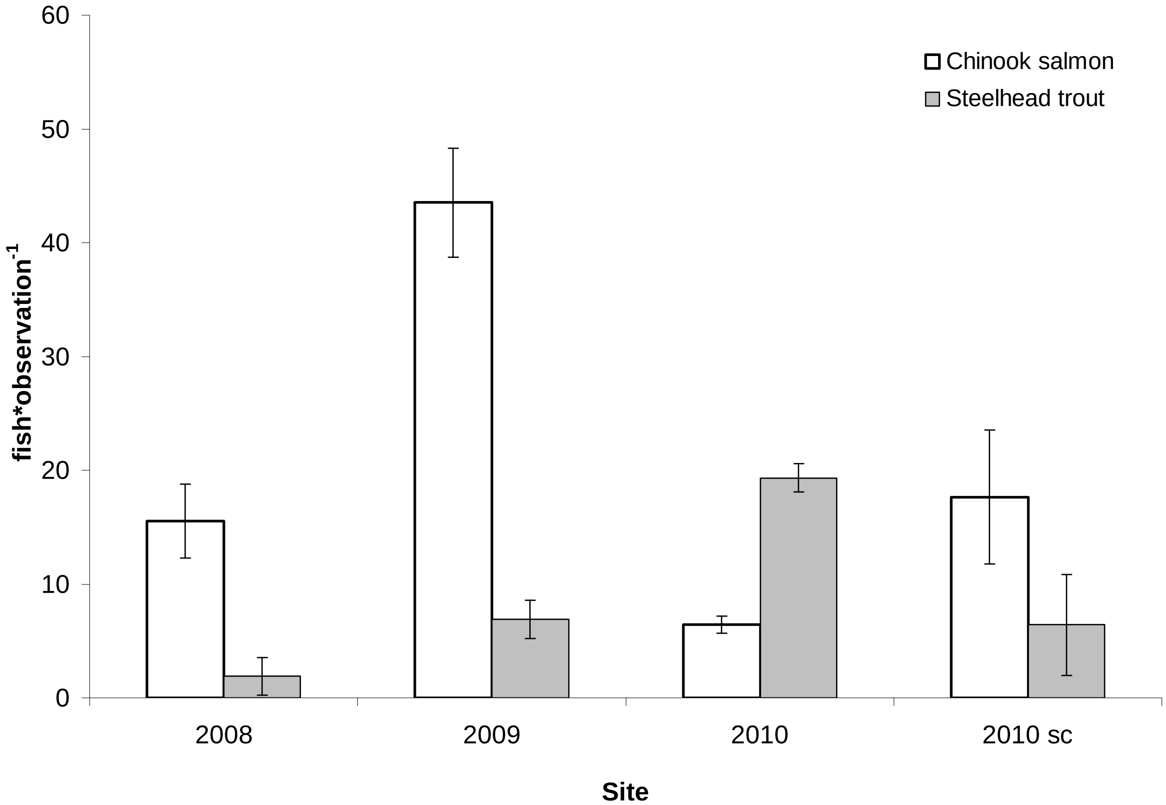


***Figure 52**. Mean number of fish•observation$^{-1}$ and standard errors for Chinook salmon and steelhead trout abundance along transects in and adjacent to the four restoration sites. 2010 sc = the 2010 side channel site.*

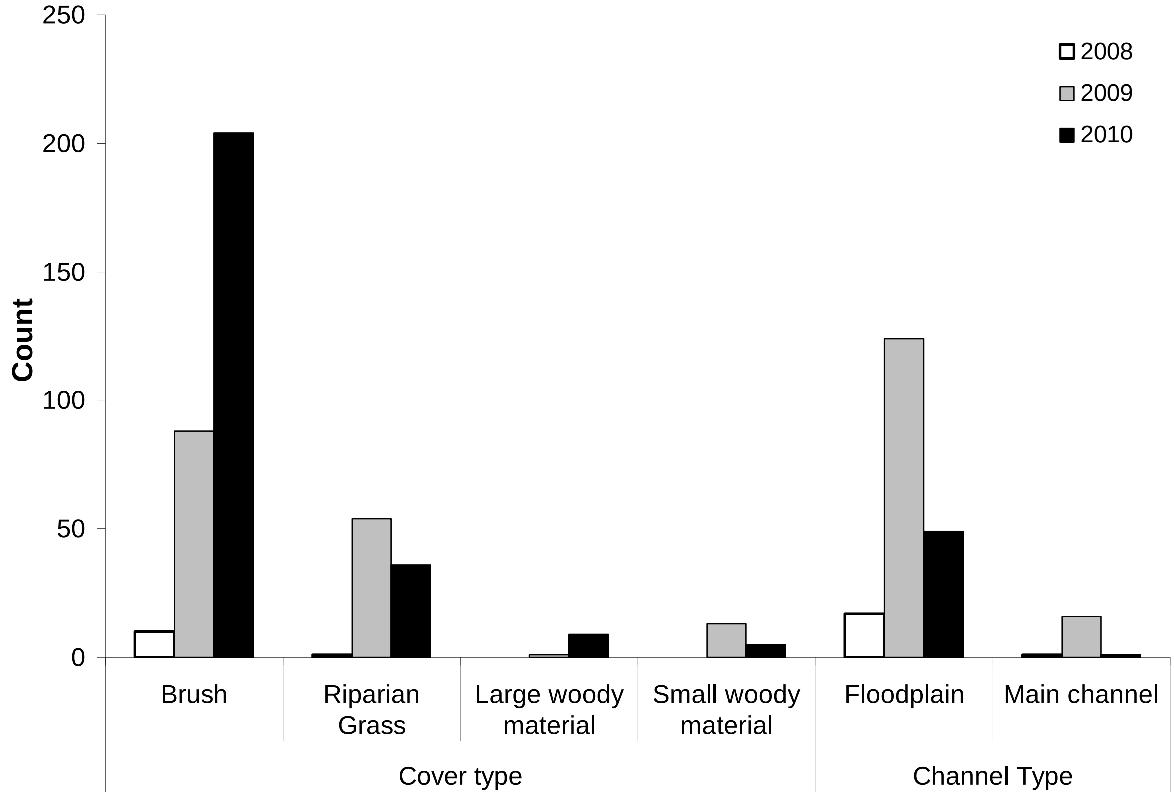


***Figure 53**. Distribution of cover types associated with fish observations within the three gravel augmentation (i.e., enhancement) sites. Cover and channel type counts were made for each observation record, regardless of number of fish•observation$^{-1}$ (e.g., if an observation record associated with brush consisted of 200 fish or 1 fish, the cover type category brush received a count of one).*

### *Species-Environment Relationships*

Canonical correspondence analysis produced two axes that explained 89% of the constrained variation in species-environment relationships (Figure 54). Axis 1 (eigenvalue = 0.36) explained 75% of the total variation and differentiated all size classes of steelhead trout from the two smallest size classes of Chinook salmon. Steelhead trout had positive scores on Axis 1 and were associated with greater velocity, whereas Chinook salmon had negative scores on Axis 1 and were associated with Floodplain channel units, Riparian Grass, larger substrates, and greater depth. Larger Chinook salmon size classes had Axis 1 scores near zero, which suggested they utilized a wider range of habitats relative to smaller Chinook salmon size classes and steelhead trout. Axis 2 (eigenvalue = 0.07) explained 14% of the total variation and differentiated larger Chinook salmon size classes associated with greater depths and velocities from smaller Chinook salmon size classes and steelhead trout associated with Riparian Grass and larger substrates (Figure 54).

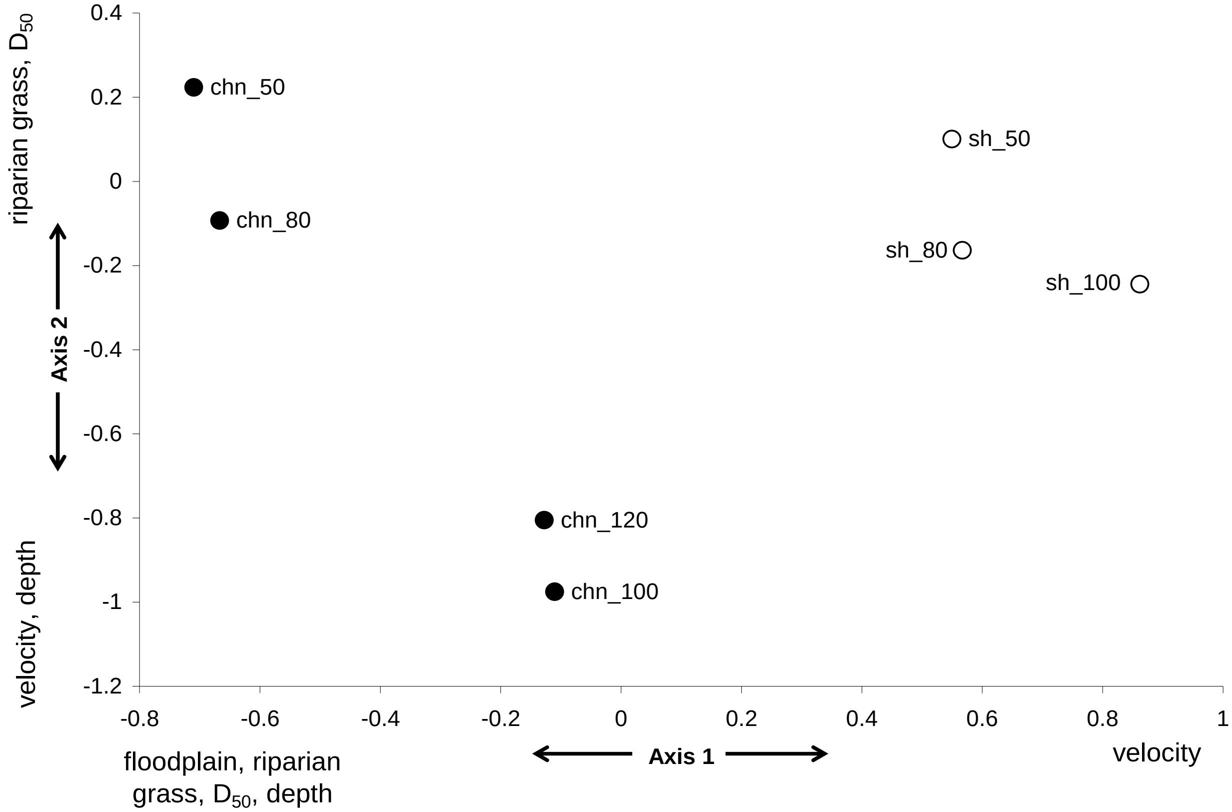


***Figure 54**. Bi-plot of species scores from canonical correspondence analysis. Axis 1 explained 75% of the total constrained variation in species-environment relationships and Axis 2 explained 14% of the total variation.*

Modeling total abundance with generalized linear mixed models yielded a significant positive relationship between total Chinook salmon abundance and depth, with significant negative relationships between total Chinook salmon abundance and Large Woody Material and Main Channel habitats (Table 4). Additionally, there was a significant site effect, with greater total Chinook salmon abundance at the 2008 and 2009 sites relative to the 2010 and 2010 side-channel sites. The steelhead trout model yielded significant negative relationships between total steelhead trout abundance and depth, velocity, Riparian Grass, and Floodplain habitats (Table 6). Additionally, there was a significant site effect. However, unlike Chinook salmon, total steelhead trout abundance was greater at the 2010 and 2010 side-channel sites.

***Table 4**. Parameter estimates for predictor variables included in the generalized linear mixed model. The response variable in the model is total Chinook salmon abundance.*

| Variable | Estimate | z | *p* |
|---|---|---|---|
| $D_{50}$ | -0.008 | -0.922 | 0.356 |
| Depth | 4.942 | 6.569 | < 0.001* |
| Velocity | -1.460 | -1.506 | 0.132 |
| Brush | 0.267 | 1.107 | 0.268 |
| Riparian grass | -0.271 | -0.640 | 0.522 |
| Large woody material | -2.670 | -2.009 | 0.045* |
| Small woody material | 0.668 | 1.166 | 0.244 |
| Main channel | -2.512 | -3.967 | <0.001* |
| Floodplain | -0.447 | -0.696 | 0.486 |

***Table 5**. Parameter estimates for predictor variables included in the generalized linear mixed model. The response variable in the model is total steelhead trout abundance.*

| Variable | Estimate | z | *p* |
|---|---|---|---|
| $D_{50}$ | 0.003 | 0.517 | 0.605 |
| Depth | -2.472 | -5.202 | < 0.001* |
| Velocity | -4.135 | -6.044 | < 0.001* |
| Brush | -0.009 | -0.059 | 0.953 |
| Riparian grass | -0.864 | -2.375 | 0.018* |
| Large woody material | -0.672 | -1.156 | 0.248 |
| Small woody material | 0.515 | 1.044 | 0.297 |
| Main channel | -1.693 | -1.701 | 0.089 |
| Floodplain | -1.024 | -2.091 | 0.036* |

## *Benthic Macroinvertebrate Community*

### *Environmental Conditions*

For pooled samples (i.e., all samples combined), depths at control sites ranged from 0.22–0.41 m (average = 0.31 m), whereas depths at treatment sites ranged from 0.08–0.32 m (average = 0.19 m). Velocities at control sites ranged from 0.00–0.98 m•s$^{-1}$ (average = 0.61 m•s$^{-1}$), whereas velocities at treatment sites ranged from 0.00–1.60 m•s$^{-1}$ (average = 0.29 m•s$^{-1}$). Based on ANOVA results, both average depths ($F = 41.329$; $df = 1, 76$; $p < 0.0001$) and velocities ($F = 14.373$; $df = 1, 76$; $p = 0.0003$) were significantly greater at control sites when compared to treatment sites (Figure 55). For upstream samples (2008 and 2009), depths at control sites ranged from 0.22–0.37 m (average = 0.28 m), whereas depths at treatment sites ranged from 0.10–0.30 m (average = 0.21 m). Velocities at control sites ranged from 0.49–0.98 m•s$^{-1}$ (average = 0.70 m•s$^{-1}$), whereas velocities at treatment sites ranged from 0.00–0.67 m•s$^{-1}$ (average = 0.23 m•s$^{-1}$). Based on ANOVA results, both average depths ($F = 9.474$; $df = 1, 36$; $p = 0.0040$) and velocities ($F = 39.987$; $df = 1, 36$; $p < 0.0001$) were significantly greater at control sites when compared to treatment sites (Figure 55). For downstream samples (2010 and

2010 side-channel), depths at control sites ranged from 0.27–0.41 m (average = 0.35 m), whereas depths at treatment sites ranged from 0.08–0.32 m (average = 0.18 m). Velocities at control sites ranged from 0.00–0.98 m•s$^{-1}$ (average = 0.52 m•s$^{-1}$), whereas velocities at treatment sites ranged from 0.01–1.60 m•s$^{-1}$ (average = 0.34 m•s$^{-1}$). Based on ANOVA results, average depths were significantly greater at control sites when compared to treatment sites ($F$ = 37.599; $df$ = 1, 38; $p$ <0.0001). Velocities were not significantly different ($F$ = 1.423; $df$ = 1, 38; $p$ = 0.2404; Figure 55). When depths and velocities were compared among control sites, downstream control sites were significantly deeper than upstream control sites ($F$ = 6.093; $df$ = 1, 14; $p$ = 0.0271). However, velocities were not significantly different ($F$ = 1.910; $df$ = 1, 14; $p$ = 0.1887; Figure 56). When depths and velocities were compared among treatment sites, upstream treatment sites were significantly deeper than downstream treatment sites ($F$ = 4.123; $df$ = 1, 60; $p$ = 0.0467). However, velocities were not significantly different ($F$ = 2.070; $df$ = 1, 60; $p$ = 0.1554; Figure 56).

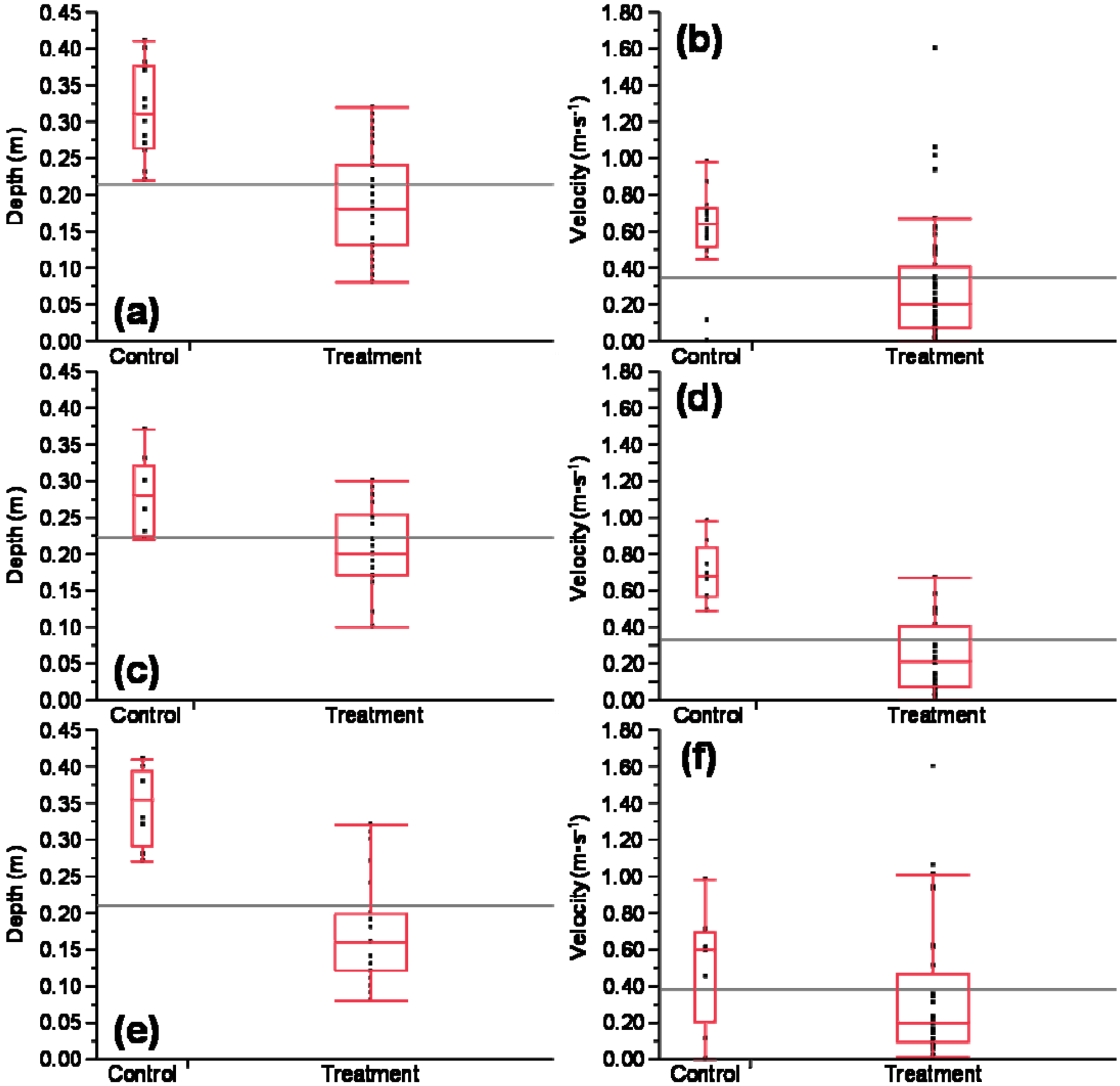


***Figure 55****. Depths and velocities at control and treatment sites based on pooled (a and b), upstream (c and d), and downstream (e and f) samples. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

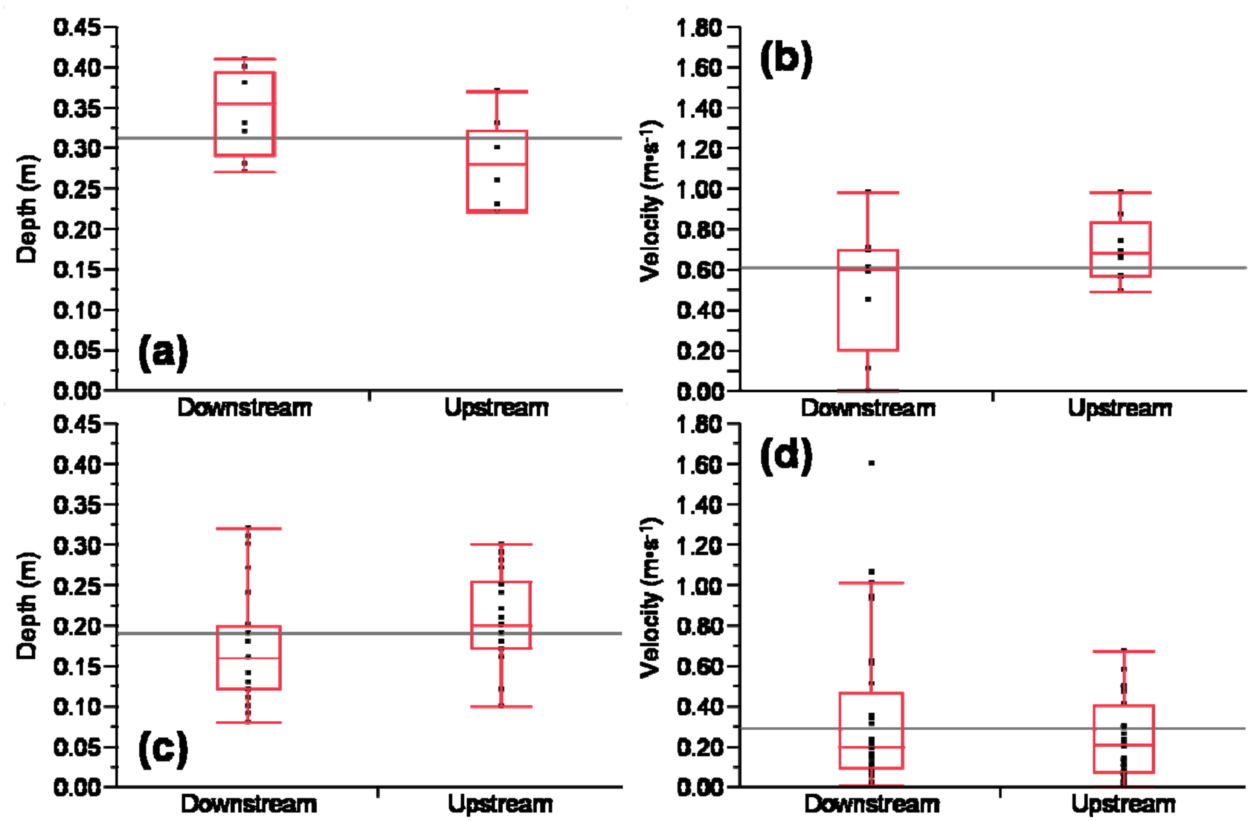


***Figure 56**. Depths and velocities at upstream and downstream control (a and b) and treatment (c and d) sites. Whiskers indicate 1% and 99% quantiles, wheras boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

### *Overall Benthic Macroinvertebrate Density and Biomass*

For pooled samples (i.e., all samples combined), overall density at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 141 individuals/m$^2$; range = 12–361 individuals/m$^2$), peaked at 8.9 weeks (62 days) inundation (average = 8,238 individuals/m$^2$; range = 2,267–20,070 individuals/m$^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 5,705 individuals/m$^2$; range = 2,279–14,767 individuals/m$^2$; Figure 57). Average treatment densities at 5.6, 8.9, and 10.0 weeks (39, 62, and 70 days) inundation (range = 4,885–8,238 individuals/m$^2$) were significantly greater (pairwise Tukey tests $_p$ <0.0001–0.0146) than average treatment densities at 1.3 and 3.3 weeks (9 and 23 days) inundation (range = 141–856 individuals/m$^2$). However, average treatment density at 8.9 weeks (62 days) inundation showed the only significant difference (pairwise Tukey test $_p$ = 0.0019) when compared to average control density (average = 3,441 individuals/m$^2$; range = 954–7,674 individuals/m$^2$). Overall biomass at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 0.36 g/m$^2$; range = 0.00–2.52 g/m$^2$) and peaked at 10.0 weeks (70 days) inundation (average = 1.89 g/m$^2$; range = 0.49–5.87 g/m$^2$; Figure 57). Average biomass was not significantly different among treatment groups. However, average treatment biomass at 1.3, 3.3, and 5.6 weeks (9, 23, and 39 days) inundation (range = 0.36–0.66 g/m$^2$) was significantly lower (pairwise Tukey tests $_p$ = 0.0005–0.0029) than average control biomass (average = 2.80 g/m$^2$; range = 0.25–14.43 g/m$^2$). Average treatment biomass did not surpass average control biomass at any time during the sampling period (~1–10 weeks). When overall density and biomass were compared among control sites, no significant differences in density ($F = 1.339$; $df = 1, 14$; $p = 0.2665$) or biomass ($F = 1.293$; $df = 1, 14$; $p = 0.2746$) were found between upstream and downstream sites (Figure 58). When overall density and biomass were compared among treatment sites, density was significantly greater at upstream sites when compared to downstream sites ($F = 4.750$; $df = 1, 76$; $p = 0.0324$). However, biomass was not significantly different ($F = 0.007$; $df = 1, 76$; $p = 0.9361$; Figure 58).

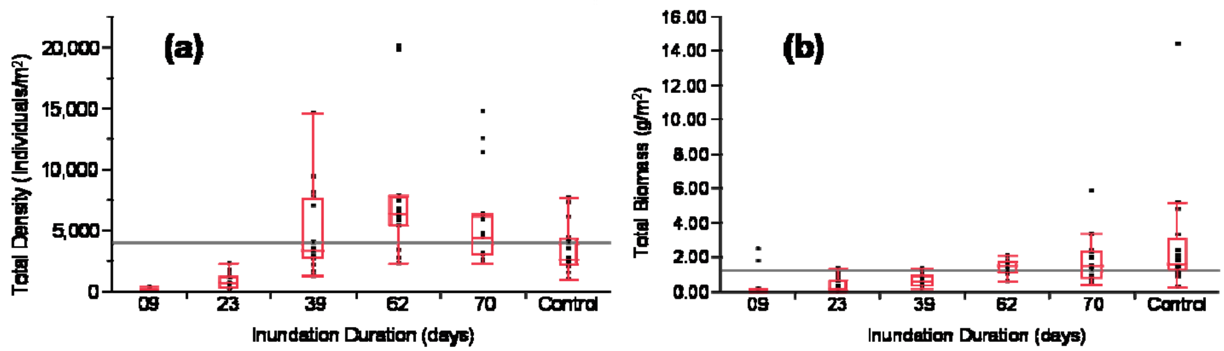


***Figure 57**. Overall macroinvertebrate density (a) and biomass (b) at control and treatment sites. Data for treatment sites are displayed based on inundation duration. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

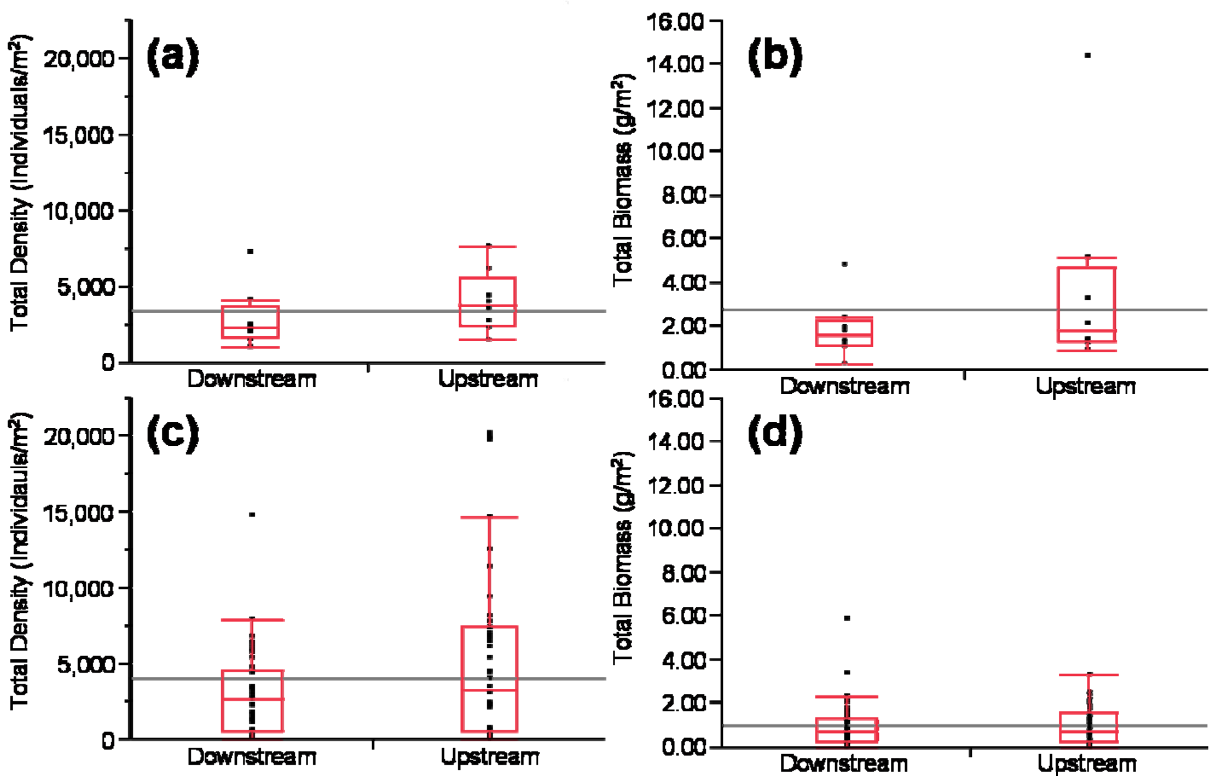


***Figure 58**. Overall macroinvertebrate density and biomass at upstream and downstream control (a and b) and treatment (c and d) sites. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

For upstream samples (2008 and 2009), overall density at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 161 individuals/m$^2$; range = 12–361 individuals/m$^2$), peaked at 8.9 weeks (62 days) inundation (average = 11,462 individuals/m$^2$; range = 5,314–20,070 individuals/m$^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 5,438 individuals/m$^2$; range = 2,279–12,523 individuals/m$^2$; Figure 59). Average treatment densities at 5.6 and 8.9 weeks (39 and 62 days) inundation (range = 7,065–11,462 individuals/m$^2$) were significantly greater (pairwise Tukey tests $_p$ <0.0001–0.0241) than average treatment densities at 1.3 and 3.3 weeks (9 and 23 days) inundation (range = 161–680 individuals/m$^2$). Similarly, average treatment density at 8.9 weeks (62 days) inundation was significantly greater (pairwise Tukey test $_p$ = 0.0374) than average treatment density at 10.0 weeks (70 days) inundation. Average treatment density at 8.9 weeks (62 days) inundation showed the only significant difference (pairwise Tukey test $_p$ = 0.0055) when compared to average control density (average = 4,023 individuals/m$^2$; range = 1,465–7,674 individuals/m$^2$). Overall biomass at treatment sites was lowest at 3.3 weeks (23 days) inundation (average = 0.39 g/m$^2$; range = 0.01–1.36 g/m$^2$) and peaked at 8.9 weeks (62 days) inundation (average = 1.62 g/m$^2$; range = 0.95–2.10 g/m$^2$). However, biomass showed a general increasing trend from 1.3 to 10.0 weeks (9 to 70 days) inundation, with average biomass at 1.3 (average = 0.45 g/m$^2$; range = 0.00–2.52 g/m$^2$) and 10.0 (average = 1.62 g/m$^2$; range = 0.49–3.28 g/m$^2$) weeks (9 and 70 days) inundation being comparable to biomass at 3.3 and 8.9 weeks (23 and 62 days) inundation, respectively (Figure 59). Average biomass was not significantly different among treatment groups. However, average treatment biomass at 1.3, 3.3, and 5.6 weeks (9, 23, and 39 days) inundation (range = 0.39–0.66 g/m$^2$) was significantly lower (pairwise Tukey tests $_p$ = 0.0201–0.0430) than average control biomass (average = 3.75 g/m$^2$; range = 0.89–14.43 g/m$^2$). Average treatment biomass did not surpass average control biomass at any time during the sampling period (~1–10 weeks).

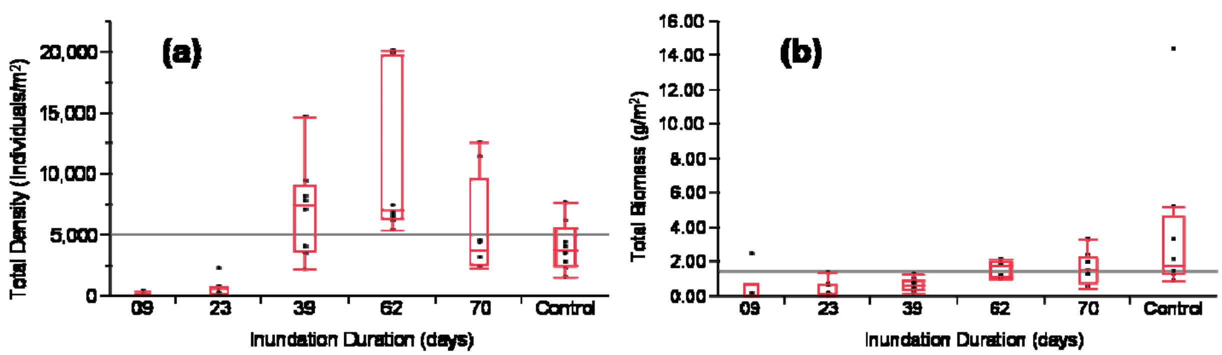


***Figure 59**. Overall macroinvertebrate density (a) and biomass (b) at upstream control and treatment sites. Data for treatment sites are displayed based on inundation duration. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

For downstream samples (2010 and 2010 side-channel), overall density at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 127 individuals/m$^2$; range = 23–267 individuals/m$^2$) and peaked at 10.0 weeks (70 days) inundation (average = 5,972 individuals/m$^2$; range = 2,674–14,767 individuals/m$^2$; Figure 60). Average treatment densities at 8.9 and 10.0 weeks (62 and 70 days) inundation (range = 5,013–5,972 individuals/m$^2$) were significantly greater (pairwise Tukey tests $_p$ <0.0001–0.0030) than average treatment densities at 1.3 and 3.3

weeks (9 and 23 days) inundation (range = 127–1,032 individuals/$m^2$). Similarly, average treatment density at 10.0 weeks (70 days) inundation was significantly greater (pairwise Tukey test $_p$ = 0.0225) than average treatment density at 5.6 weeks (39 days) inundation (average = 2,705 individuals/$m^2$). Average treatment density at 10.0 weeks (70 days) inundation showed the only significant difference (pairwise Tukey test $_p$ = 0.0336) when compared to average control density (average = 2,859 individuals/$m^2$; range = 954–7,198 individuals/$m^2$). Overall biomass at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 0.30 g/$m^2$; range = 0.01–1.78 g/$m^2$) and peaked at 10.0 weeks (70 days) inundation (average = 2.17 g/$m^2$; range = 0.69–5.87 g/$m^2$; Figure 64). Average treatment biomass at 10.0 weeks (70 days) inundation was significantly greater (pairwise Tukey tests $_p$ = 0.0052–0.0391) than average treatment biomass at 1.3, 3.3, and 5.6 weeks (9, 23, and 39 days) inundation (range = 0.30–0.66 g/$m^2$). Similarly, average treatment biomass at 1.3 and 3.3 weeks (9 and 23 days) inundation (range = 0.30–0.37 g/$m^2$) was significantly lower (pairwise Tukey tests $_p$ = 0.0312–0.0453) than average control biomass (average = 1.84 g/$m^2$; range = 0.25–4.76 g/$m^2$). Average treatment biomass surpassed average control biomass at 10.0 weeks (70 days) inundation. However, no significant difference was found.

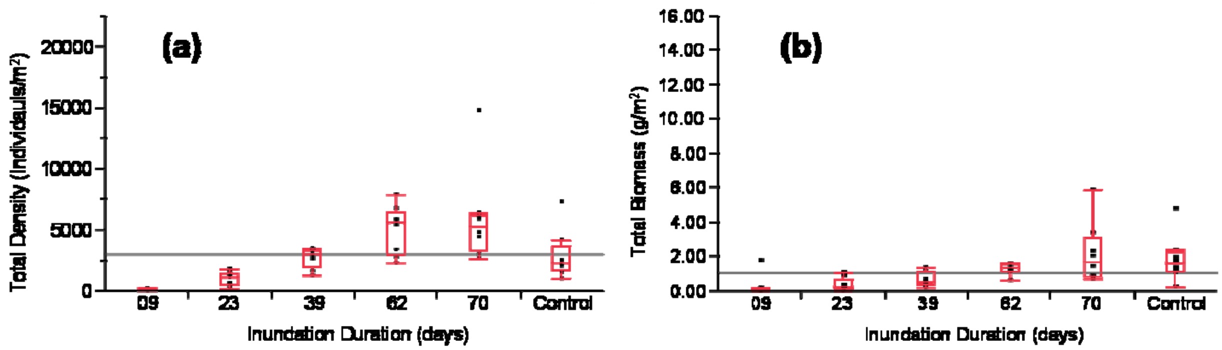


***Figure 60**. Overall macroinvertebrate density (a) and biomass (b) at downstream control and treatment sites. Data for treatment sites are displayed based on inundation duration. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

When peak treatment density and biomass were compared between upstream (8.9 weeks – 62 days inundation) and downstream (10.0 weeks – 70 days inundation) sites, average peak density for upstream sites (average = 11,462 individuals/$m^2$) was 1.92-fold greater than that of downstream sites (average = 5,972 individuals/$m^2$), whereas average peak biomass for downstream sites (average = 2.17 g/$m^2$) was 1.34-fold greater than that of upstream sites (average = 1.62 g/$m^2$). However, these differences were not significant ($_F$ = 0.707–3.807; $_{df}$ = 1, 14; $_p$ = 0.0714–0.4145; Figure 61).

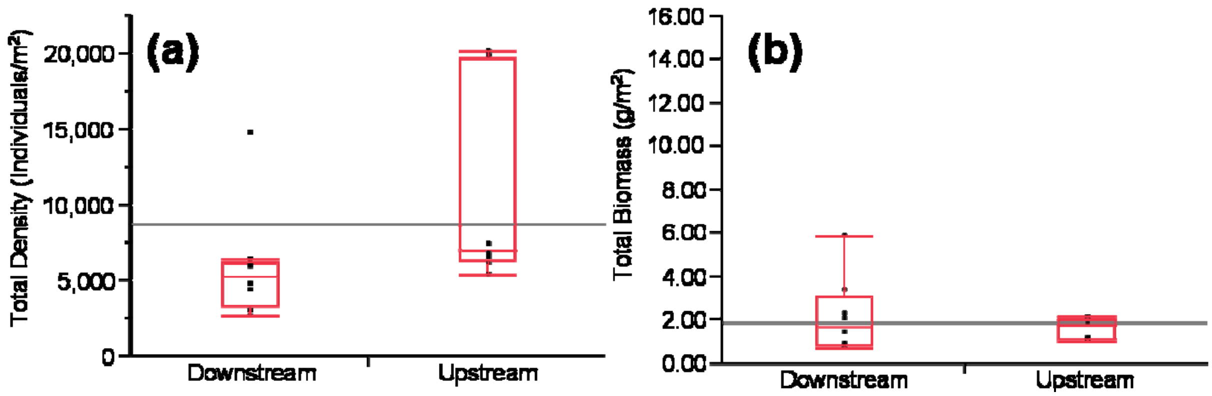


***Figure 61****. Peak overall macroinvertebrate density (a) and biomass (b) at upstream and downstream treatment sites. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

### *Baetidae Density and Biomass*

For pooled samples (i.e., all samples combined), Baetidae density at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 5 individuals/m$^2$; range = 0–12 individuals/m$^2$), peaked at 8.9 weeks (62 days) inundation (average = 435 individuals/m$^2$; range = 12–1,547 individuals/m$^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 274 individuals/m$^2$; range = 0–1,035 individuals/m$^2$; Figure 62). Average treatment density at 8.9 weeks (62 days) inundation was significantly greater (pairwise Tukey tests $p$ = 0.0039–0.0455) than average treatment densities at 1.3, 3.3, and 5.6 weeks (9, 23, and 39 days) inundation (range = 5–110 individuals/m$^2$). Similarly, average treatment density at 1.3 weeks (9 days) inundation was significantly lower (pairwise Tukey test $p$ = 0.0240) than average control density (average = 369 individuals/m$^2$; range = 0–1,174 individuals/m$^2$). Average treatment density surpassed average control density at 8.9 weeks (62 days) inundation. However, no significant difference was found. Baetidae biomass at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 0.00 g/m$^2$; range = 0.00–0.00 g/m$^2$), peaked at 8.9 weeks (62 days) inundation (average = 0.08 g/m$^2$; range = 0.00–0.27 g/m$^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 0.03 g/m$^2$; range = 0.00–0.26 g/m$^2$; Figure 62). Average treatment biomass at 8.9 weeks (62 days) inundation was significantly greater (pairwise Tukey tests $p$ = 0.0078–0.0330) than average treatment biomass at 1.3 and 3.3 weeks (9 and 23 days) inundation (range = 0.00–0.01 g/m$^2$). Similarly, average treatment biomass at 1.3 and 3.3 weeks (9 and 23 days) inundation was significantly lower (pairwise Tukey tests $p$ = 0.0079–0.0337) than average control biomass (average = 0.08 g/m$^2$; range = 0.00–0.26 g/m$^2$). Average treatment biomass surpassed average control biomass at 8.9 weeks (62 days) inundation. However, no significant difference was found. When Baetidae density and biomass were compared among control sites, average Baetidae biomass was significantly greater at downstream sites when compared to upstream sites ($F$ = 6.967; $df$ = 1, 14; $p$ = 0.0194). However, Baetidae density was not significantly different ($F$ = 4.281; $df$ = 1, 14; $p$ = 0.0575; Figure 63). When Baetidae density and biomass were compared among treatment sites, both average Baetidae density ($F$ = 11.909; $df$ = 1, 76; $p$ = 0.0009) and biomass ($F$ = 10.687; $df$ = 1, 76; $p$ = 0.0016) were significantly greater at downstream sites when compared to upstream sites (Figure 63).

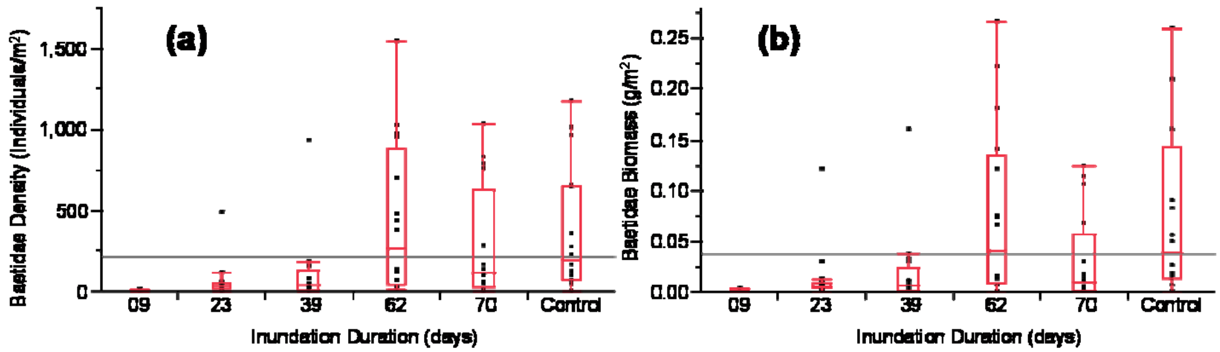


***Figure 62****. Baetidae density (a) and biomass (b) at control and treatment sites. Data for treatment sites are displayed based on inundation duration. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

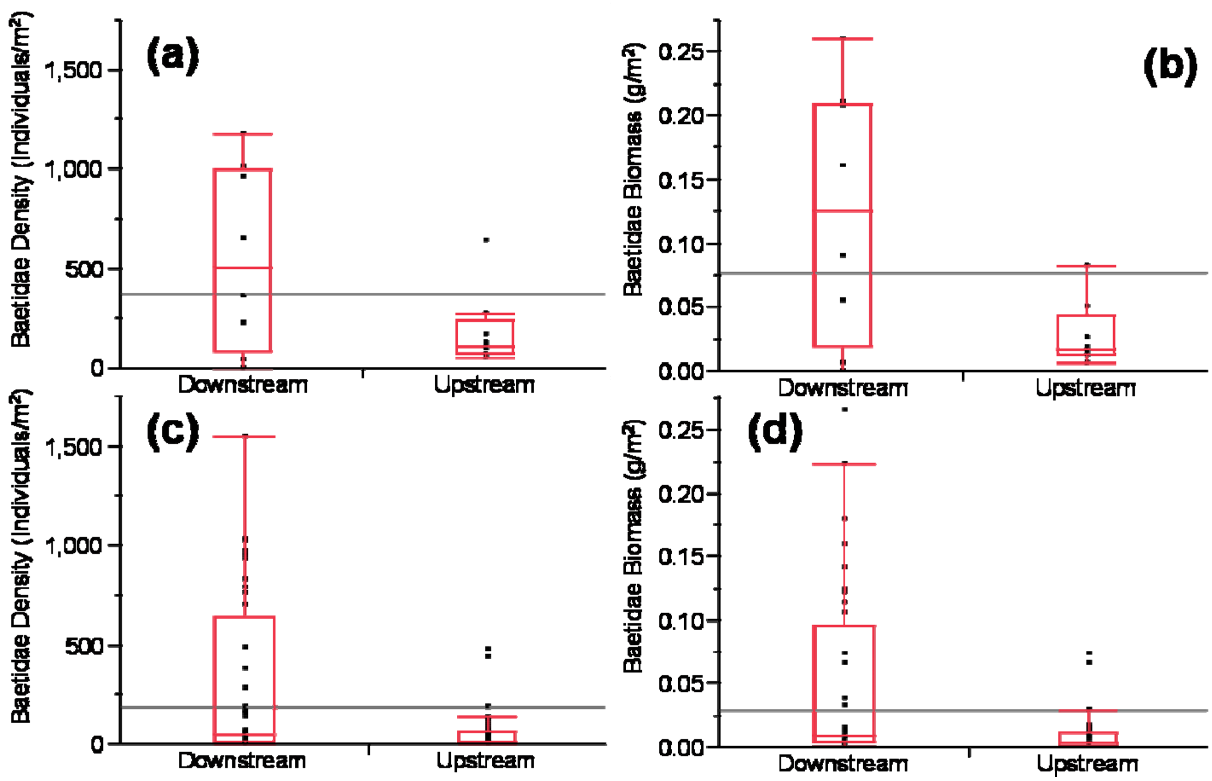


***Figure 63****. Baetidae density and biomass at upstream and downstream control (a and b) and treatment (c and d) sites. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

For upstream samples (2008 and 2009), Baetidae density at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 0 individuals/m$^2$; range = 0–0 individuals/m$^2$), peaked at 8.9 weeks (62 days) inundation (average = 163 individuals/m$^2$; range = 12–477 individuals/m$^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 49 individuals/m$^2$; range = 0–140 individuals/m$^2$; Figure 64). Average density was not significantly different among treatment groups. Similarly, average treatment group densities were not significantly different when compared to average control density (average = 185 individuals/m$^2$; range = 47–640 individuals/m$^2$). Average treatment density did not surpass average control density at any time during the sampling period (~1–10 weeks). Baetidae biomass at treatment sites was lowest at 3.3 weeks (23 days) inundation (average = 0.00 $g/m^2$; range = 0.00–0.00 $g/m^2$), peaked at 8.9 weeks (62 days) inundation (average = 0.02 $g/m^2$; range = 0.00–0.08 $g/m^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 0.01 $g/m^2$; range = 0.00–0.03 $g/m^2$; Figure 64). Average biomass was not significantly different among treatment groups. Similarly, average treatment group biomass was not significantly different when compared to average control biomass (average = 0.03 $g/m^2$; range = 0.01–0.08 $g/m^2$). Average treatment biomass did not surpass average control biomass at any time during the sampling period (~1–10 weeks).

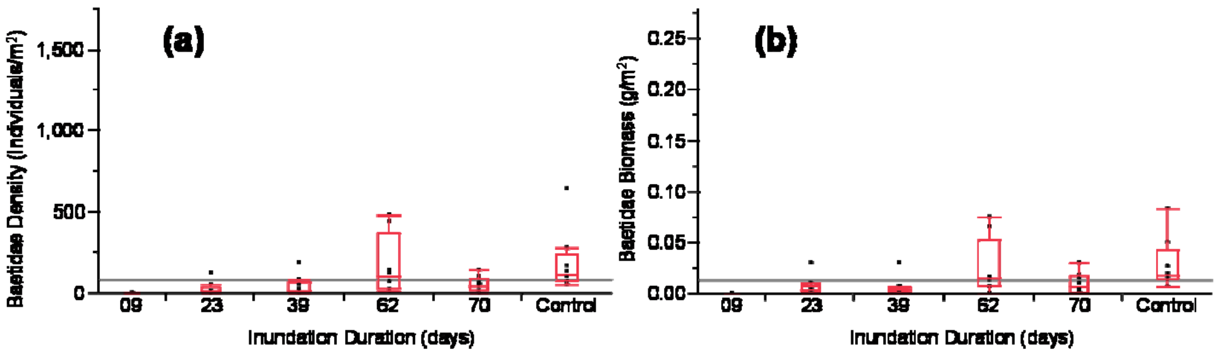


***Figure 64**. Baetidae density (a) and biomass (b) at upstream control and treatment sites. Data for treatment sites are displayed based on inundation duration. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

For downstream samples (2010 and 2010 side-channel), Baetidae density at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 9 individuals/$m^2$; range = 0–12 individuals/$m^2$), peaked at 8.9 weeks (62 days) inundation (average = 708 individuals/$m^2$; range = 12–1,547 individuals/$m^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 499 individuals/$m^2$; range = 0–1,035 individuals/$m^2$; Figure 65). Average treatment density at 8.9 weeks (62 days) inundation was significantly greater (pairwise Tukey tests $_p$ = 0.0042–0.0160) than average treatment densities at 1.3 and 3.3 weeks (9 and 23 days) inundation (range = 9–95 individuals/$m^2$). Average treatment density surpassed average control density (average = 552 individuals/$m^2$; range = 0–1,174 individuals/$m^2$) at 8.9 weeks (62 days) inundation. However, no significant difference was found. Baetidae biomass at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 0.00 $g/m^2$; range = 0.00–0.00 $g/m^2$), peaked at 8.9 weeks (62 days) inundation (average = 0.13 $g/m^2$; range = 0.00–0.27 $g/m^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 0.05 $g/m^2$; range = 0.00–0.13 $g/m^2$; Figure 65). Average treatment biomass at 8.9 weeks (62 days) inundation was significantly greater (pairwise Tukey tests $_p$ = 0.0062–0.0302) than average treatment biomass at 1.3 and 3.3 weeks (9 and 23 days) inundation (range = 0.00–0.02 $g/m^2$). Similarly, average treatment biomass at 1.3 and 3.3 weeks (9 and 23 days) inundation was significantly lower (pairwise Tukey tests $_p$ = 0.0085–0.0397) than average control biomass (average = 0.12 $g/m^2$; range = 0.00–0.26 $g/m^2$). Average treatment biomass surpassed average control biomass at 8.9 weeks (62 days) inundation. However, no significant difference was found.

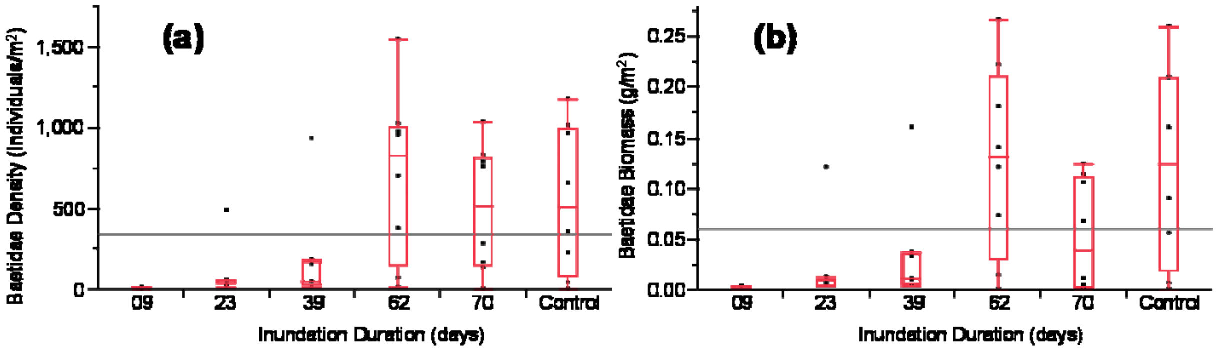


***Figure 65**. Baetidae density (a) and biomass (b) at downstream control and treatment sites. Data for treatment sites are displayed based on inundation duration. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

When peak treatment density and biomass (8.9 weeks – 62 days inundation) were compared between upstream and downstream sites, average peak density for downstream sites (average = 708 individuals/m$^2$) was 4.34-fold greater than that of upstream sites (average = 163 individuals/m$^2$), whereas average peak biomass for downstream sites (average = 0.13 g/m$^2$) was 6.50-fold greater than that of upstream sites (average = 0.02 g/m$^2$). These differences were significant for both density ($F = 7.624$; $df = 1, 14$; $p = 0.0153$) and biomass ($F = 8.629$; $df = 1, 14$; $p = 0.0108$; Figure 66).

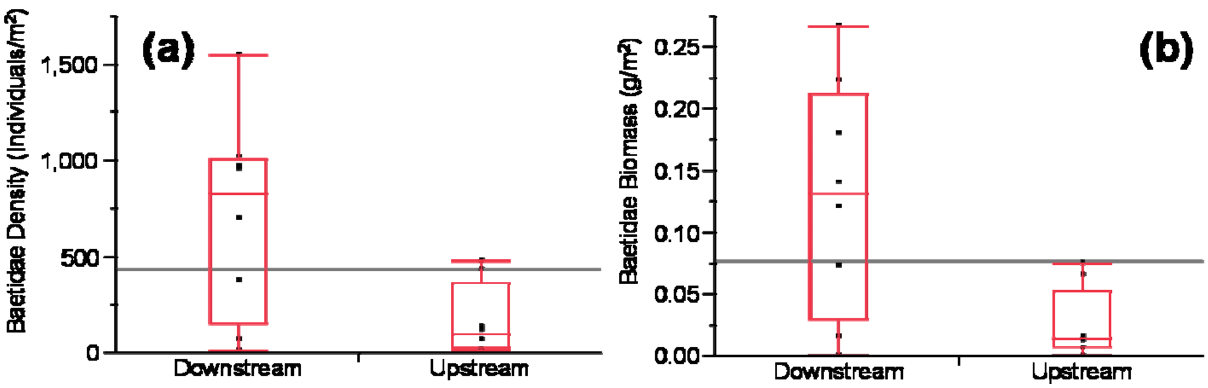


***Figure 66**. Peak Baetidae density (a) and biomass (b) at upstream and downstream treatment sites. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

### *Chironomidae Density and Biomass*

For pooled samples (i.e., all samples combined), Chironomidae density at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 57 individuals/m$^2$; range = 0–174 individuals/m$^2$), peaked at 8.9 weeks (62 days) inundation (average = 7,176 individuals/m$^2$; range = 1,244–19,663 individuals/m$^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 4,281 individuals/m$^2$; range = 1,686–12,128 individuals/m$^2$; Figure 67). Average treatment densities at 5.6, 8.9, and 10.0 weeks (39, 62, and 70 days) inundation (range = 4,281–7,176 individuals/m$^2$) were significantly greater (pairwise Tukey tests $p$ <0.0001–0.0188) than average treatment densities at 1.3 and 3.3 weeks (9 and 23 days) inundation (range = 57–522 individuals/m$^2$). Average treatment densities surpassed average control density (average = 1,528 individuals/m$^2$; range = 384–4,151 individuals/m$^2$) at 5.6, 8.9, and 10.0 weeks (39, 62, and 70 days) inundation. However, average treatment density at 8.9 weeks (62 days) inundation showed the only significant difference (pairwise Tukey test $p$ <0.0001) when compared to average control density. Chironomidae biomass at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 0.01 g/m$^2$; range = 0.00–0.01 g/m$^2$), peaked at 8.9 weeks (62 days) inundation (average = 0.64 g/m$^2$; range = 0.11–1.68 g/m$^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 0.40 g/m$^2$; range = 0.14–1.04 g/m$^2$; Figure 67). Average treatment biomass at 5.6, 8.9, and 10.0 weeks (39, 62, and 70 days) inundation (range = 0.39–0.64 g/m$^2$) was significantly greater (pairwise Tukey tests $p$ <0.0001–0.0072) than average treatment biomass at 1.3 and 3.3 weeks (9 and 23 days) inundation (range = 0.01–0.05 g/m$^2$). Average treatment biomass surpassed average control biomass (average = 0.14 g/m$^2$; range = 0.04–0.38 g/m$^2$) at 5.6, 8.9, and 10.0 weeks (39, 62, and 70 days) inundation. However, average treatment biomass at 8.9 weeks (62 days) inundation showed the only significant difference (pairwise Tukey test $p$ <0.0001) when compared to average control biomass. When Chironomidae density and biomass were compared among control sites, no significant differences in density ($F$ = 0.205; $df$ = 1, 14; $p$ = 0.6573) or biomass ($F$ = 0.218; $df$ = 1, 14; $p$ = 0.6476) were found between upstream and downstream sites (Figure 72). When Chironomidae density and biomass were compared among treatment sites, both average Chironomidae density ($F$ = 8.429; $df$ = 1, 76; $p$ = 0.0048) and biomass ($F$ = 8.624; $df$ = 1, 76; $p$ = 0.0044) were significantly greater at upstream sites when compared to downstream sites (Figure 68).

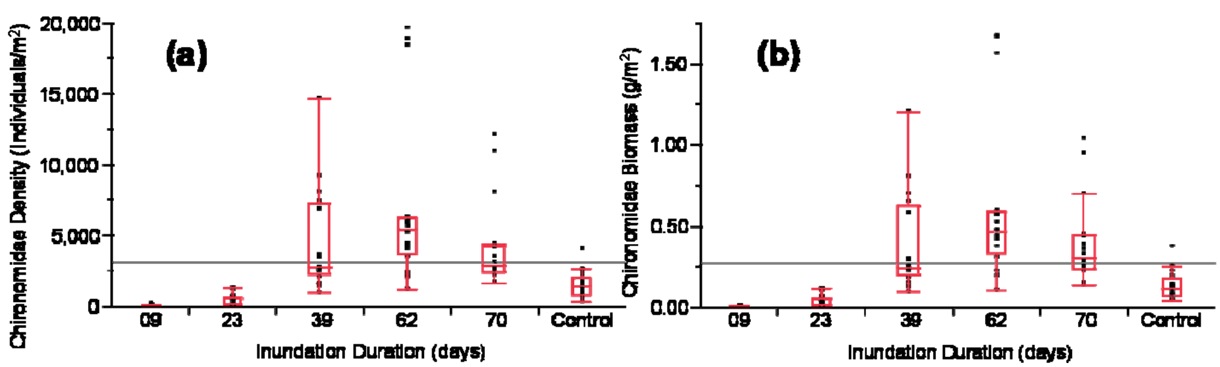


***Figure 67**. Chironomidae density (a) and biomass (b) at control and treatment sites. Data for treatment sites are displayed based on inundation duration. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

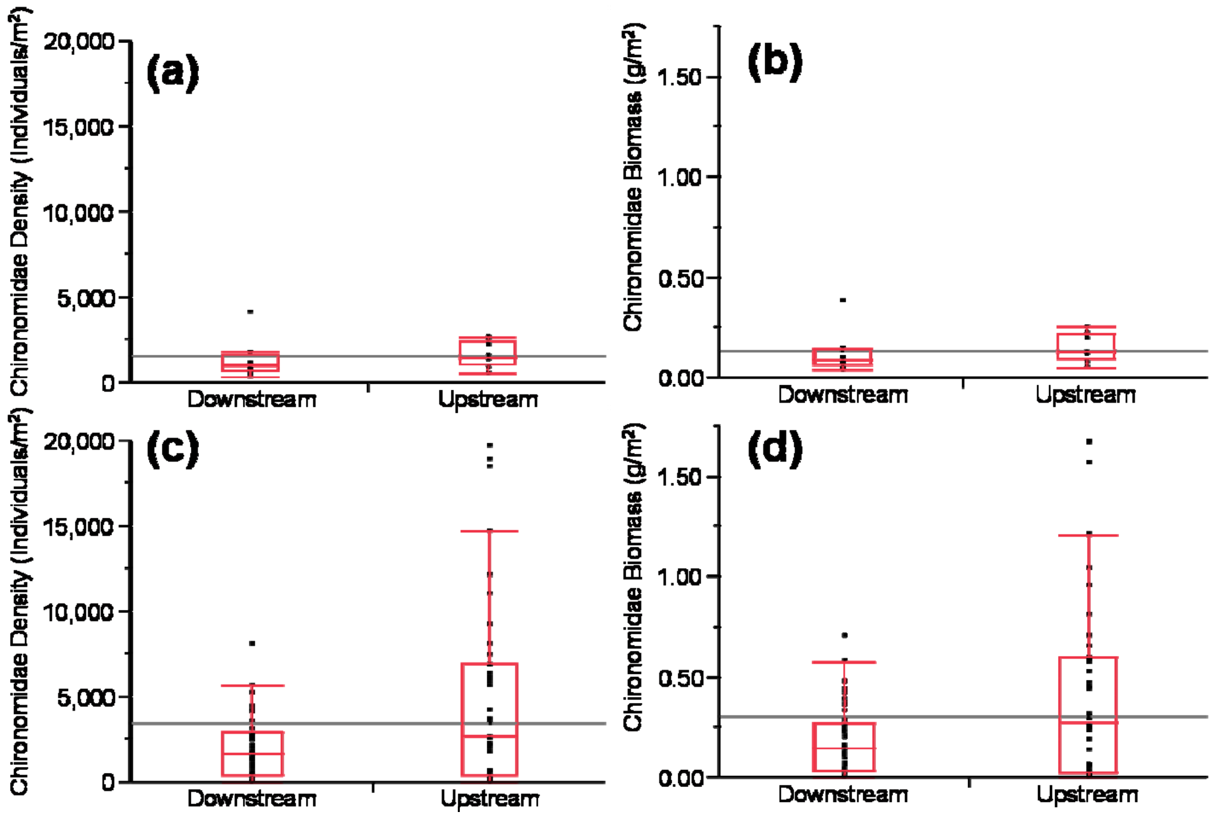


***Figure 68****. Chironomidae density and biomass at upstream and downstream control (a and b) and treatment (c and d) sites. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

For upstream samples (2008 and 2009), Chironomidae density at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 74 individuals/$m^2$; range = 0–174 individuals/$m^2$), peaked at 8.9 weeks (62 days) inundation (average = 10,576 individuals/$m^2$; range = 3,593–19,663 individuals/$m^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 4,907 individuals/$m^2$; range = 1,756–12,128 individuals/$m^2$; Figure 69). Average treatment densities at 5.6 and 8.9 weeks (39 and 62 days) inundation (range = 6,917–10,576 individuals/$m^2$) were significantly greater (pairwise Tukey tests $p$ <0.0001–0.0227) than average treatment densities at 1.3 and 3.3 weeks (9 and 23 days) inundation (range = 74–375 individuals/$m^2$). Average treatment densities surpassed average control density (average = 1,642 individuals/$m^2$; range = 558–2,674 individuals/$m^2$) at 5.6, 8.9, and 10.0 weeks (39, 62, and 70 days) inundation. However, average treatment density at 8.9 weeks (62 days) inundation showed the only significant difference (pairwise Tukey test $p = 0.0005$) when compared to average control density. Chironomidae biomass at treatment sites was lowest at 3.3 weeks (23 days) inundation (average = 0.01 g/$m^2$; range = 0.00–0.01 g/$m^2$), peaked at 8.9 weeks (62 days) inundation (average = 0.92 g/$m^2$; range = 0.31–1.68 g/$m^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 0.47 g/$m^2$; range = 0.14–1.04 g/$m^2$; Figure 69). Average treatment biomass at 5.6 and 8.9 weeks (39 and 62 days) inundation (range = 0.59–0.92 g/$m^2$) was significantly greater (pairwise Tukey tests $p$ <0.0001–0.0194) than average treatment biomass at 1.3 and 3.3 weeks (9 and 23 days) inundation (range = 0.01–0.03 g/$m^2$). Average treatment biomass surpassed average control biomass (average = 0.15 g/$m^2$; range =

0.05–0.25 g/m$^2$) at 5.6, 8.9, and 10.0 weeks (39, 62, and 70 days) inundation. However, average treatment biomass at 8.9 weeks (62 days) inundation showed the only significant difference (pairwise Tukey test $_p$ <0.0003) when compared to average control biomass.

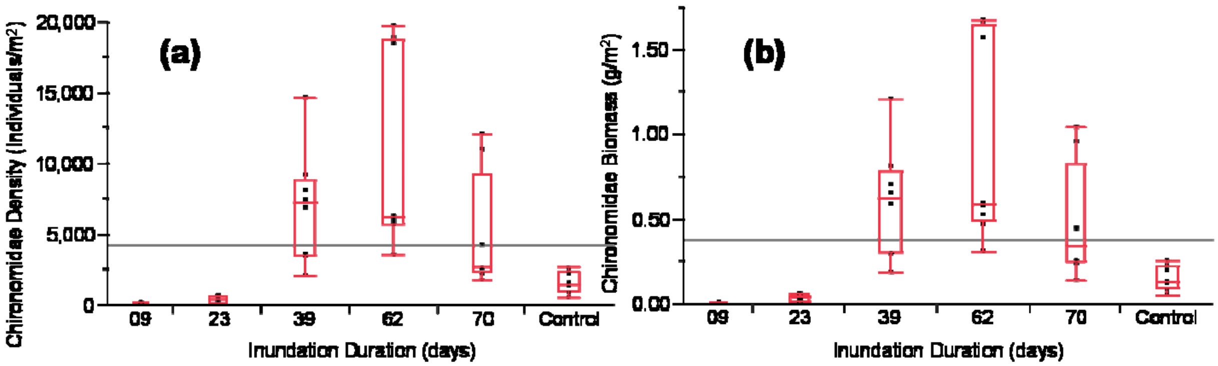


***Figure 69***. *Chironomidae density (a) and biomass (b) at upstream control and treatment sites. Data for treatment sites are displayed based on inundation duration. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

For downstream samples (2010 and 2010 side-channel), Chironomidae density at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 44 individuals/m$^2$; range = 0–105 individuals/m$^2$), peaked at 8.9 weeks (62 days) inundation (average = 3,776 individuals/m$^2$; range = 1,244–5,721 individuals/m$^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 3,654 individuals/m$^2$; range = 1,686–8,058 individuals/m$^2$; Figure 70). Average treatment densities at 8.9 and 10.0 weeks (62 and 70 days) inundation were significantly greater (pairwise Tukey tests $_p$ <0.0001–0.0002) than average treatment densities at 1.3 and 3.3 weeks (9 and 23 days) inundation (range = 44–669 individuals/m$^2$). Similarly, average treatment density at 5.6 weeks (39 days) inundation (average = 2,176 individuals/m$^2$; range = 1,012–2,802 individuals/m$^2$) was significantly greater (pairwise Tukey test $_p$ = 0.0148) than average treatment density at 1.3 weeks (9 days) inundation. Average treatment densities surpassed average control density (average = 1,414 individuals/m$^2$; range = 384–4,151 individuals/m$^2$) at 5.6, 8.9, and 10.0 weeks (39, 62, and 70 days) inundation. However, average treatment densities at 8.9 and 10.0 weeks (62 and 70 days) inundation showed the only significant differences (pairwise Tukey tests $_p$ = 0.0052–0.0092) when compared to average control density. Chironomidae biomass at treatment sites was lowest at 3.3 weeks (23 days) inundation (average = 0.00 g/m$^2$; range = 0.00–0.01 g/m$^2$), peaked at 8.9 weeks (62 days) inundation (average = 0.35 g/m$^2$; range = 0.11–0.57 g/m$^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 0.32 g/m$^2$; range = 0.15–0.70 g/m$^2$; Figure 70). Average treatment biomass at 8.9 and 10.0 weeks (62 and 70 days) inundation was significantly greater (pairwise Tukey tests $_p$ <0.0001–0.0003) than average treatment biomass at 1.3 and 3.3 weeks (9 and 23 days) inundation (range = 0.00–0.06 g/m$^2$). Similarly, average treatment biomass at 5.6 weeks (39 days) inundation (average = 0.20 g/m$^2$; range = 0.10–0.26 g/m$^2$) was significantly greater (pairwise Tukey test $_p$ = 0.0147) than average treatment biomass at 1.3 weeks (9 days) inundation. Average treatment biomass surpassed average control biomass (average = 0.12 g/m$^2$; range = 0.04–0.38 g/m$^2$) at 5.6, 8.9, and 10.0 weeks (39, 62, and 70 days) inundation. However, average treatment biomass at 8.9 and 10.0 weeks (62 and 70 days) inundation showed the only significant differences (pairwise Tukey tests $p = 0.0023–0.0096$) when compared to average control biomass.

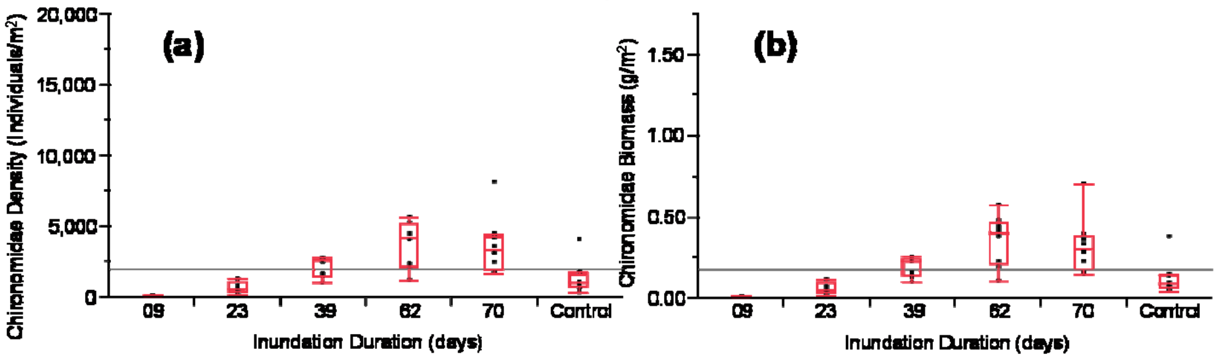


***Figure 70****. Chironomidae density (a) and biomass (b) at downstream control and treatment sites. Data for treatment sites are displayed based on inundation duration. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

When peak treatment density and biomass (8.9 weeks – 62 days inundation) were compared between upstream and downstream sites, average peak density for upstream sites (average = 10,576 individuals/m$^2$) was 2.80-fold greater than that of downstream sites (average = 3,776 individuals/m$^2$), whereas average peak biomass for upstream sites (average = 0.92 g/m$^2$) was 2.63-fold greater than that of downstream sites (average = 0.35 g/m$^2$). These differences were significant for both density ($F = 7.061$; $df = 1, 14$; $p = 0.0188$) and biomass ($F = 6.830$; $df = 1, 14$; $p = 0.0204$; Figure 71).

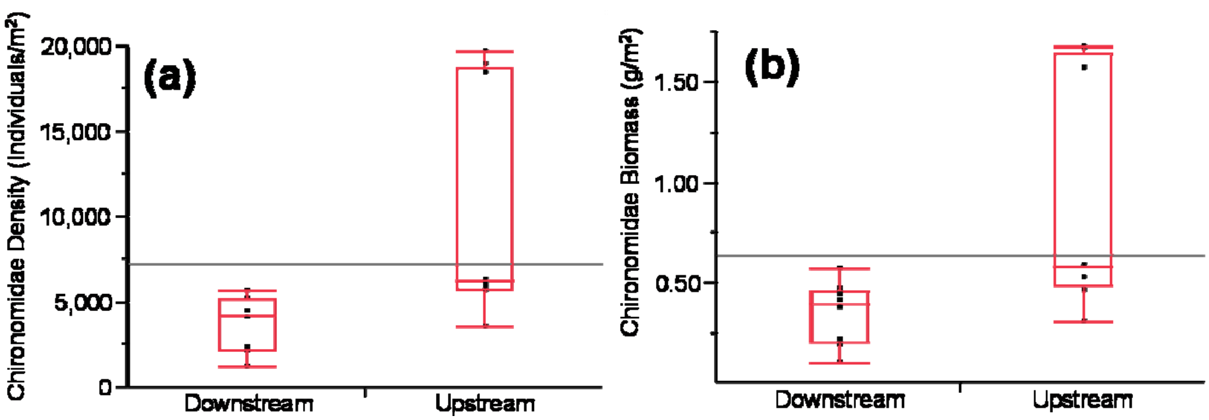


***Figure 71****. Peak Chironomidae density (a) and biomass (b) at upstream and downstream treatment sites. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

### *Hydropsychidae Density and Biomass*

For pooled samples (i.e., all samples combined), Hydropsychidae density at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 4 individuals/m$^2$; range = 0–35 individuals/m$^2$) and peaked at 10.0 weeks (70 days) inundation (average = 28 individuals/m$^2$; range = 0–163 individuals/m$^2$; Figure 72). Average density was not significantly different among treatment groups. However, average treatment densities at all inundation durations (range = 4–28 individuals/m$^2$) were significantly lower (pairwise Tukey tests $p$ = 0.0064–0.0111) than average control density (average = 398 individuals/m$^2$; range = 12–3,000 individuals/m$^2$). Hydropsychidae biomass at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 0.01 g/m$^2$; range = 0.00–0.12 g/m$^2$), peaked at 8.9 weeks (62 days) inundation (average = 0.12 g/m$^2$; range = 0.00–0.77 g/m$^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 0.11 g/m$^2$; range = 0.00–0.76 g/m$^2$; Figure 76). Average biomass was not significantly different among treatment groups. However, average treatment biomass at all inundation durations (range = 0.01–0.12 g/m$^2$) was significantly lower (pairwise Tukey tests $p$ = 0.0102–0.0209) than average control biomass (average = 1.35 g/m$^2$; range = 0.03–10.67 g/m$^2$). When Hydropsychidae density and biomass were compared among control sites, no significant differences in density ($F$ = 0.108; $df$ = 1, 14; $p$ = 0.7474) or biomass ($F$ = 0.151; $df$ = 1, 14; $p$ = 0.7032) were found between upstream and downstream sites (Figure 73). When density and biomass were compared among treatment sites, both average density ($F$ = 12.914; $df$ = 1, 76; $p$ = 0.0006) and biomass ($F$ = 11.795; $df$ = 1, 76; $p$ = 0.0010) were significantly greater at downstream sites when compared to upstream sites (Figure 73).

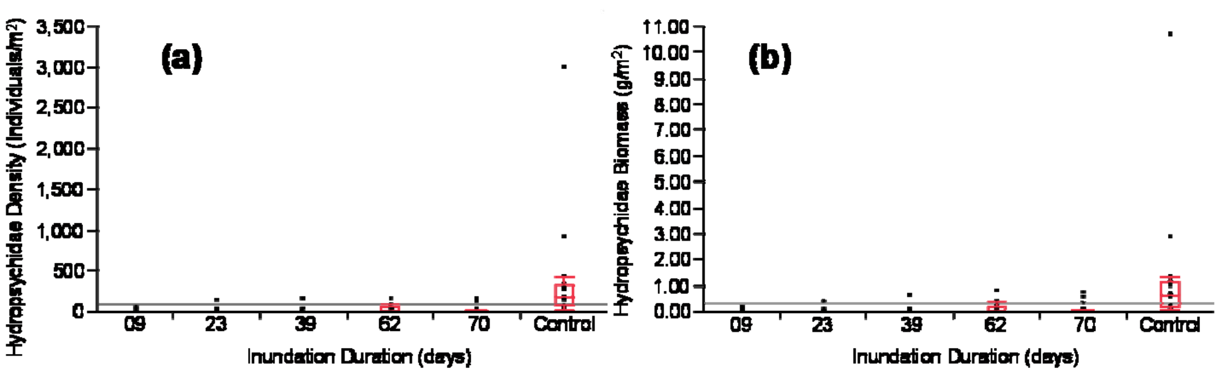


***Figure 72**. Hydropsychidae density (a) and biomass (b) at control and treatment sites. Data for treatment sites are displayed based on inundation duration. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

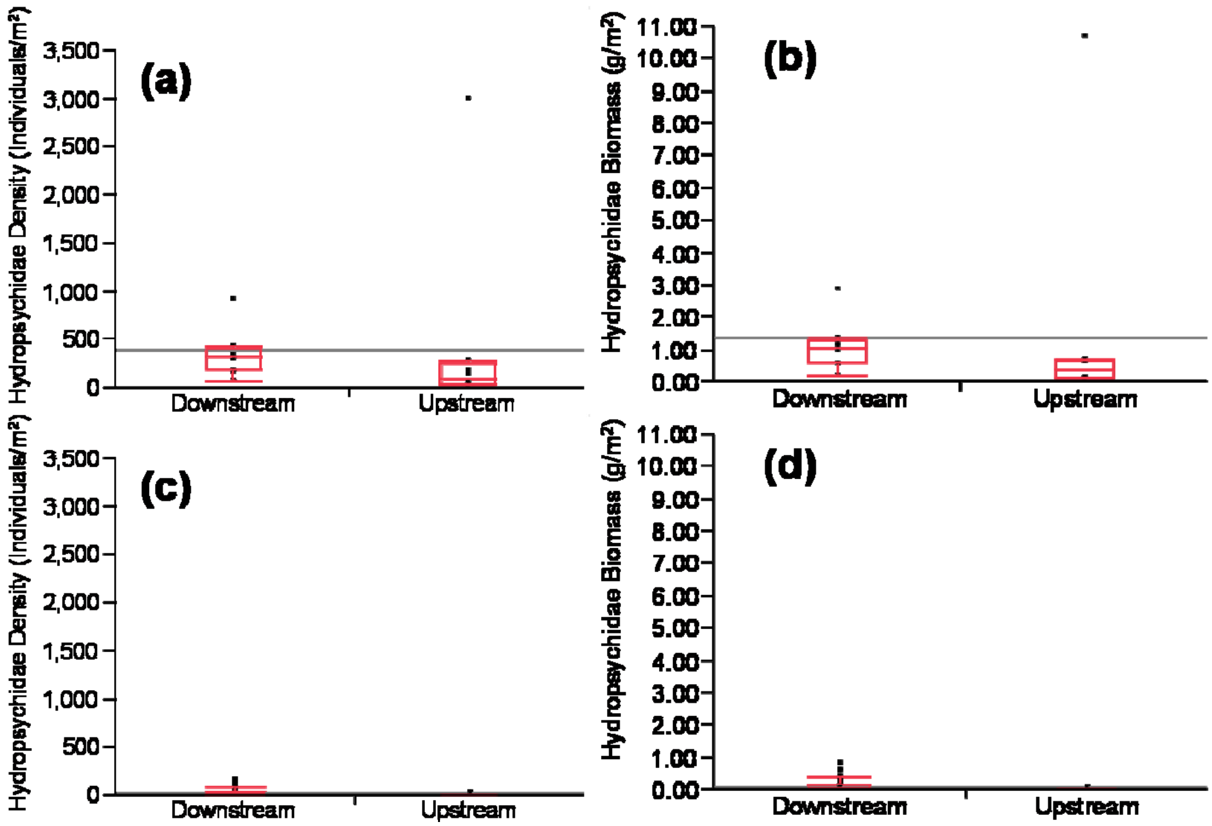


***Figure 73***. *Hydropsychidae density and biomass at upstream and downstream control (a and b) and treatment (c and d) sites. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

For upstream samples (2008 and 2009), average Hydropsychidae density was relatively low (range = 0–2 individuals/m$^2$) for all treatment groups, and showed no real trends (Figure 74). Average density was not significantly different among treatment groups, and no significant differences were found when average treatment group densities were compared to average control density (average = 459 individuals/m$^2$; range = 12–3,000 individuals/m$^2$). Average treatment density did not surpass average control density at any time during the sampling period (~1–10 weeks). Average Hydropsychidae biomass was relatively low (range = 0.00–0.01 g/m$^2$) for all treatment groups, and showed no real trends (Figure 74). Average biomass was not significantly different among treatment groups, and no significant differences were found when average treatment group biomass was compared to average control biomass (average = 1.61 g/m$^2$; range = 0.03–10.67 individuals/m$^2$). Average treatment biomass did not surpass average control biomass at any time during the sampling period (~1–10 weeks).

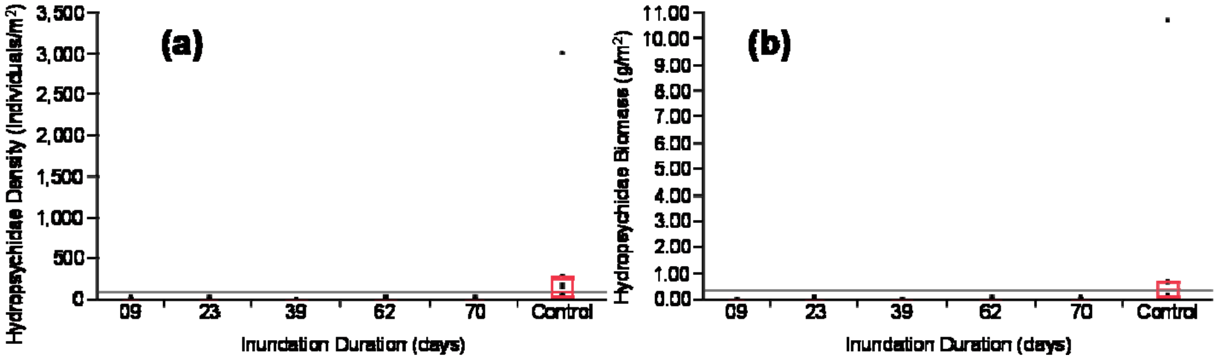


***Figure 74**. Hydropsychidae density (a) and biomass (b) at upstream control and treatment sites. Data for treatment sites are displayed based on inundation duration. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

For downstream samples (2010 and 2010 side-channel), Hydropsychidae density at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 6 individuals/m$^2$; range = 0–35 individuals/m$^2$) and peaked at 10.0 weeks (70 days) inundation (average = 55 individuals/m$^2$; range = 0–163 individuals/m$^2$; Figure 75). Average density was not significantly different among treatment groups. However, average treatment densities at all inundation durations (range = 6–55 individuals/m$^2$) were significantly lower (pairwise Tukey tests $_p$ <0.0001–0.0003) than average control density (average = 336 individuals/m$^2$; range = 58–919 individuals/m$^2$). Hydropsychidae biomass at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 0.02 g/m$^2$; range = 0.00–0.12 g/m$^2$), peaked at 8.9 weeks (62 days) inundation (average = 0.24 g/m$^2$; range = 0.00–0.77 g/m$^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 0.22 g/m$^2$; range = 0.00–0.76 g/m$^2$; Figure 75). Average biomass was not significantly different among treatment groups. However, average treatment biomass at all inundation durations (range = 0.02–0.24 g/m$^2$) was significantly lower (pairwise Tukey tests $_p$ <0.0001–0.0009) than average control biomass (average = 1.09 g/m$^2$; range = 0.16–2.89 g/m$^2$).

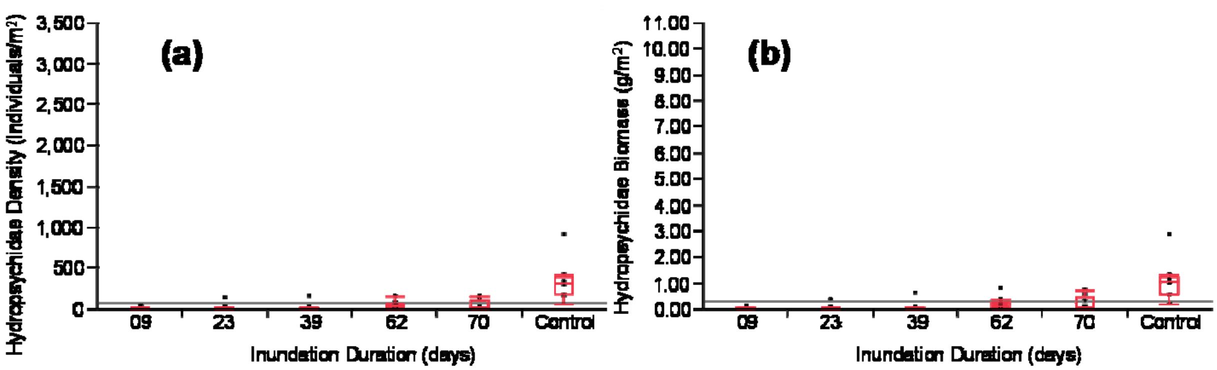


***Figure 75**. Hydropsychidae density (a) and biomass (b) at downstream control and treatment sites. Data for treatment sites are displayed based on inundation duration. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

When peak treatment density and biomass were compared between upstream and downstream sites, average peak density for downstream sites (10.0 weeks – 70 days inundation; average = 55 individuals/m$^2$) was 27.50-fold greater than that of upstream sites (1.3 weeks – 9 days inundation; average = 2 individuals/m$^2$), whereas average peak biomass for downstream sites (8.9 weeks – 62 days inundation; average = 0.24 g/m$^2$) was 24.00-fold greater than that of upstream sites (3.3 weeks – 23 days inundation; average = 0.01 g/m$^2$). These differences were significant for biomass ($F$ = 6.716; $df$ = 1, 14; $p$ = 0.0213), but not for density ($F$ = 3.567; $df$ = 1, 12; $p$ = 0.0834; Figure 76).

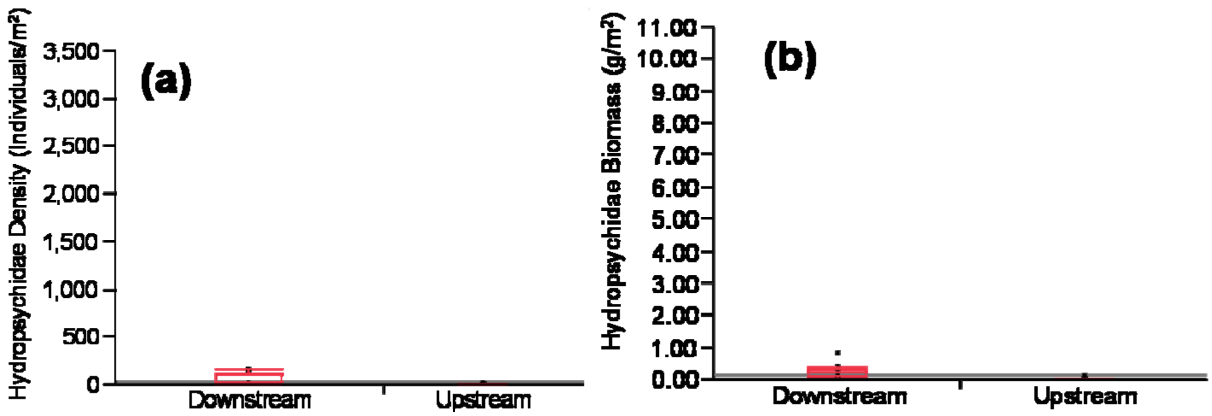


***Figure 76**. Peak Hydropsychidae density (a) and biomass (b) at upstream and downstream treatment sites. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

### *Total Baetidae, Chironomidae, and Hydropsychidae (BCH) Density and Biomass*

For pooled samples (i.e., all samples combined), total BCH density at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 66 individuals/m$^2$; range = 0–174 individuals/m$^2$), peaked at 8.9 weeks (62 days) inundation (average = 7,639 individuals/m$^2$; range = 2,000–19,686 individuals/m$^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 4,583 individuals/m$^2$; range = 1,814–12,140 individuals/m$^2$; Figure 77). Average treatment densities at 5.6, 8.9, and 10.0 weeks (39, 62, and 70 days) inundation (range = 4,583–7,639 individuals/m$^2$) were significantly greater (pairwise Tukey tests $p$ <0.0001–0.0086) than average treatment densities at 1.3 and 3.3 weeks (9 and 23 days) inundation (range = 66–591 individuals/m$^2$). However, average treatment density at 8.9 weeks (62 days) inundation showed the only significant difference (pairwise Tukey test $p$ = 0.0001) when compared to average control density (average = 2,294 individuals/m$^2$; range = 616–6,081 individuals/m$^2$). Total BCH biomass at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 0.02 g/m$^2$; range = 0.00–0.13 g/m$^2$), peaked at 8.9 weeks (62 days) inundation (average = 0.83 g/m$^2$; range = 0.22–1.68 g/m$^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 0.54 g/m$^2$; range = 0.15–1.15 g/m$^2$; Figure 77). Average biomass was not significantly different among treatment groups. However, average treatment biomass at 1.3 and 3.3 weeks (9 and 23 days) inundation (range = 0.02–0.09 g/m$^2$) was significantly lower (pairwise Tukey tests $p$ = 0.0043–0.0049) than average control biomass (average = 1.56 g/m$^2$; range = 0.11–10.97 g/m$^2$). Average treatment biomass did not surpass average control biomass at any time during the sampling period (~1–10 weeks). When total BCH density and biomass were compared among control sites, no significant differences in density ($F$ = 0.000; $df$ = 1, 14; $p$ = 0.9849) or biomass ($F$ = 0.106; $df$ = 1, 14; $p$ = 0.7498) were found between upstream and downstream sites (Figure 78). When total BCH density and biomass were compared among treatment sites, average total BCH density was significantly greater at upstream sites when compared to downstream sites ($F = 6.669$; $df = 1, 76$; $p = 0.0117$). However, total BCH biomass was not significantly different ($F = 0.697$; $df = 1, 76$; $p = 0.4063$; Figure 78).

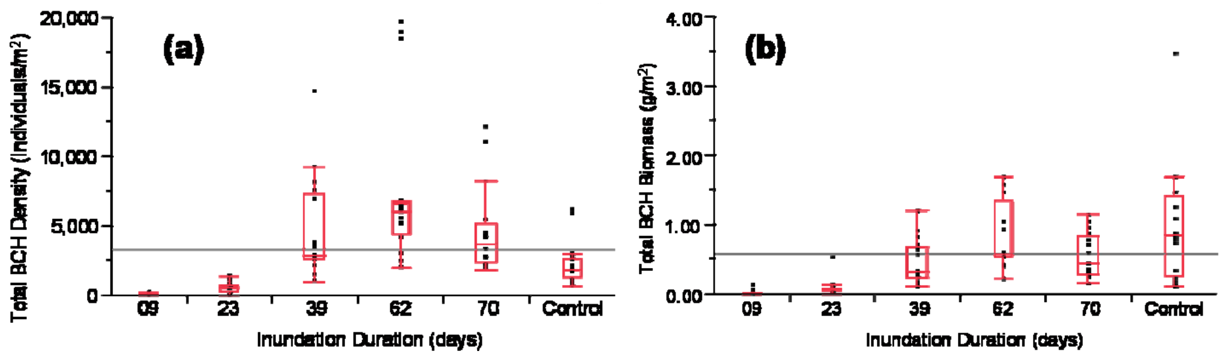


***Figure 77**. Total BCH density (a) and biomass (b) at control and treatment sites. Data for treatment sites are displayed based on inundation duration. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

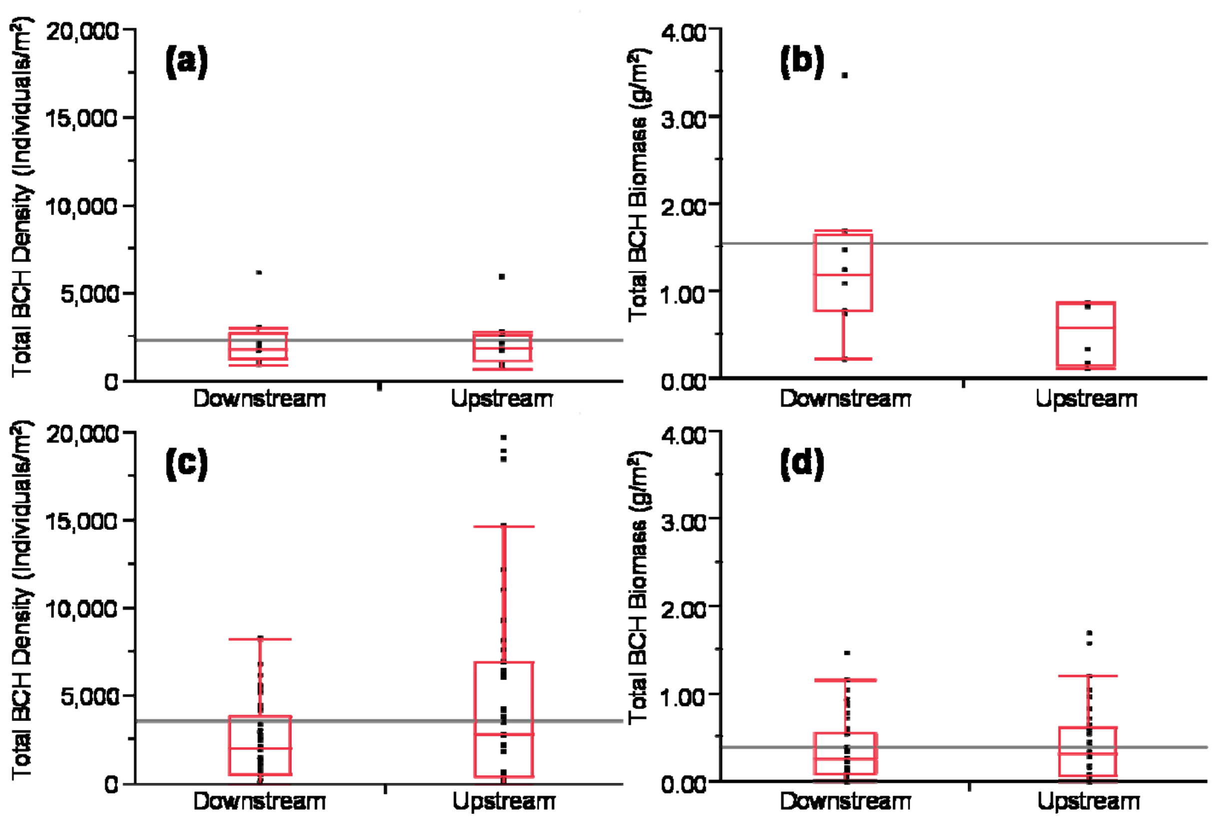


***Figure 78**. Total BCH density and biomass at upstream and downstream control (a and b) and treatment (c and d) sites. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

For upstream samples (2008 and 2009), total BCH density at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 76 individuals/$m^2$; range = 12–174 individuals/$m^2$), peaked at 8.9 weeks (62 days) inundation (average = 10,740 individuals/$m^2$; range = 4,081–19,686 individuals/$m^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 4,958 individuals/$m^2$; range = 616–5,942 individuals/$m^2$; Figure 79). Average treatment densities at 5.6 and 8.9 weeks (39 and 62 days) inundation (range = 6,959–10,740 individuals/$m^2$) were significantly greater (pairwise Tukey tests $_p$ <0.0001–0.0216) than average treatment densities at 1.3 and 3.3 weeks (9 and 23 days) inundation (range = 76–401 individuals/$m^2$). Similarly, average treatment density at 8.9 weeks (62 days) inundation was significantly greater (pairwise Tukey test $_p$ = 0.0469) than average treatment density at 10.0 weeks (70 days) inundation. Average treatment density at 8.9 weeks (62 days) inundation showed the only significant difference (pairwise Tukey test $_p$ = 0.0010) when compared to average control density (average = 2,286 individuals/$m^2$; range = 616–5,942 individuals/$m^2$). Total BCH biomass at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 0.01 g/$m^2$; range = 0.00–0.01 g/$m^2$), peaked at 8.9 weeks (62 days) inundation (average = 0.95 g/$m^2$; range = 0.42–1.68 g/$m^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 0.48 g/$m^2$; range = 0.15–1.04 g/$m^2$; Figure 79). Average biomass was not significantly different among treatment groups. Similarly, average treatment biomass was not significantly different from average control biomass at any time during the sampling period (~1–10 weeks). Average treatment biomass did not surpass average control biomass at any time during the sampling period (~1–10 weeks).

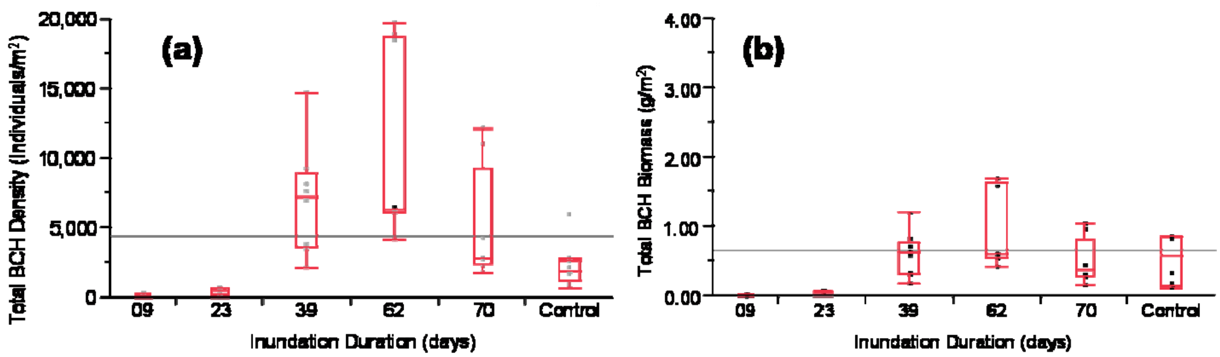


***Figure 79****. Total BCH density (a) and biomass (b) at upstream control and treatment sites. Data for treatment sites are displayed based on inundation duration. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

For downstream samples (2010 and 2010 side-channel), total BCH density at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 58 individuals/$m^2$; range = 0–116 individuals/$m^2$), peaked at 8.9 weeks (62 days) inundation (average = 4,538 individuals/$m^2$; range = 2,000–6,779 individuals/$m^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 4,208 individuals/$m^2$; range = 1,849–8,198 individuals/$m^2$; Figure 80). Average treatment densities at 8.9 and 10.0 weeks (62 and 70 days) inundation were significantly greater (pairwise Tukey tests $_p$ <0.0001–0.0001) than average treatment densities at 1.3 and 3.3 weeks (9 and 23 days) inundation (range = 58–781 individuals/$m^2$). Similarly, average treatment density at 5.6 weeks (39 days) inundation (average = 2,375 individuals/$m^2$) was significantly greater (pairwise Tukey test $_p$ = 0.0149) than average treatment density at 1.3 weeks (9 days) inundation. Additionally, average treatment density at 8.9 weeks (62 days) inundation was significantly greater (pairwise Tukey test $_p$ = 0.0273) than average treatment density at 5.6 weeks (39 days) inundation. Average treatment density at 1.3 weeks (9 days) inundation was significantly (pairwise Tukey test $p = 0.0199$) lower than average control density (average = 2,302 individuals/m$^2$; range = 849–6,081 individuals/m$^2$), whereas average treatment density at 8.9 weeks (62 days) inundation was significantly (pairwise Tukey test $p = 0.0206$) greater than average control density. Total BCH biomass at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 0.03 g/m$^2$; range = 0.00–0.13 g/m$^2$), peaked at 8.9 weeks (62 days) inundation (average = 0.72 g/m$^2$; range = 0.22–1.46 g/m$^2$), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 0.60 g/m$^2$; range = 0.15–3.47 g/m$^2$; Figure 80). Average biomass was not significantly different among treatment groups. However, average treatment biomass at 1.3, 3.3, 5.6, and 10.0 weeks (9, 23, 39, and 70 days) inundation (range = 0.03–0.60 g/m$^2$) was significantly lower (pairwise Tukey tests $p$ <0.0001–0.0340) than average control biomass (average = 1.34 g/m$^2$; range = 0.22–3.47 g/m$^2$). Average treatment biomass did not surpass average control biomass at any time during the sampling period (~1–10 weeks).

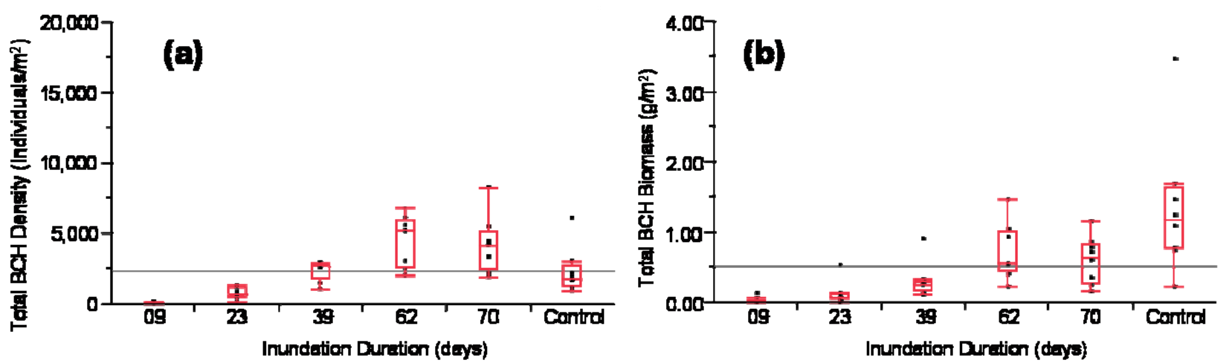


***Figure 80**. Total BCH density (a) and biomass (b) at downstream control and treatment sites. Data for treatment sites are displayed based on inundation duration. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

When peak treatment density and biomass (8.9 weeks – 62 days inundation) were compared between upstream and downstream sites, average peak density for upstream sites (average = 10,740 individuals/m$^2$) was 2.37-fold greater than that of downstream sites (average = 4,538 individuals/m$^2$), whereas average peak biomass for upstream sites (average = 0.95 g/m$^2$) was 1.32-fold greater than that of downstream sites (average = 0.72 g/m$^2$). These differences were significant for density ($F = 6.032$; $df = 1, 14$; $p = 0.0277$), but not for biomass ($F = 0.913$; $df = 1, 14$; $p = 0.3557$; Figure 81).

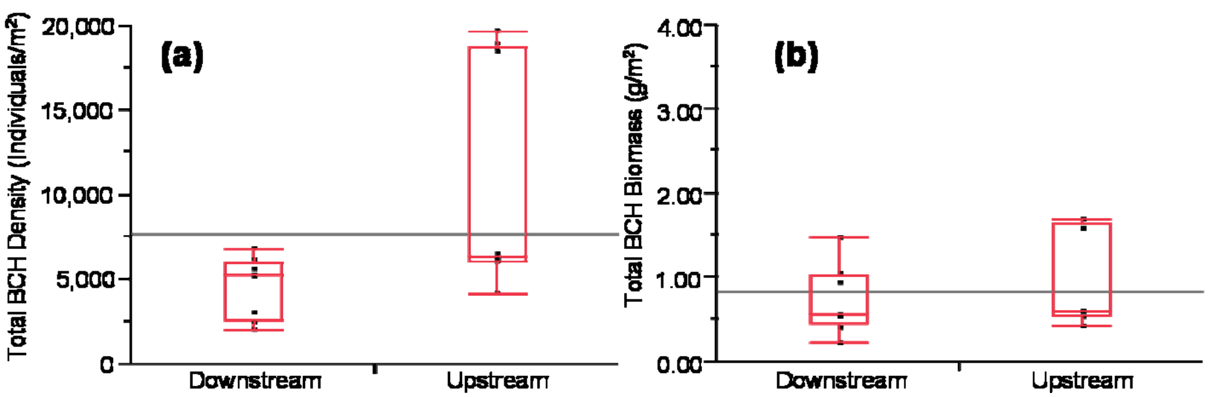


***Figure 81***. *Peak total BCH density (a) and biomass (b) at upstream and downstream treatment sites. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

***Proportional Baetidae, Chironomidae, and Hydropsychidae (BCH) Density and Biomass***

For pooled samples (i.e., all samples combined), proportional Baetidae ($X^2 = 1{,}424.948$; $df = 1, 199{,}721$; $p < 0.0001$) and Hydropsychidae ($X^2 = 9{,}131.514$; $df = 1, 153{,}151$; $p < 0.0001$) density were significantly higher in control samples when compared to treatment samples taken at peak treatment densities, whereas proportional Chironomidae ($X^2 = 6{,}290.492$; $df = 1, 326{,}128$; $p < 0.0001$) and total BCH ($X^2 = 1{,}796.723$; $df = 1, 345{,}791$; $p < 0.0001$) density were significantly lower (Figure 82). When biomass was used in place of density, proportional Hydropsychidae ($X^2 = 7.051$; $df = 1, 92$; $p = 0.0079$) biomass was significantly higher in control samples, whereas proportional Chironomidae ($X^2 = 10.299$; $df = 1, 80$; $p = 0.0013$) biomass was significantly lower. No significant differences were found for proportional Baetidae ($X^2 = 0.245$; $df = 1, 70$; $p = 0.6206$) or total BCH ($X^2 = 0.003$; $df = 1, 106$; $p = 0.9565$) biomass (Figure 82).

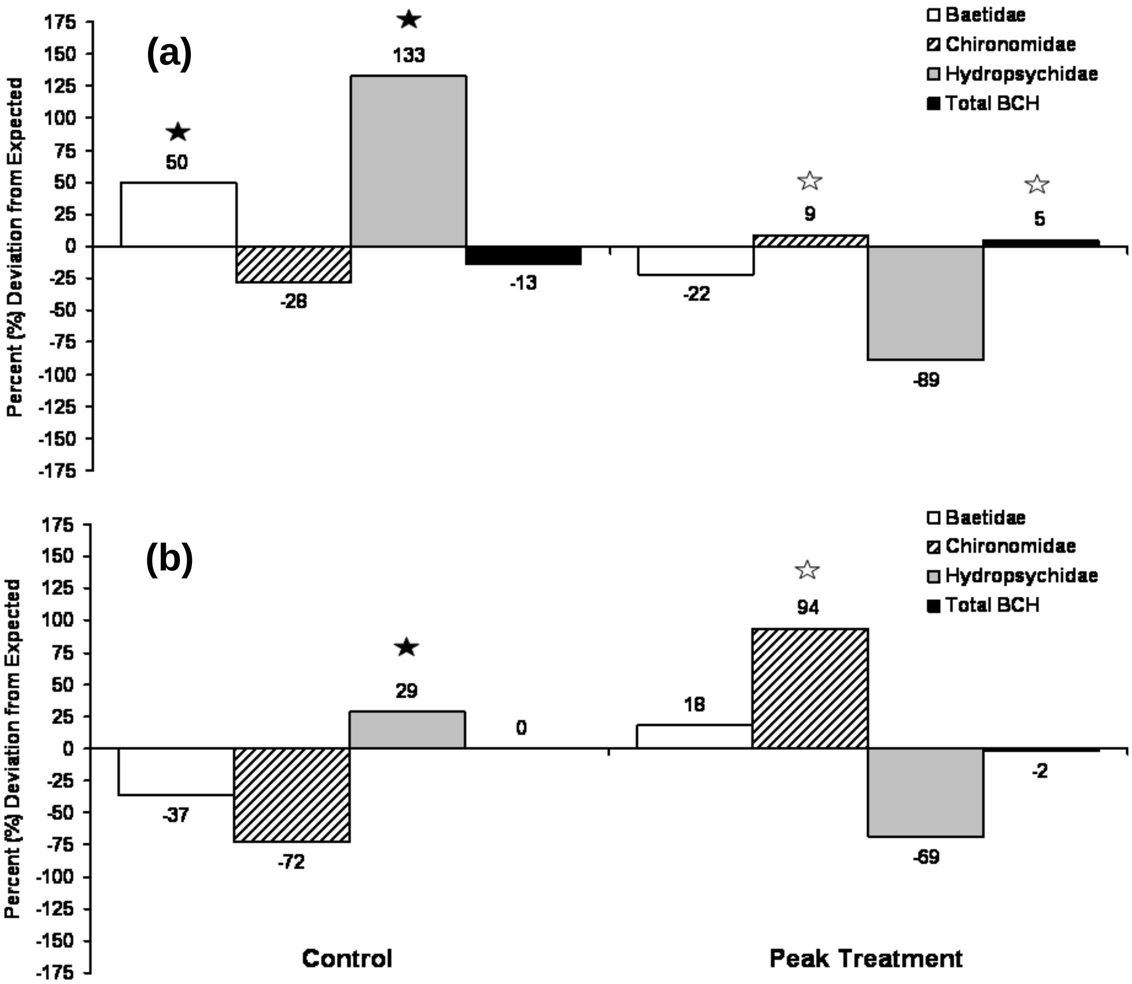


***Figure 82**. Percent (%) deviation of observed values from expected values based on chi-square analyses for proportional density (a) and biomass (b) using pooled data (i.e., upstream and downstream). Figures compare control samples to treatment samples taken at the time of peak density or biomass. Stars represent samples with significantly higher proportional density or biomass, with solid stars indicating significantly higher proportional density or biomass in control samples and open stars indicating significantly higher proportional density or biomass in treatment samples.*

For upstream samples (2008 and 2009), proportional Baetidae ($\chi^2$ = 899.418; $df$ = 1, 126,663; $p$ <0.0001) and Hydropsychidae ($\chi^2$ = 133.160; $df$ = 1, 36,837; $p$ <0.0001) density were significantly higher in control samples when compared to treatment samples taken at peak treatment densities, whereas proportional Chironomidae ($\chi^2$ = 5,452.869; $df$ = 1, 221,628; $p$ <0.0001) and total BCH ($\chi^2$ = 2,363.897; $df$ = 1, 228,093; $p$ <0.0001) density were significantly lower (Figure 83). When biomass was used in place of density, proportional Chironomidae ($\chi^2$ = 9.688; $df$ = 1, 52; $p$ = 0.0019) biomass was significantly lower in control samples. No significant differences were found for proportional Baetidae ($\chi^2$ = 0.050; $df$ = 1, 43; $p$ = 0.8238), Hydropsychidae ($\chi^2$ = 1.743; $df$ = 1, 46; $p$ = 0.1867), or total BCH ($\chi^2$ = 0.140; $df$ = 1, 65; $p$ = 0.7083) biomass (Figure 83).

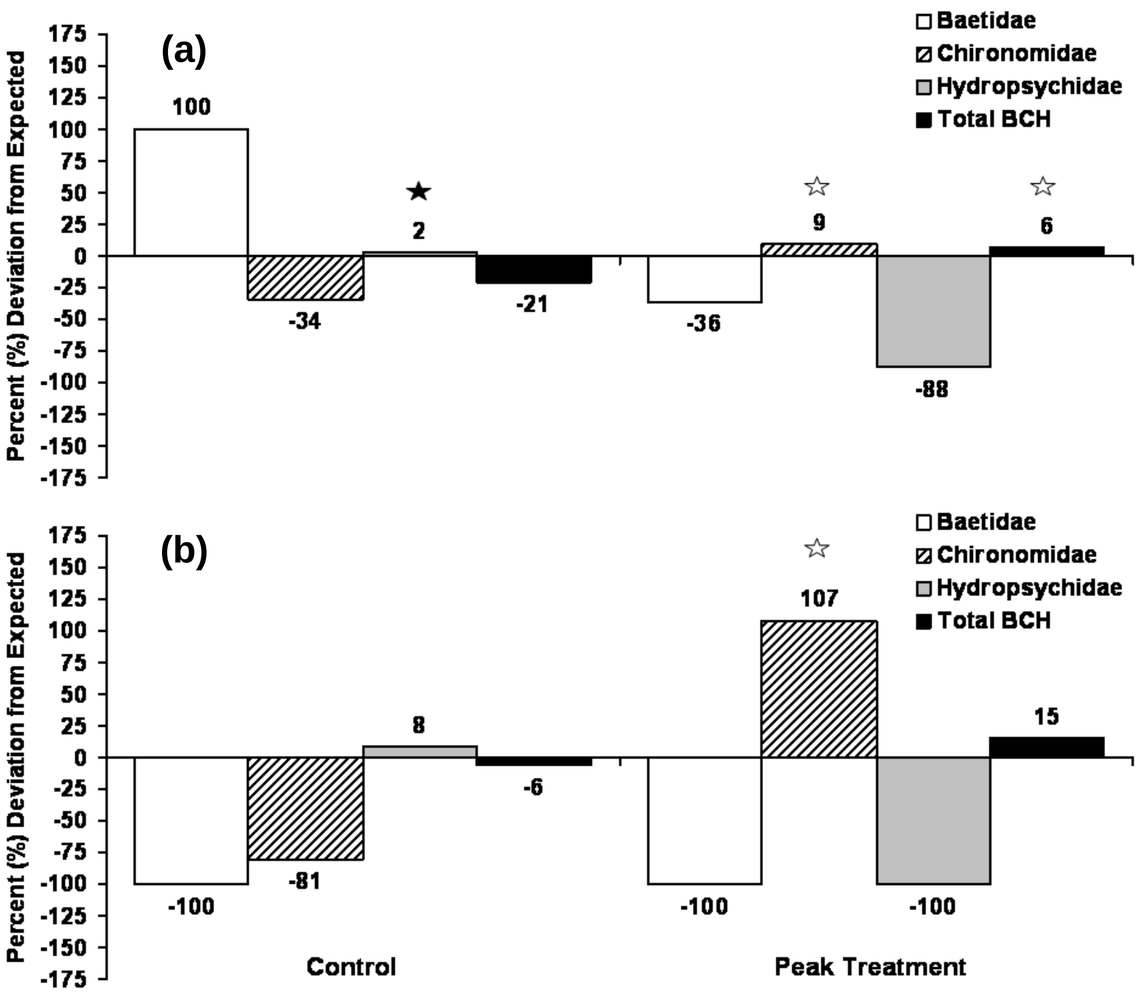


***Figure 83***. *Percent (%) deviation of observed values from expected values based on chi-square analyses for proportional density (a) and biomass (b) using upstream data. Figures compare control samples to treatment samples taken at the time of peak density or biomass. Stars represent samples with significantly higher proportional density or biomass, with solid stars indicating significantly higher proportional density or biomass in control samples and open stars indicating significantly higher proportional density or biomass in treatment samples.*

For downstream samples (2010 and 2010 side-channel), proportional Baetidae ($\chi^2$ = 206.044; $df$ = 1, 73,058; $p$ <0.0001) and Hydropsychidae ($\chi^2$ = 3,685.474; $df$ = 1, 73,779; $p$ <0.0001) density were significantly higher in control samples when compared to treatment samples taken at peak treatment densities, whereas proportional Chironomidae ($\chi^2$ = 947.245; $df$ = 1, 104,500; $p$ <0.0001) and total BCH ($\chi^2$ = 90.999; $df$ = 1, 117,698; $p$ <0.0001) density were significantly lower (Figure 84). When biomass was used in place of density, no significant differences were found for proportional Baetidae ($\chi^2$ = 0.068; $df$ = 1, 27; $p$ = 0.7941), Chironomidae ($\chi^2$ = 1.452; $df$ = 1, 29; $p$ = 0.2282), Hydropsychidae ($\chi^2$ = 1.959; $df$ = 1, 36; $p$ = 0.1616), or total BCH ($\chi^2$ = 0.175; $df$ = 1, 42; $p$ = 0.6761) biomass (Figure 84).

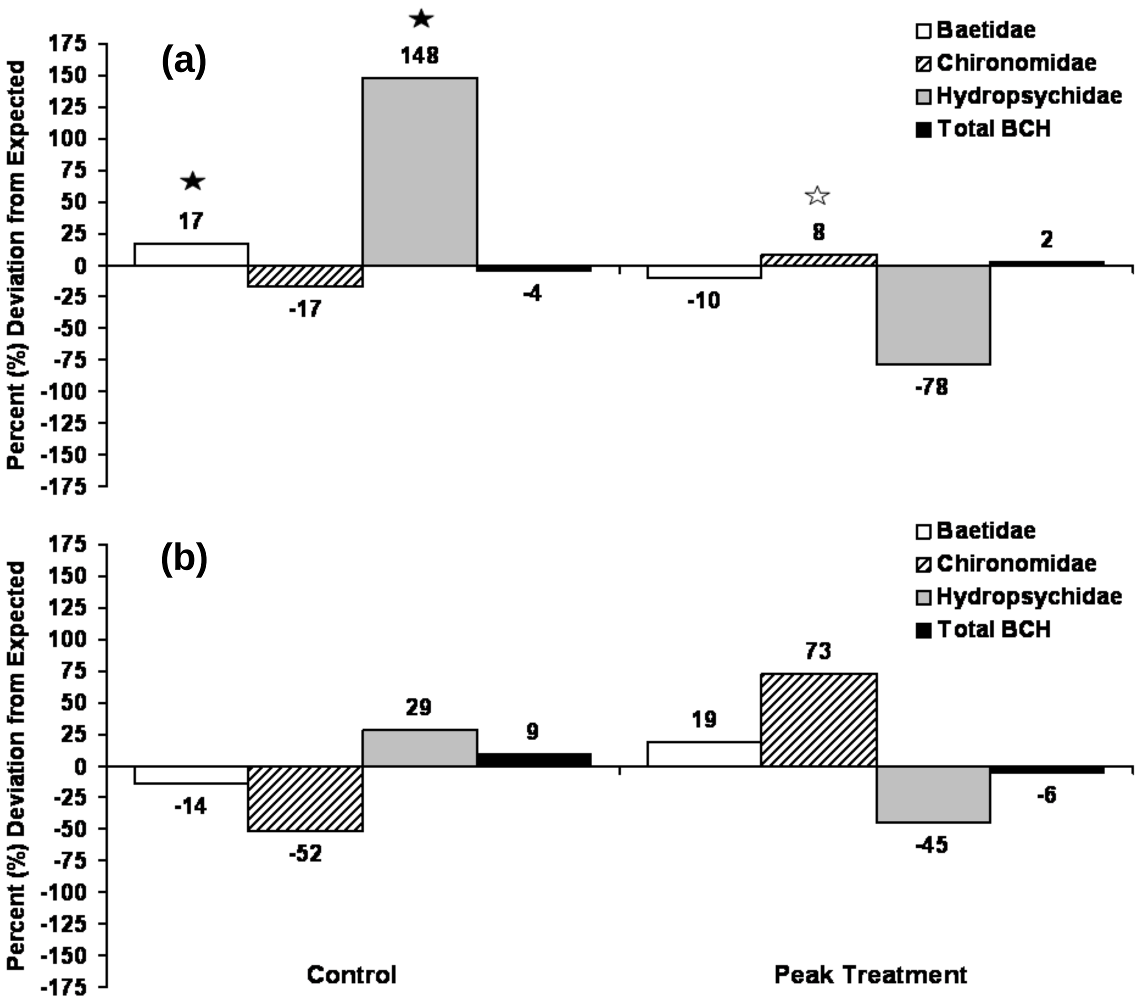


***Figure 84**. Percent (%) deviation of observed values from expected values based on chi-square analyses for proportional density (a) and biomass (b) using downstream data. Figures compare control samples to treatment samples taken at the time of peak density or biomass. Stars represent samples with significantly higher proportional density or biomass, with solid stars indicating significantly higher proportional density or biomass in control samples and open stars indicating significantly higher proportional density or biomass in treatment samples.*

### *Benthic Macroinvertebrate Richness and Diversity*

For pooled samples (i.e., all samples combined), richness on floodplain (treatment) sites was lowest at 1.3 weeks (9 days) inundation (average = 3.1 families; range = 1–6 families), peaked at 8.9 weeks (62 days) inundation (average = 7.0 families; range = 5–10 families), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 6.9 families; range = 4–12 families; Figure 85). Average richness at 8.9 and 10.0 weeks (62 and 70 days) inundation was significantly greater (pairwise Tukey tests $_p$ <0.0001–0.0464) than average richness at 1.3, 3.3, and 5.6 weeks (9, 23, and 39 days) inundation (range = 3.1–4.8 families). However, average richness at all inundation durations was significantly lower (pairwise Tukey tests $_p$ <0.0001–0.0222) than average control richness (average = 9.3 families; range = 4–14 families). Simpson's Diversity Index values at treatment sites showed a "bowl-shaped" curve, with localized peaks at 1.3 weeks (9 days) inundation (average = 2.27; range = 1.00–4.17) and 10.0 weeks (70 days) inundation (average = 1.78; range = 1.04–2.71), and a valley at 5.6 weeks (39 days) inundation (average = 1.29; range = 1.00–3.02; Figure 85). Average treatment Simpson's Diversity Index values did not surpass average control Simpson's Diversity Index values at any time during the sampling period (~1–10 weeks). When richness was compared among control sites, no significant difference ($F = 0.159$; $df = 1, 14$; $p = 0.6960$) was found between upstream and downstream sites. Simpson's Diversity Index values were also similar when compared among control sites (Figure 86). In contrast, when richness was compared among treatment sites, richness was significantly greater ($F = 12.406$; $df = 1, 76$; $p = 0.0007$) at downstream sites when compared to upstream sites. Similarly, Simpson's Diversity Index values were generally greater at downstream sites when compared to upstream sites (Figure 86).

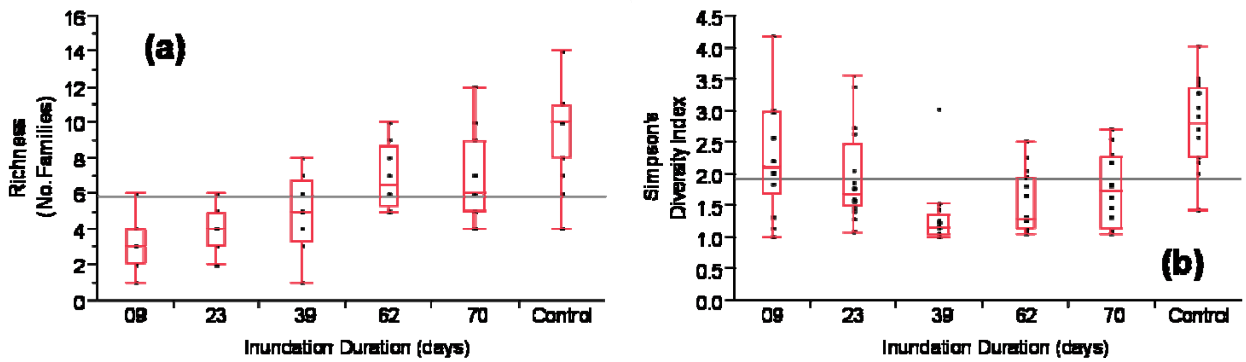


***Figure 85**. Richness (a) and Simpson's Diversity Index (b) at control and treatment sites. Data for treatment sites are displayed based on inundation duration. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

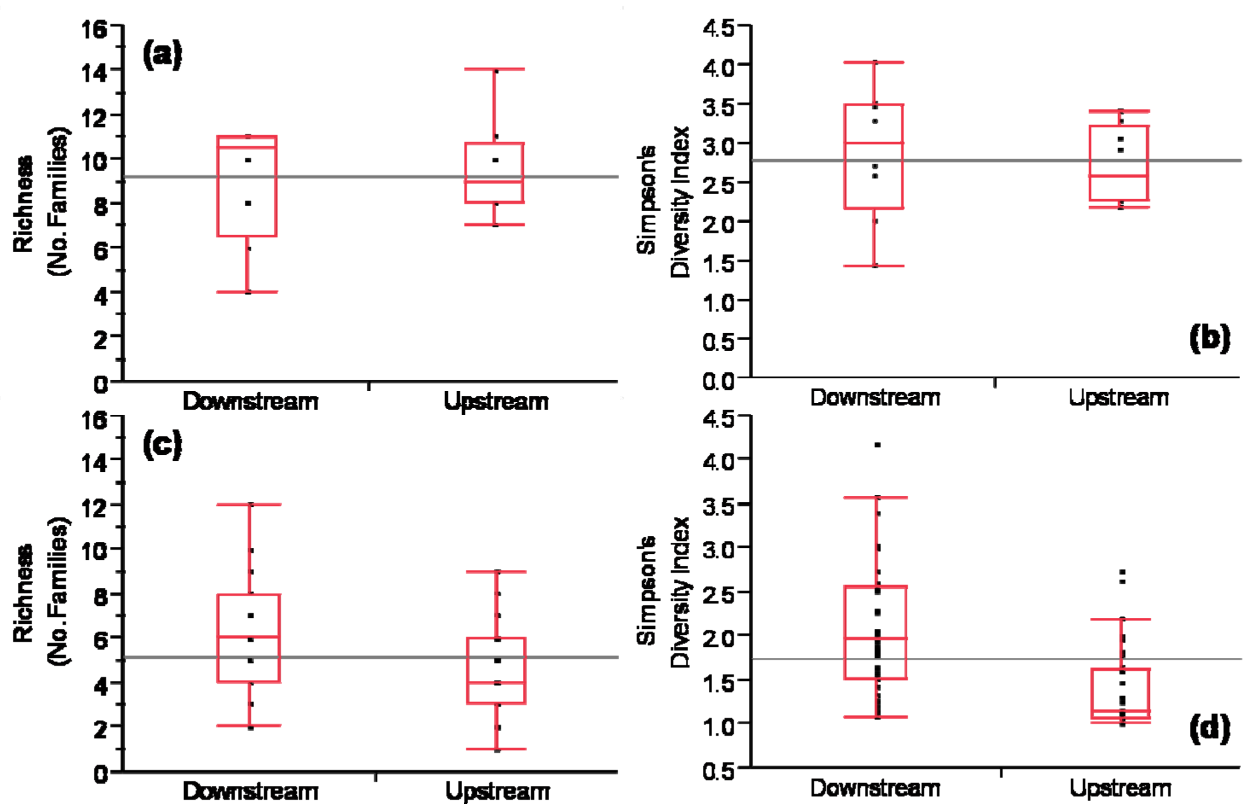


***Figure 86****. Richness and Simpson's Diversity Index at upstream and downstream control (a and b) and treatment (c and d) sites. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

For upstream samples (2008 and 2009), richness at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 2.3 families; range = 1–4 families), peaked at 8.9 weeks (62 days) inundation (average = 6.6 families; range = 5–9 families), and then fell but remained relatively high at 10.0 weeks (70 days) inundation (average = 5.3 families; range = 4–7 families; Figure 87). Average richness at 8.9 weeks (62 days) inundation was significantly greater (pairwise Tukey tests $_p$ = 0.0001–0.0035) than average richness at 1.3, 3.3, and 5.6 weeks (9, 23, and 39 days) inundation (range = 2.3–3.5 families). Similarly, average richness at 10.0 weeks (70 days) inundation was significantly greater (pairwise Tukey test $_p$ = 0.0158) than average richness at 1.3 weeks (9 days) inundation. Average richness at all inundation durations was significantly lower (pairwise Tukey tests $_p$ <0.0001–0.0086) than average control richness (average = 9.5 families; range = 7–14 families). Simpson's Diversity Index values at treatment sites showed a "bowl-shaped" curve, with localized peaks at 3.3 weeks (23 days) inundation (average = 1.76; range = 1.07–2.71) and 10.0 weeks (70 days) inundation (average = 1.35; range = 1.04–2.19), and a valley at 5.6 weeks (39 days) inundation (average = 1.05; range = 1.00–1.22; Figure 87). Average treatment Simpson's Diversity Index values did not surpass average control Simpson's Diversity Index values at any time during the sampling period (~1–10 weeks).

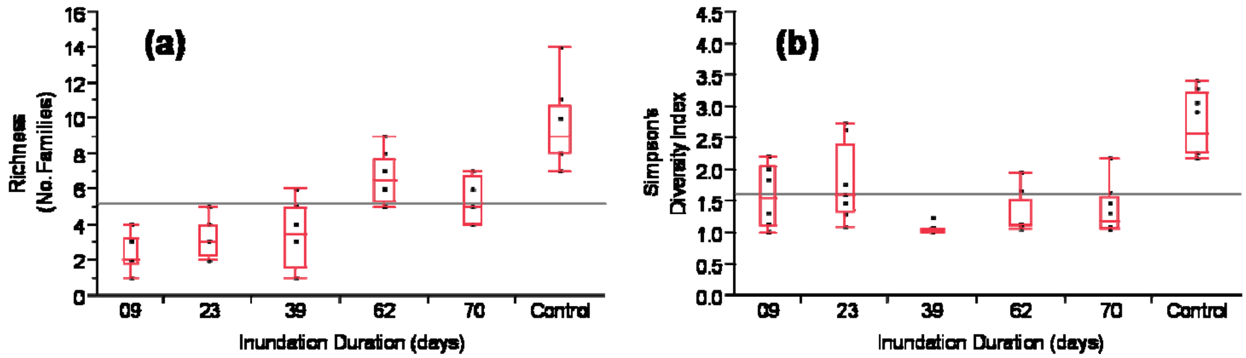


***Figure 87***. *Richness (a) and Simpson's Diversity Index (b) at upstream control and treatment sites. Data for treatment sites are displayed based on inundation duration. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

For downstream samples (2010 and 2010 side-channel), richness at treatment sites was lowest at 1.3 weeks (9 days) inundation (average = 3.6 families; range = 2–6 families) and peaked at 10.0 weeks (70 days) inundation (average = 8.5 families; range = 6–12 families; Figure 88). Average richness at 10.0 weeks (70 days) inundation was significantly greater (pairwise Tukey tests $_p$ = 0.0002–0.0051) than average richness at 1.3 and 3.3 weeks (9 and 23 days) inundation (range = 3.6–4.8 families). Similarly, average richness at 8.9 weeks (62 days) inundation was significantly greater (pairwise Tukey test $_p$ = 0.0051) than average richness at 1.3 weeks (9 days) inundation. Average treatment richness was lower than average control richness (average = 9.0 families; range = 4–11 families) at all inundation durations. However, these differences were only significant (pairwise Tukey tests $_p$ <0.0001–0.0011) at 1.3 and 3.3 weeks (9 and 23 days) inundation. Simpson's Diversity Index values at treatment sites showed a "bowl-shaped" curve, with localized peaks at 1.3 weeks (9 days) inundation (average = 2.79; range = 2.00–4.17) and 10.0 weeks (70 days) inundation (average = 2.21; range = 1.64–2.71), and a valley at 5.6 weeks (39 days) inundation (average = 1.52; range = 1.13–3.02; Figure 88). Average treatment Simpson's Diversity Index values did not surpass average control Simpson's Diversity Index values at any time during the sampling period (~1–10 weeks).

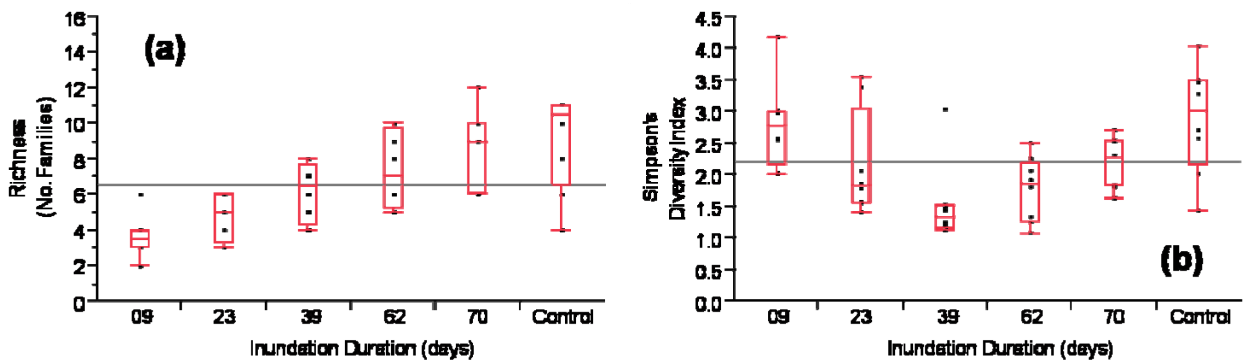


***Figure 88***. *Rrichness (a) and Simpson's Diversity Index (b) at downstream control and treatment sites. Data for treatment sites are displayed based on inundation duration. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

When peak treatment richness was compared between upstream (8.9 weeks – 62 days inundation) and downstream (10.0 weeks – 70 days inundation) sites, average peak richness for downstream sites (average = 8.5 families) was 1.29-fold greater than that of upstream sites (average = 6.6 families). However, this difference was not significant ($F$ = 3.947; $df$ = 1, 14; $p$ = 0.0669; Figure 89). Similarly, when peak treatment Simpson's Diversity Index values were compared between upstream (3.3 weeks – 23 days inundation) and downstream (1.3 weeks – 9 days inundation) sites, the average peak Simpson's Diversity Index value at downstream sites (average = 2.79) was 1.59-fold greater than that of upstream sites (average = 1.76 families; Figure 89).

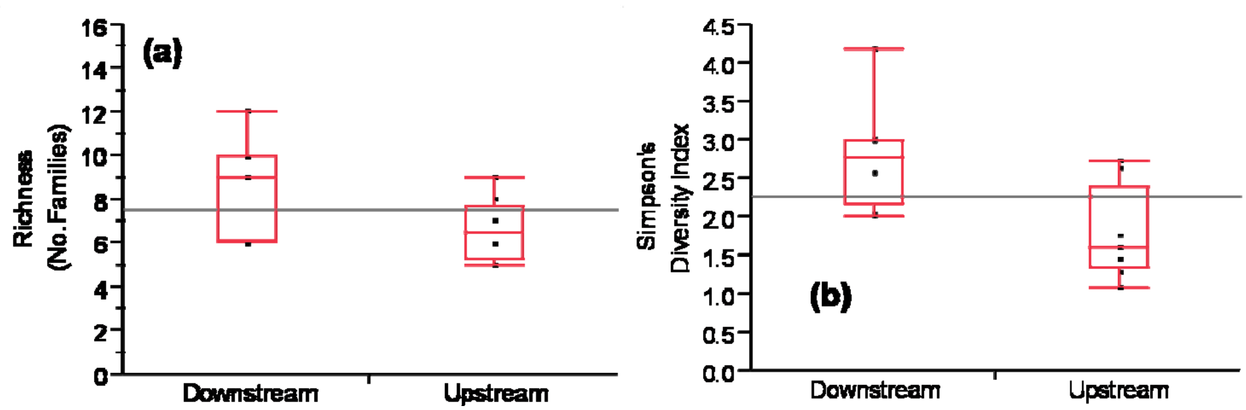


***Figure 89****. Peak richness (a) and Simpson's Diversity Index (b) at upstream and downstream treatment sites. Whiskers indicate 1% and 99% quantiles, whereas boxes indicate 25%, 50% (median), and 75% quartiles. Gray line indicates the grand mean.*

## DISCUSSION

### *Floodplain inundation*

Past investigations by DFG using rotary screw traps near Watt Avenue (Snider and Titus 1995; Snider and others 1997, 1998 as reported in Williams 2001) show that the overwhelming majority of juvenile Chinook salmon leave LAR spawning areas shortly after emerging, with emigration usually peaking in February. Whether this is the result of competition for limited rearing space driving fry from the spawning reach or a successful strategy for LAR juvenile production is unclear. Central Valley fall-run Chinook salmon populations appear to exhibit two temporal pulses of underyearling outmigration, designated as fry size (30–50 mm FL), peaking in February–March, and smolt size (>60 mm FL), peaking in May–June. The clear bimodal pattern suggests two distinct life-histories rather than a continuous migration. The proportion of fish following the two respective pathways differs by population. Williams (2006) suggested that most Sacramento River fall-run Chinook salmon migrate downstream early at fry sizes, whereas most San Joaquin River fall-run populations migrate later at smolt sizes. There is good evidence for genetic variation in the propensity of underyearling ocean-type Chinook salmon to migrate as fry (Carl and Healey 1984). However, year-to-year variation in the proportion of fry migrants in some Central Valley rivers suggests that environmental factors also matter (Williams 2006). Vogel and Marine (2000) and Workman (2002) presented data suggesting that fish are more likely to emigrate as fry when densities are high. Other likely determinants include growth opportunity and physical factors such as temperature and flow. Whether a genetic or environmental threshold may influence the likelihood of one life-history being expressed over another is still open to debate.

Juvenile rearing habitat requirements estimated in our study largely correspond to the early and late juvenile emigration peaks observed for fall-run Chinook salmon populations throughout the Sacramento River system (Williams 2006). Our results suggest that the relative timing of peak rearing habitat requirements in the LAR remains largely unchanged from "critical" to "wet" water years. However, the relative magnitude of the two peaks appears to vary dramatically, with higher proportional rearing habitat requirements for the May–June peak in "wet" water years and lower proportional rearing habitat requirements for the May–June peak in "critical" water years. Floodplain inundation mapping also suggests that an example "wet" water year release strategy provides a higher degree of correlation between available inundated areas and the early peak in LAR juvenile rearing habitat requirements when compared to an example "critical" water year release strategy. In short, water management strategies can be tailored to increase the availability of beneficial habitats to rearing juvenile salmonids at times when these early life stages are prevalent throughout the LAR.

Although our results are preliminary and based on only two representative data years for these two water-year types (1994 and 1999), they are consistent with the idea of juvenile rearing habitat limitation in the LAR, and suggest that additional inundated areas available during the early fry peak in "wet" water years may lead to increased demand for rearing habitat during the later smolt peak. This concept is consistent with the results of Vogel and Marine (2000) and Workman (2002), and suggests that changes in release strategy or the creation of additional juvenile rearing habitat throughout the LAR may be required to increase the diversity of juvenile life-history strategies expressed by the LAR fall-run Chinook salmon population. Given the limited number of adults spawning during the 1993–1994 and 1998–1999 monitoring seasons (Snider and Vyverberg 1995), changes in release strategy and additional rearing habitat area may become even more important in the future as managers move toward AFRP adult production targets (160,000 spawning adults; available: http://www.fws.gov/stockton/afrp/ws_stats.cfm?code=AMERR). Currently, additional inundated areas adjacent to the 2008–2010 gravel augmentation sites provide an average of <1.0% of daily juvenile rearing habitat requirements assuming a "critical" release strategy and an average of 2.4% of daily juvenile rearing habitat requirements assuming a "wet" release strategy (both based on expected juvenile production from 160,000 spawning adults). If more LAR rearing habitat were added, would this lead to a greater variety of juvenile life-history strategies, including genetic variation? As habitat manipulation continues on the LAR it is important to evaluate how juvenile life-history strategies respond to these activities, and if future responses will be in-line with restoration goals and buffer LAR fish against environmental variability (e.g., Lindley et al 2007). Additional monitoring over several years is needed to improve and validate preliminary model results presented herein.

### *Spawning habitat use*

The 530 total annual Chinook salmon redds observed in 2010–2011 represented an increase from a low of 316 in 2009–2010, but were still below the mean of 2,181 for all seven survey years.

Overall, post-enhancement utilization rates of placed gravel and newly-inundated side-channel habitats increased dramatically compared to pre-enhancement rates. However, not all enhancement sites experienced similar levels of utilization. The 2008 enhancement site had the highest utilization rates (33%), followed by the newly-inundated side-channel (7.9%) and the 2009 enhancement site (6.4%). No redds were observed in the 2010 enhancement site. The 2008 and 2009 sites and the newly-inundated side-channel all experienced significant increases in utilization compared to pre-enhancement conditions. However, the 2010 enhancement site exhibited significant decreases in utilization. It should be noted that only two aerial surveys were conducted in 2010 compared to three in most other years, and high flow levels in December precluded regular sampling during the entire Chinook salmon spawning season, likely affecting these results. Mean daily flow continually exceeded 10,000 $ft^3 \bullet s^{-1}$ (283 $m^3 \bullet s^{-1}$) from 13 December 2010 through 4 January 2011, and peaked at over 31,425 $ft^3 \bullet s^{-1}$ (890 $m^3 \bullet s^{-1}$) on 16 December 2010.

The 92 total annual steelhead trout redds observed in 2010–2011 represented a slight increase from a low of 79 in 2009–2010, but were still below the mean of 146 for all eight survey years. Unlike for Chinook salmon, steelhead trout post-enhancement utilization rates of placed gravel and newly-inundated side-channel habitats only increased at the 2008 site (40.2%) when compared to pre-enhancement rates. The newly-inundated side-channel accounted for ~10% total utilization and was similar to pre-enhancement utilization rates (range: 9% to ~23%). Likewise, utilization rates at the 2009 enhancement site (7.6% in 2009-2010 and 0% in 2010-2011) did not differ significantly from pre-enhancement rates (range: 1.9% to 9.7%). Utilization remained extremely low at the 2010 enhancement site (1.1%) and was comparable to pre-enhancement rates (range: 0% to 0.6%). Mean daily flow was less than 3,000 $ft^3 \bullet s^{-1}$ (85 $m^3 \bullet s^{-1}$) from 20 January through 28 February, and exceeded 9,600 $ft^3 \bullet s^{-1}$ (272 $m^3 \bullet s^{-1}$) continuously from 14 March 2011 through 15 April 2011; a peak flow of 26,259 $ft^3 \bullet s^{-1}$ (744 $m^3 \bullet s^{-1}$) occurred on 17 March 2010.

Extreme high flow levels can have multiple effects on spawning salmonids. First, fish may be displaced or precluded from spawning by excessively high velocities and increased depths, thereby making areas normally utilized for spawning inaccessible. Second, completed redds may experience increased rates of scour, possibly resulting in increased mortality (Montgomery et al. 1996) and eggs and embryos being washed away (McNeil 1966). During bed mobilization events, redds may also be covered with additional layers of substrate, thereby limiting incubation and hatching sucess. In the Greenwater River, Washington, Schuett-Hames and Adams (2003) found strong relationships between peak flow and both mean scour depths and mean bed elevation change. The authors also investigated the occurrence of >15-cm scour depths (the top of Chinook salmon egg pocket depth) and found that potentially damaging scour events occured 25%, 50%, and 75% of the time during >2-yr, 4-yr, and 7-yr incubation flow return intervals, respectively. In the Skagit River, Washington, egg-to-smolt survival ranged from 1% during the highest flow year to 22% during the lowest flow year (based on peak incubation flows; Seiler et al. 2000). Therefore, it is possible that high flow events resulting in high degrees of scour could have devastating effects on existing redds. Similarly, egg and embryo mortality could increase with substrate filling over existing redds. May et al. (2009) found the risk of aggradation to be greater than the risk of scour since deposition is more uniform across the channel whereas scour tends to be more localized. The effect of increased fill over redds and subsequent survival to emergence is less understood. It is important to note that scour and deposition are essential physical processes that help develop and maintain a healthy river ecosystem, including salmonid spawning and rearing habitat. Future work should assess when and how managed flows can meet flood requirements and maintain these processes without damaging previously established salmonid redds.

Similar to depth and velocity assessments (Watry and Merz 2009; CFS 2010), this analysis of substrate size preferences was based on data from one year and only for the 2008–2010 augmentation sites. Therefore, we were working with a relatively limited range of available substrates with a large proportion contained within discrete $D_{50}$ and $D_{85}$ size categories. This could potentially lead to relatively narrow substrate size curves. However, this preliminary analysis suggests that an estimate of maximum movable substrate sizes should be used to inform future spawning habitat enhancement design criteria. For example, this analysis suggests that average-sized Chinook salmon could spawn in the 2008 and 2010 sites but not the 2009 site based on site-specific placed gravel $D_{95}$s. In contrast, average-sized steelhead trout could spawn in the 2008 site but not the 2009 or 2010 sites. From this analysis, we hypothesize that the limited spawning that does occur in some substrate-limited sites (i.e., 2009) likely occurs in patches without larger substrate material that would limit spawning use. The reduced utilization rate of the 2009 site this year by both Chinook salmon and steelhead trout may indicate a reduction in these patches due to high flows mobilizing the usable smaller material and leaving larger material behind.

No significant relationships were observed in this analysis of the effects of substrate size on Chinook salmon redd characteristics. It is possible that sampling results for Chinook salmon were confounded by depth limitations and w2ill require additional data to complement and validate these findings. Similarly, tailspill length showed the only significant relationship for steelhead trout. However, a number of other characteristics showed trends that would likely be significant with greater sample size. When chi-squares were used to compare observed proportions to expected proportions, redd observations in larger substrates always contained a greater than expected proportion of smaller redd characteristics, whereas smaller substrates always contained a greater than expected proportion of larger redd characteristics. For steelhead trout, this suggests that larger substrate sizes do influence redd length, width, depth, and tailspill length and width. However, more data will be needed to test these hypotheses conclusively for both species.

### *Juvenile salmonid habitat preferences*

Canonical correspondence analysis and GLMMs revealed divergent patterns of habitat use between juvenile Chinook salmon and steelhead trout, and among different size classes of Chinook salmon. Habitat partitioning between salmonid species that overlap in space and time is well known (Harvey and Nakamoto 1996), and this analysis indicated that there is habitat to support both species within the Sailor Bar study area. However, certain restoration sites provided better habitat for Chinook salmon or steelhead trout as evidenced by the significant site effect in the generalized linear mixed models. Chinook salmon exhibited ontogenetic shifts in habitat affinity suggesting that a diversity of habitat types are required to support the variety of species and size classes of salmonids present in the LAR.

All steelhead trout size classes were more abundant in shallower, higher velocity habitats relative to fry and parr size Chinook salmon. One of the results of gravel augmentation is a reduction in depth and increase in velocity that could provide additional habitat for steelhead trout. However, this relationship should be viewed with caution because velocities that are too high can negatively impact steelhead trout juveniles. Although steelhead trout used higher velocity habitats relative to Chinook salmon, there was a significant negative relationship between total steelhead trout abundance and velocity in the GLMM. Total Chinook salmon abundance was negatively associated with Main Channel habitats in the GLMM and CAA suggesting a positive relationship between smaller size classes of Chinook salmon and floodplain habitats. Sommer et al. (2001) found that growth and survival of Chinook salmon was greater in floodplain relative to main channel habitats in the Sacramento River. The increase in floodplain habitat resulting from gravel augmentation on the LAR is likely to benefit Chinook salmon in these size classes provided that the frequency and duration of flow pulses is long enough for fry and parr to exploit the additional ephemeral habitat. Chinook salmon smolts (≥ 80mm) were found in deeper, higher velocity habitats relative to smaller Chinook salmon and steelhead trout.

Substrate size did not show a significant relationship with total steelhead trout or Chinook salmon abundance. However, small size classes of Chinook salmon were associated with larger substrates than juvenile steelhead trout in the CCA. The lack of strong relationships between abundance and substrate size suggests that gravel placement affects juvenile salmonids indirectly through changes in depth and velocity rather than directly through changes in substrate size. However, depths and velocities in large portions of the main channel were unsuitable for juveniles during most of the rearing period.

Although there was no significant relationship between Large or Small Woody Material and steelhead trout abundance, Chinook salmon abundance showed a negative relationship with Large Woody Material. This result was surprising because many studies have reported positive associations with these habitat features (Whiteway et al. 2010; Roni and Quinn 2001). A partial explanation for this relationship is that there is very little woody material of either size class in the restoration area (availability of both types of woody material combined was ~1.1% of the total surveyed area for all transects). Out of 642 total fish observations, Large and Small Woody Material were observed only 10 (1.6%) and 18 times (2.8%), respectively. Although these utilization rates were low, these habitats were utilized at a slightly higher rate than their availability. As a result, additional observations are required to thoroughly evaluate juvenile habitat use and needs in the restoration area.

Although there was variability in the proportion of juvenile Chinook salmon and steelhead trout utilizing the four restoration sites, there was clearly habitat available for both species within the study area. Palm et al. (2007) found that juvenile brown trout density increased more through rehabilitation of spawning habitat than by increasing juvenile habitat, and concluded that their study stream was limited by spawning habitat. A similar situation may exist on the LAR where spawning gravels have been lost, yet juvenile habitat remains. However, these habitats have not been fully evaluated for their ability to meet restoration goals for both species.

### *Benthic macroinvertebrates*

Similar to other studies (e.g., Wise and Molles 1979; Boulton et al. 1988; Quinn et al. 1998; McCabe and Gotelli 2000; Zuellig et al. 2002; Merz and Chan 2005; Watry and Merz 2009), these results indicate that benthic macroinvertebrates rapidly colonize newly inundated floodplain habitats on the LAR. In general, overall benthic macroinvertebrate density in LAR floodplain habitats surpassed density in main channel habitats at 5.6 weeks (39 days) inundation. Floodplain densities peaked and were significantly greater than density in main channel habitats at 8.9 weeks (62 days) inundation. These floodplain densities then declined but remained greater than main channel densities at 10.0 weeks (70 days) inundation. In slight contrast, 2010-2011 results suggest that overall benthic macroinvertebrate biomass in floodplain habitats gradually increased until it was comparable to but did not surpass biomass in main channel habitats at 8.9 and 10.0 weeks (62 and 70 days) inundation. A limited sampling period (~1–10 weeks) prevented any further analysis of colonization trends beyond 10.0 weeks (70 days) inundation. The observed trend in overall density was consistent with the colonization of newly placed main channel spawning gravel in the lower Mokelumne and American rivers (Merz and Chan 2005; Watry and Merz 2009). However, peak estimates of density in this study occurred ~2–3 weeks (7–14 days) later than previously reported suggesting some factor or combination of factors delayed floodplain colonization when compared to main channel habitats. The observed trend in overall biomass was consistent with the results of Merz and Chan (2005), where peak biomass occurred at 12 weeks (84 days) inundation, but differed from the results of Watry and Merz (2009), where biomass peaked at seven weeks (49 days) inundation and subsequently declined. Estimates of peak overall benthic macroinvertebrate density and biomass reported by Merz and Chan (2005) and Watry and Merz (2009) were ~20–370-fold greater than estimates for control and treatment sites in this study, which suggests that intra- and inter-annual or inter-system variation in productivity may drive differences in both overall benthic macroinvertebrate density and biomass.

With the exception of Hydropsychidae density, we found that Baetidae, Chironomidae, and total BCH density and biomass in newly inundated floodplain habitats on the LAR were lowest at 1.3 weeks (9 days) inundation, peaked at 8.9 weeks (62 days) inundation, and then fell but remained relatively high at 10.0 weeks (70 days) inundation. Baetidae, Chironomidae, and total BCH density and biomass in treatment samples taken at 5.6, 8.9, and 10.0 weeks (39, 62, and 70 days) inundation were generally similar to density and biomass in control samples, but significantly greater than density and biomass in treatment samples taken at 1.3 and 3.3 weeks (9 and 23 days) inundation. Both Baetidae density and biomass in treatment samples taken at 8.9 weeks (62 days) inundation surpassed density and biomass in control samples. However, differences were not significant. In contrast, both Chironomidae density and biomass in treatment samples taken at 5.6, 8.9, and 10.0 weeks (39, 62, and 70 days) inundation surpassed density and biomass in control samples. However, significant differences were only found for treatment samples taken at 8.9 weeks (62 days) inundation. Hydropsychidae density and biomass in treatment samples were not significantly different among inundation durations. However, both density and biomass in control samples were significantly greater than density and biomass in treatment samples taken at all inundation durations. Trends in total BCH density were identical to and driven by trends in Chironomidae density, whereas trends in total BCH biomass were similar to trends in Baetidae biomass, but did not surpass control biomass at any time during the sampling period (~1–10 weeks).

Overall trends in family-specific (i.e., Baetidae, Chironomidae, and Hydropsychidae) density were consistent with those reported for the colonization of newly placed main channel spawning gravel in the LAR (Watry and Merz 2009). However, similar to overall benthic macroinvertebrate density, peak estimates of density in this study occurred ~2 weeks (14 days) later than previously reported for two of three families (Chironomidae and Hydropsychidae), which suggests some factor or combination of factors delaying BCH colonization of floodplain habitats when compared to main channel habitats. Data were not available to compare overall trends in family-specific biomass. In general, significant differences in both BCH density and biomass among treatment groups suggests that inundation durations of ~1–4 weeks (7–28 days) would provide relatively little BCH production to benefit feeding juvenile salmonids. However, inundation durations of ~5–10 weeks (35–70 days) would provide substantial BCH production. Consistent declines in both BCH density and biomass after 8.9 weeks (62 days) inundation suggest that floodplain habitats on the LAR may provide an initial influx of BCH production to benefit feeding juvenile salmonids. However, these benefits likely decline beyond ~8–9 weeks (62–69 days) inundation. Therefore, the greatest benefit to feeding juvenile salmonids is likely to come from floodplain inundation durations of ~8–9 weeks (62–69 days), with fishery managers targeting inundation durations of ~5–10 weeks (35–70 days) when peak juvenile salmonid densities are expected in the system. Based on previously reported LAR RST data (Snider and Titus 1995; Snider and Titus 2002) and required habitat calculations which can be used as a rough estimate of juvenile salmonid food requirements (see above), fishery managers should begin floodplain inundation for benthic macroinvertebrate production ~5 weeks (35 days) prior to February (the peak of fry presence) and maintain floodplain inundation through the end of March. This may shift if production and life stage development shift significantly from future management scenarios.

Results from 2010-2011consistently suggest both Control-Treatment and Upstream-Downstream effects on overall benthic macroinvertebrate density and biomass and multiple indices of BCH density and biomass. Overall benthic macroinvertebrate community structure was not specifically analyzed, so generalizations about early-late colonizers and relative size-proportional density cannot be made. However, BCH community structure was analyzed and can provide some insight into why Control-Treatment and Upstream-Downstream effects on density and biomass were observed. In general, control samples contained a greater proportion of larger late colonizers (i.e., Baetidae and Hydropsychidae – 33% BCH density), whereas treatment samples contained a greater proportion of smaller early colonizers (i.e., Chironomidae – 94% peak BCH density). Within treatment samples, upstream samples generally contained a greater proportion of smaller early colonizers (i.e., Chironomidae – 98% peak BCH density), whereas downstream samples contained a greater proportion of larger late colonizers (i.e., Baetidae and Hydropsychidae – 83% peak BCH density). The Upstream-Downstream effect on density and biomass was not observed for control samples. Based on proportional BCH density and biomass, 2010-2011 results suggest that Control-Treatment and Upstream-Downstream effects on density and biomass are likely related to the relative succession of the LAR benthic macroinvertebrate community, with control samples containing a more "even" stable-state community (see Merz and Chan 2005) when compared to treatment samples, and downstream treatment samples approaching a more "even" stable-state community at a faster rate when compared to upstream samples. These findings contrast the findings of Merz and Chan (2005) and Watry and Merz (2009), where no significant differences in "evenness" were found between newly placed main channel spawning gravel and control sites in the lower Mokelumne and American rivers after two and seven weeks (14 and 49 days), respectively. Factors which may account for differences between this study and the previous studies include: (1) reduced macroinvertebrate drift in habitats outside of the main channel, thereby resulting in a limited number of individuals available for floodplain colonization (Waters 1972); (2) limited connectivity of floodplain habitats compared to main channel habitats, thereby resulting in limited ability of macroinvertebrates to colonize newly inundated areas (Paillex et al. 2009); and (3) intra- and inter-annual variation in macroinvertebrate life stage timing, thereby resulting in variable colonization ability (Elliot 1967a; Elliot 1967b). Similarly, factors which may account for differences between upstream and downstream treatment samples in this study include: (1) proximity to an artificial dam (i.e., Nimbus Dam), which may alter the macroinvertebrate community immediately downstream or limit the availability of drifting macroinvertebrates (Stevens et al. 1997), thereby altering colonization ability and community composition at upstream floodplain sites; and (2) limited connectivity of upstream floodplain sites compared to downstream floodplain sites (i.e., upstream sites represent a more classic floodplain disconnected from the main channel whereas downstream sites represent a more connected floodplain closer to the main channel), which may provide a greater opportunity for colonization at downstream sites, thereby increasing colonization rates of less flight-dependent macroinvertebrates (i.e., Baetidae and Hydropsychidae – 3.6% and 23.1% aerial colonization Ephemeroptera and Trichoptera, respectively) when compared to more flight-dependent marcroinvertebrates (i.e., Chironomidae – 34.8% aerial colonization rate; Williams and Hynes 1976). Alternatively, Control-Treatment and Upstream-Downstream effects may be related to systematic differences in substrate size, depth, or velocity among sample sites, whereby control and downstream sample sites contain substrate sizes, depths, and velocities more preferred by Baetidae and Hydropsychidae when compared to Chironomidae (see Watry and Merz 2009). The relatively high proportional Chironomidae density and biomass in treatment (i.e., floodplain) samples in this study are consistent with the results of Sommer et al. (2001), where Diptera (primarily Chironomidae) densities were consistently an order of magnitude higher in floodplain drift samples collected from Yolo Bypass when compared to samples collected from the Sacramento River. Similarities between this study and Sommer et al. (2001) suggest that enhanced proportional Chironomidae density may be characteristic of most Central Valley floodplains regardless of floodplain type or location (i.e., Yolo Bypass represents a low-elevation floodplain with flooded vegetation and fine substrates, whereas this study sampled a relatively higher elevation floodplain with limited flooded vegetation and course substrates). This hypothesis should be evaluated more extensively in future studies.

## CONCLUSIONS AND RECOMMENDATIONS

Continued monitoring of the three LAR augmentation sites demonstrated variable success for enhancing Chinook salmon and steelhead trout spawning habitat. While the 2008 site continued to support significantly high spawning use by both species, the 2009 site only supported moderate use by Chinook salmon after its second season in existence. The 2010 site demonstrated limited use by both species. Distance from a terminal hatchery and changes in project design, including depth, velocity, and substrate size parameters offer possible reasons for relatively low site use with distance downstream. Considering the limited LAR salmonid returns in 2010–2011, simple population dynamics may have confounded these results. High flows and the low number of aerial Chinook salmon redd surveys may have also confounded the ability to adequately monitor spawning. Regardless, the augmentation sites accounted for 47% and 51% of total LAR Chinook salmon and steelhead trout redds, respectively, which suggests that gravel augmentation has a role in continued long-term restoration success. As Central Valley salmonid populations rebound, continued monitoring will inform evaluations of design effectiveness and construction implementation to provide stronger indicators of what aspects of spawning gravel enhancement do and do not work.

Some important caveats should be stressed for this year's report: (1) these analyses of salmonid spawning and rearing preferences are based on one season of data (i.e., limited and needs data from multiple years); and (2) data for spawning Chinook salmon are only from the 2008–2010 augmentation sites, whereas data for steelhead trout are from the entire LAR. Therefore, data for steelhead trout likely represent a good range of habitats sampled, whereas data for Chinook salmon do not. This may explain why the steelhead trout depth curve is relatively wide compared to the Chinook salmon depth curve, and why velocity curves are similar in width. As we continue to collect information on spawning and rearing preferences (e.g., depth, velocity, and substrate size), we will continue to populate the data sources used for future restoration site models. Therefore, these results should be interpreted with caution until more detailed and expansive habitat preference data are available for the LAR.

Preliminary results on substrate size associated with redd characteristics suggests some very interesting and important trends. Specifically, the results imply that steelhead trout redd characteristics may be more sensitive to substrate size than Chinook salmon. Chinook salmon showed no strong trends whereas steelhead trout showed a consistent trend for larger redd measurements in smaller substrates. To better clarify the effects of substrate size on spawning Chinook salmon and steelhead trout, more observations in a greater variety of substrate categories could be very useful over the upcoming monitoring season. Future analyses could also benefit from categorization of data based on 25% substrate size increments instead of a single break point at the mean. This would allow comparison of the bottom 25% of the data to the top 25% of the data and eliminate data in the middle (i.e., where redd characteristics from larger and smaller substrates would potentially run together, thereby confounding the results).

We observed a significant problem with the ability to evaluate complexity (e.g., cover), especially as it pertained to spawning preferences by both species. For instance, placed gravel contained most spawning activity. However, placed gravel was generally not associated with cover features. Regardless, we observed a relatively high percentage of redds associated with cover. While we caution activities implemented simply for the sake of hypothesis testing, the continued incorporation of structural complexity into site enhancement will allow us to better understand its benefit (or lack there of) in future LAR restoration work.

Future directions in habitat restoration assessment include evaluating how changes in spawning habitat suitability change the value (e.g., quantity and quality) of additional inundated areas; can gravel augmentation projects also be designed to improve or enhance rearing habitat? Specifically, can these additional areas be designed for optimum HSI values to support the greatest number of juveniles possible? Such evaluations should include the quantification of juvenile production at gravel placement sites if those sites were fully seeded with adults. If rearing habitat is a limiting factor in LAR production, future gravel augmentation may be required to not only focus on the quantity and quality of spawning habitat, but also the potential to support juveniles produced from those and other restoration sites to ensur enough rearing habitat to handle increased juvenile production and adult spawning success. It is important to note that with limited water supplies, increased floodplain inundation has the potential to alter temperature during periods of juvenile rearing and this effect would need to be more fully evaluated in subsequent studies.

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