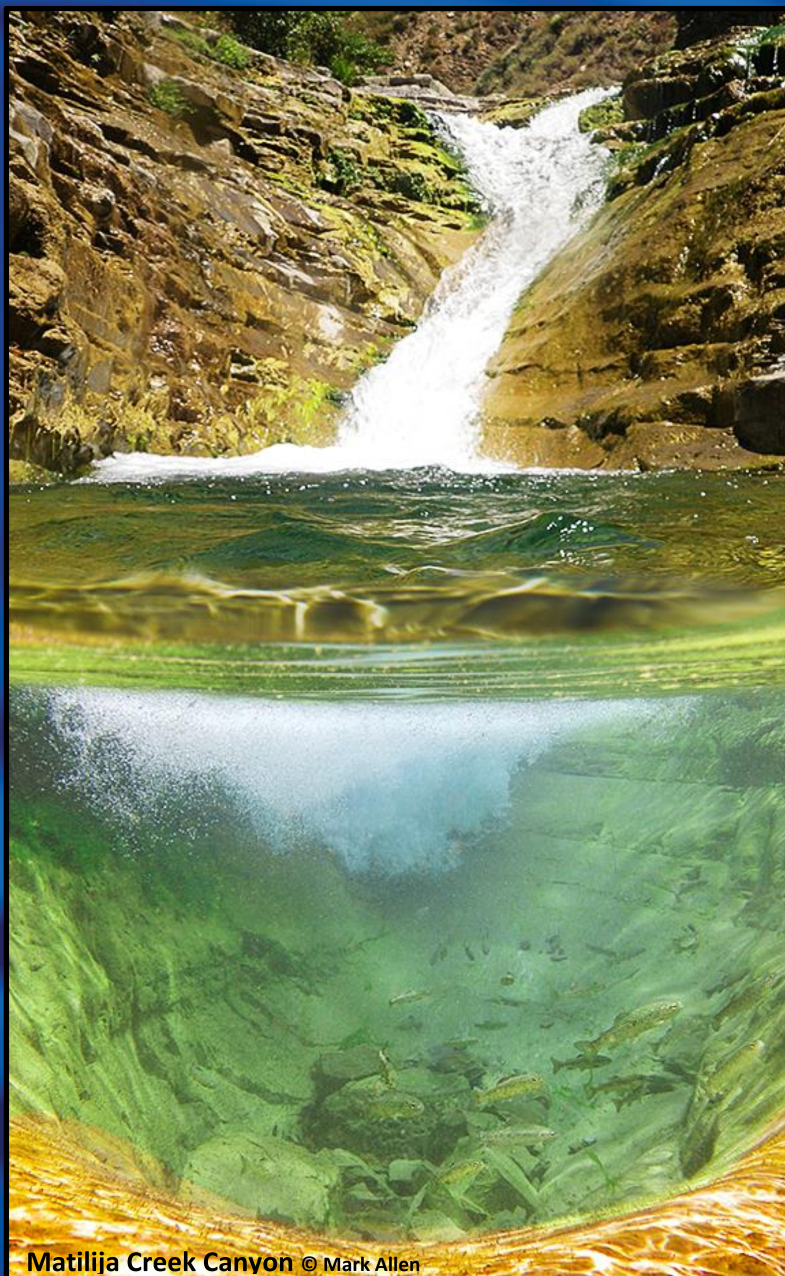
Steelhead Population and Habitat Assessment in the Ventura River / Matilija Creek Basin 2006-2012

FINAL REPORT



Matilija Creek Canyon © Mark Allen

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Submitted On:

March 31, 2015

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1.0 Executive Summary

Stratified random sampling was conducted in 4 to 14 study sites encompassing 46 to 308 sampling units distributed throughout the Ventura River Basin over a seven year period (2006-2012). Abundance of *O. mykiss* was estimated using dive counts and electrofishing under the Method of Bounded Counts protocols in pools, flatwaters, and riffles in most years to produce overall abundance estimates at study site and basin segment spatial scales according to fish size class (fry at <10cm and juvenile+ at $\geq$10cm). Abundance estimates displayed significant spatial and temporal variation, with consistently highest abundance and densities (#/100 ft^2) in the upper segment above Matilija Dam (resident rainbow trout only) and in the middle segment between Robles Diversion Dam and Matilija Dam (mixture of resident and anadromous *O. mykiss*). Maximum estimated densities of 3-7 *O. mykiss* fry/100 ft^2 and 1-2 juvenile+/100 ft^2, were routinely observed in the upper North Fork and lower North Fork Matilija Creek study sites, with zero or near zero densities in the lowermost Ventura River study sites. Densities of fry were consistently highest in riffle habitats and lowest in pool habitats (by factors of 2-5x), whereas juvenile+ *O. mykiss* were more evenly distributed among habitat types.

Annual variation was also substantial, with positive (but statistically non-significant) trends in abundance of *O. mykiss* in all three segments. Maximum abundance occurred in 2012, with 2,137 captured or observed *O. mykiss* producing a total estimated abundance of 24,134 fish in the Ventura River Basin (excluding San Antonio Creek). Total abundance was less than 15,000 fish in most other years, with a minimum estimate of 12,271 fish in 2007. The high annual variability in abundance of *O. mykiss* fry was reflected in C.V.'s exceeding 170% in the lower Ventura study sites and in the lower Matilija Creek study site, with C.V.'s over 100% for juvenile+ fish in mainstem Ventura River and San Antonio Creek study sites. In comparison, variation in annual abundance in most tributary and headwater study sites was less with C.V.s for both size classes typically between 30% and 70%. Further assessment of the annual abundance data suggested that a minimum of 7-10 years would be necessary to detect an annual decrease in abundance of 10% per year in the headwater and tributary study sites, whereas 15-20 years of sampling may be required to detect a comparable decline in the lower mainstem reaches. Longer time series would be required to detect declines in abundance using a pool only or a representative reach sampling design (compared to the habitat stratified design used here), or to detect a 10% annual increase in abundance.

Habitat data was collected in 2006, 2007, 2011, and 2012 to evaluate the relationship between study site Habitat Suitability Index (HSI) scores produced by an existing USFWS HSI model (Raleigh et al. 1984) and observed densities of *O. mykiss*. Linear regression showed statistically significant relationships, with best fit for the non-compensatory model option and worst fit using the equal components option. Despite the positive relationship, poor separation between study sites supporting low densities of *O. mykiss* with sites that were consistently absent of *O. mykiss* along with a relatively narrow range of calculated HSI scores led to the development of an alternative habitat model, termed the Southern Steelhead HSI model (or, SS HSI). New habitat variables and new model formulations produced a model with generally better fit and a wider range in calculated HSI scores. Although overall fit was improved and the model appeared appropriate to the Ventura River Basin, the model has not been validated elsewhere and several of the HSI variables were highly qualitative in nature and should be assessed with actual data prior to application in future studies.

2.0 Introduction

The Ventura River Basin is a large southern California watershed that historically provided abundant habitat for the now endangered southern steelhead (*Oncorhynchus mykiss*) (Moore 1980). Ocean migrant steelhead are reported to have utilized the mainstem Ventura River, as well as the principal subbasins including the Coyote Creek Basin, the San Antonio Creek Basin, the lower North Fork Matilija Creek Basin, and the upper Matilija Creek Basin (NMFS 2007). The amount of habitat available to anadromous steelhead for spawning and rearing declined over time following the construction of water supply facilities, such as Matilija Dam in 1947 (blocking access to the upper Matilija Basin), Casitas Dam in 1957 (blocking access to the Coyote Creek Basin), and Robles Diversion Dam in 1958, which until recently blocked access to the upper portion of the Ventura River and the lower North Fork Matilija Creek. In 2004, a new fish passage facility was constructed in Robles Diversion Dam, which gives access to several miles of important spawning and rearing habitat (TRPA 2004), and sets the stage for the restoration of upper Matilija Creek. Matilija Dam was constructed for the purpose of supplying water storage and flood control, but reservoir sedimentation and construction of newer projects has reduced the necessity of the dam, and efforts are currently underway to restore access to the upper Matilija Basin through removal of Matilija Dam (NMFS 2007).

Apparent declines in steelhead populations throughout southern California waters led to the federal listing of steelhead as “endangered” in 1997 for the Southern California Steelhead ESU (Federal Register 1997). The California Department of Fish & Wildlife (CDFW) identified the Ventura River basin as a high-priority watershed having important ecological effects on the health of the Southern California Steelhead ESU. Consequently, this study was largely funded by CDFW through the California Steelhead Restoration Grant Program, through the sponsorship of Surfrider Foundation, with supplemental funding from the Ventura County Watershed Protection District and Patagonia, Inc., with the following principal goals:

- to assess the annual abundance and spatial distribution of *O. mykiss* (both anadromous and resident forms) in the Ventura River Basin; and
- to test the validity of Habitat Suitability Index (HSI) models to assess habitat quality for southern California *O. mykiss*.

The published HSI model for rainbow trout / steelhead consists of five components with 18 variables (Raleigh et al. 1984). The HSI model was chosen to assess habitat quality because the model utilizes a wide range of habitat variables that are summarized into a single quantitative value (the HSI score), which can be easily and consistently compared among streams. The rainbow trout / steelhead HSI model incorporates several variables that are particularly important to *O. mykiss* populations in the southern portion of their range, such as water temperature, pool habitat characteristics, and riparian coverage. An alternative model (the Southern Steelhead HSI) was subsequently developed using site-specific habitat data in 2012 to encompass additional variables thought to be important to steelhead in southern California Basins.

Although this report summarizes data from all 7 study years, additional details can be found in previous reports (TRPA 2003, 2004, 2007, 2008, 2009, 2010, Normandeau 2011, 2012), most of which are available online at www.matilijadam.org.

3.0 Study Area

The Ventura Basin drains a watershed of approximately 228 mi^2, of which about 25% is above Matilija Dam and not currently accessible to anadromous steelhead (Figure 1). Below Matilija Dam, steelhead have access to the entire 16 miles of mainstem Ventura River, except during the summer and fall months of most years when six miles of channel below Robles Diversion Dam goes dry. At the bottom of the dry reach immediately upstream of San Antonio Creek, upwelling groundwater produces a consistent source of cooler water that provides over summering rearing habitat in the mainstem Ventura River for fry, juvenile and adult (resident) *O. mykiss*. In most years, *O. mykiss* are found in the mainstem downstream to the confluence of Coyote Creek (Normandeau 2012), but the fish community in the lower six miles of mainstem to the terminal lagoon is dominated by arroyo chub (*Gila orcutti*) and stickleback (*Gasterosteus aculeatus*), both native species, and common carp (*Cyprinus carpio*), an exotic species (TRPA 2007, 2008, Normandeau 2011).

A large homeless community inhabits the lowermost three miles of the Ventura River floodplain, which results in significant impacts to water quality and to aquatic and riparian habitats. Streamflows in the lower mainstem are augmented by a release of about two cfs of tertiary treated wastewater from a treatment facility 5½ miles above the lagoon. The Ventura River Lagoon is open to the Pacific Ocean following winter and spring storm events, but may be closed off by a sand berm during the summer and fall months. During drought years, the lagoon may remain closed (or closed at low tide) throughout most of the year (CMWD 2013). The lagoon has been highly altered by the crossing of highway and railway bridges, a rip-rap armored bank along the east shoreline, and inputs of poor water quality from upstream sources.

San Antonio Creek is the only known tributary to the lower mainstem Ventura River that supports significant spawning and rearing habitat for steelhead, although in some years much of the seven mile anadromous reach become intermittent during summer and fall months (Figure 1). Coyote Creek is dammed to form Lake Casitas 2½ miles upstream of the Ventura River confluence, but the lower channel is largely unsuitable for spawning or rearing of *O. mykiss* (Capelli 1997). A second major spawning and rearing tributary for steelhead is the lower North Fork Matilija Creek, which merges with the mainstem Matilija Creek to form the Ventura River one-half mile below Matilija Dam. The Lower North Fork contains about four miles of habitat accessible to adult migrant steelhead, up to a currently impassable barrier located at a road crossing within the U.S. Forest Service's Wheeler Gorge Campground (TRPA 2003). In addition to steelhead, resident rainbow trout have been observed spawning throughout the lower North Fork Matilija Creek (TRPA 2003).

Hot springs and mineral seeps are relatively common in the lower North Fork and in the mainstem near Matilija Dam (both areas historically supported commercial mineral bath resorts), and during periods of extended low flows (e.g., dry water years) the instream substrate becomes encrusted with tufa mineral deposits that can cement gravels and reduce spawning habitat (Minear 2003). The tufa encrustation also appears to exert negative effects on invertebrate prey abundance.

Above Matilija Dam, which blocks passage by steelhead, Matilija Creek exhibits alternating reaches of perennial and intermittent (in summer and fall) flows for 6½ miles, at which point the mainstem enters a narrow canyon where surface flows are persistent for another two miles to a series of high (15-50 ft) waterfalls (Figure 1, cover image). Resident *O. mykiss* are common in the upper canyon below the 15 ft falls, and in the lower mainstem in the vicinity of Murietta Creek. According to reports large trout, as well as largemouth bass (*Micropterus salmoides*) and sunfish (*Lepomis spp.*), inhabit Matilija Reservoir and all three species are found in the lower reaches of mainstem Matilija Creek when surface flows are present.

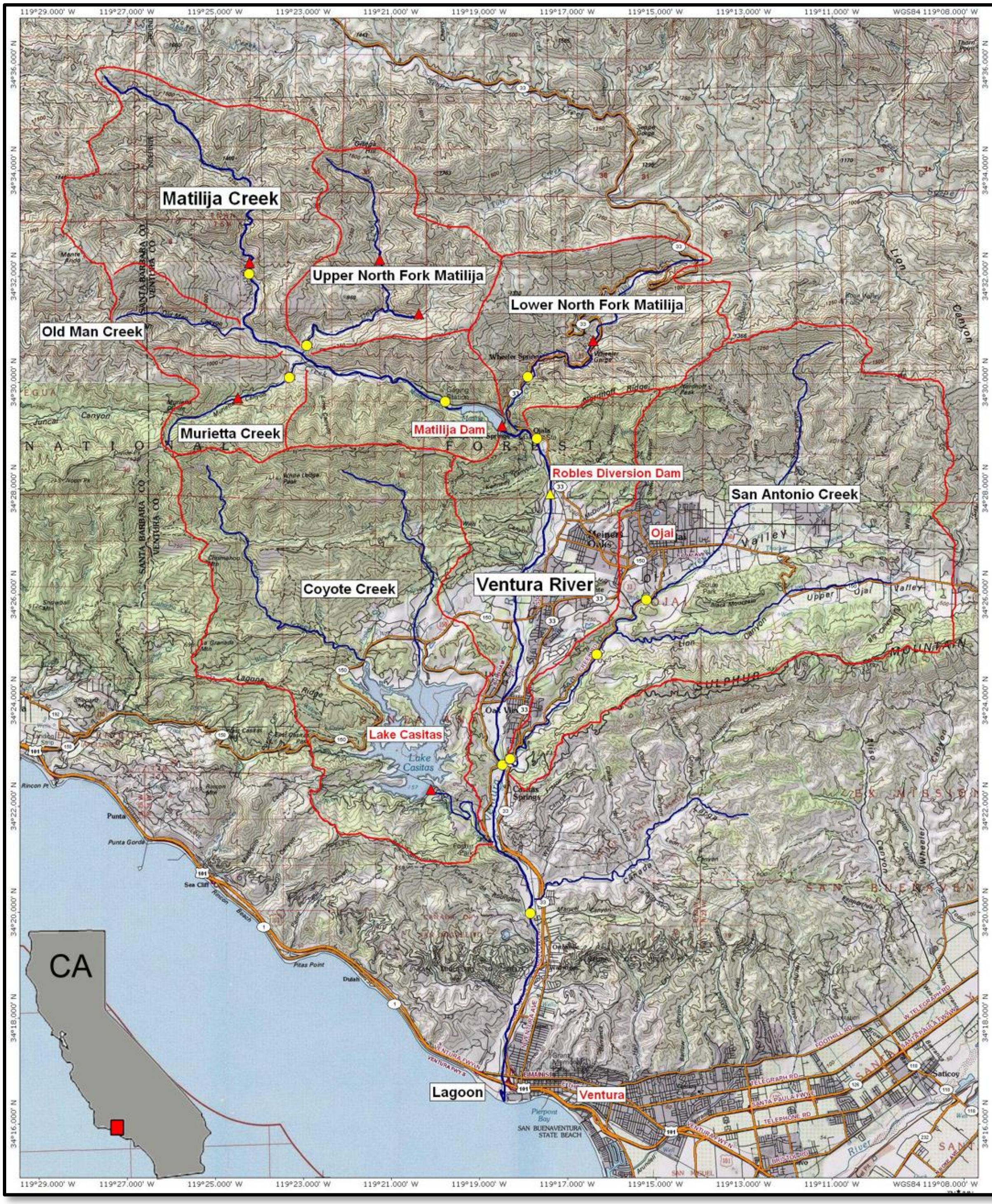


Figure 1. Watershed map showing sub-basins, barriers to upstream migration (red triangles), water temperature datalogger locations (yellow circles), and landscape features.

The three principal tributaries to the Matilija Creek study area are Murietta Creek, the upper North Fork Matilija Creek, and Old Man Creek (Figure 1). Old Man Creek was described as having marginal habitat for *O. mykiss* (TRPA 2003), however Murietta Creek and the upper North Fork contain suitable habitat and both harbor stream-resident trout. The upper North Fork and associated tributary provides approximately five miles of potential steelhead habitat, given passage beyond Matilija Dam, however the upper North Fork and the mainstem Matilija Creek both exhibit significant tufa mineralization of substrate, particularly during dry years and in intermittent reaches. In contrast, Murietta Creek appeared relatively free of tufa deposits, but alternating reaches of flowing and dry channels and the presence of several boulder cascades likely restricts potential steelhead habitat to approximately 1½ miles. Additional details regarding the geology, hydrology, and land-use characteristics of the Ventura Basin are detailed in Cardno-Entrix (2012).

4.0 Methods

4.1 Study Design & Stratifications

The instream habitat and fish population characteristics are described on the basis of a hierarchical framework of design stratifications according to basin segment, stream reach, study site, and habitat type. The basic sampling units are individual habitat units of a given habitat type.

4.1.1 Basin Segments

The Ventura River Basin was partitioned into three "segments" based on accessibility to anadromous steelhead and the continuum of river channel characteristics. The lower segment, which has been historically accessible to steelhead (given adequate surface flows), extends upstream from the Ventura River Lagoon to Robles Diversion Dam at *approximate*[1] River Mile (RM) 14.6, and is mostly characterized as a low gradient, unconfined alluvial valley stream with significant anthropogenic influence (Figure 2). Although the lower segment contains several tributaries (Cañada Larga, Coyote Creek, and San Antonio Creek), only San Antonio Creek is known to contain significant spawning and rearing habitat for steelhead.

The middle segment (Figure 3) includes the remaining 1.5 miles of Ventura River above Robles Diversion Dam, one-half mile of Matilija Creek between the North Fork confluence (RM 16.3) and Matilija Dam (RM 16.9), and four miles of the Lower North Fork Matilija Creek. Access to this segment by steelhead was effectively blocked following construction of the diversion dam in 1958, but was restored after installation of a new fish ladder in 2004. Also, access up the lower North Fork has been intermittent due to persistent landslides at the Ojai Quarry near the Ventura River confluence, and also perhaps due to public construction of large swim dams immediately above the quarry (TRPA 2008). This middle segment has intermediate characteristics to the lower and upper segments, but most *O. mykiss* spawning habitat in this segment occurs in the lower North Fork Matilija Creek, which is similar to the mountainous and more pristine habitat in the upper segment.

The upper segment (Figure 3) is entirely above Matilija Dam, and displays a wide continuum of open, alluvial channels in the lowest reaches to high gradient, confined and densely vegetated channels in headwater reaches. The mainstem Matilija Creek above Matilija Reservoir extends just over eight miles to the first definite barrier to upstream migration (although resident trout are

[1] River Mile designations are approximate due to shifting channels in alluvial mainstem reaches

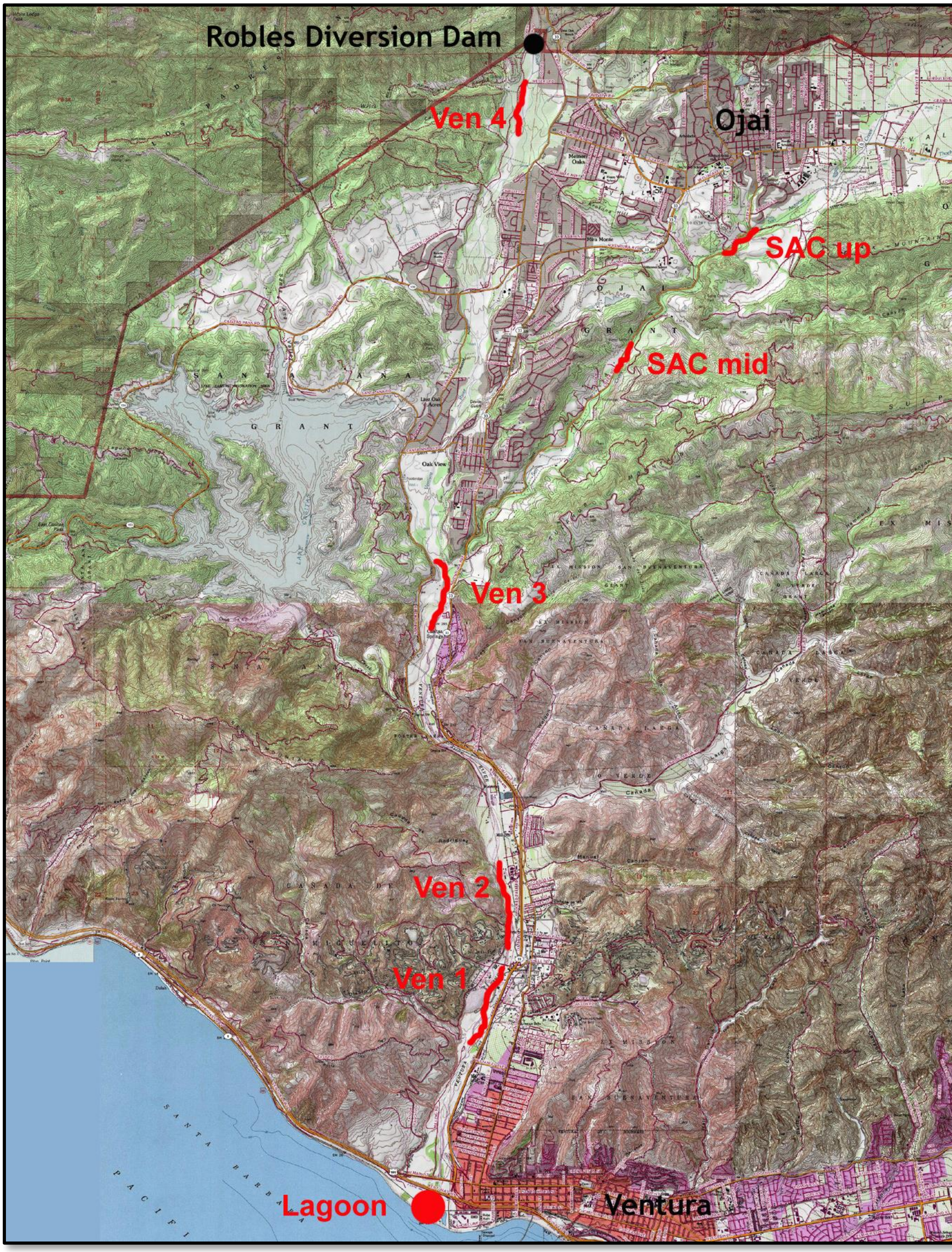


Figure 2. Map of lower segment showing study site locations and landscape features.

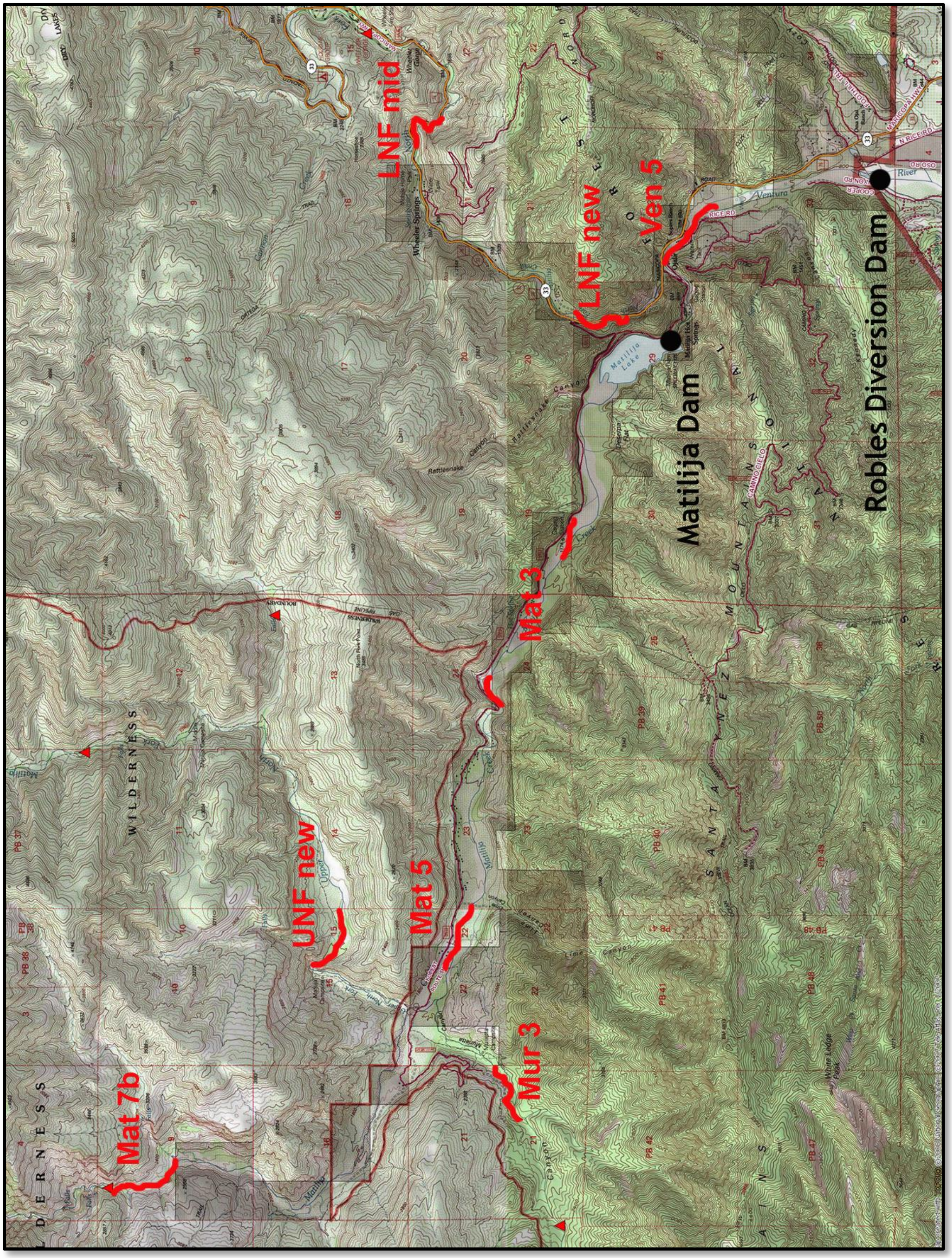


Figure 3. Map of middle and upper segments showing study site locations, barriers to upstream migration (red triangles and Matilija Dam), and landscape features.

present in low numbers above the first barrier). The upper North Fork Matilija Creek contains approximately five miles of habitat potentially accessible to steelhead given passage past Matilija Dam. Murietta Creek contains occasional intermittent sections below the first definite barrier two miles above its confluence, but *O. mykiss* are common in perennial stream reaches and, unlike many other reaches, Murietta Creek appears largely free of tufa mineral deposition.

4.1.2 Stream Reaches

The 2003 HSI study identified 31 reaches within the Ventura River Basin, encompassing the mainstem Ventura River and the lower North Fork Matilija Creek below Matilija Dam, as well as the mainstem Matilija Creek, the upper North Fork Matilija Creek, Murietta Creek, and Old Man Creek above Matilija Dam (TRPA 2003). Coyote Creek and San Antonio Creek, both major tributaries to the lower Ventura River, were not included in the original habitat surveys. Reach boundaries were based on a variety of factors, including location of impassable barriers, confluence of major tributaries, changes in channel type, presence of perennial flow, and/or changes in riparian vegetation.

Eleven of the 31 reaches were selected in 2003 for detailed habitat mapping using the HSI methodology (TRPA 2004): four in the lower segment, three in the middle segment, and four in the upper segment. These 11 reaches were selected based on several factors, including persistence of surface flows through the summer low flow period (one intermittent reach was selected), similarity to adjacent (unselected) reaches, and legal access. Two additional reaches were delineated in San Antonio Creek in 2008, making a total of 13 sampled reaches from 2008-2011. A final study site was selected in Murietta Creek in 2012, making a total of 14 sampled reaches for that year.

4.1.3 Study Sites

Each of the 14 selected reaches were subsequently divided into one mile or one-half mile sections, depending on channel size, in order to encompass a minimum of 40-50 habitat units within each section. Mainstem Ventura River reaches below San Antonio Creek were divided into one mile sections; all other reaches utilized one-half mile sections. A single section was selected within each reach to represent a study site for the collection of HSI data (in 2003) and fish abundance data (starting in 2006). Most of the study sites were selected randomly; however a few study sites were subjectively selected in order to account for distinctive habitat features or known presence of steelhead. For example, the upper San Antonio Creek study site was selected based on the location of known steelhead spawning activity (Mark Capelli, NMFS, personal communication). Two of the original 2003 study sites (Ven 2 and Ven 4, Table 1) were also selected subjectively based on the presence of large bedrock pools known to be historically important habitat for holding adult steelhead (Mark Capelli, NMFS, personal communication). Most of the selected study sites were sampled each year over the course of this seven-year study, however two of the study sites sampled in 2006 were replaced with new study sites in 2007 in order to test the 2006 HSI model with 2007 data (LNF low changed to LNF new, and UNF up to UNF new). Access into private property at Mat 7 was denied in 2010, therefore the next available section upstream (Mat 7b) was selected and sampled in 2011 and 2012. Physical characteristics of all sampled study sites and the years in which they were sampled are shown in Table 1; GPS coordinates for the 2012 study site boundaries are given in Appendix A.

Fish population sampling has not been conducted in other principal tributaries, such as Cañada Larga, Coyote Creek, or Old Man Creek. Consequently, the fish population estimates (and HSI scores) described in this report *do not* include potential fish abundance or habitat in any of those non-sampled tributaries, or in any mainstem or tributary reaches above impassable barriers (except for Matilija Dam) described in TRPA 2003 and shown in Figures 1 and 3.

Table 1. Characteristics of study sites sampled for O. mykiss abundance, 2006-2012.

Study	Study		Elevation ft msl			Average [1]		Sampling Date [2]						
Segment	Site	Length ft	btm	top	Gradient	Width ft	Flow cfs	2006	2007	2008	2009	2010	2011	2012
Lower	[3,4] Ven 1	5083	72	121	1.0%	26.8	20.3	7/15	7/17	9/3	X	7/7	6/22	6/4
	[4] Ven 2	5306	121	164	0.8%	30.3	18.4	7/12	7/19	9/6	X	7/10	6/23	6/8
	[4] Ven 3	4730	279	315	0.8%	32.1	13.8	7/19	7/22	9/4	8/11	7/15	6/28	6/10
	[4,5] SAC mid	2102	479	492	0.6%	16.4	3.4	X	8/11	9/5	X	8/21	7/20	7/7
	SAC up	2530	624	640	0.6%	11.3	3.0	X	X	X	X	8/6	7/19	7/10
	[4] Ven 4	2889	670	719	1.7%	39.7	0.5	7/10	X	X	X	7/13	6/21	X
Middle	[4] Ven 5	2717	860	915	2.0%	27.0	11.0	7/24	6/26	9/8	8/12	7/31	7/16	6/20
	[6] LNF low	2047	1353	1385	1.6%	14.2	4.6	8/21	X	X	X	X	X	X
	LNF new	2160	1085	1155	3.2%	15.0	1.7	X	7/25	9/10	8/14	8/3	7/14	7/10
	LNF mid	2181	1527	1614	4.0%	10.7	1.6	8/21	7/25	9/10	8/14	8/3	7/14	7/10
Upper	[4] Mat 3	2635	1142	1377	3.3%	28.6	7.3	8/12	7/31	9/8	X	8/20	8/2	6/22
	[4] Mat 5	2313	1505	1542	1.6%	21.2	4.9	8/8	7/28	9/9	X	8/20	8/2	6/24
	[7] Mat 7	2327	2023	2140	5.0%	16.6	3.4	8/16	8/7	9/10	X	X	X	X
	Mat 7b	2951	2198	2349	5.1%	13.1	2.8	X	X	X	X	X	8/4	7/12
	[6] UNF up	1741	2156	2268	7.2%	10.2	3.1	8/18	X	X	X	X	X	X
	UNF new	2665	1744	1845	3.8%	8.9	1.3	X	8/1	9/11	X	8/21	8/3	7/12
	Mur 3	2352	1751	1917	7.1%	9.9	0.5	X	X	X	X	X	X	6/27

[1] widths and flows averaged over all years of sampling
[2] date sampling was initiated in study site, "X" indicates no sampling that year
[3] Ven 1 was moved upstream in 2010 & again in 2011 due to encroachment of homeless camps
[4] limited pool sampling also occurred in these study sites in late-April 2010 and/or mid-May 2011 (see text for details)
[5] SAC mid was qualitatively 'spot shocked" in 2007, quantitative sampling began in 2008
[6] replaced with "new" site in 2007
[7] replaced with Mat 7b in 2011

4.1.4 Habitat Types

Each study site was mapped into mesohabitat types in 2003, 2006, and 2011 using the CDFG *Level III* classification of 19 individual types (Table 2), excluding subchannel units (Flosi et al. 1998). Mapping was repeated in 2006 and 2011 due to high winter or spring flows that resulted in significant changes to channel characteristics and habitat units. See Appendix B for 2011 mapping data. Prior to selection of sampling units for collecting fish abundance and habitat measurements, the mesohabitat units were pooled into the three *Level II* mesohabitat types: pools (*PL*), flatwaters (*FW*), and riffles (*RF*). Habitat characteristics of the *Level II* mesohabitat types, as used in this study, are:

> *Pools (PL)*. Deeper reaches with pronounced areas of bottom scour, dominated by slow velocities, smooth surface, and substrates including fines.
>
> *Flatwaters (FW)*. Moderately to swiftly flowing reaches of uniform depth (glides) or with a shallow thalweg (runs), with low (glides) to moderate (runs) turbulence, and substrate ranging from fines and gravel (glides) to cobble-boulder substrates (runs).
>
> *Riffles (RF)*. Shallow reaches of swift, turbulent water with gravel, cobble, boulder, or bedrock substrates. Cobbles and boulders often emergent during the low flow period.

Cascades and other habitat types that are not suitable for sampling by dive counts or electrofishing were placed into a fourth *Level II* habitat category termed "non-response" (*NS*) habitat. These habitat units were excluded from selection, and consequently abundance estimates for each study site or study segment do not include fish that may occupy such habitat types. The relative frequency of non-response habitat types is discussed in following sections of this report. See Flosi et al. (1998) for detailed descriptions of the *Level III* sub-types listed in Table 2.

Table 2. Level II and Level III habitat types, from Flosi et al. (1998).

Level II	Level III	Habitat Types
Pools	TRP	trench pool
(PL)	MCP	mid-channel pool
	CCP	channel confluence pool
	STP	step pool
	CRP	corner pool
	LSL	lateral scour pool - log enhanced
	LSR	lateral scour pool - root w ad enhanced
	LSBk	lateral scour pool - bedrock formed
	LSBo	lateral scour pool - boulder formed
	PLP	plunge pool
	DPL	dammed pool
Flatw aters	POW	pocketw ater
(FW)	GLD	glide
	RUN	run
	SRN	step run
Riffles	LGR	low gradient riffle
(RF)	HGR	high gradient riffle
	CAS	cascade
	BRS	bedrock sheet

Within each study site a sample of eight (occasionally seven or nine) individual mesohabitat units were randomly selected from each Level II habitat type (pools, flatwaters, and riffles) for fish sampling and HSI measurements, giving a total of 24 sampling units in most study sites. Because the habitat mapping was intended to select units for fish sampling in addition to habitat measurement, all flatwater and riffle habitats less than 20 ft in length were combined with the adjacent unit of most similar type, in order to prevent selection of extremely short units for fish sampling and to minimize the displacement of fish out of sample units while diving or setting block nets (Peterson et al. 2005). Likewise, in order to prevent selection of very long habitat units that would require lengthy electrofishing passes, all flatwater and riffle habitats longer than 100-150 ft (which mostly occurred in the lower mainstem Ventura River) were partitioned into multiple sampling units using natural feature breaks as unit boundaries. Pools typically do not contain natural feature breaks and were always sampled by the more rapid diving protocol; consequently pools were sampled in their entirety even if less than 20 ft or greater than 150 ft in length. After selection of sampling units, unit boundaries were delineated by handheld GPS units and marked with labeled flags prior to fish sampling or measurement of habitat features.

4.1.5 Ventura Lagoon

One-day qualitative sampling was conducted in the Ventura Lagoon in 2006, 2007, and 2011. These supplemental surveys were intended to yield "snapshot" information on *O. mykiss* presence in the lagoon during summer months, but were not sufficient to assess the extent or importance of lagoon rearing for juvenile steelhead.

4.2 The USFWS HSI Model

The U.S. Fish & Wildlife Services' HSI model for rainbow trout / steelhead (Raleigh et al. 1984) consists of five components with 18 variables (Figure 4). The five components address four life stages (adult, juvenile, fry, and embryo), with an "other" component that includes additional variables not specific to a single life stage. The previous HSI assessments in the Ventura River (TRPA 2004, 2007, 2008) used the "equal-components" option to calculate HSI scores, which assumed that each of the five components exerts equal influence in determining the overall HSI score. This report assesses the equal-components design as well as other model configurations, including unequal component methods with and without limiting variable restrictions (explained below). Each HSI variable is expressed in the form of a curve or (for categorical variables) stepped functions, where "optimal" conditions are given a suitability value of 1.0, "unsuitable" conditions are rated as 0.0, and "usable" conditions have intermediate suitability values. Habitat variables measured or estimated in each study site were compared to the corresponding HSI curve to determine that variable's score for that study site. Overall HSI values were calculated for each study site using the steelhead model in anadromous reaches, or the resident trout model for reaches above Matilija Dam. The steelhead model differs from the resident trout model in the addition of three variables associated with migration of steelhead adults or smolts. Note that the original Ventura Basin HSI model developed in 2003 (TRPA 2004) used the steelhead model for all reaches, including those above Matilija Dam, in order to assess the potential benefits of dam removal for restoring steelhead to the upper watershed.

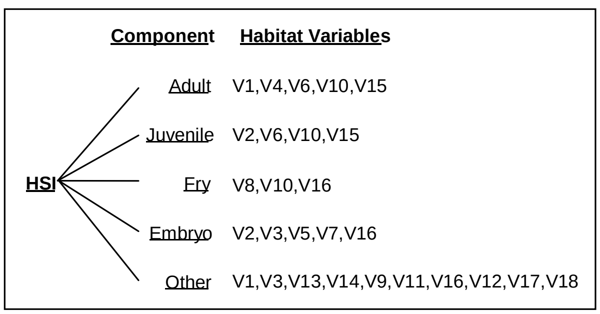


Figure 4. Model components and variable labels in the USFWS rainbow trout/steelhead HSI model.

In addition to the 18 original HSI variables listed below, new habitat variables were measured in 2012 and assessed for use in a new, alternative HSI model.

4.2.1 HSI Variables

The original 18 HSI variables are listed in Table 3 and shown in Figure 5. Raleigh et al. (1984) contains descriptions of each habitat variable as well as all model formulas; however several of the HSI curves were modified from the "original" published curves (see Table 3) for application in the Ventura River Basin, as described below. Modifications to HSI curves are encouraged by the model authors if available information suggests that site-specific curves are more appropriate. Modifications were deemed necessary for several of the variables due to the potential difference in tolerances of *O. mykiss* in the southern portion of their range to the harsh environmental conditions characteristic of southern and central California. Without such modifications, HSI scores will frequently calculate to zero suitability, despite the persistence of *O. mykiss* populations. Most of the modified curves described below were developed and applied during previous HSI studies (TRPA

2004, 2007, 2008). General descriptions of field procedures used to estimate each variable are found in those reports.

Table 3. Description of HSI model variables (modified curves are described in text). See Raleigh et al. 1984 for more details and for model formulas.

Variable Label	Variable Description	Model Component	O. mykiss Lifestage	HSI Curve Modified ?
V1 a,b	Avg Max Water Temperature	Other, Adult	rearing, migration (adult)	Y
V2 a,b	Avg Max Water Temp (Smolts & Eggs)	Juvenile, Embryo	migration (smolt), incubation	Y
V3	Avg Min Dissolved Oxygen	Embryo, Other	incubation, rearing	N
V4	Avg Thalweg Depth	Adult	rearing	N
V5	Avg Velocity Over Spawning Areas	Embryo	incubation	Y
V6 a,j	% Instream Cover	Adult, Juvenile	rearing	N
V7	Avg Substrate Size in Spawning Areas	Embryo	incubation	N
V8	% Large Substrate	Fry	overwintering, rearing	Y
V9	Dominant Substrate in Riffles	Other	food production	N
V10	% Pools	Adult, Fry, Juvenile	rearing	N
V11	Avg % Vegetation & Canopy Coverage	Other	food production	N
V12	Avg % Rooted Veg or Rock on Banks	Other	all	N
V13	Annual Max/Min pH	Other	all	N
V14	Avg Annual Base Flow	Other	rearing	N
V15	Pool Class Rating	Adult, Juvenile	rearing	N
V16 i,f	% Fines in Riffles and Spawning Areas	Fry, Embryo, Other	incubation, food prod	N
V17	% Overhead Shading	Other	rearing, food prod	Y
V18	Avg % Flow During Adult Migration	Other	adult migration	N

Variable Explanations:

V1 a	avg max temp during fry, juv, and adult rearing
V1 b	avg max temp during adult steelhead upstream migration
V2 a	avg max temp during smolt downstream migration
V2 b	avg max temp during egg incubation
V3	avg min DO during egg incubation and during fry, juv, and adult rearing
V4	avg thalweg depth during low flows (only small stream curve shown)
V5	avg velocity over spawning areas during incubation (modified to distinguish steelhead from rainbows)
V6 a	% instream cover at depths >30cm and vels <15 cm/s during low flows for adult steelhead (velocity criteria ignored)
V6 j	% instream cover at depths >15cm and vels <15 cm/s during low flows for juvenile steelhead (velocity criteria ignored)
V7	avg substrate size in spawning areas
V8	% of substrate 10-40 cm diameter for fry and juv overwintering and escape cover (modified to include larger boulders)
V9	predominant substrate size in riffle-run food producing areas (3 classes: rubble & sml boulders dominant = best score, fines or bedrock or lrg boulders dominant = worst score, gravel dominant or even mixture of all types = medium score)
V10	% pools during low flows
V11	avg % vegetation ground cover and canopy closure along streambanks during low flows (shrubs give highest rating, grass medium, and trees lowest)
V12	avg % stable streambanks due to rooted vegetation or rock substrate
V13	annual max or min pH value (use lowest HSI score)
V14	ratio of avg base flow : avg annual flow
V15	pool class rating during low flows (3 classes: large/deep w cover highest, small/shallow w/out cover lowest)
V16 i	% fines (<3mm) in spawning areas during low flows
V16 f	% fines (<3mm) in riffle-run food producing areas during low flows
V17	% of stream channel shaded between 1000-1400 hrs (modified to allow full shading)
V18	ratio of avg flow during adult steelhead upstream migration : avg annual flow

The majority of variables listed in Table 3 are best measured during low flow conditions that typically exist from late summer into early winter, but the migration and spawning variables are best

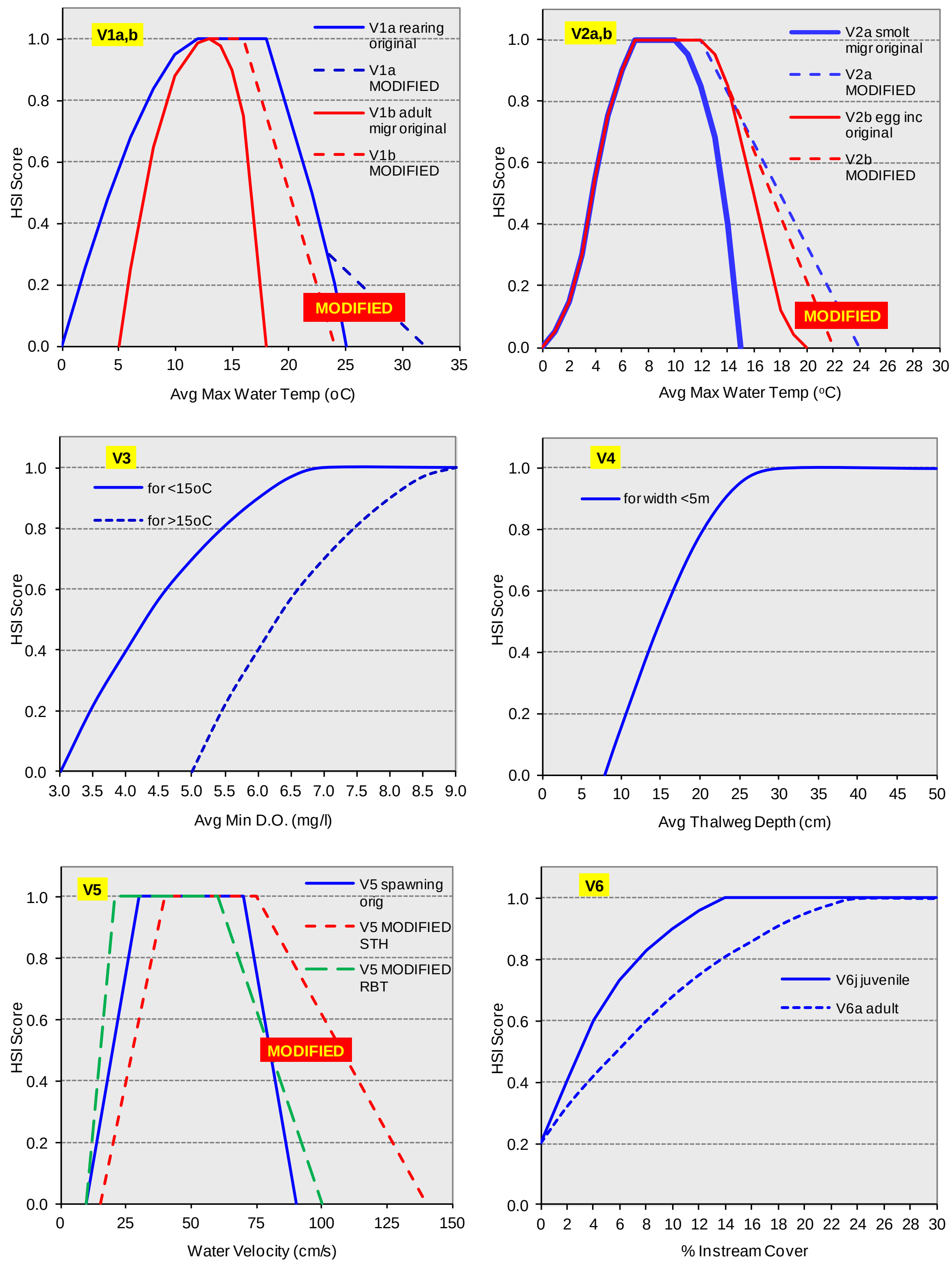


Figure 5. Original HSI variable curves from Raleigh et al. (1984). Curves modified for use in this study are shown. See Table 3 for variable descriptions.

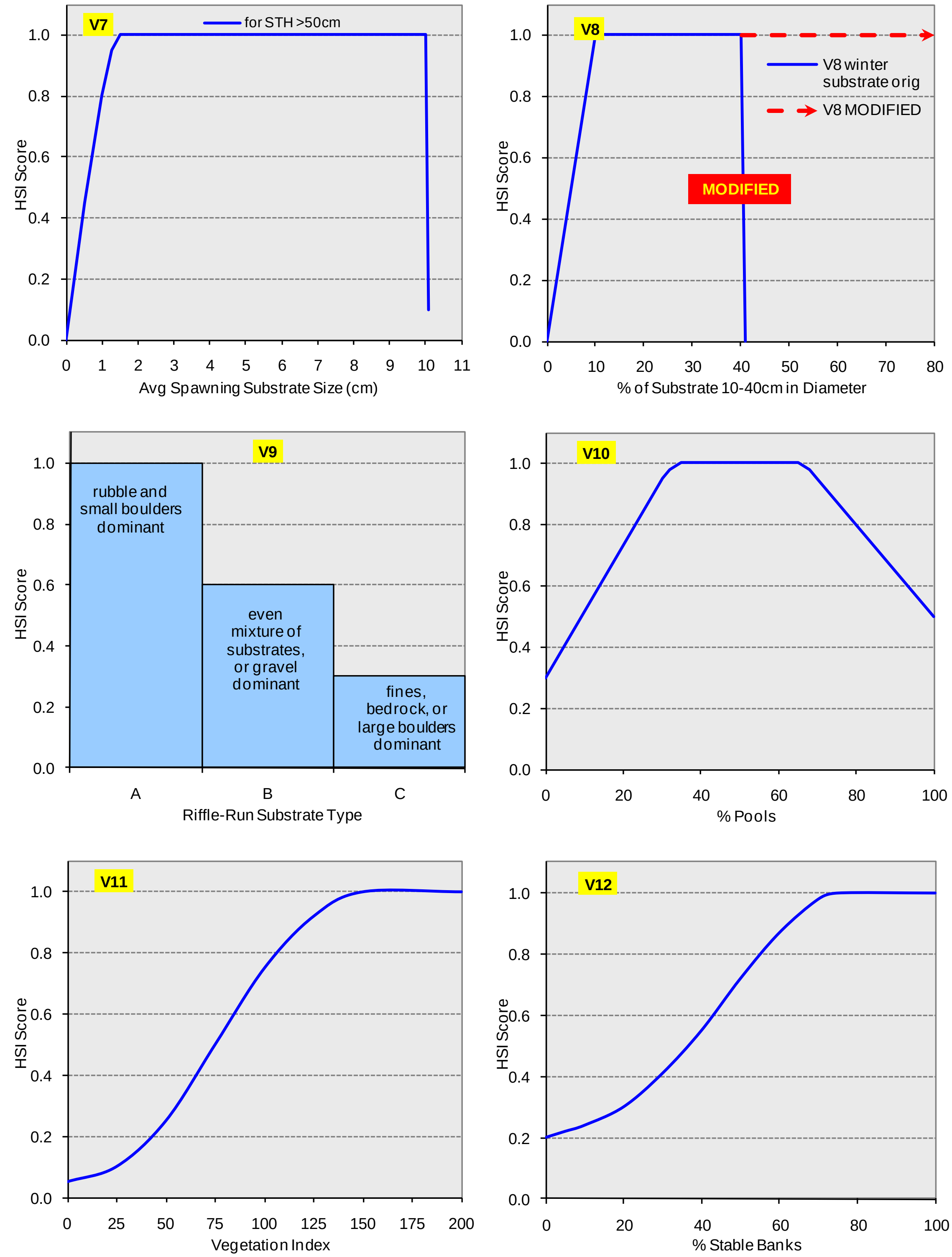


Figure 5. (continued).

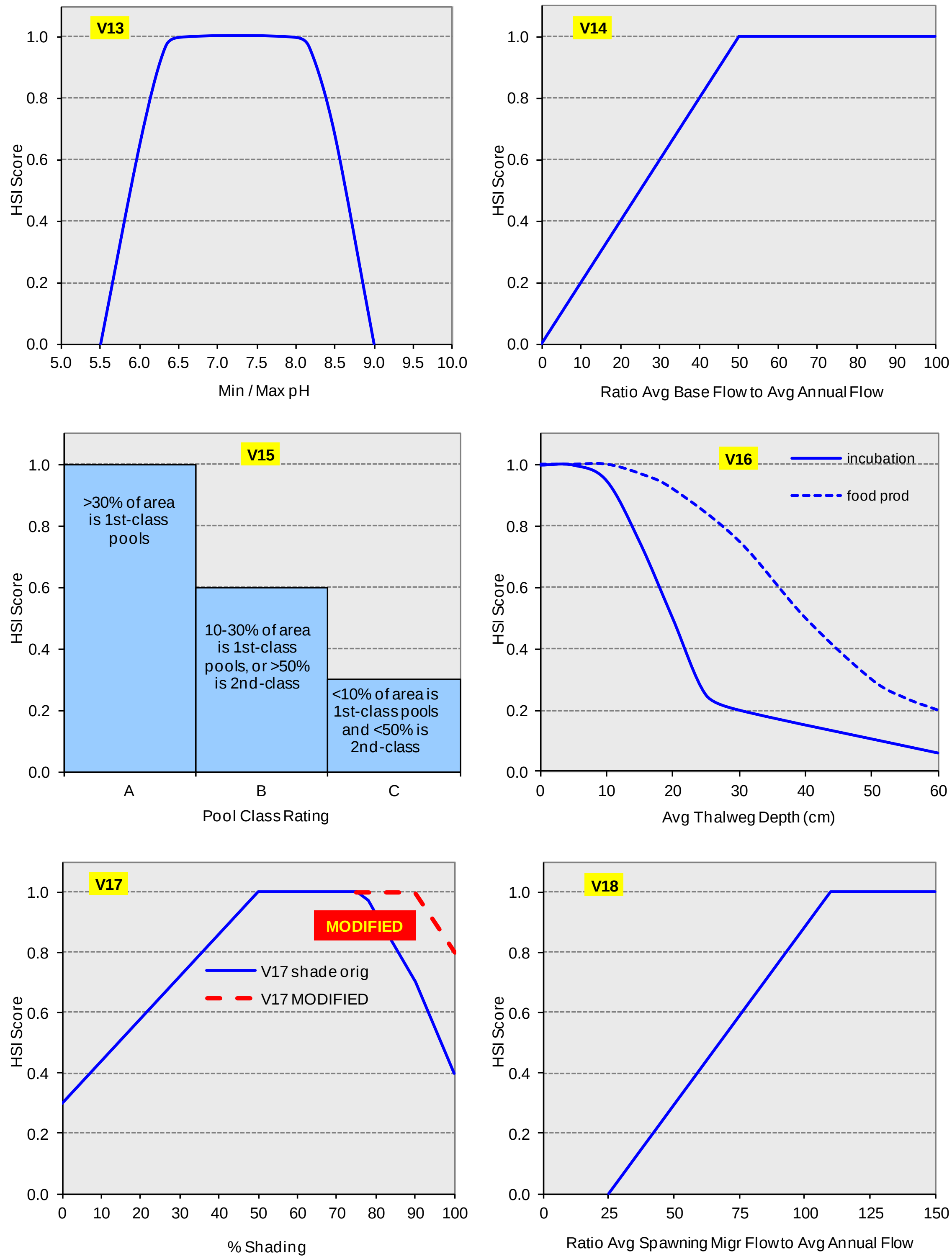


Figure 5. (continued).

measured during spring under when higher flows and cooler temperatures occur. Continuous recording data loggers allowed the collection of springtime water temperatures, but spring dissolved oxygen and spawning variables (gravel patch characteristics) were collected during summer or during intermittent springtime spot checks.

The full suite of HSI field data was collected in 2003, 2006, and 2011, using largely identical protocols and personnel in each year of study. Thalweg depths (*V4*, Table 3) were typically re-measured each year, and riparian vegetation (*V13*) was re-measured in 2007 due to rapid growth following the channel-scouring flows of 2005. New HSI data were also collected whenever new study sites were added (e.g., LNF new and UNF new in 2007, SAC mid and SAC up in 2010, Mat 7b in 2011, and Mur 3 in 2012). Water temperature variables were calculated in 2010 through 2012 from Onset U-22 temperature probes logging data at 30 minute intervals that were deployed throughout the Ventura River Basin (Figure 1), whereas temperature variables in previous years were estimated using spot measurements by fish field crews and by the Santa Barbara Channelkeeper's Ventura River Stream Team (http://www.sbck.org). Additional continuous temperature data from other locations was provided in 2012 by Casitas Municipal Water District (CMWD) (Scott Lewis, personal communication). The temporal periods over which continuous water temperatures were collected at each HSI study site are listed in Table 4.

Table 4. Periodicity of temperature dataloggers deployed throughout the Ventura River Basin.

Study Site	Duration of Deployment [1]
Ven 1 [2]	01/01/10 - 07/05/12
Ven 2	05/02/10 - 09/29/10 and 05/12/11 - 09/30/12
Ven 3	04/30/10 - 09/29/10 and 05/12/11 - 09/30/12
SAC low [3]	03/28/12 - 09/30/12
SAC mid [4]	05/13/11 - 03/27/12 and 08/16/12 - 09/30/12
SAC up	05/02/10 - 09/30/12
Ven 5	05/01/10 - 09/30/12
LNF	05/02/10 - 09/30/12
Mat 3	05/02/10 - 09/30/12
Mat 5	05/02/10 - 09/30/12
Mat 7	05/13/11 - 09/30/12
UNF new	05/01/10 - 09/29/10
Mur 3	03/27/12 - 09/30/12

[1] many loggers remained deployed at conclusion of study (contact Paul Jenkin of Surfrider for data)

[2] data from CMWD logger at Main St Bridge

[3] surface flow appeared to cease on 07/08/12

[4] data corrupted 03/28/12 - 08/15/12, dry thereafter

4.2.2 Modified HSI Variables

Average Maximum Water Temperature for Rearing (V1a)

The warm stream temperatures prevalent in most southern and central California steelhead streams and the "cool" temperature HSI curves ("original" curves in Figure 5) proposed by Raleigh et al. (1984) frequently produced zero HSI scores (TRPA 2004). Given the continued persistence and sometimes high densities of trout or steelhead in many such streams, the "original" HSI curves did not appear to adequately represent temperature suitability for southern or south-central *O. mykiss*.

Because of this unrealistic fit and because of the high genetic variability and the ability of California populations to exist in seemingly unfavorable environments (Moyle 2002), the HSI curves for average maximum temperatures (*V1* and *V2*) were modified from those in Raleigh et al. (1984).

These curves were modified using professional judgment and temperature data from several warm streams in California known to contain abundant *O. mykiss*. For example, the rearing curve (*V1a*) was modified using available temperature data from the Ventura River (this study), the lower Klamath River at Seiad Valley (USFWS Arcata, website data), Topanga Creek (Spina 2007), and maximum temperatures reported in Moyle (2002) and Myrick and Cech (2000). Based on the above data, the upper end of the temperature curve was extended from the original suitability of 0.3 at 23.5^{o}C to a new zero point at 32^{o}C (Figure 5). Note that Sloat and Osterback (2013) predicted occupation of *O. mykiss* in pools with temperatures just over 32^{o}C. It is recognized that these HSI curve modifications are not based on rigorous scientific experiments, and they may not account for a fish's ability to actively seek out temperature refuges and thereby avoid some of the maximum temperatures described above. Although the temperature requirements of southern steelhead during various life stages is poorly understood, it appears that the temperature graphs presented by Raleigh et al. (1984) are inappropriate for southern and central populations of *O. mykiss* for several life stages, including freshwater rearing (*V1a*), adult upstream migration (*V1b*), smolt out-migration (*V2a*), and egg incubation (*V2b*).

For study sites reaches not containing a temperature probe, data was obtained either from CMWD (Ven 1), or data was estimated by linear regression using data from the most similar study site containing a data logger. For example, temperature data were not available for the entire 2012 incubation period (Jan-April) from the Murietta study site, so the available data were used to estimate incubation temperatures by regression with temperature data from the LNF study site. Comparative rearing and incubation temperatures in the LNF and UNF study sites in 2010 were also used to estimate missing incubation and rearing temperatures in 2012 for the UNF study site (due to the loss of the UNF logger over the winter of 2010-11). Finally, incubation and rearing temperatures for the Mat 5 study site, which did not have a data logger, "borrowed" data from the Mat 7b study site. Although the Mat 5 study site was located well downstream of Mat 7b, inflow from Murietta Creek and groundwater from the Matilija and NF Matilija confluence provided relatively cool inflow to the Mat 5 study site.

For each HSI study site, the average maximum water temperature for rearing (*V1a*) was estimated in by calculating the mean value of weekly average maximum water temperatures over the period of July through August 2012, based on water temperatures either measured with the temperature data loggers or estimated by the procedures described above.

Average Maximum Water Temperature for Adult Upstream Migration (V1b)

The original HSI curve was modified using water temperature data from the lower Klamath River in August-September, when summer-run steelhead ascend the mainstem river (USFWS web data), and from December to March data from San Luis Obispo Creek (TRPA unpub data). This variable was calculated using the mean value of the weekly average maximum water temperatures logged from January through April 2012 (Figure 5), but only for those study sites located in the anadromous zone (i.e., the lower and middle study segments).

Although adult migrants in upstream study sites like the LNF and SAC study sites must pass through lower, warmer reaches of the mainstem Ventura River, this study assumed that adult steelhead will, given adequate flows, migrate rather quickly through the lower reaches and will spend the majority of their pre-spawning time in the vicinity of the spawning location (e.g., within the study site in question). Consequently, HSI migration variables (*V1b* and *V2a*) for all anadromous study sites utilized temperature data from within the site, and not for downstream reaches, which may be warmer. This assumption would be less appropriate for larger basins with longer mainstem migration corridors than exists in the Ventura River Basin.

Average Maximum Water Temperature for Smolt Downstream Migration (V2a)

The original HSI curve was modified (Figure 5) using March through May temperature data from San Luis Obispo Creek in central California (TRPA unpub data), and April to May data from the lower Klamath River (USFWS web data), the Mad River (Sparkman 2002, 2003), Redwood Creek (Sparkman 2002, 2003, 2004), and Bear Creek (Ricker 2002) in coastal northern California. This variable was also calculated using the mean value of the weekly average maximum water temperatures logged from March through May 2012, and only for study sites in the anadromous zone (i.e., the lower and middle study segments). As described above for variable *V1b*, HSI scores for this variable were based on migration temperatures from within the study site, and not from downstream study sites, under the assumption that smolts will migrate rather rapidly through lower reaches whereas the time spent during the actual smoltification process would be much longer and would likely occur within the study site itself.

Average Maximum Water Temperature for Incubation (V2b)

The modification procedures described for the *V1* variables were again applied to variable *V2b,* however information describing incubation temperatures in warm salmonid streams was not located, therefore the shown modification was drawn entirely by eye and the proposed change is relatively minor, giving a shift in the zero point from 20^{o}C to 22^{o}C (Figure 5). The mean of weekly average maximum temperatures was calculated over the period of January to April 2012, using either site-specific logger data or estimated data as described for variable *V1a*.

Spawning Area Velocity (V5)

Raleigh et al (1984) proposed a single curve to represent the suitability of water velocity over spawning gravels for both rainbow trout and the (typically) much larger steelhead. The original curve appeared too restrictive for steelhead, which are commonly known to spawn in velocities faster than indicated by the original HSI curve, and too rapid for smaller stream resident trout inhabiting small headwater streams. NAI’s habitat suitability library contains a large collection of habitat suitability curves that represent velocities selected by spawning steelhead and resident trout. These curves were plotted against the original HSI curve (TRPA 2007), and the HSI curve was modified by professional judgment to better represent suitability for spawning in headwater streams either by adult resident trout or anadromous steelhead (Figure 5).

Mean velocities over potential spawning areas were either measured with a mini current meter on a handheld rod or were visually assessed by estimating the distance and speed at which floating objects (e.g., sticks or leaves) passed over gravel patches. Because spawning gravels were mostly assessed during the summer survey, and not under higher flows that are typical during the spring spawning season, the summer velocities were doubled prior to comparison with the HSI spawning velocity curve, as per TRPA (2007). Also, all spawning gravels located in riffles, flatwaters, or pool tails were given a minimum velocity suitability value of 0.25, to further account for the lower velocities encountered during the summer sampling period. Limited springtime data was collected over spawning patches in Ven 2, Ven 5, and Mat 5 in 2010, and in Mat 7b in 2011. These springtime velocities were *not* doubled prior to calculating HSI values.

Percent Instream Cover (V6)

Raleigh et al. (1984) suggested that instream cover should be at least 15cm in depth and occur in velocities <15cm/s in order to be useful. The HSI curve for percent cover was not modified, however we disregarded the velocity criteria, since instream cover typically possesses velocity shelters, and instead assessed cover regardless of velocity. Also, for study sites in anadromous segments (lower and middle segments), we re-defined minimum depth to be 30cm for adult steelhead, but kept the 15cm depth criteria for juveniles.

Percent Large Rearing Substrate (V8)

Winter hiding substrate was defined by Raleigh et al. (1984) as substrate particles 10cm to 40cm in diameter, but suitability for larger sizes was not defined. Because overwintering salmonids are frequently observed to utilize larger cover elements (e.g., boulders, rip-rap, woody debris, etc.), we re-defined winter cover as any substrate particle or woody vegetation >10cm in diameter, thus including larger cover elements as well as undercut banks (Figure 5). Although the relative importance of this variable for overwintering in mild southern California streams is uncertain, this variable was retained in the analysis.

Percent Overhead Shading (V17)

Midday shading was eye-estimated from one or more locations in each selected habitat unit, with the number depending upon unit size and riparian complexity. The HSI curve used in this study was modified from the original curve presented in Raleigh et al. (1984), by extending the area of maximum habitat suitability to include areas with greater canopy closure (Figure 5). Although closed canopies would typically result in lower invertebrate production, the added benefit of cooling the water temperatures in southern and central California streams might be expected to offset the reduced food production. Consequently, the HSI score of 1.0 was extended to include shade values from 75% to 90%.

"Tributary Effects" Variable

The USFWS HSI model assumes, through the spawning variable (*Vs*) and water quality parameters related to egg survival, that recruitment of fish into the study area occurs solely by spawning and emergence within the survey area. The *Vs* variable is calculated by scoring the quality of the *V5, V7*, and *V16sp* variables at each gravel patch, then weighting the combined patch score by the patch size. Previous HSI studies have shown that the *Vs* variable is highly influential on the overall HSI score (TRPA 2007), but no account is made for recruitment of fish into a study area from upstream (or downstream) sources. Consequently, if spawning habitat or incubation conditions are limiting, the model may yield a low overall suitability even if rearing conditions are suitable for immigrant fry, juvenile, or adult fish. Three of the HSI study sites occur in close proximity to spawning tributaries (Ven 3, Ven 5 and Mat 5), and the latter two sites have limited spawning gravels. Because the *Vs* variable resulted in low overall HSI scores that did not appear consistent with the observed densities of *O. mykiss* in those reaches, a new variable was added to help account for recruitment of fish from nearby spawning areas.

If fry densities within a study site are mostly associated with in-site spawning and emergence, no relationship would be expected between unit-specific fry densities and distance to spawning tributary. However, if a negative relationship is observed between fry densities and distance to spawning area, fry recruitment from tributaries may be in effect and could potentially compensate for limitations in local spawning and incubation habitat. This appeared to be the case in several mainstem study sites that were situated immediately downstream of spawning tributaries. Consequently, as an alternative to using only the *Vs* and incubation variables to represent the recruitment potential of fry into a study site, fish abundance data from three study sites were evaluated to assess the potential effects of recruitment from the upstream spawning tributary. This "tributary effects" variable was evaluated using 2007, 2010, and 2011 abundance estimates of *O. mykiss* fry <10cm from riffle and flatwater habitats in the Ven 5 study site due to recruitment from the Lower North Fork Matilija Creek, in the Mat 3 and Mat 5 study sites due to recruitment from the Upper North Fork Matilija Creek and Murietta Creek, and in the Ven 3 study site due to recruitment from San Antonio Creek.

The relationship between recruitment tributaries and fish abundance was evaluated by plotting riffle-specific fry densities in each study site against distance from the upstream tributary. Only fry were used in this analysis because the tributary effects variable was intended to supplement the embryo component as an alternative to the *Vs* variable; juvenile and adult fish are assessed by separate components (Raleigh et al. 1984). The assessment also only included riffles since *O. mykiss* fry were typically much less abundant in pools and flatwaters. Prior to plotting, fry densities in individual units were first normalized by year and study site for each spawning tributary (e.g., Ven 3 below San Antonio Creek, Mat 3/Mat 5 below Murietta Creek, etc.) where the highest density was set to 1.0, and all lower densities were scaled accordingly. Fry densities were normalized in an attempt to minimize differences due to year and emphasize differences due to distance below spawning tributaries. The relationship between normalized fry densities and distance from the tributary was then fitted with a logarithmic regression curve, which was also normalized to yield a maximum suitability factor of 1.0 at the highest estimated density (Figure 6).

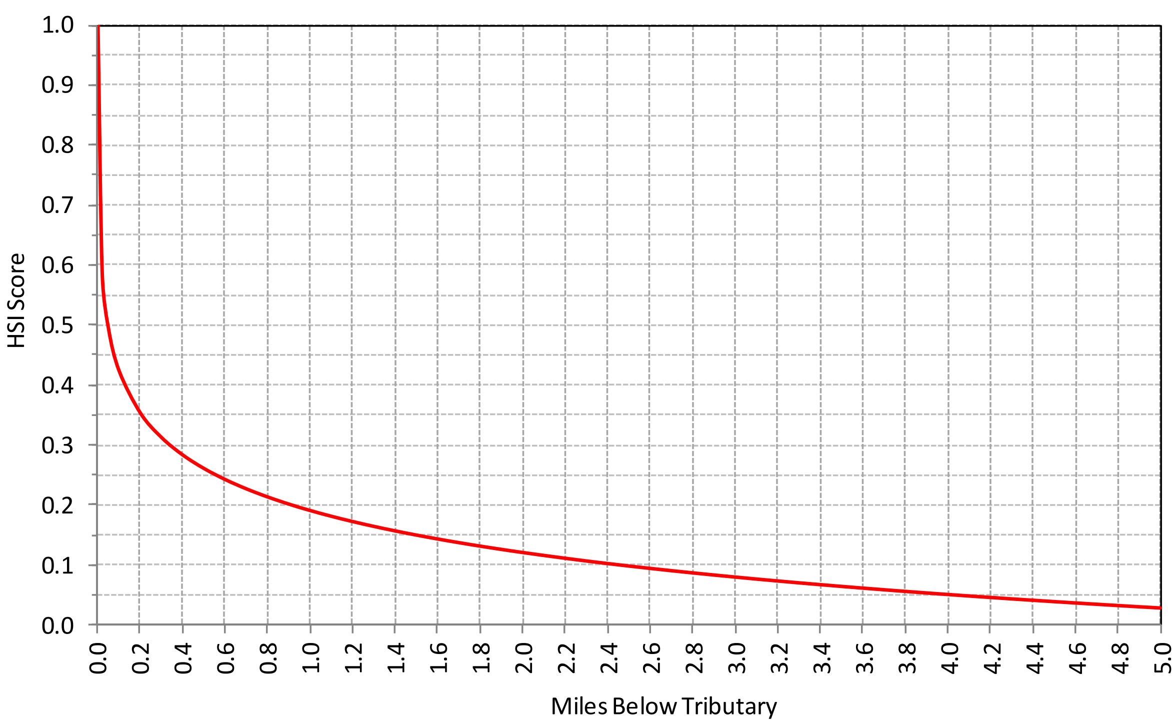


Figure 6. Tributary effects HSI curve.

An HSI variable score for “trib effects” was then estimated for all mainstem study sites by reference to that curve, according to the distance from the nearest spawning tributary to the middle of the HSI study site (i.e., the mean distance). The trib effects value was then compared to the original embryo component score (which was the minimum of the *Vs*, incubation temperature, and incubation D.O. scores) for that study site, and the maximum of those two scores was used to represent the embryo component of the HSI model. The trib effects HSI score was set to zero for all study sites within spawning tributaries so that the embryo component score was based only on localized spawning and incubation habitat suitability.

The trib effects curve produces a high suitability value only for study sites in the immediate proximity of spawning tributaries, reflecting the apparently short spatial effects of tributary recruitment on mainstem fry abundance. This HSI variable curve should be re-evaluated using mainstem data from other southern California basins.

4.2.3 Calculation of USFWS HSI Scores

HSI scores were calculated for each study site using several alternative model formulas and assumptions. In prior Ventura Basin HSI assessments (TRPA 2007, 2008), the steelhead equal components HSI model was used for all sites below Matilija Dam, and the resident trout equal components model for all sites above Matilija Dam. The equal components formula gives equal weight to each of the five model components (adult, juvenile, fry, incubation, and other) when calculating the overall HSI score (Raleigh et al. 1984). The HSI score calculations reported in those prior reports did not utilize the “limiting variable” option, which sets a component HSI score or the overall HSI score to a minimum value if a subcomponent or component score is less than 0.4.

The 2012 HSI assessment again utilized the USFWS equal components model, both with and without applying the limiting variable option. In addition, to fully compare different options presented in the USFWS HSI model, 2012 HSI scores were also calculated using the unequal components options (where components can be weighted differently) with either the compensatory or the non-compensatory assumptions. The compensatory option assumes that limitations in some HSI variables can be partially compensated for by other variables; this option is suggested for larger rivers or where water quality problems are temporary (Raleigh et al. 1984). The non-compensatory option implies that poor water quality cannot be overridden by good physical habitat; this option is suggested for smaller streams or where poor water quality is persistent.

4.3 Southern Steelhead (SS) HSI Model

The USFWS HSI model with the curve modifications described above was evaluated in 2006 and 2007 by comparing *O. mykiss* abundance estimates with HSI scores in each study site (TRPA 2007, 2008). In both years, the HSI models were statistically significant and the HSI scores explained between 60% and 71% of the variation in abundance of fry and juvenile+ *O. mykiss*. However, the models did not adequately distinguish between reaches having low abundance of fish and reaches rarely containing fish. Also, the model produced HSI scores that were clumped in the middle to high range and did not show a breadth of suitability consistent with visual assessment of habitat quality. Consequently, additional habitat variables were collected in 2012 and assessed in order to determine if a new HSI model, termed the “Southern Steelhead HSI Model” (SS HSI), could be developed that might be tested against independent data from within the Ventura River Basin or other southern California steelhead streams.

The SS HSI model retains many of the variables and some of the form of the original USFWS *O. mykiss* HSI model, but adds new variables, new components, and different mechanisms of combining the component scores into an overall HSI score (Figure 7). SS HSI scores are derived for five habitat-related components:

1) *physical habitat characteristics at the habitat-unit scale (depth, velocity, and cover), by habitat type and size/age class;*
2) *physical habitat, biological, and flow-related characteristics at the reach scale (all age classes);*
3) *recruitment (fry only);*
4) *water quality (all age classes);*
5) *migration (for anadromous steelhead adults and smolts only)*.

Consistent with the USFWS HSI model, the SS HSI model also allows calculation of HSI scores for each age class, defined as follows:

1) *O. mykiss fry (0+)- whether of resident or anadromous origin;*
2) *juveniles (representing 1+ or 2+ fish up to smolt size) - whether of resident or anadromous origin;*
3) *adult resident rainbow trout;*
4) *adult migratory steelhead.*

The habitat unit component scores are calculated separately for fry, juvenile, and resident adult trout age classes due to the different microhabitat requirements of rearing fish; whereas the broader-scale HSI components (e.g., reach and water quality) are not based on age classes. As noted above, the migration HSI component only represents the two anadromous life-forms exhibiting large-scale migrations: steelhead adults and smolts.

An overall SS HSI score is calculated for each study area by combining all pertinent component HSI scores (see Section 4.3.6). For resident trout reaches, the migration component is excluded; for anadromous reaches, the migration component is included but the resident adult sub-component of the habitat unit component is not, assuming that juveniles will emigrate as smolts.

Following is a description of the variables and associated HSI curves for these five habitat-related components.

4.3.1 Habitat Unit Component

A wide variety of habitat variables were measured or estimated in 2012 to describe the suitability of individual habitat units (pools, flatwaters, or riffles) for rearing fry, juvenile, and (resident) adult *O. mykiss*.

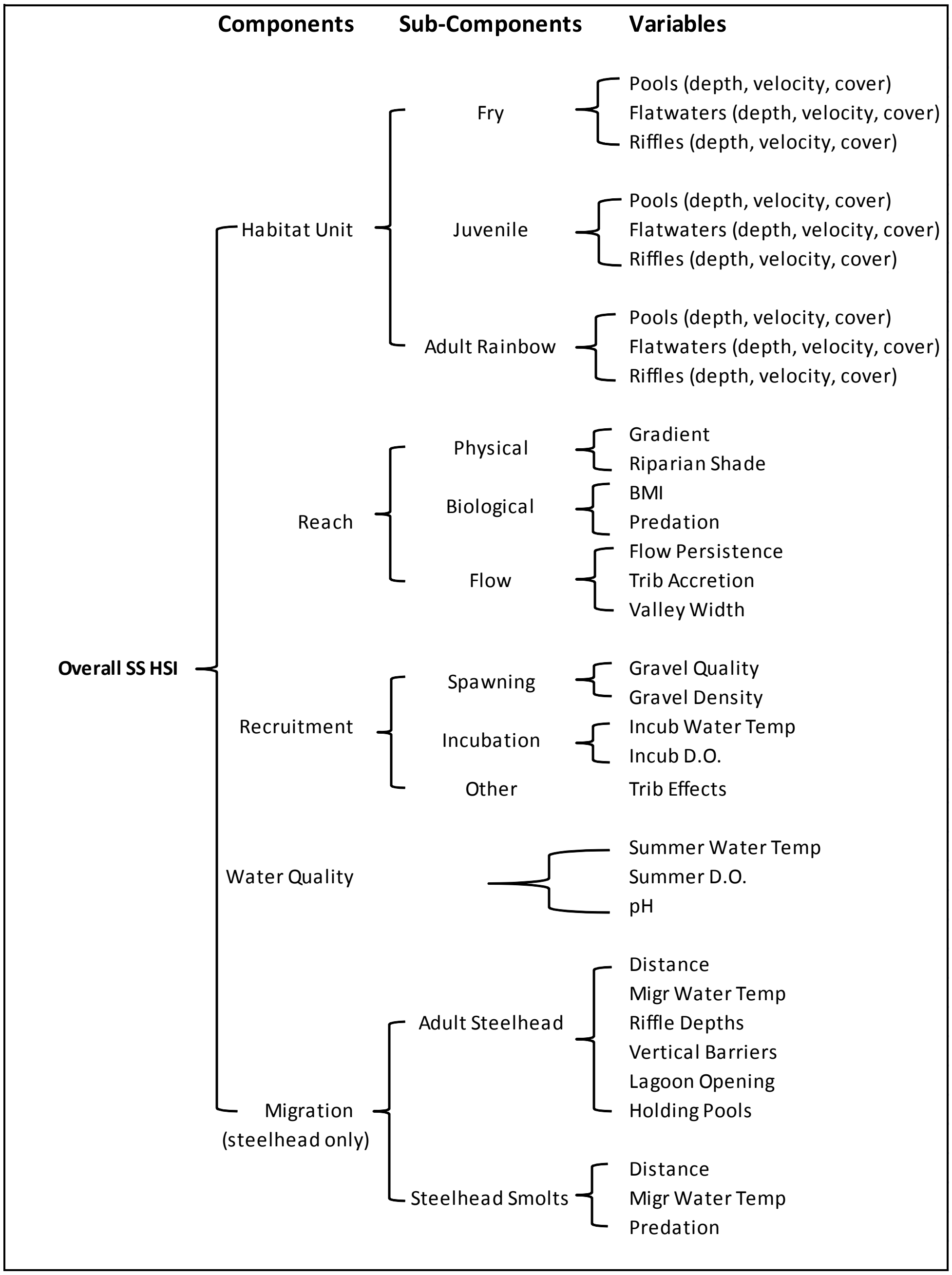


Figure 7. Dendritic chart showing structure of SS HSI model. The habitat unit component was also stratified by channel size (mainstem vs. tributary).

CDFG Habitat Typing Variables

Additional habitat variables not represented by current HSI variables, but encompassed within the CDFG habitat typing protocols (Flosi et al. 1998), were collected in each sampling unit in 2012. These variables included a measurement of pool tail crest depth, a count of large woody debris (LWD, defined as pieces ≥6 ft long x ≥1 ft in diameter), a visual estimate of the percentage of each sampling unit containing undercut bank, small woody debris (SWD, <ft in diameter), LWD, roots, riparian vegetation, aquatic vegetation, entrained air (bubble curtain), and unembedded boulders, and an estimate of overall shelter rating. See Flosi et al. 1998 for additional description of these variables.

Depth/Velocity/Cover Variables

Data from this study as well as other studies on the Ventura River (TRPA 2009b) and Central California streams (e.g., Spina et al. 2005, TRPA 2007b) have shown positive associations between juvenile *O. mykiss* abundance and pool habitat characteristics, particularly depth. Although water depth is undoubtedly an important variable, adequate velocities are also important for providing invertebrate drift to feeding salmonids, particularly in the warm, low flow conditions typically encountered in southern California streams. Combinations of depth, velocity, and instream shelter are all expected to be important components of habitat suitability, thus their addition (in various forms) to habitat assessments such as the HSI model and the CDFG habitat typing protocol. To further assess the relationship between habitat unit depths, velocities, and cover attributes with *O. mykiss* abundance, a transect-based mapping protocol was initiated in 2012 to estimate the total surface area of each habitat unit that was composed of various combinations of depth, velocity, and instream or overhead cover.

Physical habitat was mapped within each sampling unit in all study sites (except for Ven 1 where *O. mykiss* are rarely observed) along five evenly spaced, cross-sectional transects. Three transects were used for units <20 ft in length, and 10-12 transects were used in 6 habitat units to assess the effects of transect number on accuracy of area estimates. The distance across each transect that contained water depths of <1 ft, >1 ft, >2 ft, and >3 ft was measured using a depth rod by reference to an overhead tape measure or (in small channels) a stadia rod laid bank-to-bank across the channel (Figure 8). In a like manner, the distance across each transect containing surface water velocities <0.5 fps, >0.5 fps, and >1 fps were estimated by observing floating objects as they passed across the transect, with reference to the overhead tape and a hand-held ruler. Finally, the distance across each transect that was composed of cover elements including unembedded cobble or boulder substrate, instream branches, overhead branches (w/in 18 inches of water surface), aquatic vegetation, or surface turbulence, was likewise recorded for each transect.

The cover variables were subsequently condensed into in-water cover (cobble/boulder, instream branches, and aquatic vegetation) or overhead cover (turbulence or overhead branches). Location data for the 4 depth categories, 3 velocity categories, and 2 cover categories were overlaid for each transect in order to estimate the length of the transect that was composed of all combinations of depth, velocity, and cover categories. These lengths were summed across all transects and expanded by the unit's total surface area to produce a total estimate of the surface area (or total percentage) in each sampling unit according to each depth/velocity/cover type. In addition to estimating depth/velocity/cover combinations, the transect depth and velocity data was used to estimate mean depth and mean velocity for each sampling unit (except in Ven 1).

Ranking and Selection of Habitat Unit HSI Variables

To assess the utility and performance of adding new variables to the *O. mykiss* HSI model, the large suite of new habitat unit variables collected in 2012 (12 CDFW variables, 43 depth/velocity/cover combinations, and unit mean depth and mean velocity), in addition to the 18 original HSI variables (many re-measured in 2012), required initial assessment and data reduction prior to model development and evaluation. This assessment was conducted by a multi-stage process. First, Pearson product-moment correlation coefficients were calculated between pertinent unit-specific variables and *O. mykiss* densities within habitat units, according to life stage (fry, juvenile, or resident adult) and channel size (mainstem vs. tributaries study sites). Variables that produced correlations that were consistently near zero were subsequently dropped from further consideration.

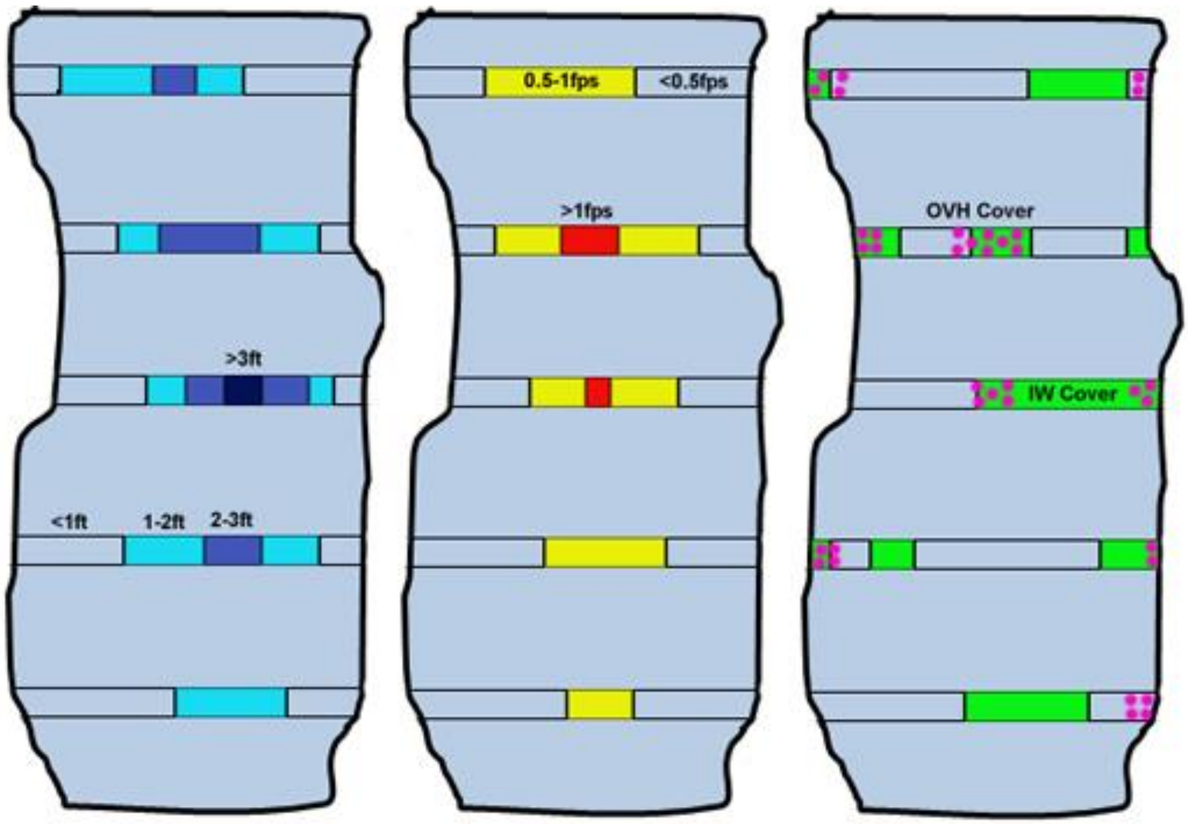


Figure 8. Example map of habitat unit showing 5 transects and boundaries of depth categories (left figure), velocity categories (middle), and cover categories (right). Superimposition of data will yield area estimates of depth, velocity, and cover combinations.

The remaining variables were then assessed for complexity and redundancy. The three-parameter variable combinations (e.g., the percentage of habitat containing a combination of, for example, depth >1 ft with a velocity >0.5 fps and instream branches) were dropped from consideration due to the difficulty of estimating those combined values, as well as the rarity of most three-variable combinations in sampling units. In contrast, the two-parameter variables were more commonly encountered and were deemed feasible to estimate in the field, whether quantitatively (via direct measurement as in this study) or qualitatively (via visual assessments).

Next, all remaining variables were assessed for redundancy. For example, all three variable sets (FWS HSI variables, CDFW habitat typing variables, and new HSI variables) contained variables associated with substrate cover elements. The USFWS HSI model included a variable for percent juvenile or adult cover (which is mostly substrate related in the Ventura Basin) as well as riffle/run substrate type; the CDFW habitat typing protocol included a percent boulder variable; and the new HSI variable dataset contained a percent cobble/boulder variable. Similar redundancies occurred among other cover and depth-related variables. Inspection of correlation coefficients was used to retain the redundant variable that produced the highest average value. An AllCov variable, which combined the four primary cover variables (CB, TURB, IW BR, and OH VEG, Table 5), was excluded from selection in most cases due to redundancy, however several models for adult resident trout could not be developed without this combined cover variable.

The foregoing steps reduced the habitat unit variable list from over 50 depth, velocity, and cover variables to 17 variables (Table 5). Stepwise multiple regressions were then used to further reduce the variable list and to create habitat unit HSI values by predicting *O. mykiss* densities in habitat units according to the unit's habitat attributes. Stepwise regression models were developed for each life stage and habitat type according to channel size, using unit-specific depth, velocity, and cover habitat attributes as predictor variables and *O. mykiss* densities as response variables. This process resulted in 18 models based on three life stages (fry, juvenile, and resident adult), two channel sizes (mainstem and tributary), and three habitat types (pool, flatwater, and riffle). Models were developed for each life stage due to the well-established differences in habitat requirements among salmonids of different size and age classes (e.g., Everest & Chapman 1972, Moyle & Baltz 1984, Campbell & Neuner 1985). Models were also developed for mainstem versus tributary study sites due to much-improved correlation coefficients between unit *O. mykiss* densities and habitat attributes when stratified by channel size, likely due to broader (reach-wide) habitat and water quality effects that are assessed in other model components (see below).

Table 5. Variables used for estimating the suitability of individual habitat units.

Variable ID	Description
AV DEP	Mean Depth ft
MX DEP	Maximum Depth ft
D1	% Area >1 ft
D2	% Area >2 ft
D3	% Area >3 ft
AV VEL	Average Velocity fps
V05	% Area >0.5 fps
V1	% Area >1.0 fps
CB	% Area w cobble/boulder cover
TURB	% Area w turbulence cover
IW BR	% Area w inwater branches/roots/woody debris
OH VEG	% Area w overhead vegetation or undercut bank (w/in 18 in of surface)
FINES	% Area w fine substrate (sand, silt, clay)
V05,IW	% Area >0.5 fps and inwater cover (CB or IW BR)
V05,OW	% Area >0.5 fps and overhead cover (TURB or OH VEG)
V1,IW	% Area >1.0 fps and inwater cover (CB or IW BR)
V1,OW	% Area >1.0 fps and overhead cover (TURB or OH VEG)

Finally, separate models were developed for each habitat type because the consistently large differences in *O. mykiss* densities observed between habitat types was used to assign different weighting factors for each habitat type's HSI score (see below), which necessitated developing regression models for each of the three habitat types. Also, it was expected that habitat parameters that are important determinants of, say, riffle quality for fry, may be different than parameters that determine quality of pool habitats for fry. For example, instream branches or undercut banks may serve a relatively unimportant role for *O. mykiss* inhabiting riffles which typically contain abundant substrate and turbulence cover, whereas wood or bank cover may be important in pool habitats where turbulence or large substrate elements are less abundant. In like manner, velocity may be a more important component for providing drift to feeding *O. mykiss* in pools in comparison to riffles, which typically contain an abundance of higher velocity water and potential feeding stations. This expectation was supported by correlation coefficients that were much stronger between *O. mykiss* densities and unit habitat attributes when stratified by habitat type (e.g., combining data among habitat types yielded much lower correlations).

Stepwise regression models were performed in S-plus software (MathSoft 2000), using a forward and backward stepping routine until further refinement in the predictive relationship was no longer achieved. Termination of the variable selection process was based on the magnitude of the Akaike information criteria (termed the C_p statistic in S-plus), which evaluates the goodness of fit of the model at each step by rewarding model accuracy while penalizing model complexity.

Habitat Type Weighting

Multiple years of sampling in the Ventura River Basin and in a wide variety of other basins have consistently shown that habitat type can exert significant effects on local *O. mykiss* densities, and that different life-stages respond differently to habitat type effects. For example, comparison of *O. mykiss* densities in pools, flatwaters, and riffles has shown that fry occur at much higher densities in riffles than in other habitat types (TRPA 2005, 2007, Normandeau 2013). In contrast, larger adult resident trout or holding adult steelhead in smaller streams are more commonly found in pool habitats. Although this trend was highly consistent within both smaller tributary study sites and larger mainstem study sites in the Ventura River Basin, some differences in proportional densities did exist according to channel size, likely due to size-related differences in flow, depth, and water temperatures. Consequently, the SS HSI Model was designed to produce individual HSI scores for each life stage by habitat type and channel size, then those habitat type scores were combined using size-class specific weighting factors to produce an overall habitat unit score.

For example, because *O. mykiss* fry occur at highest densities in riffles and lowest densities in pools, the combined habitat unit HSI score gives a higher weighting for the riffle HSI score, an intermediate weighting for the flatwater HSI score, and a lower weighting for the pool HSI score. These factors differed slightly by channel size, where fry densities were proportionally higher in mainstem riffles than in tributary riffles, with the opposite effect for pools. These weighting factors for each life stage (fry, juvenile, and resident adult *O. mykiss*), channel type, and habitat type were determined using mean relative densities from the five years (2006, 2007, 2010, 2011, and 2012) in which full sampling (riffles, flatwaters, and pools) was conducted in the Ventura Basin. The calculated mean values were then rounded to the nearest 0.05, such that the three habitat type weighting factors per channel size summed to 1.00.

4.3.2 Reach Component

The reach-scale component includes variables for a physical habitat sub-component (e.g., habitat characteristics that are more descriptive at the reach scale than at the habitat unit scale), including channel gradient and riparian shading; a biological sub-component (benthic macroinvertebrates and predation); and a flow sub-component (flow persistence, valley width, and tributary proximity) (Figure 7).

Channel Gradient

Channel gradient is well known to influence salmonid habitat and abundance (e.g., Chisholm & Hubert 1986, Rich et al. 2003). Gradient affects abundance directly through the associated variables of water depth, velocity, and substrate characteristics, which are parameters utilized in the habitat unit component described above. Gradient also exerts significant and direct population effects through habitat accessibility, as well as indirect effects on water quality, riparian vegetation, etc. Despite this overlap with other variables included in this new HSI model, channel gradient is an easily measured attribute that is expected to improve assessments of habitat suitability for southern steelhead.

This new gradient HSI curve was developed using metadata relating *O. mykiss* abundance and channel gradient was compiled from a variety of studies, with emphasis on southern California basins. Although the NMFS presented a gradient suitability relationship for *O. mykiss* in their (Boughton and Goslin 2006), the HSI curve presented here is dominated by local information adapted from site-specific data collected in the Ventura River (this study), Sespe Creek (Dvorsky 2000), Piru Creek (Weaver & Mehalick 2009a,b), and the Santa Ynez River (BOR 2013). Additional data used to develop this HSI curve was taken from several south-central California basins including Morro Bay tributaries (TRPA 2001, 2007b), the San Luis Obispo Creek Basin (TRPA 2004b), and the upper Arroyo Grande Creek Basin (TRPA 2011). Finally, six curve points representing the NMFS curve (points defining the consensus, the 95% envelope, and the complete dataset) were also included in the gradient HSI analysis.

The gradient HSI curve was developed by plotting normalized *O. mykiss* densities of fry and/or juveniles and fitting a non-parametric tolerance limits (NPTL) curve to the grouped data. The NPTL approach is a distribution-free methodology (Wilks 1941) commonly employed in instream flow studies for developing habitat suitability curves from fish observation data (Bovee 1986, Bovee and Zuboy 1988). The NTPL curve was derived using the 90% confidence level and assigning the central 50% of density values to a suitability of 1.0. Suitability values of 0.5 and 0.2 were assigned to the central 75% and 90% of density values, respectively. Suitability was set to zero at gradients of zero and 16.3%, which were the endpoints used in the NMFS curve.

The gradient HSI curve gives maximum suitability to gradients between 1.2 and 6.9%, which is somewhat more restrictive than the NMFS curve (Figure 9). Despite the more restrictive curve, only 7 of the 92 datapoints fall outside of the NPTL curve, six of which were from the San Luis Obispo Creek watershed.

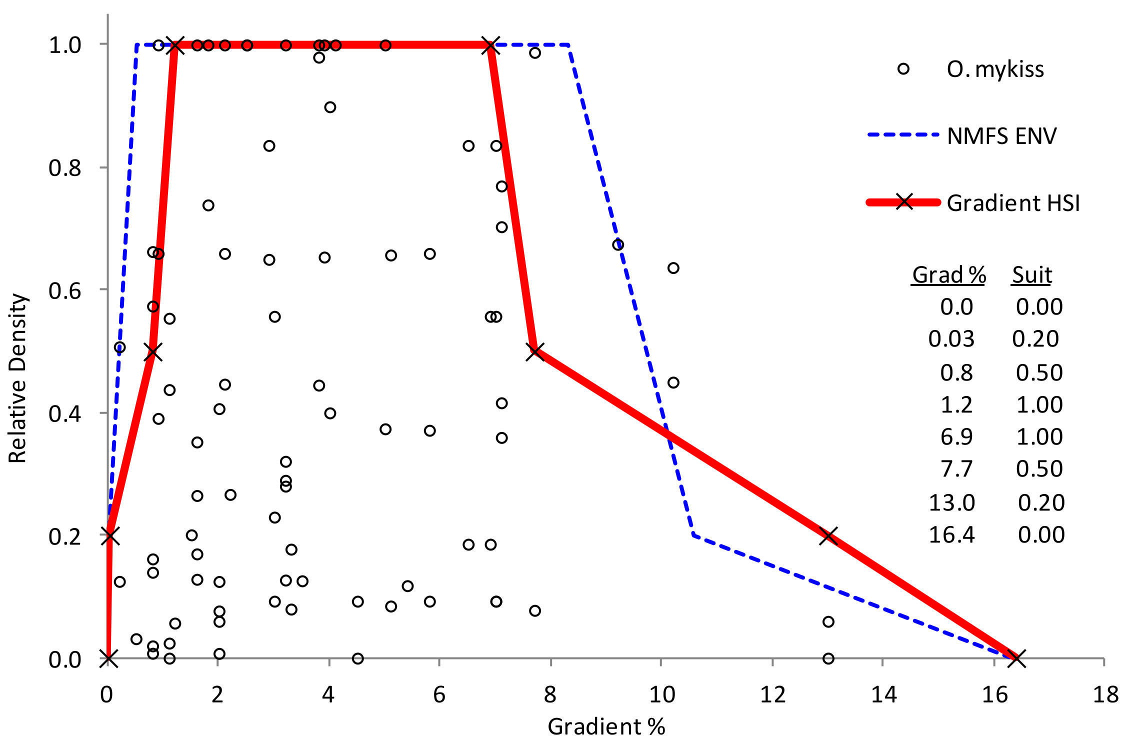


Figure 9. Gradient HSI curve plotted with O. mykiss density data and NMFS envelope curve.

Riparian Shading

Riparian shading is a particularly important parameter in southern California streams due to its influence in moderating water temperatures, which are often a limiting factor in southern California streams. The SS HSI model uses the modified USFWS HSI curve shown in Figure 5, which was modified in order to give greater suitability for higher shading values, despite the potential loss in primary and secondary (invertebrate) production.

Benthic Macroinvertebrate Index

Benthic macroinvertebrates are a primary constituent of *O. mykiss* prey items and are also a significant indicator of stream health. An index of benthic macroinvertebrate diversity and abundance, or BMI, is routinely used as an Index of Biological Integrity (IBI), and consequently BMI surveys have been conducted throughout southern California. These surveys have yielded a southern California benthic IBI (B-IBI) score which is scaled between 0 and 100. These scores can thus be easily transformed into a 0 to 1 scale to represent an HSI variable.

Although the USFWS HSI model includes a food subcomponent intended to represent potential fish food production, comparison of calculated food HSI scores from eight Ventura Basin study sites with actual B-IBI scores (2001-2005 means) from nearby benthic sites (ABC 2006) showed no relationship (R^2=0.00). Similarly, no relationship was evident between calculated food HSI scores and *O. mykiss* abundance in those same study sites, which suggests that the USFWS HSI food subcomponent is not effective in representing BMI abundance nor is it correlated with fish abundance. In contrast to the HSI food subcomponent scores, the mean B-IBI scores did show a positive relationship with mean (2007-2012) *O. mykiss* densities in the eight comparative Ventura study sites (R^2=0.45, P=0.07).

Because invertebrate abundance is expected to directly influence *O. mykiss* abundance, in contrast to surrogate habitat variables that may have less explanatory value, the BMI variable in the SS HSI model uses, wherever possible, a site-specific B-IBI score rescaled between 0 and 1. In the Ventura Basin, mean B-IBI scores were available from locations in close proximity to eight of the 14 HSI study sites (ABC 2006). B-IBI scores for the remaining 6 sites were borrowed from the nearest or most similar study site for calculation of SS HSI scores.

Where site-specific BMI data is not available for use in a SS HSI study, a surrogate value may be estimated based on applicable habitat and/or water quality data. As noted above, however, the food subcomponent in the USFWS HSI model did not correlate with mean B-IBI scores previously estimated from nearby sites in the Ventura Basin. Also, comparison of mean B-IBI scores with habitat and water quality data collected concurrently from those same sites (which is included in the bioassessment protocols) in the Ventura River Basin (ABC 2006) showed relatively little correspondence (R^2=0.11, P=0.42). Finally, a limited attempt was subsequently made to relate B-IBI scores to the new habitat variables collected in 2012, however no usable models were produced. Although more research is warranted, these results suggest that a habitat-based BMI variable may be difficult to develop. Consequently, site-specific B-IBI surveys are recommended for use in the SS HSI model.

Predation

Predation can exert significant effects on *O. mykiss* abundance, particularly for fry and juveniles (Brown & Moyle 1991, Nakamoto & Harvey 2003). Predacious exotic species are relatively common in warm mainstem reaches of southern California steelhead streams, including portions of the Ventura Basin. Large slow-velocity pool habitats, whether natural, beaver-formed, or man-made (e.g., ponds, diversion pools, etc.), are particularly favorable habitats for most piscivorous fish species. The primary fish predators of fry and juvenile steelhead in southern California streams include largemouth and smallmouth bass (*Micropterus salmoides and dolomieui*, respectively) and bullheads (*Ictalurus* spp.). Green sunfish (*Lepomis cyanellus*) and bullfrogs (*Rana catesbeiana*) may also be significant exotic predators of *O. mykiss* fry.

The extensive development of reservoirs and ponds in southern California basins suggests that predation may be an important component in assessing habitat suitability for steelhead, however quantitative data is generally lacking for relating the effects of predator densities on densities of *O. mykiss* fry or juveniles. Consequently, the SS HSI model contains a qualitative predation variable based on professional judgment. This variable and its associated HSI curve, like several others in the SS HSI model, is highly subjective and should be carefully evaluated for each application. Significant refinement of the predation HSI curve may be required as this model is utilized in different basins and stream reaches.

The predation HSI curve (Figure 10) is utilized in both the rearing subcomponent and in the steelhead (smolt) migration subcomponent, and is based on a subjective assessment of densities of exotic fish or frog predators (not including bird or mammal predators). If only smaller exotic predators are present (e.g., green sunfish and bullheads), this predation variable may not be applicable to larger steelhead smolts. Also, the effects of pool-dwelling predators are expected to vary according to stream flow and the availability of riffle habitats. In very small and low-flow stream reaches, *O. mykiss* fry and juveniles are more likely to be forced into direct contact with pool-dwelling predators. In larger channels or higher-flow stream reaches, segregation may limit contact prior to smolt outmigration.

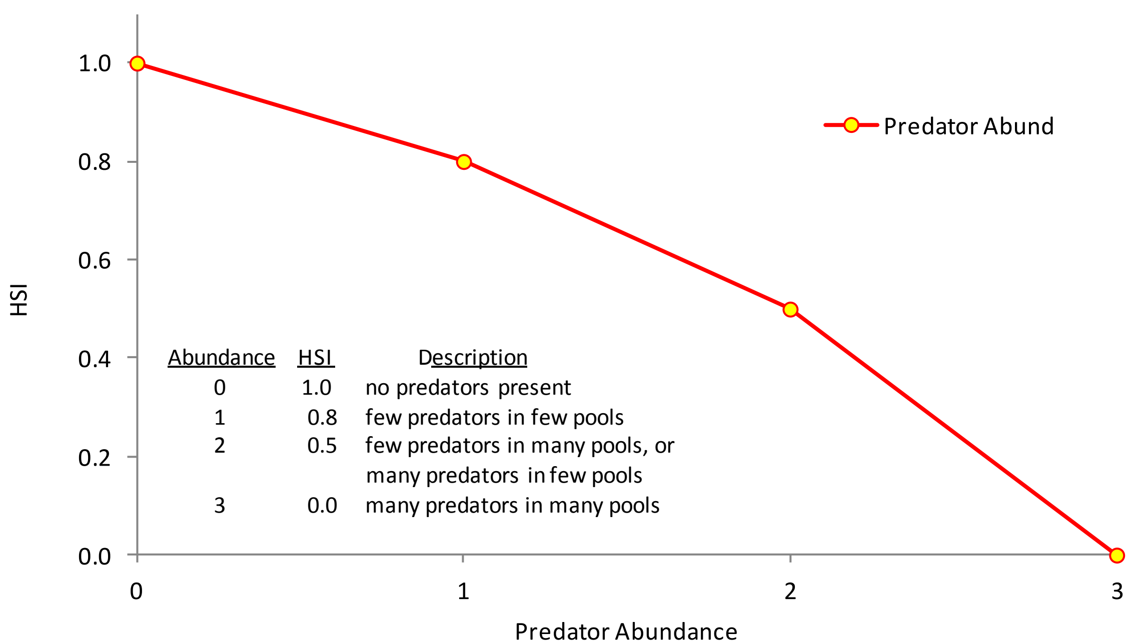


Figure 10. Predation HSI curve for rearing and migration (smolt) subcomponents.

Flow Persistence

The dry Mediterranean climate in southern California steelhead watersheds produces highly intermittent rainfall events and variable streamflow regimes. Most southern California streams not influenced by dam releases contain significant lengths of dry or intermittent channels during the summer and fall low flow periods. Annual variability in rainfall can lead to significant differences in the extent of dry or intermittent reaches. *O. mykiss* fry, juveniles, and adults can survive short periods of isolation in intermittent pool habitats if water temperatures and other water quality parameters remain suitable, but such fish are particularly prone to predation by exotic aquatic predators or by native bird or mammal predators. Stream reaches that frequently become dry or intermittent during low flow periods typically contain little or sparse riparian vegetation, thus exacerbating water temperature problems. Also, some southern California watersheds, including the Ventura Basin (Minear 2003), are subject to mineralization of bottom sediments. This process can cement the substrate which then reduces the suitability for spawning and also appears to inhibit invertebrate production. Such tufa deposition is most severe in open channel areas subject to intense sunshine and low flows. Although intermittent stream reaches are known to provide an important function for spawning and early rearing in some locales (Allen 1986, Boughton et al. 2009), several of the annually intermittent stream reaches in the Ventura Basin, and likely other southern California basins with high mineral content, do not appear to provide high quality habitat for spawning, nor for over-summer rearing even during wet years.

Adequate streamflow is obviously a critical component of suitable steelhead habitat, however the SS HSI model does not include flow magnitude as an HSI variable because in many watersheds the highest densities of *O. mykiss* occur in the headwater tributaries, where the volume of flow magnitude is typically lower than in downstream reaches which contain greater flow but lower densities of *O. mykiss*. The USFWS HSI model recognized this and utilized a single variable to represent rearing flow (*V14*) and another variable to represent migration flow (*V18*). Both of the USFWS flow variables utilized ratios between mean flow over specific time periods (e.g., summer rearing or winter/spring migration) and mean annual flow (Table 3). Estimating these ratios was often difficult due to a lack of site-specific discharge time series data, particularly for smaller tributaries where most *O. mykiss* over-summer. For the SS HSI model, three alternative variables were derived to represent the stability of flow in a given stream reach. The first of these variables, flow persistence, is a qualitative variable that, like the predation HSI curve above, may need revision as this model is tested in different watersheds.

The flow persistence HSI variable is a simple linear curve that gives maximum suitability to perennial stream reaches, zero suitability to consistently dry (or severely intermittent) reaches, and intermediate suitability based on the probability of maintaining surface flow through the summer and fall months (Figure 11). Estimating this probability may be via a qualitative method, such as reviewing time series of aerial photographs (e.g., using Google Earth timeline images), data from nearby stream reaches possessing similar flow characteristics, interviews with local residents, or combinations of the above. In many cases the presence or absence of riparian vegetation along a specific reach will indicate the likelihood of surface flow. Quantitative estimates may be acquired from long-term field studies or from streamflow gaging stations (if present). In highly studied basins, groundwater models may be capable of estimating the recurrence interval of surface flow conditions.

It should be noted that steelhead are adaptable fish and in wet years may utilize reaches that are typically dry (although Ventura data suggests *O. mykiss* densities remain very low); thus this HSI variable is meant to be a measure of longer-term suitability and may not reflect suitability or potential *O. mykiss* abundance over a single wet or dry year.

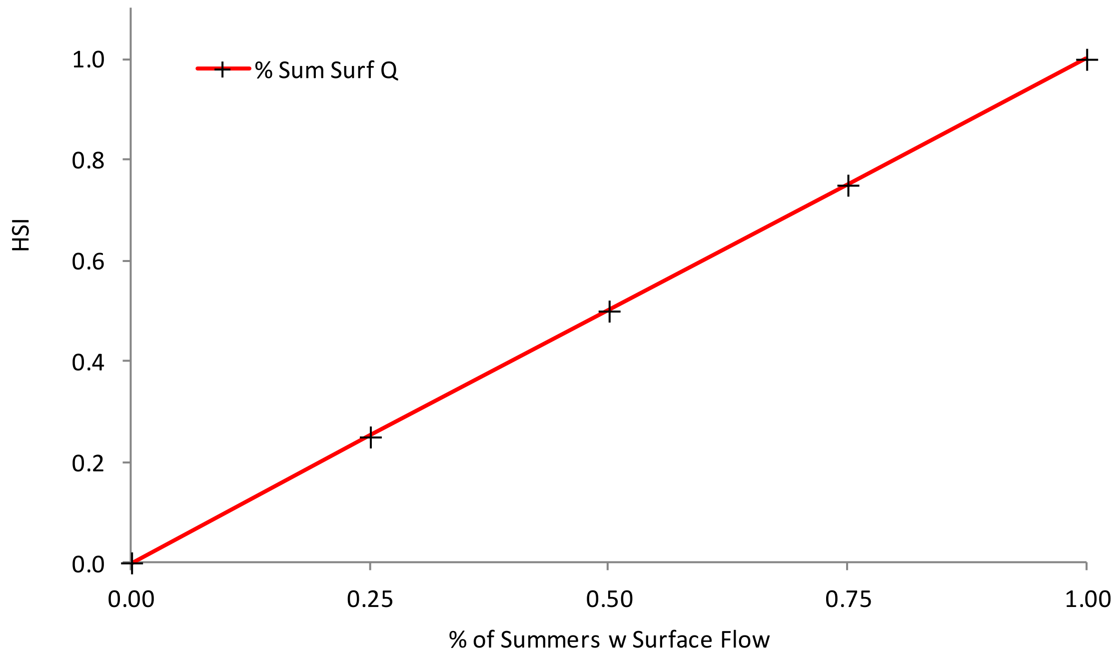


Figure 11. Flow persistence HSI curve for over-summer rearing.

Valley Width

Changes in valley width, or valley confinement, have been shown to have significant effects on surface flow characteristics in many watersheds. A downstream constriction in valley width may force cooled groundwater towards the surface, whereas an expansion in valley width may result in a lessening in surface flow as water percolates into the larger area of alluvium. In Santa Paula Creek, an adjacent watershed to the Ventura River, local-scale changes in valley width was found to be associated with local differences in flow volume and water temperature (Sloat and Osterback 2013). Metrics describing valley width were also used by Dvorsky (2000) to help explain the occurrence of steelhead in nearby Sespe Creek. The Santa Clara River, located immediately south of the Ventura River, has a prominent series of valley expansions and contractions that highly influence surface flow conditions throughout the low flow period. This effect of changes in valley width, in addition to expected changes in the elevation of subsurface bedrock, appears to explain in part the variability in surface flow characteristics in several reaches of the Ventura River Basin.

Seven of the 14 study sites in the Ventura River Basin are spatially associated with nearby changes in valley width that may in part explain observed differences in surface flow characteristics and, thereby, *O. mykiss* abundance. The Ven 3 study site is frequently referred to as the "living reach" due to its perennial, cool surface flows. Immediately upstream of the Ven 3 site is a six mile stretch of channel that remains dry throughout much of the year except during storm events or following particularly wet water years. Although the presence of the "living reach" may be largely explained by a subsurface bedrock sill that forces groundwater towards the surface, this area is also characterized by converging valley sides (these two geomorphic features are likely related). An estimate of valley width three-quarters of a mile upstream of the Ven 3 study site is approximately 2,000 ft, whereas the valley width near the top of Ven 3 (just above the San Antonio Creek confluence) is only about one-third the width, at 750 ft. The opposite trend occurs in the downstream direction, where the valley width increases again to over 2,000 ft. In some years, the mainstem Ventura River becomes intermittent between the Ven 3 study site and Foster Park, but it returns to perennial flow through Foster Park where valley width narrows again to less than 1,000 ft.

A similar scenario occurs just above the top of Mat 5, where valley widths decrease and surface flows emerge from the dry channel just upstream (but above the Murietta confluence), but valley widths increase again towards the bottom of the study site where flows frequently become intermittent. In like manner, the perennial SACup study site is located in an area of converging valley slopes just downstream of a wide, intermittent reach, whereas the frequently intermittent SACmid study site is located in an area of expanding valley width. Valley widths narrow markedly from the top of the UNF study site (at 700 ft), where flow emerges from the dry channel upstream, to less than 100 ft in the area of perennial flow near the reach bottom. Similar relationships are evident with widening valley widths and decreasing flows below the Ven 5 and Mat 3 study sites.

These valley width:flow patterns were assessed by approximating valley widths at 1,000 ft, 2,000 ft, and (for mainstem reaches) 3,000 ft upstream and downstream of the upper boundary of each study site, using ArcGIS to measure the lateral distance perpendicular to the channel where elevation increased 5 meters above the channel low point. The elevations were plotted against distance from the study site boundary and separate regression slopes were calculated for the upstream and downstream directions (Figure 12, top). The slopes were then plotted by study site and visually assessed to estimate what slope appeared to be consistent with observed changes in flow, with emphasis on the seven study sites described above (Figure 12, middle and bottom). Inspection of these plots suggested that positive slopes of 0.05 or greater (indicating a narrowing valley) were likely to produce or maintain surface flows and thus was given a suitability of 1.0 (Table 6). Negative slopes of 0.05 or greater (indicating a widening valley) were likely to result in lessening flows and were given a suitability of 0.5. Slopes between 0.05 and -0.05 were given a suitability of 0.8. A combined study site score was determined by an average of the upstream slope HSI and the downstream slope HSI, with double weight to the upstream HSI. Consequently, maximum suitability (1.0) was attributed to a study reach that was located where the valley width narrowed immediately upstream of the reach and continued to narrow through the study reach. Minimum suitability (0.5) occurred where the valley width widened both above and within the study reach.

Table 6. Valley width HSI values for upstream and downstream portions of study sites. Overall reach HSI score is calculated as ([HSI upstrm*2 + HSI dwnstrm]/3).

Valley Width	HSI	Slope
Narrowing	1.00	> 0.05
No change	0.80	-0.05-0.05
Widening	0.50	< -0.05

Although the related changes in depth of underlying bedrock may have more direct effect on surface flow conditions than valley width, the latter metric can be easily estimated using GIS analysis, whereas estimated depths of alluvium are not readily available for many basins and would require specialized studies to assess. Future assessment of the SS HSI model should include refinement of this valley width parameter and its associated slope criteria.

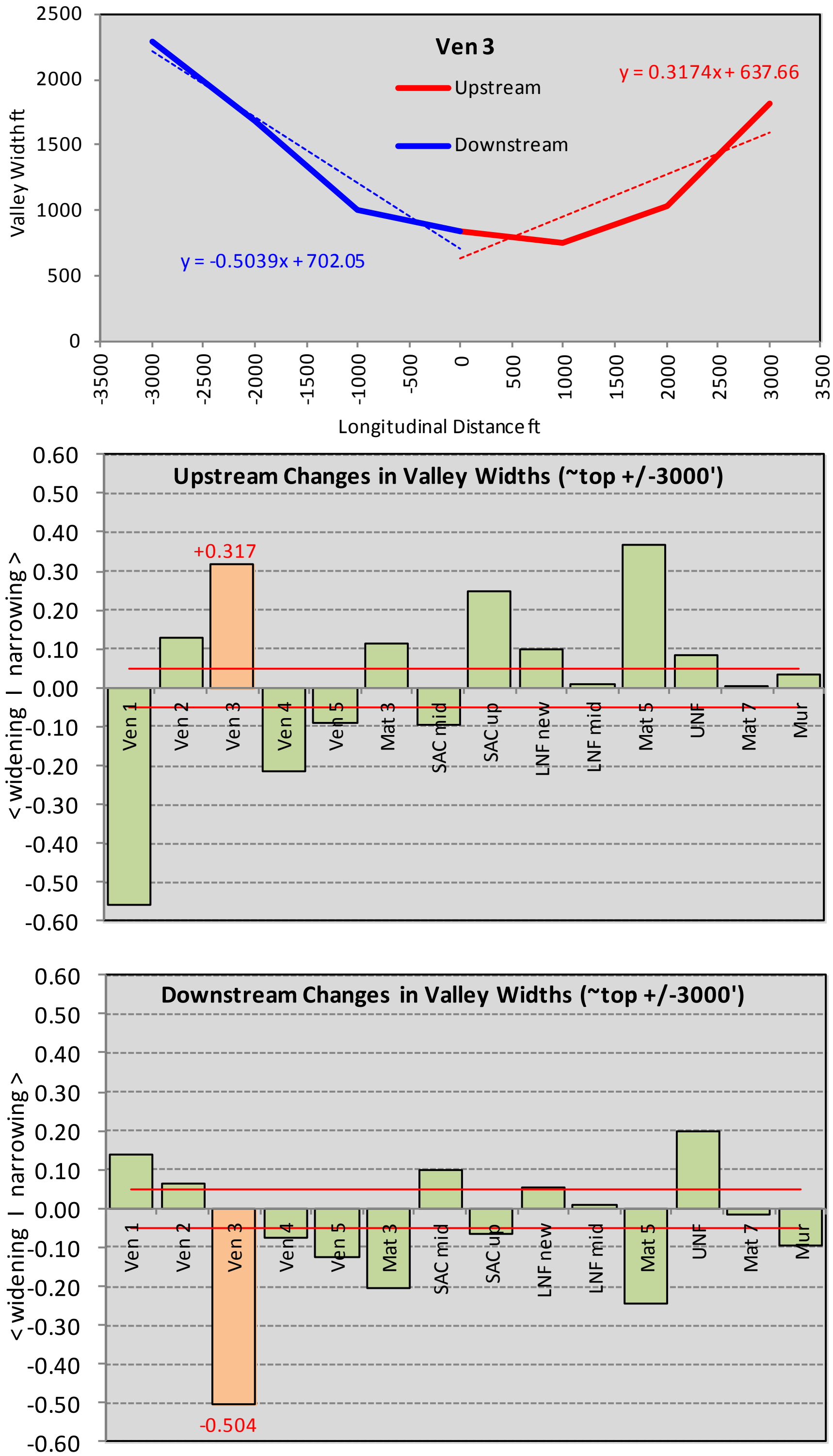


Figure 12. Determination of valley width HSI criteria. Top figure shows upstream/downstream valley widths and slopes for the Ven 3 study site; middle figure shows upstream slopes for all study sites; lower figure shows downstream slopes for all study sites (Ven 3 site highlighted).

Tributary Accretion

This final flow-related HSI variable for the reach component is a simple parameter that gives a maximum suitability of 1.0 for any mainstem (or non-spawning tributary) study reach that contains a confluence with a perennial tributary, given its expected effect on surface flow characteristics, or zero if no confluence is present. This value is also set to zero for study reaches within a perennial tributary, which effectively removes this variable from affecting the reach component HSI (due to the equation used to calculate the reach HSI value, see Section 4.3.6).

4.3.3 Recruitment Component

The USFWS HSI model included an embryo component which contained several variables associated with spawning habitat quality, and two water quality variables associated with successful egg incubation. The recruitment component of the SS HSI model uses a similar form and directly relies upon the USFWS HSI curves for several variables, including gravel quality, incubation water quality, and the "tributary effects" variable developed for use in the USFWS HSI model and shown in Figure 6. The SS HSI model also adds a gravel quantity variable.

Gravel Quality

Gravel quality is assessed using a weighted mean of the USFWS' *Vs* parameter, which is calculated from HSI curves for the average water velocity (*V5*), average particle size (*V7*), and percentage of fines (*V16sp*) within patches of spawning gravels (Figure 5), weighted by patch size. In the SS HSI model this variable does not include the USFWS restrictions on cumulative gravel quantity, or the mean *Vs* division by total habitat area; instead gravel quantity is treated separately in the following variable. As noted in Section 4.2.2, the spawning velocity HSI curve was modified from the original USFWS curve.

Gravel Quantity

The suitability of a particular stream reach to support recruitment of *O. mykiss* fry is dependent in part upon the quantity of spawning gravels, not just the quality. The USFWS HSI model incorporates a measure of gravel quantity, however that measure was not utilized in the calculating the embryo component of the previous Ventura Basin HSI scores (only the mean quality was used). Consequently, the SS HSI model reinserts a gravel quantity component based on the percentage of the wetted area (during typical spring spawning flows) that is composed of potentially spawnable gravel, using the range of suitable particle sizes, water velocities, and % fines illustrated in Figure 5.

The HSI curve for gravel quantity maximizes at 3% of wetted area and remains at 1.0 for greater densities (Figure 13). This criteria was subjectively chosen based on the estimates of gravel densities in the five study sites judged to have the most optimal spawning habitat in the Ventura Basin (in bold red font, Table 7). Each of these sites contains approximately 3% or more of potentially suitable spawning gravels. Note that this criterion is slightly less than the 5% criterion described in Raleigh et al. (1984), consequently future validation studies should evaluate the appropriateness of this value.

Table 7. Percent of wetted area containing potential spawning gravels.

Study Site	% Spawning Gravel
Ven 1	0.3
Ven 2	0.4
Ven 3	**2.7**
SACmid	2.4
SACup	**5.2**
Ven 4	0.1
Ven 5	0.5
LNFnew	**3.1**
LNFmid	**4.2**
Mat 3	0.6
Mat 5	1.9
Mat 7	0.5
Mur	1.5
UNF	**7.1**

Incubation Water Temperature and Dissolved Oxygen

The SS HSI model utilizes the same HSI curves for egg incubation water quality (Figure 5), including the modification to the incubation temperature curve. As previously stated, the temperature modifications to the incubation (and other temperature curves) were subjective and should be validated with additional data, preferably using *O. mykiss* stocks from southern California.

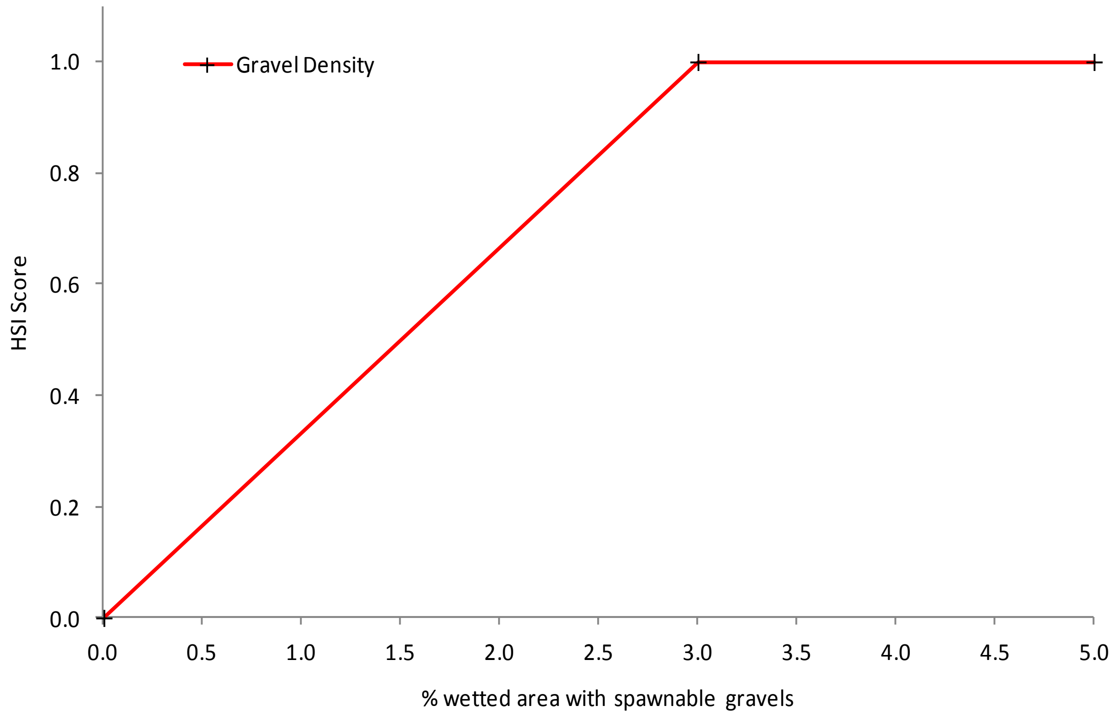


Figure 13. HSI curve for gravel quantity.

Tributary Effects

The tributary effects HSI curve (Figure 6) was developed from Ventura Basin data for use with the USFWS HSI model, and is described in detail in Section 4.2.2. This variable was added to the USFWS HSI and incorporated into the SS HSI model to help account for recruitment of fry from nearby spawning tributaries into mainstem reaches.

4.3.4 Water Quality Component

The water quality component includes variables primarily associated with over-summer rearing. Water quality parameters associated with spawning and egg incubation were treated in the recruitment component discussed above, and water quality for steelhead adult and smolt migration is addressed separately below. The three variables included in the SS HSI models water quality component utilize the same HSI curves as the USFWS HSI model for rearing temperature (*V1r*), rearing D.O. (*V3r*), and rearing pH (*V13*), including the modifications made to the rearing temperature curve (Figure 5) which, as stated for other variables, should be re-assessed in future HSI studies.

4.3.5 Migration Component

The USFWS HSI model did not put much emphasis on the suitability of migration conditions for the steelhead life form, but instead only included water temperature criteria for migrating adults and smolts, and a flow ratio criterion to represent passage conditions. Because both water quality and passage conditions can be highly limiting in southern California basins, the SS HSI model also utilizes the migration temperature HSI curves and adds several passage-related variables. The adult steelhead migration HSI sub-component includes migration distance (to study reach), migration temperature, and three passage-specific variables (lagoon opening, riffle depths, and migration barriers). The smolt migration HSI sub-component also uses migration temperature and migration distance, and adds predation.

Distance

The migration component HSI scores for both adult steelhead and smolts utilizes a distance factor which determines the suitability of a particular reach based on its distance from the ocean. The distance HSI is based on an arbitrary model that assumes a decline in suitability of approximately 1% per mile (Figure 14). This rate of decline was based on an initial assumption that one-third of all migrating steelhead (smolts or adults) would survive over one of the longest potential migration routes in southern California (upper Piru Creek, assuming no dams), or 33% over 100 miles. According to this assumed mortality rate, survival to or from the upper headwaters (again assuming no man-made barriers) of shorter southern California watersheds, such as the San Juan, San Mateo, Santa Paula, and Ventura, would be approximately 75%. The longer migration lengths for the Sespe, Santa Ynez, or Sisquoc would produce an estimated survival and HSI of about 50% to the upper headwaters.

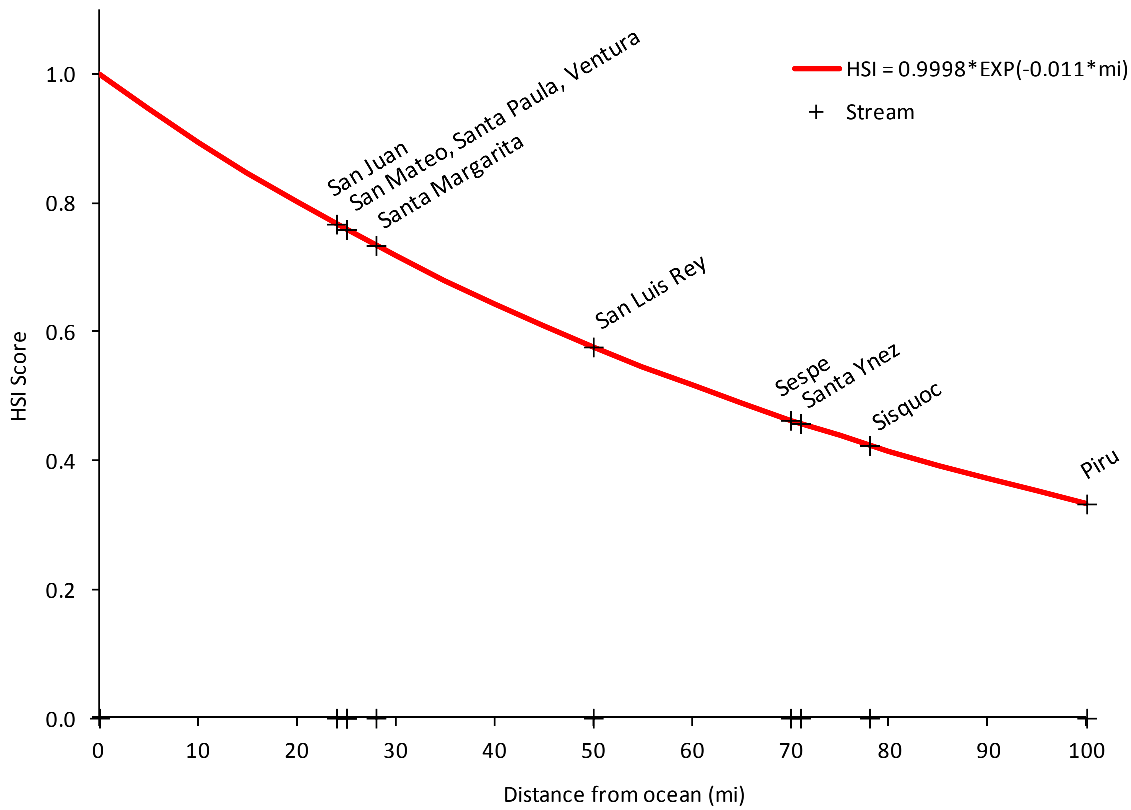


Figure 14. HSI curve for migration distance, showing the approximate distance to headwaters of several southern California basins.

This mortality rate is undoubtedly much higher than rates observed on larger, northern streams, but given the short duration of most migration flow events in southern California, it is likely that migration to headwaters of larger basins will be interrupted by potentially lengthy periods of low flows with limited ability to continue migration (particularly for larger adult steelhead), with its associated risks. Because this migration distance relationship is not based on actual migration data from southern California steelhead streams, this HSI curve should be re-evaluated as data becomes available.

Migration Temperature

The migration component of the SS HSI model utilizes the modified water temperature curves for adult steelhead upstream migration (*V1b*) and smolt downstream migration (*V2a*), as illustrated in Figure 5 and described in Section 4.2.2.

Lagoon Opening

The sand berms that form across most southern California lagoons must be open for adult steelhead to initiate its spawning migration. Lagoon berms typically open following significant storm events, however the frequency and duration of such openings will vary considerably between basins and between years. The lagoon opening HSI curve is a simple 1:1 linear relationship based on the estimated percentage of time within the adult migration season that the lagoon is open to the ocean (Figure 15). Many lagoons in southern California are routinely monitored, but for some lagoons this parameter may need to be estimated using data from another monitored lagoon with similar opening dynamics. One potential change to this HSI curve that might produce a more realistic suitability relationship would be to estimate the percent opening only during migration flow events, since most upstream migration is likely to be closely linked with a period of elevated flows.

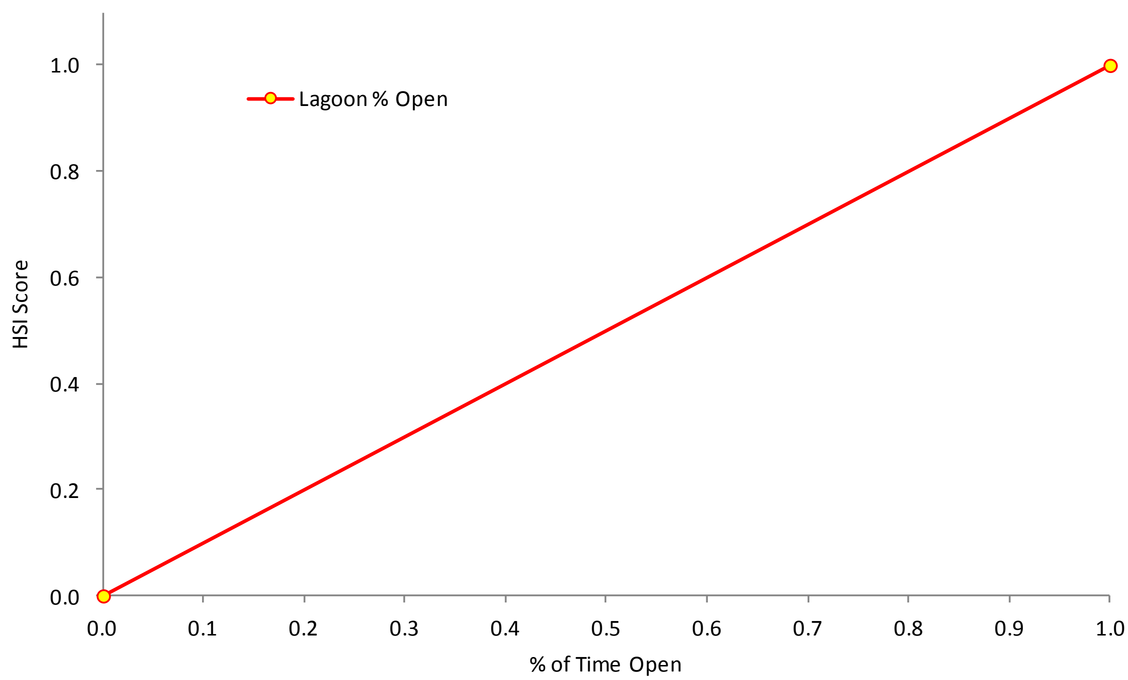


Figure 15. Lagoon opening HSI curve.

Although an open lagoon is also requisite for smolts to complete their downstream migration, a closed lagoon could provide extended rearing until the next available opening, or else smolts could re-ascend the river and follow a resident trout pathway. Consequently, the SS HSI model currently does not include this variable for assessing the juvenile (smolt) HSI component.

Riffle Depth

Riffle depths may be limiting to adult upstream migration in many southern California streams due to extended periods of low flow, even during the winter/spring migration period. Ideally, the suitability of a particular stream reach would be assessed by conducting a formal riffle passage analysis at identified critical riffles downstream of the study reach, using protocols such as a critical riffle analysis (Thompson 1972, CDFG 2013), 1-D hydraulic modeling (TRPA 2005b, Stillwater 2012), or 2-D hydraulic modeling (Holmes et al. 2015, Grantham 2011). Combining a critical riffle analysis or a hydraulic model, which will estimate minimum flows for passage, with a flow duration analysis encompassing the migration period, would allow estimation of the percentage of time that passable flows are available. This estimate could then be scaled for use as an HSI variable. For example, in the productive and largely pristine Big Sur River, passage flows were predicted to naturally occur for over 50% of the period of adult upstream migration (Holmes et al. 2015). Given the more Mediterranean climate of southern California coastal streams, naturally occurring passage flows in steelhead basins such as the Ventura are likely to occur at a much lower frequency; thus scaling the flow duration relationship with riffle depth to a habitat suitability index should be validated with southern California data.

Alternative methodologies requiring minimal or no field assessments, include the regional regression formula (SWRCB 2014), which utilizes a formula (derived from upon northern California streams) incorporating drainage area and unimpaired mean annual flow at each point of interest. The riffle crest thalweg methodology (McBain and Trush 2013) utilizes depth measurements at the thalweg of multiple riffle crests. These measurements taken over a range of flows can provide a rough estimate of passability over defined riffle crests.

In lieu of a formal critical passage and flow duration analysis, the SS HSI model employed two alternative riffle depth HSI curves that were developed in a manner similar to the riffle crest thalweg methodology, except riffle thalweg depths were taken throughout the length of the riffle rather than at a single point at the riffle crest. If critical riffles are identified within or downstream of the study reach, and can be assessed during one or more migration flows, the HSI curve sets maximum suitability for a mean critical riffle thalweg depth of 0.6 ft, with zero suitability for a mean thalweg depth of 0.4 ft (Figure 16, blue dashed line). The maximum suitability depth of 0.6 ft is based on criteria initially proposed for adult steelhead by Thompson (1972), but is slightly shallower than the criteria (0.7 ft) adopted by the CDFW SOP (CDFG 2013). The above criteria are intended to provide for unimpeded passage, but adult steelhead are commonly observed to successfully negotiate depths significantly less than the criteria listed above; consequently the shallower depth of 0.6 was selected for this HSI curve.

Note that a single measurement of critical riffle depths will not allow assessment of the flow necessary to allow passage, it will only allow assessment if the measured flow is likely to allow passage. Also note that this method does not account for the length of critical riffles, which is an important parameter not addressed in either the traditional "Oregon Method" passage analysis (Thompson 1972) or the CDFW SOP. Neither does the riffle depth HSI curve address channel width, which is also not adequately treated by the above two protocols. For small headwater spawning tributaries, a continuous channel through a critical riffle that also meets depth criteria need only be wide enough for an adult fish to pass through (say, one to two feet).

The alternative riffle depth HSI curve is a subjective curve based on the average thalweg depth of riffles during the summer low flow period, which goes from zero suitability at a depth of 0.5 ft to maximum suitability at 1.0 ft (Figure 16, red line). This curve was based on 95 measured riffle depths in 13 reaches of the Ventura River, with a subjective assessment of how likely an adult steelhead could migrate through those riffles. Using summer riffle depths to assess passage during higher migration flows is obviously a speculative process, however being on-site to measure depths during an actual passage event is particularly difficult in southern California due to the highly intermittent nature and short duration of most migration flows. Consequently this alternative curve, based on data collection during low, stable flows, is offered as a logistically convenient alternative, but the appropriateness of this HSI curve should be carefully evaluated in future HSI studies. A comparison of summer riffle depths in a variety of tributary streams known to be consistently accessible (or else not accessible) to adult steelhead could be used to validate or modify this curve.

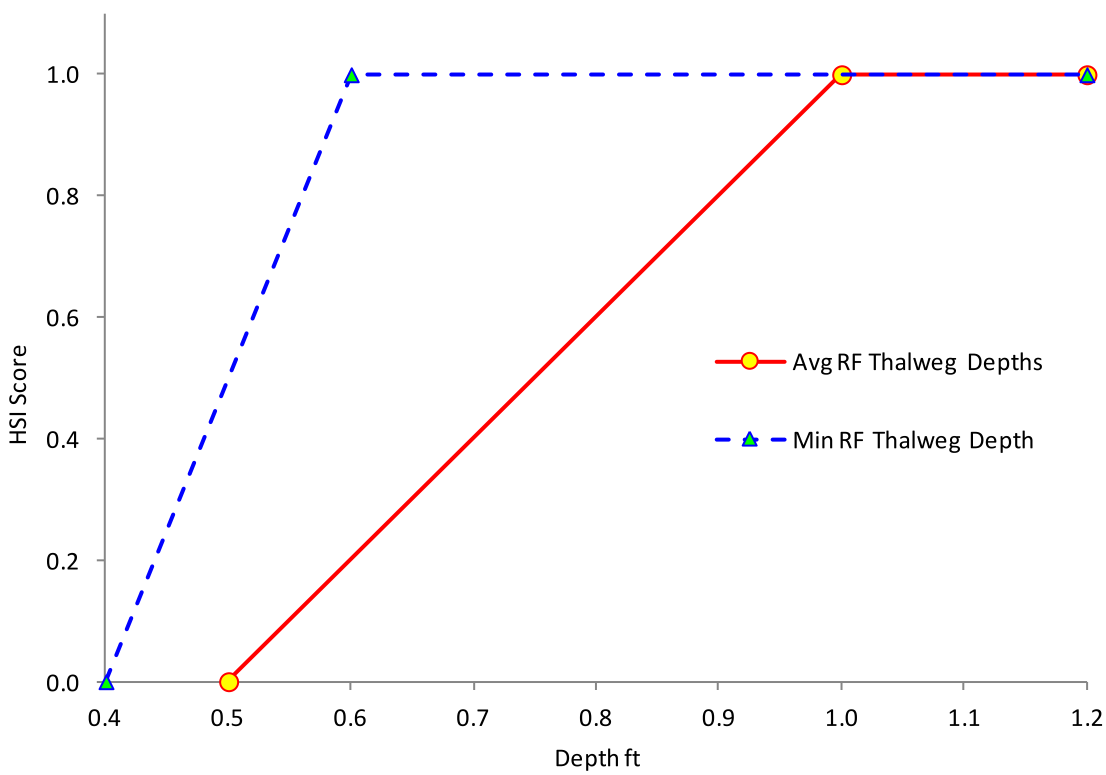


Figure 16. Riffle depth HSI curves for passage of adult steelhead, based on minimum depth of critical riffle thalwegs during migration flows (blue line), or mean thalweg depths of typical riffles during the summer low flow period (red line).

Vertical Barriers

This HSI variable is intended to represent the cumulative probability of adult upstream migrant steelhead successfully ascending vertical passage impediments such as waterfalls, steep cascades, dams, or culverts (with or without jumps). Wherever possible, a formal barrier assessment should be conducted that will estimate what flows, if any, would provide passage over the barrier. For example, Fish Xing software (USFS 2006) can be used to estimate the range of passage flows for culvert barriers. Next, a flow duration curve could be derived to estimate the probability that passage flows will occur in a given year (during the migration season), which could then be rescaled to represent an HSI score. Formal analyses could also be conducted for dams, ladders, or other design structures to similarly estimate passage probability.

Various assessment methodologies have been developed for predicting passage over natural barriers, such as steep cascades, chutes, and falls. Most such analyses, including the quantitative methods described in Reiser et al. (2006) or the more qualitative assessment in Bain and Stevenson (1999), rely heavily on fish performance data described in Powers and Orsborn (1985). As noted above, repeated measurements at multiple flows will help to predict what range of flows, if any, would allow passage over a given barrier. Combining that data with flow duration data would yield an estimate of the probability of suitable passage flows occurring at that site in a given year.

A qualitative and field-friendly protocol based on Powers and Orsborn (1985) leaping curves for steelhead (see their Figure 7) utilized a modified jump chart for evaluating natural barriers in the Ventura Basin (TRPA 2003). The jump chart (Figure 17) represents a composite of vertical and horizontal jump distances for adult steelhead in “bright” and “good” condition, where barriers falling within the “passable” area should be manageable by fish in either condition, and barriers within the “possible” area are most likely to be ascended by “bright” fish fresh from the ocean. The jump chart was further partitioned for this study to assign intermediate HSI scores based on the estimated probability of passage.

In addition to jumping distances, successful passage over a natural feature is also dependent upon depth of the jump pool and velocity at the barrier crest (for jumping barriers), or chute velocity for swimming barriers. Optimal jump pool depth has been described as at least 1.25 times the total barrier height (Stuart 1962); whereas minimal pool depths are characterized as being at least equal to fish length (e.g., 2 - 3 ft for southern steelhead) and deep enough such that plunge pool turbulence does not extend to the pool bottom.

The barrier crest where a fish is expected to re-enter the water must be at least as deep as the fishes body depth (~0.4-0.6 ft) and the crest water velocity cannot exceed a steelheads maximum burst swimming speed, which has been cited as ranging from 13-27 fps, depending on fish size and physical condition (Reiser et al. 2006, USFS 2006). The higher burst rates may be more appropriate for large, “bright” fish encountering barriers reasonably close to the ocean, whereas the lower burst speed might represent small adults or fish at barriers far from the ocean. Additional barrier characteristics that effect passage success, and should be considered in estimating the probability of passage, include the presence and location of a standing wave (which is where the horizontal jumping distance is measured from), and the amount of turbulence at the jump site. Standing waves will enhance jumping ability if not too distant from the barrier crest, whereas turbulence and (especially) whitewater will degrade jumping ability.

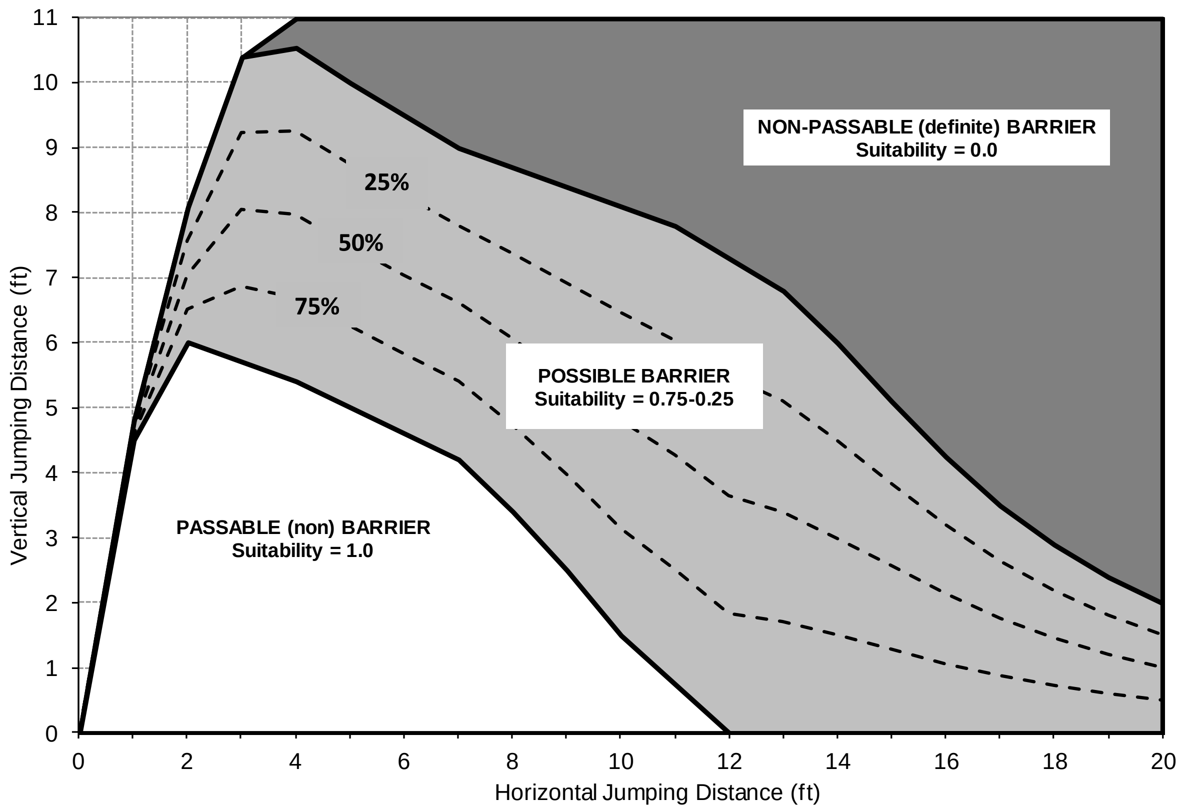


Figure 17. Vertical barrier jump chart showing HSI values based on vertical and horizontal jump distances (adapted from Powers and Orsborn 1985, Figure 7).

The overall vertical barrier HSI score for any point in the watershed is dependent on the cumulative probability of passing all barriers downstream of the study location. This is determined by multiplying the estimated passage probability of a particular barrier by the cumulative probability up to that location, as shown in Table 8. Thus, if all downstream barriers are judged as fully passable, the cumulative HSI score remains at 1.0, whereas if any single barrier is judged as non-passable, the HSI goes to zero for all reaches above that barrier.

Table 8. Calculation of overall HSI score for vertical barriers, based on number and difficulty of down-stream barriers.

Barrier #	Passage Probability	Cumulative HSI
1	0.75	0.75
2	0.75	0.56
3	1.00	0.56
2	0.75	0.42
5	0.50	0.21

Finally, as described for the riffle depth variable, a barrier assessment at a single flow may not provide an accurate estimation of passability. Ideally a barrier would be reassessed at a variety of flows that are typical of conditions during upstream migration in order to determine the range of passable conditions, and then combine that information with hydrology to estimate the probability of achieving those flows during the migration period.

Holding Pools

Adult migrant steelhead are large, conspicuous fish that are highly vulnerable to harassment or predation in the small channels typical of southern California spawning reaches. Consequently, the adult steelhead migration HSI sub-component also includes a variable describing the presence and quality of holding pools within a given reach. Studies describing the habitat requirements of adult steelhead in small, southern California streams are rare, if present. Consequently, this HSI curve is subjective and based on the underwater and bank-side observations of about 16 steelhead-sized (≥40 cm) *O. mykiss* observed in the Casitas Springs reach of the Ventura River during the spring of 2012. One deep pool (eye estimated to be 8 ft deep) contained 4 adult steelhead that milled about in the center of the pool, but all other adult fish were located by snorkeling and were observed holding immediately beneath thick cover comprised of willow or Arundo branches at depths ranging from 2.5 to 6.0 ft.

The adult holding pool HSI curve was subjectively based on the above data which gave higher suitability, at a given depth, for pools with dense cover than pools without cover (Figure 18). Pools containing dense cover reach maximum suitability at 3 ft, whereas pools devoid of dense cover achieve maximum suitability at 8 ft in depth. Note that the "with cover" HSI value is based on the maximum depth measured *beneath* areas of thick cover, whereas the "without cover" HSI value is based on the pools maximum depth away from cover; the pools HSI score is based on the larger of these two values. Calculation of a reach-specific HSI score for holding pools is based on the weighted mean HSI value for individual pools, with weights based on pool dimension (e.g., pool length, surface area, or volume). This HSI relationship for pool depth and cover should be assessed with additional adult holding data from small spawning streams.

Predation

Steelhead smolts are subject to predation during their downstream migration, consequently the smolt migration sub-component incorporates, along with distance and water temperature, the predation HSI curve utilized in the reach component and presented in Figure 10. Although the reach HSI component considered green sunfish and bullfrogs as potential predators for *O. mykiss* fry, aquatic predators on larger steelhead smolts would be limited to larger species, such as adult bass or bullheads.

4.3.6 Calculation of SS HSI Scores

Southern steelhead HSI scores were calculated within each study site for each of the 5 components (4 components for resident trout reaches above Matilija Dam), as well as an overall combined HSI score. These study site HSI scores represented the HSI for its respective study reach (see Section 4.1 for project area stratifications). Overall study segment HSI scores were calculated for each of the 3 segments (below Robles diversion Dam, between Robles and Matilija dams, and above Matilija Dam), by combining the encompassed study reach HSI scores, weighted by reach lengths. The overall SS HSI score for the Ventura River Basin was calculated by weighting each study segment HSI score by the segment lengths. Comparisons of SS HSI or USFWS HSI scores with abundance of *O. mykiss* were conducted using data at the study site scale.

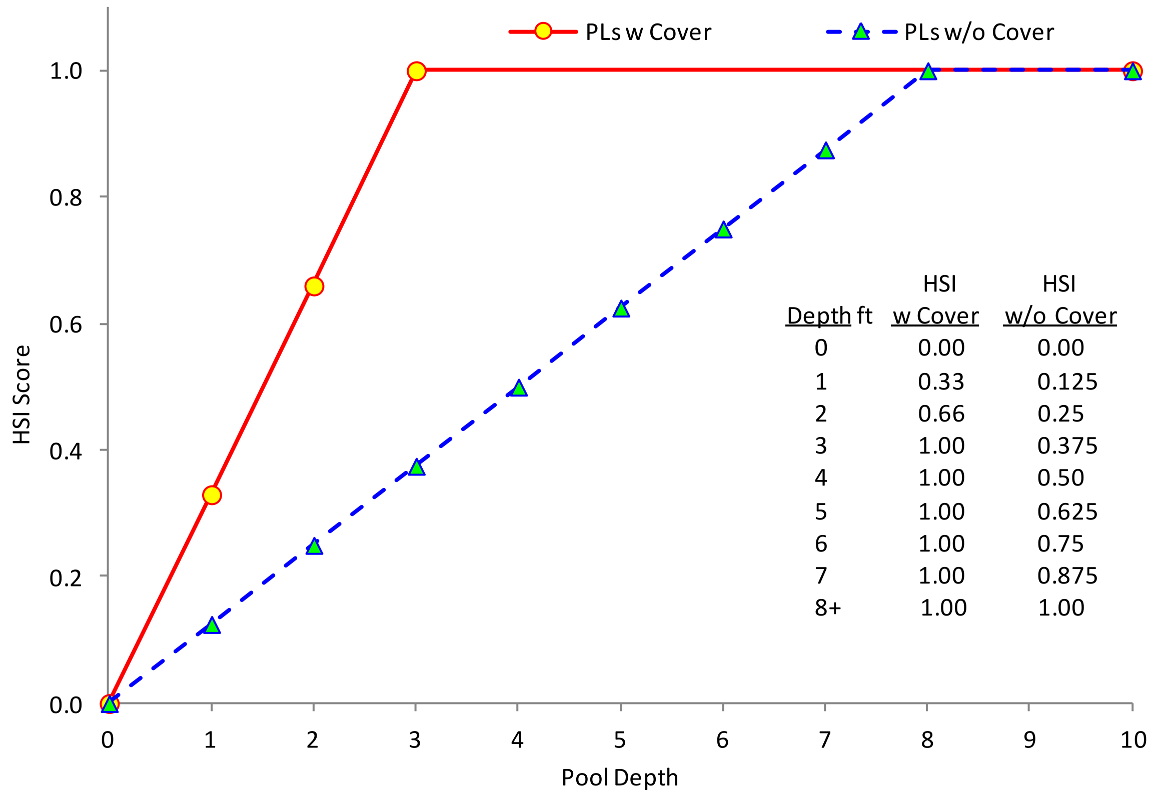


Figure 18. HSI curves for holding pools with or without dense instream cover.

Selection of Model Formulas

The SS HSI model, like the USFWS HSI model, uses a variety of equation types to calculate HSI scores for each component based on that components suite of HSI variables (Table 6). Component scores are calculated from combinations of variable scores using geometric means, minimum values, and maximum values, depending upon how each variable or combination of variables was expected to influence habitat suitability and, consequently, *O. mykiss* abundance. The geometric mean differs from the arithmetic mean by, in effect, normalizing the variable ranges prior to averaging. This reduces the influence of variables having a larger range in values and consequently the geometric mean will typically be lower than the arithmetic mean. However, a geometric mean cannot be calculated if any of the variable scores is zero; consequently for those component equations utilizing a geometric mean, it was sometimes necessary to add a small constant (e.g. 0.001) to a zero value. Minimum values were sometimes used in component equations where one or more of the variables was deemed to be a limiting factor and could not be compensated for by other variables. Where a high level of compensation was expected to occur between several variables, a maximum value was used.

Generally, several equation formulations were tested for calculating each component HSI score, which were then evaluated by comparing the calculated scores for each study site against several metrics. One metric involved comparing the trend in study site component scores against subjective assessments of the relative quality of that component according to study site. For example, the professional judgment of the quality of the reach component (based on gradient, shading, BMI abundance, predation, flow persistence, valley width, and tributary accretion) within each study site was compared to the alternative reach HSI scores for each study site, with the expectation that study sites judged to have higher quality would have higher scores, and lower quality study sites would have lower scores (Figure 19). The component equation producing the most subjectively "realistic" relationship of HSI scores between the different study sites would be most favored for selection.

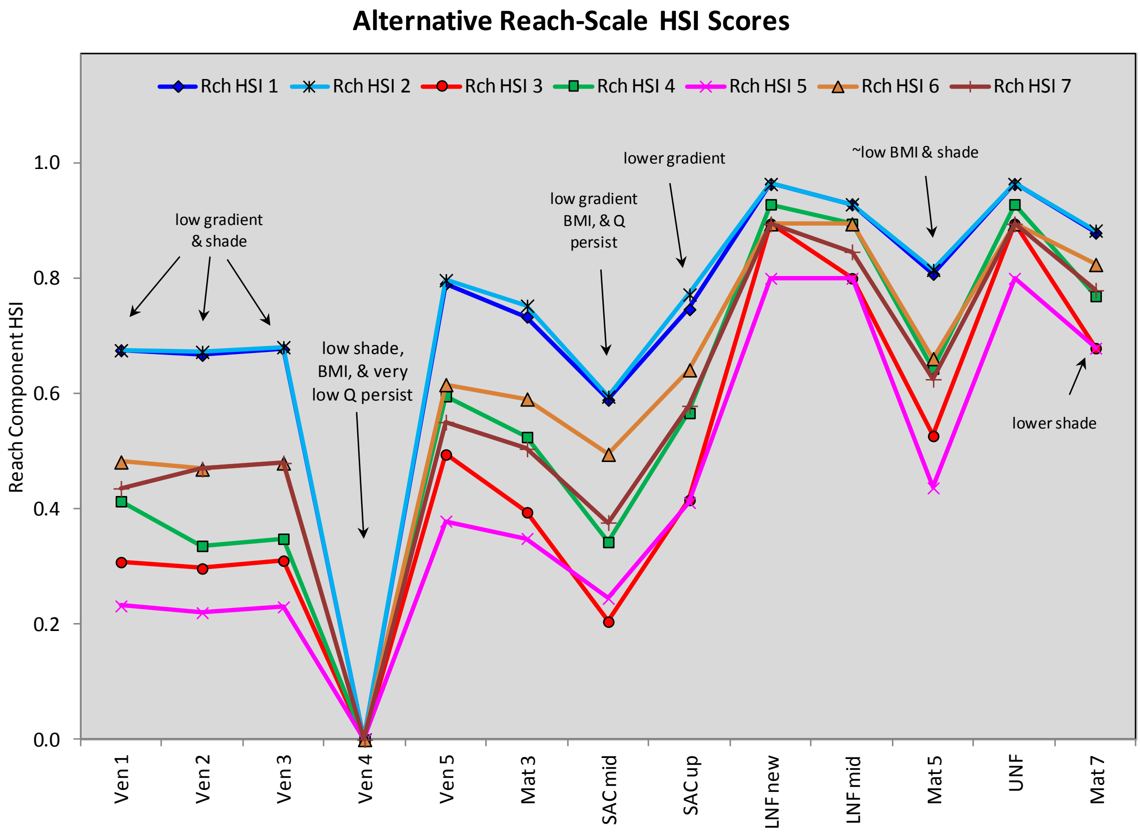


Figure 19. Example plot showing reach component scores by study site according to different equation formulations, along with notes indicating specific variables degrading the scores.

A second subjective assessment method was evaluating the magnitude of the study site component scores, where equations that produced unrealistically high or low component scores, when compared to professional judgment, were less favored for selection. Third, a quantitative evaluation utilized the correlation between study site component scores and estimated densities of *O. mykiss* in those study sites (excluding Murietta Creek as the validation study site). Equations yielding higher correlations with fish densities were generally favored over equations yielding lower correlations, while recognizing that *O. mykiss* densities are also influenced by the remaining components as well as unmeasured factors. These three methods of evaluation were used to select what was deemed the "best" equation formula for a given component.

Calculation of an overall SS HSI score for each study site was also based on comparison of different ways of combining the component scores, which typically included models using a geometric mean of all component scores, or the minimum value of the water quality score compared to a mean of the remaining component scores. These alternative overall SS HSI scores were assessed by plotting the scores for each study site with *O. mykiss* densities and evaluating the relative fit (judged by R^2 values, excluding Murietta Creek), the magnitude of the HSI scores, and the breath of the HSI scores. The Murietta Creek HSI score was plotted separately for visual validation.

Habitat Unit Component HSI Scores

As stated in Section 4.3.1, the habitat unit component HSI scores were derived for each of three life stages (fry, juvenile, and rainbow adult), three habitat types (pools, flatwaters, and riffles), and two channel sizes (mainstem vs. tributary), resulting in 18 combinations (Figure 7). Stepwise multiple regression was used to develop predictive models based on a suite of unit-specific depth, velocity, and cover variables as predictor variables and *O. mykiss* density (#/100ft^2) as the response variable. Murietta Creek data was not used for developing the tributary regression models. Final regression models and associated statistics are given in Table 9, variable descriptions are listed in Table 5.

Table 9. Stepwise regression models used to estimate HSI scores for individual habitat units.

Age Class	Channel Size	Habitat Type	n	R^2	P	Equation
fry	mainstem	flatwater	28	0.69	<0.01	Y=0.4248-0.4975*MXDEP+1.9238*D1+7.8897*V1-7.6863*V1IW
fry	mainstem	riffle	28	0.46	0.03	Y=-5.3886+27.3662*AVDEP-33.609*D1-16.673*AVVEL+26.6686*V1+72.3526*TURB-69.3611*V1OW
fry	mainstem	pool	28	0.60	0.01	Y=0.0209-1.2796*AVDEP+0.2537*MXDEP+1.3231*D1+1.2014*D2-1.8472*AVVEL+3.7884*V05-0.7468*OHVEG
fry	tributary	flatwater	35	0.18	0.04	Y=2.4009+6.1661*IWBR-4.4088*OHVEG
fry	tributary	riffle	30	0.53	<0.01	Y=-0.6973+3.9649*MXDEP+12.9802*V1-33.4037*TURB
fry	tributary	pool	43	0.31	<0.01	Y=0.4711+0.6483*MXDEP-8.9240*D2-5.2072*IWBR
juv	mainstem	flatwater	28	0.37	0.02	Y=-0.1123+0.4476*AVVEL+5.7276*TURB+1.8289*IWBR-6.6759*V1OW
juv	mainstem	riffle	28	0.67	<0.01	Y=-0.8703+2.9899*AVDEP-3.4351*D1-1.0207*AVVEL+1.5659*V1+7.1232*TURB-6.6998*V1OW
juv	mainstem	pool	28	0.60	<0.01	Y=-0.0556-0.5496*AVDEP+0.1112*MXDEP+0.6083*D1+0.8735*D3+0.1644*CB
juv	tributary	flatwater	35	0.43	<0.01	Y=-0.7384+5.2342*AVDEP-1.4229*MXDEP+1.6108*CB+9.8514*TURB+5.4651*IWBR
juv	tributary	riffle	30	0.50	<0.01	Y=-0.157+1.5506*CB+5.7403*IWBR-1.5668*OHVEG-7.77256*V1IW+8.4159*V1OW
juv	tributary	pool	43	0.36	<0.01	Y=-0.302+0.257*MXDEP-0.5759*D1+4.3581*V05+0.5659*CB-8.9895*V05IW
adult	mainstem	flatwater	28	0.33	0.26	Y=0.0464-0.0725*D1-4.6108*CB-4.8291*TURB-4.6684*IWBR-4.6213*OHVEG+4.6442*ALLCOV-0.2159*FINES
adult	mainstem	riffle	28	0.58	<0.01	Y=0.0376+0.5721*D1-0.1524*CB-1.3889*TURB-0.4103*OHVEG+0.2999*V1IW+1.3022*V1OW
adult	mainstem	pool	28	0.29	<0.01	Y=-0.0373+0.0179*MXDEP
adult	tributary	flatwater	35	0.27	0.14	Y=0.2465-0.4655*AVDEP+15.2263*CB+14.4507*TURB+14.8701*IWBR+15.5466*OHVEG-15.226*ALLCOV
adult	tributary	riffle	30	0.35	0.03	Y=0.1552-0.3026*AVVEL-2.0539*TURB-0.3344*OHVEG+2.7548*V1OW
adult	tributary	pool	43	0.13	0.10	Y=0.0225+0.2901*D2+0.6321*AVVEL-1.4042*V05

Converting the regression-predicted *O. mykiss* densities into HSI scores for each habitat unit used the following process. The maximum *O. mykiss* density predicted from each of the 18 models was multiplied by 0.8 to represent the expected density in an "optimal" habitat unit. The predicted densities in all habitat units within a given fish/channel/habitat strata was normalized to that "optimum" value, which resulted in most unit-specific HSI scores falling between 0 and 1. For those units that had predicted densities over 80% of the maximum predicted density, those units were given an HSI value of 1.0. This process assumed that at least one habitat unit in each strata contained "optimum" habitat (based on depth, velocity, and cover).

In the example plot above (Figure 20), the regression model (Table 9) explained approximately 50% of the variation in densities of juvenile *O. mykiss* among tributary riffles. Eighty percent of the maximum predicted density was 1.23 fish/100ft^2. Two of the 30 test riffles had predicted densities exceeding that value, which according to this assumption were assumed to represent optimal riffle habitat in tributaries. The Murietta data points are also shown as a validation of the regression model. In this example, the fit for Murietta Creek was reasonable except for the single unit that contained the highest density of juvenile *O. mykiss* observed in all 2012 sampling units (4.39 juveniles/100 ft^2).

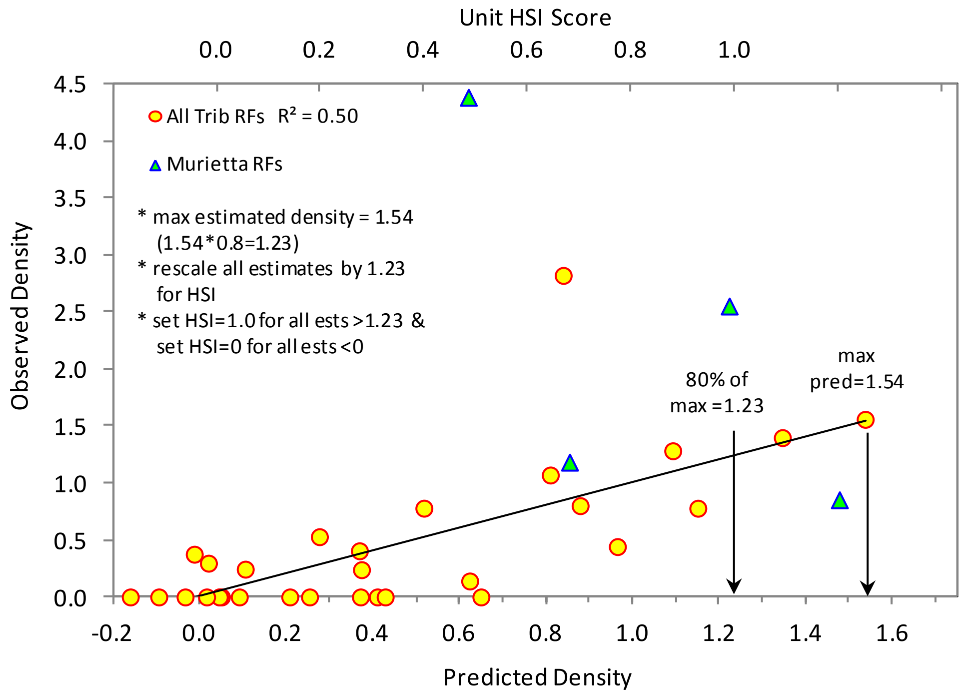


Figure 20. Example plot showing observed and predicted densities of juvenile O. mykiss in tributary riffles (n=30), with conversion of unit densities to unit HSI scores. Note Murietta data was plotted to assess model validity, but was not used to develop the regression models.

In general, this 80% rule resulted in an average of 4 units scaled to maximum suitability for fry (among the 2 channel size and 3 habitat type strata), 2.6 units at maximum for juveniles, and 2 units for adults. This relationship appeared reasonable given the more rigorous habitat requirements for larger fish (e.g., deeper water), however comparison of densities and habitat characteristics in the Ventura River Basin with data from other southern California steelhead basins may suggest that optimal habitat should support higher densities than (in the above example) 1.23 juveniles/100 ft^2.

After rescaling all habitat unit predictions to HSI values between 0 and 1, a weighted mean HSI score was calculated for each study site within channel size, habitat type, and fish age class strata using unit surface area as the weighting factor. As an example, Table 10 shows the weighted HSI scores for juvenile *O. mykiss* in riffles within tributary study sites; which produced low habitat unit HSI component scores for riffles in the two San Antonio Creek study sites largely due to low abundance of cobble substrate, versus riffles in the UNF and Mat 5 study sites which contained much higher proportions of cobbles as well as in-water branches.

Table 10. Estimated HSI scores for juvenile O. mykiss in tributary riffle habitats.

Study Site	RF HSI
SAC mid	0.07
SAC up	0.10
LNF new	0.27
LNF mid	0.18
UNF	0.53
Mat 5	0.54
Mat 7	0.34
Mur	0.84

The final step in calculating the habitat unit HSI component score for a given study site and fish age class involves combining the HSI scores for pools, flatwaters, and riffles using the habitat type weighting factors described in Section 4.3.1 and the proportion of each habitat type in the study reach. The habitat type weighting factors were determined for each channel size using mean relative densities from the five years in which sampling was conducted in all three habitat types. The calculated mean densities were then rounded to the nearest 0.05, such that the three habitat type weighting factors per channel size summed to 1.00 (Table 11). For example, *O. mykiss* fry averaged twice as abundant (in #/100 ft^2) in mainstem riffles than in mainstem flatwaters, and six times more abundant than in pools. Overall, the habitat type weighting factors for fry and juveniles gave more importance to riffle habitats than to pools (particularly in mainstem reaches), whereas adult rainbow trout weighting factors emphasized pool habitats over flatwaters and riffles in both channel sizes.

Table 11. Habitat type weighting factors used to calculate habitat unit component HSI scores.

Channel	HabType	Fry O.m.	Juv O.m.	Adult RBT
	PL	0.10	0.20	0.50
Mainstem	FW	0.30	0.30	0.25
	RF	0.60	0.50	0.25
	PL	0.20	0.25	0.50
Tributaries	FW	0.35	0.35	0.25
	RF	0.45	0.40	0.25

The final calculation of the habitat unit component HSI for each study site is determined by the average of two weighted calculations:

$$HSI_{habitat\ unit} = \left(\sum_1^3(\%SA_{HT} \times HSI_{HT}) + \sum_1^3(wt_{HT} \times HSI_{HT})\right)/2 \qquad (1)$$

where *HT* = habitat type, *%SA* = percent occurrence of habitat type by surface area, *HSI*=habitat type HSI score (e.g., Table 10 for juveniles in tributary riffles) , and *wt*=habitat type weighting factor (e.g., 0.40 for juveniles in tributary riffles, Table 11).

Reach Component HSI Score

The reach component HSI score was calculated for each life-stage and each study site using 7 variables (Section 4.3.2) within 3 sub-components (Figure 7). The physical habitat sub-component HSI was calculated as the geometric mean of the gradient (Figure 9) and shade (Figure 5) variable HSI scores, or:

$$HSI_{physical} = (HSI_{gradient}\ x\ HSI_{shade})^{1/2}$$

The biological sub-component was calculated as the minimum value of the BMI and the predation (Figure 10) variable HSI scores:

$$HSI_{biological} = min\ (HSI_{BMI}\ , HSI_{predation})$$

whereas the flow sub-component was calculated as the geometric mean between the flow persistence HSI variable (Figure 11), which is a direct assessment of flow stability, and the maximum of the valley width (Table 6) and tributary accretion HSI variables, which are indirect assessments:

$$HSI_{flow} = (HSI_{Q\ persistance}\ x\ \max(HSI_{valley\ width}, HSI_{trib\ accretion}))^{1/2}$$

These 3 sub-component HSI scores are combined into the study sites overall reach component HSI scores by:

$$HSI_{reach} = (HSI_{physical}\ x\ \min(HSI_{biological}, HSI_{flow})) \quad (2)$$

Recruitment Component HSI Score

The recruitment component HSI score is only utilized for the fry life-stage due to the direct effects of spawning habitat quantity and quality, incubation water quality, and proximity of spawning tributaries on the immediate recruitment of fry into a study reach. Although the local abundance of fry may also influence the local abundance of juveniles in the following year, and the abundance of adult rainbow trout may also be associated with the proximity of spawning habitat, dispersion of *O. mykiss* away from spawning areas is also common. In addition, the correlations between the overall study site HSI scores and the abundance of juvenile and adult *O. mykiss* were generally stronger when the recruitment HSI was only applied towards the fry life-stage.

The recruitment component HSI score was based upon 3 sub-component scores (Section 4.3.3): spawning habitat, incubation water quality, and tributary recruitment (Figure 7). The spawning habitat subcomponent was calculated as the geometric mean of the gravel quality (the *Vs* score) and quantity (Figure 13) HSI scores:

$$HSI_{spawning} = (HSI_{gravel\ quality}\ \ x\ HSI_{gravel\ quantity})^{1/2}$$

Because of the potentially overriding effects of poor water quality (Figure 5) on successful incubation of eggs, the incubation sub-component was calculated using the minimum value, or:

$$HSI_{incubation} = min\ (HSI_{incubation\ temperature}\ , HSI_{incubation\ DO})$$

A study reach can support a significant number of fry through recruitment from a nearby spawning tributary (Figure 6), even if spawning and incubation habitat is unsuitable. Consequently, a proximal tributary can compensate for low spawning and incubation scores, so the overall recruitment component HSI score is calculated by:

$$HSI_{recruitment} = max\ ((HSI_{spawning}\ x\ HSI_{incubation})^{1/2},\ HSI_{trib\ proximity}) \quad (3)$$

Water Quality Component HSI Score

The water quality component HSI score utilizes the three rearing water quality parameters of the original USFWS HSI model (Table 3 and Figure 7): maximum temperature, minimum D.O., and pH (Section 4.3.4), with subsequent modifications to the temperature HSI curve (Figure 5). The SS HSI model calculates this component score for all 3 life-stages (fry, juvenile, and adult rainbow) using the geometric mean of each variable:

$$HSI_{water\ quality} = (HSI_{max\ temp}\ \ x\ HSI_{min\ D.O.}\ x\ HSI_{pH})^{1/3} \quad (4)$$

Migration Component HSI Score

The migration component is utilized in the overall HSI score to assess habitat suitability for anadromous steelhead (Section 4.3.5). One HSI equation involving 3 variables is used to assess downstream migration of juvenile smolts (Figure 7); this score is then combined with the HSI score for juvenile rearing to represent the life-history requirements of anadromous juveniles. Another equation involving 6 variables is used to assess upstream migration of adult steelhead; this score is not combined with the adult resident trout score because of the limited time that adult steelhead reside in freshwater (and the adult migration HSI does include a holding pool variable).

Both equations utilize the water temperature variables described by the USFWS HSI model, but with the curve modifications shown in Figure 5. Both the smolt and adult steelhead equations also use a distance variable (Figure 14) that gives maximum suitability for short migrations and declining suitability as migration distance increases. Unlike the application of migration temperatures in the USFWS HSI model (Section 4.2.3), the migration HSI scores in the SS HSI model also accounts for downstream effects, since migration to or from a specific study site will by necessity also involve conditions experienced in all downstream reaches.

The migration component HSI score for smolts is a two-part equation based on water temperature and predation within the given study site and all downstream study sites. The temperature/distance sub-component is calculated as:

$$HSI_{smolt\ temp} = ((HSI_{temp\ 1}\ \ x\ HSI_{temp\ 2}\ x\ \ldots\ HSI_{temp\ i})^{1/i})\ x\ HSI_{dist}$$

Where *i* represents the number of reaches traversed by the smolt on its way to the ocean (which could include a score for lagoon temperature). Note that this equation does not assume that a single reach having warm temperatures and a very low HSI value will not limit the overall smolt temperature score, but instead that low HSI value is averaged with the remaining reach values to represent a “mean” condition. Also note that the reach-specific temperatures are not weighted by reach length, so a combination of a very long and warm reach with a short and cool reach will produce the same temperature HSI as two short reaches. If possible, long warm reaches should be represented by multiple shorter reaches.

This equation choice was selected because outmigration of smolts is largely expected to occur during flow pulses, which will generally produce lower temperatures (and thus higher suitability values) than would a temperature based on weekly maxima without regard to flow events. Consequently, if a smolt must traverse a long, very warm reach (even under elevated flows), this equation may produce an unrealistically high HSI score.

The smolt predation sub-component is likewise calculated using predation HSI scores (Figure 10) for each reach traversed by a smolt:

$$HSI_{smolt\ pred} = (HSI_{pred\ 1}\ \ x\ HSI_{pred\ 2}\ x\ \ldots\ HSI_{pred\ i})^{1/i}$$

The overall smolt migration HSI score is a simple geometric mean of these two HSI scores:

$$HSI_{smolt\ migration} = (HSI_{smolt\ temp}\ \ x\ HSI_{smolt\ pred}\ )^{1/2} \tag{5}$$

Note that the smolt migration component HSI score does not include any variables for physical barriers to downstream passage, such as dry reaches, falls, dams, or unscreened water diversions. HSI scores may need to be further degraded for study sites located above such barriers.

The adult steelhead migration component HSI score, in like manner to the smolt score, also utilizes an averaged temperature score multiplied by a distance HSI score (Figure 14), or:

$$HSI_{adult\ temp} = ((HSI_{temp\ 1}\ x\ HSI_{temp\ 2}\ x\ \ldots\ HSI_{temp\ i})^{1/i})\ x\ HSI_{dist}$$

Unlike for smolts, the adult migration HSI score does not include a predation variable, but it does include 3 passage variables as well as a holding pool variable (Figure 7). The passage HSI score is calculated as a minimum score based on the assumption that any limiting barrier downstream will affect all upstream reaches, using the passage variables and cumulative passage probability calculations described in Section 4.3.5. Consequently, the adult passage HSI score for a given study site is calculated as:

$$HSI_{adult\ passage} = min\ (HSI_{lagoon\ opening}\ \ x\ HSI_{riffle\ depths}\ x\ HSI_{vertical\ barriers})$$

Finally, the overall adult steelhead migration component HSI score is calculated simply as the geometric mean of the two sub-component scores listed above and the HSI value representing holding pools in a study site (Figure 18), or:

$$HSI_{adult\ migration} = (HSI_{adult\ temp}\ \ x\ HSI_{adult\ passage}\ x\ HSI_{holding\ pools}\ )^{1/3} \qquad (6)$$

Calculating Fry, Juvenile, and Adult HSI Scores

Using various combinations of the above component equations, HSI scores are developed within each study site for *O. mykiss* fry, juvenile, outmigrant steelhead smolt, resident rainbow adult, or upstream migrant adult steelhead.

The *O. mykiss* fry HSI score was calculated using the geometric mean of the habitat unit component HSI score, the reach component HSI score, the recruitment component HSI score, and the water quality component HSI score, or:

$$HSI_{fry} = (HSI_{habitat\ unit\ (fry)}\ x\ HSI_{reach}\ x\ HSI_{recruitment}\ x\ HSI_{water\ quality})^{1/4} \qquad (7)$$

This equation represents any *O. mykiss*, whether of resident trout or anadromous steelhead origin. Note that the habitat unit HSI scores are unique for each life-stage, whereas the reach and water quality HSI scores are common to each life-stage.

The juvenile *O. mykiss* HSI score, representing the freshwater rearing life-stage only (e.g., not accounting for smolt outmigration), is similarly calculated using the geometric mean, including the juvenile-specific habitat unit score, but excluding the recruitment component score:

$$HSI_{juvenile} = (HSI_{habitat\ unit\ (juvenile)}\ x\ HSI_{reach}\ x\ HSI_{water\ quality})^{1/3} \qquad (8)$$

To represent juvenile *O. mykiss* choosing an anadromous pathway as smolts, the HSI score is calculated by combining the juvenile HSI score with the smolt migration score:

$$HSI_{juvenile\ (smolt)} = (HSI_{juvenile}\ x\ HSI_{smolt\ migration})^{1/2} \qquad (9)$$

Use of the geometric mean to calculate the smolt HSI score may result in a higher score for the anadromous life-form than for the resident life-form, which may seem counterintuitive. In southern

California steelhead streams, many juvenile steelhead reach smolt size and outmigrate after only one-year in freshwater, whereas a juvenile choosing a resident trout pathway will likely require a second year to reach maturity. Consequently, emigrating as smolts after a single year of freshwater residency may be advantageous to juveniles that rear in suboptimal habitat. In contrast, juveniles rearing in cooler, higher elevation headwater areas may require a second year to smolt, but in such cases the smolt HSI score may be lower than the HSI score for resident juveniles.

Resident adult rainbow trout are represented by an equation similar to the juvenile equation, but with the resident adult's unique habitat unit HSI score:

$$HSI_{resident\ adult} = (HSI_{habitat\ unit\ (adult)}\ x\ HSI_{reach}\ x\ HSI_{water\ quality})^{1/3} \qquad (10)$$

Because adult steelhead are only expected to reside in freshwater for a relatively short time period, the HSI score for this life-stage is represented by the adult migration HSI score (equation 6), or:

$$HSI_{adult\ steelhead} = HSI_{adult\ migration} \qquad (11)$$

Calculating an overall *O. mykiss* HSI score for either resident rainbow trout or anadromous steelhead is done using the geometric mean of the appropriate life-stages.

For resident rainbow trout:

$$HSI_{resident\ rainbow} = (HSI_{fry}\ x\ HSI_{juvenile}\ x\ HSI_{resident\ adult})^{1/3} \qquad (12)$$

and for steelhead:

$$HSI_{steelhead} = (HSI_{fry}\ x\ HSI_{juvenile\ (smolt)}\ x\ HSI_{adult\ steelhead})^{1/3} \qquad (13)$$

Calculating HSI Scores at Different Spatial Scales

HSI scores for any of the above components can be calculated for different spatial scales, such as study sites, reaches, segments, sub-basins, or basins, by weighting each individual HSI score by its spatial extent, such as length or surface area. For example, if a given basin has 3 sub-basins, each with different spatial extents and HSI scores:

$$HSI_{basin} = \sum_1^3 (Length_i\ x\ HSI_i) / \sum_1^3 Length_i \qquad (14)$$

where *i* = sub-basin 1, 2, or 3.

4.4 Fish Abundance Sampling

For threatened and endangered species, state and federal agencies prefer passive fish sampling methods, such as direct observation (i.e. snorkeling), wherever feasible. In small to medium sized streams under low flow conditions, such as the Ventura River and Matilija Creek during the summer months, snorkeling is most effective where depths are sufficient for divers to navigate upstream. However, snorkeling is not effective where shallow depths prevent the diver from moving effectively through the unit. In such areas electrofishing can be highly effective to generate abundance estimates. For this study, sampling by direct observation was the preferred methodology and was used in those habitats where diving was feasible. Water depths in all of the mainstem Ventura River reaches was sufficient to allow direct observation in pools, but electrofishing was employed in all riffles. Flatwaters were sampled by diving in mainstem reaches in most years, but electrofishing was used in years with low flows (Table 12). In smaller channels where flatwaters were too shallow to conduct dive counts, electrofishing was used in both riffles and flatwaters, whereas dive counts were only employed in pools. The 2012 survey was an exception to this rule due to reaching our permitted take limit via electrofishing; consequently dive counts were used in some tributary flatwater and riffle habitats.

Sampling generally progressed from downstream study sites to upstream sites, with the majority of surveys occurring in mid-summer, generally beginning around the July 4^{th} weekend (Table 1). However, sampling was significantly later in 2008 due to contracting delays, whereas sampling was somewhat accelerated in 2012 due to particularly low flows and the desire to maximize the use of dive counts in flatwater habitats. With the exception of 2008 and 2012, the difference in sampling dates at a given study site in the remaining 5 years was typically within 3 to 4 weeks.

Each year sampled habitat units were located using habitat mapping data and handheld GPS units, generally one to three weeks prior to fish sampling. Top and bottom boundaries were flagged with plastic flagging labeled with habitat unit type and unit number. Unit lengths were verified using laser rangefinders or hipchain with biodegradable string. In most years, unit widths and depths were measured at 3 to 7 cross-sections, depending on unit length, using a laser rangefinder, tape measure, or stadia rod. Unit depths were also measured at the maximum thalweg depth (for the HSI analysis). As noted in Section 4.3.1, units widths, depths, and other variables were measured in 2012 using a modified transect protocol. Unit length, width, and depth measurements were used to calculate unit surface area and volume, as well as to assess suitability of individual habitat units.

During this pre-survey and also immediately prior to fish sampling with the backpack electrofisher, the margins of the sampling unit were carefully searched for the presence of any frog tadpoles, juveniles, or adults. If observed, those organisms were recorded and their locations were communicated to the fish crew so that any subsequent electrofishing passes would avoid those margin areas.

4.4.1 Direct Observation Dive Counts

Because conventional dive counts only represent an *index* estimate of abundance and not an estimate of *total* abundance, a random subsample of the units sampled by diving was re-sampled in order to calibrate the dive count index estimates to produce estimates of total abundance. The protocols and formulas used to calibrate the dive counts, and to generate basin-wide estimates of steelhead abundance, were taken from Mohr and Hankin's Method of Bounded Counts (MBC) unpublished manuscript. Habitat units that were sampled using electrofishing as the primary sampling methodology (described below) did not need calibration because multiple-pass electrofishing provides estimates of total abundance.

Table 12. Annual sampling frequency and methodology according to basin segment, study site, habitat type, and year. PL=pools, FW=flatwaters, RF=riffles, DO=direct observation (snorkeling), EF=backpack electrofishing, BS=bag seine, UV=underwater video, X=not sampled (red X's represent estimated abundance - see Section 4.5.1).

Basin Segment	Study Site	Habitat Type	2006	2007	2008	2009	2010	2011	2012
Lower	Ven 1 [1]	PL	DO	DO	DO	X	DO	DO	X
		FW	DO	DO	DO	X	EF	DO	EF
		RF	EF	EF	X	X	EF	EF	EF
	Ven 2 [2]	PL	DO	DO	DO	X	X	DO	X
		FW	DO	DO	DO	X	EF	DO	EF
		RF	EF	EF	X	X	EF	EF	EF
	Ven 3	PL	DO	DO	DO	DO	DO	DO	DO
		FW	DO	EF	DO	DO	EF	DO	DO
		RF	EF	EF	X	X	EF	EF	EF
	SAC mid [3]	PL	X	EF	DO	X	DO	DO	DO
		FW	X	EF	X	X	X	EF	DO
		RF	X	EF	X	X	X	EF	DO
	SAC up [2]	PL	X	X	X	X	DO	DO	DO
		FW	X	X	X	X	X	EF	DO
		RF	X	X	X	X	X	EF	DO
	Ven 4 [2]	PL	DO	dry	dry	dry	DO	DO	dry
		FW	DO	dry	dry	dry	EF	DO	dry
		RF	EF	dry	dry	dry	EF	EF	dry
Middle	Ven 5	PL	DO	DO	DO	DO	DO	DO	DO
		FW	DO	EF	DO	DO	EF	DO	DO
		RF	EF	EF	X	X	EF	EF	EF
	LNF low/new [4]	PL	DO	DO	DO	DO	DO	DO	DO
		FW	EF	EF	X	X	EF	EF	DO
		RF	EF	EF	X	X	EF	EF	DO/EF
	LNF mid	PL	DO	DO	DO	DO	DO	DO	DO
		FW	EF	EF	X	X	EF	EF	DO
		RF	EF	EF	X	X	EF	EF	EF
Upper	Mat 3	PL	DO	DO	DO	X	DO	DO	DO
		FW	EF	EF	X	X	EF	DO	DO
		RF	EF	EF	X	X	EF	EF	EF
	Mat 5	PL	DO	DO	DO	X	DO	DO	DO
		FW	EF	EF	X	X	EF	DO	EF
		RF	EF	EF	X	X	EF	EF	EF
	Mat 7/7b [2,5]	PL	DO	DO	DO	X	X	DO	DO
		FW	EF	EF	X	X	X	DO	EF
		RF	EF	EF	X	X	X	EF	EF
	UNF up/new [6]	PL	DO	DO	DO	X	DO	DO	DO
		FW	EF	EF	X	X	EF	DO	EF
		RF	EF	EF	X	X	EF	EF	EF
	Murietta	PL	X	X	X	X	X	X	DO
		FW	X	X	X	X	X	X	EF
		RF	X	X	X	X	X	X	EF
	Lagoon		BS,UV,EF	BS,UV	X	X	X	BS,UV	X

[1] Ven 1 was moved upstream in 2010 & again in 2011 due to encroachment of homeless camps

[2] non-random study sites

[3] qualitative, single-pass "spot" shocking in 2007

[4] LNF low was replaced by LNF new in 2007

[5] Mat 7 was replaced by Mat 7b in 2011

[6] UNF up was replaced by UNF new in 2007

Each pool or flatwater unit selected for conducting dive counts was sampled by one to four biologists, depending on unit width and water clarity. Divers cautiously entered the lower end of each habitat unit in pre-specified dive lanes, then proceeded together upstream to the unit head counting fish as they passed downstream of the diver. Diver position and observation area within each unit was determined prior to each unit being sampled. Each diver enumerated all *O. mykiss* in their dive lane by size class, with reference to a wrist-mounted ruler. Data were recorded onto underwater slates during the dive counts, and then transferred to data sheets after each dive.

Divers classified all *O. mykiss* as either fry (<10cm FL) or juvenile+ (≥10cm FL). The 10 cm criterion used in this study was consistent with the size class utilized in previous studies in the South-Central California Coastal ESU for steelhead in Morro Bay tributaries (TRPA 2001, 2007b) and the San Luis Obispo Creek watershed (TRPA 2004b), where length-frequency distributions suggested that most young-of-year fry were less than 10cm FL in the summer months. Young-of-year *O. mykiss* in the Sespe watershed were also reported to be less than 10 cm in length (Dvorsky 2000), and mean length of young-of-year *O. mykiss* fry in Topanga Creek were approximately 10-11cm by fall (Bell et al. 2011). In Soquel Creek, tributary to Monterey Bay, *O. mykiss* <9cm by October were classified as young-of-year (Sogard et al. 2009). The larger *O. mykiss* size class was further divided in 2011 and 2012 into smolt-sized juveniles 10-20 cm FL versus resident adult-sized trout >20 cm FL. Approximately 85% of 231 smolts trapped at the nearby Freeman Diversion Dam on the Santa Clara River in 2009-2011 were between 13-20 cm FL; the remaining 15% were larger than 20 cm FL (UWCD 2009, 2010, 2011). Most *O. mykiss* in Topanga Creek were less than 25cm by the fall of their 3rd year of life (Bell et al. 2011). Three years of downstream trapping in San Luis Obispo Creek revealed smolt lengths ranging from 12-23 cm (Spina et al. 2005), however only 6% of smolts were >20 cm FL (Normandeau, unpublished data). Thus, fish already >20 cm in length during a summer survey in the Ventura River Basin would likely exceed smolt-size by the following spring, and were consequently expected to remain, mature, and reproduce in freshwater as stream-resident “rainbow trout”, rather than follow an anadromous pathway as “steelhead”.

After conducting the single-pass dive count, divers determined if the sampling unit was selected for a second-stage calibration survey by removing a label concealing a “yes” or “no” previously recorded for each unit (but unknown to the divers). If the unit was not selected for calibration, the divers continued upstream to the next selected pool (or flatwater). If the unit was selected for calibration, the divers conducted three more independent dive counts according to the MBC protocols, for a total of four counts (or two more counts if no *O. mykiss* were observed on any pass). Each repetitive count was conducted after the water visibility had cleared sufficiently to produce visibility conditions similar to the first dive count.

In most study sites, five of the eight units of each habitat type were selected by simple random sampling for repeat count calibration. Thus, second-stage calibration was generally conducted on 50% or more of units that were selected for first-stage dive counts. All calibration surveys were conducted using the repeat dive counts; electrofishing was not used because the vast majority of first pass counts in calibration units were less than the maximum count (20 fish per species/size strata) recommended for calibration by direct observation methods, according to the MBC protocols, and to minimize the application of the more stressful electrofishing protocols.

Additional information collected at each habitat unit included starting and ending dive times, water temperature, underwater visibility, and digital photographs. Water visibility was measured just downstream of the sampling unit immediately prior to diving, by recording the distance at which a diver could clearly identify a two-inch trout colored lure.

4.4.2 Multiple-Pass Electrofishing

Multiple-pass electrofishing was employed as the primary fish sampling methodology in all riffles and in also in flatwaters for those stream reaches that were too shallow to effectively dive (Table 12). Electrofishing surveys were conducted by trained personnel using procedures consistent with guidelines established by NOAA Fisheries for protecting listed species of salmonids (NMFS 2000), except that electrofishing was frequently conducted at stream temperatures higher than the maximum recommended temperature of 18^{o}C, and at conductivities higher than 350µS/cm. At virtually all of the mainstem Ventura River study sites, and several of the mainstem Matilija Creek sites, summer water temperatures in the morning hours already exceeded the NOAA recommended maximum, and specific conductivities throughout the entire basin were typically over 700µS/cm. Consequently, it would not be possible to utilize electrofishing within the study area under the federal guidelines. We notified NOAA of this problem and were authorized to continue with our intended sampling procedures based on several observations and procedural safeguards:

- Southern steelhead appear to be more tolerant of warmer water conditions than steelhead in more northerly areas;
- Repeated electrofishing in 2006, 2007, and 2010 under high temperatures resulted in low immediate mortality (2%), with no short-term mortality of 14 *O. mykiss* confined overnight in a net pen;
- At the warmest study sites all captured *O. mykiss* were kept in buckets separated from other species and containing a portable aerator and self-contained ice-pack to reduce stress; one individual was specifically assigned to ensure that the bucket water was continually refreshed, aerated, and remained cooler than the river water.

Due to the known or potential presence of listed amphibians (California red-legged frogs, *Rana draytonii*) in several study reaches, specific protocols were also employed to identify locations where tadpoles or adult frogs were present, and to avoid those areas during walking and electrofishing. These specific protocols are described in Appendix C.

Prior to electrofishing, block nets were placed at the upper and lower unit boundaries in order to prevent emigration out of the study site during sampling. On smaller habitat units, great care was taken to place the block nets in a manner to minimize displacement of fish prior to sampling. Maintenance of a minimum habitat unit length (for riffles and small channel flatwaters only) of 20 ft during the mapping also helped to minimize this potential disturbance. Occasionally, the upper boundary of the sampling unit contained a natural barrier, such as a cascade or high gradient riffle, where an upper block net was not required.

Each unit was sampled using one or two backpack electrofishers (Smith-Root models 11-A and 12-A) with one to two netters per shocker. The voltage and frequency settings used during electrofishing were adjusted for each stream reach to provide efficient capture of fish and to minimize physical injury to the fish. Each sampled pool received a minimum of three electrofishing passes, unless salmonids were not captured in either of the first two passes. All captured fish from each pass were temporarily held in an aerated bucket or transferred into an instream live-car until all electrofishing passes were completed. Equal effort was maintained among passes by careful attention to repeating each pass (by the same individual) in a similar manner and in a similar time frame. The "shocking seconds" and the beginning and ending times were also recorded for each electrofishing pass. After electrofishing, all captured salmonids were anesthetized with CO_2 (using a 3:1 solution of water:club soda or alka-seltzer tablets dissolved in water) in order to reduce stress associated with measurement.

The following data were recorded at each study site: number of fish captured (by species) during each pass, the fork length (to nearest mm) and weight (to nearest gram) of each *O. mykiss*, the number of mortalities (if any), and any unusual features of captured *O. mykiss* (e.g., abnormalities, black-spot disease, etc.). Weights were not measured for non-salmonid species. After data collection, all fish were revived in fresh water and released back into the sampling unit following the final pass. In addition to the capture data, water temperature and conductivity was measured at each electrofishing unit, and digital photographs were recorded.

4.4.3 Estimation of Fish Abundance

The abundance and density (number/100 ft^2 of stream channel) of *O. mykiss* by size class was estimated at three spatial scales: within individual habitat units, within study sites (by habitat type and combined across habitat types), and within basin segments (habitat types combined).

Estimating Abundance within Habitat Units

For units sampled by diving, single pass dive counts were used to estimate an *index* of abundance. For the (typically) five dive units that were calibrated by the MBC (Mohr and Hankin, unpublished manuscript), estimates of *total* abundance were calculated for each size class according to the bounded count formula:

$$\tilde{y}_B = D_{[m]} + (D_{[m]} - D_{[m-1]}) \tag{15}$$

where $\tilde{y}_B$ is the *biased* bounded count estimate of true abundance, $D_{[m]}$ is the largest of the four dive counts, and $D_{[m-1]}$ is the second largest of the four dive counts. An adjustment factor is used to correct for the negative bias that is typically associated with dive counts. To estimate this bias, the diver observation probability ($\hat{p}$) is first required:

Definitions	Variable
diver observation probability in unit *k*	p_k
the i[th] diver count in unit *k*	D_{ik}
number of repeat counts (4)	m_D
overall diver observation probability	$\hat{p}$
number of calibration units where $\overline{D}_k$ is >0	n_2*

$$\hat{p}_k = 1 - \frac{s_k^2(D)}{\overline{D}_k}$$

where $\overline{D}_k = \sum_{i=1}^{m_D} D_{ik} / m_D$ and $s_k^2(D) = \sum_{i=1}^{m_p} (D_{ik} - \overline{D}_k)^2 / (m_D - 1)$

and the overall observation probability for all calibrated dive units within a given habitat type and study site for each size class is:

$$\hat{p} = \frac{1}{n_2^*} \sum_{k=1}^{n_2^*} \hat{p}_k$$

The bias-adjusted estimate of abundance $\tilde{y}_B *$ is then calculated using:

Definitions	Variable
original bounded count estimate	$\tilde{y}_B$
bias-corrected bounded count estimate	$\tilde{y}_B *$
number of repeat counts	m

$$\tilde{y}_B * = \tilde{y}_B - \sum_{u=0}^{\tilde{y}_B - 1} \hat{F}\ (u)^{m-1}\ (m - (m+1)\hat{F}(u)) \qquad (16)$$

$$\text{where} \quad \hat{F}(u) = \sum_{j=0}^{u} \binom{\tilde{y}_B}{j} \hat{p}^j (1-\hat{p})^{\tilde{y}_B - j}$$

The abundance of *O. mykiss* in riffles and shallow flatwater units sampled by electrofishing was estimated for each size class using the following bias-adjusted jackknife estimator:

Definitions	Variable
bias-adjusted jackknife estimate	$\hat{y}_j^*$
electrofishing capture on pass *i*	C_i
total capture on all passes	C_m
number of repeat electrofishing passes	m
estimated capture probability	$\hat{p}$

$$\hat{y}_j^* = \sum_{i=1}^{m-1} C_i + \frac{C_m}{\hat{p}} \qquad (17)$$

where the capture probability ($\hat{p}$) is represented by a pooled estimate (within habitat type and size class strata) based on the removal sequence for all *k* sampled units containing fish, per:

$$\hat{p} = 1 - \frac{\sum_{k=1}^{n} \sum_{i=1}^{m} C_i(k) - \sum_{k=1}^{n} C_1(k)}{\sum_{k=1}^{n} \sum_{i=1}^{m} C_i(k) - \sum_{k=1}^{n} C_m(k)}$$

These bias-corrected estimates of true abundance in dive units ($\tilde{y}_B *$) or electrofishing units ($\hat{y}_j^*$) are inserted into the equation for $\hat{t}_{y,D}$ or $\hat{t}_{y,DA}$ (see below) and its associated variance to produce estimates of abundance and variance of fry or juvenile+ *O. mykiss* within a given habitat type for each study site.

Estimating Abundance within Study Sites

For estimation of fish abundance and densities at the study site scale, jackknife electrofishing estimates, or dive counts calibrated by MBC, were calculated for each sampling unit according to the equations presented above. Study site estimates are expanded estimates that represent total fish abundance by size class within the entire one mile or one-half mile study site according to habitat type. Because the estimates of abundance and variances were independently derived for each habitat type, the overall study site estimates (combined habitat types) were calculated by simply adding together the respective habitat type estimates of abundance and variance. Abundance estimates do not account for non-sampleable habitat, such as cascades, falls, or habitat units containing so much brush or vegetation that they could not be effectively sampled. Such non-sampleable habitats typically represented <5% of the study site by length, but equaled 10-15% in two study sites due to cascades (in Mat 7b) or *Arundo donax* thickets (in SAC up).

To estimate the total abundance of fry or juvenile+ *O. mykiss* within a study site according to habitat type, the following definitions apply:

Definitions	Variable
total number of available sampling units	N
number of units sampled	n_1
number of calibrated units	n_2
mean diver counts in sampled units	$\bar{x}_1$
mean diver counts in calibrated units	$\bar{x}_2$
mean "true" abundance in calibrated units	$\bar{y}_2$
ratio of true abundance to 1st pass counts in calibrated dive units	B_x
mean length of all available sampling units	$\bar{z}_U$
mean length of all sampled units	$\bar{z}_1$
mean length of all calibrated units	$\bar{z}_2$

Habitat unit length (*z*) was tested as an auxiliary variable in ratio estimators to see if a positive correlation between numbers of fish and unit size would increase precision of the abundance estimates (Hankin 1984, Hankin and Reeves 1988, Mohr and Hankin, unpublished manuscript). A high, positive correlation will increase the precision of ratio estimators (Cochran 1977), and thus improve the ability to detect differences among spatial and temporal scales. The specific estimate (i.e., with or without the auxiliary variable) used to represent *O. mykiss* abundance for each year, study site, habitat type, and size class varied according to performance, as judged by the calculated variance. Ratio estimators using unit length as an auxiliary variable ($\hat{t}_{y,DA}$) were used where the correlation increased precision of the abundance estimate (e.g., produced a narrower confidence interval); estimators without auxiliary variables ($\hat{t}_{y,D}$) were used when the correlation was poor (common in low-density study sites).

The estimated abundance for a given size class and habitat type within a study site, where unit length was not correlated with abundance ($\hat{t}_{y,D}$), was calculated as:

$$\hat{t}_{y,D} = N\bar{y}_2\left(\frac{\bar{x}_1}{\bar{x}_2}\right) \quad (18)$$

with a variance of:

$$\hat{V}\left(\hat{t}_{y,D}\right) = N^2\left(1-\frac{n_1}{N}\right)\frac{s_e^2(\bar{y}_2)}{n_1} + N^2\left(1-\frac{n_2}{n_1}\right)\left(\frac{\bar{x}_1}{\bar{x}_2}\right)^2\frac{s_e^2(\bar{y}_{2,x})}{n_2} \quad (19)$$

where $$s_e^2(\bar{y}_2) = \sum_{k=1}^{n_2} \frac{(y_k - \bar{y}_2)^2}{n_2 - 1}$$

and $$s_e^2(\bar{y}_{2,x}) = \sum_{k=1}^{n_2} \frac{(y_k - \hat{B}_x x_k)^2}{n_2 - 1} \quad \text{with} \quad \hat{B}_x = \frac{\bar{y}_2}{\bar{x}_2}$$

The corresponding formulas for those strata where dive counts or electrofishing captures were positively correlated with unit lengths ($\hat{t}_{y,DA}$) were:

$$\hat{t}_{y,DA} = N\bar{y}_2\left(\frac{\bar{x}_1}{\bar{x}_2} + \frac{\bar{z}_U - \bar{z}_1}{\bar{z}_2}\right) \tag{20}$$

and $$\hat{V}(\hat{t}_{y,DA}) = N^2\left(1 - \frac{n_1}{N}\right)\left(\frac{\bar{z}_U}{\bar{z}_1}\right)^2 \frac{s_e^2(\bar{y}_{2,z})}{n_1} + N^2\left(1 - \frac{n_2}{n_1}\right)\left(\frac{\bar{x}_1}{\bar{x}_2}^2\right)\frac{s_e^2(\bar{y}_{2,x})}{n_2} \tag{21}$$

95% confidence intervals for both abundance estimates were calculated as:

$$\hat{t}_{y,..} \pm t_{n-1}\sqrt{\hat{V}(\hat{t}_{y,..})} \tag{22}$$

Estimated densities of fry and juvenile+ *O. mykiss* were calculated by dividing the estimated abundance by the total length (in miles) or surface area (SA, in 100ft^2) of a given habitat type within each study site, as determined from the habitat mapping. Variance for these density estimates were calculated by:

$$\frac{\hat{V}(\hat{t}_{y,...})}{(mi\ or\ SA)^2}$$

Estimating Abundance within Basin Segments

Estimated abundance at the segment scale was calculated by summing the abundances and variances from each study site within each segment, then expanding those summed estimates to represent the total length of each segment. Note that although study sites were initially selected at random from within reaches, which were subsets of segments (see Section 4.1.2), this stratification level was ignored during segment estimation and study sites were treated as being randomly selected from within segment strata. This treatment is not unlike the common application of systematic sampling to ensure longitudinal coverage of sampling locations, after which estimates are often generated under the assumption of random (rather than systematic) selection (Hankin & Reeves 1988).

Note that the *O. mykiss* abundance estimates for stream segments do not include portions of tributaries above impassable barriers (TRPA 2003), or tributaries that were not quantitatively sampled, such as Old Man Creek in the upper segment, or Coyote Creek or in the lower segment. The expanded segment estimates do include the entire length of the mainstem Ventura River, all of the mainstem Matilija Creek (up to the first impassible barrier), and both forks (upper and lower) of the North Fork Matilija Creek up to the first impassible barriers identified in 2003 (but ignoring the quarry barrier near the mouth of the Lower North Fork). Comparisons of annual abundance estimates between segments also do not include data from Murietta Creek or San Antonio Creek, which were only sampled during the latter years (Table 12).

4.4.4 Ventura Lagoon Sampling

The Ventura River lagoon was sampled over a single day in mid to late August 2006 and 2007, and in late June 2011. Eight to 16 beach seine sets were made throughout the shallower portions of the lagoon using a kayak to deploy a 4 ft by 100 ft seine with a ½ inch mesh size (Figure 21).

The larger mesh size was used to avoid capture of the listed tidewater goby (*Eucyclogobius newberryi*), but may have also prevented capture of small *O. mykiss* fry. In most years the seine sets extended over a range of tidal heights, although the lagoon outlet to the Pacific Ocean was closed-off in 2007, which limited tidal influence on the lagoon's physical and chemical properties. Limited electrofishing was also conducted in the flowing section at the head of the lagoon in 2006, but all subsequent surveys were restricted to the stillwater portions of the lagoon where salinity, depth, and/or expansive open water made electrofishing infeasible. All captured fish were identified to species, enumerated, and released back into the lagoon.

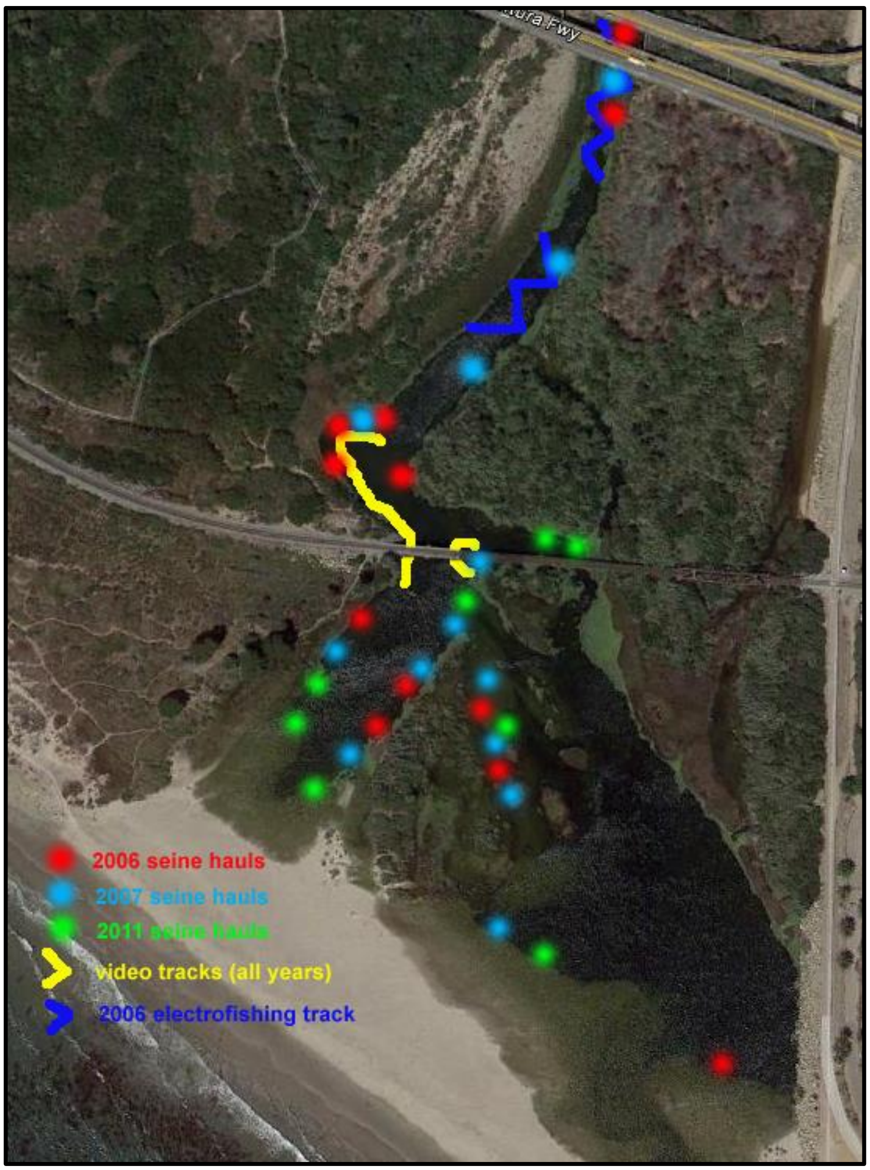


Figure 21. Map of Ventura River Lagoon showing annual locations of seine hauls, underwater video surveys, and electrofishing transects (image from August 2012).

Seining could not be conducted along the deep, rip-rap lined channel under the railroad bridge or in scour holes associated with bridge abutments and woody debris, and therefore we deployed a pole-mounted, high-resolution underwater video camera (Outland Technology UWC-300, low-lux B&W) to search for fish in the deeper water. The camera was probed along the riprap bank from the shoreline, and along the bridge abutments from a kayak. Video images were both viewed in real-time and recorded onto a portable DVR (Archos AV-700). The recorded video clips were reviewed a second time in the office at the conclusion of each year's survey.

4.5 Data Analysis

This section addresses the methodologies used to assess the significance of temporal changes in abundance estimates of *O. mykiss* as well as differences in abundance at various spatial scales, and how variability in those estimates affects the power to detect differences. Methods used to evaluate the relationship between HSI results and *O. mykiss* abundance is also described.

4.5.1 Assessment of Trends in Abundance

Annual Trends in Abundance

Changes in annual abundance of *O. mykiss* were assessed for each size class using two methodologies. Most comparisons of annual changes utilized a conservative approach by assessing the overlap of 95% confidence intervals (Equation 22) around each abundance estimate. Overlapping confidence intervals, such as those in 2006 and 2007 in Figure 22, were judged to represent a non-significant change in abundance between estimates. This approach is conservative because the likelihood that, for example, the true value in 2006 is actually near 50 while at the same time the true value for 2007 is near 52, is expected to be less than 5%.

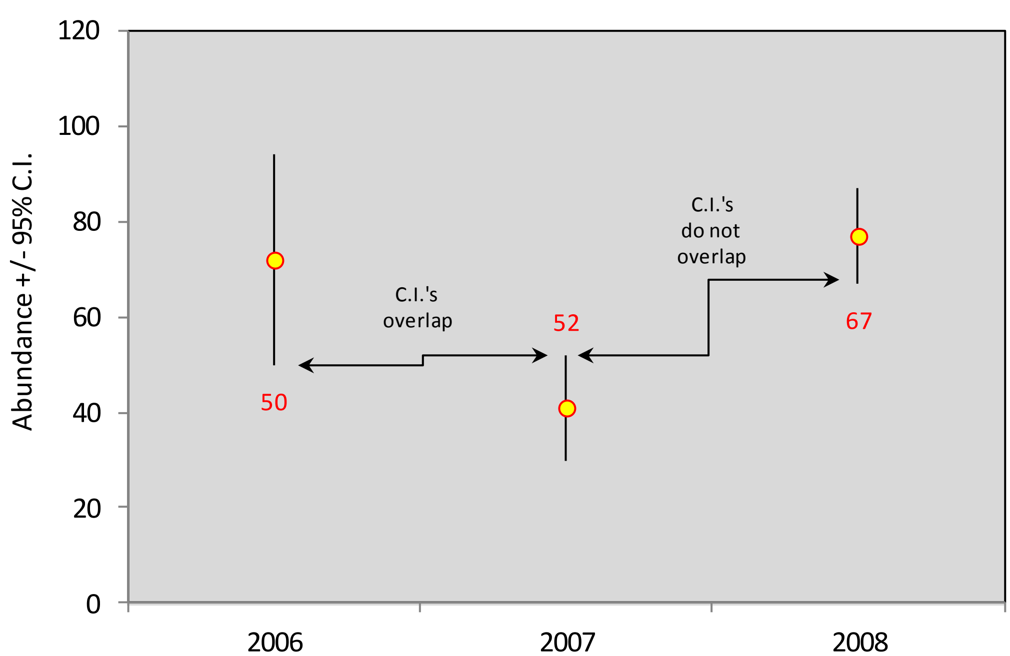


Figure 22. Example data showing assessment of differences in estimates by overlap of 95% confidence intervals.

A quantitative assessment of changes between years was employed when consecutive survey years utilized the same sampling units and sampling methodologies (e.g., diving versus electrofishing). Difference estimators (Cochran 1977, Des Raj 1968) were applied to consecutive-year estimates for pools and riffles, which were consistently surveyed by diving and electrofishing, respectively. Difference equations were less frequently applied to flatwater estimates due to occasional changes in sampling methods (e.g., more diving in wetter years, more electrofishing in drier years). Difference estimates could not be calculated when counts were zero in all units of a specific study site/habitat type strata; when study sites were remapped and thus new sampling units were selected (e.g., in 2011); or for data pooled among all three habitat types. Where difference equations could not be employed to assess significance, overlap in confidence intervals was used instead.

Difference estimators utilize the expected correlation in abundance within specific habitat units between years to increase the precision of the estimated change in abundance. The estimated difference in abundance between, for example, 2006 and 2007 for a given study strata (e.g., *O. mykiss* fry in Ven 3 riffles) is calculated as:

$$D\hat{I}F = \hat{Y}_{07} - \hat{Y}_{06} \tag{23}$$

with a variance of:

$$\hat{V}(D\hat{I}F) = \hat{V}(\hat{Y}_{06}) + \hat{V}(\hat{Y}_{07}) - 2C\hat{O}V(\hat{Y}_{06}, \hat{Y}_{07}) \tag{24}$$

where the covariance term is:

$$C\hat{O}V(\hat{Y}_{06}, \hat{Y}_{07}) = r(\hat{Y}_{06}, \hat{Y}_{07})\sqrt{\hat{V}(\hat{Y}_{06})}\sqrt{\hat{V}(\hat{Y}_{07})}$$

and the correlation between years (r) is estimated by a Pearson product-moment correlation. The stronger the correlation, the smaller the variance around the estimated difference, and the greater the power to detect changes in abundance.

The 95% confidence interval for the estimated difference between years is:

$$95\%CI = t_{n-1}\sqrt{\hat{V}(D\hat{I}F)} \tag{25}$$

A statistically significant difference in abundance between years is indicated if the 95% confidence intervals for the difference <u>*does not*</u> encompass zero.

Treatment of Missing Data

Table 12 reveals that all habitat types were not sampled in all years. For example, only pools and mainstem flatwaters were sampled in 2008 due to delays in acquiring a permit for electrofishing. Sampling in 2009 was further restricted due to limited funding. Other data gaps occurred when access was lost (e.g., Mat 7 in 2010), or when elevated turbidity in lower mainstem pools made dive counts ineffective (e.g., Ven 1 in 2012, Ven 2 in 2010 and 2012). These data gaps produce interruptions in the time series data and they impact the assessment of annual trends in abundance. Consequently, linear regression was used to estimate abundance of fry and juvenile+ *O. mykiss* for many of these missing strata (red X’s in Table 12).

For example, riffles were not sampled in Ven 3 in 2008 and 2009, however all three habitat types were sampled in the remaining 5 years. The relationship between abundance of *O. mykiss* in riffles versus abundance in pools plus flatwaters was assessed using linear regression for the 5 years of full data. The resulting regression model (for fry) was:

$$\hat{Y}_{RF} = -16.522 + 0.688\ x\ \hat{Y}_{PL+FW}$$

which produced an R^2=0.82 with a P=0.03. This model was then used to predict the number of fry in riffles in 2008 and 2009, using the available pool + flatwater abundance estimates from those years. The variance associated with each predicted estimate for riffles was combined with the variance from the sampled habitat type(s) to yield an estimate of abundance in all habitat types with associated confidence intervals, and to yield a full 7-year time series of abundance estimates for combined habitat types. In a similar manner, the relationship between abundance of *O. mykiss* in LNF new pools versus abundance in flatwaters and riffles was used for the five years of full sampling to estimate the number of *O. mykiss* in flatwaters and riffles (combined) in 2008 and 2009, when these habitat types were not sampled (Figure 23).

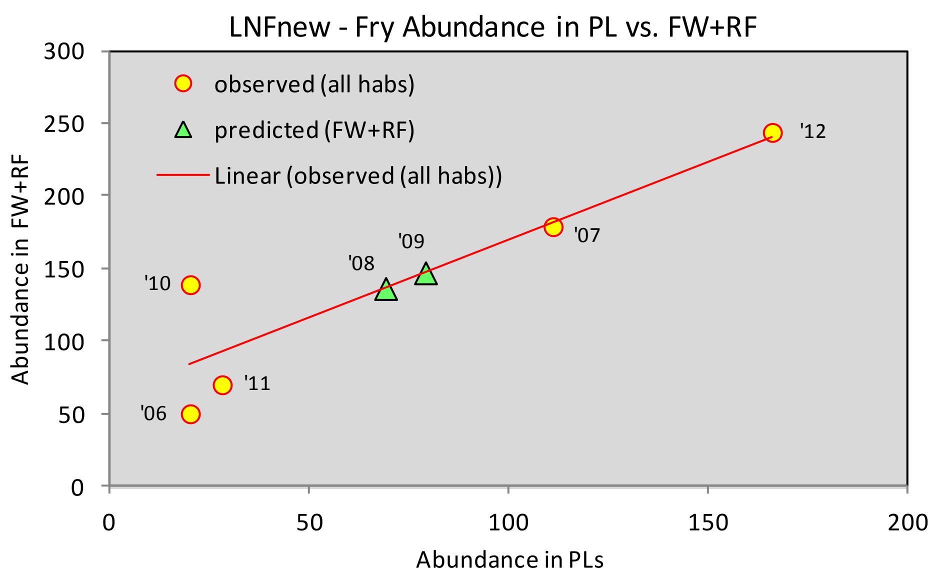


Figure 23. Example showing prediction of missing abundance estimates.

In general, most regression models produced a reasonable fit for fry, with an average (of 8 models) R^2 of 0.71. Models predicting abundance of juvenile+ were less successful, with an average (of 10 models) R^2 of only 0.37.

Spatial Trends in Abundance

Chi-square goodness of fit tests were utilized to assess if the estimated abundance of fry or juvenile+ *O. mykiss* among habitat types was simply proportional to the availability of each habitat type (e.g., surface area of pools, flatwaters, and riffles), or if *O. mykiss* were non-randomly distributed among habitat types (e.g., showed selectivity for certain habitat types). Data only included those years with full sampling among habitat types, and where observed or expected frequencies exceeded 5 fish (Sokal and Rolf 1969). Pie charts were also used to visually compare densities of *O. mykiss* according to habitat type. Comparison of *O. mykiss* abundance between study sites or between basin segments was visually assessed by overlap of confidence intervals, as previously described.

4.5.2 Comparison of Fish Abundance and HSI Scores

The relationship between estimates of fish abundance and HSI scores was assessed for each study site by simple linear regression, using the HSI score as the predictor variable and density (#/100ft^2) of either fry or juvenile+ *O. mykiss* as the response variable. HSI and fish abundance data were also pooled among study sites to represent the three study segments, and this relationship was visually assessed using scatterplots.

4.5.3 Assessing Power and Sample Size Requirements

An important component for planning a distribution and abundance study includes assessment of the spatial and temporal variability of the fish population, in order to estimate the sample size requirements to detect significant trends in population parameters (e.g., abundance). This seven-year study provided abundance information stratified by year, segment, study site, habitat type, and fish size class, and may be useful to assist with planning future steelhead studies in other Southern California basins. Consequently, abundance data were input into the program *Trends* (Gerrodette 1993) to assess sample size requirements and the associated power to detect annual trends in abundance. The excel sample size and power assessment tool *SSPow2Samples.xls*, developed by Ken Gerow (Gerow 2007), was used to assess sample size requirements and power to detect significant differences in paired-year or paired-reach studies.

Assessing Annual Trends in Abundance

The program *Trends* (Gerrodette 1993) was used to assess the relationship between the number of years of sampling and the power to detect a statistically significant increase or decrease in abundance. Inputs to the program included the model of the expected change in abundance (linear or exponential), the direction of change (increase or decrease), the test design (1-tailed or 2-tailed), the relationship between variation and abundance (C.V. either constant with abundance [A], proportional to sqrt[A], or proportional to 1/sqrt[A]), the distribution type (*z* or *t* distribution), the Type-I error rate, or significant level (α), and the initial (or observed) C.V.'s. The number of years of sampling is then input to estimate the power to detect a significant change given the above input parameters.

For this assessment, *Trends* was utilized to estimate power to detect a *10% annual change in abundance* (either positive or negative) using a 2-tailed test of a linear change with an α = 0.1 and the C.V.'s constant with abundance (e.g., variation increases directly as abundance increases). A conservative Type-I error rate (α) of 10% was selected over the traditional 5% rate because the Type-I error, interpreted in this context as the probability of detecting a *false* trend in abundance, was not considered as vital for endangered species as the Type-II error rate (β), which is the probability of **not** detecting a *true* trend in abundance. The power of a statistical test, or 1- β, thus describes the ability a test to accurately detect a real trend if it is in fact occurring.

The 10% per year change in abundance criteria was arbitrarily chosen based on the assumption that a 10% annual decrease would pose a significant threat to an endangered species, such as southern steelhead. The C.V. to abundance relationship (constant with abundance) was based on an assessment of the 7-year dataset. The initial C.V. inputs were also based on observed C.V.'s from the 7-year dataset according to each given strata, however C.V. inputs were truncated at 1.0 (e.g., higher estimated C.V.'s were set to 1) to allow power estimation. The Ventura Basin data was assessed according to segment strata (lower, middle, upper, or combined), habitat type sampling design (all habitats or pools-only sampling), and fish size class (fry <10cm, juvenile+ $\geq$10cm, or all *O. mykiss*). The pools-only alternative was specifically added to this assessment because the full 7-year dataset contained pool abundance data and because pools-only sampling is a common alternative used for many endangered species studies due to the less invasive nature and ease of permitting for diving-only studies. The iterative estimation of power for a given number of years of sampling was used to create power curves, while also noting the specific number of years required to detect a 10% change with 80% power.

Assessing Change in Abundance Between Two Consecutive Years

The above assessment was used to evaluate longer-term trends, with guidance for predicting the number of years of sampling to detect a linear increase or decrease in abundance. An alternative assessment was used to estimate the number of individual sampling units that may be required to detect a difference in abundance within a single reach or segment over a 2-year period. This evaluation may be used to guide sample size determinations when comparing the potential effects of a restoration action intended to increase abundance, or the effects of a projected impact with its potential to decrease abundance.

The excel macro program *SSPow2Samples.xls*, developed by University of Wyoming statistics professor Ken Gerow, was used for this assessment because it allowed the selection of various sampling designs, including a paired (e.g., fixed index site) design option (Gerow 2007). In a paired design, such as that used in the Ventura studies, the same sampling units are surveyed each year, and correlations in abundance within individual sampling units between years can potentially be used to increase the power to detect actual changes.

The *SSPow2Samples.xls* program was used in a 2-tailed design assuming paired sampling between any two years, with the tested criteria being a change in abundance (either an increase or a decrease) of 25% between the two years (Figure 24). The 25% criteria was chosen arbitrarily and was greater than the 10% annual change described above, which seemed appropriate for a long-term study but too ambitious (i.e., difficult to detect) for a 2-year study given the levels of annual variability seen in most salmonid studies. In contrast, although a 50% change in abundance between any two years may occur in nature, that level of change, if due to restoration or impact activities, did not seem sufficient to allow protection of a population experiencing a decrease in abundance.

To estimate the power to detect a 25% change in abundance using the 2-tailed, paired design, *SSPow2Samples.xls* requires selecting the Type-I error rate (α=0.05 was used in this assessment) and an option describing the manner of variation with abundance (constant variance, variance proportional to mean abundance [A], or standard deviations proportional to mean A). The latter measure of variation appeared most appropriate for the Ventura data (Ken Gerow, personal communication). This assessment stratified the Ventura data into two spatial groups (headwater and tributary study sites *versus* mainstem study sites), three habitat sampling designs (all habitat types *versus* a pools-only *versus* a representative reach design), but only assessed abundance for all *O. mykiss* combined (e.g., not by individual size classes). The headwater/tributary study sites used in this analysis were the two LNF study sites (LNF new and LNF mid), the UNF study site, and the Mat 7 study site. The mainstem study sites included Ven 3, Ven 5, Mat 3, and Mat 5.

For each study site, the ratio of the standard deviation (SD) of abundance to the mean abundance was calculated for each year when all habitat types were sampled (5-7 years). SD/mean ratios were also calculated for pool-only data for each year when all pools were sampled (6-7 years). The Pearson correlation coefficient (*r*) was calculated between the unit-specific abundance estimates for each year-pair when the same habitat units were sampled (e.g., 2006-07, 2007-08, . . . 2011-12), whether for all habitat units combined or for pools-only. Strong positive correlations will result in increased power (and thus lower sample size requirements), whereas low correlations will provide little benefit to the paired design. The average ratio of SD/mean and the average correlation was calculated for each study site and input into *SSPow2Samples.xls* to calculate the power associated with each increase in sample size (Figure 24), according to the spatial strata (headwater/tributary or mainstem) and sampling design (all habitats or pools-only). Power curves and the estimated number of habitat units required to achieve 80% power were determined based on these calculations.

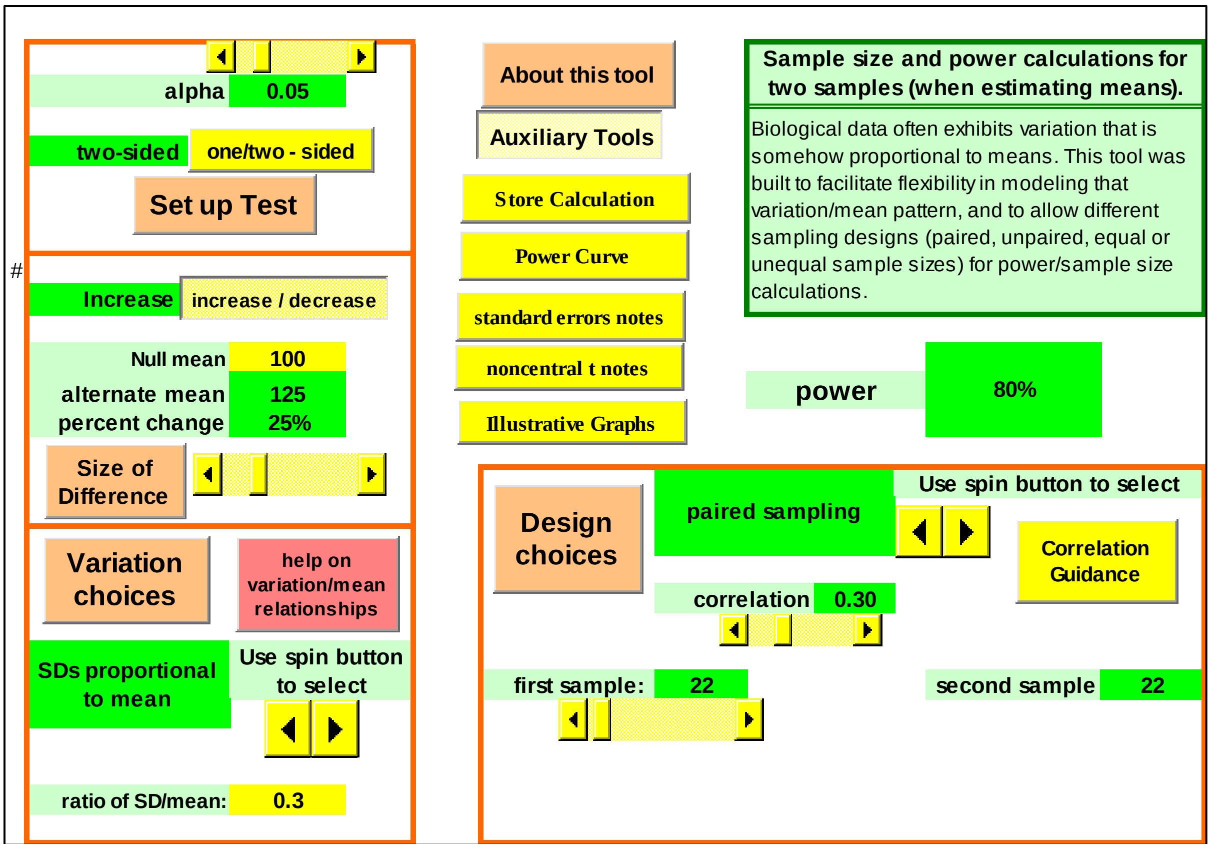


Figure 24. Screenshot of SSPow2Samples.xls with inputs for paired-year assessment.

Assessing Change in Abundance Between Two Reaches

The *SSPow2Samples.xls* program was also used to assess the number of sampling units that may be required to detect a 25% difference in abundance between two reaches in a given year, such as might be desired in a control *versus* impact design. Because sampling units are located in two separate reaches, and because this assessment looked at differences in abundance over a single year, this was not a paired design, so the design selected in *SSPow2Samples.xls* was the 'independent, equal sample sizes" model, which again used a 2-tailed design with *α*=0.05 and data-derived estimates of the average SD/mean ratio, but did not include correlations due to the nonpaired design. In other respects this assessment was identical to the paired-year design described above.

Assessing Changes Using a Representative Reach Design

This study used a habitat-stratified design with independent sampling within pool, flatwater, and riffle habitat types. This design was employed because of the expected (and observed) variability in *O. mykiss* densities according to habitat type (see Section 4.3.1), and the desire to account for that variation to improve the precision of annual estimates of abundance. However, many traditional fish distribution and abundance studies do not use a habitat-stratified design, but instead use a representative reach (RR) design, where a (typically) 100 yard or 100 meter study reach is selected that encompasses multiple habitat types, presumably (but often not validated) in proportion to each habitat type's availability in that study area. In order to utilize the Ventura's habitat-stratified data to estimate power and sample size requirements under an alternative, RR design, the Ventura data was pooled to simulate a series of ~100-200 yard RRs in each study site, then assessed using *SSPow2Samples.xls* under the two scenarios described above (paired comparison between 2 years, or independent samples from two reaches).

To create simulated RRs from the Ventura data, the mean length of habitat units was calculated for 7 study sites - the 4 headwater/tributary locations described above, and 3 of the 4 mainstem locations. The Mat 5 study site was not included in this analysis because the variability in flowing versus dry channels in that reach did not allow for consistent pooling of unit data into RRs. Habitat units in each of the 4 headwater/tributary study sites and the Ven 5 mainstem study sites averaged 40-60 ft in length, whereas habitat units in the Ven 3 and Mat 3 study sites averaged 80-100 ft in length. Simulated RRs for each study site were derived by adding unit-specific *O. mykiss* abundance estimates from 2 randomly selected units of each habitat type, or 6 units total per RR. The random selection procedure was repeated to yield abundance estimates for 24 simulated RRs (corresponding to the number of individual habitat units assessed above). Note that although these simulated RRs did not include continuous series of pools, flatwaters, and riffles, the encompassed units were frequently in close proximity and some units were in fact contiguous to one another.

The combined abundance estimates for *all O. mykiss* (size classes combined) within each of the 24 RRs were then used to calculate the average SD/mean abundance ratios and the average correlations (for the paired-year assessment), and input into *SSPow2Samples.xls* to estimate the power to detect a 25% difference in abundance according to the number of sampled RRs. Power curves and the specific number of RRs estimated to achieve 80% power were derived based on these calculations.

4.5.4 Other Data Analysis

Length-frequency distributions of fish captured by electrofishing were created for each stream reach in order to assess possible differences in local population characteristics, and to evaluate the appropriateness of the 10 cm FL size criterion for separating fry (young-of-year) from juvenile+ *O. mykiss* (yearling or older). Tracking cohorts over time was also assessed via scatterplots with linear regression to determine how much variation in abundance of an older size class was explained by abundance of the younger age class the preceding year.

The relationship between first-pass dive counts and four-pass MBC estimates was assessed with scatterplots and linear regression for each calibrated sampling unit, in order to validate the expected correlation between first-pass counts and total fish abundance. A limited number of sampling units were calibrated using both repeat dive counts and multiple-pass electrofishing; these comparative abundance estimates were assessed with scatterplots and correlation analysis.

5.0 Results

The 2006-2012 *O. mykiss* distribution and abundance surveys encompassed 11 to 14 study sites in most years (only 4 sites in 2009), involving 109 to 308 individual sampling units (46 in 2009) distributed throughout the Ventura River Basin and evenly split among pool, flatwater, and riffle habitats. See Table 1 for details regarding sampling dates and physical habitat parameters of each study site. Specific sample sizes by year and study site along with all abundance and density estimates can be found in Appendix D.

5.1 Environmental Conditions

The Ventura River Basin exists near the southern limit of steelhead, and like other southern California steelhead basins the Ventura Basin possesses a dry, warm climate that is punctuated by intermittent and sometimes intensive rainfall events. The typically dry, sunny climate in combination with little or no rainfall over the summer months leads to high water temperatures in most mainstem and lower elevation tributaries. Low flows and high water temperatures are principal factors presently limiting the distribution and abundance of *O. mykiss* in southern California watersheds (NMFS 2012), and these factors can be further exacerbated by the withdrawal of surface or sub-surface water and input of poor water quality due to agricultural and urban development.

5.1.1 Rainfall and Streamflows

Water year types and annual rainfall amounts from 2002 to 2012, based on upper (wet) or lower (dry) quartiles of annual rainfall in the City of Ventura from 1873-2012 (140 years), are listed in Table 13, and show alternating patterns of wet, normal, and dry years, with a 4 year period of normal or wet years from 2008 to 2011.

Table 13. Annual cumulative rainfall in the City of Ventura from 2002-2012, number of days with peak flows in the lower Ventura River (USGS gage #8500), and mean base flows from July-November. Water year types based on upper and lower quartiles from City's long-term (1873-2012) rainfall data.

Year	Rainfall (in)	Days >1000 cfs	Base Flow cfs	Water Year
2002	7.2	0	1.1	dry
2003	19.9	1	4.5	wet
2004	11.6	1	2.3	normal
2005	35.9	16	21.2	wet
2006	18.1	5	16.8	normal
2007	6.7	0	2.3	dry
2008	14.1	6	7.3	normal
2009	10.4	0	3.2	normal
2010	16.2	0	6.4	normal
2011	19.7	3	12.8	wet
2012	8.9	0	1.8	dry

Rain-gage data from Matilija Dam illustrates that the vast majority of rainfall occurs over the December to March period, but with a median rainfall of less than 0.1 inch per month over a 6 month period from May to September (Figure 25). Median monthly flows in the lower Ventura River (USGS gage #8500) show a one month delay from rainfall with highest flows (15-30 cfs) in February, March, and April and lowest flows ($\leq$2 cfs) in late summer and fall (August-November). The highly variable occurrence and potentially extreme magnitude of high flow events is clearly illustrated in Figure 26, where peak daily mean flows in the lower Ventura River exceeded 1,000 cfs on 13 separate rain events between 2002 and 2012, with 13 days exceeding 1,000 cfs in 2005 alone (Table 13). Maximum mean daily flows reached approximately 6,000 cfs in 2006, 2008, and 2011, and exceeded 20,000 cfs in 2005. High flow events were notably absent in 2002, 2007, 2009, and 2012, when maximum mean daily flows ranged from a low of only 7cfs in 2002 to 65 cfs in 2009.

Median monthly flows during the dry season in the lower Ventura River remained less than 10 cfs from May through December in 2002, 2004, 2007, 2009, and 2012, but flows maintained at or above 10 cfs in 2005 and 2006 (Figure 27). Although rainfall in 2006 was just below the "wet year" classification, base flows remained high due in part to the abundance of rain in 2005 (the 4th highest rainfall on record). In contrast, median base flows were less than 3 cfs in 2002, 2004, 2007, and 2012, with gage readings of <1 cfs in each year. During wet years the Ventura River flows generally reach their minimum during November or December, but dry and normal year's exhibit more extended periods of minimum flow beginning in August or September.

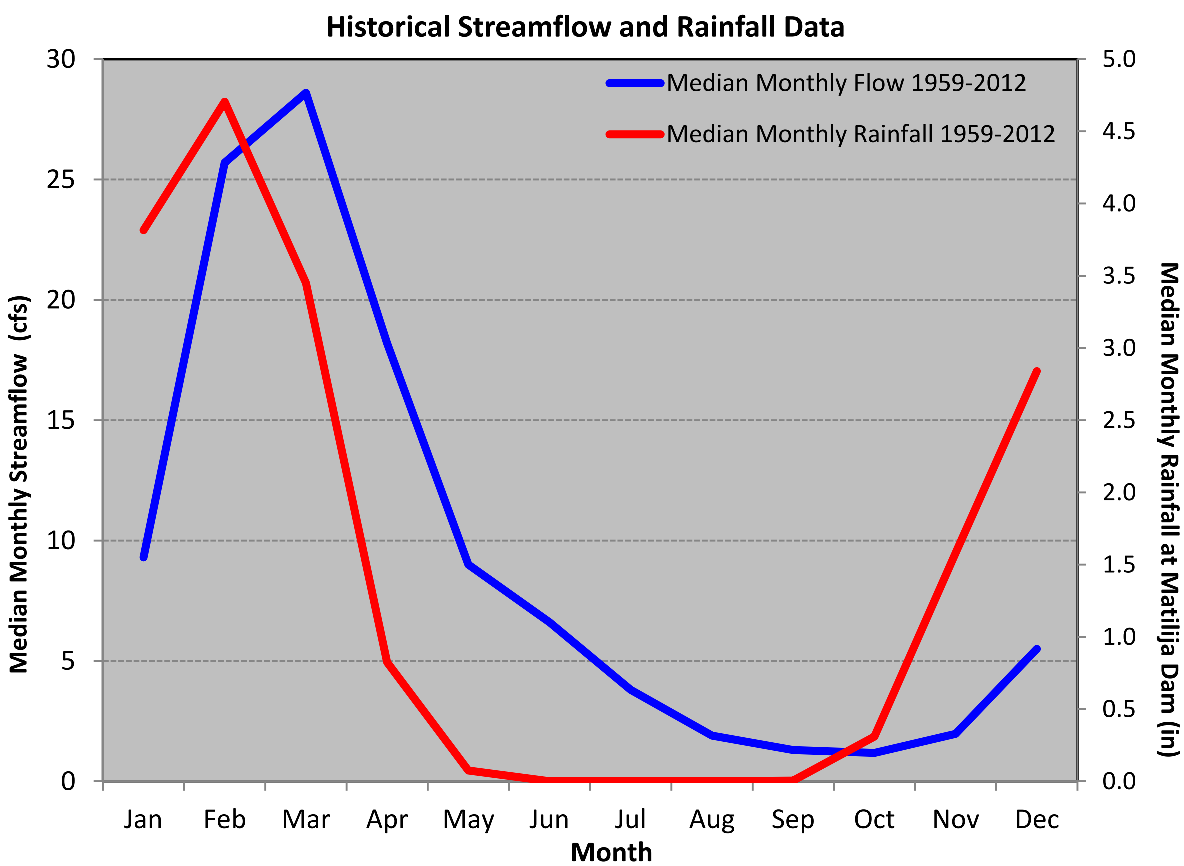


Figure 25. Median monthly streamflows at Ventura USGS gage #8500 and median rainfall at Matilija Dam, 1959-2012.

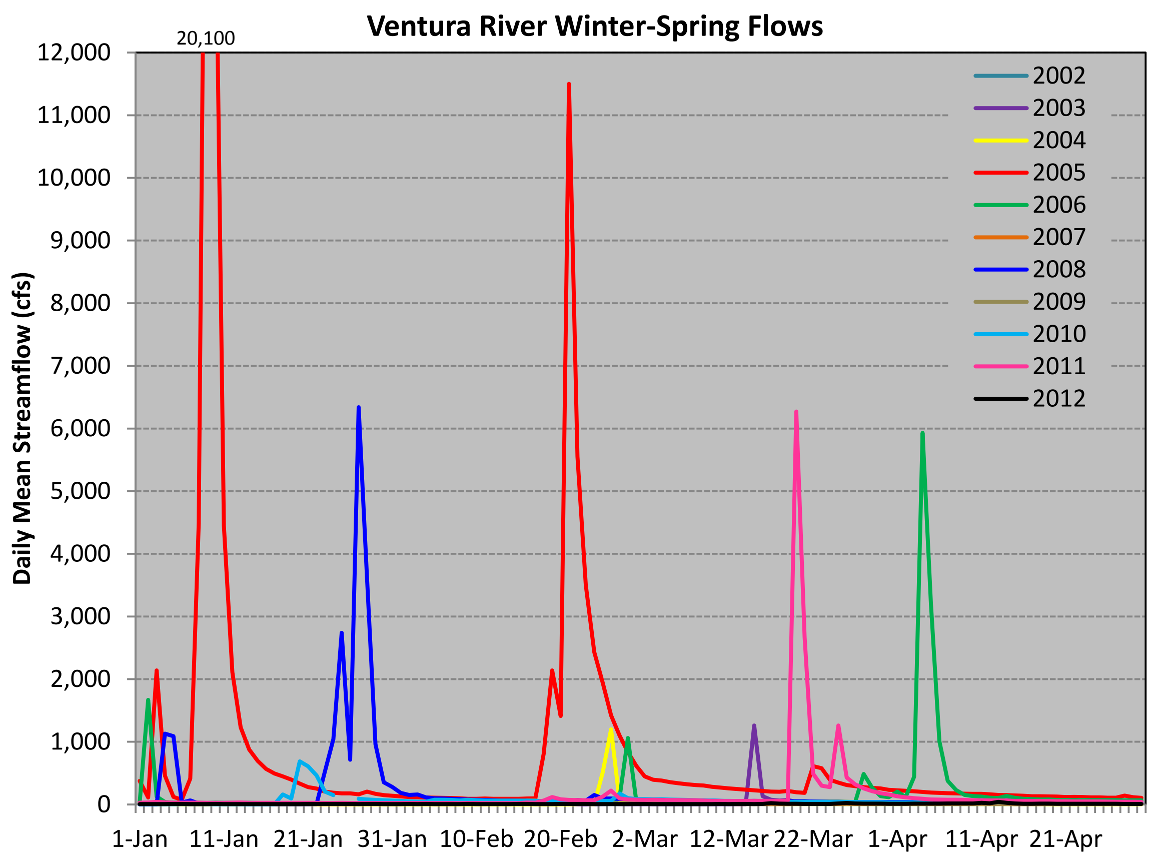


Figure 26. Mean daily streamflows at Ventura USGS gage #8500 during January through April, 2002-2012.

Average estimated streamflows at each study site during abundance surveys over the 7 years of study are given in Table 1, and generally show the lower magnitude of surface flow in the headwater and tributary reaches. Average flows during sampling were typically over 10 cfs in the mainstem Ventura River study sites, 5-10 cfs in the mainstem Matilija Creek sites, and 1-4 cfs in the headwater and tributary study sites. Exceptions included the mainstem Ven 4 study site, which was dry by mid-summer in 2007, 2008, 2009, and 2012. Intermittent and dry stream reaches are common in southern California steelhead watersheds, including the Ventura River Basin. Although the number and extent of intermittent reaches varies each year within the basin due to differences in rainfall, groundwater levels, and water withdrawal from diversions and water pumps, the typical location of dry or intermittent reaches is shown in Figure 28. Neither Matilija Dam or Robles Diversion Dam alter streamflows in the Ventura River during the summer base flow period, although Coyote Dam captures all flow into Casitas Reservoir with no downstream release. Flow augmentation does occur, however, from a wastewater treatment facility which releases approximately 2 cfs into lower 5½ miles of the mainstem Ventura River. Base flows are altered by groundwater pumping for municipal, residential, and agricultural uses, which are most likely to influence surface flows in the mainstem Ventura River and in San Antonio Creek. Limited effects of groundwater pumping on base flows may also occur in the lower mainstem Matilija Creek and in the lower North Fork Matilija Creek.

Although Figure 27 and Table 13 appears to suggest that this study occurred over a period of relatively typical flow and environmental conditions, a more intensive assessment of long-term trends in rainfall and streamflows suggests that the Ventura River Basin was in the midst of a longer dry spell, beginning in roughly 1996 (Leydecker 2014). Although not encompassed in this report, extreme low flows occurred throughout the Ventura Basin in 2013 and 2014, which resulted in greater extents of dry or intermittent channels than depicted in Figure 28. Much of the typically perennial and highly productive Ven 3 reach became dry or stagnant (Paul Jenkin, personal communication), which was expected to have severe impacts on the locally resident O. mykiss population (see Section 5.5.2).

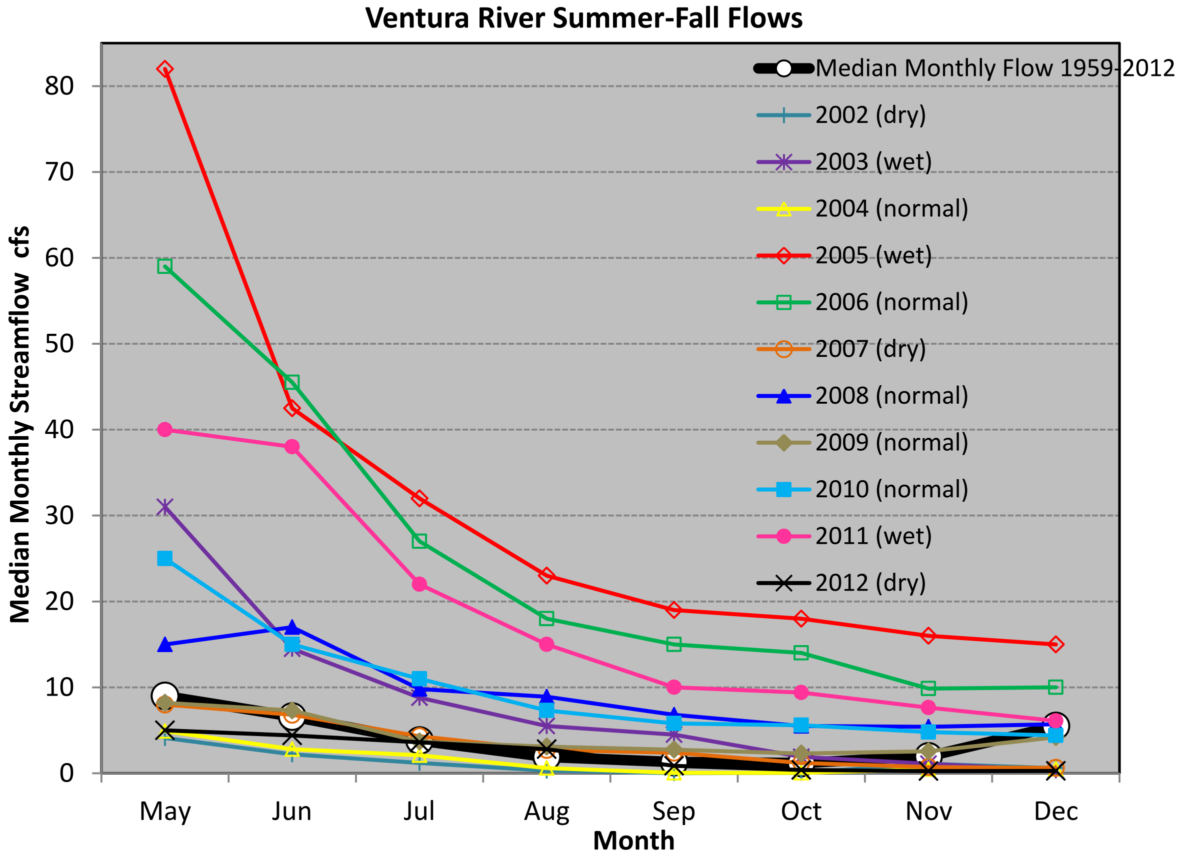


Figure 27. Median monthly base flows at Ventura USGS gage #8500 from May to December 2002-2012, with historical flows (1959-2012 median) and water year designations.

5.1.2 Water Temperature

O. mykiss require cool, clean water to grow, mature, and reproduce. Water temperature is one of the primary limiting factors influencing the current distribution of *O. mykiss* in southern California watersheds, along with adequate surface flows and access to spawning tributaries. As stated in Section 4.2.2, the HSI curves for *O. mykiss* life-stages appeared too restrictive for southern California populations (hence their modification), however significant portions of the Ventura River Basin exhibit spring, summer, and fall water temperatures that are clearly suboptimal.

Water temperature data was collected using spot measurements during fish sampling in each year of study, however the following results are based on continuous time series data collected from 13 water temperature data loggers deployed throughout the Ventura River Basin over the last 3 years of sampling (Figure 1, Table 4), representing a normal water year (2010), a wet year (2011), and a dry year (2012). A comparison of mean weekly average temperatures (MWAT) in 2 warmer mainstem and 2 cooler tributary study sites during a wet year (2011) shows minimum mean

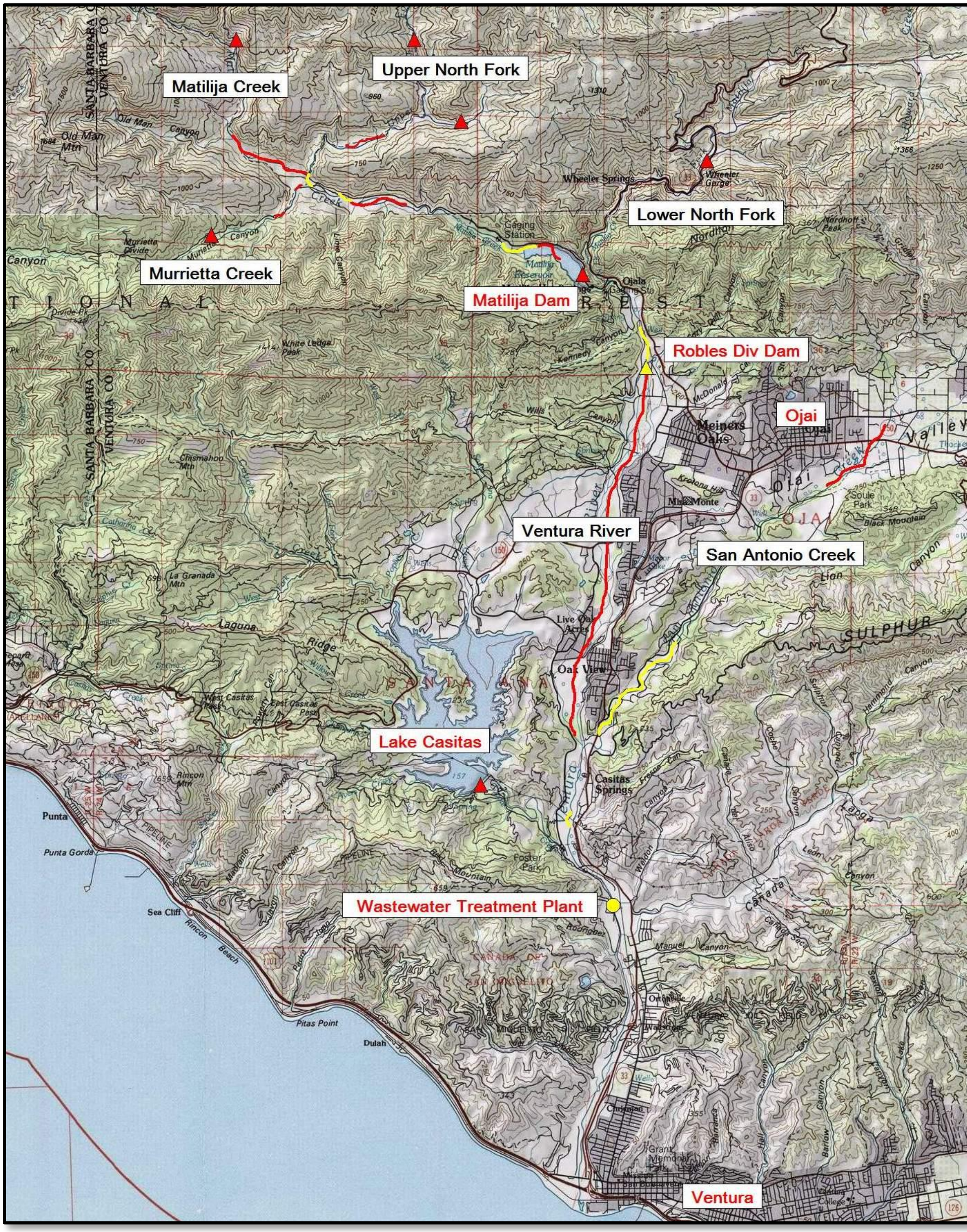


Figure 28. Approximate location of dry or intermittent stream reaches in normal water years (red lines). Yellow lines show potential expansion of dry channels in drier years. Red triangles are approximate locations of known barriers to upstream migration.

temperatures in December through February, and maximum mean temperatures in July through September at all 4 sites (Figure29) . Of these 4 representative sites, upper San Antonio Creek was generally the warmest through the winter months but the coolest over the summer, in contrast to the Ven 5 site which was coolest in the winter but warmest in the summer. The upper San Antonio site is influenced by rising groundwater upstream of the data logger, which likely moderates both summer and winter water temperatures. The Ven 5 site is influenced by Matilija Reservoir, whichmay act in an opposite manner by retaining heat over the summer yet remaining cool over the winter. Differences in mean temperature between the warmest and coolest of these 4 sites were generally 3-5°F throughout the year.

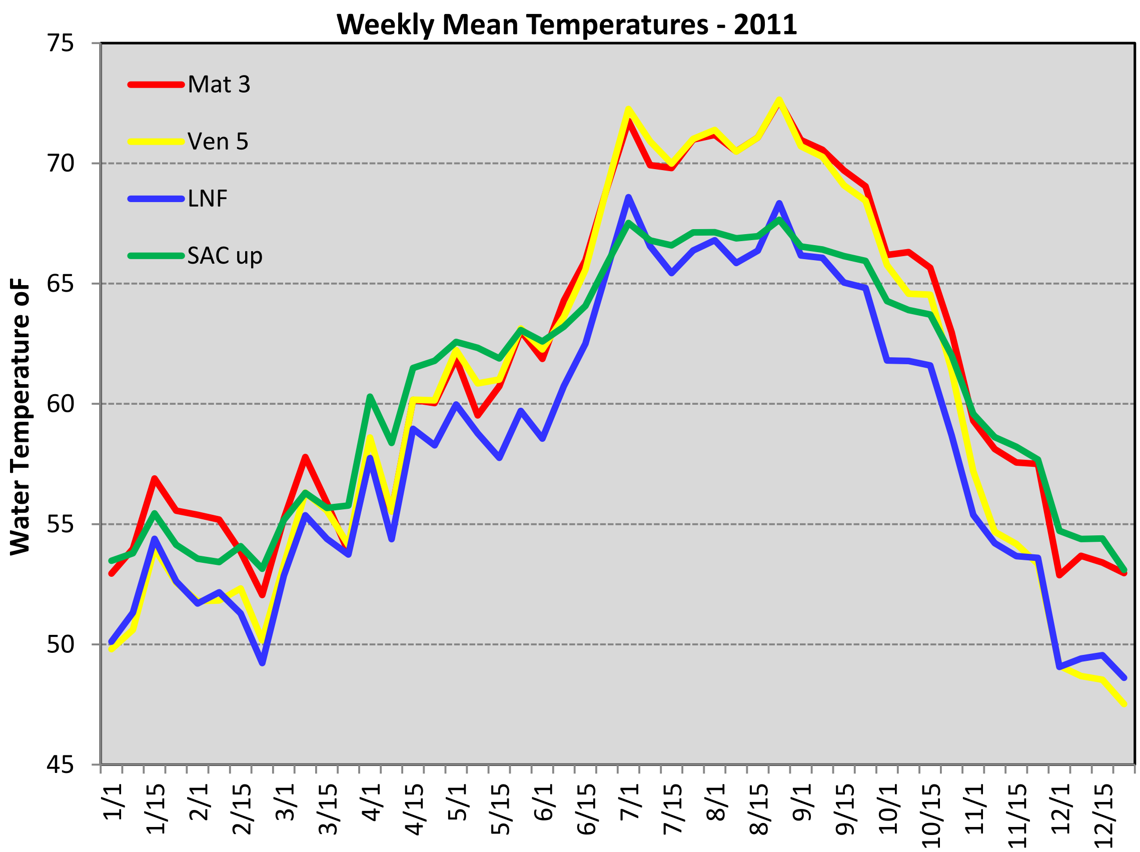
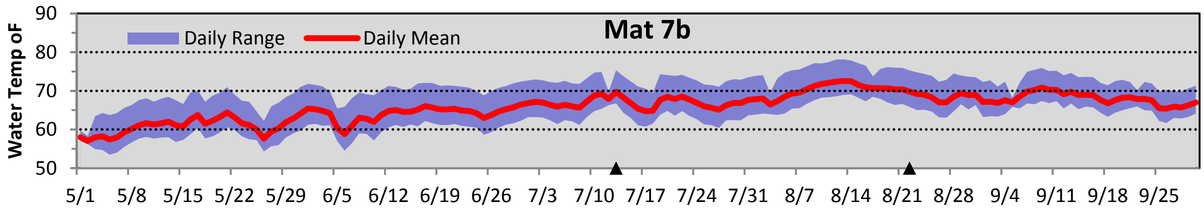
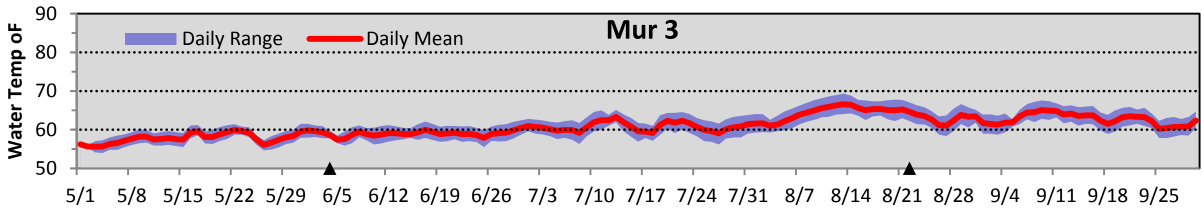
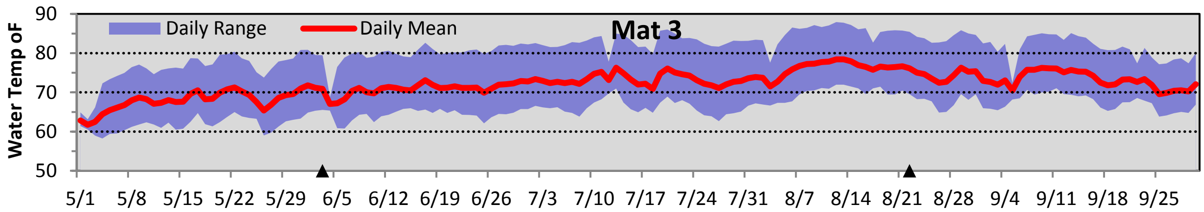


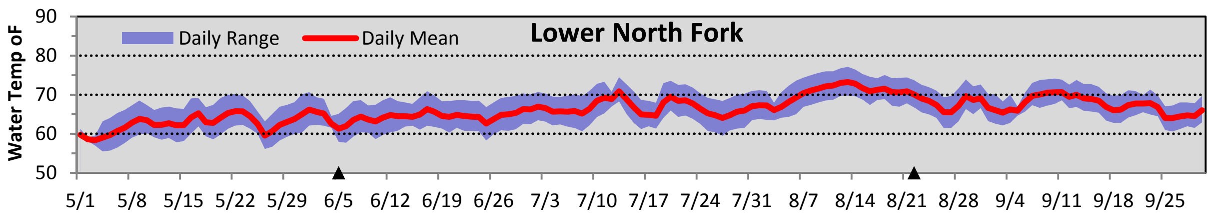
Figure 29. Mean weekly average temperatures at 4 study sites throughout 2011, a wet year.

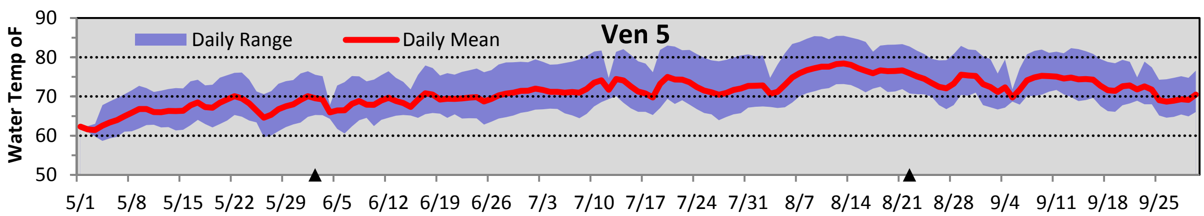
Inspection of daily minimum, mean, and maximum temperatures at 9 of the study sites during May through September 2012 (a dry year) reveals significant site-specific differences in the magnitude and range of daily temperatures (Figure 30). Daily fluctuations are particularly evident at the Mat 3 and Ven 5 study sites, which also regularly exceeded 80°F (27°C). The largest average range in summer (June through September) daily temperatures was 15.5°F in Mat 3, followed by Ven 5 at 11.8°F. The narrowest range occurred in Murietta Creek at 4.4°F, with mean daily ranges of 6-7°F in the upper San Antonio Creek (June-July only), the lower North Fork, and the Ven 3 study sites. The daily range at Ven 3 was well below the range in Ven 5 farther inland, and was also less than Ven 2 (at 9.5°F), which is located downstream and is more directly influenced by coastal fog. The more constant temperature regime in Ven 3 is somewhat unique for an inland mainstem reach, and this characteristic in combination with the more constant flow regime is undoubtedly responsible for the potentially high fish production observed in that reach (see Section 5.5.2 below). Although the

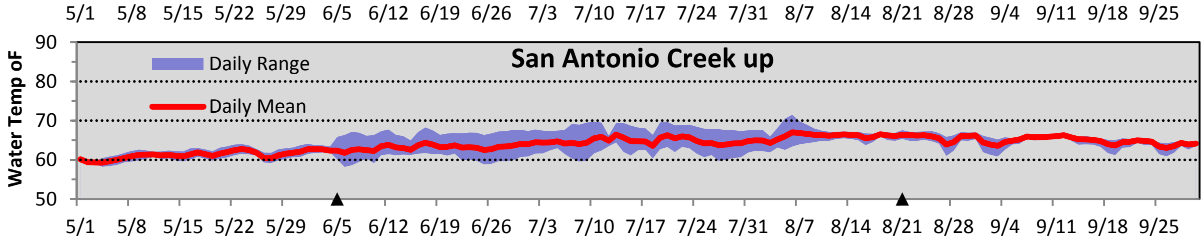

Mat 7b
Daily Range
Daily Mean
Water Temp oF
90 80 70 60 50
5/1 5/8 5/15 5/22 5/29 6/5 6/12 6/19 6/26 7/3 7/10 7/17 7/24 7/31 8/7 8/14 8/21 8/28 9/4 9/11 9/18 9/25

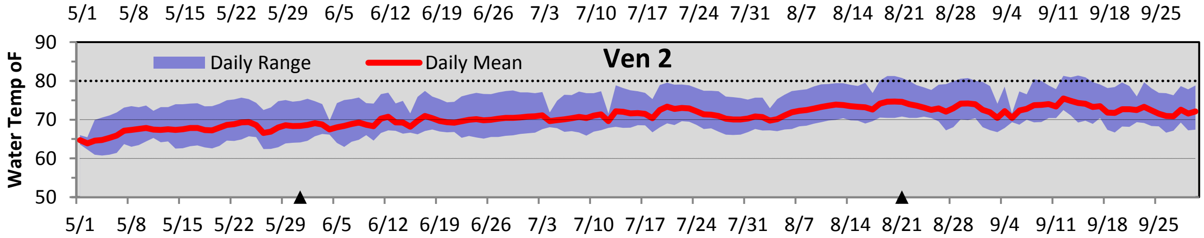

Mur 3
Daily Range
Daily Mean
Water Temp oF
90 80 70 60 50
5/1 5/8 5/15 5/22 5/29 6/5 6/12 6/19 6/26 7/3 7/10 7/17 7/24 7/31 8/7 8/14 8/21 8/28 9/4 9/11 9/18 9/25

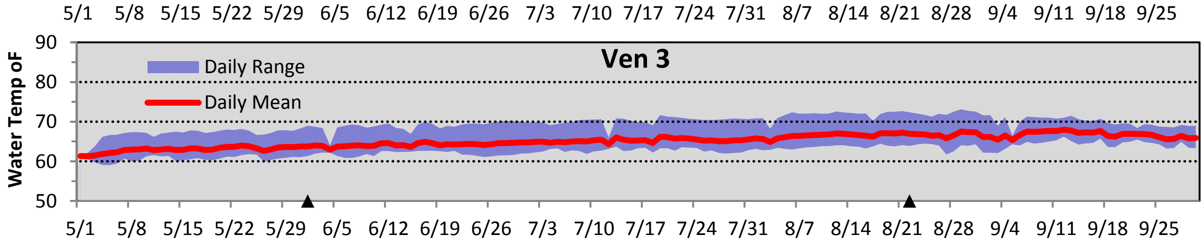

Mat 3
Daily Range
Daily Mean
Water Temp oF
90 80 70 60 50
5/1 5/8 5/15 5/22 5/29 6/5 6/12 6/19 6/26 7/3 7/10 7/17 7/24 7/31 8/7 8/14 8/21 8/28 9/4 9/11 9/18 9/25


Lower North Fork
Daily Range
Daily Mean
Water Temp oF
90 80 70 60 50
5/1 5/8 5/15 5/22 5/29 6/5 6/12 6/19 6/26 7/3 7/10 7/17 7/24 7/31 8/7 8/14 8/21 8/28 9/4 9/11 9/18 9/25


Ven 5
Daily Range
Daily Mean
Water Temp oF
90 80 70 60 50
5/1 5/8 5/15 5/22 5/29 6/5 6/12 6/19 6/26 7/3 7/10 7/17 7/24 7/31 8/7 8/14 8/21 8/28 9/4 9/11 9/18 9/25


San Antonio Creek up
Daily Range
Daily Mean
Water Temp oF
90 80 70 60 50
5/1 5/8 5/15 5/22 5/29 6/5 6/12 6/19 6/26 7/3 7/10 7/17 7/24 7/31 8/7 8/14 8/21 8/28 9/4 9/11 9/18 9/25


Ven 2
Daily Range
Daily Mean
Water Temp oF
90 80 70 60 50
5/1 5/8 5/15 5/22 5/29 6/5 6/12 6/19 6/26 7/3 7/10 7/17 7/24 7/31 8/7 8/14 8/21 8/28 9/4 9/11 9/18 9/25


Ven 3
Daily Range
Daily Mean
Water Temp oF
90 80 70 60 50
5/1 5/8 5/15 5/22 5/29 6/5 6/12 6/19 6/26 7/3 7/10 7/17 7/24 7/31 8/7 8/14 8/21 8/28 9/4 9/11 9/18 9/25

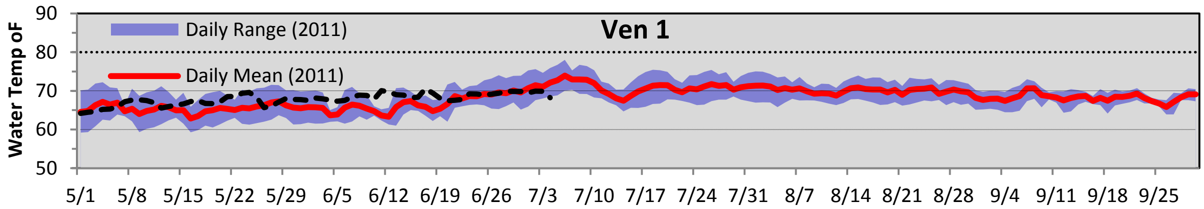


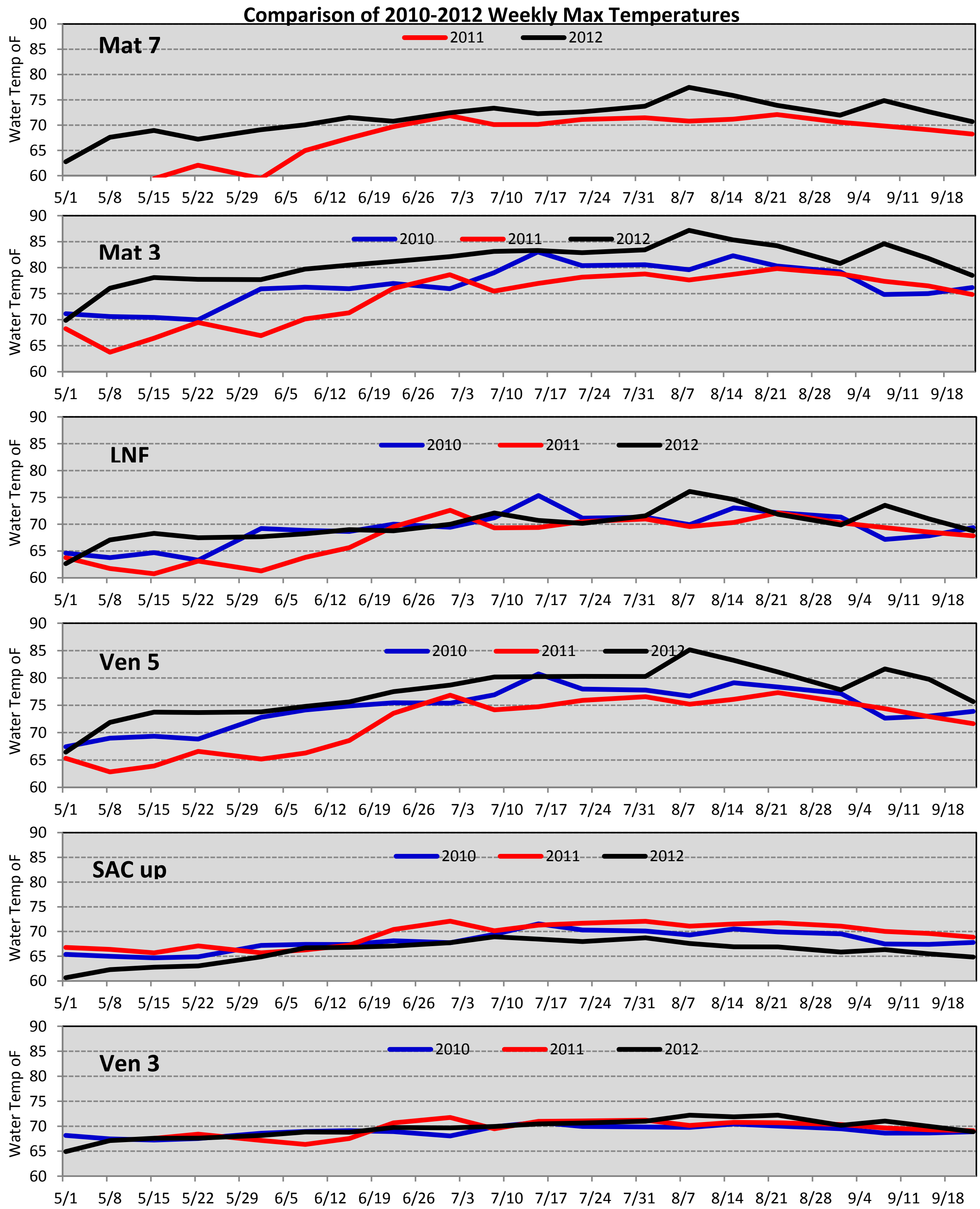
Figure 30. Weekly mean, minimum, and maximum water temperatures from May through September 2012. Black triangles indicate data downloading dates.

cooler and more constant temperature and flow regime may be due in part to the influence of San Antonio Creek, which enters at the upstream end of the study site, this desirable attribute of Ven 3 is probably most attributable to the rising groundwater which emerges from the long dry channel just upstream of the Ven 3 site (Figure 28).

The upper San Antonio Creek min:max data showed two anomalies (Figure 30), the first is the lack of daily temperature fluctuations in May, which may have been due to logger encapsulation within a dense mat of tree roots over the preceding winter and spring. This effect is suggested because immediately following retrieval and redeployment of the logger in early June (see black triangles showing download dates) the logger recorded more “normal” daily ranges in temperatures. In contrast, the sudden reduction in daily temperature fluctuations in early August was not an effect of root encapsulation, since the logger remained in the water column during the mid-August data download and redeployment. Instead, it is likely that flow went subsurface in a perennially dry channel in upper San Antonio Creek (Figure 28) with re-emergence of surface flow through the study reach, which would have resulted in a more constant temperature regime as seen in Ven 3.

Comparison of the mean of weekly maximum temperatures (MWMT) among study sites and between a dry water year (2012), a normal year (2010), and a wet year (2011) also shows several notable trends in temperatures (Figure 31), including the expectation that higher flows would result in cooler water temperatures. As noted in Table 13, base flows during those years were consistent with the rainfall-derived water year designations, with median base flows in the lower Ventura River (USGS gage #8500) of 1.8 cfs in 2012 (dry), 6.4 cfs in 2010 (normal), and 12.8 cfs in 2011 (wet). Although summer water temperatures will be influenced by other factors in addition to flow (e.g., variation in ambient air temperatures, lower river fog cover, annual riparian growth and shading, etc.), maximum temperatures were cooler under the higher flows in 2011 in comparison to the lower flows in 2012 at several sites, including the Ven 2, Ven 5, lower North Fork, Mat 3, and Mat 7 study sites. In contrast, temperatures were nearly identical in Ven 3 in all 3 years. On average, the weekly maximum temperatures from July through August were 5^{o}F higher in Ven 5 and Mat 3 during the dry year (2012) in comparison to the wet year (2011), whereas the difference between years was less than 0.5^{o}F in Ven 3.

Unlike the study sites listed above, the upper San Antonio Creek study site showed higher maximum temperatures during the wet year and lower maximum temperatures during the dry year (Figure 31). This anomaly may be associated with a higher proportion of rising groundwater during the drier 2012 season in comparison to the wetter 2011 season. In 2012, much of the surface flow with the upper San Antonio Creek study site was likely derived from rising groundwater coming out of the dry channel upstream of the study site, whereas in 2011 there may have been a higher proportion of warmer surface water flowing into the study site. Alternatively, minor differences in placement of the data logger between years (in the order of 10 ft), due to growth of alders and associated root masses, may have resulted in micro-scale differences in local groundwater effects at the logger location.



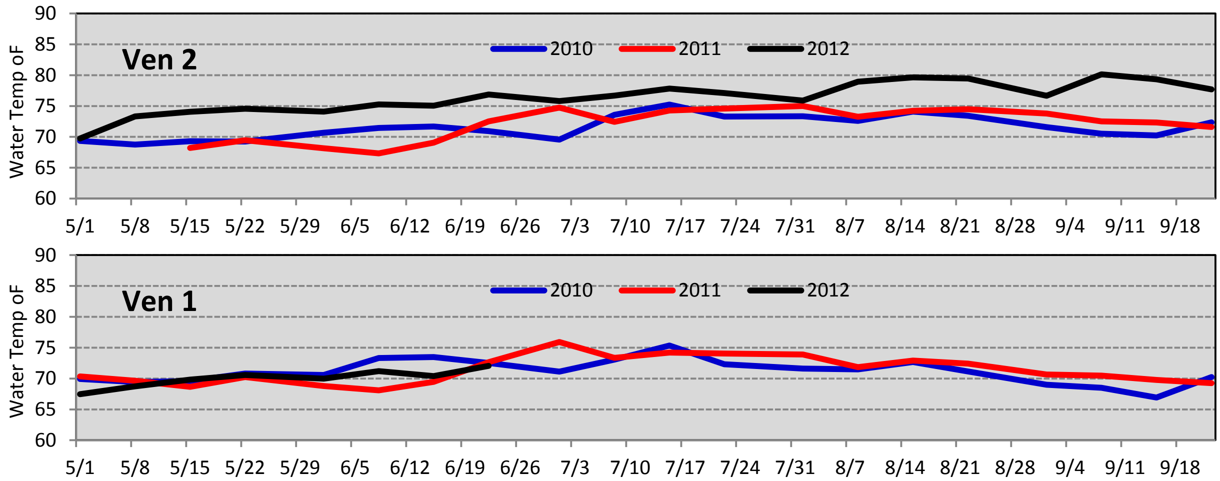


Figure 31. MWMT's from May through September in 2010 (normal year), 2011 (wet year), and 2012 (dry year).

Although the negative impacts of dry or intermittent channels on fish populations is well known, the beneficial cooling effect of subsurface flow on downstream surface waters is less commonly reported, but is readily evident in Ven 3 and also appears to be a regular benefit to *O. mykiss* inhabiting the upper San Antonio Creek, the Mat 5, and the upper North Fork study sites also, each of which are located below perennially dry channels (Figure 28). This cooling effect was also noted in Foster Park downstream of Ven 3 in 2009 where groundwater emerging into a pool below a 1,000 ft dry channel was approximately 2^{o}F cooler than the flowing reach above the dry channel (TRPA 2009b).

In study sites not influenced by rising groundwater, water temperatures regularly exceeded values listed as impairments to salmonid habitat. The U.S. Environmental Protection Agency (USEPA 2003) lists several temperature thresholds that would be expected to produce chronic effects on salmonid populations if exceeded, including a 68^{o}F (20^{o}C) MWMT threshold for adult migration, a 64.4^{o}F (18^{o}C) MWMT threshold for non-core juvenile rearing (assumed to also represent smolt migration), a 60.8^{o}F (16^{o}C) MWMT threshold for core juvenile rearing (representing locations with moderate to high juvenile densities), and a 55.4^{o}F (13^{o}C) MWMT threshold for spawning and egg incubation.

Comparing the wet year (2011) and dry year (2012) MWMT time series data from the 4 study sites containing the highest abundance of *O. mykiss* in the anadromous reaches below Matilija Dam (Ven 3, Ven 5, SACup, and LNF), with the 4 EPA temperature thresholds listed above, shows that the spawning/egg incubation and core juvenile rearing thresholds are almost always exceeded (Figure 32). The non-core juvenile threshold (assumed here to represent smolt migration) is generally met during the early half of the smolt migration period, but is frequently exceed in the latter half (April-May) migration period. In contrast, the higher threshold for adult migration (64.4^{o}F) is met at all sites, although late-emigrating kelts (post-spawned adults) might be expected to occur in the mainstem Ventura River in May or June when this threshold is exceeded (adult steelhead were observed in the lower Ventura River in July 2007, TRPA 2008).

Another EPA document lists 75.2^{o}F (24^{o}C) as a threshold that would be expected to totally eliminate salmonids from an area (EPA 1999). Spina (2007) also used a 75^{o}F (24^{o}C) MWMT criterion to evaluate temperature effects on juvenile steelhead behavior in southern California streams. Figure

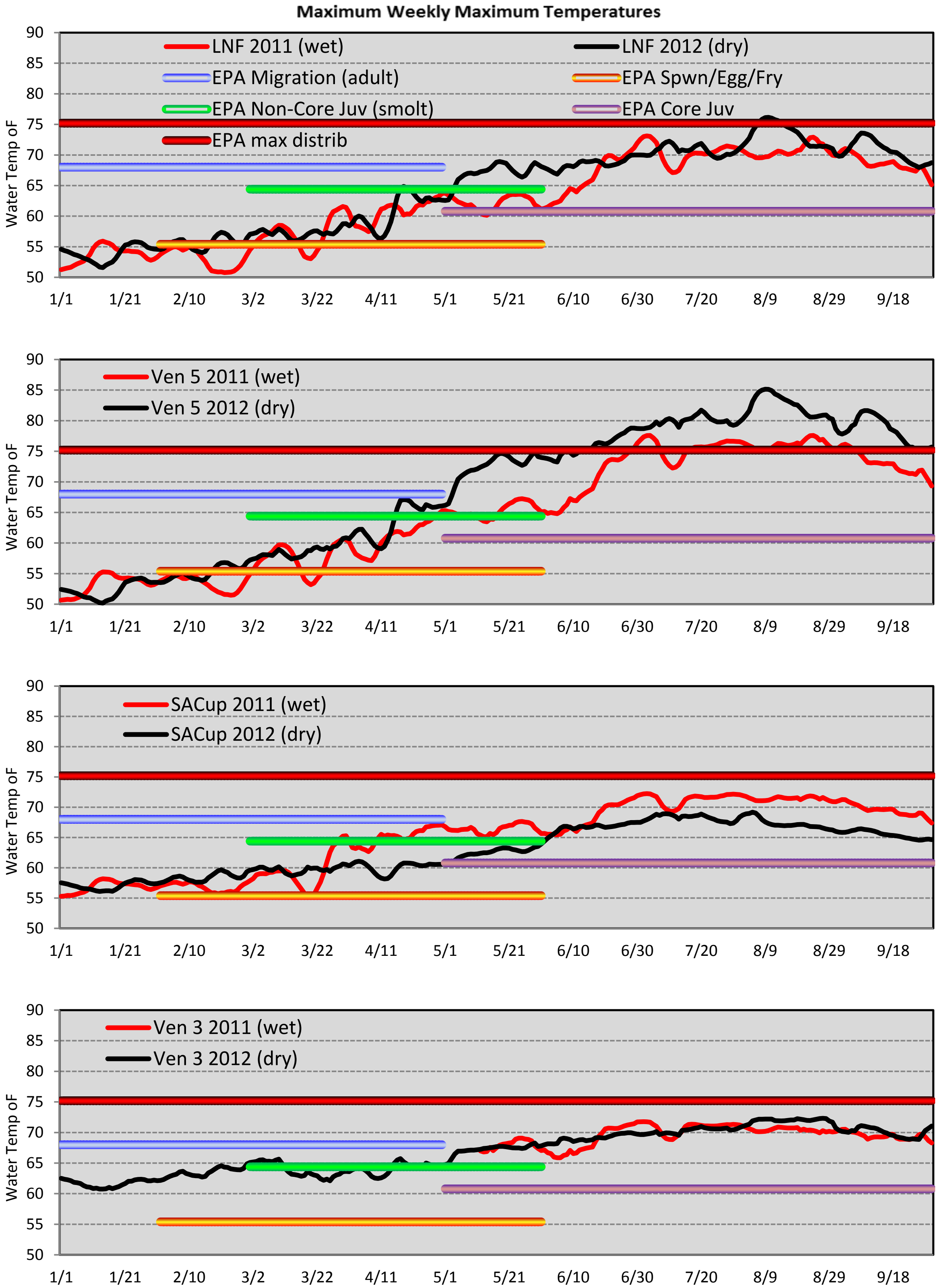


Figure 32. Comparison of annual MWMT's in a wet year (2011) and a dry year(2012) with EPA 2003 threshold temperatures for salmonid lifestages.

32 illustrates that this threshold was exceeded in the Ven 5 study site, where *O. mykiss* have been observed in low to moderate densities during most years of study. This threshold was also exceeded for extended periods of time in the Mat 3 study site in 2010, 2011, and 2012, as well as the Ven 2 study site in 2012 (Figure 31). The number of days when the daily temperature maxima exceeded the 75°F threshold by year and study site shows that the Ven 5 and Mat 3 study sites consistently exceed the threshold for durations lasting from several weeks to several months, whereas most study sites either did not exceed this threshold or it was exceeded only for short periods of time (Figure 33).

Although the 75°F MWMT threshold was exceeded for long durations in 2012 in the Ven 2, Ven 5, and Mat 3 study sites, fish sampling results indicate that *O. mykiss* fry were more abundant in all 3 of these reaches in 2012 (the warmest year) than in any preceding year (see Section 5.5 below), which suggests that the EPA's 75°F threshold may not be effective at distinguishing presence or absence of *O. mykiss* in southern California stream reaches. The relatively high densities of *O. mykiss* observed in the LNF and Ven 3 study sites in many years also suggests that the EPA threshold for core juvenile rearing (60.8°F) may not be representative of high quality habitat in southern California basins.

5.2 Physical Habitat Characteristics

The physical habitat characteristics measured or estimated in the Ventura River Basin showed substantial variation at both the spatial scale (between study sites) and at the temporal scale (between years).

5.2.1 Spatial Variation in Physical Habitat

Each of the 13 study sites exhibit differences in physical habitat, both at the mesohabitat scale (e.g., habitat types), and at the micro-habitat scale (e.g., depths, velocities, cover, etc.). In many of the study sites, each level II habitat type (pools, flatwaters, and riffles) are roughly equally represented at 20-40% by length (Figure 34). Pool habitats were more dominant, however, in the Ven 3 site (due to a single 976 ft pool) and in the higher gradient LNF, Mat 7, and Murietta sites; whereas pools were relatively infrequent in the two San Antonio Creek study sites. Flatwaters were dominant in the Ven 2 and SAC mid sites which contained long glide habitats, and in the Ven 4 site (when flowing) and the Mat 5 site, both of which contained long reaches of pocketwater habitat, which are classified as flatwaters despite the high complexity and gradient (Table 2). Riffles were the dominant habitat type only in the SAC up study site at 42%, but riffles were only infrequent in the LNF new study site, due in part to the construction of numerous swim dams which utilized cobbles and boulders at riffle locations to produce dams up to 6 ft in height (TRPA 2008). Non-sampleable habitats were typically rare (<5%), but where present were typically due to dense riparian vegetation, including *Arundo* (Ven3 and SAC up study sites), or due to high-gradient cascade habitats (Ven 5, Mat 7, and Murietta study sites).

Comparison of microhabitat characteristics collected for HSI analyses in 2006, 2007, and 2011 at most study sites (2011 only for SAC mid, 2010 and 2011 for SAC up, and 2012 only for Murietta) show large differences between study sites for some physical habitat parameters, but more moderate differences for others. Nevertheless, all means were significantly different among study sites for all variables (ANOVAs, $P<0.05$). As expected, median channel widths and thalweg depths were greatest in the mainstem Ventura River and (for widths) mainstem Matilija Creek study sites (Figure 35). The greatest median thalweg depth of 2.0 ft occurred in the lowest reach (Ven 1), due in part to the emergent aquatic vegetation (largely water primrose, *Ludwigia* spp., and watercress, *Rorippa* spp.), which effectively channelized flow in many habitats (Figure 36). Besides creating deeper and swifter riffles and flatwaters (a desirable attribute for rearing steelhead), the channelized areas also appeared more scoured of fines with improved substrate conditions. In contrast, the emergent aquatic vegetation was apparently attractive to carp which were commonly observed within the weedbeds, and during dry years the vegetation sometimes spread across the entire stream channel.

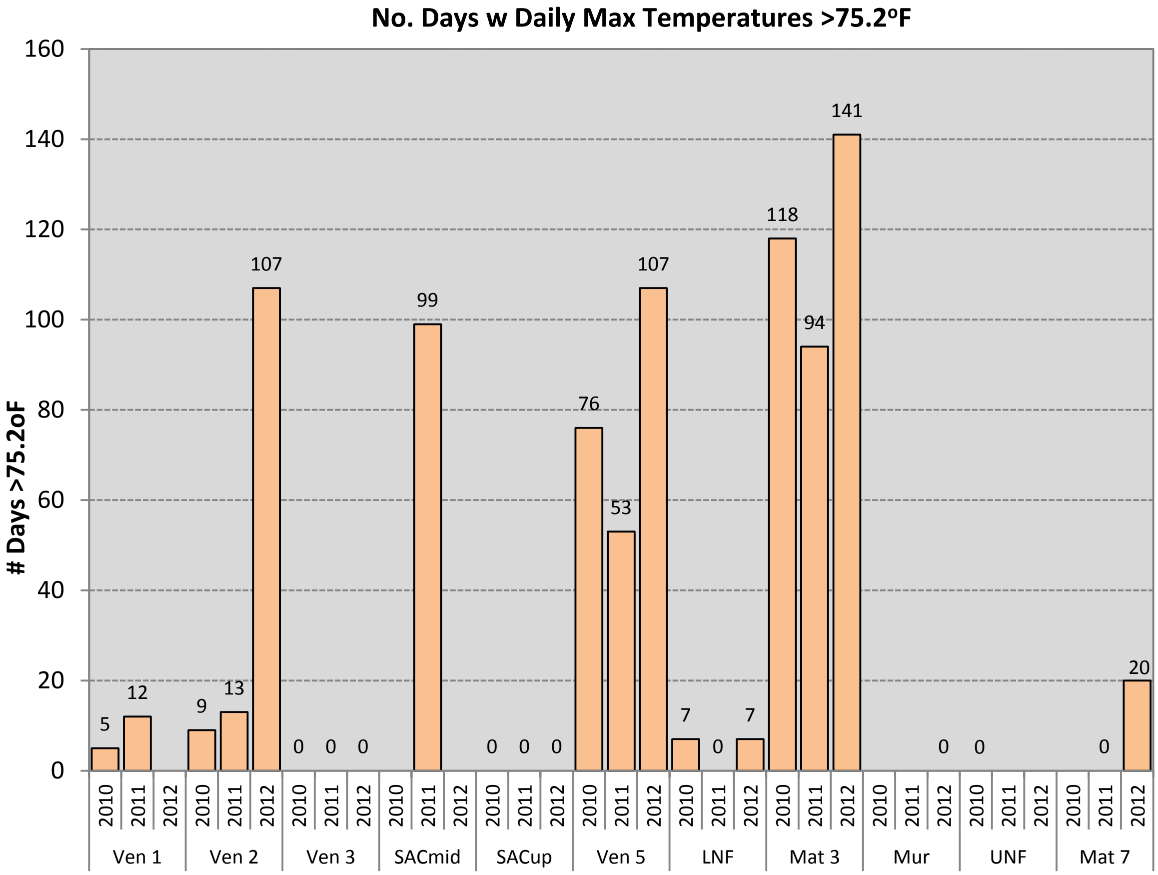


Figure 33. Number of days with maximum temperatures exceeding the 75.2oF (24oC) threshold by study site in 2010 (normal year), 2011 (wet year), and 2012 (dry year). Years without labels indicate no data.

Maximum thalweg depths (unit averages) over 6 ft occurred in Ven 3, Ven 4 (when not dry), and Ven 5, and Mat 7, with maximum pool depths equal to or greater than 10 ft in the Ven 4 and Mat 7 bedrock scour pools. The shallowest study sites were the two San Antonio Creek study sites, the upper North Fork study site, and the Murietta study site, all with median thalweg depths less than one foot (Figure 35). Those same four study sites also had the shallowest pool depths, with median maximum depths of <2 ft. The SAC mid study site also displayed the narrowest range and lowest variability in both thalweg and maximum pool depths, indicating very little diversity in depth characteristics.

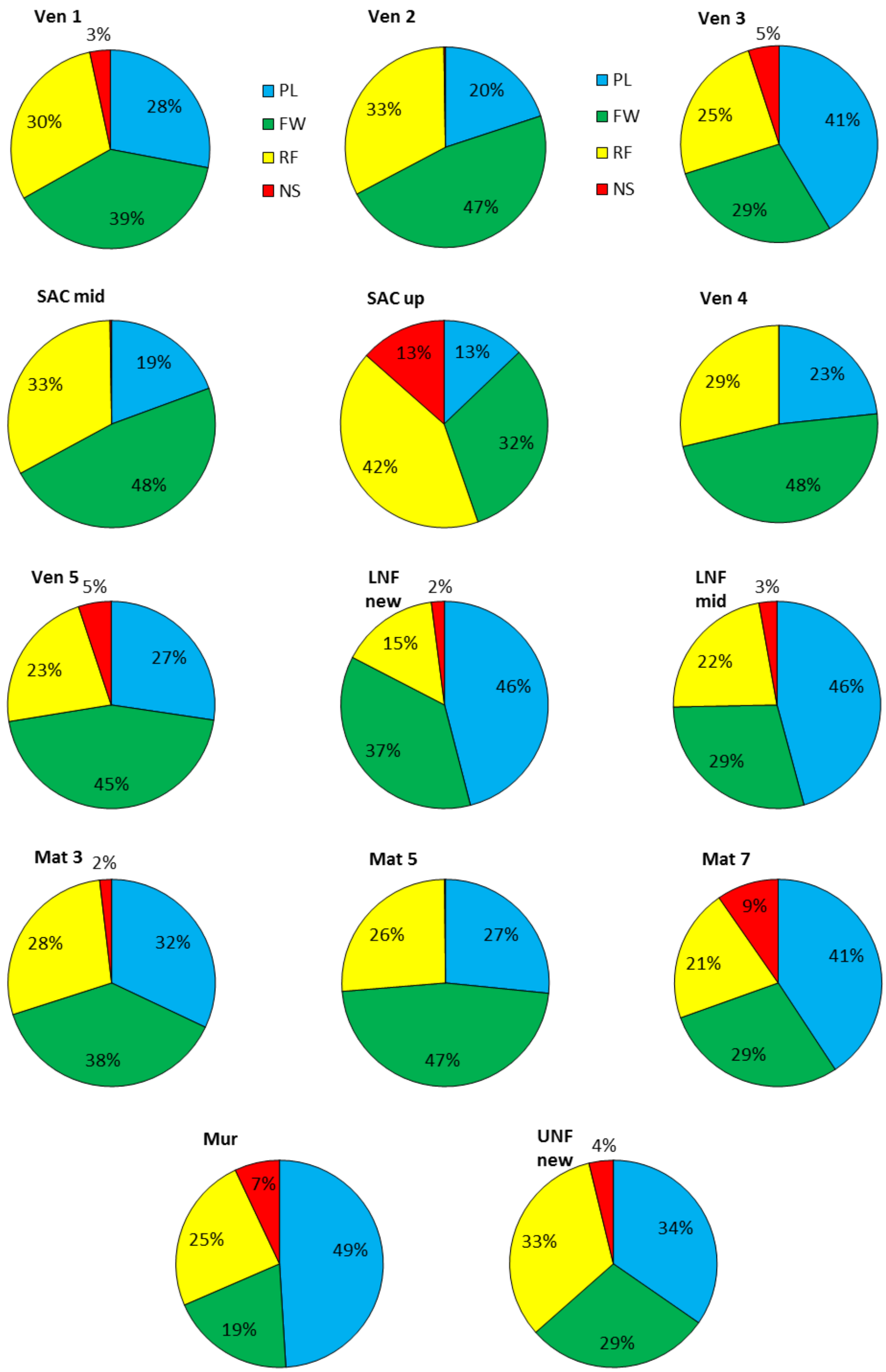


Figure 34. Relative proportion of level II habitat types in 2011 by study site (Murietta mapped in 2012).

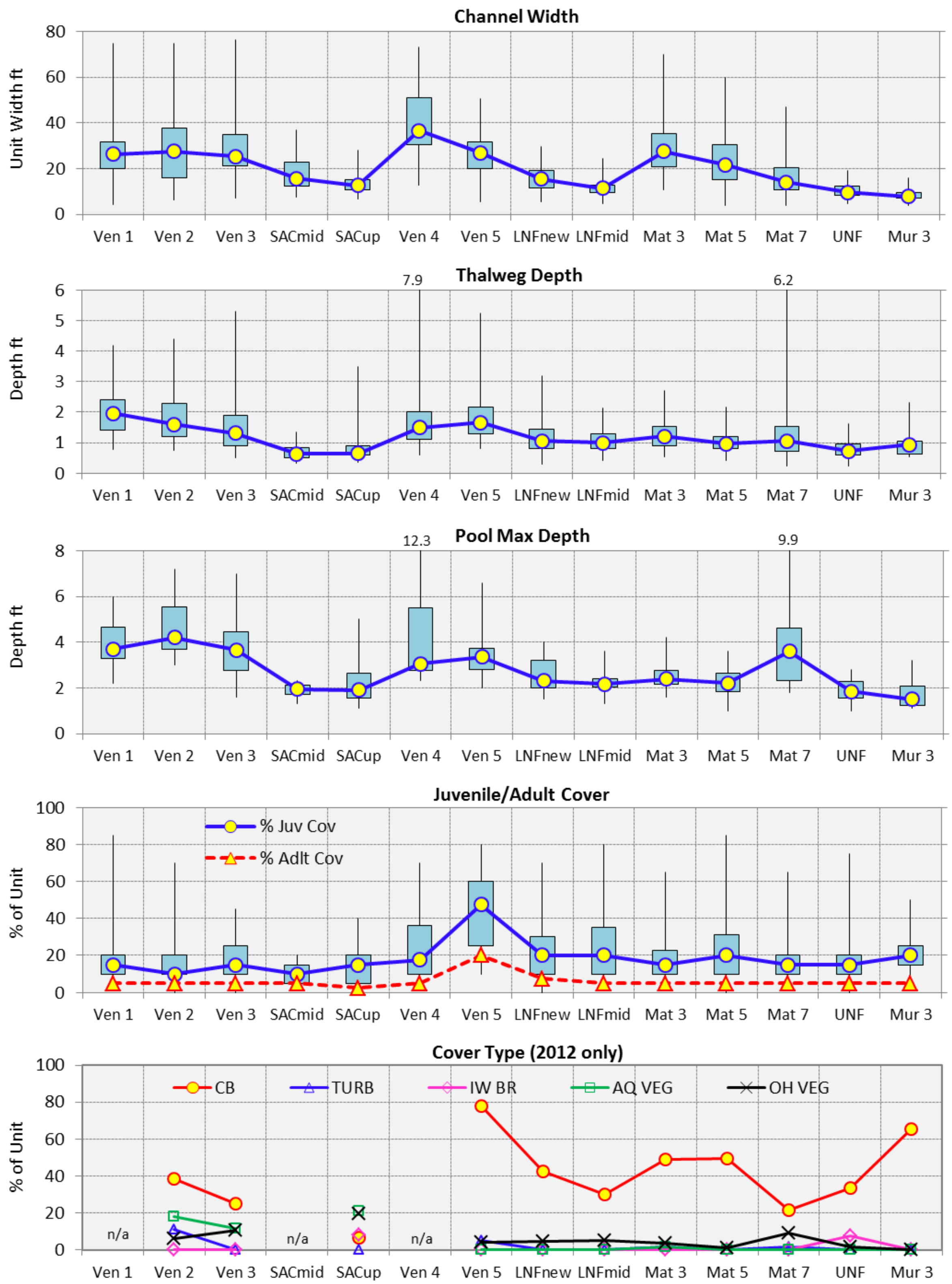


Figure 35. Comparison of physical habitat attributes according to study site (data combined across years). Circles are medians, boxes are quartiles, and whiskers are ranges. Percent adult cover and cover types are medians only.

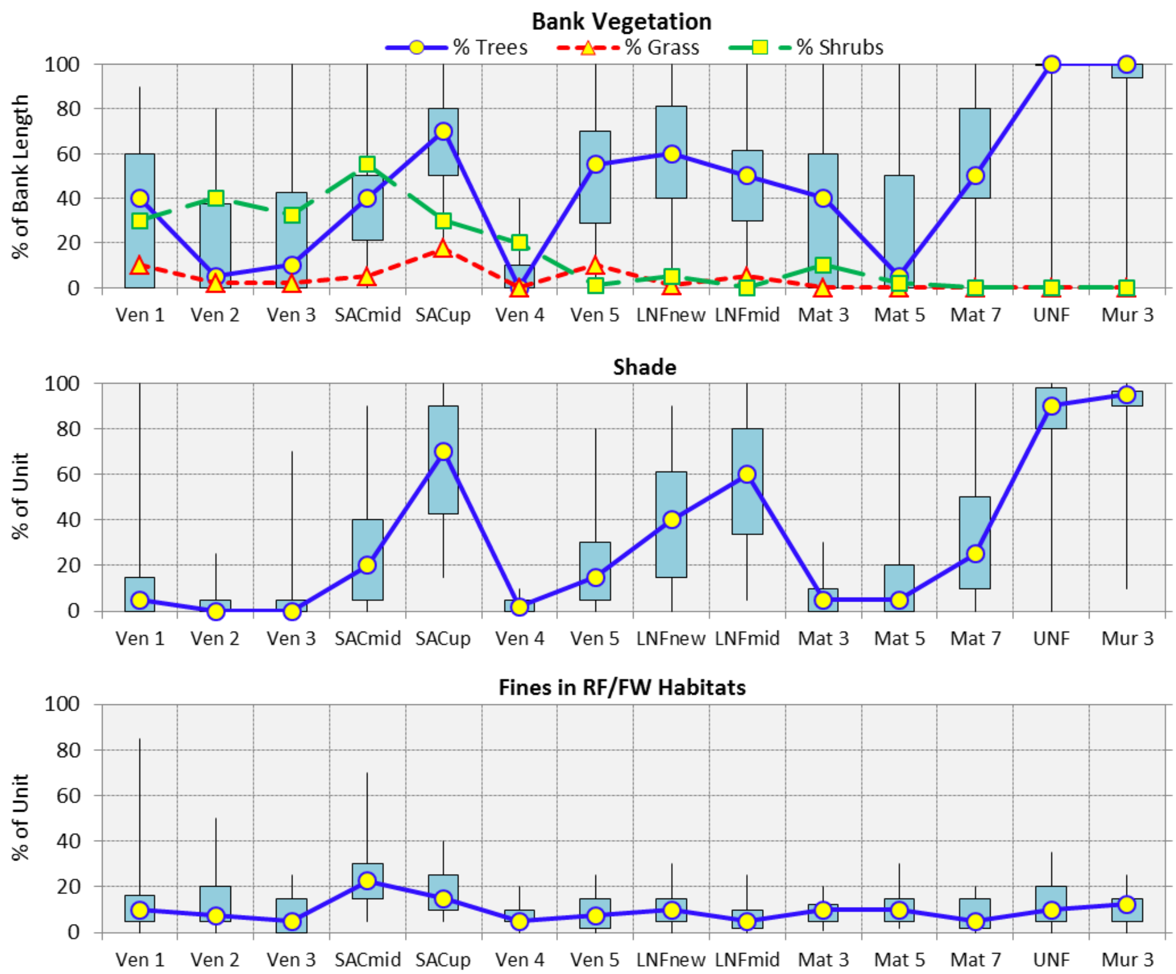


Figure 35. continued (% grass and % shrubs are medians only).

Figure 36. . Example of confined channels in the lower Ventura River due to water primrose (left) and watercress (right).

Ocular estimates of instream or overhead cover for juvenile *O. mykiss* were typically between 10% and 20% of the unit surface area, or 5-10% for adult resident fish (Figure 35). Highest values occurred for the higher gradient study sites including Ven 4, Ven 5, and the Lower North Fork study sites (Table 1), which contained more boulder substrate than did the lower gradient sites. Murietta Creek, which was only sampled during dry-year conditions in 2012, also contained relatively abundant cover due to higher gradient and associated boulder habitat. Measurements of the percentage coverage of specific cover types was conducted in 2012, which produced higher estimates of cover than the ocular method, and illustrated that cobble and boulder (CB) substrates were by far the most dominant cover type in the Ventura River Basin. Only the upper San Antonio Creek site was not dominated by substrate cover, but like the lower mainstem reaches it contained moderate amounts (10-20%) of aquatic vegetation (AQ VEG), overhead vegetation (OH VEG) within 18 inches of the water surface, and (SAC up only) in-water branches or woody debris (IW BR).

Comparison of riparian characteristics between study sites showed large differences in both vegetation type and degree of unit shading (Figure 35). The of proportion of bank length that contained trees were generally much higher in tributary study sites than in mainstem study sites, which also produced the highest shade values. Tree and shade coverage was nearly complete in the upper North Fork and Murietta study sites, with relatively high values in the upper San Antonio Creek and both lower North Fork study sites. In contrast, the percentage of trees and/or shading was lowest in the mainstem reaches of the Ventura River and Matilija Creek, particularly the Ven 4 and Mat 5 study sites which contained significant lengths of dry or intermittent channels, and in the lower three Ventura River study sites which possessed a high proportion of shrub species along the streambanks, thus providing little shade in the wide channels.

The median percentage of fines (sands, silts, and mud) in riffle and flatwater habitats was generally less than or equal to 10% (Figure 35), with higher levels in the two San Antonio Creek study sites and in the Murietta Creek study site (the latter was only sampled in 2012, a dry year). Some individual habitat units in the lower Ventura River contained high levels of fines, but the median values were probably lowered in part due to the encroachment of emergent vegetation which increased velocities in many habitats (Figure 36).

5.2.2 Annual Variation in Physical Habitat

Data collected during the 2006, 2007, and 2011 HSI assessments also allow comparison of temporal changes in habitat characteristics for the six study sites that were wetted each year and did not change locations. Annual changes in flow-related variables, such as channel width and thalweg depths, occurred as expected with wider and deeper channels in 2011 (a wet year) vs. 2007 (a dry year). On average, the percent increase in mean channel widths and thalweg depths between 2007 and 2011 was 61% and 24%, respectively. More substantial changes over time occurred in riparian characteristics, due to the scouring effect of the 2005 flood events and the subsequent regrowth of bankside vegetation. The mean percentage of tree coverage was nearly zero in the Ven 2 and Ven 3 study sites in 2006, one year following the 2005 flood events, but increased to over 40% coverage by 2011 (Figure 37). The annual change in percent tree coverage was highly significant for the mainstem Ventura River study sites (ANOVAs, $P<0.05$), however the change in percent shade was not significant, undoubtedly due to the wide stream widths where most of the channel was not overlaid by tree canopy. This relationship was essentially reversed in the upstream study sites, where the % trees did not differ significantly between years in LNFmid and Mat 5, but the change in

% shade was significant for both study sites as well as Mat 3. The decrease in % trees in 2012 at the LNF mid study site was anomalous and contrary to the increase in shading.

Another habitat parameter that showed a consistent change over time was the percentage of fines in riffle and flatwater habitats. Fines remained unchanged from 2006 to 2007, but mean values from 2011 averaged 165% of the 2006 values (Figure 37) and the differences were highly significant at most study sites (ANOVAs, $P<0.01$). It is presumed that the high flows in 2005 scoured fines from most reaches, and in the absence of major channel changing flows since then (Figure 26), the proportion of fines has been steadily increasing in most sites. The annual changes were not significant in Ven 2 due to high variation between units, nor in LNFmid, perhaps due to higher gradient. The change in fines was highly significant in the Ven 5 study site, possibly due to recruitment of fines from the quarry located just upstream in the lower North Fork.

5.3 FWS Habitat Suitability Index (HSI) Scores

Habitat suitability index (HSI) scores were developed using the USFWS model (Raleigh et al. 1984) with modified curves, for 11-13 study sites in 2006, 2007, 2011, and 2012 (Table 1). An alternative model (the SS HSI model) was also developed using 12 study sites in 2012. HSI models were also developed for many of these same study sites in 2003 (TRPA 2004), however comparative fish sampling was not conducted in that year and consequently this report will only present HSI data from 2006 to 2012. The HSI results described in previous reports (TRPA 2007, 2008) utilized one specific model option (equal components model without variable limitations) for a given year's habitat and fish abundance data, based modified curves for several HSI variables (Figure 5). However this assessment will compare the different model options presented in Raleigh et al. 1984 (e.g., equal components, compensatory, and non-compensatory with or without variable limitations), with and without the modified curves, and will also compare HSI scores with *O. mykiss* density over a single year (2012) and over a multi-year period, using mean HSI scores and mean density over the four HSI sample years.

5.3.1 Comparison of HSI Model Options

Equal vs. Unequal Components Models

As described in Section 4.2.3, the unequal components model options either allow for some variables to compensate for limitations in other variables (the compensatory option), or else the limiting variables will constrain the overall score (non-compensatory option). Figure 38 illustrates that the equal components model produced the highest HSI scores, whereas the non-compensatory model, as expected, produced the lowest HSI scores. Although the trend in scores between study sites was similar for all three model options, the equal-components model produced the least difference between the maximum score (0.88) and the minimum score (0.63, excluding the zero score for Ven 4), a difference of 40%. In contrast, the non-compensatory option produced the greatest difference with a maximum score (0.55) almost three times the minimum score (0.19).

Comparison of these scores with actual *O. mykiss* abundance will occur in a later section of this report (Section 5.6), however comparison of scores by subjective assessment of habitat quality suggests that the equal-components model scores study sites too high, whereas the non-compensatory model scores study sites too low.

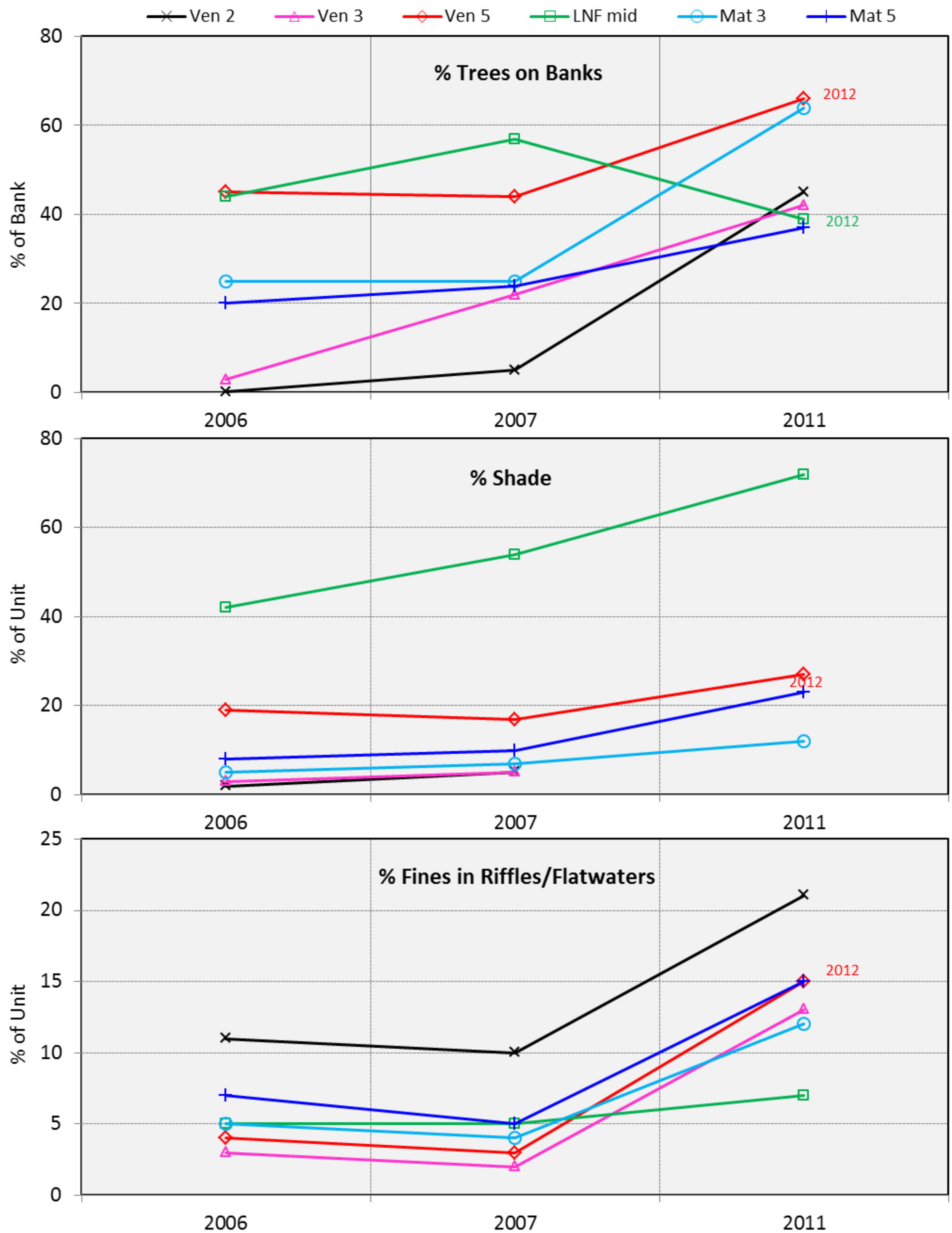


Figure 37. Mean percent tree coverage, percent shade, and percent fines in 2006, 2007, and 2011 (except where noted) according to study site.

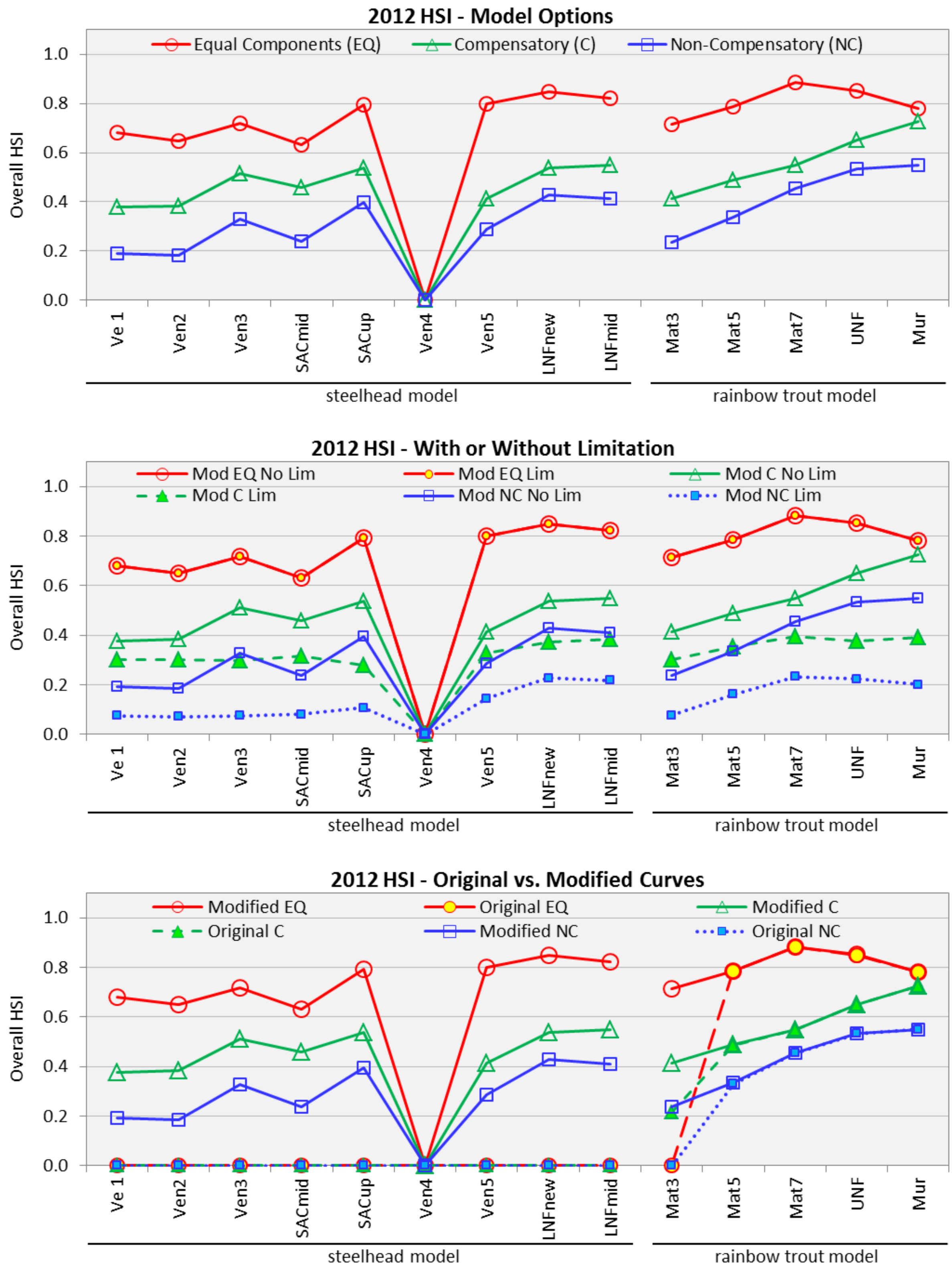


Figure 38. Comparison of 2012 HSI scores using different model equation or HSI curve options.

Inclusion of Variable Limitation Option

The USFWS HSI model suggests that if an individual variable score is less than or equal to 0.4, the low variable score can be assigned to that component, acting as another form of habitat limitation (Raleigh et al. 1984). In like manner, any calculated component score $\leq$0.4 can be assigned to the overall HSI score. In previous HSI assessments (TRPA 2004, 2007, and 2008), this "variable limitation" option was not utilized when calculating the equal-components HSI scores; however this option did not affect the 2012 overall HSI scores when using the equal-components model (Figure 38). The variable limitation option did reduce the overall HSI scores using the compensatory and non-compensatory models, and served to reduce variability in HSI scores between models. The homogenized HSI scores were particularly evident for the compensatory model, where all but one study site (excluding Ven 4) produced HSI scores between 0.3 and 0.4, which is contrary to the consistently observed differences in *O. mykiss* densities between study sites (Section 5.5).

Application of Modified HSI Curves

The original HSI model developed for the Ventura River Basin in 2003 recognized that many of the HSI curves presented in the USFWS publication (Raleigh et al. 1984) did not appear applicable to *O. mykiss* inhabiting streams in southern California basins, particularly those associated with water temperatures. The HSI authors encouraged researchers to modify HSI curves where appropriate to better represent habitat suitability in specific regions; consequently several HSI curves were modified using professional judgment prior to application in the Ventura River Basin (TRPA 2004). The 2012 HSI data was utilized to calculate and compare HSI scores using the modified curves (Figure 5) versus the original, unmodified curves presented in Raleigh et al. (1984).

As expected, many of the HSI scores were reduced to zero when using the original HSI curves (Figure 38). All scores from the anadromous zone produced zero scores due excessive water temperatures for smolt emigration (Ven 3, SAC up, and both LNF sites), or for both smolt emigration and juvenile rearing (Ven 1, Ven 2, Ven 4, Ven 5, and SAC mid). The Mat 3 HSI score was also reduced to zero using the equal-components and non-compensatory models, due to excessive juvenile rearing temperatures (the resident trout model does not use smolt temperature). Many of the above study sites consistently harbored juvenile and adult *O. mykiss* during the summer months, some at relatively high densities, which further demonstrated the need for curve modifications. HSI scores in the other resident trout study sites above Matilija Dam were essentially unaffected by the curve modifications.

5.3.2 Annual and Spatial Variation in HSI Scores

Comparison of HSI scores over four years (2006, 2007, 2011, and 2012), using the equal components/no limitations model for the six study sites that remained unmoved in each year, showed more variability in some of the component scores than in others (Figure 39). The spawning *Vs* score, which is a sub-component of the embryo score, clearly showed the greatest annual variation, whereas annual variation in the adult, juvenile and fry component scores were typically minor. Previous sensitivity analysis indicated that the *Vs* score was highly influential on the overall HSI score (TRPA 2007), which is further demonstrated where the year with the lowest *Vs* score also produced the lowest overall HSI score in five of the six study sites. The *Vs* scores were also highly

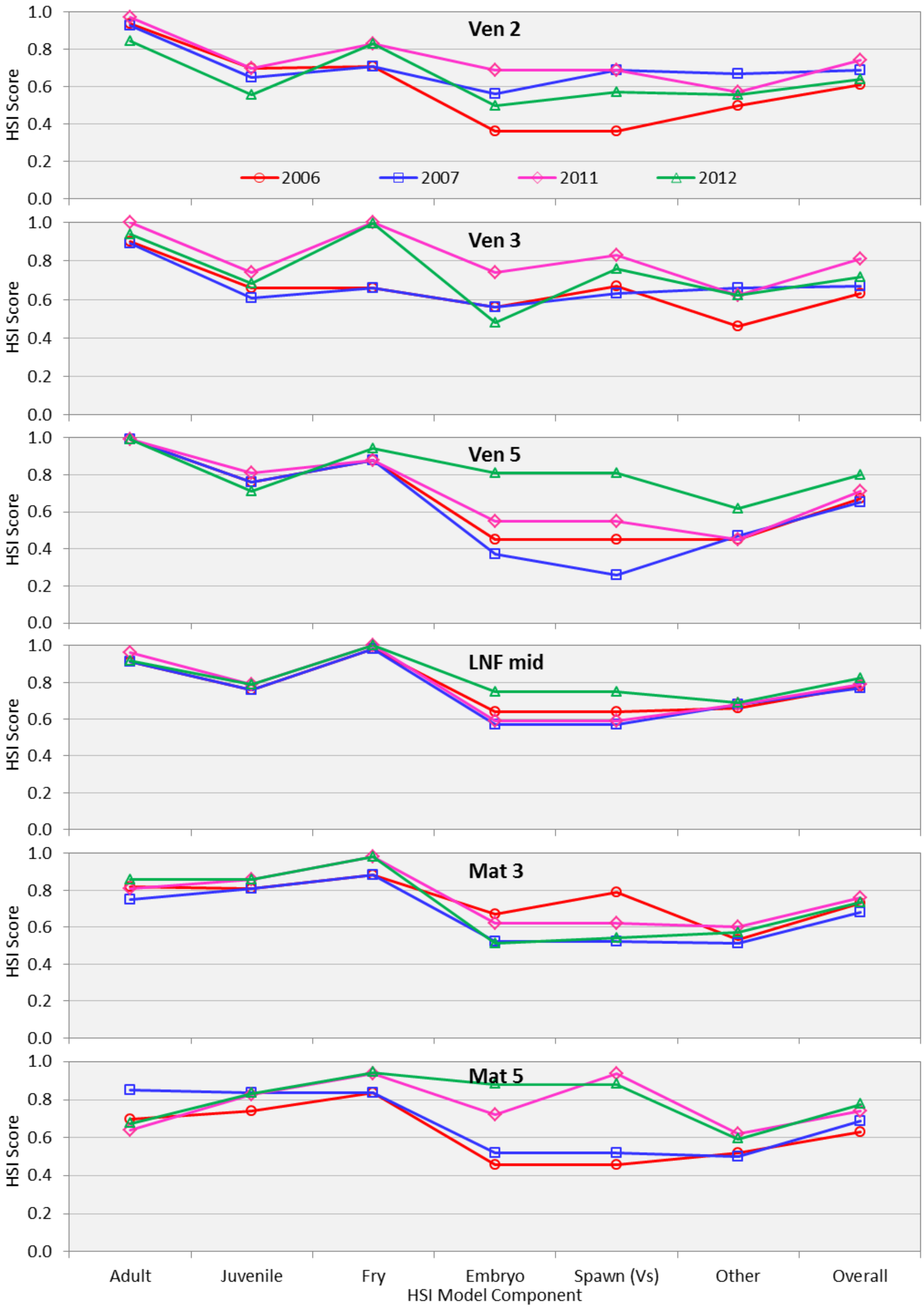


Figure 39. Comparison of component and overall HSI scores across years for six study sites.

variable across years and study sites due in part to the relative lack of spawning gravels in many sites and subsequently small sample sizes for estimating the suitability of spawning habitat. This effect lead to disparity in annual *Vs* estimates in several of the study sites (e.g., Ven 2, Ven 5, Mat 3, Mat 5). Although annual variability in the overall HSI score was generally far less than the variability in the *Vs* sub-component score, maximum overall HSI scores were up to 33% higher than minimum scores in sites with variable Vs values. In contrast, only a 7% difference in the range in HSI scores was evident in the LNF mid study site where *Vs* was relatively consistent and represented by numerous spawning patches. Although not shown in Figure 39, the other tributary study sites that also contained relatively abundant spawning habitat (but having just three years of HSI data) produced max:min differences in overall HSI scores of 11%, 15%, and 6% in the LNF new, SAC up, and UNF study sites, respectively.

Comparison of overall HSI scores from all study sites (Table 14) showed relatively consistent trends among years, with highest scores in the tributary study sites (SAC up, LNF, and UNF) and lower scores in the mainstem Ventura River, the mainstem Matilija Creek, and the SAC mid site (Figure 40). As noted above, annual variability was greater in some sites, such as Ven 3, Ven 5, Mat 5, and (especially) Mat 7. With the exception of Ven 4, which was frequently dry and thus yielded by far the most variation in scores (measured by the Coefficient of Variation, or C.V.), most study sites produced C.V.'s of annual HSI scores between 5% and 10%. C.V.'s were less than 5% for the two SAC study sites (based on only two HSI scores), the LNF mid site, the Mat 3 site, and the UNF site, but was over 15% for the Mat 7 study site. The relatively high C.V. for the Mat 7 site may have been partly due to the change in location, where the 2011 and 2012 scores were based on a reach approximately ¼ mi upstream of the reach sampled in 2006 and 2007. The difference in HSI scores between the alternative Mat 7 study sites appeared to be largely due to the higher *Vs* sub-component score calculated for the upper study site.

Table 14. Overall HSI scores by study site and year, with average scores (recalculated HSI using mean values for each variable) and C.V. of annual scores.

Study Site	2006	2007	2011	2012	Avg HSI	C.V.
*Ven 1	0.61	0.63	0.75	0.68	0.72	9.3
Ven 2	0.61	0.69	0.74	0.64	0.71	8.5
Ven 3	0.63	0.67	0.81	0.72	0.72	10.9
SAC mid	n/a	n/a	0.65	0.61	0.63	4.5
SAC up	n/a	n/a	0.77	0.79	0.77	2.1
Ven 4	0.60	0.00	0.73	0.00	0.35	117.5
Ven 5	0.67	0.65	0.71	0.80	0.74	9.4
*LNF new	0.73	0.77	0.82	0.86	0.77	7.0
LNF mid	0.78	0.77	0.79	0.82	0.75	2.9
Mat 3	0.73	0.68	0.76	0.73	0.72	4.6
Mat 5	0.63	0.69	0.74	0.77	0.73	8.8
*Mat 7	0.63	0.71	0.89	0.89	0.79	16.6
*UNF	0.87	0.82	0.87	0.87	0.84	2.9
Mur	n/a	n/a	n/a	0.78	0.78	n/a

* study sites changed location in 2007 (LNF new & UNF) or 2011 (Ven 1 & Mat 7)

It should be noted that not all HSI variables were re-measured each year, consequently true annual variation in HSI scores would be expected to be somewhat greater than observed in Figures 38 and 39. For example, the water temperature HSI used in the 2006 and 2007 HSI assessments were both derived from Stream-Team spot measurement data, whereas the temperature HSI used in 2011 and 2012 were based on year-specific data from continuous temperature dataloggers. Estimated values and HSI scores for dissolved oxygen, pH, and annual flow variables (V14 and V18, Table 3) were held constant in all four years, whereas thalweg depths and most riparian characteristics were re-measured each year. Several variables, including riffle and pool class designations, winter substrate, bank stability, and juvenile or adult cover estimates were held constant between 2011 and 2012.

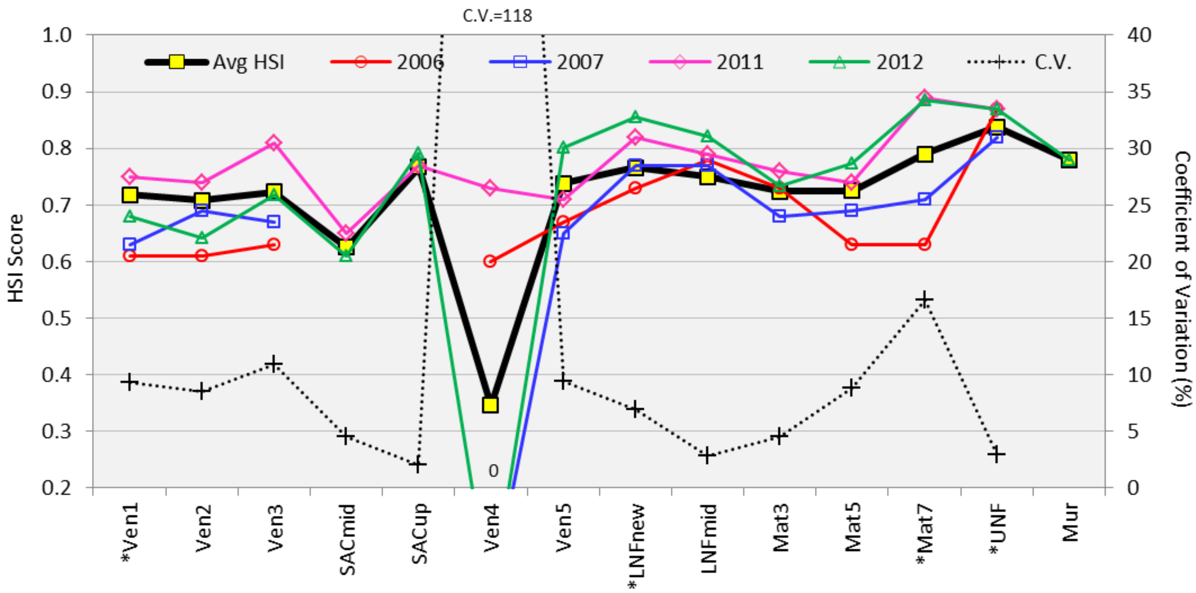


Figure 40. Comparison of overall HSI scores between study sites and years, along with mean HSI scores and C.V.'s of annual differences in scores.

5.4 Southern Steelhead (SS) Habitat Suitability Index (HSI) Scores

Like the USFWS HSI model, the southern steelhead (SS) HSI model is arranged in components, including a habitat unit component, a reach component, a recruitment component, a water quality component, and a migration component (see Section 4.3 for model formulation). The overall SS HSI score for fry is based on the habitat unit, reach, recruitment, and the water quality components, and is identical for either resident rainbow fry or steelhead fry. The HSI for stream-resident juvenile *O. mykiss* is based on the habitat unit, reach, and water quality components, and adds a smolt migration component to the resident score to represent juvenile steelhead. The adult rainbow SS HSI score is likewise based on the habitat unit, reach, and water quality components, and adds a migration component to represent adult steelhead.

5.4.1 Habitat Unit Component

The habitat unit component produces an HSI score that rates the quality of habitat units according to fish size class, e.g., fry (rainbow or steelhead), juvenile (rainbow or steelhead), and adult (rainbow only). Scores are also calculated separately by channel size (mainstem or headwater/tributary) and mesohabitat type (pool, flatwater, or riffle). The habitat unit HSI score is combined across channel and mesohabitat types to produce unit component HSI scores for each size class, due to the perceived difference in importance of different mesohabitat types for each size class (i.e., riffles of highest importance for *O. mykiss* fry, pools for adult rainbows).

Comparison of habitat unit component scores between study sites showed the greatest variability for fry and the least variability for adult rainbows (Figure 41). Fry habitat unit HSI scores were most heavily weighted by the scores for riffles, and least weighted by pool scores (Table 11), due to the consistently higher densities of fry observed in riffle habitats. Habitat unit HSI scores were highest for fry in the six tributary study sites (SAC mid, SAC up, LNF new, LNF mid, UNF, and MUR), and lowest in the warmest mainstem reaches (Ven 4, Ven 5, and Mat 3). Ven 4 produced a zero score for all three size classes due to the lack of surface flow, and the low scores for fry in Ven 5 and Mat 3 were principally due to the relatively slow velocities and shallow depths of riffles, and the lack of velocity within pool habitats, both factors associated with the low flows and drought conditions experienced in 2012.

Habitat unit component scores for juvenile *O. mykiss* were highest in the Mat 7 and UNF study sites, but lowest in the two SAC study sites (Figure 41). The relatively low abundance of cobble/boulder cover in San Antonio Creek riffles appeared to be the primary variable that produced low scores for these two sites, along with shallow depths and low proportions of in-water branches in the flatwater habitats.

HSI scores for the habitat unit component were relatively similar between study sites for adult rainbow trout, with highest scores in the Ven 2 and Ven 3 study sites and lowest scores in the Mat 3, Mat 5, and UNF sites (Figure 41). The habitat unit HSI scores for adult resident trout are most heavily weighted by the quality of pool habitats (Table 11), which in the mainstem reaches was based solely on pool maximum depth (Table 9). In tributary reaches, pool quality for adult rainbow trout was estimated using the proportion of depths >2 ft, the proportion of velocities $\geq$0.5 fps, and the pools average velocity. For mainstem reaches, pool maximum depths were greatest in Ven 2 and Ven 3 (also in Ven 4 during wet years); consequently those study sites yielded the highest habitat unit component scores. In the tributary sites, the lower scores in the Mat 5 and UNF sites were likewise mostly associated with shallow pool depths.

5.4.2 Reach Component

The reach component HSI scores were estimated based on seven variables (Figure 7), two relating to physical habitat (gradient and shading), two related to biological variables (BMI and predation), and three relating to flow (persistence, accretion, and valley width). Unlike the habitat unit component, the reach component was not estimated separately for size class or channel size, and was not based on regression models, but instead used a formula consisting of variable minima and geometric means (Equation 2). The calculated reach component scores were highest in the upper tributary study sites (both LNF sites and the UNF site), and lowest in the lower mainstem sites (Ven 1, Ven 2, Ven 3) and in the SAC mid study site (Figure 41). The low scores in the Ven 2 and Ven 3 sites were primarily due to the low gradient and the low shade values, whereas the Ven 1 study site also possessed a low BMI score. The SAC mid study site had low scores due to low gradient, low BMI, and low flow persistence. In contrast, the LNF and UNF study sites were near optimal in all reach-scale variables. The Mat 7 score was slightly degraded due to a lower shade value, whereas the Murietta score was reduced due to flow persistence.

5.4.3 Recruitment Component

The recruitment component was only applied to the fry life stage and was intended to represent habitat and quality for spawning and water quality for egg incubation, with an additional variable (tributary effects) intended to account for reaches that lack spawning habitat but rear lots of fry due to a nearby spawning tributary (Figure 7). This component was utilized for calculating the fry HSI due to the expected correlation between spawning success (or trib recruitment) within a reach and

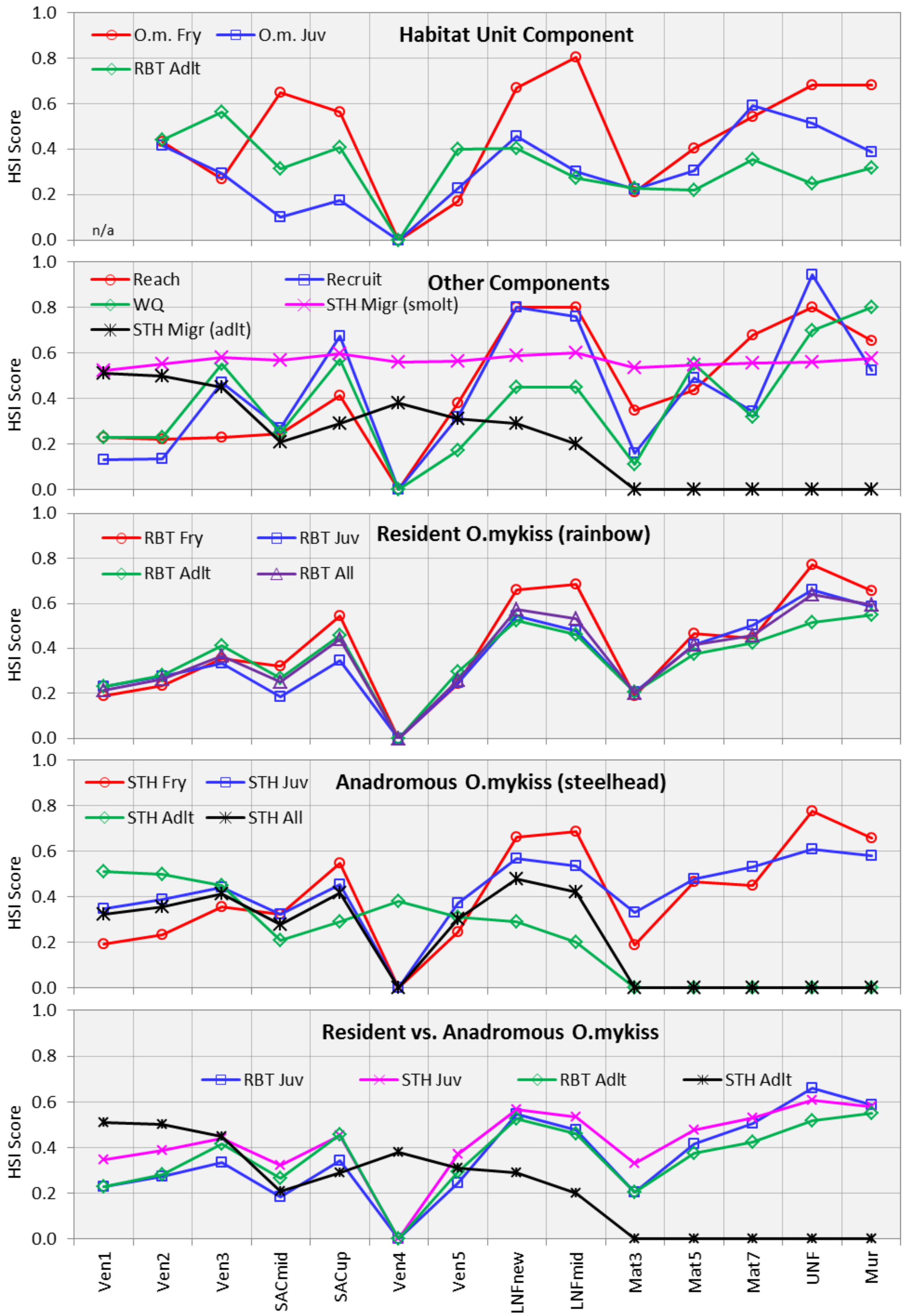


Figure 41. . Comparison of SS HSI scores among study sites according to model components, size class, and life-history form.

the abundance of fry. Not unexpectedly, the recruitment component produced high HSI scores for the SAC up, both LNF, and the UNF study sites (Figure 41), all sites that have shown an abundance of spawning habitat and relatively high densities of *O. mykiss* fry. Low recruitment scores occurred for the lower two Ventura River study sites as well as for the lowest Matilija Creek study site. The low scores were mostly due to low percentages of spawning gravel having only moderate quality, and relatively warm incubation temperatures.

5.4.4 Water Quality Component

This component compliments the water quality variables used in the recruitment component by representing temperature and oxygen requirements over the harsh summer months in southern California steelhead streams (Figure 7). The two warmest study sites in the Ventura Basin (Ven 5 and Mat 3) produced the lowest water quality component scores (Figure 41). In contrast, two of the highest elevation study sites (UNF and Mur) and two study sites that receive upwelling groundwater (Ven 3 and SAC up) all had cooler water temperatures and subsequently higher component scores.

5.4.5 Migration Component

HSI scores for juvenile steelhead were calculated by combining the juvenile *O. mykiss* score (representing pre- or non-migratory juveniles) with smolt migration variables (Figure 7). Unlike the juvenile steelhead HSI score, the HSI for adult steelhead was not based on the complimentary resident trout HSI score, since adult steelhead only reside in freshwater for a relatively short time. Instead, the adult steelhead score was based on several variables associated with passage, water quality, and migration distance.

The smolt migration component showed very little variability, with scores ranging from 0.52 in Ven 1 to 0.60 in the LNF mid study site (Figure 41). Smolt migration scores do not include a barrier variable as does the adult migration component, consequently the smolt HSI scores for the reaches above Matilija Dam assume safe passage, however those scores are slightly degraded by the high migration temperatures encountered in the lower Matilija mainstem. Also note the non-zero HSI score for Ven 4, which assumes surface flow during the winter/spring smolt migration period (as do all upstream study sites). The adult migration component did account for barriers to upstream migration; consequently all scores for study sites above Matilija Dam were zero. Lower scores also occurred for the SAC mid study site due to shallow riffle depths, and also for the LNF study sites due in part to potential partial barriers below both study sites. The higher HSI scores for adult steelhead in the lower Ventura study sites were associated with few passage impediments, a short migration distance and the abundance of deeper holding pools.

5.4.6 Size Class HSI Scores

The component HSI scores are combined to calculate size class HSI scores, as described above and in Section 4.3.6. The lowest HSI scores for *O. mykiss* fry occurred in Ven 1, Ven 2, and Mat 3 (excluding the dry Ven 4 study site) at about 0.2 (Figure 41). The Ven 1 and Ven 2 study sites had low fry scores for the reach, water quality, and recruitment components, whereas the Mat 3 scores were lowest for the habitat unit, recruitment, and water quality components. The highest fry HSI scores for fry (as well as juvenile and adult size classes) occurred in the tributary and headwater study sites, with the exception of SAC mid. The SAC mid study site produced the lowest HSI score for juvenile *O. mykiss* (after Ven 4), largely due to the low habitat unit score. Juvenile HSI scores were also low for the Ven 5 and Mat 3 study sites, largely due to very low water quality component scores. The Ven 1 and Mat 3 study sites also showed the lowest HSI scores for adult resident trout, the latter due to the water quality component score and the former due to low reach and water quality scores.

Combining the HSI scores for *O. mykiss* fry, juvenile, and adult to represent all resident rainbow trout produced HSI scores intermediate in nature but still following the trend of high scores in the tributaries and lower scores in the warmer mainstems (and SAC mid).

Adding the migration variables to the *O. mykiss* HSI scores to represent steelhead smolts and adults improved the HSI scores over the resident scores in many cases (Figure 41). The juvenile steelhead HSI scores averaged 29% greater than the juvenile rainbow scores, due to the relatively high and constant HSI scores for smolt migration, although changes were minimal or slightly negative for study sites that already possessed high HSI scores for juvenile rainbows. Although a higher score for a juvenile steelhead than a juvenile rainbow may seem counter-intuitive, the smolts will likely spend less time in freshwater than the resident juveniles, especially those that emigrate at age 1+, and thus some SoCal streams may provide better habitat for anadromous juveniles than for resident juveniles, especially where fry emigrate from small headwater streams into larger, but warmer mainstem habitats. Additional data from other basins is needed to test this relationship.

The differences between HSI scores for adult steelhead and adult rainbow trout were more variable (Figure 41), because the adult steelhead HSI was not based on the adult rainbow scores, but instead based on factors associated with migration and the short time of freshwater residency. The adult steelhead HSI consequently replicated the migration component described above, with higher scores than for resident adult trout at sites in the lower mainstem Ventura River, equal scores in the upper mainstem (Ven 5), and lower scores than resident trout in the SAC and LNF study sites due to shallow riffles and holding pools, and potential barriers. HSI scores for adult steelhead dropped to zero above Matilija Dam, whereas steelhead smolts were assumed to pass the dam on their downstream migration.

5.5 Distribution and Abundance of *O. mykiss*

Fish distribution and abundance sampling occurred over a seven-year period (2006-2012) in 11 to 13 study sites (less in 2009), each containing 22-24 individual sampling units evenly allocated among pools, flatwaters, and riffles (Table 1). All pools were sampled by direct observation (snorkeling) with count calibration by the Method of Bounded Counts (MBC); all riffles were sampled by multiple-pass electrofishing (with few exceptions); and flatwaters were sampled either by calibrated dive counts or by electrofishing, depending upon water depth. See Section 4.4 for details regarding sampling methodologies. Specific sample sizes by year and study site along with all abundance and density estimates can be found in Appendix D.

5.5.1 Length-Frequency Distributions

Detailed length-frequency distributions of *O. mykiss* are available from all years, study sites, and habitat units where electrofishing was employed (Table 12), whereas generalized size-class distributions, e.g. fry versus juvenile+, are only available from units sampled by diving. For the remainder of this report, the term “fry” will be used to represent *O. mykiss* <10 cm in fork length, “juvenile” represents fish from 10-20 cm, and “adult” represents fish >20 cm in length. “Juvenile+” represents all *O. mykiss* $\geq$10 cm (e.g., juvenile and adult resident rainbows). Although fry are expected to mostly represent young-of-year (0+) and juveniles represent fish in their second summer of life (1+), the length-frequency distributions illustrate that many *O. mykiss* >10 cm are likely 0+ fish, particularly in the lower, warmer mainstem reaches. Also, data is generally insufficient to confidently distinguish between 1+ juveniles from 2+ juveniles or adults, although some data suggests that 15-18 cm FL might be a more appropriate cutoff between 1+ and 2+ *O. mykiss* in the

Ventura River Basin. As described in Section 4.4.1, *O. mykiss* >20 cm in the summer months are likely to exceed smolt size by the following spring, and thus are expected to follow a resident-trout pathway (hence the label "adult"). Note that dive counts only distinguished fish >20 cm in 2011 and 2012. Given the above uncertainty, the terms fry, juvenile, and adult are expected to be only approximate representations of age classes.

The electrofishing-based length frequency distributions from 2006, 2007, and 2010-2012 show differences in distributions between study sites and between years. Sampling in 2012 occurred approximately one month earlier than in previous years (Table 1), due to permitting requirements and drought conditions, and consequently much of the difference in length-frequency distributions with previous years is due to the change in sampling date. Sample sizes from study sites Ven 1 and Ven 2 were too low to draw firm conclusions about the size of fish occupying those reaches, except that 2012 was the only year when either fry or adult *O. mykiss* (>20 cm) were captured (Figure 42).

Few *O. mykiss* were present in the Ven 3 study site in either 2006 or 2007, but were abundant in the remaining years. Although few fish were captured by electrofishing in riffles in 2011, length-frequency distributions in 2010 and 2012 each showed prominent modes (Figure 43), with a longer mode at 115-135 mm in 2010 and a shorter mode at 75 mm in 2012, which was undoubtedly influenced by the much earlier sampling date in 2012. The lack of fry in the 2011 distribution might suggest poor recruitment that year, a hypothesis somewhat supported by results from several tributary study sites, but it is also possible that the higher flows in 2011 resulted in more rearing habitat in San Antonio Creek (the primary spawning tributary to Ven 3), which may have lessened emigration from the tributary into the mainstem Ventura River. Although the electrofishing-based length-frequency distributions for Ven 3 did not show any adult-sized *O. mykiss* >20 cm, the size class distributions based on dive counts in deeper habitats (flatwaters and pools) showed that adult-sized fish were common in 2011 and even dominant in 2012 (Figure 44).

The Ven 5 length-frequency distributions were dominated by a single mode in 2007, 2010, and 2012, presumably representing fry, whereas multiple size classes were evident in the 2011 distributions (Figure 43). The size class pie charts based on dive counts also showed that larger, adult-sized *O. mykiss* were particularly prominent in 2011, a wet year with summer temperatures lower than temperatures recorded in either 2010 or 2012 (Figure 31).

The length-frequency distributions from riffles and flatwaters in the two lower North Fork study sites showed very similar patterns, with strong modes representing fry (with most fish <10 cm) in 2006, 2007, 2010, and 2012 (Figure 45). The size distribution in 2011 was relatively equal between fry and juvenile+ size classes. The size class pies based on dive counts in pools also showed a clear dominance of fry in 2007 and 2012 (as well as in 2009 when electrofishing was not performed), however juvenile+ *O. mykiss* were more abundant in pools than were fry in 2006, 2008, and 2011 (Figure 44). The size class pies further suggested that adult sized fish (>20 cm) were relatively uncommon in both study sites, except in 2012 when 9% of the *O. mykiss* abundance estimate (in pools) was composed of larger fish. However HSI mapping in March 2003 revealed that small, resident adult spawners were relatively common in the lower North Fork (TRPA 2003), which suggests that *O. mykiss* in Ventura Basin tributaries may mature at sizes less than 20 cm.

Above Matilija Dam, *O. mykiss* in the Mat 3 study site showed dominant modes for fry in 2007 and 2012, with broader size distributions in 2006 and 2010 (Figure 46). Similar patterns were observed

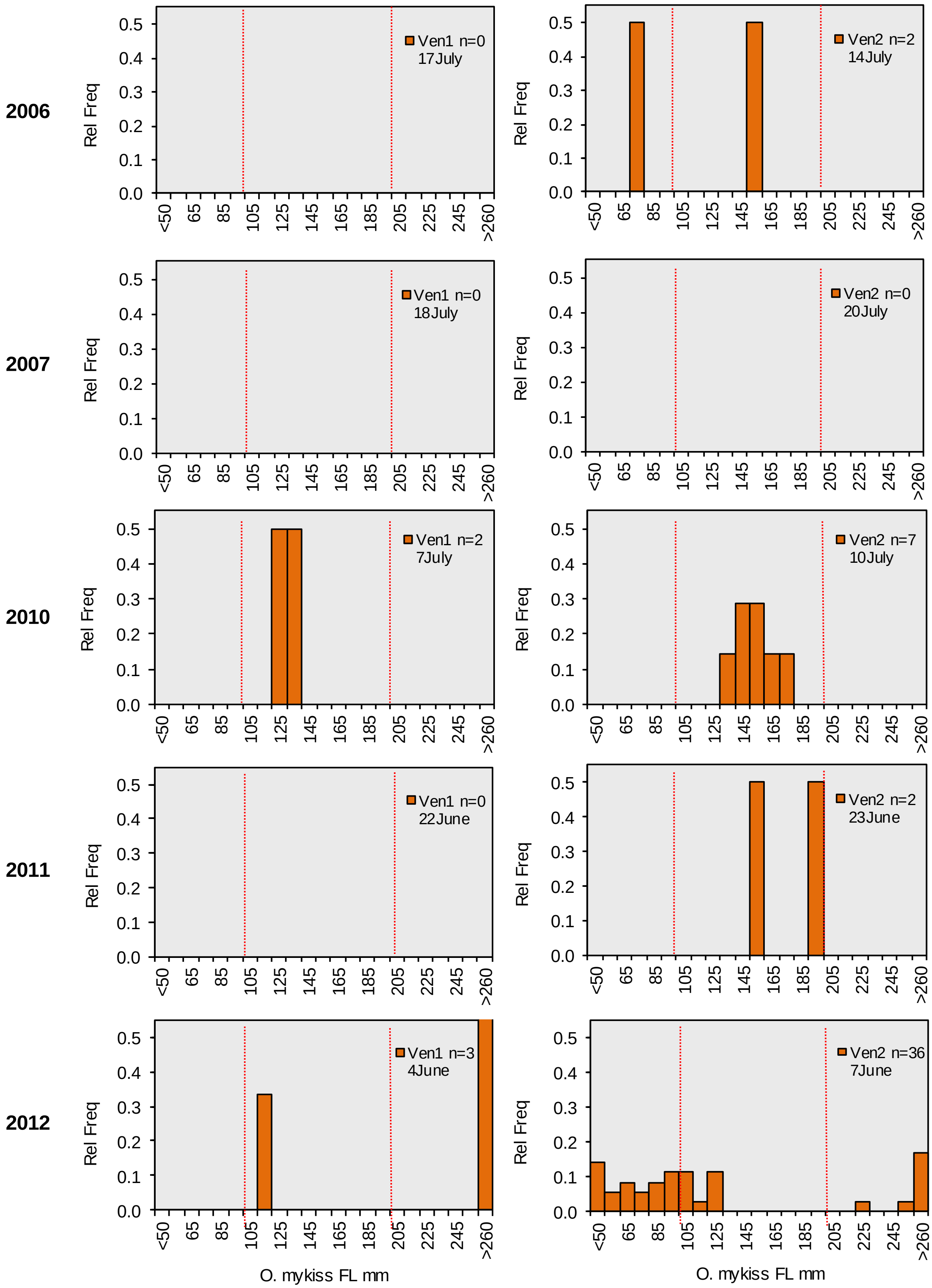


Figure 42. Length-frequency distributions of O. mykiss based on electrofishing captures in Ven 1 and Ven 2 according to year. Dive count size class criteria are also shown.

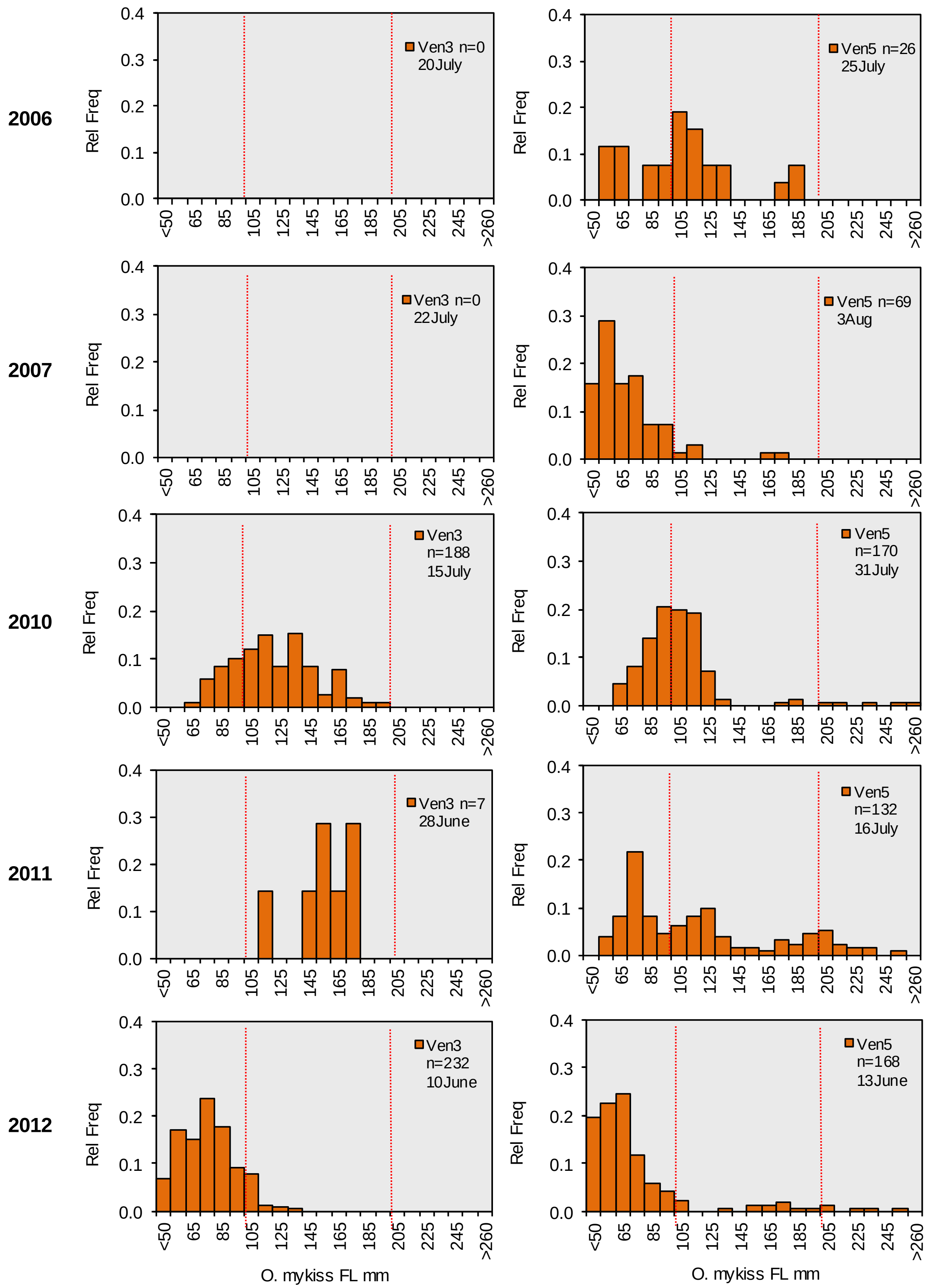


Figure 43. Length-frequency distributions of O. mykiss based on electrofishing captures in Ven 3 and Ven 5 according to year. Dive count size class criteria are also shown.

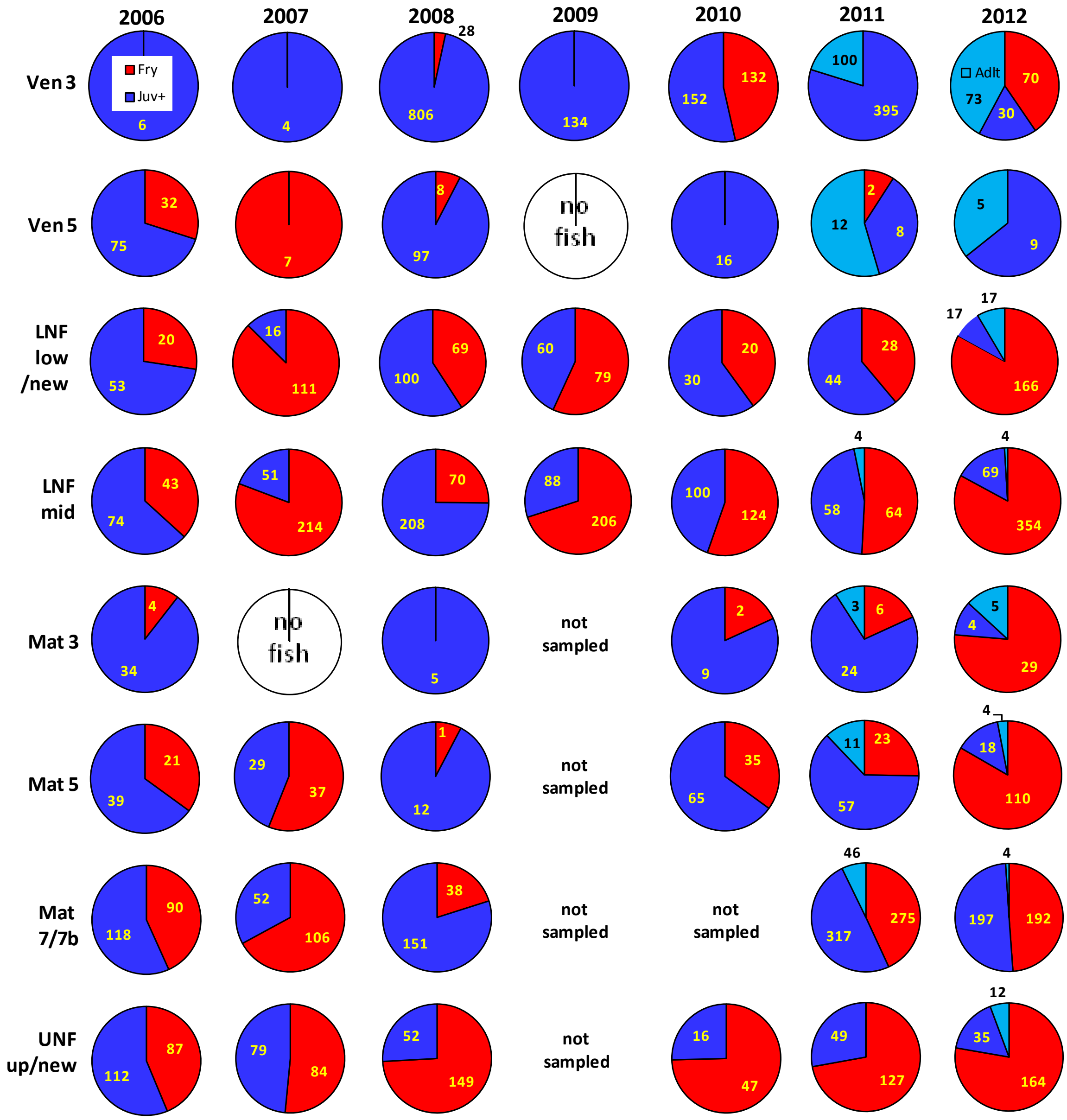


Figure 44. Relative abundance of size classes of O. mykiss based on dive count estimates according to year and study site ("Adult" size class >20cm was only distinguished in 2011-2012).

in the length-frequency distributions in the Mat 5 study site, which also showed a dominance of fry in 2010. The lower abundance in 2011 obscured length-frequency relationships, but data suggested poorer recruitment of fry in that year. The size class pies from dive counts showed that juvenile+ *O. mykiss* were dominant in pools in 2006, 2008, 2010, and 2011 (Figure 44).

Length-frequency distributions for the two uppermost sites sampled each year showed dominant fry age classes in 2006 and 2007 in the Mat 7 study site, and dominant fry in all years in the UNF site (Figure 47). The 10 cm length criterion appeared to be an accurate measure for distinguishing 0+ and 1+ *O. mykiss* in these headwater study sites, as well as the LNF tributary sites. Juvenile+ *O.*

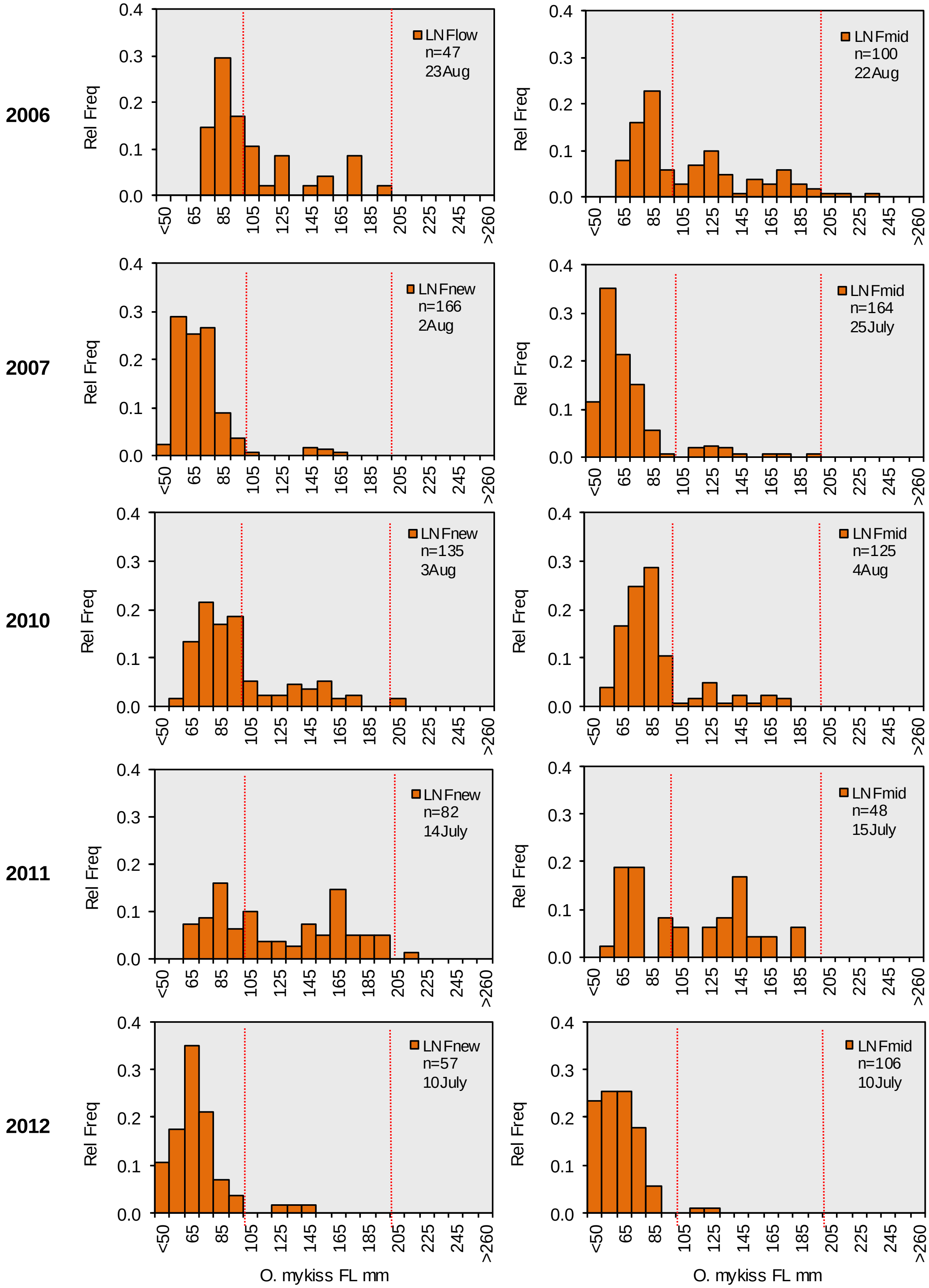


Figure 45. Length-frequency distributions of O. mykiss based on electrofishing captures in the LNF study sites according to year. Dive count size class criteria are also shown.

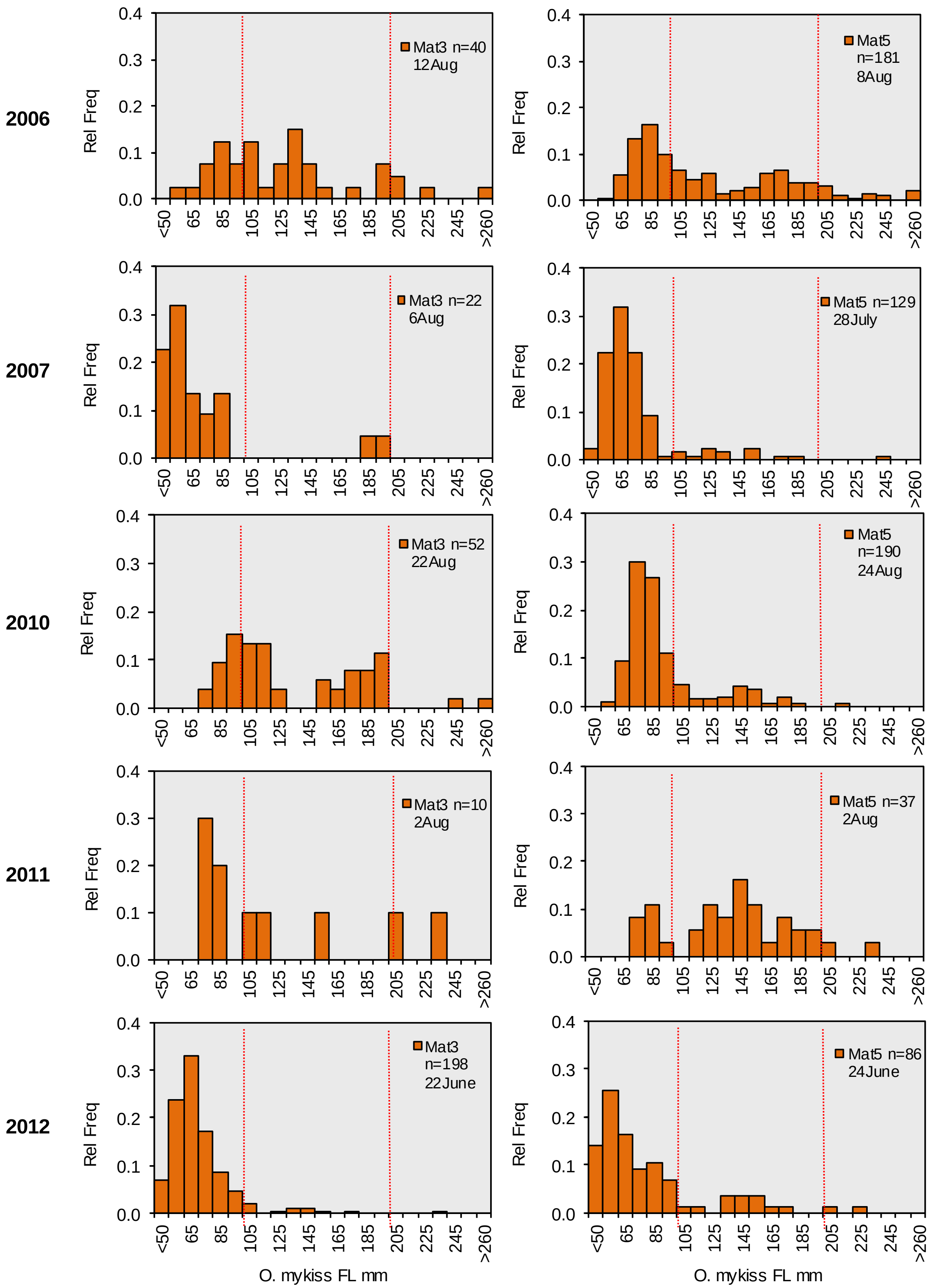


Figure 46. Length-frequency distributions of O. mykiss based on electrofishing captures in Mat 3 and Mat 5 according to year. Dive count size class criteria are also shown.

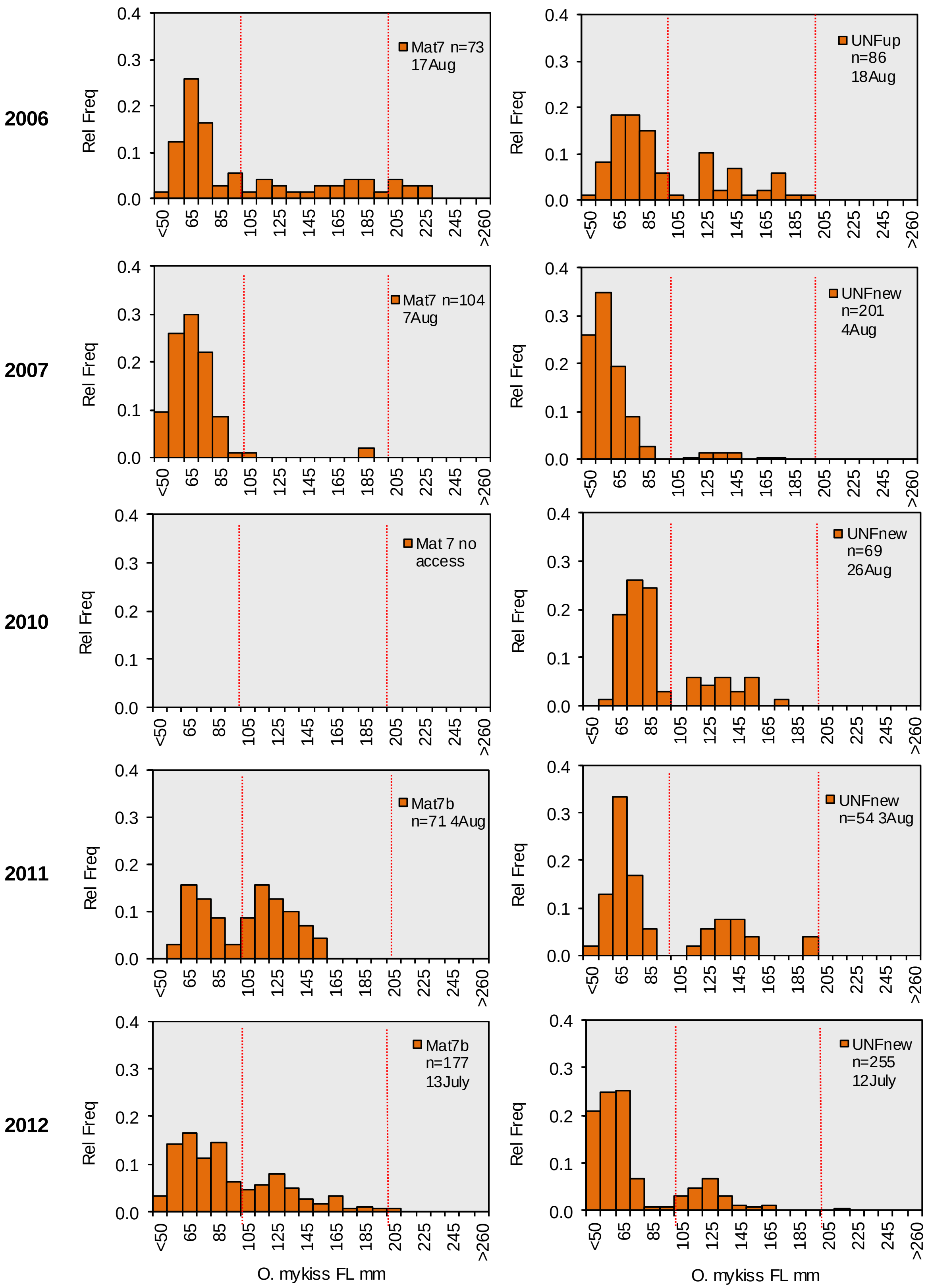


Figure 47. Length-frequency distributions of O. mykiss based on electrofishing captures in Mat 7 and the UNF according to year. Dive count size class criteria are also shown.

mykiss were relatively rare in riffles and flatwaters in 2007 (a dry year), but were frequently captured in those habitats in most other years, including the other dry year (2012). In Mat 7 pool habitats, juvenile+ fish were dominant or of equal abundance with fry in each sample year except 2007, when fry were dominant. In contrast, fry were clearly the dominant size class in UNF pools in 2008, 2010, 2011, and 2012 (Figure 44). Adult-sized *O. mykiss* only comprised 1% to 7% of fish in Mat 7 pools in 2011 and 2012, and were less prevalent (0-6%) in the UNF study site, although small resident spawners have been observed during spring surveys in both study sites.

Sample sizes were generally too small to assess length-frequency distributions in the San Antonio Creek study sites, however size class distributions in 2011 and 2012 generally showed a mix of juvenile and adult or fry, juvenile and adult *O. mykiss* in SAC up pools, with a strong dominance of fry in Murietta Creek pools in 2012, the only year of sampling (Figure 48).

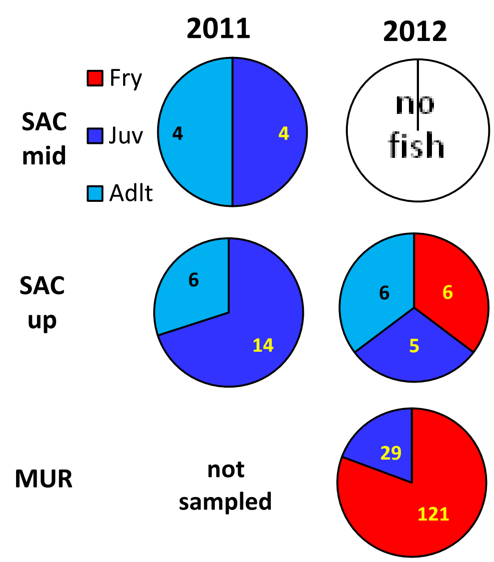


Figure 48. Relative abundance of size classes of O. mykiss based on dive count estimates in San Antonio Creek and Murietta Creek in 2011 and 2012.

In sum, the length-frequency and age class distributions suggest strong recruitment of fry in 2007 and 2012, both dry water years, with some study sites also showing good recruitment in 2010. Unexpectedly, the three years with the highest flows (2006, 2008, and 2011) generally showed relatively low recruitment of fry. The low proportion of fry in 2006 could be associated with the flood events that occurred in 2005, which may have depressed the abundance of adults available for spawning in 2006. Alternatively, a moderate flow event of 6,000 cfs occurred in April 2006, which could have led to mortality of *O. mykiss* eggs or fry (Figure 26). The only other year that possessed a high flow event during the spring spawning and incubation period was 2011, when a mean daily flow of 6,270 cfs occurred in late March.

The strong recruitment of fry in 2007 and 2012 may have been associated with the lack of scouring flow events during the previous winter and spring, and/or to an apparently strong return of adult steelhead in those years. The spring and summer of 2007 was noted for numerous sightings of adult steelhead in several southern California rivers, including the San Luis Rey River, Trabuco Canyon, and the Ventura River, where up to four large (~45-60 cm) *O. mykiss* were observed as late as mid-July and early August (TRPA 2008, Capelli 2007). Likewise in 2012, when seven *O. mykiss* from 45 to 51 cm in length were observed in 20 pools in the lower Ventura River between Foster Park and San Antonio Creek (encompassing all of Ven 3, Figure 2), as well as another group of large fish in a deep pool downstream of the study area (Normandeau 2012). Although the apparent abundance of adult spawners in the Casitas Springs area may have contributed to the strong fry class seen in Ven 3 in 2012 (Figure 43), fry were not abundant in the Ven 3 reach in 2007, but fry were the dominant size class in the three reaches farther upstream (Ven 5 and the two lower North Fork study sites).

In addition to the "steelhead-sized" *O. mykiss* observed in the Ven 3 reach of the lower Ventura River, numerous other adult sized fish 25-40 cm FL in length were observed over-summering in the Ven 3 study site, and "trout-sized" redds and small (<25 cm) spawners were observed during spring surveys in several mainstem and tributary reaches. These observations support the conclusion that a significant proportion of *O. mykiss* in the lower and middle segments of the Ventura Basin, both accessible to steelhead, is composed of resident life-forms.

5.5.2 Annual Estimates of Abundance

The abundance estimates for *O. mykiss* fry and juvenile+ over the course of this seven-year study is arranged by basin segment (Table 1). Abundance estimates with their associated variances and confidence intervals were calculated according to the MBC formulas presented in Section 4.4.3. Assessing the significance of changes in abundance between consecutive years, and treatment of missing data is discussed in Section 4.5.1. Missing data occurred where limitations in funding or permitting restrictions prevented sampling in specific study sites or habitat types; these missing data are identified in Table 12. Those missing data that were estimated using linear regression, in order to produce a full 7-year time series of total abundance, are also indicated in Table 12 by red font, and are identified in the following annual trend figures as open, rather than filled, circles. Statistically significant differences between adjacent estimates, whether assessed by non-overlap of confidence intervals (Figure 22) or the more rigorous difference equations (equations 23 and 24), are identified the in the following figures by asterisks. Statistical comparisons were not made where study sites were moved between years, e.g., LNF low vs. LNF new and UNF up vs. UNF new in 2006 and 2007.

Lower Segment

The lower segment represents the historically accessible anadromous reaches of the mainstem Ventura River from the mouth upstream to Robles Diversion Dam (Figure 2), which includes study sites Ven 1, Ven 2, Ven 3, and (when not dry) Ven 4 (Table 1). Abundance estimates from SAC mid and SAC up are also presented for recent years, although the combined study site estimates representing the entire lower segment do not include San Antonio Creek, in order to present a comparable sampling frame.

Ven 1

O. mykiss fry were not observed in the Ven 1 study site in any of the six sample years, although Ven 1 was not sampled in 2009 and pools were too turbid to sample in 2012 (Table 12, Figure 49). Low numbers of juvenile+ *O. mykiss* were observed in Ven 1 in each of the last three years of sampling, however estimates of abundance in the one-mile study site were less than 10 fish each year (Figure 50). Most of those fish observed or captured were in flatwaters or riffles.

Ven 2

A single *O. mykiss* fry was captured in the Ven 2 study site in 2006, however fry were captured in one flatwater and four riffles in 2012, which produced an overall abundance estimate of 50 fry, not

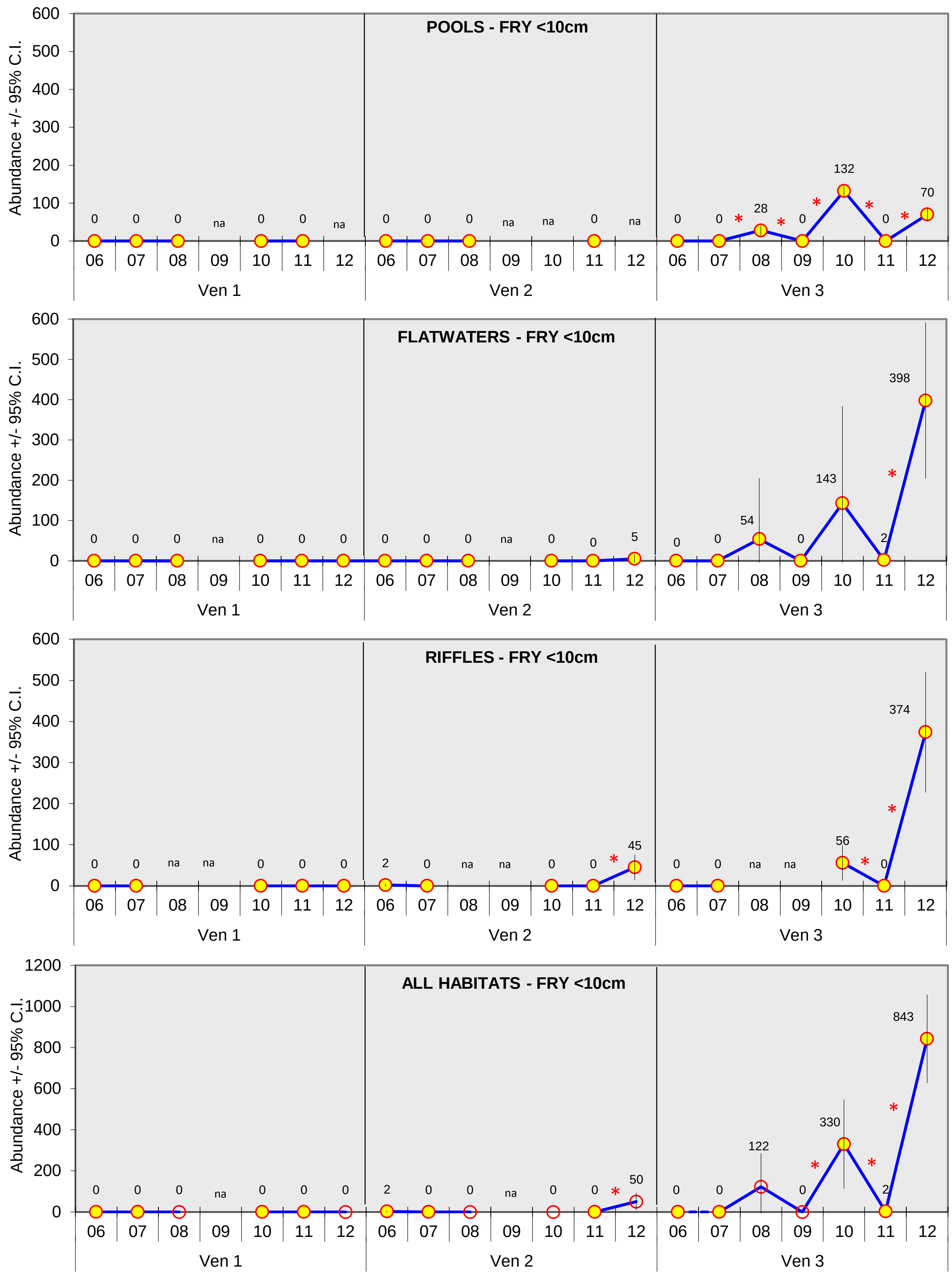


Figure 49. Estimated abundance (w 95% C.I.'s) of O. mykiss fry <10cm in the lower segment according to habitat type, study site, and year. Asterisks indicate statistically significant difference between adjacent years.

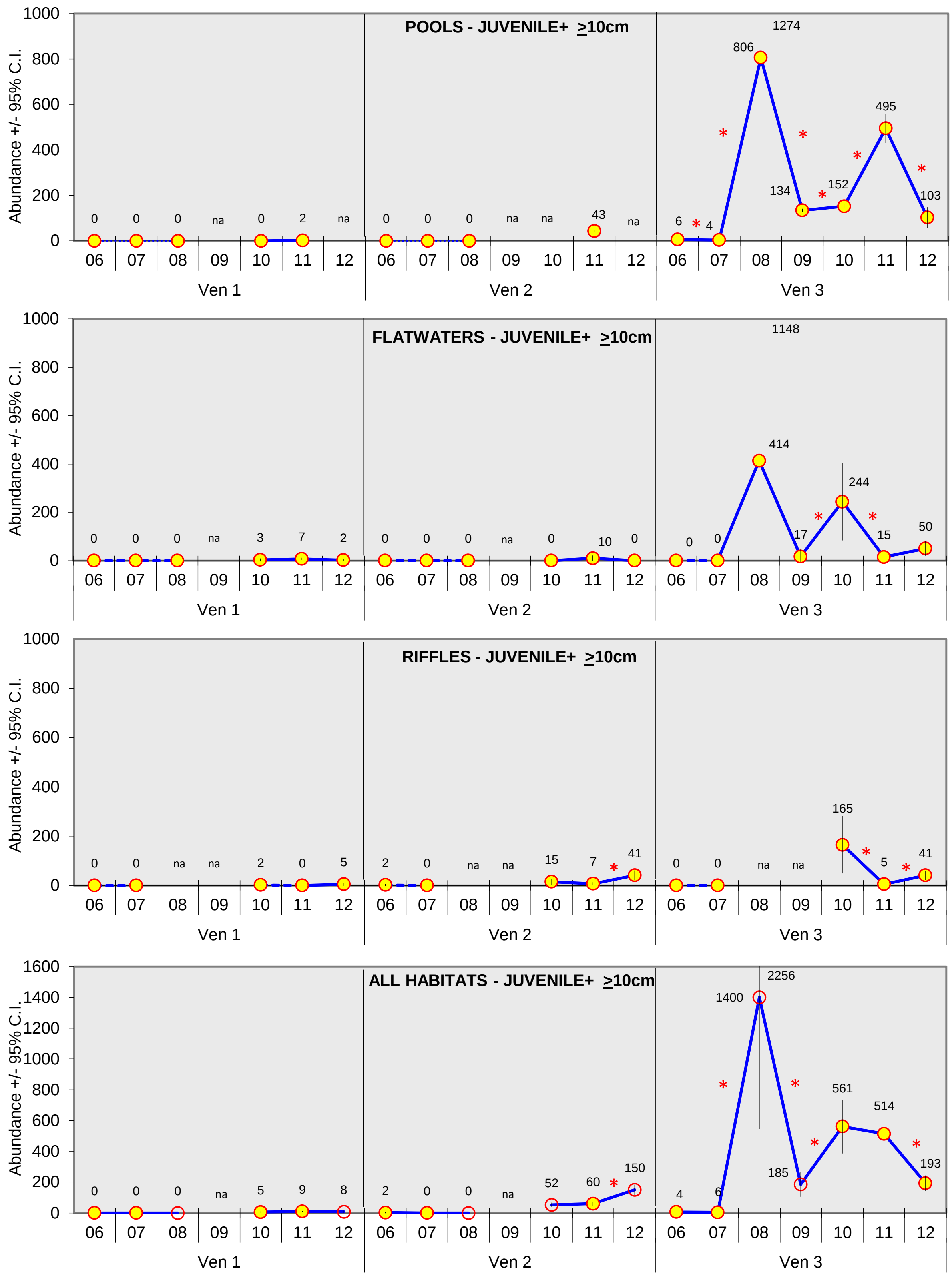


Figure 50. Estimated abundance (w 95% C.I.'s) of O. mykiss juvenile+ >10cm in the lower segment according to habitat type, study site, and year. Asterisks indicate statistically significant difference between adjacent years.

including pools which were too turbid to sample. The increase from zero abundance in 2011 was statistically significant for the riffles and combined habitat types estimates (Figure 49). As seen for Ven 1, juvenile+ *O. mykiss* were more abundant in Ven 2 than were fry, with fish observed or captured in 2006 (only 1 fish), 2010 (7 fish), 2011 (31 fish), and 2012 (16 fish, excluding pools). Estimated abundance of juvenile+ increased from two fish in 2006 to 150 fish in 2012, with the 2011-2012 change significant for riffles and combined habitats (Figure 50).

Ven 3

The Ven 3 study site is historically known to support abundant rearing of *O. mykiss* juveniles (Moore 1980, Capelli 1997), due in part to the presence of cool upwelling groundwater near the upper boundary of the study site (Figure 31), and to the confluence of San Antonio Creek, a spawning tributary also near the upstream boundary (Figure 2). During the course of this study, Ven 3 has supported the highest abundance of both size classes of *O. mykiss*, as well as the highest mean densities of fry and juvenile+ (both at 0.17 fish/100 ft^2) of all study sites in the lower segment. However, Ven 3 has also shown high variability in abundance, with estimated abundance ranging from less than 10 fry and juvenile+ *O. mykiss* in 2006 and 2007 to maxima of 800 to 1,500 fish in 2008, 2010, and 2012. Although not reported here, personal observations by local biologists have indicated that the extreme dry conditions in 2013 and 2014 produced lengths or dry or stagnant conditions in Ven 3 (Paul Jenkin, personal communications), and may have effectively “reset” the local population of *O. mykiss* in this reach to lower abundance as seen in the first two years of this study. The coefficients of variation (C.V.’s) calculated from the annual abundance estimates for fry shows that Ven 3, along with the other lower mainstem study sites (e.g., Ven 2, Ven 4, and Mat 3), were highly variable, averaging 3-4 times the variability seen in the headwater and tributary sites (Figure 51). C.V.’s for juvenile+ *O. mykiss* were lower and more consistent, with most mainstem study sites showing C.V.’s approximately twice that of the headwater and tributary study sites.

Given the changes in annual abundance estimates in Ven 3, many of the comparative estimates were significantly different despite typically wide confidence intervals (Figures 48 and 49). The most dramatic changes in abundance were the increases in fry in 2010 and 2012, and the increases in juvenile+ in 2008 and 2010. The overall abundance of fry in 2012 (at 843 fry) was 155% greater than the second highest estimate of 330 fry in 2010, and represented a dramatic increase from only 2 fry in the preceding year. As previously noted, the size class criteria of 10 cm does not accurately represent age classes in the lower, warmer mainstem reaches; consequently the increase in estimated abundance from 2 fry in 2011 to 843 fry in 2012 is likely influenced by the earlier sampling in 2012 (by 3 weeks, Table 1) and by the smaller proportion of fish >10 cm classed as juvenile+ in that year (Figure 44). For juvenile+ *O. mykiss*, the maximum abundance estimate of 1,400 fish in 2008 was almost three times the next highest estimates of 561 and 514 fish from 2010 and 2011, respectively. The high abundance in 2008 is particularly notable given the zero or near zero estimates for all *O. mykiss* in the preceding two years. The year 2007 was the second driest year during the seven year study with a mean base flow (at the Casitas USGS gage) of only 2.3 cfs (Figure 27), which could have limited over-summer survival of *O. mykiss*; yet fish were more abundant in 2012 under even lower base flows of 1.8 cfs. It is probable that the extreme high flow events during 2005, and perhaps the late spring flow event in 2006 (Figure 26), effectively eliminated the *O. mykiss* population in the lower Ventura River, which carried over to yield low abundance in 2007.

Comparison of abundance data among the three habitat types in 2010, 2011, and 2012 generally showed highest estimates in flatwaters for fry and in pools for juvenile+ *O. mykiss* (Figures 48 and

49), however fry densities (#/100 ft^2) were similar among habitat types in 2010 or were highest in riffles in 2012 (Figure 52). Juvenile+ fish occurred at highest densities in riffles in 2010, much higher densities in pools in 2010, and evenly distributed among habitat types in 2012. Greater use of pool

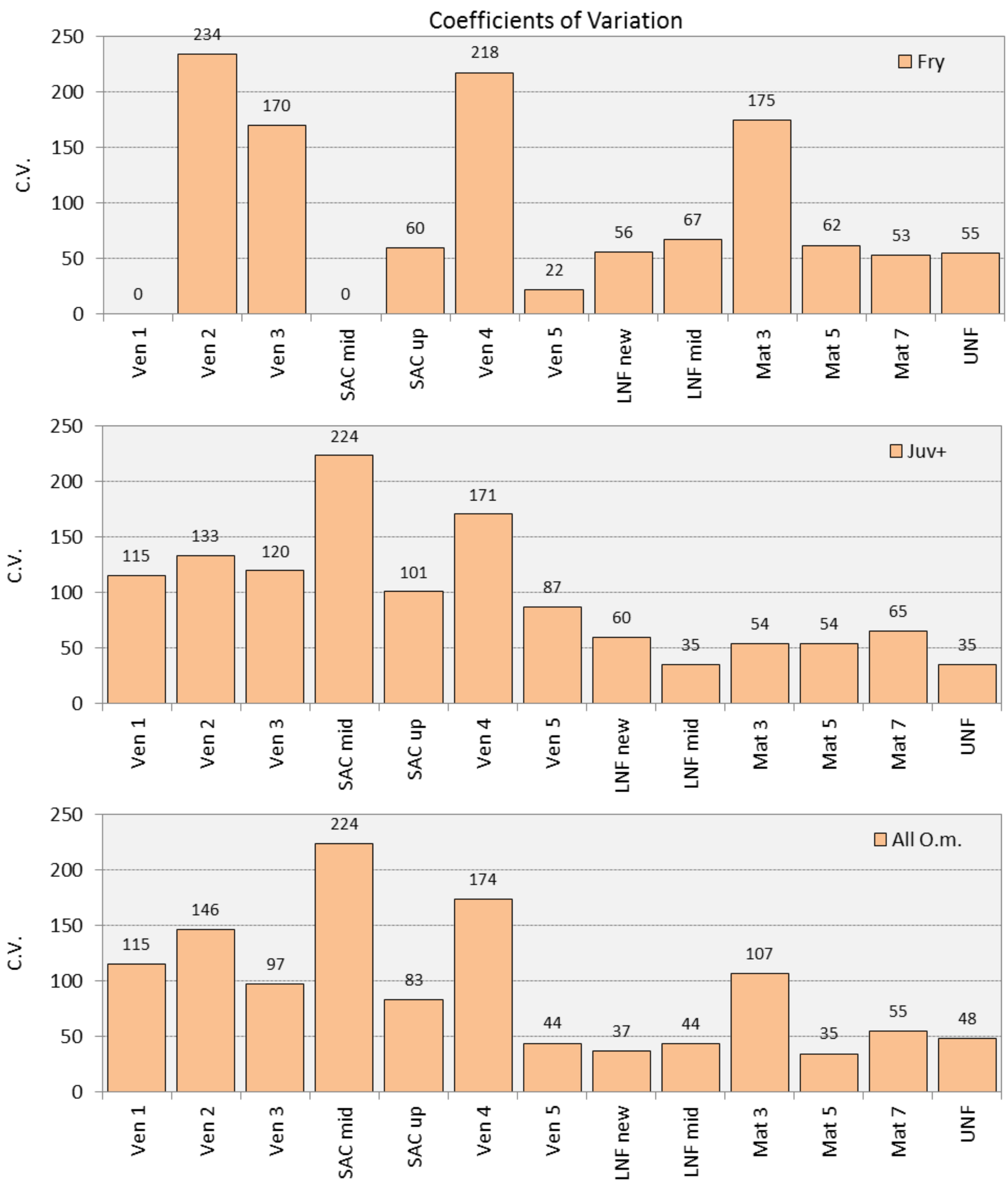


Figure 51. Coefficients of variation (C.V.'s) of annual abundance of O. mykiss according to size class and study site.

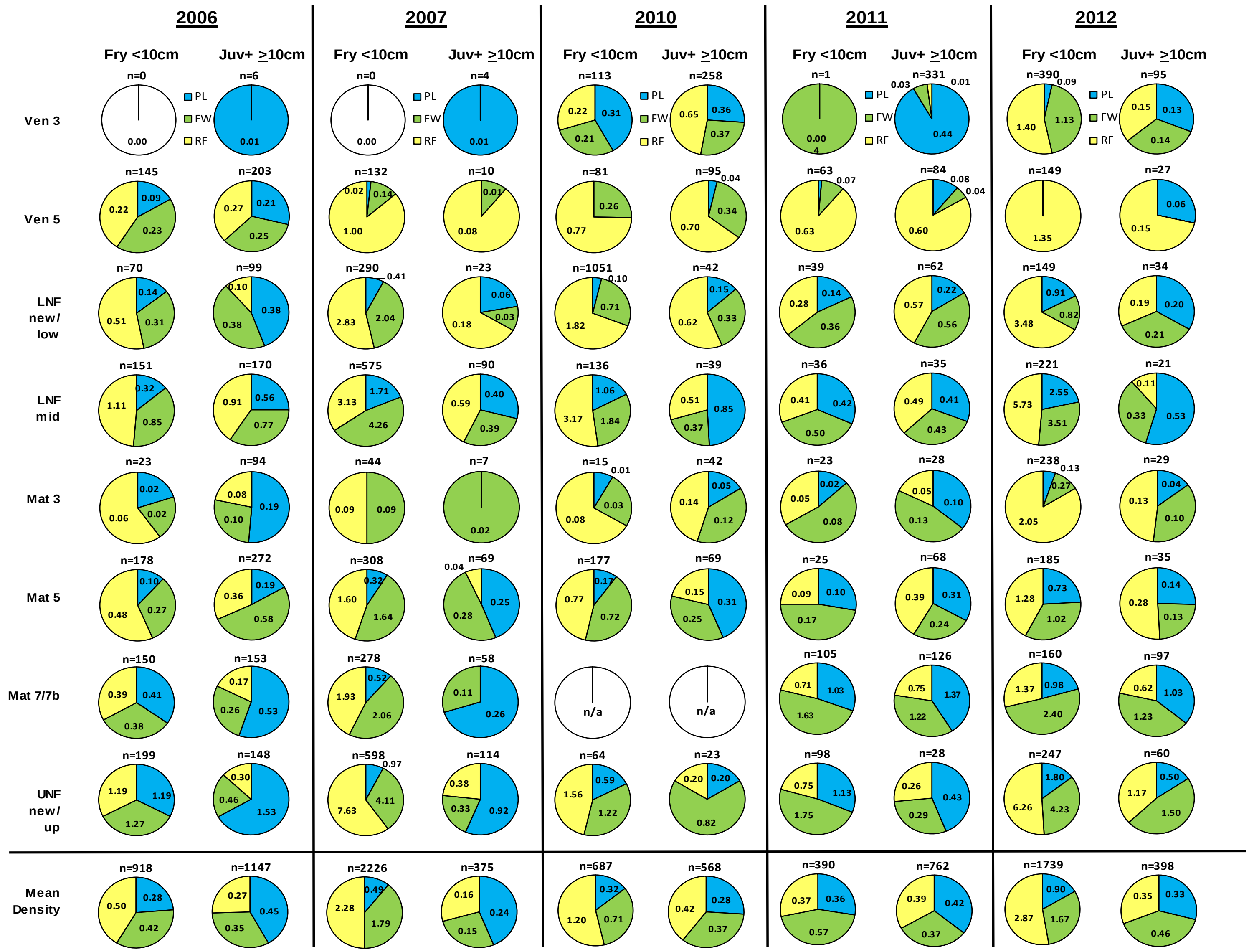


Figure 52. Estimated densities (#/100ft2) of O. mykiss by year, study site, size class, and habitat type.

habitats during a wet year (2011) than in a dry year (2012), when riffles were deeper and presumably more available to larger fish, is somewhat unexpected. Mean densities among all years and all study sites (those with five years of full sampling) show that fry occurred at highest densities in riffles in all years except 2011 (the wet year), whereas juvenile+ fish were either most densely populated in pools (2006 and 2007) or were evenly distributed in all habitat types (2010, 2011, 2012), apparently without regard to base flow conditions.

SAC mid

The SAC mid study site was sampled by qualitative electrofishing in 2007, followed by random sampling in most of the remaining years. No *O. mykiss* fry were observed or captured in any year (Figure 53), but juvenile+ fish were present in one pool , three flatwaters, and three riffles in 2011, yielding an estimated total abundance of 33 fish in the half-mile study site (Figure 54). One of the juvenile+ fish observed in July 2011 was an adult steelhead over 50 cm in length, which may have been stranded over the summer low flow period. The reaches of San Antonio Creek below Lion Canyon (and the uppermost reaches) are subject to dry or intermittent flows in dry years (Figure 28); although surface flow was present during early August 2007 (a dry year), portions of the study site were dry by late August 2012, one month following fish sampling.

SAC up

The perennial study site on upper San Antonio Creek is located downstream of an annually dry channel (Figure 28), and consequently is influenced by upwelling groundwater and mediated water temperatures (Figure 30). This location is a known spawning area for adult steelhead (Mark Capelli, NMFS, personal communication), and consequently was expected to provide rearing for fry and juvenile+ *O. mykiss*. Pool dive counts were conducted in 2010, 2011, and 2012, whereas flatwaters and riffles were sampled only in the latter two years (Table 12). Permit take limitations in 2012 prevented electrofishing in shallow water habitats in this study site; consequently those habitats were sampled by dive counts which, due to the low flows and shallow conditions, were expected to yield minimum abundance estimates. Rearing *O. mykiss* were observed in all three years, with the highest abundance of fry (26 fish) in 2012 (Figure 53) and the highest abundance of juvenile+ (167 fish) in 2010 (Figure 54). The annual changes in abundance in pools and in all habitats combined were statistically significant in all years. Densities, like abundance, were highest in runs and riffles for fry, but densities of juvenile+ were consistently higher in the deeper pool habitats.

Ven 4

The Ven 4 study site is located within a six mile intermittent channel of the mainstem Ventura River, immediately downstream of the Robles Diversion Dam (Figure 2). This site is unique due to the presence of several large, deep, bedrock-formed pools which historically were important holding habitats for adult migrant steelhead . The Ven 4 study site was completely dry and therefore unsampled during the 2007, 2008, 2009, and 2012 survey periods (Table 1). Sampling was possible during 2006, 2010, and 2011, and fry or juvenile+ *O. mykiss* were present during the latter two surveys. The maximum estimated abundance of fry was only 10 fish in 2010 (Figure 53), whereas 19 juvenile+ fish were estimated in Ven 4 in both 2010 and 2011 (Figure 54).

Combined Study Sites

Annual changes in abundance of *O. mykiss* were generally consistent among lower segment study sites; consequently the pooled estimates of abundance representing the entire lower segment, which included the entire mainstem Ventura River below Robles Diversion Dam but excluded San Antonio Creek (due to lack of consistent time-series data) and other tributaries (e.g., Cañada Larga and Coyote Creek), showed a similar trend with near zero abundance in 2006 and 2007, and highest abundance of fry (2,348 fish) in 2012 and juvenile+ (3,739 fish) in 2008 (Figure 55). Although

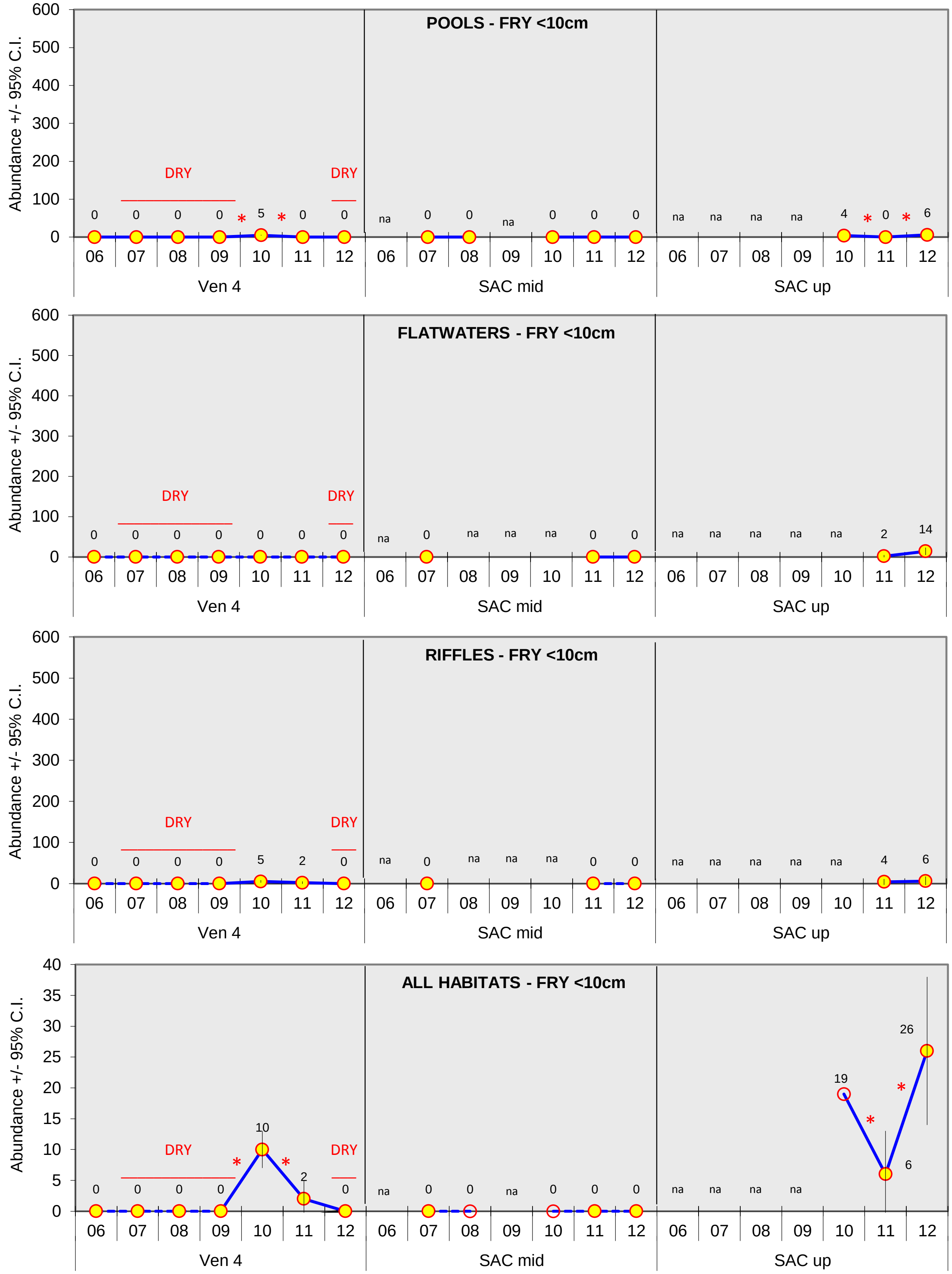


Figure 53. Estimated abundance (w 95% C.I.'s) of O. mykiss fry <10cm in the lower segment according to habitat type, study site, and year. Asterisks indicate statistically significant difference between adjacent years (part 2).

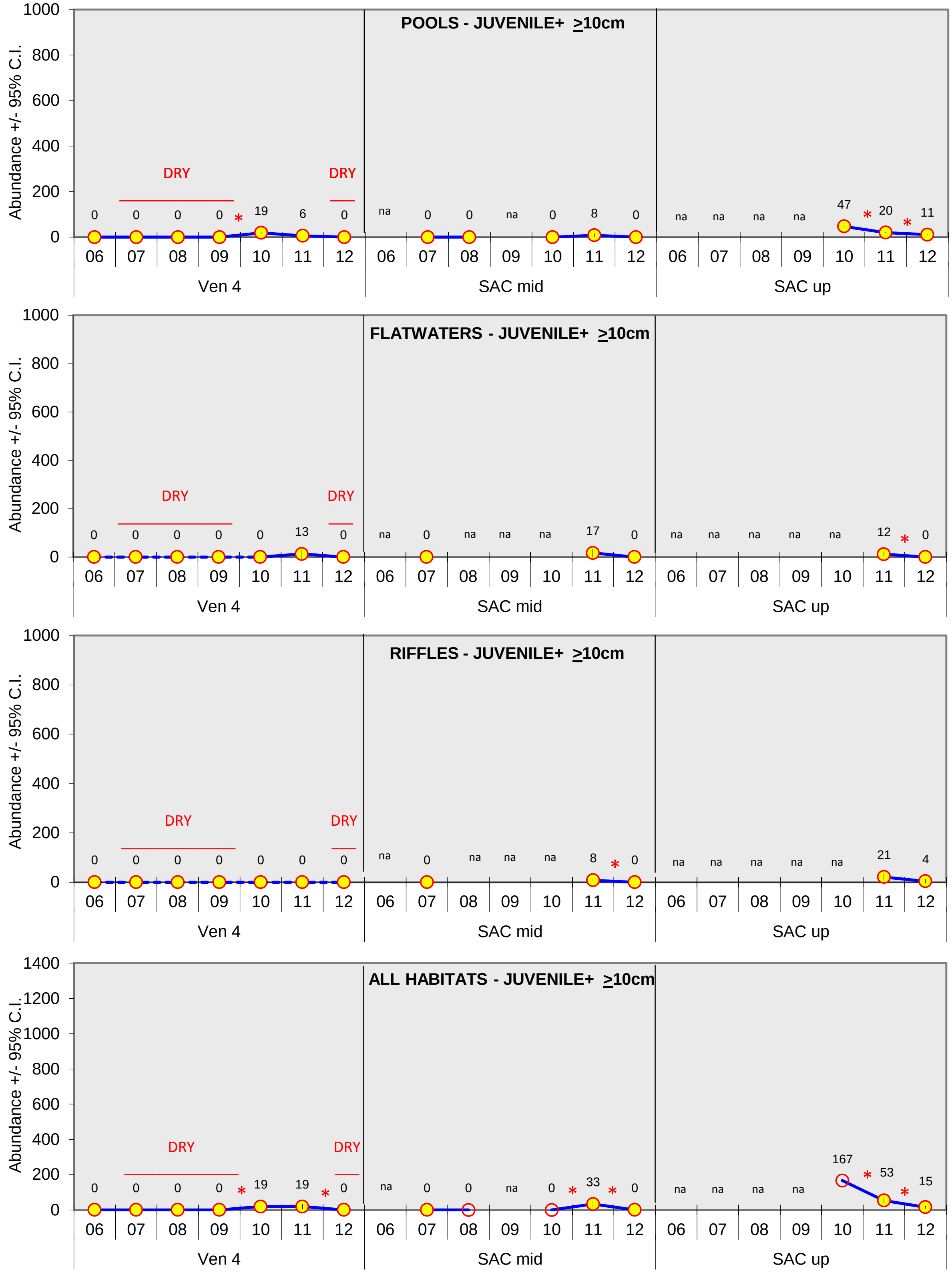


Figure 54. Estimated abundance (w 95% C.I.'s) of O. mykiss juvenile+ >10cm in the lower segment according to habitat type, study site, and year. Asterisks indicate statistically significant difference between adjacent years (part 2).

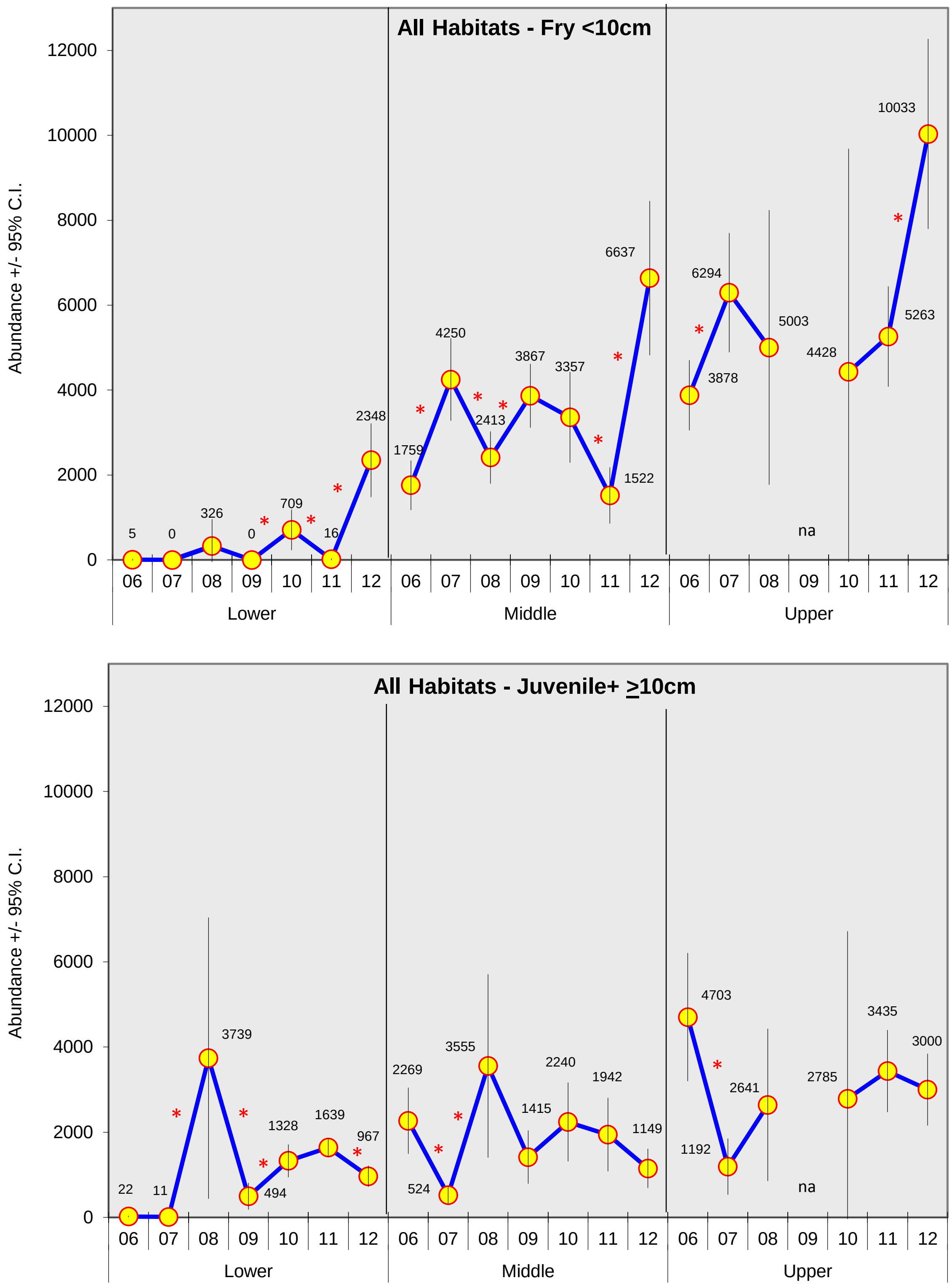


Figure 55. Estimated abundance (w 95% C.I.'s) of O. mykiss fry and juvenile+ according to basin segment and year. Asterisks indicate statistically significant difference between adjacent years.

confidence intervals for some estimates were very broad, annual changes were significant for fry from 2009 to 2010, 2010 to 2011, and 2011 to 2012. Likewise, annual changes were significant for juvenile+ in all consecutive years except 2006 to 2007 (when estimates were near zero) and 2010 to 2011. Overall the general trend for both size classes was an increase in abundance over time, however those trends were highly influenced by the low abundance in the first two years (likely due to the 2005 flood event) and also by the high abundance of fry in 2012.

Middle Segment

The middle segment is currently accessible to anadromous steelhead, following construction of the fish ladder at Robles Diversion Dam in 2004. This segment includes one mainstem Ventura River study site (Ven 5) and two sites in the lower North Fork Matilija Creek (LNF new and LNF up). Note that the LNF low study site was sampled only in 2006, and was then moved downstream to the LNF new site which was sampled in all remaining years (Table 1, Figure 3).

Ven 5

The Ven 5 study site has consistently produced moderate abundance of *O. mykiss* fry with little annual variation (Figure 51); however like Ven 3, abundance of juvenile+ showed significant variation between years. Although few of the annual changes in abundance of fry were statistically significant (Figure 56), most annual changes in juvenile+ were significant (Figure 57). The Ven 5 study site is the warmest of all sites with thermographs below Matilija Dam (Figure 1), with weekly maximum water temperatures exceeding 75^{o}F (24^{o}C) in most years (Figure 31). The highest abundance estimates of juvenile+ came from 2006, 2008, and 2010, when base flows were approximately double the flows in intervening years (Figure 27); however juvenile+ abundance remained relatively low in 2011 despite high base flows in that year. Of all study sites, Ven 5 was most consistent in showing the highest densities of both fry and juvenile+ *O. mykiss* in riffle habitats, with substantially lower densities in flatwaters and (in most years) pools (Figure 52). The warmer water could be in part responsible for the reliance on riffle habitats given the increased metabolism of fish and their need for greater food resources.

As will be noted elsewhere (Section 5.5.6), the correlation between abundance of fry in one year and juvenile+ *O. mykiss* in the following year may help to explain much of the observed variation in juvenile+ abundance estimates. For example, Ven 5 is located immediately downstream of the confluence with the lower North Fork Matilija Creek, a major spawning tributary (Figure 3), and the estimated abundance of fry in the LNF study sites was highest in 2007, 2009, and 2012 (Figure 56). It is possible that the strong recruitment of fry in those years led to emigration of juvenile+ fish downstream into the Ven 5 study site in the following year, hence producing the high juvenile+ abundance estimates in Ven 5 in 2008 and 2010.

LNF low/new

The LNF low study site was sampled in 2006, and then moved to a new site (LNF new) for the remainder of this study; consequently observed changes in annual abundance from 2006 to 2007 may be due to spatial rather than temporal effects. In both LNF study sites and in all study sites above Matilija Dam, flatwaters and riffles were not sampled in either 2008 or 2009; consequently the estimated abundance in all habitat types combined was based on pool dive counts using linear regression, as noted in Section 4.5.1. The abundance of fry in the LNF new study site generally showed little change from 2007 to 2011, but the 2012 estimates for pools and riffles were significantly higher than the preceding year, and was the highest estimate from all years of study (Figure 56). The abundance of juvenile+ *O. mykiss* also showed relatively little change in most years, with highest abundance in 2008 (based only on pool data), with a decline in abundance from 2011

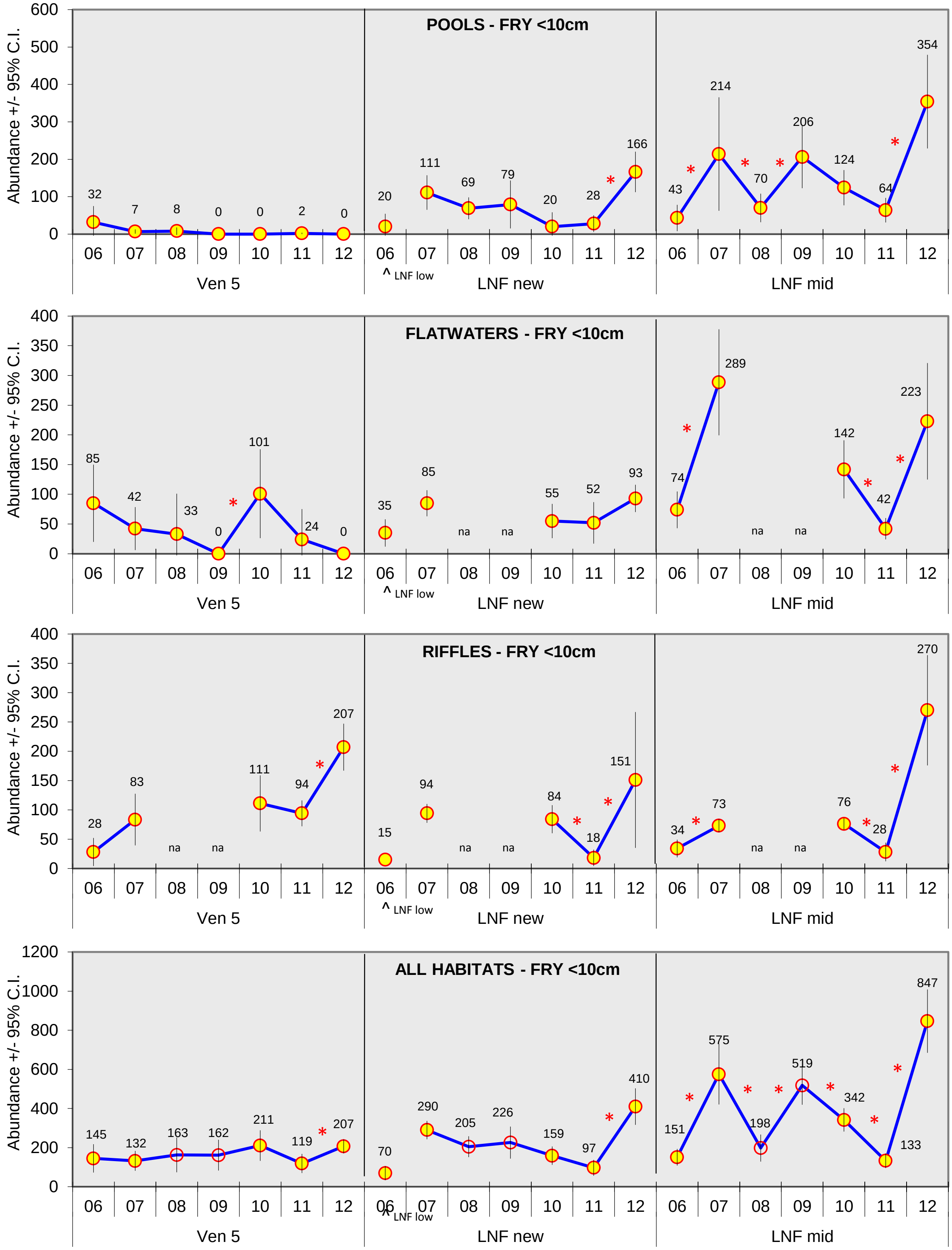


Figure 56. Estimated abundance (w 95% C.I.'s) of O. mykiss fry <10cm in the middle segment according to habitat type, study site, and year. Asterisks indicate statistically significant difference between adjacent years.

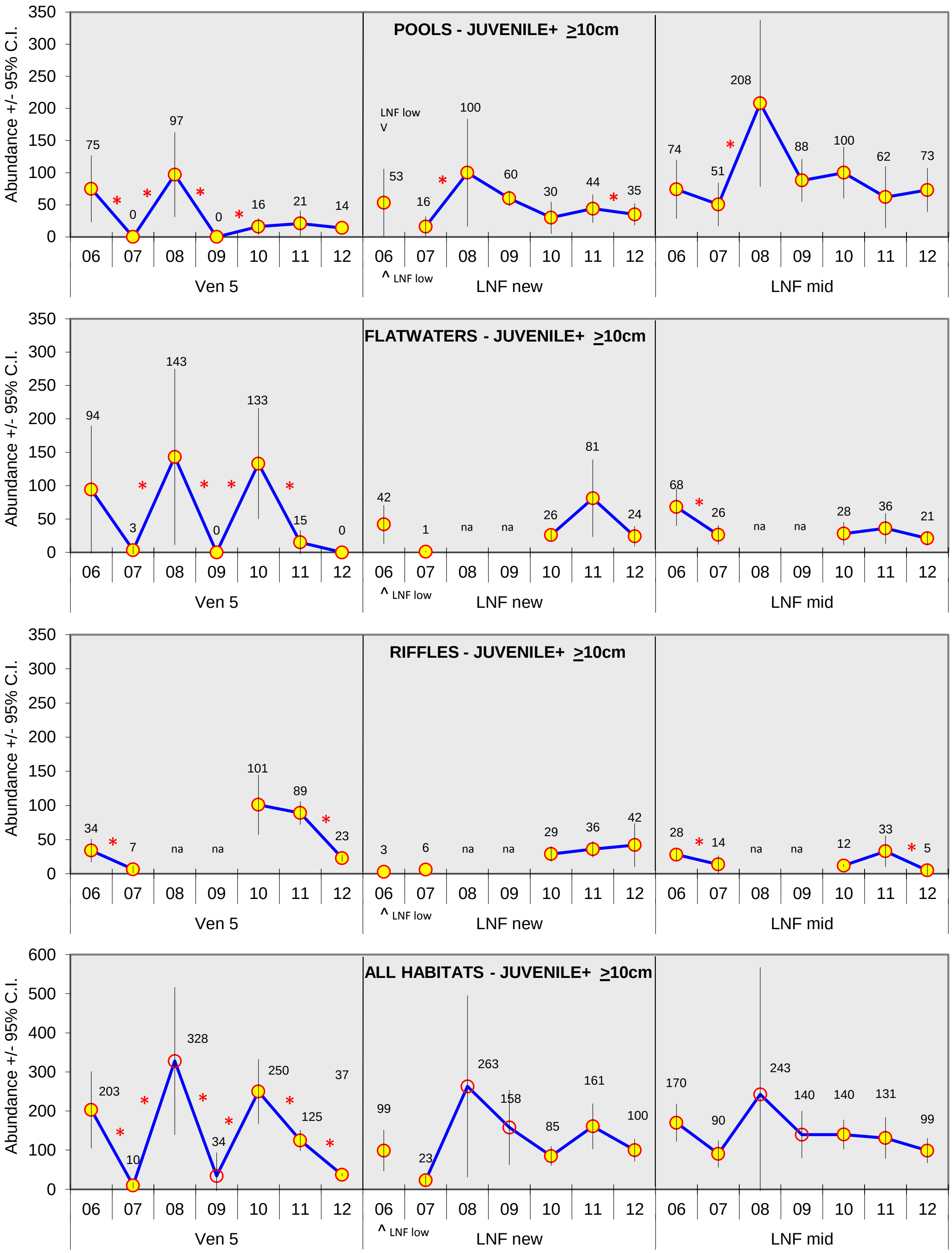


Figure 57. Estimated abundance (w 95% C.I.'s) of O. mykiss juvenile+ >10cm in the middle segment according to habitat type, study site, and year. Asterisks indicate statistically significant difference between adjacent years.

to 2012 (Figure 57). In most years densities of fry were highest in riffles and lowest in pools, typically by several orders of magnitude (Figure 52). Juvenile+ *O. mykiss* also tended to occur at higher densities in riffles or flatwaters than in pools, but the differences were less extreme and more variable.

The abundance of both fry and juvenile+ *O. mykiss* was typically lower in LNF new pool habitats than in LNF mid pools, despite attempts by local swimmers to construct large and deep swimming holes in the LNF new site. Pools in the LNF new study site in 2011 averaged 66% larger in surface area than pools in the LNF mid study site (*t-test, P*=0.06), although only 21% deeper (*t-test, P*=0.16). Despite shallower depths in the LNF mid pools, estimated densities of *O. mykiss* fry and juvenile+ were 255% and 393% greater, respectively, in LNF mid pools than in the larger and deeper LNF new pools (see Appendix D for density data). Differences in abundance of fry between the two sites were also evident in flatwaters and riffles, which suggest that differences in spawning habitat or other non-pool features may be factors; however, unlike for pools the abundance of juvenile+ in flatwaters and riffles was similar between the two reaches.

LNF mid

The LNF mid study site showed both higher abundance as well as higher annual variability (Figure 51) of *O. mykiss* fry than did the LNF new study site, although such was not the case for juvenile+ fish, which occurred at similar abundance in most years. As noted for the LNF new study site, fry were typically most densely populated in riffles and least in pools, whereas juvenile+ fish were more evenly distributed among habitat types or else occurred at highest densities in pools (Figure 52). Annual abundance was more variable for fry than for juvenile+ *O. mykiss*, with highest abundance in 2007, 2009, and 2012, with the 2011-2012 increase statistically significant for all habitat types and combined habitats (Figure 56). Juvenile+ abundance remained very constant between about 100 and 150 fish, except in 2008 where the combined estimate (based on linear regression) was 243 fish (Figure 57). Few of the annual changes were significant.

Combined Study Sites

Combining abundance estimates from the three middle segment study sites showed statistically significant increases in abundance of *O. mykiss* fry in 2007, 2009, and 2012, with significant decreases in 2008 and 2011 (Figure 55). Maximum abundance of fry was estimated at 6,637 fish in 2012, a 65% increase over the abundance estimates from 2007 and 2009. Annual changes for juvenile+ were significant and variable from 2006 to 2008, but declined from the 2008 maximum abundance of 3,555 fish to lower and relatively constant abundance estimates from 1,100 to 2,200 fish in 2009-2012. Comparison of annual trends in abundance of fry in the lower segment with trends in the middle segment showed few similarities, except for the changes in 2011 and 2012 (a decrease and an increase, respectively). Changes in abundance of juvenile+ fish were more similar between segments, with both showing strong increases in 2008, followed by decreases in 2009 and little change or a slight decrease over the last three years of sampling.

Upper Segment

The upper segment lies above Matilija Dam (Figure 3) and consequently is inhabited only by the stream-resident form of *O. mykiss* (rainbow trout), although recent spawning surveys suggest that adfluvial trout may inhabit Matilija Reservoir and ascend Matilija Creek to spawn. It is also possible that resident-derived juveniles smolt and pass over (or through) Matilija Dam, however such information has not been located. Study sites above Matilija Dam include the lower and middle mainstem sites Mat 3 and Mat 5, the headwater mainstem site Mat 7, and the tributary study sites in upper North Fork Matilija Creek and (in 2012 only) Murietta Creek (Table 1, Figure 3). Although two *O. mykiss* were observed in Old Man Creek during a March 2003 survey (TRPA 2003), this tributary is mostly dry during summer months and is expected to provide little rearing habitat and minimal contribution to basin population estimates.

Note that only pools were sampled in the upper segment in 2008, thus the combined habitat estimates in that year were derived using linear regression (Table 12). Also, sampling was not conducted in any upper segment habitats in 2009, and the Mat 7 study site was denied access in 2010, requiring selection of a new study site (immediately upstream) for sampling in 2011 and 2012. Finally, note that the UNF up study site was sampled only in 2006, and was then moved downstream to the UNF new site which was sampled in all remaining years.

Mat 3

Sampling in Mat 3 produced low estimates of abundance (<50 fish) of *O. mykiss* fry in all years except 2012, when abundance significantly increased to 421 fry (Figure 58). Juvenile+ *O. mykiss* were likewise present in low abundance (<100 fish), however that size class did not show an increase in 2012 (Figure 59). As previously discussed (Section 5.1.2), Mat 3 (along with Ven 5) showed the highest temperatures recorded among all data loggers deployed for this study (Figure 29). The water temperature data logger was located in the lower of two Mat 3 sub-reaches, which were divided by a one mile section of private property (Figure 3). Within this private reach exists a series of hot springs (98^{o}F on 8-11-06) that increased mid-day water temperatures in the lower half of Mat 3 by approximately 7-9^{o}F (4-5^{o}C) during summer surveys.

Although separate abundance estimates were not calculated for the sampling units above versus below the hot springs, a comparison of relative densities of fry and juvenile+ *O. mykiss* clearly shows the higher densities above the hot spring's inflow (Figure 60). On average, densities of fry in riffles and flatwaters in the lower, warmer reach were 10-15% lower than densities above the springs; likewise for juvenile+ fish whose densities in the lower half were only 13-63% of densities in the upper half. These differences in densities are not likely to be due solely to water temperature, as the upper half of Mat 3 possesses a steeper, more confined channel with bedrock-related scour elements. Also, the lower half contains more potential predator species (e.g., Centrarchids) and upper half is closer to known sources of fry recruitment, such as Murietta Creek and the upper North Fork Matilija Creek. Nevertheless it is likely that, given the common occurrence of maximum temperatures in lower Mat 3 that are well above EPA criteria for *O. mykiss* distributions (Figure 33), the hot springs are likely to be contributing to the reduced densities in the lower half of Mat 3.

Mat 5

Like Mat 3, the Mat 5 study site possessed a disparity in habitat character, with a lower half consisting of a single channel with relatively sparse riparian vegetation, which then split into two channels in the upper 1,000 ft of the study site. The main split was heavily vegetated and was significantly influenced by a cool, groundwater-derived tributary at the channels top boundary, whereas the other split was open, with little vegetation and lower, warmer (by 1-2^{o}F) flow. Also, the lower half of the Mat 5 study site was a losing reach and was partially dry or intermittent during sampling in 2007 and 2008.

Abundance of *O. mykiss* fry from the Mat 5 study site showed annual fluctuations but with overall estimates mostly ranging between 100 and 400 fish, punctuated by significant decreases in abundance from 2010 to 2011 followed by significant increases again to maximum observed abundance in 2012 (Figure 58). Juvenile+ *O. mykiss* showed less variability, with maximum abundance (272 fish) in the first year of study, and minimum abundance (87 fish) in the final year of

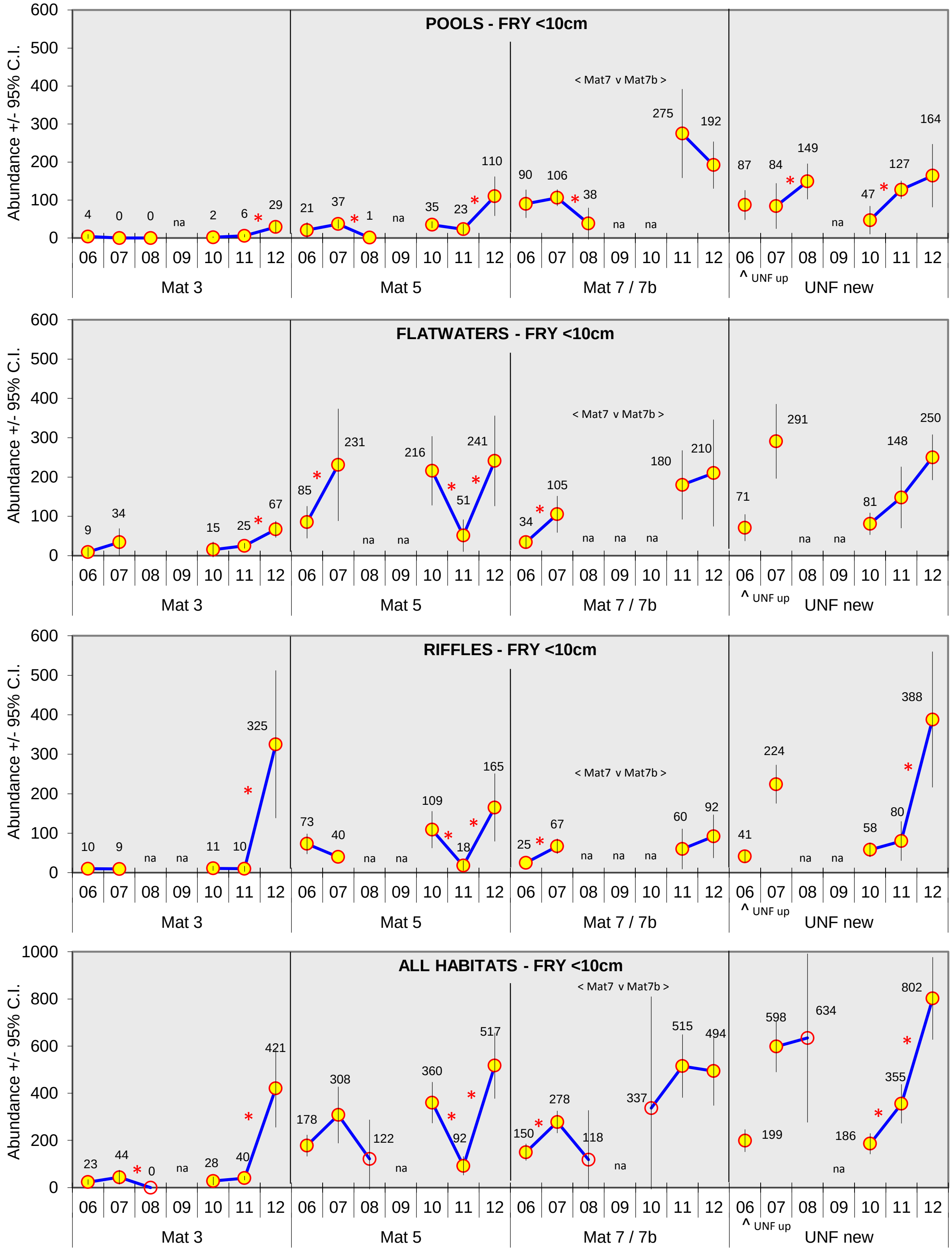


Figure 58. Estimated abundance (w 95% C.I.'s) of O. mykiss fry <10cm in the upper segment according to habitat type, study site, and year. Asterisks indicate statistically significant difference between adjacent years.

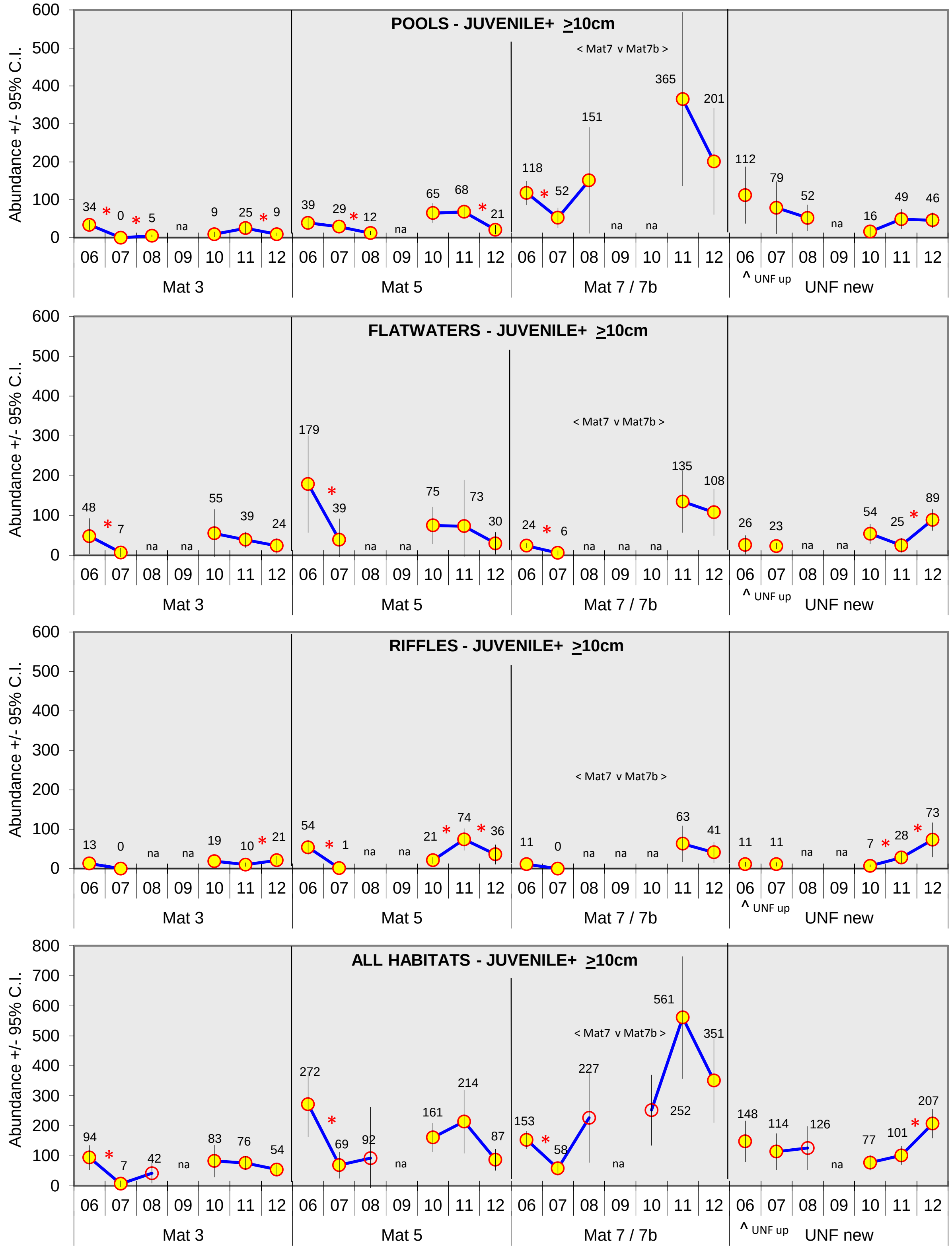


Figure 59. Estimated abundance (w 95% C.I.'s) of O. mykiss juvenile+ >10cm in the upper segment according to habitat type, study site, and year. Asterisks indicate statistically significant difference between adjacent years.

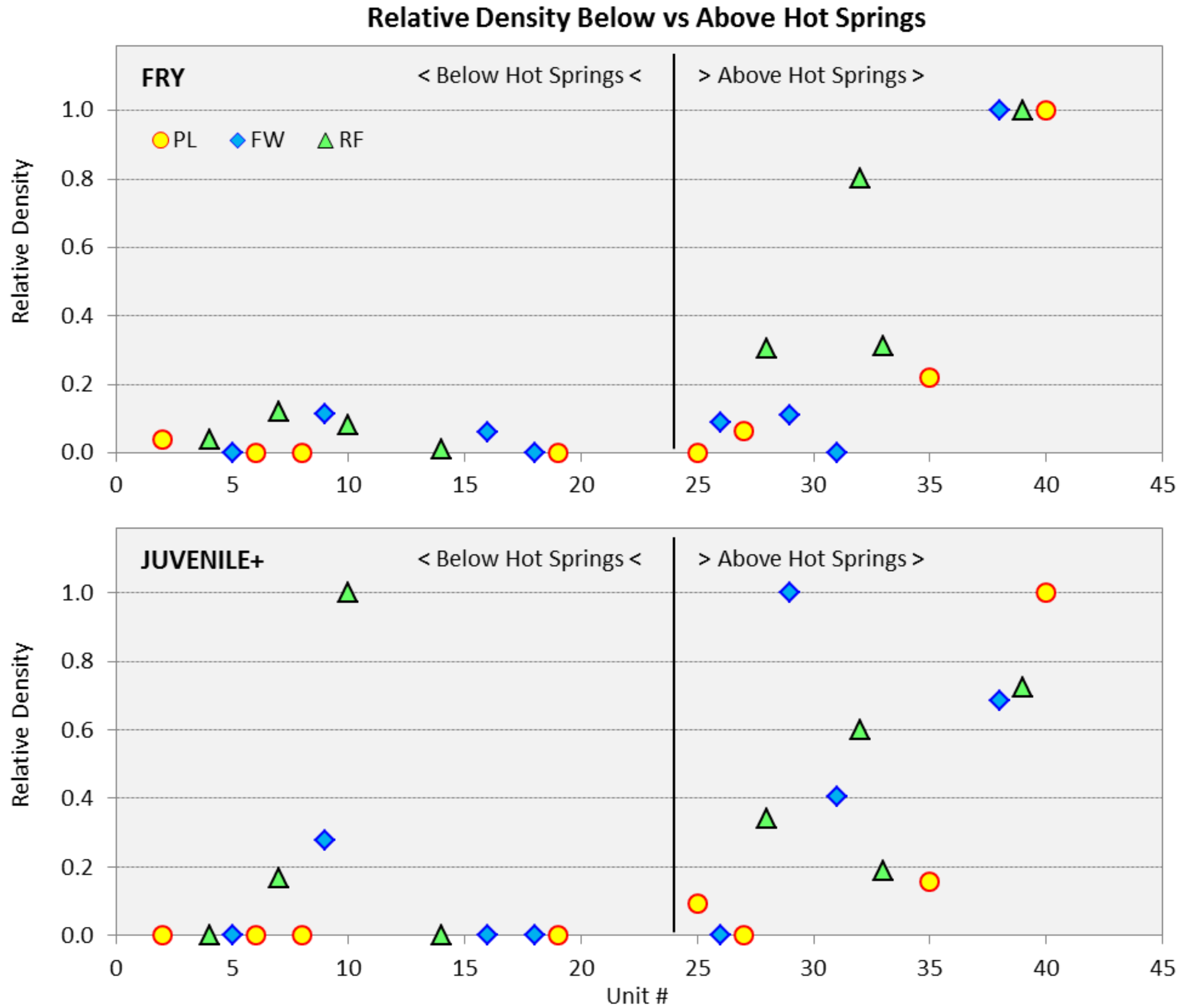


Figure 60. Comparative densities of O. mykiss fry and juvenile+ above and below the Mat 3 hot springs in summer 2012. Relative densities are normalized to maximum density by habitat type.

study (Figure 59). As with most other study sites, densities of *O. mykiss* fry were generally highest in riffles and lowest in pools, whereas juvenile+ fish occurred at highest densities in pools or flatwaters in most years (Figure 52).

Mat 7

As previously described, the one-half mile Mat 7 study site sampled in 2006 through 2008 was moved upstream about 4,000 ft (and 180 ft in elevation) for sampling in 2011 and 2012. In general the habitat characteristics appeared similar in the two study sites, except that the upper site (Mat 7b) terminated at a deep pool with an impassable cascade, which appeared to produce aggregations of larger *O. mykiss* in 2011 (see cover image). For example, most Mat 7b pools in 2011 contained an average of nine juvenile+ *O. mykiss*, but the top pool contained an estimated 91 fish; this data was therefore treated as an outlier and the pool abundance estimate was calculated without this sampling unit. Such aggregation was not evident the following year and consequently all pool counts were used to estimate abundance in 2012. In order to help fill-in the data gaps from lack of access to the private property, linear regression was used to predict the total abundance of *O.*

mykiss fry and juvenile+ (all habitats combined) in 2010 (Table 12), based on the relationship between Mat 7 abundance and abundance in the Mat 5 and UNF study sites in the remaining years.

The estimated abundance of *O. mykiss* fry suggested a difference in the two alternative study sites, with generally lower (100-300 fry) abundance in the lower site and higher abundance (300-500 fry) in the upper site (Figure 58). This was also evident for juvenile+, where the mean abundance from 2011 and 2012 (excluding the 2010 predicted abundance) was three times higher than the average abundance in the lower study site (Figure 59). Relatively few of the consecutive-year abundance comparisons were statistically significant, with the exception of the increase in fry and concurrent decrease in juvenile+ fish from 2006 to 2007. Maximum abundance estimates for both fry and juvenile+ *O. mykiss* exceeded 500 fish in 2011. Unlike most other study sites, *O. mykiss* fry did not occur at highest densities in riffle habitats, but were typically highest in flatwaters (Figure 52). Densities of juvenile+ fish were consistently greatest in pools and lowest in riffles.

UNF

The UNF study site was moved from its 2006 location 2.8 miles above its confluence with Matilija Creek (UNF up) to a new study site (UNF new) 0.7 miles from the confluence (Figure 3); this new study site was sampled in the six remaining years. Abundance estimates for *O. mykiss* fry in pool habitats were generally consistent between about 80 and 150 fish each year, but abundance was more variable in flatwater and riffle habitats with estimates ranging from 71 to 291 fry in flatwaters and 41 to 388 fry in riffles (Figure 58). Abundance estimates for all habitat types combined increased significantly each year from 2010 to 2012 to a maximum of 802 fry in 2012. Juvenile+ *O. mykiss* showed much less variability between years, with most combined habitat estimates ranging from 100 to 200 fish, with a significant increase to maximum abundance (207 fish) in 2012 (Figure 59).

Comparative densities of fry among habitat types again showed highest fry densities in the shallower and faster riffle or flatwater habitats, and typically lower densities in pools (Figure 52). Juvenile+ fish occurred at highest densities in pools in three years and flatwaters in two years.

MUR

The Murietta Creek study site was only sampled in 2012, a dry water year. Although sampled in early summer (Table 1), flows were already $\leq$0.5 cfs and were continuing to drop. A 500 ft section in the middle of the study site contained five sampling units that each held 1-12 *O. mykiss* on June 27th and 28th, however these habitat units were completely dry by July 11th. Murietta Creek was the only study site where numerous electrofishing mortalities occurred (11 of 170 captured *O. mykiss*), which may have been associated with the decreasing flows, although water temperatures remained cool throughout the 2012 summer months (Figure 30).

The estimated abundance of fry and juvenile+ *O. mykiss* in late-June 2012 was 340 and 169 fish, respectively, which despite the low flows represented the 3rd highest densities of fry and juvenile+ fish in the Ventura Basin in 2012. Somewhat surprisingly, despite the shallow depths the densities of both size classes were lowest in pools and highest in flatwaters, with intermediate densities in riffles. This relationship again illustrates the relative importance of shallow non-pool habitats for rearing *O. mykiss*, even in headwater tributaries in dry water years. Riffle depths in the Mur, UNF new, and the LNF mid study sites, where the three highest *O. mykiss* fry densities occurred, averaged less than four inches.

Combined Study Sites

The combined study site abundance estimates for *O. mykiss* fry in the upper segment were higher than estimates in either of the two lower study segments, which was also the case for juvenile+ estimates except in 2008 when high pool counts in the Ven 3 and LNF study sites produced higher (regression-based) estimates for combined habitat types in the lower and middle segments (Figure 55). Because of the wide confidence intervals in 2008 and 2010 (mostly due to the regression estimates of missing data), only the increases in abundance of fry from 2006 to 2007 and from 2011 to 2012 were statistically significant. For juvenile+ fish, only the decrease from 2006 to 2007 was significant.

The higher abundance estimates in the upper segment are largely due to the higher average densities of *O. mykiss* in the reaches above Matilija Dam, which encompass approximately one-half of the stream miles that are currently available for rearing below the dam (not including dry channels). The result further illustrates the potential rearing habitat and benefits of providing access to steelhead above Matilija Dam.

5.5.3 Spring vs. Summer Dive Counts

Limited spring sampling was conducted in several intermittent and warm mainstem stream reaches in order to assess if those reaches were potentially important rearing areas for *O. mykiss* during periods of higher flow and cooler water temperatures. Spring dive counts were conducted in four pools in each of six study sites in late April 2010, followed by the summer dive counts (Table 1) in most of the same pools. Spring (early May) and summer dive counts were conducted in 2011 in 5 to 8 pools or flatwaters (mostly different units) in six study sites as well as in Mat 6, a perennially dry channel between Mat 5 and Mat 7 (Figure 3).

Although summer counts (mean per unit) were generally higher than spring counts for both size classes in 2010 and 2011 (Figure 61), the differences were not significant due to high variability in mean counts (unpaired t-test, most P's >0.5). Although this test likely possesses little statistical power due to the sampling of different units in spring versus fall (due in part to remapping in 2011), this limited dataset does not suggest that significantly greater numbers of *O. mykiss* utilize the warmer mainstem or tributary study sites in spring in comparison to summer, despite the somewhat higher flows and cooler temperatures prevalent during the spring months.

5.5.4 Utilization of Intermittent Stream Reaches

In a related assessment, fish sampling was conducted in several reaches that are frequently dry or intermittent in surface flows during the summer base flow period. Intermittent channels are known to be productive for steelhead spawning and rearing in many areas (Allen 1986, Faudskar 1980), including drainages in central California (Boughton et al. 2009). However, most of the Ventura Basin's intermittent reaches possess a high degree of substrate mineralization due to the combination of high mineral content of the basin waters, and the extended periods of low, sluggish flow with high solar radiation (from limited riparian growth). Cursory inspection of substrate characteristics in these intermittent reaches suggested poor spawning habitat due to firmly cemented gravels, and low densities of invertebrate prey species; both factors would be expected to limit abundance of *O. mykiss* even during cooler, spring months. Sampled stream reaches and study sites that are frequently subject to dry channels or intermittent flows (Figure 28) include Ven 4, SAC mid, Mat 2 (just downstream of Mat 3), and Mat 6 (between Mat 5 and Mat 7).

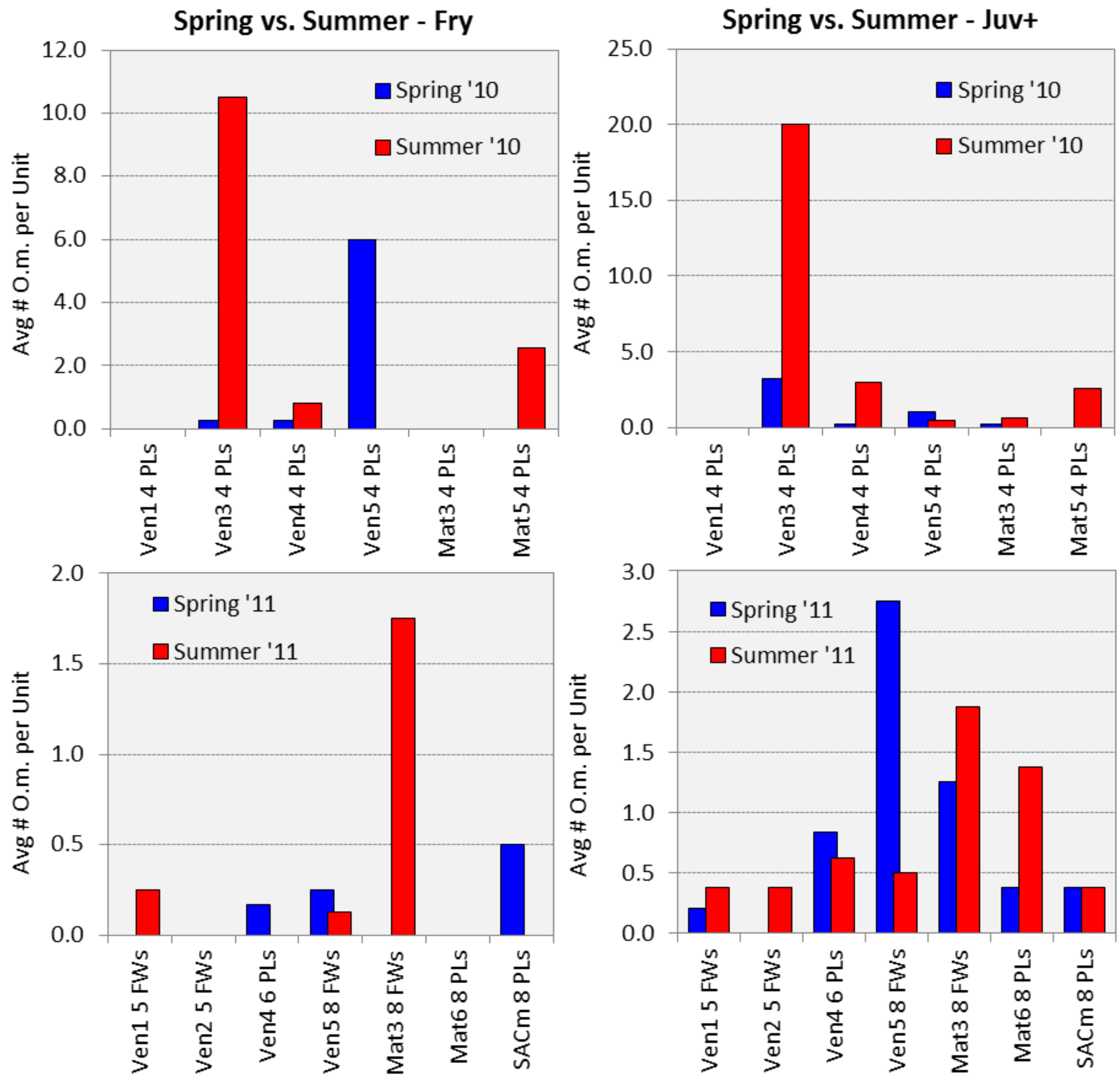


Figure 61. Comparative mean density (#/habitat unit) of O. mykiss during spring vs. summer dive counts according to year, size class, and habitat type.

The Ven 4 study site was sampled during the summers of 2006, 2010, and 2011 (Table 12), and in the spring of the latter two years (see above for spring versus summer comparisons). As seen in Figures 52 and 53, the estimated abundance of *O. mykiss* in Ven 4 during the three years of summer sampling showed consistently low numbers of fry (0-10) and juvenile+ (0-19) fish. These abundance estimates translate to densities of less than 0.03 fish/100ft^2, which are far below equivalent densities from adjacent study sites upstream (Ven 5) or downstream (Ven 3). Although smolts derived from upstream sources (Ven 5 or LNF) must pass through Ven 4 towards the ocean, and limiting rearing may occur in the Ven 4 study site, the potential productivity of this intermittent reach is further limited by rapid and accelerated flow reduction, presumably due to groundwater pumping by adjacent agricultural activities (Paul Jenkin, Surfrider Foundation, personal communication). Fisheries field crews also noted a rapid drop in flow and water surface elevations during the two day sampling period in mid July 2010, which appeared unnatural and artificially induced.

The SAC mid study site occurs in an open, broadening channel that is also subject to high insolation and low or intermittent flows. Sampling was conducted in the summers of 2007, 2010, 2011, and 2012, with a spring survey in 2011 (described above). In each year summer abundance estimates have been well below estimates from the SAC up study site (Figures 52 and 53), which is a perennial reach with thick riparian vegetation located 1.75 miles upstream of SAC mid. The lack of *O. mykiss* fry in the summer of 2012 was particularly surprising given the large redd that was observed in the SAC mid study site in March of that year. However, by the time of summer sampling (early July), habitat units had become extremely shallow and thick with algae and other aquatic vegetation. Also, permitted take levels for this project were reached prior to sampling SAC in 2012, which restricted sampling of all habitat types by dive counts only, which are highly ineffective in very shallow riffle and flatwater habitats. Despite these sampling limitations, the complete lack of fish observations in 2012 suggested that conditions in SAC mid had become inhospitable for *O. mykiss* and any surviving fry or juvenile+ fish may have emigrated to more suitable habitat.

The Mat 2 reach is a 0.6 mile long channel above Matilija Reservoir that frequently goes dry during summer months, however the higher flows present in 2011 (a wet year) allowed qualitative dive counts in six pools in late July. Despite high water temperatures (78^{o}F, 25.5^{o}C), one to five juvenile+ *O. mykiss* were observed in several pools, with a school of 20 *O. mykiss* (along with largemouth bass, sunfish, chubs, and turtles) in a deep, stratified pool that received a cold-water (67^{o}F) inflow from a bankside spring. Whether those *O. mykiss*, including a 35 cm fish, were immigrants from upstream reaches or from the downstream reservoir, is unknown.

The Mat 6 reach was sampled during the springs of 2010 and 2011 (described above), with a follow-up sample in the summer of 2011 under very low flow and warm (77^{o}F, 25^{o}C) water conditions. Dive counts in July 2011 yielded observations of *O. mykiss* in six of eight pools totaling zero fry and 11 juvenile+ fish. Expanding the dive counts to represent all available pools in the lower one mile of the 1.5 mile Mat 6 reach produced an abundance estimate (using simple random sampling formulas) of 45 (±31) juvenile+ *O. mykiss*. Converting to density produced an estimate of 0.14 juvenile+ per 100 ft^2 of pool habitat, which is approximately one-half the density of *O. mykiss* in Mat 5 pools and one-tenth the density in Mat 7 pools. Most of these juvenile+ fish were observed in the upper half of the sample area, closer to the upper Matilija Canyon and the Mat 7 study sites (Figure 28). The harsh conditions experienced by these fish in the largely open, non-vegetated channel is illustrated by the heavy infestation of black spot on a juvenile+ fish observed in lower Mat 6 in May 2010 (Figure 62).

In the summer of a wet year (2011) *O. mykiss* were somewhat abundant in the typically intermittent Mat 2 and Mat 6 reaches, whereas they remained relatively rare in the SAC mid and Ven 4 study sites. In each of those reaches, the estimated densities of *O. mykiss* were well below the estimated densities in adjacent, perennial reaches. These results suggest that, although intermittent reaches may contribute to *O. mykiss* production during the spring months or during summer months of wet years, their contribution remains well below that of perennial stream reaches.

5.5.5 Utilization of Lagoon Habitat

Qualitative, one-day sampling events were conducted in the Ventura Lagoon in August 2006 and 2007, and in late June 2011 (Table 12). The lagoon mouth was open during the 2006 and 2011 sampling, which occurred on both incoming and outgoing tides in 2006 and during outgoing tide in 2011, but the lagoon mouth was closed during the 2007 sampling. Most sampling effort utilized a

Figure 62. A 12 cm O. mykiss in Mat 6 showing heavy black spot infestation, May 2010.

4 ft x 100 ft beach seine with ½ inch mesh for 16 net hauls in 2006 and 2007 and 8 hauls in 2011 in locations throughout the lagoon. In addition to seining, 20-60 minutes of underwater video was recorded in deeper areas along the railroad's rip-rap bank, bridge abutments, and woody debris piles (Figure 21). Limited electrofishing was also conducted in the upper, riverine portion of the lagoon in 2006.

Seining locations ranged from 1.5 ft to 6 ft in depth, at water temperatures ranging from 64°F to 77°F (18°C to 25°C). Salinities approached seawater in some locations, but most were 5-20 ppt, with lower salinities (<2 ppt) in surface waters near the head of the lagoon. Dissolved oxygen levels recorded in 2011 were at or above saturation in both surface and bottom locations. No *O. mykiss* were captured or observed in any of the three surveys, although topsmelt (*Atherinops afinis*) were commonly captured in seine hauls in each year (Table 15). California killifish (*Fundulus parvipinnis*) were also commonly captured in 2007, but were rare or not observed in other years. No tidewater gobies were observed or captured, although the seine mesh was intentionally large (½ inch) to avoid capture of this listed species. Other fish that were occasionally captured were shiner surfperch (*Cymatogaster aggregate*), threespine stickleback, staghorn sculpin (*Leptocottus armatus*), prickly sculpin (*Cottus asper*), and arroyo chub. Striped mullet (*Mugil cephalus*) were abundant in the lagoon but were highly adept at escaping the net. Carp were sometimes observed during a seine haul but also avoided capture. Underwater video surveys along deeper portions of the lagoon revealed numerous schools of shiner surfperch and occasional sticklebacks, with aggregations of adult carp around bridge pilings and woody debris. The limited electrofishing conducted near the 101 bridge in 2006 resulted in the capture of chub, stickleback and sculpins.

Table 15. Number and species of fish captured in seining surveys in the Ventura Lagoon.

Date	Mouth	# sets	O. mykiss	Top-smelt	CA Killifish	Arroyo Chub	Shiner Perch	Prickly Sculpin	Staghorn Sculpin	Stickle-back	Striped Mullet	Carp
8/25/06	open	16	0	634	0	1	5	1	0	0	0*	0*
8/11/07	closed	16	0	234	36	0	0	1	2	1	0*	0*
6/30/11	open	8	0	188	1	0	0	0	1	0	0*	0

* fish observed at net but avoided capture

Although lagoon and estuary environments are known to important rearing habitat for *O. mykiss* and other anadromous salmonids in many west coast locations (Smith 1987, Miller & Sadro 2003, Quinones & Mulligan 2005), it is unknown to what degree the Ventura Lagoon may have supported extended rearing and growth of juvenile steelhead. Increased growth rates in lagoons and larger size at ocean entry has been found to result in greater ocean survival and increased returns of adult spawners (Reimers 1973, Ward & Slaney 1988, Bond et al. 2008). Anthropogenic impacts to lagoon physical habitat, including loss of wetland vegetation, channelization and bank armoring, and other impacts, may reduce the suitability of lagoons for *O. mykiss* rearing. Elevated water temperatures through reduced flows or loss of riparian vegetation may also reduce productivity of lagoon habitats for juvenile steelhead.

In the Scott Creek Lagoon, approximately 250 miles north of the Ventura River, juvenile steelhead reared through the summer months and achieved high growth rates (Bond 2006). Although most downstream migrant juveniles in the spring appeared to emigrate directly into the ocean without over-summering in the lagoon, the lagoon-reared juveniles were found to comprise 85% of the returning adult spawners over the following years. Water temperature or other water quality information was not provided for the Scott River Lagoon, but summer temperatures in the Ventura Lagoon exceeded 23^{o}C during all three years of sampling, and it is unknown to what degree water temperatures or other physio-chemical characteristics may be limiting use of the Ventura Lagoon by juvenile *O. mykiss*. In addition to potential water quality problems, the Ventura Lagoon is situated several miles downstream of any stream reaches found to contain substantial numbers of rearing juveniles, thus recruitment of juveniles into the lagoon may be limited except during the spring smolt migration season, when these larger individuals would be expected to pass through the lagoon into the ocean.

5.5.6 Correlations in Fish Abundance with Other Parameters

Correlations between parameters can be very useful in helping to explain observed trends in abundance or habitat associations, and can be explicitly used to improve abundance estimates. For example, correlations were evaluated and sometimes used to improve abundance estimates within study sites by using habitat unit length as an auxiliary variable (Section 4.4.3). Correlations in abundance of *O. mykiss* within individual habitat units between years were used in difference estimators to improve the ability to detect the significance of annual changes in abundance (Section 4.5.1). Correlations between age classes were noted above to be useful in explaining annual trends in abundance of juvenile+ *O. mykiss*. Lastly, correlations between abundance of *O. mykiss* in individual habitat units with physical parameters of those habitat units were utilized (via multiple regression) to formulate the habitat unit component of the SS HSI model (Section 4.3.1).

Correlations Between Abundance and Habitat Unit Lengths

The Method of Bounded Counts (MBC) estimators allow the use of auxiliary variables to improve the precision of abundance estimates (equations 20 and 21). Habitat unit length was utilized as a potential auxiliary variable because lengths were available for all habitat units mapped in each study site, and because the number of fish within a habitat unit is frequently correlated with a measure of unit size (e.g., length). The MBC protocols can calculate abundance estimates either with or without the use of auxiliary variables (Mohr and Hankin, unpublished manuscript). In this study, auxiliary variables were used wherever they increased the precision of the abundance estimate; where unit lengths were not correlated with unit abundances the auxiliary variable was not used. Unit length was found to be consistently correlated with *O. mykiss* abundance in several rivers studied to assess annual trends, including the upper Sacramento River following the Cantara Spill (TRPA 2005) and the North Fork Feather River following staged flow increases at Pacific Gas & Electric dams (Normandeau 2013), and these correlations were utilized in unequal-probability sampling designs to improve precision of abundance estimates and to better identify annual changes.

In the Ventura River Basin, correlations between unit lengths and *O. mykiss* abundance were highly variable and frequently near zero (and sometimes negative). However, of approximately 115 correlation coefficients (*r*) calculated for each size class using pool, flatwater, and riffle data from 2006, 2007, and 2010-2012, 71% and 63% were positive for fry and juvenile+, respectively, with approximately 30% of coefficients both positive and statistically significant (e.g., *r*'s >0.48 for mainstem study sites and *r*'s >0.40 for tributary sites). The distribution of mean correlation coefficients arranged by channel size (mainstem vs. tributary) and habitat type (pool, flatwater, and riffle) shows that correlations were consistently higher in tributary study sites than in mainstem study sites (Figure 63). All mean correlations for tributary habitat types exceeded 0.2, with correlations in flatwaters approaching 0.4. In contrast, mean correlations in mainstem habitats were low (<0.12) or negative except for juvenile+ fish in pools (mean r = 0.30). Correlations were slightly higher for fry than for juvenile+ in tributaries, but differences between size classes were highly variable in mainstem habitats.

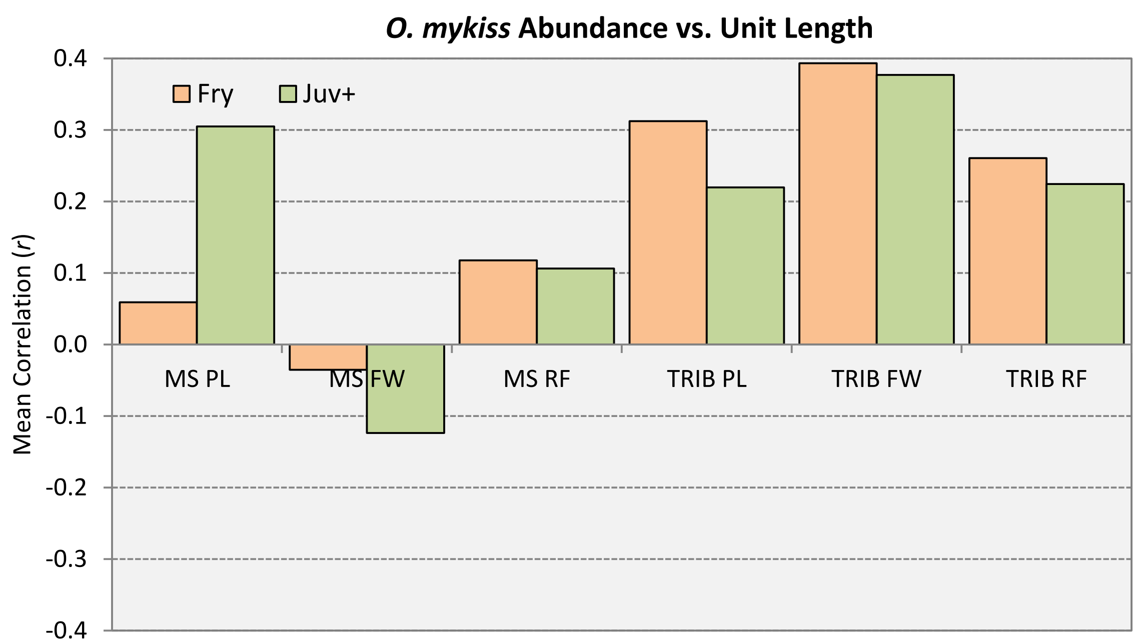


Figure 63. Mean correlations between abundance of O. mykiss and habitat unit lengths by size class, channel size and habitat type.

In summary, the use of unit length as an auxiliary variable or in a length-based unequal probability design is expected to improve estimator performance in locations where *O. mykiss* are relatively abundant (e.g., Ventura Basin tributary study sites), but locations containing relatively few fish (e.g., most mainstem study sites) produced poor correlations and may not yield improved performance. Although pool depths or pool volumes were not assessed as potential auxiliary variables in this study, studies utilizing a pool-only sampling design in tributaries may benefit from incorporating these alternative variables given the consistently good correlations between pool depth and *O. mykiss* (especially juvenile+) abundance (Spina et al. 2005, TRPA 2007b, 2009b).

Correlations in Unit Abundance Between Years

As stated above, the difference estimators described in Section 4.5.1 allowed a direct assessment of annual changes in abundance on a unit-by-unit basis. Because habitat units that contain relatively abundant *O. mykiss* in one year would be expected to contain relatively abundant fish in following years (assuming no major habitat changes), application of difference estimators (or other index-based approaches) that utilize these expected correlations can improve the ability to detect annual changes. Many of the adjacent-year abundance comparisons shown in Figures 49-59 show significant differences despite substantial overlap in confidence intervals, which illustrate the increased power of difference estimators to detect change. However the performance of difference estimators is dependent upon the correlations in abundance within sampling units between years.

Correlations in abundance of fry and juvenile+ *O. mykiss* within individual habitat units (Figure 64) sampled in adjacent years were mostly positive (64% and 76% of *r*'s by size class, respectively), with stronger correlations for juvenile+ fish (mean r = 0.32) than for fry (mean r = 0.22). Many of the poorer correlation values were based on units having low counts of less than 10 fish; this effect was most evident for juvenile+ *O. mykiss* where the mean correlation for units with higher counts (>10 fish) increased to 0.42. As seen for unit lengths, correlations between adjacent year abundance were also higher in tributaries than in mainstem study sites, with somewhat higher correlations for flatwater habitats than for pools and riffles.

Correlations in Abundance Between Cohorts

Several years of data suggest that strong recruitment of *O. mykiss* fry in the Ventura Basin may lead to relatively high abundance of juvenile+ the following year. A cohort comparison of fry abundance in year t with juvenile+ abundance in year $t+1$, for the four year-pairs where most study sites were sampled (2006-07, 2007-08, 2010-11, and 2011-12), shows strong and statistically significant (or nearly significant) relationships in most years (Figure 65). Only in the 2007-08 year-pair is the relationship weak (R^2=0.0, excluding Ven 3). The high variability in abundance estimates in the Ven 3 study site was previously discussed, and this variability also appears to influence the fry:juvenile+ cohort relationship. In several years the Ven 3 and Ven 5 study sites produced a higher year $t+1$ abundance of juvenile+ than the preceding years estimate of fry abundance, which supports prior observations that these mainstem reaches may be important rearing areas for fry immigrating from upstream tributaries, such as San Antonio Creek and the lower North Fork Matilija Creek.

Because a variable and unknown (but likely small) proportion of the fry size class may actually represent 1+ *O. mykiss*, and some unknown proportion of the juvenile+ size class represents 2+ or older fish, these cohort relationships are only approximate; nevertheless the data from this study (and others, e.g., TRPA 2005) suggests that information on fry abundance may be useful for estimating the abundance of older, smolt-sized *O. mykiss* in the subsequent year, and further emphasizes that limitations on fry production may lead to limitations in production of smolts and perhaps, ultimately, on returns of adult steelhead.

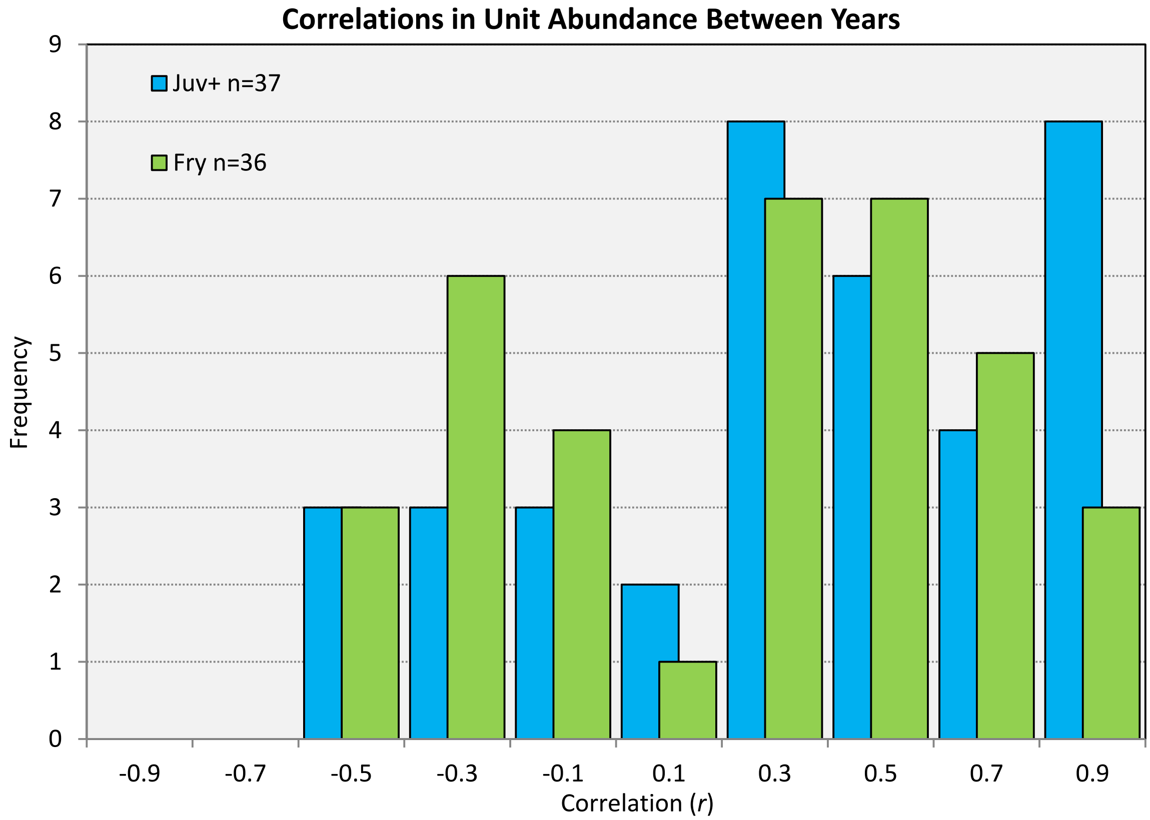


Figure 64. Mean correlations in abundance of O. mykiss in individual sampling units between adjacent years, according to size class.

Correlations in Abundance with Habitat Parameters

The stepwise regression models utilized in the development of the habitat type component of the SS HSI model (Section 5.4.1) relied on correlations between unit-specific abundance of *O. mykiss* and habitat parameters in those units. Due in part to autocorrelations between habitat variables (e.g., the proportion of velocities >0.5 fps was highly correlated with the proportion of velocities >1 fps), some variables that were correlated with *O. mykiss* abundance were not added to a stepwise model; and likewise some variables that did not individually appear correlated with abundance were useful in explaining abundance and were thus added to a model. Because the SS HSI model and its associated suite of habitat parameters (Table 5) was only developed using variables collected in 2012, the following analysis relies only on 2012 data. Note that the dry water year in 2012 resulted in low base flows during sampling despite sampling earlier in the summer compared to previous surveys. Consequently it is possible that habitat limitations (e.g., riffle depths, pool velocities) may have influenced correlations between abundance and habitat parameters, and that these results may be more representative of drought conditions than for normal or wet water years.

Table 16 shows an example correlation matrix for the mainstem riffle habitat class, not including variables that were ultimately discarded prior to fitting stepwise regressions (see Section 4.3.1 for discussion of redundant or rare variables). The correlations between density (#/100 ft^2) of fry (<10 cm), juvenile (10-20 cm), and adult (>20 cm) *O. mykiss* were generally positive for depth, velocity,

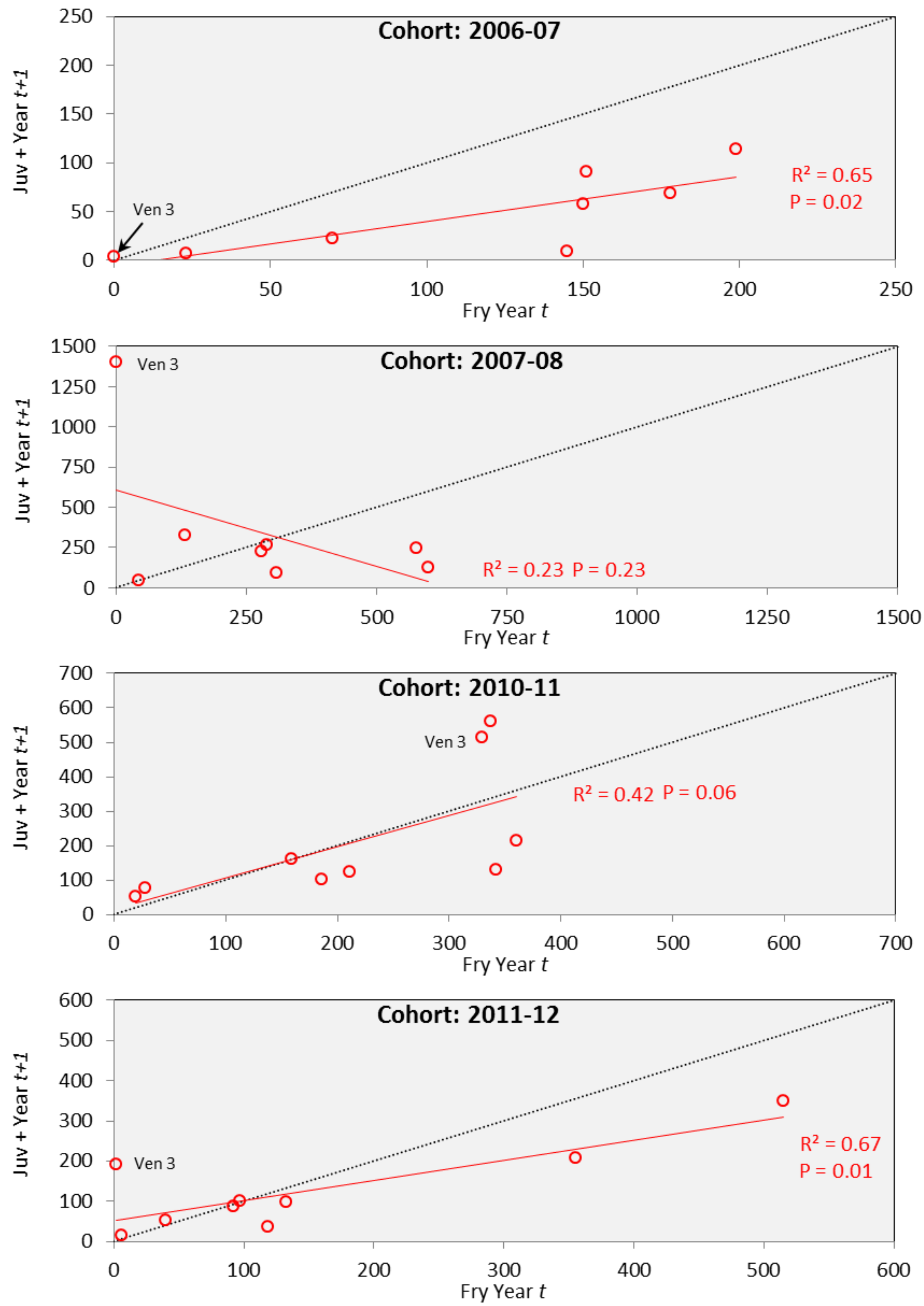


Figure 65. Relationship between abundance of O. mykiss fry in year *t* with abundance of juvenile+ in year *t+1*.

and many cover variables, whereas correlations were weakly negative for percent fines. These relationships were not always evident for the other habitat classes where, for example, correlations with depth variables were typically negative in mainstem and tributary flatwater habitats (Appendix E).

Summarizing these habitat correlations by averaging correlation coefficients among habitat types, but separated by channel size and fish size class, produced several evident trends. For the depth variables, fry occurred at highest densities in shallow units dominated by depths less than one foot in both mainstem and tributary study sites (Figure 66, upper half). Juvenile *O. mykiss* also appeared to prefer shallow habitats in the mainstem, but were slightly correlated with deeper habitats in tributary sites. Densities of adult fish were consistently correlated with deeper habitat units, particularly in the smaller tributary study sites. For the velocity variables (Figure 66, lower half), fry and juveniles occurred at highest densities in units containing higher proportions of velocities exceeding 0.5 fps in both mainstem and tributary study sites, consistent with the higher densities observed in riffle habitats. Likewise for juveniles in mainstem sites, but densities of juveniles in tributaries and adults in both mainstem and tributaries showed little relationship with unit velocities.

The relationships between *O. mykiss* densities and percent cover was generally weak, except for the percentage of cobble/boulder substrate for fry and juveniles in both mainstem and tributary study sites, and for turbulence and all cover (combined) in mainstem sites (Figure 67, upper half). Adult *O. mykiss* showed little relationship to any cover types, although it should be noted that fish >20 cm in length were typically uncommon, especially in the tributaries. Consequently, the mean correlations for that size class (and overall regression results, Section 5.4.1) should be viewed with caution. The combined velocity and cover variables (Table 5) showed consistently strong (~0.5) correlations with densities of fry and juveniles in mainstem habitats, with lesser correlations with adults (Figure 67, lower half). These variables showed little correlation with any size classes in tributary habitats.

Correlations in Abundance Between Survey Methodologies

The abundance estimates in pool habitats and in many mainstem flatwater habitats relied upon the MBC protocols (Mohr and Hankin, unpublished manuscript) for assessing differences in abundance between study sites and between years. The repeat dive count methodology and associated bias-adjustment formulas (Section 4.4.3) are used to estimate the diver observation probabilities and apply those probabilities to estimate total abundance of fish within a given study strata (e.g., all pools in Ven 3). Because this method has not been extensively applied in other southern California steelhead streams, limited assessments were made to compare bias-adjusted dive count estimates with follow-up estimates based on multiple-pass electrofishing using the jackknife estimator.

Comparative bounded dive counts and multiple-pass electrofishing passes were conducted within five pools in 2006 and in four flatwaters in 2011. Only one mainstem study site was represented, which did not contain any fish; all other sites contained 1-10 fish and were located in tributary or headwater study sites. The relationships between estimates of abundance of fry and of juvenile+ *O. mykiss* were positive but weak (R^2 <0.2), with generally higher dive count estimates for fry and higher electrofishing estimates for juvenile+ fish (Figure 68, upper graph). When combined across size classes, the 2006 estimates were highly comparable (R^2=0.95), with slightly higher estimates from electrofishing (Figure 68, lower graph), but the relationship in flatwaters in 2011 remained poor with higher estimates derived from dive counts.

Table 16. Correlation table showing Pearson correlation coefficients between O. mykiss density and habitat parameters for mainstem riffle habitats. Significant correlations (P<0.05) with density are shown in yellow highlight. See Table 5 for variable descriptions; correlation tables for other channel/habitat type strata are shown in Appendix E.

	Depth Variables			Velocity Variables			Cover Variables							Combination Variables			
VAR	AvDep	MaxDep	D1	AvVel	V05	V1	CB	Turb	IWBR	OHVeg	RipVeg	AllCov	Fines	V05,IW	V05,OW	V1,IW	V1,OW
Fry	0.10	0.21	-0.02	0.26	0.17	**0.39**	0.27	**0.37**	-0.19	-0.27	-0.25	0.37	-0.17	0.24	0.30	**0.39**	0.35
Juvenile	0.23	0.31	0.02	0.27	0.24	**0.38**	0.21	**0.38**	-0.22	-0.28	-0.27	0.33	-0.02	0.32	0.34	**0.42**	0.35
Adult	**0.54**	0.25	**0.50**	0.16	0.12	0.21	0.27	0.15	-0.09	-0.22	-0.13	0.28	-0.07	0.31	0.09	**0.37**	0.13
AvDep		0.68	0.89	-0.14	-0.24	-0.08	0.53	0.07	-0.34	0.15	0.16	0.53	0.32	0.25	0.06	0.22	0.05
MaxDep			0.60	-0.27	-0.28	-0.18	0.51	0.05	-0.35	-0.15	0.17	0.37	0.17	0.23	0.03	0.18	0.00
D1				-0.21	-0.30	-0.11	0.43	0.09	-0.25	0.10	0.15	0.44	0.35	0.07	0.03	0.07	0.05
AvVel					0.82	0.93	-0.14	0.79	-0.03	0.18	0.04	0.34	-0.58	0.24	0.73	0.44	0.80
V05						0.79	-0.17	0.67	0.19	0.03	-0.15	0.18	-0.53	0.45	0.67	0.47	0.69
V1							-0.04	0.88	-0.15	0.11	0.04	0.44	-0.57	0.30	0.81	0.51	0.90
CB								-0.02	-0.32	-0.18	0.12	0.80	-0.10	0.69	-0.01	0.64	-0.01
Turb									-0.14	0.16	0.09	0.52	-0.40	0.19	0.95	0.37	0.99
IWBR										-0.01	-0.14	-0.35	0.08	-0.18	-0.15	-0.17	-0.15
OHVeg											0.54	0.07	0.21	-0.21	0.27	-0.17	0.22
RipVeg												0.20	0.04	-0.08	0.17	0.02	0.13
AllCov													-0.22	0.61	0.52	0.67	0.53
Fines														-0.34	-0.35	-0.46	-0.42
V05,IW															0.24	0.90	0.21
V05,OW																0.35	0.96
V1,IW																	0.37
V1,OW																	

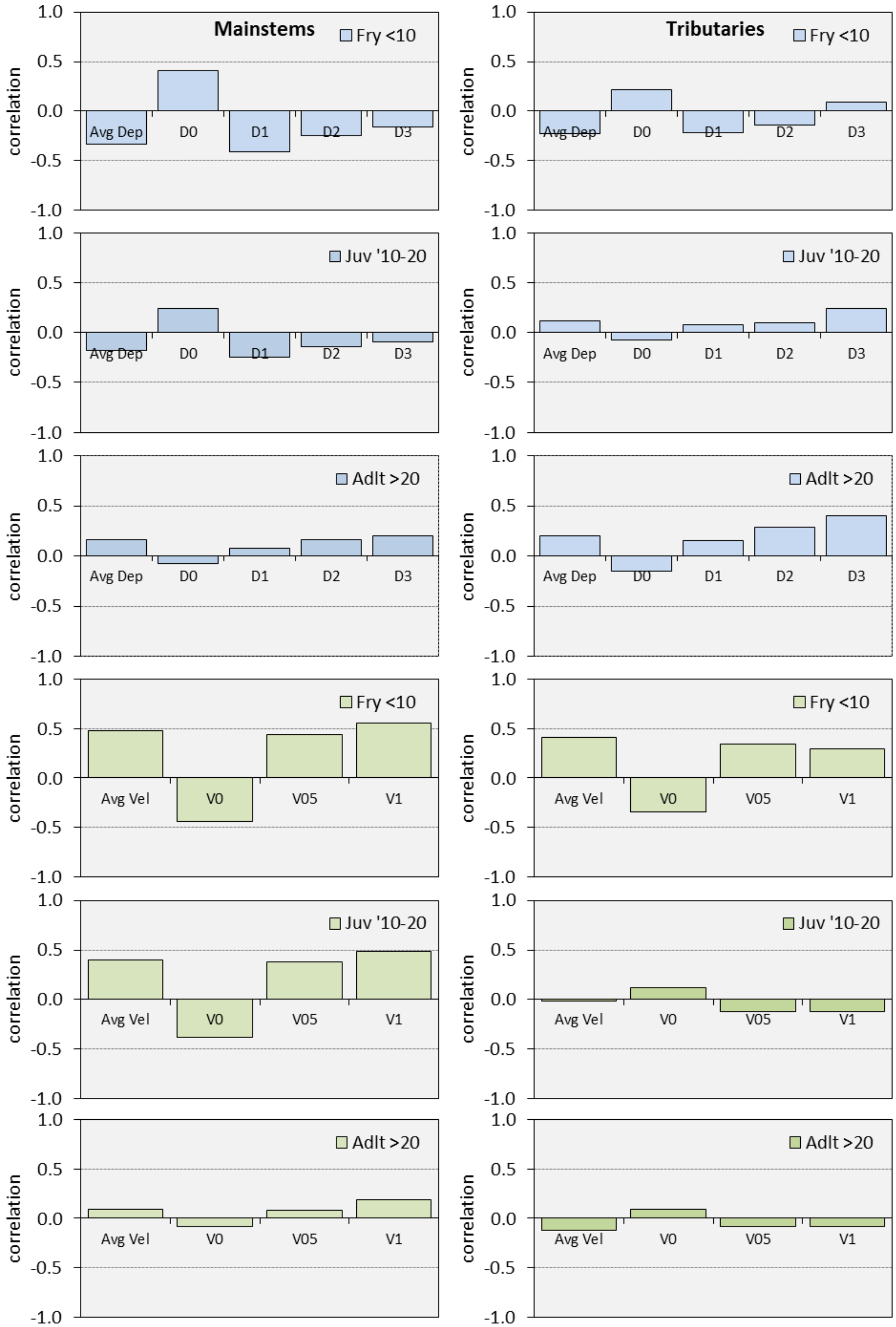


Figure 66. Mean correlations between density (#/100 ft2) of O. mykiss in 2012 and depth (upper 3 rows) or velocity (lower 3 rows) variables, according to channel size and fish size class. See Table 5 for variable definitions.

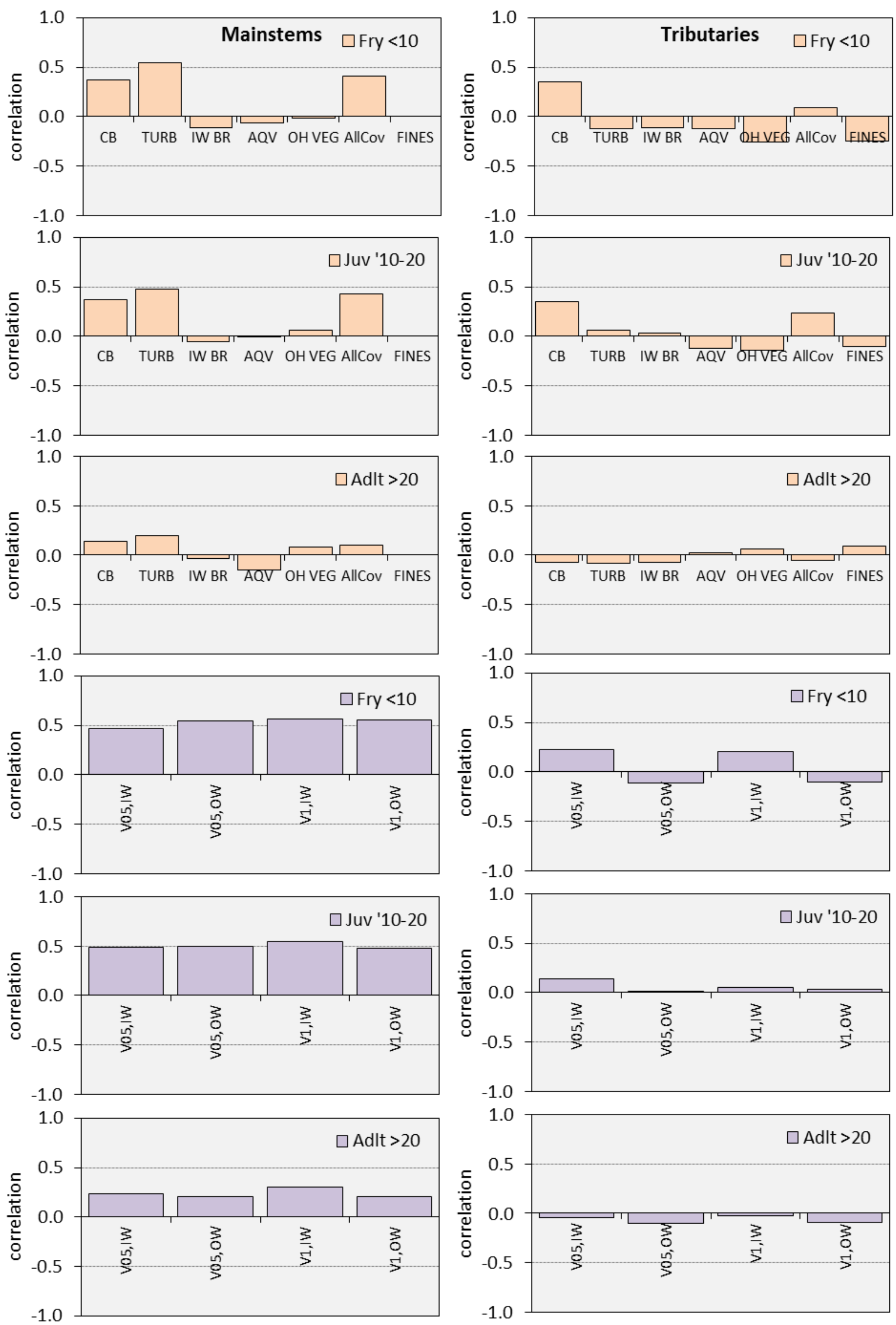


Figure 67. Mean correlations between density (#/100 ft2) of O. mykiss in 2012 and cover (upper 3 rows) or combined velocity/cover (lower 3 rows) variables, according to channel size and fish size class. See Table 5 for variable definitions.

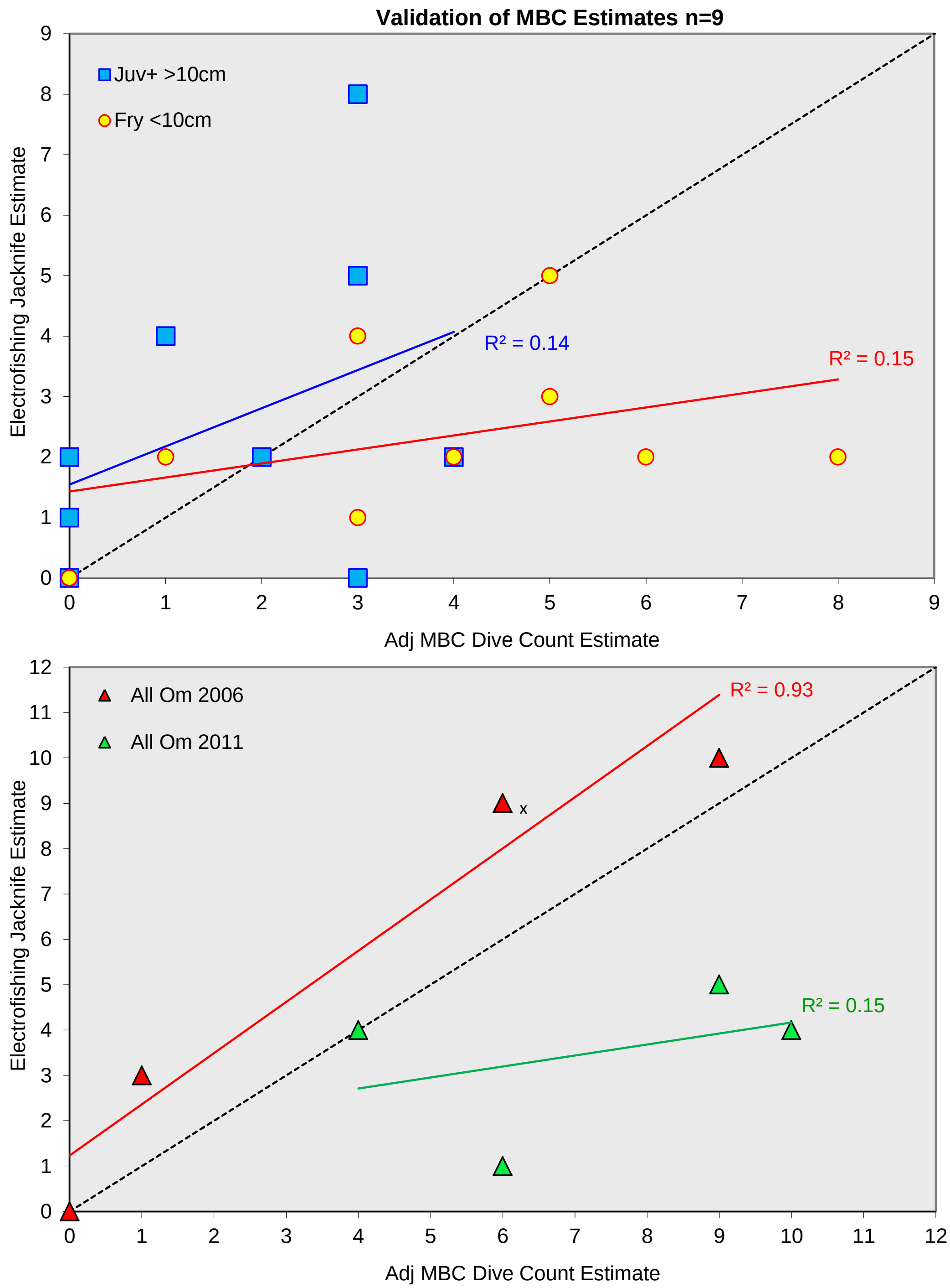


Figure 68. Comparison of abundance estimates using bounded dive counts and multiple-pass electrofishing.

Additional data is clearly needed to better assess the reliability of using repeat dive counts for estimating abundance of *O. mykiss* in southern California streams, however it should be noted that most habitat units sampled by diving in the Ventura Basin were relatively shallow with clear water and little vegetation or woody debris cover, thus providing near ideal conditions for conducting dive counts in pool and deeper flatwater habitats. Although bounded counts were conducted in shallow flatwaters and riffles in 2012 due to permit restrictions (Figure 69), dive counts in such shallow water habitats are not likely to be representative and should be avoided whenever possible (i.e., these are best assessed using electrofishing).

Correlations in Annual Trends Based on Pool-Only Surveys vs All Habitat Surveys

Given the endangered status of Southern Steelhead and the relative simplicity, low cost, and ease of permitting for conducting surveys based on direct observation (snorkel) methodologies, an increasing number of steelhead surveys are relying strictly on dive counts to assess the relative distribution and abundance of fish in many salmonid watersheds. However, it is our experience that dive counts are not reliable in riffle habitats except in larger mainstem rivers, nor in flatwater habitats in most tributary and headwater reaches (Figure 69). Consequently, a diving-only survey protocol conducted in most central and southern California stream reaches that support significant numbers of *O. mykiss* may only be effective in pool habitats.

Information presented above indicated that *O. mykiss* fry were found at densities up to six-times higher in riffles than in pools (Table 11), with a similar (but less extreme) difference noted for juveniles 10-20 cm in length. This density versus habitat type relationship was consistent across years and was present even within small headwater reaches containing very shallow riffle habitats (Figure 52). If riffle and flatwater habitats make-up a significant proportion of habitat in a given stream reach, as is the case in all Ventura study sites, Figure 34), then application of a pool-only dive count protocol may only assess a minor proportion of fish that are present in a given reach or basin.

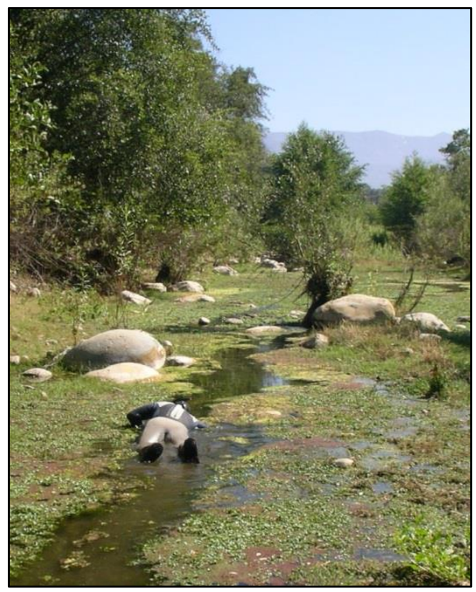


Figure 69. Example of ineffective dive count in shallow water habitat.

To assess whether a survey protocol involving only dive counts in pools and large channel flatwater habitats will effectively represent annual trends in abundance, a comparison was made between annual abundance estimates (maxima normalized to 1.0) from all habitat types combined (using dive counts and electrofishing) with estimates based only on dive counts (note that riffles and tributary flatwaters were not sampled in 2008 and 2009, and the all habitat estimates for those years were based on linear regression, Section 4.5.1).

In general, annual trends showed good correspondence (Figure 70), with correlation coefficients of 0.87 for fry and 0.74 for juvenile+ *O. mykiss* (all segments combined). Of 32 adjacent-year changes in abundance, only 4 direction changes (e.g., increases or decreases) were not consistent: the 2010-11 (middle segment) and 2011-12 (upper segment) changes for fry, and the 2009-10 and 2010-11 (middle segment) changes for juvenile+.

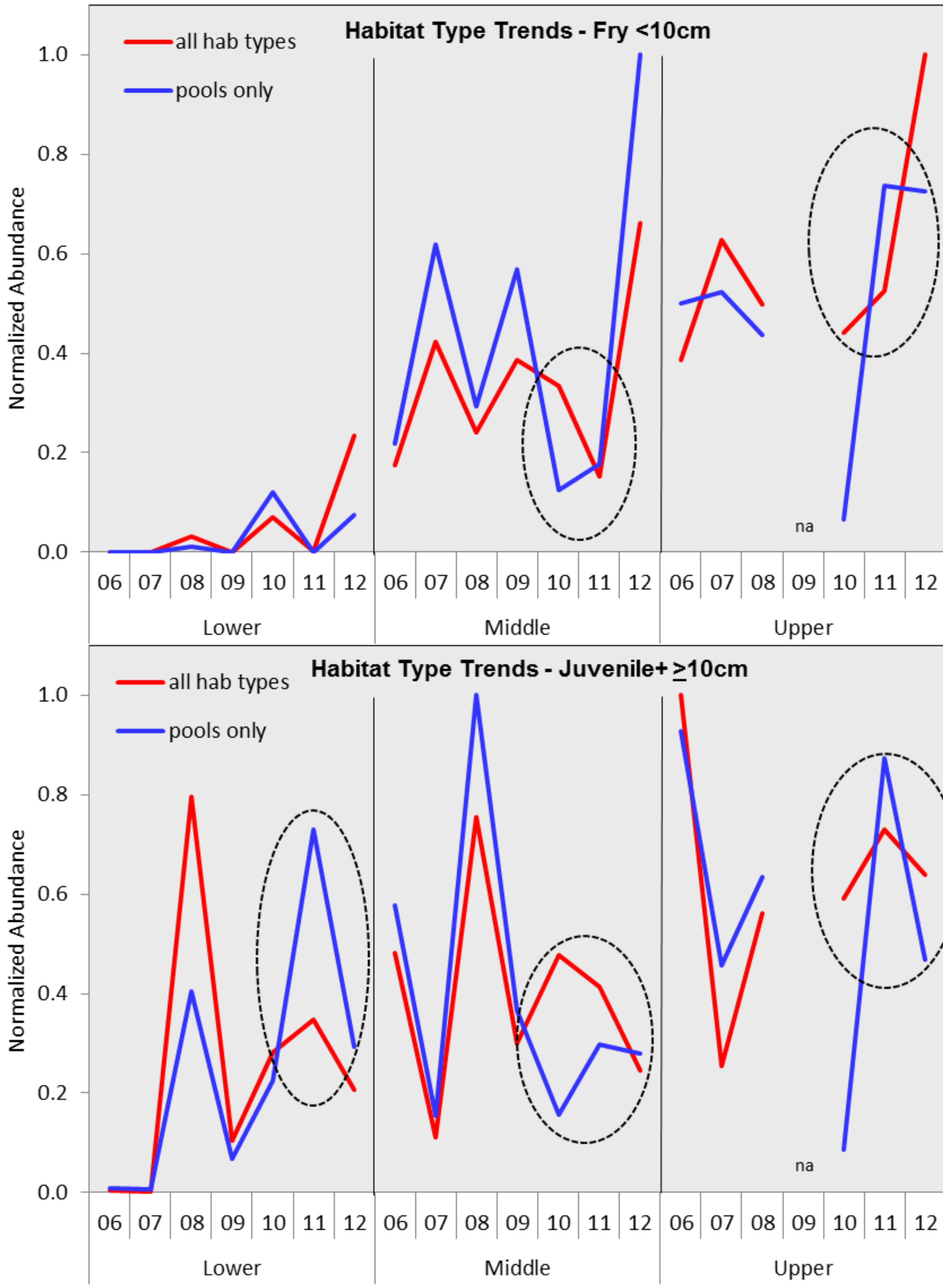


Figure 70. Comparison of annual trends in normalized abundance based on sampling all habitats versus sampling pools-only, by size class. Dotted circles illustrate differences discussed in the text.

However, in many cases, the relative magnitude of changes differed substantially between the all habitats and pool-only estimates, particularly the changes observed from 2010-11 and 2011-12 for both size classes, where the pools-only estimates suggested far more extreme fluctuations in abundance than did the estimates based on all habitat types. In another case, the all-habitats estimates for juvenile+ suggested that maximum abundance occurred in 2008 in the lower segment, whereas the pool-only estimates suggested maximum abundance in the lower segment occurred in 2011. This difference may be due in part to the lack of electrofishing data in 2008, and the uncertainty surrounding the regression-based estimates of abundance in riffle habitats in that year.

In summary, the available data suggest that pool-only sampling designs may be effective in describing general trends in annual abundance, although overall abundance estimates will always be lower (often by orders of magnitude), and year-to-year fluctuations are likely to be exaggerated, when compared to estimates based on all habitat types.

5.6 Relationships Between *O. mykiss* Abundance and HSI Scores

The abundance of *O. mykiss* estimated each year in each study site (Section 5.5) along with the development and/or revision of HSI scores at each site in 2006, 2007, 2011, and 2012 (Sections 5.3 and 5.4) allow a direct comparison of the relationship between estimates of habitat quality and fish populations. In theory, a strong positive relationship would provide confidence that the environmental variables encompassed by the HSI model are effective surrogates for the fish abundance information, which is typically more difficult and expensive to acquire. As noted in Section 5.3, previous assessments of the USFWS HSI model resulted in statistically significant relationships with *O. mykiss* abundance during the same years (TRPA 2007, 2008); however the pattern and distribution of HSI scores across the study sites displayed a narrow range of values that did not appear consistent with visual estimates of habitat quality, and did not effectively distinguish between study sites containing relatively low abundance with sites that rarely held fish. Consequently, the USFWS HSI models were reassessed in 2012 using alternative model options, and the Southern Steelhead (SS) HSI model was developed in an attempt to account for limitations and missing variables in the USFWS model.

Two datasets were used in this assessment (Table 17): a comparison of 2012 HSI scores versus 2012 *O. mykiss* densities (#/100 ft^2) by study site; and a comparison of mean values from the four years of HSI data collection (2006, 2007, 2011, and 2012) with mean *O. mykiss* densities over the five years of full pool, flatwater, and riffle sampling (same as HSI sampling years plus 2010). This latter comparison using mean values was used to provide a more representative estimate of typical habitat characteristics and fish population abundance rather than relying on a single year. Although 2012 was characterized by high abundance of *O. mykiss*, the very low flows which occurred that year may have resulted in habitat limitations. Note that the SS HSI model was only available from 2012 data collection; consequently the 2012 SS HSI scores were compared to both the 2012 *O. mykiss* densities and also to the five-year mean density estimates. Also note that some mean densities, such as the SAC and Mur values, are based on fewer years of data (see Table 1 for sampling periodicities).

The comparison using only the 2012 data shows best fit for the SS HSI and the USFWS Non-Compensatory (NC) resident trout models (Figure 71, top), both of which explain 50-60% of the observed variation in 2012 *O. mykiss* densities (all size classes combined). Both models gave HSI scores of zero to the Ven 4 study site due to the dry channel, and these zero scores degraded the explained variation by about 10%. The SS HSI model for resident trout produced the widest range (0.44) in HSI scores of all four model options, from 0.22 (Ven 1) to 0.64 (UNF), excluding the zero score for Ven 4. The USFWS model options produced narrower ranges of 0.25, 0.35, and 0.38 for the equal components (EQ), compensatory (C), and NC models, respectively. As stated above, the narrow ranges in HSI scores for the original USFWS EQ model did not appear consistent with visual assessments of the habitat quality among the study sites; consequently the wider range in scores of the SS HSI model may be more realistic and consistent with the wide range in observed densities of *O. mykiss*.

Table 17. Comparative O. mykiss densities and HSI scores depending on model type (FWS or SS alternatives), fish life-history type (resident rainbow trout or steelhead), and time frame (2012 only or mean of multiple years), according to study site.

Study Site	O. mykiss Density #/100 ft2			FWS HSI EQ Model			FWS HSI NC Model		FWS HSI C Model		SS HSI Model	
	RBT 12	RBT 07-12	STH 12	RBT 12	RBT 07-12	STH 12	RBT 12	RBT 07-12	RBT 12	RBT 07-12	RBT 12	STH 12
Ven 1	0.01	0.00	0.00	0.77	0.72	0.68	0.21	0.24	0.35	0.41	0.22	0.37
Ven 2	0.05	0.02	0.04	0.71	0.71	0.64	0.20	0.25	0.37	0.43	0.26	0.42
Ven 3	0.75	0.34	0.70	0.78	0.72	0.72	0.36	0.33	0.50	0.52	0.37	0.50
SAC mid	0.00	0.06	0.00	0.64	0.63	0.61	0.24	0.24	0.46	0.46	0.25	0.37
SAC up	0.16	0.23	0.14	0.79	0.77	0.79	0.40	0.37	0.54	0.53	0.44	0.54
Ven 4	0.00	0.01	0.00	0.00	0.35	0.00	0.00	0.04	0.00	0.28	0.00	0.00
Ven 5	0.35	0.36	0.33	0.85	0.74	0.80	0.30	0.28	0.39	0.47	0.26	0.40
LNF new	1.50	0.88	1.42	0.83	0.77	0.86	0.42	0.40	0.54	0.56	0.58	0.62
LNF mid	3.79	2.31	3.75	0.81	0.75	0.82	0.41	0.40	0.55	0.58	0.53	0.57
Mat 3	0.75	0.25	0.74	0.72	0.72	-	0.24	0.28	0.41	0.46	0.20	-
Mat 5	1.17	0.89	1.15	0.79	0.73	-	0.34	0.36	0.49	0.56	0.42	-
Mat 7	2.41	1.68	2.39	0.88	0.79	-	0.46	0.46	0.55	0.63	0.46	-
UNF	4.75	2.73	4.68	0.85	0.84	-	0.53	0.54	0.65	0.67	0.64	-
Mur	2.34	2.34	2.34	0.78	0.78	-	0.55	0.55	0.72	0.72	0.60	-

Repeating this comparison using mean HSI scores and mean *O. mykiss* densities produced somewhat better fits for all four models (Figure 71, bottom), with 60-65% of variation explained by the SS HSI (R^2=0.61), USFWS C (R^2=0.65), and the USFWS NC (R^2=0.63) models. The 2012 SS HSI model produced a range in scores that doubled the range from the averaged USFWS EQ model (0.44 vs. 0.21, excluding Ven 4), and was 50% greater than ranges produced by the averaged USFWS C and USFWS NC models (both ranges 0.31). Note that the averaged HSI scores produced non-zero scores for the Ven 4 study site since habitat was available during the wetter summers of 2006, 2007, and 2011.

Incorporating the anadromous components of both the USFWS HSI model (EQ option only) and the SS HSI model for those study sites below Matilija Dam with the 2012 HSI data produced similar but poor fits (R^2=0.15 for RBT and STH models) for the USFWS EQ model, and also a much degraded fit (R^2=0.26 vs. 0.59) for the SS HSI model (Figure 72). The SS HSI score for steelhead was degraded by the loss of upper basin scores (which were out of the anadromous zone) as well as the zero HSI score for Ven 4 and the relatively high *O. mykiss* densities in the LNF mid study site. As has been noted previously, many of the *O. mykiss* that are present in the anadromous zone below Matilija Dam are fish that appear to have chosen a resident trout pathway; consequently the rainbow trout HSI model alternative may be a more appropriate option for comparison with actual *O. mykiss* densities in the lower and middle segments of the Ventura River Basin.

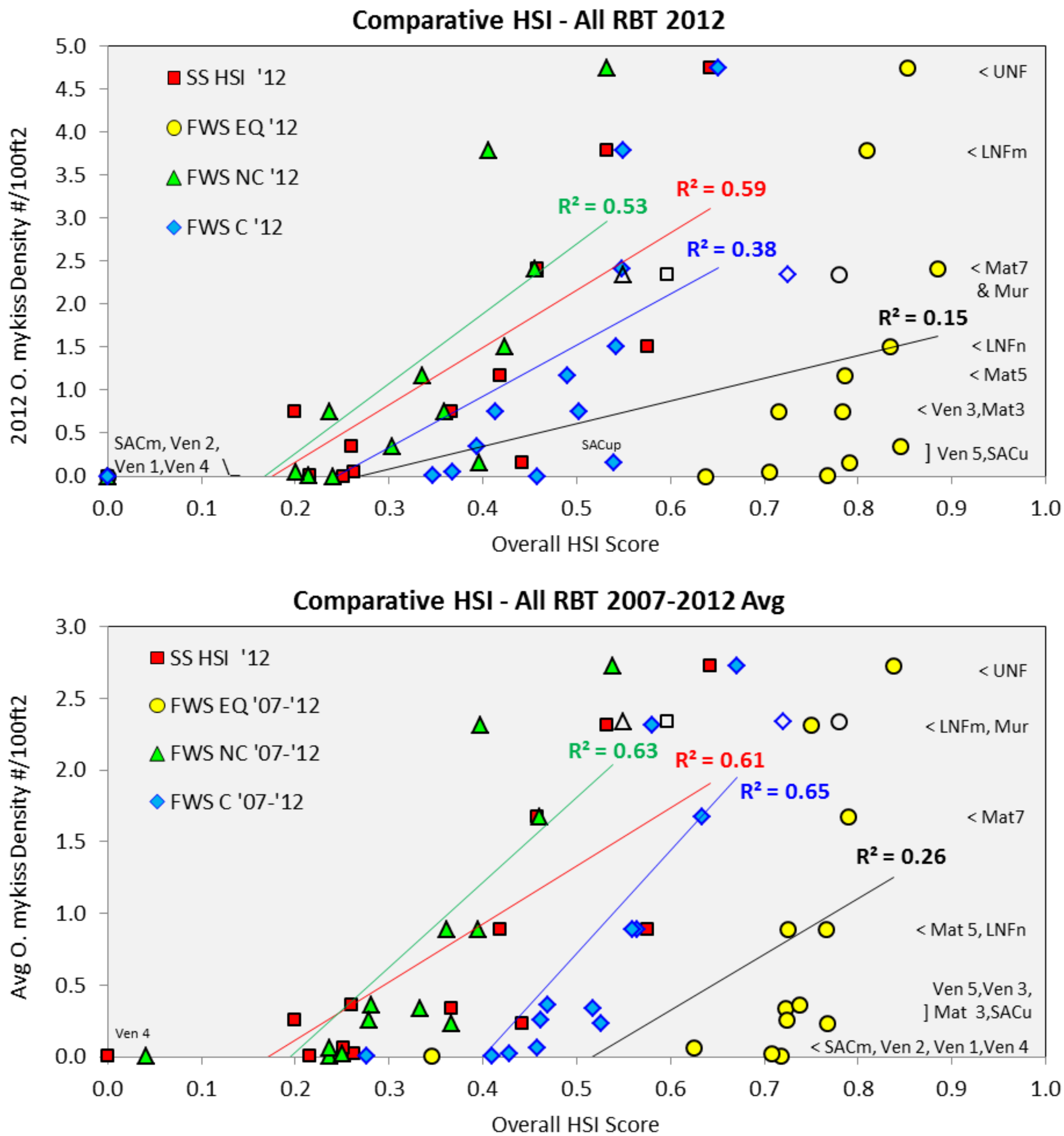


Figure 71. Comparison of resident trout HSI scores with O. mykiss densities (all sizes combined) according to model options. Note the Murietta HSI scores (unfilled symbols) were not used to fit the regressions.

In conclusion, the SS HSI model produced the best fit to the observed densities of *O. mykiss* in 2012 and a similar fit as two of the USFWS HSI models when using mean habitat and density values, and in addition the SS HSI model produced a wider range in estimated HSI scores that appeared more consistent with visual assessments of habitat quality among the HSI study sites. Incorporating steelhead-related parameters into the SS and USFWS HSI models resulted in reduced performance with poor fit. The proposed SS HSI model, which was developed and "fit" to the observed abundance data in the Ventura River Basin, should be carefully assessed to determine its utility in other southern California watersheds.

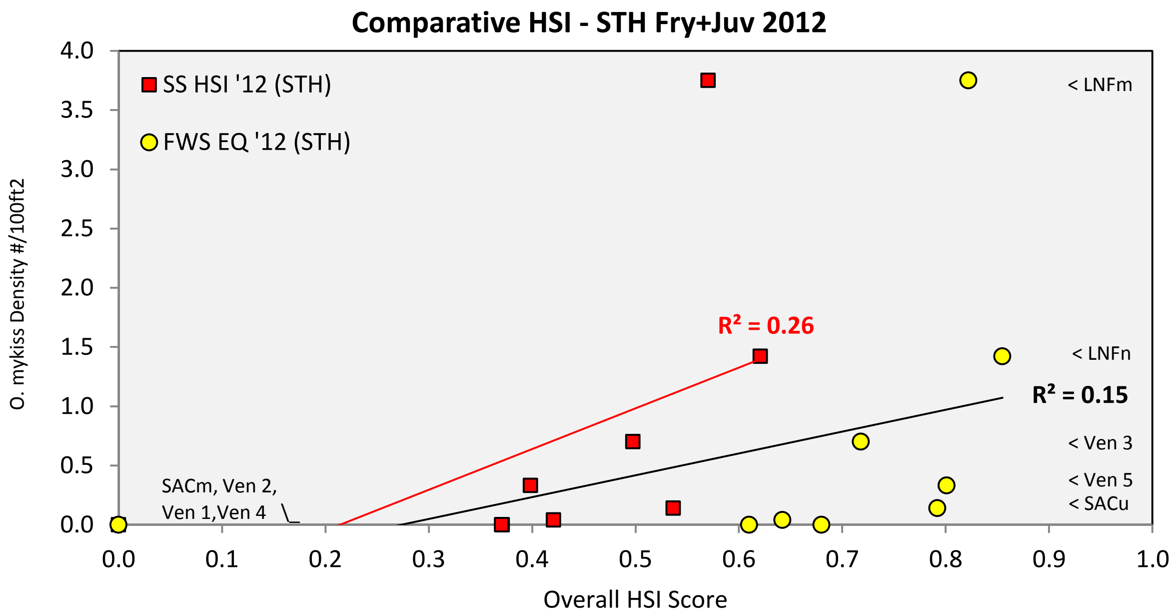


Figure 72. Comparison of steelhead HSI scores with O. mykiss densities (fry and juveniles combined) according to model options.

5.7 Sample Size Requirements to Detect Trends in Abundance

Given the endangered status of steelhead in the Southern California ESU and the projected increases in ambient temperatures over the next century, the detection of real trends in abundance is of vital concern. Unfortunately, data describing the annual trends of *O. mykiss* populations in southern and south-central California basins in recent years are relatively rare, of short term, or based on highly qualitative estimates of abundance. Multi-year datasets that contain quantitative abundance estimates, such as data from this study, reveal high variability in both spatial and temporal aspects, and such variability will hinder efforts to confidently assess trends in abundance. Recent sampling designs intended to monitor abundance trends, such as the California Monitoring Plan (CMP), are long-term in nature and have only just begun to be applied in central and southern California steelhead basins (CDFG 2011).

Assessing Annual Trends in Abundance

Information from this quantitative, seven-year study can be used to assess the sample size requirements that may be necessary to detect actual trends in abundance according to various spatial scales (e.g., low density/mainstem reaches vs. high density/headwater reaches), sampling designs (e.g., pool-only dive counts vs. all habitats diving/electrofishing), and cohorts (e.g., fry, juveniles, all *O. mykiss*). As described in Section 4.5.3, the program *Trends* (Gerrodette 1993) was used to evaluate the variability of *O. mykiss* abundance in the Ventura Basin and predict the number of years of sampling that may be required to detect an annual trend of 10%, given specified error rates and structure of variation in abundance. The power curves presented below were calculated in *Trends* using a two-tailed test with a type-I (α) error rate of 0.1 with the assumptions of linear change with the C.V. of abundance proportional to abundance. Each curve displays an estimate of the statistical power to detect a 10% change in abundance based on the number of years of sampling, assuming a level of sampling effort and sampling design efficiency similar to that used in this study (e.g., a randomized design using segment, reach, habitat type, and fish size class strata).

Because the C.V.'s of abundance estimates increased with the magnitude of abundance (based on analysis of the Ventura data), the power to detect an increase in abundance for a given sample size (number of years) was considerably lower than the power to detect a decrease in abundance (in which case C.V.'s would also decrease). In other words, a shorter sampling time frame would be required to detect a decrease in abundance, which is a more critical result with endangered species, than would be required to detect an increase in abundance. Also, as would be expected, those study strata that display higher levels of annual variation in abundance (e.g., lower segment sites such as Ven 3) would require a longer time period to detect a statistically significant change than would a study reach that shows less annual variation (e.g., the UNF).

Consistent with these expectations, output from the program *Trends* shows that it may take twice as long to detect a 10% increase in annual abundance at a typical level of power (say, 80%) than to detect a 10% decrease in abundance (Table 18 and Figure 73). Shorter time frames may be required to detect a 10% decrease in abundance when all size classes of *O. mykiss* are combined as opposed to estimates for fry or juvenile+ fish alone. Also, less time may be required if all habitat types (pools, flatwaters, and riffles) are sampled versus sampling pools only. Specifically, the Ventura dataset suggests that a 10% decrease in abundance can be detected with 80% power over a 7 to 8 year period in the upper and middle segments if all habitat types are sampled (versus 10 years for pool-only sampling), whereas 15 to 20 years may be required to detect the same decline in the highly variable lower segment.

Table 18. Estimated time to detect a 10% change in abundance of O. mykiss with 80% power, according to segment, sampled habitat, and fish size class.

		# Yrs to Detect 10% DECREASE			# Yrs to Detect 10% INCREASE		
Segment	Habitat	All O.m.	Juv 10+	Fry <10	All O.m.	Juv 10+	Fry <10
All	All Habitats	8	10	10	12	18	>20
	Pools Only	9	10	10	16	>20	>20
Upper	All Habitats	7	9	9	11	16	16
	Pools Only	10	10	10	20	>20	20
Middle	All Habitats	8	10	10	12	>20	>20
	Pools Only	10	12	12	19	>20	>20
Lower	All Habitats	16	20	20	>20	>20	>20
	Pools Only	15	20	20	>20	>20	>20

The results listed in Table 18 suggest that the 7-year study described in this report would be nearly sufficient to statistically detect a 10% decrease in abundance of *O. mykiss* (all sizes combined) in all habitats (combined) and in the entire basin or in the upper and middle segments, if such a decline existed. Consequently, linear regression was used to assess the observed trends in estimated abundance of all *O. mykiss* from 2006 to 2012, according to those spatial scales.

None of the regression models describing annual trends were statistically significant, with slopes not significantly different from zero (Figure 74). Although all segments (including combined segments) showed a positive slope, these apparently increasing trends were almost wholly due to the high abundance estimates from the last year of sampling (2012), as regression slopes were essentially flat when the 2012 data was removed. In comparison to the stated criteria of a 10% increase in

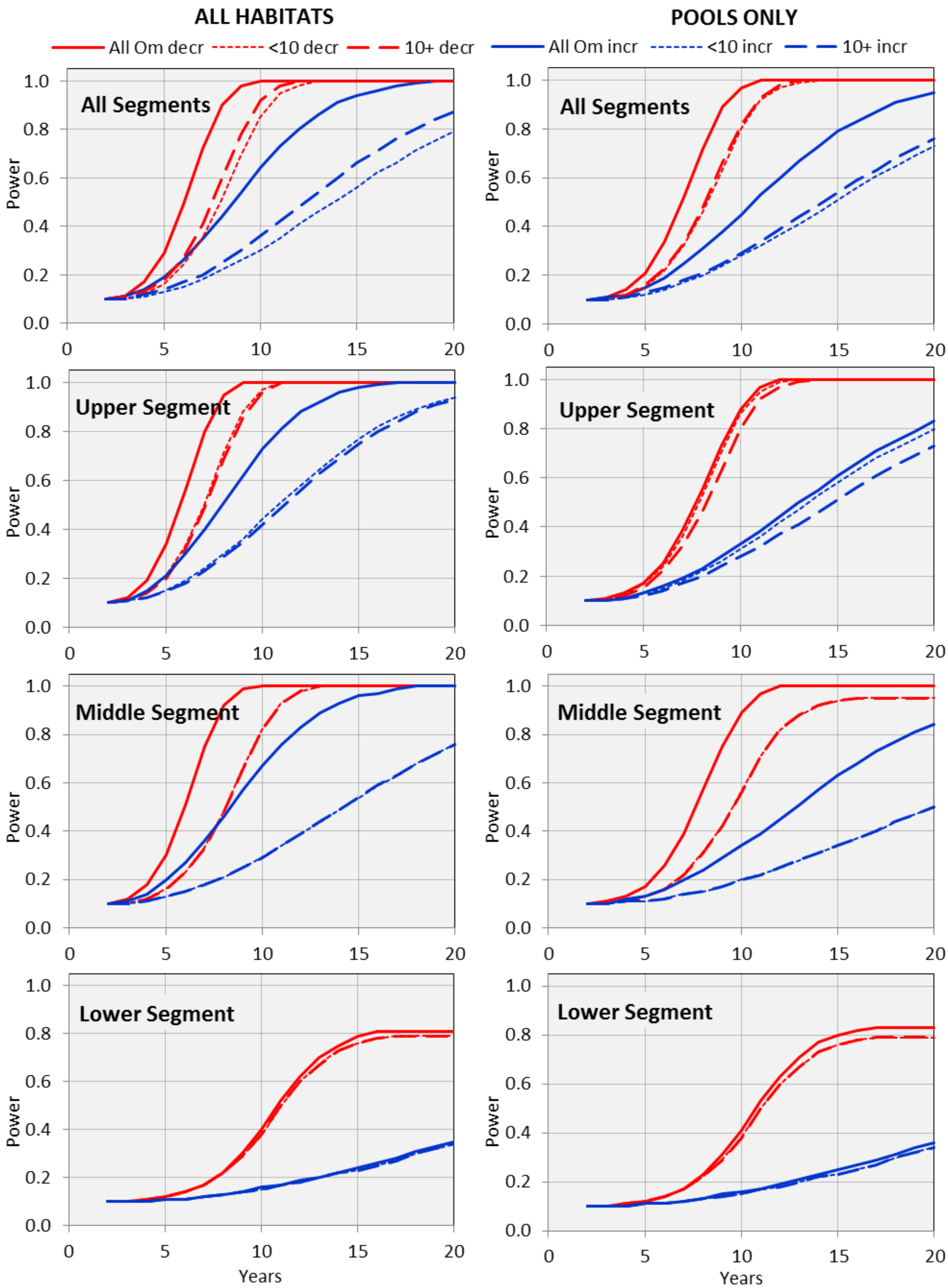


Figure 73. Power curves for detecting a 10% increase (blue lines) or decrease (red lines) in abundance of O. mykiss by size class and basin segment based on sampling all habitats (pools, flatwaters, and riffles) or pools only.

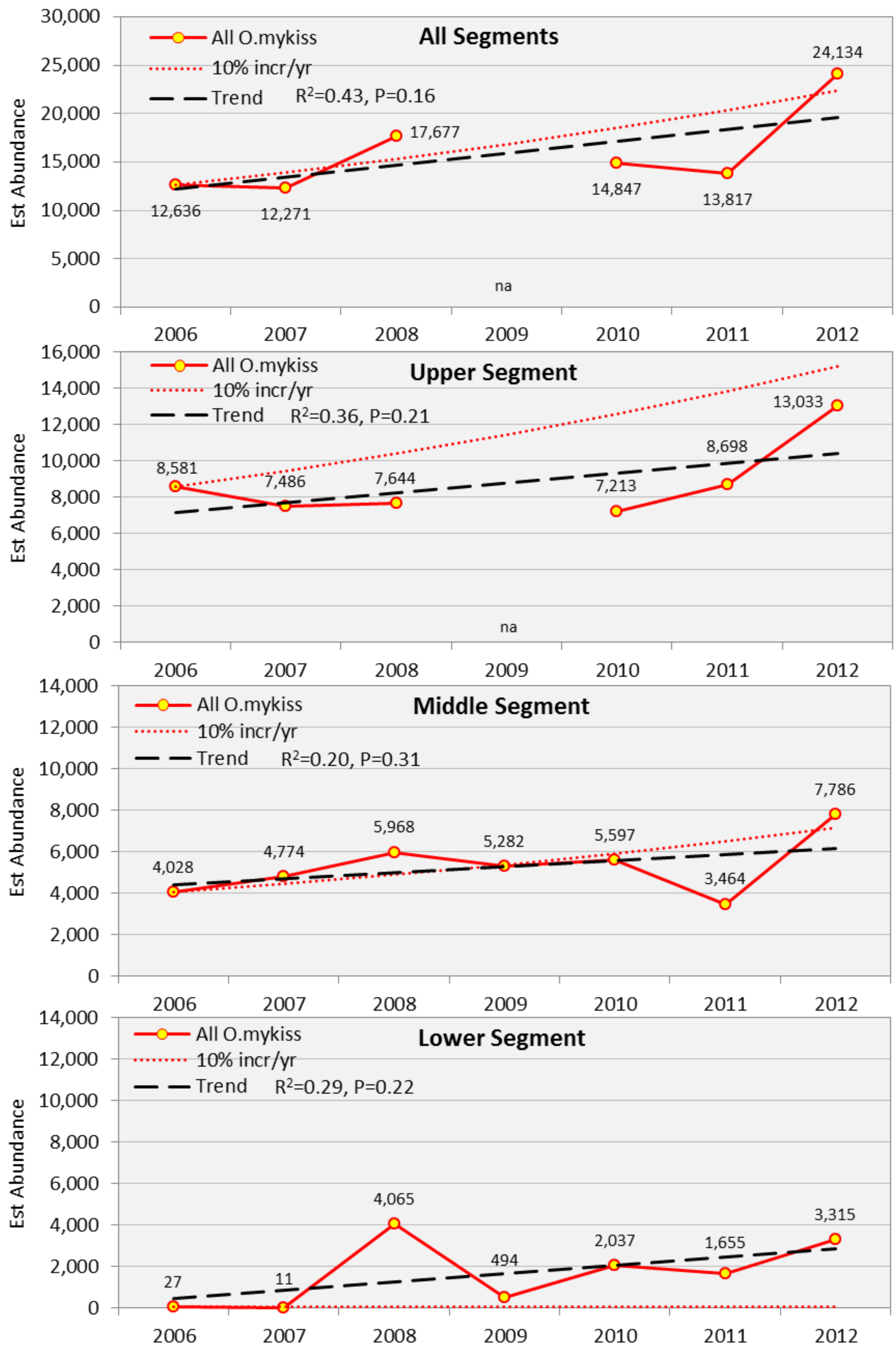


Figure 74. Linear trends in annual abundance of all O. mykiss (sizes combined) in all habitat types (combined), according to segment. Red dotted line shows 10% increase/year from 2006.

abundance per year (based on the initial abundance in 2006), many of the annual estimates in the upper segment and combined segments remained well below the 10% projection (red dotted line, Figure 74). In contrast, the observed abundance in the middle segment was mostly above the 10% line, yet the annual variability (e.g., the low estimate from 2011) rendered the linear trend non-significant. The observed estimates from the lower segment were all well above the 10% projection due to the near-zero initial estimate from 2006, but the high estimate from 2009 contributed to the non-significant linear trend.

Assessing Change in Abundance Between Two Consecutive Years

The program *SSPow2Samples.xls* (Gerow 2007) was used to estimate the number of sampling units necessary to detect a 25% change in abundance of *O. mykiss* (all sizes combined) between any two years, based on Ventura Basin data representing either headwater/tributary study sites (LNFnew, LNFmid, UNF, and Mat 7) or mainstem study sites (Ven 3, Ven 5, Mat 3, Mat 5), and either sampling all habitat types or sampling pools only according to a paired sampling design (see Section 4.5.3 for details). As expected given the pattern of observed variation (SDs proportional to mean abundance), and consistent with the annual trend analysis described above, calculated sample sizes required to detect an increase in abundance at a specified power were greater than sample sizes needed to detect a decrease (Figure 75).

Differences were noted in the power curves comparing all habitat sampling versus pool-only sampling, depending on study location. For study sites located in the headwaters and tributaries, greater power was achieved with fewer sampling units when all habitat types were sampled compared to pools only, whereas in mainstem reaches there was little difference in power whether all units or only pools were sampled (Figure 75). This result was largely due to a lower SD/mean abundance ratio for the all habitats dataset than for the pools only dataset in headwater study sites, whereas both the ratios and the correlations from the mainstem study sites were similar between the two sampling scenarios. Specifically, sample sizes of only 9 to 13 units were estimated to be necessary to detect a decrease or an increase in abundance (respectively) of *O. mykiss* between two consecutive years in headwater/tributary locations when all habitat types were sampled, whereas 26 to 41 units may be required if only pools are sampled (Table 19). In mainstem reaches, 17 to 19 units (whether all types or pools only) may be required to detect a decrease in abundance, whereas 26-30 units may be required to detect an increase.

Assessing Change in Abundance Between Two Reaches

The Ventura Basin data from the same 4 headwater/tributary and 4 mainstem study sites was also used to assess the sample size requirements to detect a 25% difference in abundance of all *O. mykiss* (size classes combined) between two alternative reaches in a given year, again assuming a sampling design encompassing discrete habitat units of all three principal types or alternatively a pools-only design (see Section 4.5.3 for details). This scenario could represent a study to compare a “test” reach with a “control” reach, where a test reach that received habitat restoration work might be expected to show higher abundance in comparison to a control reach. Or, if the test reach was subject to a foreseen impact (e.g., construction activities, logging, channelization, etc.), it might be expected to display lower abundance than a control reach.

This power analysis also utilized the *SSPow2Samples* program (Gerow 2007), but assumed an independent sampling design with no pairing and equal sampling units per reach. The same SD/mean abundance ratios used in the above power analysis were again used in this analysis, but this independent design did not incorporate the correlations used in the 2-year paired sampling

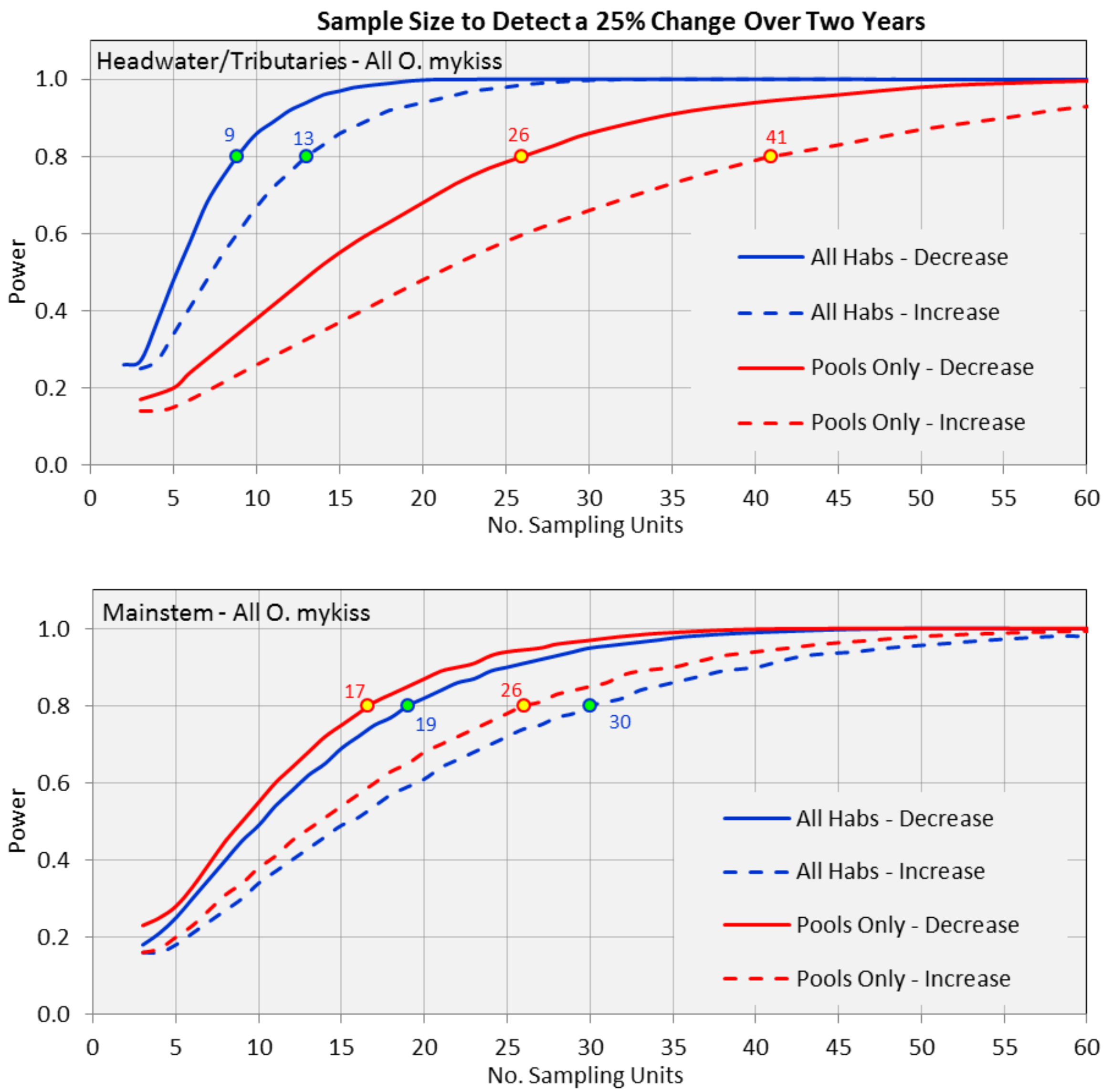


Figure 75. Power curves for detecting a 25% change in abundance (increase or decrease) of O. mykiss (all size classes combined) between consecutive years according to basin location (headwater/tributary or mainstem) and sampling all habitats (pools, flatwaters, and riffles) or pools only, based on a paired sampling design. N's required to achieve 80% power are shown.

design. Consequently, the number of sampling units required to achieve a specified level of power (e.g., 80%) was greater than with the paired design (Figure 76). Also, as expected, a greater number of sampling units may be required to detect if a test reach contained 25% higher abundance than a control reach (e.g., a restoration scenario), whereas fewer sampling units may be required to detect if a test reach contained 25% lower abundance than a control reach (e.g., an impact scenario).

Again, as noted for the above test, the difference in sample sizes required to achieve, say, 80% power was more disparate for the headwater/tributary study sites than for the mainstem study sites. In headwater/tributary sites, 11 to 17 habitat units (all types combined) may be necessary to detect a 25% lower or higher (respectively) abundance between two reaches, whereas if only pools are sampled, 34 (for lower abundance) to 55 pools (for higher abundance) may be required (Figure 76, Table 19). In mainstem study sites, 21-23 units (all habitats or pools only) may be required to detect a 25% lower abundance, versus 34-36 units to detect a higher abundance.

Table 19. Estimated number of sampling units (pools only, pools/flatwaters/riffles, or representative reaches) to detect a 25% difference in abundance of O. mykiss (all size classes combined) with 80% power, according to design, basin location, and direction of change.

Comparative Design	Location	Direction of Change	Pools Only	All Habitats	Rep Reaches
Paired	Headwater/Tribs	Decrease	26	9	12
Yr 1 vs Yr 2		Increase	41	13	18
	Mainstem	Decrease	17	19	24
		Increase	26	30	37
Independent	Headwater/Tribs	Decrease	34	11	15
Rch 1 vs Rch 2		Increase	55	17	24
	Mainstem	Decrease	21	23	25
		Increase	34	36	40

Assessing Changes Using a Representative Reach Design

In an attempt to utilize the habitat-stratified data from this study to assess sample size requirements for a more basic representative reach (RR) sampling approach, *O. mykiss* abundance data was pooled to simulate sampling at a 100 to 200 yd or m spatial scale, typical of RR survey lengths (see Section 4.5.3 for details). This power assessment utilized the *SSPow2Samples* program (Gerow 2007), assuming an independent sampling design with equal numbers of RRs per study area, in order to assess how many RRs may be necessary to detect a 25% difference in abundance between two years (using a paired deign) or between two study areas over a single year (using an independent design).

The estimated number of RRs required to achieve a specified level of power (e.g., 80%) to detect a 25% difference in abundance were again higher when assessing a study area for an increase or a higher level of abundance, in comparison to lower or decreased abundance. Overall, the RRs approach, using the simulated RR data from the Ventura Basin, showed less power than the habitat-stratified approach encompassing all habitat types for both the year to year paired design and the independent two reaches design. An estimated 12-18 RRs (each encompassing multiple habitat units) would be required to detect a 25% decrease or increase (respectively) in abundance over two years, whereas the habitat stratified/all habitats design would require 9 to 13 individual units to achieve the same power (Figure 77, Table 19). However, the representative reach approach did show more power than the pools only design in headwater/tributary study sites (e.g., 12-18 RRs vs 26-41 pools), but less power for the mainstem study sites (e.g., 24-37 RRs vs 17-26 pools).

As noted above, the disparity between power among designs and the advantages of an all habitats stratified design was most apparent in the headwater tributary locations, where most *O. mykiss* are expected to reside in the warmer basins of southern California. Although the method of simulating RRs from the Ventura’s habitat-stratified data (see Section 4.5.3) may have introduced additional uncertainty into this comparative assessment, in actual practice fixed-length RRs will often contain highly variable proportions of each habitat type, whereas the pooled data used in this assessment contained a more consistent proportion of each habitat type in the simulated representative reaches, which would be expected to result in reduced sample size requirements.

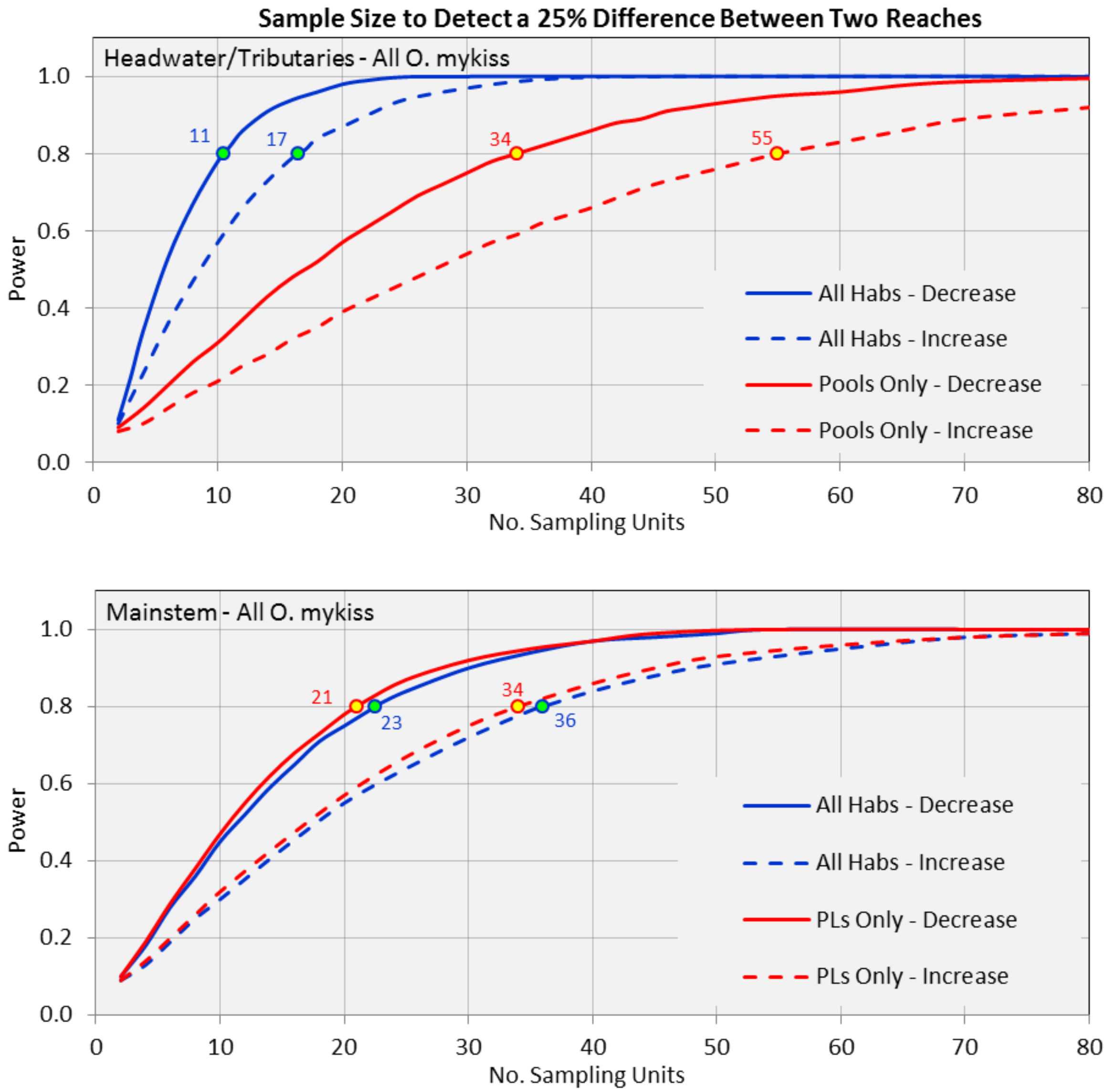


Figure 76. Power curves for detecting a 25% difference in abundance (higher or lower) of O. mykiss (all size classes combined) between two study areas according to basin location (headwater/tributary or mainstem) and sampling all habitats (pools, flatwaters, and riffles) or pools only, based on an independent sampling design. N's required to achieve 80% power are shown.

Comparison of Sampling Designs

Although the power to detect real changes is a critical component of a fish population study, other factors must be considered and may significantly influence the feasibility and cost to detect trends. Although the pools only design generally showed less power to detect trends or differences in abundance for a given sample size, a pools only design possesses several significant advantages over a design that incorporates flatwater and riffle habitats. The primary benefit of a pools only design is the ability to estimate abundance (typically as an index) using dive counting protocols, which are

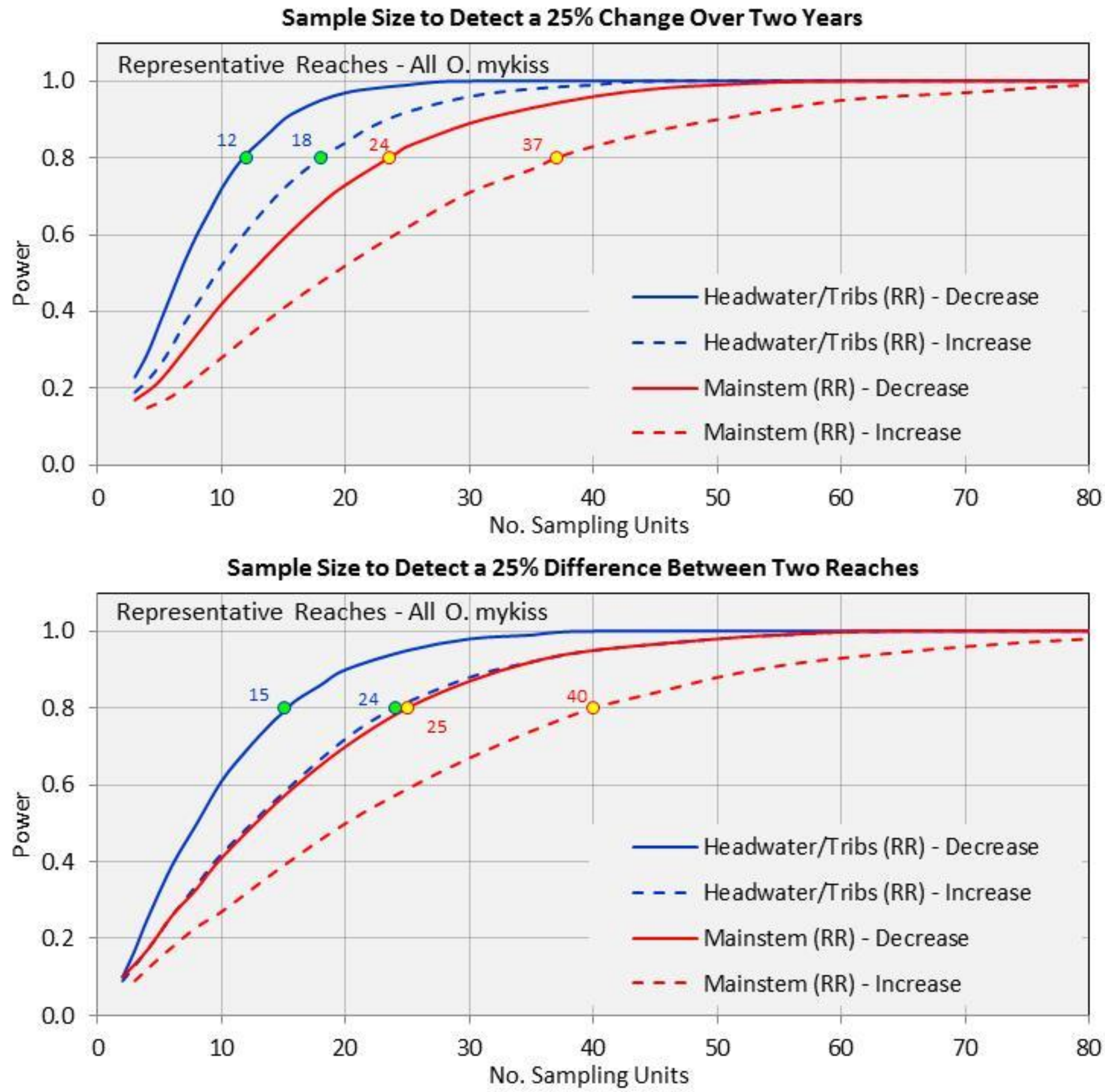


Figure 77. Power curves for detecting a 25% difference in abundance of O. mykiss (all size classes combined) between years (upper) or between two study areas (lower) using a representative reach (RR) sampling approach, according to basin location (headwater/tributary or mainstem). N's required to achieve 80% power are shown.

more rapid and less costly than electrofishing protocols that are generally necessary to provide reliable data in shallower flatwaters and riffles that are encompassed in an all habitats or representative reaches design. A diving only protocol is non-intrusive in nature and consequently is highly desirable when assessing endangered species; this attribute also greatly simplifies the cost and time for permitting. A reduction in necessary personnel and gear is another benefit, particularly when assessing large and remote basins. Overall, the reduced cost of diving allows for increased sample sizes that could lead to increased power over an all habitats or representative reaches design.

Disadvantages of a pools only design includes the lack of data associated with fish capture and handling, such as detailed length, weight, condition, food consumption, and fish tagging or other tag-associated data (e.g., growth rates, movement, etc.). Data from the Ventura Basin also suggests that a pools only design may only account for a small fraction of the existing population, particularly the young-of-year or fry component which occur at much higher densities in riffles. Studies focused on evaluation of spawning success, for example, may be ineffective if only pools are sampled. The specific goals of a study must account for these factors when assessing sampling protocols.

The comparative attributes of a habitat-stratified design involving all habitat types versus a representative reach design are less clear. The differences in densities of *O. mykiss* between habitat types suggests that a habitat stratified design would be most effective in reducing variance estimates, and therefore in assessing trends in abundance. If RRs accurately and consistently encompassed an appropriate proportion of each habitat type (which they often do not), variances could be minimized. Longer representative reaches are more likely to result in consistent habitat proportions than the 100 yd or 100 m reaches that are typically utilized. Although RR designs are regularly employed, sites are often non-randomly selected and used to provide several independent index estimates of abundance rather than expanded estimates encompassing an entire basin or subbasin (which would require a formal sampling design). Accomplishing the latter goal using a RR design would require selection of a significant number of RRs, each of which are longer and more time consuming to sample that are the individual habitat units encompassed in the habitat stratified approach. In contrast, effectively netting-off individual flatwater and riffle units for electrofishing in small headwater tributaries is time consuming and is subject to significant fish displacement and associated bias (Peterson et al. 2005). In sum, the RR approach applied in a randomized design may be an effective protocol in smaller headwater streams, whereas a habitat stratified design may be more appropriate in larger tributary and mainstem habitats.

6.0 Conclusions

Stratified random sampling was conducted in 4 to 14 study sites encompassing 46 to 308 sampling units distributed throughout the Ventura River Basin over a seven year period (2006-2012). Abundance of *O. mykiss* was estimated using dive counts and electrofishing under the Method of Bounded Counts protocols in pools, flatwaters, and riffles in most years to produce overall abundance estimates at study site and basin segment spatial scales according to fish size class (fry at <10cm and juvenile+ at $\geq$10cm). Abundance estimates displayed significant spatial and temporal variation, with consistently highest abundance and densities (#/100 ft^2) in the upper segment above Matilija Dam (resident rainbow trout only) and in the middle segment between Robles Diversion Dam and Matilija Dam (mixture of resident and anadromous *O. mykiss*).

Maximum estimated densities of 3-7 *O. mykiss* fry/100 ft^2 and 1-2 juvenile+/100 ft^2, were routinely observed in the upper North Fork and lower North Fork Matilija Creek study sites, with zero or near zero densities in most of the lowermost Ventura River study sites. An exception to the lower mainstem sites was Ven 3 through Casitas Springs, a reach characterized by cool upwelling groundwater and inflow from San Antonio Creek, and important spawning tributary. *O. mykiss* abundance fluctuated dramatically in this reach from near zero in 2006 and 2007 to approximately 1,000 fish in 2008, 2010, and 2012. Although not encompassed in this study, extremely dry conditions occurred in 2013 and 2014 that may have “re-set” abundance of O. mykiss to the low levels observed in the first two years of study (Paul Jenkin, personal communication).

In almost every year and every reach studied, densities of *O. mykiss* fry were consistently highest in riffle habitats and lowest in pool habitats by factors of 2-5 times, whereas juvenile+ *O. mykiss* were more evenly distributed among habitat types. Larger (>20cm) and presumably residualized *O. mykiss* occurred at highest densities in pool habitats.

Annual variation was also substantial, with positive (but statistically non-significant) trends in abundance of *O. mykiss* in all three segments. Maximum abundance occurred in 2012, with 2,137 captured or observed *O. mykiss* producing a total estimated abundance of 24,134 fish in the Ventura River Basin (excluding San Antonio Creek and several minor tributaries). Total estimated abundance was less than 15,000 fish in most other years, with a minimum estimate of 12,271 fish in 2007. The high annual variability in abundance of *O. mykiss* fry was reflected in C.V.'s exceeding 170% in the lower Ventura study sites and in the lower Matilija Creek study site, with C.V.'s over 100% for juvenile+ fish in mainstem Ventura River and San Antonio Creek study sites. In comparison, variation in annual abundance in most tributary and headwater study sites was less with C.V.s for both size classes typically between 30% and 70%. Further assessment of the annual abundance data suggested that a minimum of 7-10 years would be necessary to statistically detect an annual decrease in abundance of 10% per year in the headwater and tributary study sites, whereas 15-20 years of sampling may be required to detect a comparable decline in the lower mainstem reaches. Longer time series would be required to detect declines in abundance using a pool only or a representative reach sampling design (compared to the habitat stratified design used here), or to detect a 10% annual increase in abundance.

Habitat data was collected in 2006, 2007, 2011, and 2012 to evaluate the relationship between study site HSI scores produced by an existing USFWS HSI model (Raleigh et al. 1984) and observed densities of *O. mykiss*. Linear regression showed statistically significant fits, with best fit for the non-compensatory model option and worst fit using the equal components option. Despite the positive relationship, poor separation between study sites supporting low densities of *O. mykiss* with sites that were consistently absent of *O. mykiss* along with a relatively narrow range of calculated HSI scores led to the development of an alternative habitat model, termed the Southern Steelhead HSI model (or, SS HSI). New habitat variables and new model formulations produced a model with generally better fit and a wider range in calculated HSI scores. Although overall fit was improved and the model appeared appropriate to the Ventura River Basin, the model has not been validated elsewhere and several of the HSI variables were highly qualitative in nature and should be assessed with actual data prior to application in future studies.

The variability in juvenile abundance described above, and its effects on steelhead survey design, sampling periodicity, and habitat requirements, illustrates the difficulties associated with assessing trends in distribution, abundance, and habitat in southern California basins. Some assessments of historical rainfall trends suggest that the Ventura Basin may currently be experiencing a long-term dry spell, which combined with the expected effects of climate change and increased population growth, with its associated water demands, is likely to produce yet greater stress on limited water supplies and more challenges for native cool-water species such as the endangered Southern California steelhead.

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Appendix A

GPS Coordinates at Study Site Boundaries

Study Site	Deg N	Min N	Deg W	Min W
Ven1 Top	34	18.989	-119	17.762
Ven1 Btm	34	18.233	-119	18.155
Ven2 Top	34	20.000	-119	17.817
Ven2 Btm	34	19.150	-119	17.717
Ven3 Top	34	22.901	-119	18.539
Ven3 Btm	34	22.141	-119	18.555
SACmid Top	34	24.979	-119	16.251
SACmid Btm	34	24.734	-119	16.439
SACup Top	34	26.106	-119	14.777
SACup Btm	34	25.889	-119	15.194
Ven4 Btm	34	27.035	-119	17.609
Ven4 Top	34	27.531	-119	17.497
Ven5 Top	34	29.117	-119	17.999
Ven5 Btm	34	28.833	-119	17.600
LNFnew Top	34	29.606	-119	18.327
LNFnew Btm	34	29.327	-119	18.341
LNFmid Top	34	30.359	-119	16.989
LNFmid Btm	34	30.499	-119	17.197
Mat3 (low) Top	34	29.696	-119	19.976
Mat3(low) Btm	34	29.624	-119	19.713
Mat3 (up) Top	34	30.021	-119	20.966
Mat3 (up) Btm	34	30.097	-119	20.796
Mat5 Top	34	30.336	-119	22.767
Mat5 Btm	34	30.197	-119	22.335
Mur3 Top	34	29.915	-119	23.779
Mur3 Btm	34	30.062	-119	23.435
Mat7b Top	34	32.213	-119	24.239
Mat7b Btm	34	31.854	-119	24.053
UNF Top	34	30.919	-119	22.444
UNF Btm	34	31.083	-119	22.723

Appendix B

2011 Habitat Mapping

(Murietta Creek mapped in 2012)

Selected units are boxed with bold font

Study Site	Unit #	Habitat Level II	Type Level III	Length ft	Width ft	Waypoint #	Comments
Ven 1	1	FW	GLD	183			map 6/21/11, Q~46cfs
Ven 1	2	FW	**RUN**	127	25.0	78	LB brushy
Ven 1	3	PL	**MCP**	48	27.7	77	LB brushy
Ven 1	4	FW	RUN	50			deep/fast
Ven 1	5	NS	HGR	22			
Ven 1	6	RF	LGR	95			top at 6' bldr by RB
Ven 1	7	RF	**LGR**	115	24.0	76	
Ven 1	8	FW	RUN	74			old FW#46
Ven 1	9	RF	**HGR**	70	19.0	75	split 20' abv btm, up RC
Ven 1	10	NS	LGR	17			2nd sml split
Ven 1	11	FW	**RUN**	82	19.0	74	access to levee
Ven 1	12	RF	LGR	74			w RN
Ven 1	13	RF	LGR	152			opp upper levee gate
Ven 1	14	FW	**RUN**	94	19.0	73	split ends at top
Ven 1	15	FW	GLD	177			
Ven 1	16	FW	**RUN**	78	25.7	72	
Ven 1	17	RF	LGR	61			
Ven 1	18	FW	**RUN**	142	15.7	71	w sml RF break
Ven 1	19	RF	LGR	165			w GLD,drain w access to bike path
Ven 1	20	PL	**LSBK**	157	28.0	70	bank formed(no bedrk), btm RN/GLD
Ven 1	21	FW	**RUN**	90	17.0	69	
Ven 1	22	RF	**LGR**	126	16.3	68	onto big open bar LB, LWD
Ven 1	23	NS	HGR	64			LWD
Ven 1	24	PL	**MCP**	491	66.6	67	very wide, oil rig behind LB
Ven 1	25	RF	LGR	88			wide
Ven 1	26	FW	RUN	154			
Ven 1	27	FW	GLD	145			top 25' PL
Ven 1	28	FW	RUN	88			top at concrete w rebar
Ven 1	29	NS	RUN	44			
Ven 1	30	NS	LGR	23			short break
Ven 1	31	PL	**LSL**	126	25.0	66	brushy, hiQ chan RB w bedrk PL
Ven 1	32	RF	**LGR**	113	13.3	65	narrow/fast
Ven 1	33	FW	RUN	49			short RF break at top
Ven 1	34	PL	**MCP**	298	29.0	64	hiQ chan at btm RB, rip-rap RB
Ven 1	35	FW	GLD	81			btm at RB concrete slab
Ven 1	36	PL	**MCP**	191	52.0	62	OVH oil pipeline
Ven 1	37	RF	**LGR**	50	22.0	61	
Ven 1	38	FW	**RUN**	186	24.0	60	concrete w metal pipe top RB
Ven 1	39	RF	**LGR**	20	16.3	59	lrg concrete slab RB
Ven 1	40	FW	**RUN**	58	18.0	58	
Ven 1	41	RF	**LGR**	52	23.7	57	
Ven 1	42	FW	RUN	30			short flatwater
Ven 1	43	RF	HGR	94			
Ven 1	44	RF	**LGR**	124	32.0	56	OVH pipes at top
Ven 1	45	FW	RUN	87			
Ven 1	46	PL	**MCP**	111	28.0	55	finish ~1230
Ven 1	47	RF	LGR	117			split nr top, top 40' blw bridge
Ven 2	1	FW	**RUN**	160	30.8	102	map 6/22/11, Q~35cfs, OVH lines
Ven 2	2	RF	**LGR**	104	25.6	101	~RN
Ven 2	3	FW	RUN	126			
Ven 2	4	PL	**LSBK**	67	21.4	100	cables along cliff
Ven 2	5	RF	**LGR**	83	19.5	99	fresh slide LB top
Ven 2	6	PL	**LSBK**	56	24.7	98	short/fast
Ven 2	7	RF	**HGR**	41	35.7	97	scalloped bedrock (mudrock)
Ven 2	8	FW	GLD	121			deep
Ven 2	9	PL	**LSBK**	165	27.0	96	
Ven 2	10	RF	HGR	68			top at hole in bedrock
Ven 2	11	RF	LGR	117			widens at top, w HGR
Ven 2	12	RF	LGR	122			w RN
Ven 2	13	FW	RUN	76			w RF

Study Site	Unit #	Habitat Level II	Type Level III	Length ft	Width ft	Waypoint #	Comments
Ven 2	14	RF	LGR	104	16.5	95	
Ven 2	15	PL	LSBK	140	37.9	94	thick veg plug at top,OVH lines
Ven 2	16	FW	RUN	170	23.8	93	btm 30' ~PL, too deep to net?
Ven 2	17	RF	LGR	104			w RN & veg clump in middle
Ven 2	18	RF	LGR	126			TRV top
Ven 2	19	FW	GLD	120	52.8	92	top cattail clump RB, **MID TRAIL**
Ven 2	20	FW	GLD	138	63.9	91	
Ven 2	21	FW	GLD	176	36.5	90	top at MC concrete w willow, beware
Ven 2	22	FW	RUN	193	26.9	89	top at split w TRV RF_ IW metal cable
Ven 2	23	RF	LGR	139			up left chan, btm HGR
Ven 2	24	FW	RUN	89			top 10' abv LB cattail clump
Ven 2	25	PL	LSBK	99	25.0	88	top at LB bedrock w grass patch
Ven 2	26	FW	RUN	90			deep/fast
Ven 2	27	RF	LGR	88	19.0	87	HGR btm, RN top
Ven 2	28	FW	RUN	95			w RF
Ven 2	29	FW	GLD	142	20.2	86	open gravel bar top RB,RF top, lean-to
Ven 2	30	FW	GLD	109			gravel,top at sml bedrock outcrop LB
Ven 2	31	FW	GLD	157			top at 14" log RB
Ven 2	32	FW	GLD	148	52.8	85	top at 4' bldr RB
Ven 2	33	FW	RUN	217			bedrk LB, OVH lines
Ven 2	34	RF	LGR	154			top RB RN, top at TRV crest
Ven 2	35	RF	LGR	136	30.8	84	sml split, rt chan HGR
Ven 2	36	FW	GLD	89			top at 6' oblong bldr LB, lrg log RB
Ven 2	37	PL	MCP	209	46.7	83	top at lrg metal object LB
Ven 2	38	RF	LGR	87	27.3	82	
Ven 2	39	FW	RUN	95			**UPPER TRAIL**
Ven 2	40	PL	LSBK	322	33.8	81	btm deep GLD,top RN, temp logger
Ven 2	41	RF	LGR	79	15.5	80	deep, upper half RN
Ven 2	42	NS	CAS	13			
Ven 2	43	RF	LGR	172			top at bank-bank bedrock sill
Ven 3	1	FW	GLD	100	37.7	132	map 6/23/11, Q~36cfs
Ven 3	2	FW	RUN	128			
Ven 3	3	RF	LGR	83	30.7	131	
Ven 3	4	PL	LSBK	166	34.0	130	scour on dirt bank, not bedrock,~RN
Ven 3	5	RF	LGR	167	44.0	129	TRV btm, w RN along LB arundo
Ven 3	6	FW	RUN	108			RF at low Q?
Ven 3	7	FW	GLD	174	48.7	128	shallow btm
Ven 3	8	RF	LGR	94	46.3	127	
Ven 3	9	PL	LSR	132	29.0	125	scour under LB tree roots
Ven 3	10	FW	RUN	51			btm 20' blw LB arundo clump
Ven 3	11	RF	HGR	111			braided
Ven 3	12	PL	LSBO	976	76.2	124	move to btm long PL
Ven 3	13	NS	HGR	20			manipulated
Ven 3	14	FW	RUN	54	23.0	123	up rt chan
Ven 3	15	RF	LGR	80			top at split
Ven 3	16	RF	LGR	79	19.7	122	rt chan, top at MC rock
Ven 3	17	RF	LGR	128			~RN
Ven 3	18	FW	RUN	65			
Ven 3	19	PL	LSL	67	22.7	121	~fast RN, LB backwater
Ven 3	20	NS	LGR	25			short break
Ven 3	21	FW	GLD	72			along levee, OVH lines
Ven 3	22	FW	RUN	81			LB shallow
Ven 3	23	FW	RUN	91			LB shallow ~RF
Ven 3	24	FW	RUN	118	30.3	119	w deep pockets along riprap
Ven 3	25	RF	LGR	126	30.3	117	mid 1/2 RN
Ven 3	26	NS	LSL	58			downed tree in middle
Ven 3	27	NS	HGR	11			short break
Ven 3	28	FW	RUN	75	19.3	116	~RF, tree xing at top
Ven 3	29	NS	LGR	66			deep, fast, brushy
Ven 3	30	PL	LSL	187	48.0	115	temp logger pool, split chan/BWP

Study Site	Unit #	Habitat Level II	Type Level III	Length ft	Width ft	Waypoint #	Comments
Ven 3	31	RF	LGR	40			
Ven 3	32	FW	**RUN**	88	23.0	114	lower 1/2 PL
Ven 3	33	PL	**CCP**	164	54.0	113	btm a swim dam
Ven 3	34	RF	**LGR**	36	30.0	112	top at swim dam
Ven 3	35	FW	**RUN**	75	29.3	111	
Ven 3	36	FW	**RUN**	79	31.0	110	
Ven 3	37	RF	LGR	85			w RN, top start of TRV RF
Ven 3	38	RF	**LGR**	78	51.0	109	rt half RN, top at tree/arundo
Ven 3	39	NS	LGR	42			
Ven 3	40	RF	**LGR**	66	28.3	108	btm deeper RN,top sml MC willow
Ven 3	41	PL	**LSBO**	155	29.7	106	thick arundo LB
Ven 3	42	NS	GLD	17			short break between PLs
Ven 3	43	PL	**LSL**	112	31.0	105	tree/bank formed, big trout in '10
Ven 4	1	PL	**MCP**	100	38	41	map 6/20/11, Q~?cfs, 71°F
Ven 4	2	FW	**RUN**	30	23	40	
Ven 4	3	PL	**LSBO**	21	32	39	8' bldr LB
Ven 4	4	FW	RUN	40			lower 1/3 ~RF
Ven 4	5	RF	**LGR**	77	31	38	top wide w LB HGR,TRV top
Ven 4	6	FW	GLD	94			top sml break by 12' bldr LB
Ven 4	7	FW	GLD	103			top 6' triang bldr MC
Ven 4	8	FW	RUN	73			lower half GLD
Ven 4	9	RF	**LGR**	98	36	37	some RN LB
Ven 4	10	FW	RUN	42			~LGR
Ven 4	11	RF	**LGR**	41	34	36	top 8' hanging bldr RB, w RN
Ven 4	12	FW	RUN	46			narrow, ~RF
Ven 4	13	RF	**HGR**	43	28	35	
Ven 4	14	RF	**LGR**	74	36	34	btm 25' pocket PL
Ven 4	15	FW	RUN	78			wide, left half GLD
Ven 4	16	FW	GLD	61			
Ven 4	17	PL	**LSBK**	169	62	33	big swimming hole
Ven 4	17b	FW	RUN	5			
Ven 4	18	PL	**TRP**	71	26	32	~RN/trench pool, fast top
Ven 4	19	RF	HGR	14			
Ven 4	20	RF	LGR	40			wide
Ven 4	21	FW	**GLD**	107	53	31	top 20' blw swim dam
Ven 4	22	PL	**LSBK**	137	50	29	2nd swimming hole, split at top
Ven 4	23	RF	HGR	48			up LC
Ven 4	24	FW	RUN	27			2nd split, both channels RN
Ven 4	25	RF	**HGR**	28	37	28	
Ven 4	26	RF	**LGR**	49	33	27	
Ven 4	27	FW	RUN	32			
Ven 4	28	RF	LGR	23			SC enters top RB
Ven 4	29	FW	**SRN**	101	20	26	top 1/2 narrow/deep, Q enters RB
Ven 4	30	RF	HGR	63			narrow, brushy
Ven 4	31	RF	LGR	42			
Ven 4	32	PL	**MCP**	103	50	25	
Ven 4	33	FW	**RUN**	85	40	24	
Ven 4	34	FW	RUN	79			main split chan leaves at top
Ven 4	35	FW	**GLD**	119	59	23	very wide
Ven 4	36	PL	**LSBO**	37	66	22	short pocket pool
Ven 4	37	RF	**LGR**	51	42	20	w POW at top
Ven 4	38	FW	RUN	49			
Ven 4	39	PL	**MCP**	36	28	18	4' bldr MC top
Ven 4	40	RF	LGR	67			left half RN
Ven 4	41	FW	**RUN**	84	57	17	left 1/2 GLD
Ven 4	42	FW	**RUN**	63	53	16	
Ven 4	43	RF	LGR	70			w RN
Ven 4	44	FW	**RUN**	69	42	15	OVH lines, finish 1415
Ven 5	1	FW	RUN	17			map 6/24/11, Q~25cfs
Ven 5	2	NS	CAS	30			braided

Study Site	Unit #	Habitat Level II	Type Level III	Length ft	Width ft	Waypoint #	Comments
Ven 5	3	FW	**RUN**	46	16.1	161	braided
Ven 5	4	PL	PLP	23			braided
Ven 5	5	NS	CAS	4			part sw immers dam
Ven 5	6	PL	**MCP**	129	47.4	160	sw imming hole
Ven 5	7	FW	RUN	56			
Ven 5	8	PL	**MCP**	70	27.8	159	
Ven 5	9	RF	**LGR**	84	18.6	158	w RN
Ven 5	10	FW	**POW**	40	30.3	157	top at 8' bldr RB
Ven 5	11	PL	LSR	32			scour under RB trees
Ven 5	12	FW	**RUN**	27	31.3	156	
Ven 5	13	PL	**LSR**	33	15.5	155	undercut RB, 8' bldr at top
Ven 5	14	FW	RUN	25			
Ven 5	15	PL	MCP	59			upper 25' LSR w temp logger
Ven 5	16	FW	**RUN**	75	26.1	154	w pockets, old unit #15
Ven 5	17	PL	**LSBO**	30	25.2	153	
Ven 5	18	FW	RUN	39			
Ven 5	19	RF	**HGR**	74	12.5	152	LGR at top
Ven 5	20	FW	RUN	70			w ide, LB half GLD
Ven 5	21	FW	GLD	110			btm 25' blw 1st clump, top TRV RF
Ven 5	22	RF	**LGR**	34	46.7	151	~1/2 RN
Ven 5	23	FW	RUN	58			
Ven 5	24	PL	**LSR**	29	34.0	150	top at concrete plunge
Ven 5	25	NS	CAS	6			
Ven 5	26	PL	**LSBK**	48	26.5	149	along concrete apron
Ven 5	27	NS	CULV	42			Camino Cielo bridge
Ven 5	28	NS	RUN	25			affected by bridge
Ven 5	29	NS	CAS	5			
Ven 5	30	FW	**RUN**	49	31.5	148	w pockets, w ide
Ven 5	31	FW	RUN	58			narrow er
Ven 5	32	RF	LGR	103			mid RN, old unit 30
Ven 5	33	FW	RUN	90			btm 6' bldr nr RB
Ven 5	34	RF	**HGR**	57	20.3	146	
Ven 5	35	RF	**LGR**	100	22.8	144	
Ven 5	36	PL	MCP	95			~POW
Ven 5	37	FW	POW	50			
Ven 5	38	FW	**RUN**	57	33.9	142	~ deep GLD
Ven 5	39	PL	MCP	62			top 20' blw bridge
Ven 5	40	FW	RUN	40			OVH foot bridge
Ven 5	41	NS	CAS	20			
Ven 5	42	PL	LSBO	38			12' bldr LB, top at 3' MC bldr
Ven 5	43	FW	**RUN**	32	12.2	140	
Ven 5	44	NS	CAS	7			
Ven 5	45	FW	RUN	44			
Ven 5	46	RF	**HGR**	61	27.4	139	
Ven 5	47	FW	**POW**	55	36.0	138	
Ven 5	48	RF	**LGR**	50	32.1	137	very w ide. ~POW
Ven 5	49	FW	POW	67			
Ven 5	50	PL	**LSBO**	40	26.9	135	12' bldr at top, old unit #50
Ven 5	51	FW	POW	45			
Ven 5	52	PL	**LSBO**	55	33.0	134	15' bldr top RB, pipe up RB
Ven 5	53	FW	RUN	31			
Ven 5	54	RF	**LGR**	47	20.0	133	~RN, top at plunge, pipe up RB
Ven 5	55	FW	RUN	44			top LNF confl, LNF 70oF, Ven 6 72oF
SACmid	1	FW	RUN	57			map 7/20/11, btm Rt chan
SACmid	2	RF	LGR	82			total Q~10cfs (RC/LC Q ~50:50)
SACmid	3	RF	**LGR**	59	10.7	170	w HGR
SACmid	4	FW	**GLD**	42	22.3	171	~PL below MC bldr
SACmid	5	FW	**RUN**	63	10.0	172	
SACmid	6	PL	**MCP**	49	13.0	173	old "PL1R"
SACmid	7	FW	RUN	14			short break

Study Site	Unit #	Habitat Level II	Type Level III	Length ft	Width ft	Waypoint #	Comments
SACmid	8	PL	LSBO	29			old "PL2R"
SACmid	9	RF	LGR	68			some RUN
SACmid	10	FW	RUN	27			upper part RF
SACmid	11	PL	LSR	16			old "PL3R", bldr scour at top
SACmid	12	NS	CAS	7			
SACmid	13	PL	**LSBO**	38	12.0	174	old "PL4R", ~RN w pocket Pl at top
SACmid	14	FW	RUN	14		175	
SACmid	15	**PL**	**LSBO**	18	7.7		old "PL5R", ~RN
SACmid	16	FW	SRN	82			brushy, green garden hose
SACmid	17	RF	**LGR**	43	7.3	176	lower RN, upper HGR
SACmid	18						skip over to main channel
SACmid	19	FW	**GLD**	73	29.7	177	start mainchannel, top at 2' MC bldr
SACmid	20	PL	**MCP**	147	32.0	178	old "PL7", btm still ~GLD
SACmid	21	FW	RUN	51			
SACmid	22	RF	**LGR**	29	31.7	179	
SACmid	23	PL	**LSBO**	29	13.0	180	~RN but w scour
SACmid	24	RF	HGR	13			
SACmid	25	PL	**MCP**	70	23.0	181	brushy hole LB, OVH pipe
SACmid	26	FW	RUN	95			deep GLD, top fast
SACmid	27	RF	**HGR**	40	14.7	182	top half LGR
SACmid	28	FW	**RUN**	132	21.7	183	~GLD
SACmid	29	RF	LGR	101			wide, top at concrete & old rd xing
SACmid	30	FW	GLD	120			top lrg willow LB, just below parking
SACmid	31	RF	**LGR**	94	37.0	184	w RN/GLD in middle
SACmid	32	FW	**GLD**	111	34.0	185	top at 6' bldr LB, hi Q chan top RB
SACmid	33	PL	**MCP**	65	20.7	186	narrows and deepens, ~GLD/PL
SACmid	34	FW	RUN	39			short RF break at top
SACmid	35	FW	**GLD**	55	17.0	187	top ~PL
SACmid	36	RF	**LGR**	61	14.0	188	
SACmid	37	PL	LSL	42			top just below OVH lines
SACmid	38	FW	RUN	45			start btm Lft chan, w RF
SACmid	39	FW	**GLD**	39	17.7	169	
SACmid	40	RF	LGR	128			w RN
SACmid	41	FW	GLD	77			
SACmid	42	FW	RUN	34			sml pocket pool at top
SACmid	43	FW	GLD	42			
SACmid	44	RF	**LGR**	37	16.3	168	w RN
SACmid	45	FW	RUN	32			
SACmid	46	FW	**POW**	66	15.0	167	
SACmid	47	RF	LGR	28			~shallow POW
SACmid	48	FW	POW	29			
SACmid	49	PL	**MCP**	41	14.3	166	old "PL 6"
SACmid	50	RF	**HGR**	74	12.0	165	
SACmid	51	RF	LGR	59			top at btm main channel, end at 11:30
SACup	1	FW	**RUN**	34	9.8	421	map 7/18/11, Q~7cfs, under bridge
SACup	2	NS	LGR	34			brushy
SACup	3	PL	**LSBO**	42	14.1	422	concrete slab UCB
SACup	4	RF	LGR	34			
SACup	5	FW	**GLD**	51	11.8	420	top at 2' daim willow RB
SACup	6	RF	**LGR**	83	7.6	419	w RN, 40' error on GPS
SACup	7	NS	LGR	29			arundo
SACup	8	PL	**MCP**	59	13.2	418	btm GLD, top RN
SACup	9	RF	LGR	36			
SACup	10	FW	RUN	85			shallow, top arundo cave RB
SACup	11	RF	LGR	54			redd flag nr top LB
SACup	12	FW	**GLD**	66	10.8	417	top MC bldr
SACup	13	FW	RUN	41			5' RF break at top
SACup	14	PL	**MCP**	58	10.3	416	rusty iron pipe nr top LB
SACup	15	RF	**LGR**	29	7.1	415	
SACup	16	PL	**MCP**	19	7.1	414	PL under OVH willow

Study Site	Unit #	Habitat Level II	Type Level III	Length ft	Width ft	Waypoint #	Comments
SACup	17	NS	RUN	50			arundo
SACup	18	NS	LGR	38			arundo
SACup	19	NS	LGR	62			arundo
SACup	20	RF	LGR	32			
SACup	21	NS	RUN	42			arundo
SACup	22	FW	RUN	48			~RF, top at sw im dam
SACup	23	PL	**LSBK**	56	13.1	413	concrete/boulder apron LB by road
SACup	24	RF	**LGR**	25	10.1	412	
SACup	25	FW	RUN	25			
SACup	26	RF	LGR	58			HGR at btm w stump
SACup	27	FW	**RUN**	43	11.0	411	low er half GLD, upper ~PL
SACup	28	RF	**LGR**	58	10.8	410	
SACup	29	FW	GLD	46			access LB, old campfire
SACup	30	NS	GLD	45			arundo
SACup	31	RF	LGR	54			arundo but doable?
SACup	32	NS	GLD	41			arundo
SACup	33	PL	**LSR**	48	15.3	409	maybe CCP?
SACup	34	RF	**LGR**	113	14.0	408	
SACup	35	FW	**RUN**	80	8.6	407	~RF, GPS error 65'
SACup	36	RF	LGR	44			almost HGR
SACup	37	RF	LGR	63			dirt cliff RB, ~GLD, gravel
SACup	38	FW	**GLD**	115	15.6	406	upper 30' RN
SACup	39	RF	**LGR**	75	11.7	405	arundo LB, GPS error 60'
SACup	40	FW	RUN	36			short flat in RF
SACup	41	RF	**LGR**	105	10.3	404	aggregate bldr mid RB
SACup	42	FW	**GLD**	54	16.0	403	top at w hite pipe
SACup	43	PL	**MCP**	43	13.5	402	~GLD
SACup	44	RF	LGR	96			top at MV aggregate bldr
SACup	45	FW	**RUN**	82	6.7	401	lrg sycamore/root w ad RB, ~RF
SACup	46	RF	**LGR**	99	12.3	400	top at rd xing, end at 11:45
LNFnew	1	PL	DPL	33			map 7/13/11,Q~4cfs, under bridge
LNFnew	2	NS	CAS	1			
LNFnew	3	PL	PLP	23			2 PLs separated by concrete abut
LNFnew	4	NS	CAS	1			
LNFnew	5	PL	PLP	40			
LNFnew	6	NS	CAS	1			
LNFnew	7	FW	RUN	20			
LNFnew	8	RF	HGR	7			
LNFnew	9	PL	MCP	33			
LNFnew	10	NS	HGR	14			
LNFnew	11	PL	LSBK	55			
LNFnew	12	NS	CAS	4			
LNFnew	13	**FW**	**POW**	17	23.0	4	
LNFnew	14	FW	GLD	14			
LNFnew	15	FW	RUN	12		5	LB ~PL
LNFnew	16	**RF**	**HGR**	13	15.4		
LNFnew	17	PL	**LSBK**	68	20.9	6	dammed at btm, RB split starts-map LC
LNFnew	18	NS	CAS	1			
LNFnew	19	PL	STP	44			2 PLs w 2ft CAS in middle
LNFnew	20	NS	CAS	11			
LNFnew	21	PL	**STP**	54	7.4	7	5 short bedrk DPLs
LNFnew	22	FW	RUN	26			
LNFnew	23	NS	CAS	1			
LNFnew	24	PL	**STP**	28	15.2	8	
LNFnew	25	FW	SRN	52			
LNFnew	26	PL	STP	41			2 PLs
LNFnew	27	FW	**SRN**	46	19.4	9	end RB split
LNFnew	28	PL	MCP	44			
LNFnew	29	FW	RUN	24			narrow
LNFnew	30	NS	CAS	1			

Study Site	Unit #	Habitat Level II	Type Level III	Length ft	Width ft	Waypoint #	Comments
LNFnew	31	FW	SRN	53	9.2	10	
LNFnew	32	FW	GLD	24			
LNFnew	33	RF	HGR	59			
LNFnew	34	FW	RUN	47			
LNFnew	35	RF	LGR	26	10.3	11	~RN in middle
LNFnew	36	PL	LSBK	44			short RF in mid
LNFnew	37	NS	CAS	1			
LNFnew	38	PL	PLP	24	15.1	12	
LNFnew	39	NS	CAS	3			
LNFnew	40	PL	DPL	42	20.8	13	bldr/LWD dam at btm
LNFnew	41	NS	CAS	2			
LNFnew	42	FW	GLD	36			
LNFnew	43	FW	SRN	46			
LNFnew	44	RF	LGR	37	15.5	14	
LNFnew	45	FW	RUN	47			
LNFnew	46	PL	LSBO	34			
LNFnew	47	RF	HGR	37	9.3	19	
LNFnew	48	FW	RUN	30			
LNFnew	49	PL	LSBK	23			
LNFnew	50	FW	SRN	31	7.7	21	top at 4' RF
LNFnew	51	PL	LSBK	56			bldr dam at btm
LNFnew	52	RF	HGR	53	14.3	30	
LNFnew	53	NS	CAS	1			
LNFnew	54	PL	LSBK	45			shallow
LNFnew	55	FW	RUN	68	18.4	42	
LNFnew	56	NS	CAS	2			
LNFnew	57	PL	LSBO	21	18.6	43	
LNFnew	58	FW	SRN	56	12.0	44	
LNFnew	59	RF	HGR	23	10.4	45	
LNFnew	60	FW	RUN	35	13.8	46	BRIDGE concrete apron at btm
LNFnew	61	PL	LSBK	54			
LNFnew	62	FW	RUN	20	6.5	47	
LNFnew	63	RF	HGR	32	10.0	48	
LNFnew	64	FW	RUN	61			
LNFnew	65	PL	LSBK	36	13.7	49	~PLP
LNFnew	66	FW	SRN	20			
LNFnew	67	RF	HGR	16			
LNFnew	68	PL	STP	46			
LNFnew	69	FW	RUN	6			short & shallow
LNFnew	70	PL	LSBK	105	18.6	50	
LNFnew	71	RF	HGR	29	18.5	51	
LNFmid	2	RF	HGR	57			map 7/13/11 DOWNSTREAM, Q~2.5cfs
LNFmid	3	PL	MCP	57			2ft dam at btm
LNFmid	4	FW	GLD	40	11.6	249	added 200 to WPts to avoid overlap
LNFmid	5	RF	LGR	6			
LNFmid	6	NS	CAS	2			2'
LNFmid	7	PL	PLP	19			small trib RB
LNFmid	8	FW	POW	21			
LNFmid	9	NS	CAS	2			1.5'
LNFmid	10	FW	POW	6			
LNFmid	11	PL	MCP	43			more trib braid RB
LNFmid	12						no unit entered
LNFmid	13	FW	RUN	16			box RB
LNFmid	14	RF	LGR	8			
LNFmid	15	FW	RUN	9			
LNFmid	16	NS	CAS	2			1'
LNFmid	17	PL	PLP	7			
LNFmid	18	NS	CAS	5			8'
LNFmid	19	PL	PLP	11			
LNFmid	20	PL	MCP	49	14.6	248	

Study Site	Unit #	Habitat Level II	Type Level III	Length ft	Width ft	Waypoint #	Comments
LNFmid	21	FW	**RUN**	30	6.0	247	w ater line
LNFmid	22	PL	**LSBK**	10	6.6	246	
LNFmid	23	NS	CAS	2			1.5', rock w all LB
LNFmid	24	PL	LSBK	17			
LNFmid	25	NS	CAS	2			2.5'
LNFmid	26	FW	SRN	3			
LNFmid	27	NS	CAS	2			2'
LNFmid	28	FW	POW	12			
LNFmid	29	NS	CAS	2			
LNFmid	30	PL	LSBK	75			red flag btm LB
LNFmid	31	RF	LGR	20			
LNFmid	32	NS	BRS	10			
LNFmid	33	FW	**RUN**	29	8.6	245	~PL
LNFmid	34	NS	CAS	3			5'
LNFmid	35	PL	PLP	24			
LNFmid	36	FW	**SRN**	27	7.4	244	
LNFmid	37	PL	LSBK	50			
LNFmid	38	RF	HGR	15			
LNFmid	39	PL	MCP	26			
LNFmid	40	RF	**LGR**	25	12.3	243	
LNFmid	41	PL	MCP	22			
LNFmid	42	FW	POW	30			
LNFmid	43	NS	CAS	2			1'
LNFmid	44	PL	MCP	24			
LNFmid	45	RF	LGR	3			
LNFmid	46	**PL**	**STP**	6	13.0	242	2 STPs
LNFmid	47	NS	CAS	1			
LNFmid	48	PL	**STP**	13			
LNFmid	49	RF	HGR	5		230	
LNFmid	50	**PL**	**MCP**	26	18.1		gravel conglomerate
LNFmid	51	RF	**LGR**	33	5.3	221	
LNFmid	52	NS	CAS	2			1'
LNFmid	53	PL	LSBK	20			
LNFmid	54	FW	RUN	21			w PL at btm
LNFmid	55	RF	HGR	7			
LNFmid	56	PL	MCP	30			
LNFmid	57	RF	**LGR**	32	5.1	219	top concrete w all
LNFmid	58	PL	LSBK	56			
LNFmid	59	RF	LGR	34			
LNFmid	60	FW	RUN	12		214	
LNFmid	61	**PL**	**STP**	15	12.5		
LNFmid	62	NS	BRS	12			8'
LNFmid	63	PL	**LSBK**	45	12.0	213	
LNFmid	64	NS	CAS	1			break
LNFmid	65	PL	LSBK	70			
LNFmid	66	FW	GLD	49			
LNFmid	67	NS	CAS	2			1'
LNFmid	68	PL	**STP**	20	17.5	212	
LNFmid	69	**FW**	**SRN**	41	9.1	211	backw ater PL RB
LNFmid	70	PL	MCP	14			
LNFmid	71	RF	LGR	17			
LNFmid	72	FW	RUN	16			
LNFmid	73	PL	MCP	36			
LNFmid	74	RF	**LGR**	38	7.5	210	
LNFmid	75	PL	STP	28			2 PLs
LNFmid	76	NS	CAS	3			4'
LNFmid	77	FW	**RUN**	39	11.8	209	
LNFmid	78	PL	LSR	43			
LNFmid	79	FW	**GLD**	31	17.7	208	
LNFmid	80	**RF**	**LGR**	16	18.7	207	

Study Site	Unit #	Habitat Level II	Type Level III	Length ft	Width ft	Waypoint #	Comments
LNFmid	81	FW	POW	15			
LNFmid	82	RF	**HGR**	36	14.2	206	
LNFmid	83	PL	LSBK	39			
LNFmid	84	FW	GLD	47			
LNFmid	85	RF	HGR	32			
LNFmid	86	FW	POW	19		205	
LNFmid	87	**RF**	**LGR**	27	8.9		
LNFmid	88	NS	CAS	6			3'
LNFmid	89	PL	MCP	32		204	
LNFmid	90	**FW**	**POW**	33	9.1		
LNFmid	91	RF	**LGR**	30	5.3	WPt lost	
LNFmid	92	PL	**LSBK**	72	13.6	WPt lost	
LNFmid	93	FW	GLD	64			
LNFmid	94	FW	RUN	20			small log MC
LNFmid	95	RF	LGR	50		LNFMIDBTM	
Mat 3	1	FW	RUN	95			map btm half 7/23/11, Q~12cfs
Mat 3	2	PL	**MCP**	156	35.0	559	GLD BTM; TEMP LOGGER,TOP 8' BLD
Mat 3	3	FW	RUN	43			
Mat 3	4	RF	**LGR**	48	17.5	558	RB TOO BUSHY TO EFISH
Mat 3	5	FW	**RUN**	134	24.4	557	GLD; MINERAL SEEPS NR TOP MC
Mat 3	6	PL	**MCP**	66	20.5	556	TROUT; UPPER 20' RN
Mat 3	7	RF	**LGR**	75	20.8	555	MINERAL SEEPS RB
Mat 3	8	PL	**MCP**	39	18.7	554	
Mat 3	9	FW	**POW**	88	20.2	553	DEEP POCKET POOL MID
Mat 3	10	RF	**LGR**	43	18.3	552	FW LBANK
Mat 3	11	PL	MCP	42			TROUT
Mat 3	12	RF	HGR	37			
Mat 3	13	FW	SRN	56			TROUT;2 SHORT RUNS W/RF MID,WIDE
Mat 3	14	RF	**LGR**	82	40.3	551	WIDE ART BREAK @ CATAILS
Mat 3	15	RF	LGR	70			
Mat 3	16	FW	**RUN**	88	37.8	550	GLD; TOP 15' SWAYBACK BLDR
Mat 3	17	RF	LGR	38			TRV BTM
Mat 3	18	FW	**RUN**	69	28.9	549	FALLEN ALDER NR TOP
Mat 3	19	PL	**LSBO**	38	27.7	548	SHALLOW EDDY PL BELOW 10'BLDR
Mat 3	20	FW	RUN	80			TROUT, OLD RN30
Mat 3	21	RF	LGR	58			MAYBE MORE RN THAN RF
Mat 3	22	PL	MCP	80			GRAVEL SUPER CEMENTED
Mat 3	23	NS	LGR	25			
Mat 3	24	NS	LSBO	24			CEMENTED-top of low er half
Mat 3	25	PL	**MCP**	109	38.2	571	start upper half, SUNFISH
Mat 3	26	FW	**RUN**	79	29.6	570	SWIM DAM @ TOP, CEMENTED
Mat 3	27	PL	**LSBO**	62	27.7	569	SCOUR @HEAD
Mat 3	28	RF	**LGR**	67	22.9	568	MIDDLE @1/2 POW, HGR TOP/BTM
Mat 3	29	FW	**RUN**	57	21.5	567	
Mat 3	30	RF	HGR	14			OLD RF 40
Mat 3	31	FW	**RUN**	31	19.5	566	
Mat 3	32	RF	**LGR**	62	14.1	565	
Mat 3	33	RF	**LGR**	74	18.9	564	LOWER GRADIENT, CEMENTED
Mat 3	34	FW	GLD	73			CEMENTED
Mat 3	35	PL	**MCP**	167	29.8	563	CEMENTED,BIG POOL,20' POCKET TOP
Mat 3	36	FW	RUN	30			
Mat 3	37	PL	LSBK	32			
Mat 3	38	FW	**SRN**	79	18.1	562	3 STEPS - TOP @3' FALLS
Mat 3	39	RF	**HGR**	72	18.5	561	SPLIT UP LC, SWIM DAM TOP
Mat 3	40	PL	**LSBK**	53	14.5	560	top upper half below lrg corner pool
Mat 5	1	PL	**MCP**	333	33.8	547	map 7/22/11,Q~6cfs,62oF@9am
Mat 5	2	FW	POW	129			
Mat 5	3	PL	**LSBO**	34	14.0	546	channel w idens; PL @ LB; 1 diver
Mat 5	4	RF	**HGR**	63	30.6	545	map up left 1/2; rt 1/2 rf
Mat 5	5	PL	STP	57			2 pls w . RN mid.

Study Site	Unit #	Habitat Level II	Type Level III	Length ft	Width ft	Waypoint #	Comments
Mat 5	*6*	RF	LGR	15			
Mat 5	*7*	PL	LSBO	24			
Mat 5	*8*	FW	SRN	65			
Mat 5	*9*	RF	LGR	29			split mostly rejoins @ top,RF both sides
Mat 5	*10*	FW	**RUN**	51	23.4	544	GLD top @ swim dam
Mat 5	*11*	PL	LSBO	66			upper 1/3 rn
Mat 5	*12*	FW	GLD	58			rt 1/2 swim pl
Mat 5	*13*	FW	**RUN**	61	21.4	543	lower 1/3 PL
Mat 5	*14*	RF	**LGR**	47	18.8	542	w/pockets
Mat 5	*15*	FW	**RUN**	42	17.6	541	
Mat 5	*16*	RF	**LGR**	33	22.2	540	slight increase in gradient
Mat 5	*17*	PL	LSBO	33			w. sand bar opposite gate
Mat 5	*18*	RF	LGR	13			
Mat 5	*19*	FW	RUN	113			small pocket pool @ top
Mat 5	*20*	RF	LGR	48			1/2 RN
Mat 5	*21*	RF	**HGR**	35	10.3	539	
Mat 5	*22*	PL	**LSBO**	21	15.0	538	old rd xing
Mat 5	*23*	FW	**RUN**	33	18.8	537	almost PL, nice gravel
Mat 5	*24*	PL	**MCP**	40	15.6	536	trout?
Mat 5	*25*	FW	POW	57			top @ start split
Mat 5	*26*	RF	HGR	31			wide trvs rc62/LC64.5 degrees @0950
Mat 5	*27*	PL	**STP**	47	15.8	530	trout, 1st small PL
Mat 5	*28*	FW	SRN	53			almost stp
Mat 5	*29*	FW	RUN	40			
Mat 5	*30*	FW	POW	31			w/side PL on RB (BWP)
Mat 5	*31*	PL	**LSBO**	26	18.5	529	shady PL
Mat 5	*32*	FW	RUN	33			w. cas @ btm
Mat 5	*33*	PL	**MCP**	40	16.4	528	nice gravel in tail od lwr PL (Old PL 45)
Mat 5	*34*	NS	CAS	4			
Mat 5	*35*	PL	LSBO	21			short and wide
Mat 5	*36*	FW	**RUN**	44	11.5	527	short RF at top
Mat 5	*37*	PL	LSBO	20			
Mat 5	*38*	RF	**LGR**	47	15.2	526	upper 1/2 RN
Mat 5	*39*	FW	SRN	79			PLS, old PL 49 @ top
Mat 5	*40*	FW	RUN	42			
Mat 5	*41*	RF	**LGR**	41	8.5	525	HGR @ top
Mat 5	*42*	FW	**SRN**	64	11.8	524	
Mat 5	*43*	RF	LGR	57			
Mat 5	*44*	FW	RUN	93			white pipe RB & metal pipe; sc enters LB
Mat 5	*45*						canal enters 40' up RB
Mat 5	*46*	FW	SRN	85			btm along base TRVS RF; w/ sml PL
Mat 5	*47*	RF	**HGR**	41	10.0	531	upper 1/2 LGR
Mat 5	*48*	FW	**SRN**	87	14.6	532	w/pocket PL at top
Mat 5	*49*	RF	LGR	51			split enters btm, up mid chan
Mat 5	*50*	FW	**SRN**	64	8.8	533	w/sml PLs
Mat 5	*51*	RF	LGR	112			very narrow
Mat 5	*52*	FW	RUN	50			splits merge @ top, sml tadpoles
Mat 5	*53*	RF	LGR	97			wide, left side FW, gravel more cemented
Mat 5	*54*	PL	**MCP**	71	17.7	534	trout, OVH tree @ top - nice pool
Mat 5	*55*	RF	**HGR**	60	10.9	535	
Mat 5	*56*	FW	GLD	101			Lots of 2"-3" gravel and single chan.
Mat 7b	*1*	PL	STP	48			map 7/21/11, Q~4.4cfs, 3 steps
Mat 7b	*2*	NS	CAS	16			
Mat 7b	*3*	PL	**PLP**	35	14.8	523	incl. sml LSBO @btm, 10" trt
Mat 7b	*4*	RF	**HGR**	45	7.5	522	bedrock ledges w/pockets
Mat 7b	*5*	PL	PLP	24			
Mat 7b	*6*	NS	CAS	7			bedrock slide
Mat 7b	*7*	FW	RUN	42			lrg grvl dep LB
Mat 7b	*8*	PL	PLP	27			
Mat 7b	*9*	NS	CAS	49			w/2 sPL, giant bldr RB top

Study Site	Unit #	Habitat Level II	Type Level III	Length ft	Width ft	Waypoint #	Comments
Mat 7b	10	PL	STP	39			3 PLs, RBT
Mat 7b	11	NS	CAS	7			w/3.5' drop
Mat 7b	12	FW	SRN	29			2 steps
Mat 7b	13	PL	**LSBK**	33	12.5	521	w/bldr scour at btm, shallow
Mat 7b	14	FW	**RUN**	41	8.5	520	sml LSBO @ top
Mat 7b	15	NS	HGR	19			low 1/2 CAS,up 1/2 RN
Mat 7b	16	PL	PLP	27			w/lrg bldr scour on RB
Mat 7b	17	RF	**HGR**	41	10.2	519	w/cas @ btm, upper 1/2 LGR/GLD
Mat 7b	18	PL	LSBK	58			RBT
Mat 7b	19	RF	HGR	21			2 outlets, hand net btm
Mat 7b	20	FW	RUN	12			
Mat 7b	21	PL	LSBO	36			several RBT 4-7"
Mat 7b	22	NS	BRS	25			
Mat 7b	23						unit added to LSBK
Mat 7b	24	PL	LSBK	37			cliff LB
Mat 7b	25	RF	BRS	5			
Mat 7b	26	RF	**HGR**	23	10.7	518	
Mat 7b	27	PL	LSBO	22			
Mat 7b	28	NS	CAS	5			
Mat 7b	29	FW	GLD	30			or shallow PL
Mat 7b	30	PL	**MCP**	15	13.3	517	
Mat 7b	31	FW	SRN	32			
Mat 7b	32	FW	POW	49			
Mat 7b	33	NS	BRS	9			
Mat 7b	34	RF	LGR	43			split LC GLD
Mat 7b	35	PL	**LSBO**	20	37.0	516	8-10 RBT 4-8", deep
Mat 7b	36	FW	SRN	67			w. pocket PL top RB
Mat 7b	37	RF	BRS	43			small split
Mat 7b	38	FW	RUN	29			small split
Mat 7b	39	FW	**POW**	45	12.9	515	small split
Mat 7b	40	RF	**LGR**	53	17.2	514	small split
Mat 7b	41	FW	**RUN**	28	14.3	513	perched boulders RB
Mat 7b	42	NS	BRS	10			w/ pocket pools RC
Mat 7b	43	FW	**POW**	44	8.7	512	upper 20' run
Mat 7b	44	PL	LSBO	60			lower 2/3 alon bdrk
Mat 7b	45	RF	**HGR**	60	8.7	511	w/pocket PL in middle, upper LGR
Mat 7b	46	PL	LSBO	34			
Mat 7b	47	FW	**POW**	27	9.2	510	
Mat 7b	48	FW	GLD	32			trt
Mat 7b	49	PL	LSBO	43			LB all sloped brk, shallow
Mat 7b	50	FW	RUN	29			in bdrk channel; upper 10'pwl
Mat 7b	51	RF	**HGR**	74	4.7	509	lower 1/2 chute, upper w/ pockets; trt
Mat 7b	52	FW	RUN	19			
Mat 7b	53	PL	LSBO	9			
Mat 7b	54	RF	HGR	13			
Mat 7b	55	FW	RUN	10			downed tree @ top
Mat 7b	56	PL	STP	36			2nd pool shallow w/ww
Mat 7b	57	NS	CAS	19			
Mat 7b	58	PL	PLP	41			temp logger pl, trt
Mat 7b	59	NS	CAS	9			
Mat 7b	60	PL	**LSBK**	47	10.8	508	
Mat 7b	61	NS	BRS	17			
Mat 7b	62	NS	CAS	13			
Mat 7b	63	FW	POW	42			turtle
Mat 7b	64	PL	PLP	39			
Mat 7b	65	NS	BRS	19			steps, not chute
Mat 7b	66	PL	**LSBK**	71	15.3	507	lrg BLDR top LB, trib LB, trt
Mat 7b	67	FW	RUN	17			
Mat 7b	68	PL	STP	43			2 pools
Mat 7b	69	NS	CAS	5			

Study Site	Unit #	Habitat Level II	Type Level III	Length ft	Width ft	Waypoint #	Comments
Mat 7b	*70*	RF	HGR	21			
Mat 7b	*71*	PL	**LSBO**	31	17.0	506	
Mat 7b	*72*	FW	**SRN**	79	8.5	505	upper 1/2 RF, 3-5" trt
Mat 7b	*73*	PL	PLP	41			
Mat 7b	*74*	RF	**HGR**	24	5.8	504	hand net @ btm
Mat 7b	*75*	PL	PLP	19			
Mat 7b	*76*	FW	SRN	53			
Mat 7b	*77*	PL	STP	46			
Mat 7b	*78*	NS	BRS	32			
Mat 7b	*79*	PL	LSBO	28			
Mat 7b	*80*	NS	CAS	7			
Mat 7b	*81*	FW	RUN	10			
Mat 7b	*82*	PL	PLP	33			deep and w ide
Mat 7b	*83*	NS	CAS	5			
Mat 7b	*84*	PL	LSBO	43			10' RN mid w /huge BLDR top RB; rbt
Mat 7b	*85*	NS	CAS	12			
Mat 7b	*86*	RF	LGR	40			step RN
Mat 7b	*87*	FW	**RUN**	25	9.7	503	shallow Pl w /RF break top
Mat 7b	*88*	PL	LSBO	33			several trt 4-6"
Mat 7b	*89*	RF	**HGR**	108	16.0	502	braided, CAS @ btm
Mat 7b	*90*	FW	**SRN**	59	12.2	501	w ide w /shallow RF
Mat 7b	*91*	PL	**PLP**	84	17.5	500	CAS/falls; 50-80 RBT to 10"
UNF	*1*	PL	MCP	12			map 7/24/11, Q~3.2cfs, abv rock slide
UNF	*2*	FW	GLD	55			aggraded from rock slide, short rf top
UNF	*3*	FW	RUN	41			pcoket pls top lb/rb
UNF	*4*	RF	HGR	13			
UNF	*5*	PL	LSBK	18			
UNF	*6*	FW	RUN	30			
UNF	*7*	RF	HGR	60			split w /pocket pls;trt
UNF	*8*	FW	GLD	18			trail xing
UNF	*9*	PL	LSBK	29			trvs rf top
UNF	*10*	RF	**LGR**	61	16.7	595	trail Xing, very w ide
UNF	*11*	PL	LSBK	88			overhanging cliff LB; short rf middle
UNF	*12*	RF	**LGR**	25	4.5	594	
UNF	*13*	PL	LSBO	15			small plunge top
UNF	*14*	FW	**RUN**	37	8.9	593	SC w /trickle @ LB Btm
UNF	*15*	RF	**HGR**	58	11.7	592	or POW; trail xing top; trt
UNF	*16*	PL	**LSBK**	25	12.3	291	
UNF	*17*	FW	RUN	16			
UNF	*18*	PL	LSBK	34			w /undercut bank RB
UNF	*19*	RF	HGR	53			
UNF	*20*	PL	**LSBK**	55	11.7	590	perched dirt/gravel w /spring RB; trt
UNF	*21*	FW	**SRN**	29	9.4	589	low er RN w /pocket PL
UNF	*22*	PL	LSBO	40			
UNF	*23*	RF	LGR	35			almost RN
UNF	*24*	RF	HGR	68			w /pockets
UNF	*25*	FW	SRN	85			~15' RF in middle
UNF	*26*	PL	STP	45			2PLs, 8' slab bldr @ cascade top
UNF	*27*	NS	CAS	7			
UNF	*28*	PL	LSBO	22			
UNF	*29*	NS	CAS	15			split - up RC
UNF	*30*	PL	**STP**	47	7.7	588	3 PLs top PL w /8' hanging bldr
UNF	*31*	NS	LGR	4			w /brokern calcified gravel shelf
UNF	*32*	PL	**LSBO**	34	8.7	587	brushy, rn middle
UNF	*33*	NS	CAS	8			
UNF	*34*	FW	RUN	14			
UNF	*35*	RF	**LGR**	22	3.3	586	
UNF	*36*	NS	POW	67			dow ned trees xing channel
UNF	*37*	PL	**MCP**	23	9.0	585	UCB LB, trt
UNF	*38*	RF	HGR	19			

Study Site	Unit #	Habitat Level II	Type Level III	Length ft	Width ft	Waypoint #	Comments
UNF	39	FW	**SRN**	55	7.1	584	top by 6" downed snag xing channel
UNF	40	PL	LSBO	43			lower 1/2 RN/GLD
UNF	41	RF	LGR	15			
UNF	42	FW	**RUN**	24	9.2	583	w/pocket pools - more PL than RN
UNF	43	RF	**LGR**	55	6.5	582	w/pockets
UNF	44	PL	MCP	31			shallow
UNF	45	PL	STP	86			~5 PLs
UNF	46	RF	**HGR**	31	8.4	581	nice pocket pool top LB
UNF	47	FW	**RUN**	22	7.5	580	
UNF	48	PL	**LSBO**	72	12.5	579	15' bldr @ btm; short pocket @ top
UNF	49	FW	**POW**	54	9.7	578	or RF w/ pockets
UNF	50	FW	SRN	58			almost PLs
UNF	51	FW	**POW**	74	9.5	577	mid 1/3 HGR, upper 1/2 old sample unit
UNF	52	PL	**MCP**	43	5.8	576	6" tree xing in middle(live willow)
UNF	53	RF	LGR	50			w/cascade @ top
UNF	54	RF	HGR	61			channel does zig zag around bldr
UNF	55	FW	RUN	46			split cas at top; up LC
UNF	56	PL	LSBO	28			15' bldr @ top
UNF	57	PL	STP	67			3rd step RN
UNF	58	FW	GLD	78			gravel flat, lower 1/3 RF
UNF	59	RF	LGR	43			w/pockets; high Q channel to LB
UNF	60	PL	**LSBO**	39	8.7	575	4" trt
UNF	61	RF	**LGR**	51	4.0	574	upper 1/2 w/small pockets
UNF	62	FW	**POW**	33		573	
UNF	63	RF	LGR	12			cuts through gravel bar
UNF	64	PL	LSBO	27			enterin open channel; deep; trt?
UNF	65	RF	**LGR**	70		572	w/FW areas
UNF	66	RF	HGR	70			substrate much more calcified in sun?
MUR3	1	FW	POW	10			map 6/4/12, Q~0.5cfs
MUR3	2	PL	MCP	32			gravel, shallow in middle
MUR3	3	FW	**SRN**	66	9.1	190	w pocket pools, split at top
MUR3	4	RF	HGR	14			up Rt channel, 90% Q
MUR3	5	PL	LSBo	20			
MUR3	6	RF	**HGR**	24	3.8	189	
MUR3	7	PL	**MCP**	14	8.7	164	
MUR3	8	NS	CAS	2			
MUR3	9	RF	**LGR**	27		163	btm 10' run
MUR3	10	PL	MCP	26			
MUR3	11	PL	STP	41			2 pools
MUR3	12	NS	CAS	5			TRV LGR/CAS
MUR3	13	PL	**MCP**	25	9.7	162	
MUR3	14	FW	**RUN**	27	6.8	147	channel bends left at top
MUR3	15	NS	X				units added to LGR 17
MUR3	16	NS	X				units added to LGR 17
MUR3	17	RF	**LGR**	43	7.3	145	unit going dry 7/11/12
MUR3	18	FW	**SRN**	27	8.6	143	2 runs
MUR3	19	PL	LSBo	14			
MUR3	20	NS	LGR	5			short break
MUR3	21	PL	PLP	23			gravel
MUR3	22	NS	CAS	4			
MUR3	23	PL	MCP	32			gravel
MUR3	24	FW	POW	11			
MUR3	25	PL	MCP	28			gravel
MUR3	26	NS	CAS	5			
MUR3	27	PL	PLP	18			
MUR3	28	RF	HGR	22			
MUR3	29	PL	**LSBo**	21	0.0	141	unit dry 7/11/12
MUR3	30	NS	CAS	14			
MUR3	31	PL	LSBo	30			top half narrow
MUR3	32	NS	CAS	3			

Study Site	Unit #	Habitat Level II	Type Level III	Length ft	Width ft	Waypoint #	Comments
MUR3	33	PL	**STP**	50		136	3 pools
MUR3	34	NS	CAS	2			
MUR3	35	PL	STP	30			2 pools
MUR3	36	FW	**SRN**	37	0.0	126	gravel, unit dry 7/11/12
MUR3	37	PL	**LSRt**	25	8.2	120	tree roots at btm, no flow 7/11/12
MUR3	38	RF	**HGR**	24	0.0	118	run in middle, unit dry 7/11/12
MUR3	39	FW	**RUN**	33	11.0	107	lower 1/2 unit dry 7/11/12
MUR3	40	PL	LSBo	47			
MUR3	41	NS	CAS	7			
MUR3	42	FW	RUN	18			channel bends left at top
MUR3	43	RF	**LGR**	21	0.0	104	unit dry 7/11/12
MUR3	44	PL	DPL	13			shallow
MUR3	45	RF	**HGR**	32	0.0	103	possible redd, trail top LB, unit dry 7/11/12
MUR3	46	PL	PLP	34			gravel
MUR3	47	NS	CAS	2			
MUR3	48	FW	RUN	13			slow
MUR3	49	PL	MCP	20			trib enters just abv pool
MUR3	50	RF	HGR	28			
MUR3	51	PL	PLP	19			gravel
MUR3	52	FW	SRN	37			
MUR3	53	RF	**HGR**	49	6.9	79	low half RF w pockets, min Q
MUR3	54	PL	PLP	27			
MUR3	55	FW	**RUN**	17	7.2	67	LWD at top, full Q again 7/11/12
MUR3	56	PL	LSBo	7			pool on LB not mapped
MUR3	57	NS	CAS	7			flows under lrg bldrs
MUR3	58	PL	MCP	44			
MUR3	59	NS	CAS	13			
MUR3	60	FW	**RUN**	21	10.8	63	
MUR3	61	PL	PLP	51			
MUR3	62	FW	RUN	28			
MUR3	63	RF	LGR	59			
MUR3	64	NS	CAS	7			trail xing w flags
MUR3	65	PL	LSBo	15			
MUR3	66	RF	**LGR**	35	6.7	54	w slow channels on RB
MUR3	67	PL	LSBo	41			gravel, sml PL LB, lrg Pl RB
MUR3	68	NS	CAS	2			
MUR3	69	PL	LSBo	26			channel opens up
MUR3	70	NS	CAS	3			3 ft drop
MUR3	71	FW	**SRN**	32	7.8	53	
MUR3	72	RF	HGR	71			long w run
MUR3	73	PL	LSBo	39			lower half w break, gravel
MUR3	74	RF	LGR	10			
MUR3	75	PL	LSBo	74			4 ft deep in middle
MUR3	76	NS	CAS	37			5-7ft falls at btm
MUR3	77	PL	LSBo	16			back into tree canopy, narrow
MUR3	78	NS	CAS	14			
MUR3	79	PL	LSBo	14			
MUR3	80	RF	HGR	22			
MUR3	81	PL	**LSBo**	53	15.8	3	
MUR3	82	NS	CAS	19			
MUR3	83	RF	LGR	34			sml pools at btm
MUR3	84	PL	LSBo	25			
MUR3	85	RF	LGR	6			
MUR3	86	PL	MCP	15			short
MUR3	87	RF	HGR	55			
MUR3	88	FW	RUN	41			
MUR3	89	PL	**LSBo**	27	16.7	2	
MUR3	90	PL	**STP**	25	7.7	1	
MUR3	91	NS	CAS	9			
MUR3	92	FW	RUN	11			

Study Site	Unit #	Habitat Level II	Type Level III	Length ft	Width ft	Waypoint #	Comments
MUR3	*93*	PL	LSBo	20			
MUR3	*94*	FW	RUN	11			
MUR3	*95*	PL	STP	43			gravel, 2 pools, upper one shallow
MUR3	*96*	NS	CAS	5			
MUR3	*97*	FW	SRN	18			upper half RF
MUR3	*98*	PL	LSBo	29			confluence w major trib LB (~50% Q)

Appendix C

Protocols for Minimizing Take of California Red-Legged Frogs

MEMORANDUM

To: Matt McGoogan, NMFS
From: Mark Allen, Normandeau Associates
Date: 4-4-12

RE: Protocols to reduce impact to red-legged frogs during electrofishing

Electrofishing for salmonid population abundance studies has the potential to negatively impact amphibian species in addition to the target fish. Below is a list of protocols that Normandeau proposes to employ in an effort to minimize the potential negative impacts of backpack electrofishing on California red-legged frogs (RLFs) in the Ventura/Matilija Creek basin.

- Personnel: at least one member of the electrofishing crew will be experienced in the identification and biology of sensitive amphibian species, and all crew members will be instructed in the identification of RLFs and avoidance protocols specified in the sampling permit.
- Survey Period: electrofishing will not be conducted until early-mid July, after egg masses have hatched into free-swimming (and readily observable) tadpoles or adults (earlier surveys, if conducted, will only utilize snorkeling). Electrofishing will only be conducted in shallow/swift water areas, all deeper habitats will be surveyed using non-invasive snorkeling techniques.
- Pre-Survey: All sampling areas will be visited prior to electrofishing to delineate sampling boundaries, and at that time a careful visual survey will be conducted to ascertain the presence and location of any RLFs; if observed in the sampling area, the exact location of the tadpoles (or adults) will be carefully described and marked with bright-colored flagging in order to ensure avoidance of that area during the actual electrofishing survey. If necessary, that sampling unit will be discarded and a new sampling unit (w/out RLF sightings) will be selected in its place, however that option will affect the statistical comparisons. The pre-survey will occur immediately prior to the electrofishing by one or two (depending on channel width) biologists as they walk the entire sampling unit. All subsequent electrofishing, if conducted, will circumvent any RLF observation location by a minimum of 10 ft. The electrical field in the Ventura/Matilija Basin is typically very narrow ($\leq$5ft) due to high conductivity of the water and absorption of the current.
- Electrofisher Settings: All electrofishing will be conducted under the supervision of a crew leader with extensive experience in the use of backpack electrofishing equipment and fish sampling procedures. Prior to sampling, the backpack electrofisher controls will be set to the levels necessary to stun and capture juvenile steelhead with the minimum of impact, according to the sampling permit requirements, which should also help to minimize impact to any incidentally stunned amphibians. Although we could not locate any scientific documentation on the effects of electrofishing on frogs or tadpoles, and personal communications with other frog experts did not yield specific information, our personal anecdotal observations from many years of electrofishing in Pacific states streams have suggested that adult and subadult frogs are susceptible to electrofishing (e.g., they are

stunned as are fish), however none in our office can remember any incidents of frog mortalities. In contrast, mortalities of fish have been observed at an average rate of about 1-3%. These observations suggest to us that frogs are generally less sensitive to the adverse effects of electrofishing than are fish.

- Avoidance Procedures: During an electrofishing survey, the crew will carefully scan the upstream water column, with particular emphasis on shallow/warm margin and shallow pool habitats, in order to identify any tadpoles or adult frogs to minimize the likelihood of energizing the electrofisher in the vicinity of those animals, or injuring the animals during wading and netting activities; if observed in the sample area during the pre-survey (see above) or during the sampling, the electrofishing crew will make a wide berth around the observed RLFs to ensure that the RLFs are not subject to the electric field; if RLF are observed throughout the sampling unit, that unit will be discarded and a new sampling unit (w/out RLF sightings) will be selected in its place.
- Capture and Recovery Procedures: Although we are confident that our protocols will successfully avoid shocking a RLF, if a RLF tadpole or adult is accidentally stunned by the electrofisher, the animal will be netted and placed in a recovery unit; stunned tadpoles will be immediately transferred into the instream live-car (used for captured juvenile steelhead) until full recovery; stunned adults will be placed in a shaded bucket containing moistened vegetation; after ensuring full recovery, all captured RLF adults and tadpoles will be released back into the sampling unit following the conclusion of the electrofishing survey.
- Record Keeping: Careful records will be kept to document any sightings or captures of RLF according to lifestage; GPS coordinates will be associated with each sighting, and if desired and if authorized under the sampling permit, genetic samples may be taken and transferred to an appropriate agency. The NMFS (or FWS, depending upon permit conditions) will be contacted for further instructions if the number of captured RLFs approaches the maximum level specified under the sampling permit.

Appendix D

O. mykiss Abundance Estimates According to Year, Size Class, Habitat Type, and Study Site, 2006-2012

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2012	Fry	Pools	# Units Sampled	0	0	8	0	8	7	8	8	8	8	8	8	8	8
	<10cm		Abundance	-	-	70	-	0	6	0	166	354	29	110	192	164	121
			Variance	-	-	58	-	0	0	0	530	2798	40	489	677	1233	671
			95% C.I.	-	-	18	-	0	0	0	54	125	15	52	62	83	61
			Density (#/mi)	-	-	180.3	-	0	97	0	881.8	1,868.8	181.0	696	842	940	553
			Variance (#/mi)	-	-	380.54	-	0	0	0	14,990.86	78,161.86	1,561.59	19,657	13,064	40,356	14,067
			95% C.I. (#/mi)	-	-	46.1	-	0	0	0	289.5	661.1	93.4	332	270	475	280
			Density (#/100ft^2)	-	-	0.09	-	0.00	0.14	0.00	0.91	2.55	0.13	0.73	0.98	1.80	0.83
			Variance (#/100ft^2)	-	-	0.0001	-	0.0000	0.0000	0.0000	0.0161	0.1451	0.00080	0.0215	0.0176	0.1477	0.0318
			95% C.I. (#/100ft^2)	-	-	0.02	-	0.00	0.00	0.00	0.30	0.90	0.07	0.35	0.31	0.91	0.42
2012	Fry	Flatwaters	# Units Sampled	8	7	8	0	8	8	8	8	8	8	8	8	7	8
	<10cm		Abundance	0	5	398	-	0	14	0	93	223	67	241	210	250	114
			Variance	0	19	6736	-	0	14	0	97	1714	75	2379	3331	572	168
			95% C.I.	0	11	194	-	0	9	0	23	98	21	115	136	58	31
			Density (#/mi)	0.0	10.1	1,548.1	-	0	93	0	623	1,873	354.1	864	1,304	1,718	1,315
			Variance (#/mi)	0.00	84.01	101,676	-	0	590	0	4,344	120,372	2,093.37	30,488	128,525	26,944	22,342
			95% C.I. (#/mi)	0.0	22.4	754.0	-	0	57	0	156	820	108.2	413	848	402	353
			Density (#/100ft^2)	0.00	0.00	1.128	-	0.00	0.15	0.00	0.82	3.51	0.27	1.02	2.40	4.23	3.28
			Variance (#/100ft^2)	0.0000	0.0000	0.0540	-	0.0000	0.0016	0.0000	0.0076	0.4233	0.0012	0.0427	0.4346	0.1630	0.1387
			95% C.I. (#/100ft^2)	0.00	0.01	0.55	-	0.00	0.09	0.00	0.21	1.54	0.08	0.49	1.56	0.99	0.88
2012	Fry	Riffles	# Units Sampled	8	8	8	0	8	8	8	4	8	8	8	8	6	8
	<10cm		Abundance	0	45	374	-	0	6	207	151	270	325	165	92	388	106
			Variance	0	173	3863	-	0	19	293	1323	1567	6254	1324	544	4488	214
			95% C.I.	0	31	147	-	0	11	40	116	94	187	86	55	172	35
			Density (#/mi)	0.0	138.9	1,682.4	-	0	29	1,791	2,405	2,903	2,318.9	1,065	790	2,347	969
			Variance (#/mi)	0.00	1,622.76	78,277	-	0	484	21,915	334,556	181,190	318,400	54,902	40,210	164,536	17,988
			95% C.I. (#/mi)	0.0	95.3	661.6	-	0	54	350	1,841	1,007	1,334.3	554	474	1,043	317
			Density (#/100ft^2)	0.00	0.11	1.40	-	0.00	0.05	1.35	3.48	5.73	2.05	1.28	1.37	6.26	2.82
			Variance (#/100ft^2)	0.0000	0.0010	0.0545	-	0.0000	0.0015	0.0125	0.6993	0.7052	0.2494	0.0789	0.1214	1.1708	0.1527
			95% C.I. (#/100ft^2)	0.00	0.08	0.55	-	0.00	0.09	0.26	2.66	1.99	1.18	0.66	0.82	2.78	0.92
2012	Fry	All	# Units Sampled	23	22	24	0	24	23	24	20	24	24	24	24	21	24
	<10cm	Habitats	Abundance	0	50	843	-	0	26	207	410	847	421	517	494	802	340
			Variance	0	192	10657	-	0	33	293	1950	6079	6369	4193	4552	6293	1053
			95% C.I.	0	36	215	-	0	12	36	93	162	166	140	146	175	70
			Density (#/mi)	0.0	50.0	969.0	-	0	63	424	1,024	2,110	860	872	978	1,652	822
			Variance (#/mi)	0.00	191.06	14,095.72	-	0	193	1,227	12,144	37,705	26,553	11,946	17,853	26,684	6,137
			95% C.I. (#/mi)	0.0	35.5	246.9	-	0	29	73	233	404	339	236	289	360	169
			Density (#/100ft^2)	0.00	0.03	0.61	-	0.00	0.10	0.29	1.21	3.39	0.67	1.00	1.41	3.77	1.57
			Variance (#/100ft^2)	0.0000	0.0001	0.0056	-	0.0000	0.0005	0.0006	0.0170	0.0975	0.0159	0.0157	0.0371	0.1393	0.0223
			95% C.I. (#/100ft^2)	0.00	0.02	0.15	-	0.00	0.05	0.05	0.28	0.65	0.26	0.27	0.42	0.82	0.32

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2012	Juv	Pools	# Units Sampled	0	0	8	0	8	7	8	8	8	8	8	8	8	8
	10-20cm		Abundance	-	-	30	-	0	5	9	17	69	4	18	197	35	29
			Variance	-	-	7	-	0	0	0	9	183	0	26	3471	33	105
			95% C.I.	-	-	6	-	0	0	0	7	32	2	12	139	14	24
			Density (#/mi)	-	-	76.9	-	0	81	62.2	91.7	366.1	24.4	111	867	197	133
			Variance (#/mi)	-	-	44.39	-	0	0	0.00	243.85	5,114.24	17.89	1,038	66,970	1,071	2,207
			95% C.I. (#/mi)	-	-	15.8	-	0	0	0.0	36.9	169.1	10.0	76	612	77	111
			Density (#/100ft^2)	-	-	0.04	-	0.00	0.12	0.04	0.09	0.50	0.02	0.12	1.01	0.38	0.20
			Variance (#/100ft^2)	-	-	0.0000	-	0.0000	0.0000	0.0000	0.0003	0.0095	0.00001	0.0011	0.0904	0.0039	0.0050
			95% C.I. (#/100ft^2)	-	-	0.01	-	0.00	0.00	0.00	0.04	0.23	0.01	0.08	0.71	0.15	0.17
2012	Juv	Flatw aters	# Units Sampled	8	7	8	0	8	8	8	8	8	8	8	8	7	8
	10-20cm		Abundance	0	0	50	-	0	0	0	16	15	22	26	105	89	88
			Variance	0	0	160	-	0	0	0	34	10	71	135	610	122	454
			95% C.I.	0	0	30	-	0	0	0	14	7	20	27	58	27	50
			Density (#/mi)	0.0	0.0	194.3	-	0	0	0	106	126	118.3	95	652	609	1,012
			Variance (#/mi)	0.00	0.00	2,417	-	0	0	0	1,510	674	1,982.58	1,729	23,542	5,754	60,277
			95% C.I. (#/mi)	0.0	0.0	116.2	-	0	0	0	92	61	105.3	98	363	186	581
			Density (#/100ft^2)	0.00	0.00	0.14	-	0.00	0.00	0.00	0.14	0.24	0.09	0.11	1.20	1.50	2.52
			Variance (#/100ft^2)	0.0000	0.0000	0.0013	-	0.0000	0.0000	0.0000	0.0026	0.0024	0.0011	0.0024	0.0796	0.0348	0.3743
			95% C.I. (#/100ft^2)	0.00	0.00	0.08	-	0.00	0.00	0.00	0.12	0.12	0.08	0.12	0.67	0.46	1.45
2012	Juv	Riffles	# Units Sampled	8	8	8	0	8	8	8	4	8	8	8	8	6	8
	10-20cm		Abundance	0	21	41	-	0	4	17	39	5	20	31	41	69	52
			Variance	0	53	56	-	0	14	4	94	15	22	102	133	286	134
			95% C.I.	0	17	18	-	0	9	5	31	9	11	24	27	43	27
			Density (#/mi)	0.0	65.8	182.9	-	0	19	146	625	54	139.1	201	355	419	474
			Variance (#/mi)	0.00	500.61	1,131.21	-	0	337	272	23,864	1,735	1,110.93	4,244	9,858	10,471	11,236
			95% C.I. (#/mi)	0.0	52.9	79.5	-	0	45	39	492	98	78.8	154	235	263	251
			Density (#/100ft^2)	0.00	0.05	0.15	-	0.00	0.03	0.11	0.90	0.11	0.12	0.24	0.62	1.12	1.38
			Variance (#/100ft^2)	0.0000	0.0003	0.0008	-	0.0000	0.0010	0.0002	0.0499	0.0068	0.0009	0.0061	0.0298	0.0745	0.0954
			95% C.I. (#/100ft^2)	0.00	0.04	0.07	-	0.00	0.08	0.03	0.71	0.19	0.07	0.18	0.41	0.70	0.73
2012	Juv	All	# Units Sampled	23	22	24	0	24	23	24	20	24	24	24	24	21	24
	10-20cm	Habitats	Abundance	0	21	121	-	0	9	26	72	89	46	75	344	192	169
			Variance	0	53	223	-	0	14	4	137	208	94	263	4214	440	692
			95% C.I.	0	19	31	-	0	8	4	25	30	20	35	140	46	57
			Density (#/mi)	0.0	21.4	138.7	-	0	21	52	181	222	94	127	680	396	407
			Variance (#/mi)	0.00	53.11	294.54	-	0	79	15	852	1,288	391	750	16,529	1,867	4,036
			95% C.I. (#/mi)	0.0	18.7	35.7	-	0	19	8	62	75	41	59	278	95	137
			Density (#/100ft^2)	0.00	0.012	0.087	-	0.00	0.04	0.036	0.21	0.36	0.07	0.15	0.98	0.91	0.77
			Variance (#/100ft^2)	0.0000	0.00002	0.0001	-	0.0000	0.0002	0.0000	0.0012	0.0033	0.0002	0.0010	0.0343	0.0098	0.0146
			95% C.I. (#/100ft^2)	0.00	0.011	0.02	-	0.00	0.03	0.006	0.07	0.12	0.03	0.07	0.40	0.22	0.26

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2012	Adult	Pools	# Units Sampled	0	0	8	0	8	7	8	8	8	8	8	8	8	8
	>20cm		Abundance	-	-	73	-	0	6	5	17	4	5	4	4	12	0
			Variance	-	-	355	-	0	0	0	41	18	2	2	19	38	0
			95% C.I.	-	-	45	-	0	1	0	15	10	3	3	10	15	0
			Density (#/mi)	-	-	185.9	-	0	97	37.3	91.7	20.5	31.6	22	18	66	0
			Variance (#/mi)	-	-	2,333.11	-	0	38	0.00	1,150.68	497.93	71.54	84	370	1,241	0
			95% C.I. (#/mi)	-	-	114.2	-	0	15	0.0	80.2	52.8	20.0	22	46	83	0
			Density (#/100ft^2)	-	-	0.09	-	0.00	0.14	0.02	0.09	0.03	0.02	0.02	0.02	0.13	0.00
			Variance (#/100ft^2)	-	-	0.0006	-	0.0000	0.0001	0.0000	0.0012	0.0009	0.00004	0.0001	0.0005	0.0045	0.0000
			95% C.I. (#/100ft^2)	-	-	0.06	-	0.00	0.02	0.00	0.08	0.07	0.01	0.02	0.05	0.16	0.00
2012	Adult	Flatw aters	# Units Sampled	8	7	8	0	8	8	8	8	8	8	8	8	7	8
	>20cm		Abundance	2	0	0	-	0	0	0	8	6	2	3	3	0	0
			Variance	4	0	0	-	0	0	0	6	10	0	6	6	0	0
			95% C.I.	4	0	0	-	0	0	0	6	7	0	6	6	0	0
			Density (#/mi)	6.3	0.0	0.0	-	0	0	0	52	50	9.2	12	19	0	0
			Variance (#/mi)	25.34	0.00	0	-	0	0	0	252	674	0.00	79	233	0	0
			95% C.I. (#/mi)	11.9	0.0	0.0	-	0	0	0	38	61	0.0	21	36	0	0
			Density (#/100ft^2)	0.01	0.00	0.00	-	0.00	0.00	0.00	0.07	0.09	0.01	0.01	0.03	0.00	0.00
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	-	0.0000	0.0000	0.0000	0.0004	0.0024	0.0000	0.0001	0.0008	0.0000	0.0000
			95% C.I. (#/100ft^2)	0.01	0.00	0.00	-	0.00	0.00	0.00	0.05	0.12	0.00	0.02	0.07	0.00	0.00
2012	Adult	Riffles	# Units Sampled	8	8	8	0	8	8	8	4	8	8	8	8	6	8
	>20cm		Abundance	5	19	0	-	0	0	6	3	0	2	4	0	3	0
			Variance	7	20	0	-	0	0	1	6	0	1	9	0	9	0
			95% C.I.	6	11	0	-	0	0	2	8	0	2	7	0	8	0
			Density (#/mi)	15.8	58.5	0.0	-	0	0	52	44	0	11.6	29	0	21	0
			Variance (#/mi)	85.95	185.88	0.00	-	0	0	69	1,461	0	53.18	362	0	325	0
			95% C.I. (#/mi)	21.9	32.2	0.0	-	0	0	20	122	0	17.2	45	0	46	0
			Density (#/100ft^2)	0.01	0.05	0.00	-	0.00	0.00	0.04	0.06	0.00	0.01	0.03	0.00	0.06	0.00
			Variance (#/100ft^2)	0.0001	0.0001	0.0000	-	0.0000	0.0000	0.0000	0.0031	0.0000	0.0000	0.0005	0.0000	0.0023	0.0000
			95% C.I. (#/100ft^2)	0.02	0.03	0.00	-	0.00	0.00	0.01	0.18	0.00	0.02	0.05	0.00	0.12	0.00
2012	Adult	All	# Units Sampled	16	15	24	0	24	23	24	20	24	24	24	24	21	24
	>20cm	Habitats	Abundance	7	19	73	-	0	6	11	28	10	8	11	7	15	0
			Variance	11	20	355	-	0	0	1	52	27	3	17	25	47	0
			95% C.I.	8	11	39	-	0	1	2	15	11	4	9	11	15	0
			Density (#/mi)	7.4	19.1	83.4	-	0	14	23	69	25	17	19	14	31	0
			Variance (#/mi)	12.28	19.72	469.28	-	0	1	4	325	170	12	48	99	198	0
			95% C.I. (#/mi)	9.0	11.4	45.1	-	0	2	4	38	27	7	15	21	31	0
			Density (#/100ft^2)	0.01	0.011	0.052	-	0.00	0.02	0.016	0.08	0.04	0.01	0.02	0.02	0.07	0.00
			Variance (#/100ft^2)	0.0000	0.00001	0.0002	-	0.0000	0.0000	0.0000	0.0005	0.0004	0.0000	0.0001	0.0002	0.0010	0.0000
			95% C.I. (#/100ft^2)	0.01	0.007	0.03	-	0.00	0.00	0.003	0.05	0.04	0.01	0.02	0.03	0.07	0.00

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2012	Juv+	Pools	# Units Sampled	0	0	8	0	8	7	8	8	8	8	8	8	8	8
	≥10cm		Abundance	-	-	103	-	0	11	14	35	73	9	21	201	46	29
			Variance	-	-	362	-	0	0	0	49	201	2	28	3490	71	105
			95% C.I.	-	-	45	-	0	1	0	17	34	4	12	140	20	24
			Density (#/mi)	-	-	262.8	-	0	179	99.5	183.4	386.6	56.0	133	884	263	133
			Variance (#/mi)	-	-	2,377.50	-	0	38	0.00	1,394.54	5,612.17	89.43	1,122	67,341	2,312	2,207
			95% C.I. (#/mi)	-	-	115.3	-	0	15	0.0	88.3	177.1	22.4	79	614	114	111
			Density (#/100ft^2)	-	-	0.13	-	0.00	0.26	0.06	0.19	0.53	0.04	0.14	1.03	0.50	0.20
			Variance (#/100ft^2)	-	-	0.0006	-	0.0000	0.0001	0.0000	0.0015	0.0104	0.00005	0.0012	0.0909	0.0085	0.0050
			95% C.I. (#/100ft^2)	-	-	0.06	-	0.00	0.02	0.00	0.09	0.24	0.02	0.08	0.71	0.22	0.17
2012	Juv+	Flatw aters	# Units Sampled	8	7	8	0	8	8	8	8	8	8	8	8	7	8
	≥10cm		Abundance	2	0	50	-	0	0	0	24	21	24	30	108	89	88
			Variance	4	0	160	-	0	0	0	40	19	71	141	616	122	454
			95% C.I.	4	0	30	-	0	0	0	15	10	20	28	59	27	50
			Density (#/mi)	6.3	0.0	194.3	-	0	0	0	158	176	127.6	107	671	609	1,012
			Variance (#/mi)	25.34	0.00	2,417	-	0	0	0	1,762	1,349	1,982.58	1,808	23,775	5,754	60,277
			95% C.I. (#/mi)	11.9	0.0	116.2	-	0	0	0	99	87	105.3	101	365	186	581
			Density (#/100ft^2)	0.01	0.00	0.14	-	0.00	0.00	0.00	0.21	0.33	0.10	0.13	1.23	1.50	2.52
			Variance (#/100ft^2)	0.0000	0.0000	0.0013	-	0.0000	0.0000	0.0000	0.0031	0.0047	0.0011	0.0025	0.0804	0.0348	0.3743
			95% C.I. (#/100ft^2)	0.01	0.00	0.08	-	0.00	0.00	0.00	0.13	0.16	0.08	0.12	0.67	0.46	1.45
2012	Juv+	Riffles	# Units Sampled	8	8	8	0	8	8	8	4	8	8	8	8	6	8
	≥10cm		Abundance	5	41	41	-	0	4	23	42	5	21	36	41	73	52
			Variance	7	73	56	-	0	14	5	100	15	23	111	133	294	134
			95% C.I.	6	20	18	-	0	9	5	32	9	11	25	27	44	27
			Density (#/mi)	15.8	124.3	182.9	-	0	19	198	669	54	150.7	230	355	440	474
			Variance (#/mi)	85.95	686.49	1,131.21	-	0	337	341	25,324	1,735	1,164.12	4,607	9,858	10,796	11,236
			95% C.I. (#/mi)	21.9	62.0	79.5	-	0	45	44	506	98	80.7	160	235	267	251
			Density (#/100ft^2)	0.01	0.10	0.15	-	0.00	0.03	0.15	0.97	0.11	0.13	0.28	0.62	1.17	1.38
			Variance (#/100ft^2)	0.0001	0.0004	0.0008	-	0.0000	0.0010	0.0002	0.0529	0.0068	0.0009	0.0066	0.0298	0.0768	0.0954
			95% C.I. (#/100ft^2)	0.02	0.05	0.07	-	0.00	0.08	0.03	0.73	0.19	0.07	0.19	0.41	0.71	0.73
2012	Juv+	All	# Units Sampled	16	15	24	0	24	23	24	20	24	24	24	24	21	24
	≥10cm	Habitats	Abundance	7	41	193	-	0	15	37	100	99	54	87	351	207	169
			Variance	11	73	577	-	0	14	5	189	235	97	280	4239	487	692
			95% C.I.	8	22	50	-	0	8	4	29	32	20	36	141	49	57
			Density (#/mi)	7.4	40.5	222.1	-	0	36	75	250	247	111	146	694	427	407
			Variance (#/mi)	12.28	72.83	763.82	-	0	80	19	1,177	1,458	403	798	16,628	2,066	4,036
			95% C.I. (#/mi)	9.0	21.9	57.5	-	0	19	9	72	79	42	61	279	100	137
			Density (#/100ft^2)	0.01	0.023	0.139	-	0.00	0.06	0.052	0.30	0.40	0.09	0.17	1.00	0.98	0.77
			Variance (#/100ft^2)	0.0000	0.00002	0.0003	-	0.0000	0.0002	0.0000	0.0017	0.0038	0.0002	0.0011	0.0345	0.0108	0.0146
			95% C.I. (#/100ft^2)	0.01	0.013	0.04	-	0.00	0.03	0.006	0.09	0.13	0.03	0.07	0.40	0.23	0.26

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2012	All *O. mykiss*	All Habitats	# Units Sampled	16	15	24	0	24	23	24	20	24	24	24	24	21	24
			Abundance	7	91	1036	-	0	41	244	511	946	475	603	844	1009	509
			Variance	11	265	11234	-	0	47	297	2139	6314	6466	4473	8791	6780	1745
			95% C.I.	8	42	220	-	0	14	36	98	165	167	144	203	181	90
			Density (#/mi)	7.4	91	1,191	-	0	99	499	1,274	2,357	971	1,018	1,672	2,079	1,229
			Variance (#/mi)	12	264	14,860	-	0	273	1,246	13,321	39,164	26,955	12,744	34,482	28,750	10,174
			95% C.I. (#/mi)	9	42	254	-	0	34	73	244	412	341	244	401	373	218
			Density (#/100ft^2)	0.005	0.052	0.747	-	0.00	0.16	0.35	1.51	3.79	0.75	1.17	2.41	4.75	2.34
			Variance (#/100ft^2)	0.00001	0.00009	0.0059	-	0.0000	0.0007	0.0006	0.0187	0.1013	0.0162	0.0168	0.0716	0.1501	0.0369
			95% C.I. (#/100ft^2)	0.006	0.024	0.159	-	0.00	0.06	0.05	0.29	0.66	0.26	0.28	0.58	0.85	0.41
2011	Fry <10cm	Pools	# Units Sampled	7	7	8	8	8	7	8	8	8	8	8	7	8	0
			Abundance	0	0	0	0	0	0	2	28	64	6	23	275	127	-
			Variance	0	0	0	0	0	0	1	89	190	4	35	2273	106	-
			95% C.I.	0	0	0	0	0	0	2	22	33	4	14	117	24	-
			Density (#/mi)	0	0	0.0	0	0	0	12	146.8	337.9	36.2	143	1,209	725	-
			Variance (#/mi)	0	0	0.00	0	0	0	51	2,521.44	5,311.35	140.53	1,397	43,853	3,456	-
			95% C.I. (#/mi)	0	0	0.0	0	0	0	17	118.7	172.3	28.0	88	512	139	-
			Density (#/100ft^2)	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.14	0.42	0.02	0.10	1.03	1.13	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0022	0.0082	0.00006	0.0007	0.0319	0.0083	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.11	0.21	0.02	0.06	0.44	0.22	-
2011	Fry <10cm	All Habitats	# Units Sampled	8	8	8	8	8	8	8	8	8	8	8	8	7	0
			Abundance	0	0	2	0	0	2	24	52	42	25	51	180	148	-
			Variance	0	0	0	0	0	1	459	214	55	8	299	1373	1023	-
			95% C.I.	0	0	0	0	0	3	51	35	18	7	41	88	78	-
			Density (#/mi)	0.0	0.0	7.3	0	0	11	103	345	352	129.1	183	1,117	1,015	-
			Variance (#/mi)	0.00	0.00	0	0	0	57	8,528	9,553	3,853	233.24	3,826	52,962	48,207	-
			95% C.I. (#/mi)	0.0	0.0	0.0	0	0	18	218	231	147	36.1	146	544	537	-
			Density (#/100ft^2)	0.00	0.00	0.004	0.00	0.00	0.01	0.07	0.36	0.50	0.08	0.17	1.63	1.75	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	0.0000	0.0000	0.0001	0.0035	0.0105	0.0078	0.0001	0.0034	0.1124	0.1429	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	0.00	0.00	0.02	0.14	0.24	0.21	0.02	0.14	0.79	0.93	-
2011	Fry <10cm	Riffles	# Units Sampled	8	8	8	8	8	8	8	8	8	8	8	8	8	0
			Abundance	0	0	0	2	0	4	94	18	28	10	18	60	80	-
			Variance	0	0	0	2	0	8	84	36	43	9	31	458	454	-
			95% C.I.	0	0	0	3	0	7	22	14	16	7	13	51	50	-
			Density (#/mi)	0.0	0.0	0.0	13	0	18	812	287	296	69.6	115	516	484	-
			Variance (#/mi)	0.00	0.00	0	82	0	208	6,304	9,183	4,991	443	1,280	33,832	16,656	-
			95% C.I. (#/mi)	0.0	0.0	0.0	21	0	34	188	227	167	49.8	85	435	305	-
			Density (#/100ft^2)	0.00	0.00	0.00	0.01	0.00	0.02	0.63	0.28	0.41	0.05	0.09	0.71	0.75	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	0.0000	0.0000	0.0004	0.0038	0.0089	0.0098	0.0002	0.0009	0.0637	0.0401	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	0.01	0.00	0.05	0.15	0.22	0.23	0.04	0.07	0.60	0.47	-

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2011	Fry	All	# Units Sampled	23	23	24	24	24	23	24	24	24	24	24	23	23	0
	<10cm	Habitats	Abundance	0	0	2	2	0	5	119	97	133	40	92	515	355	-
			Variance	0	0	0	2	0	10	544	340	288	21	364	4103	1582	-
			95% C.I.	0	0	0	3	0	7	49	40	37	9	40	134	83	-
			Density (#/mi)	0.0	0.0	2.2	4	0	20	245	243	473	82	154	1,020	730	-
			Variance (#/mi)	0.00	0.00	0.00	7	0	142	2,283	2,116	3,618	86	1,038	16,092	6,711	-
			95% C.I. (#/mi)	0.0	0.0	0.0	5	0	26	99	99	130	19	67	265	171	-
			Density (#/100ft^2)	0.00	0.00	0.00	0.00	0.00	0.03	0.16	0.24	0.61	0.05	0.13	1.11	1.17	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	0.0000	0.0000	0.0002	0.0009	0.0021	0.0061	0.0000	0.0007	0.0192	0.0172	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	0.00	0.00	0.03	0.06	0.10	0.17	0.01	0.06	0.29	0.27	-
2011	Juv	Pools	# Units Sampled	7	7	8	8	8	7	8	8	8	8	8	7	8	0
	10-20cm		Abundance	2	23	395	5	4	14	8	44	58	24	57	317	49	-
			Variance	0	3	551	0	7	1	38	87	394	46	51	7496	128	-
			95% C.I.	0	4	55	0	6	2	15	22	47	16	17	212	27	-
			Density (#/mi)	7	114	1,063.5	39	40	227	60.2	235.4	307.2	150.5	363	1,392	279	-
			Variance (#/mi)	0	67	3,999.92	0	699	189	1,906.60	2,459.95	11,008.03	1,787.93	2,034	144,641	4,186	-
			95% C.I. (#/mi)	0	20	149.6	0	63	34	103.3	117.3	248.1	100.0	107	931	153	-
			Density (#/100ft^2)	0.00	0.05	0.35	0.02	0.04	0.28	0.03	0.22	0.38	0.10	0.26	1.19	0.43	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0004	0.0000	0.0006	0.0003	0.0006	0.0022	0.0171	0.00074	0.0011	0.1053	0.0101	-
			95% C.I. (#/100ft^2)	0.00	0.01	0.05	0.00	0.06	0.04	0.06	0.11	0.31	0.06	0.08	0.79	0.24	-
2011	Juv	Flatwaters	# Units Sampled	8	8	8	8	8	8	8	8	8	8	8	8	7	0
	10-20cm		Abundance	7	5	9	13	17	9	3	81	36	31	66	134	25	-
			Variance	4	18	9	62	59	12	0	595	96	69	2388	1088	55	-
			95% C.I.	5	10	7	19	18	8	0	58	23	20	116	78	18	-
			Density (#/mi)	18.2	10.0	35.0	50	68	57	13	537	302	161.4	238	834	169	-
			Variance (#/mi)	30.84	80.86	143	905	910	509	0	26,517	6,743	1,905.13	30,596	41,962	2,602	-
			95% C.I. (#/mi)	13.1	21.3	28.3	71	71	53	0	385	194	103.2	414	484	125	-
			Density (#/100ft^2)	0.01	0.00	0.02	0.02	0.06	0.07	0.01	0.56	0.43	0.10	0.22	1.22	0.29	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	0.0002	0.0007	0.0008	0.0000	0.0290	0.0137	0.0007	0.0269	0.0891	0.0077	-
			95% C.I. (#/100ft^2)	0.01	0.01	0.02	0.03	0.06	0.07	0.00	0.40	0.28	0.06	0.39	0.71	0.21	-
2011	Juv	Riffles	# Units Sampled	8	8	8	8	8	8	8	8	8	8	8	8	8	0
	10-20cm		Abundance	0	7	5	0	8	4	71	35	33	5	69	63	28	-
			Variance	0	7	4	0	8	8	49	26	94	2	133	375	34	-
			95% C.I.	0	6	5	0	6	7	17	12	23	4	27	46	14	-
			Density (#/mi)	0.0	21.9	21.9	0	43	18	614	552	349	34.8	446	544	167	-
			Variance (#/mi)	0.00	69.25	75.96	0	249	208	3,674	6,597	10,854	112.28	5,494	27,731	1,241	-
			95% C.I. (#/mi)	0.0	19.7	20.6	0	37	34	143	192	246	25.1	175	394	83	-
			Density (#/100ft^2)	0.00	0.01	0.01	0.00	0.04	0.02	0.47	0.54	0.49	0.02	0.36	0.75	0.26	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	0.0000	0.0003	0.0004	0.0022	0.0064	0.0214	0.0001	0.0037	0.0522	0.0030	-
			95% C.I. (#/100ft^2)	0.00	0.01	0.01	0.00	0.04	0.05	0.11	0.19	0.35	0.02	0.14	0.54	0.13	-

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2011	Juv	All	# Units Sampled	23	23	24	24	24	23	24	24	24	24	24	23	23	0
	10-20cm	Habitats	Abundance	9	35	408	18	29	44	82	160	127	3	193	514	101	-
			Variance	4	28	564	62	73	49	87	708	584	0	2571	8959	217	-
			95% C.I.	4	11	49	16	19	15	19	57	52	1	105	197	31	-
			Density (#/mi)	9.5	34.6	480.2	33	104	168	169	398	449	16	326	1,019	208	-
			Variance (#/mi)	4.98	28.21	779.37	209	961	721	364	4,410	7,333	13	7,325	35,139	920	-
			95% C.I. (#/mi)	4.7	11.1	58.1	30	67	58	40	143	185	8	178	391	63	-
			Density (#/100ft^2)	0.01	0.015	0.205	0.01	0.10	0.22	0.108	0.39	0.58	0.01	0.27	1.11	0.33	-
			Variance (#/100ft^2)	0.0000	0.00001	0.0001	0.0000	0.0009	0.0012	0.0001	0.0043	0.0123	0.0000	0.0051	0.0420	0.0024	-
			95% C.I. (#/100ft^2)	0.00	0.005	0.02	0.01	0.06	0.08	0.025	0.14	0.24	0.01	0.15	0.43	0.10	-
2011	Adult	Pools	# Units Sampled	7	7	8	8	8	7	8	8	8	8	8	7	8	0
	>20cm		Abundance	0	43	100	1	4	6	12	0	4	3	11	46	0	-
			Variance	0	5	187	0	7	0	15	0	18	0	2	565	0	-
			95% C.I.	0	5	32	0	6	0	9	0	10	1	3	58	0	-
			Density (#/mi)	0	214	270.5	8	40	97	87.1	0.0	20.5	15.8	67	201	0	-
			Variance (#/mi)	0	121	1,356.64	0	699	0	747.32	0.00	497.93	12.78	84	10,903	0	-
			95% C.I. (#/mi)	0	27	87.1	0	63	0	64.6	0.0	52.8	8.5	22	255	0	-
			Density (#/100ft^2)	0.00	0.09	0.09	0.00	0.04	0.12	0.05	0.00	0.03	0.01	0.05	0.17	0.00	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0002	0.0000	0.0006	0.0000	0.0002	0.0000	0.0008	0.00001	0.0000	0.0079	0.0000	-
			95% C.I. (#/100ft^2)	0.00	0.01	0.03	0.00	0.06	0.00	0.04	0.00	0.07	0.01	0.02	0.22	0.00	-
2011	Adult	Flatwaters	# Units Sampled	8	8	8	8	8	8	8	8	8	8	8	8	7	0
	>20cm		Abundance	0	5	6	0	0	4	12	0	0	9	6	1	0	-
			Variance	0	5	24	0	0	2	61	0	0	2	11	0	0	-
			95% C.I.	0	5	12	0	0	4	18	0	0	3	8	0	0	-
			Density (#/mi)	0.0	10.0	21.9	0	0	23	52	0	0	46.1	23	6	0	-
			Variance (#/mi)	0.00	23.10	358	0	0	97	1,133	0	0	58.31	146	0	0	-
			95% C.I. (#/mi)	0.0	11.4	44.7	0	0	23	80	0	0	18.1	29	0	0	-
			Density (#/100ft^2)	0.00	0.00	0.01	0.00	0.00	0.03	0.03	0.00	0.00	0.03	0.02	0.01	0.00	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0001	0.0000	0.0000	0.0001	0.0005	0.0000	0.0000	0.0000	0.0001	0.0000	0.0000	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.03	0.00	0.00	0.03	0.05	0.00	0.00	0.01	0.03	0.00	0.00	-
2011	Adult	Riffles	# Units Sampled	8	8	8	8	8	8	8	8	8	8	8	8	8	0
	>20cm		Abundance	0	0	0	0	0	0	18	1	0	5	4	0	0	-
			Variance	0	0	0	0	0	0	3	1	0	2	3	0	0	-
			95% C.I.	0	0	0	0	0	0	4	2	0	4	4	0	0	-
			Density (#/mi)	0.0	0.0	0.0	0	0	0	156	22	0	34.8	29	0	0	-
			Variance (#/mi)	0.00	0.00	0.00	0	0	0	238	133	0	112.28	123	0	0	-
			95% C.I. (#/mi)	0.0	0.0	0.0	0	0	0	37	27	0	25.1	26	0	0	-
			Density (#/100ft^2)	0.00	0.00	0.00	0.00	0.00	0.00	0.12	0.02	0.00	0.02	0.02	0.00	0.00	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0001	0.0001	0.0000	0.0001	0.0001	0.0000	0.0000	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.03	0.00	0.02	0.02	0.00	0.00	-

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2011	Adult	All	# Units Sampled	23	23	24	24	24	23	24	24	24	24	24	23	23	0
	>20cm	Habitats	Abundance	0	25	106	1	4	10	42	1	4	16	21	47	0	-
			Variance	0	7	210	0	7	2	79	1	18	5	16	565	0	-
			95% C.I.	0	6	30	0	6	3	18	2	9	4	8	50	0	-
			Density (#/mi)	0.0	24.8	124.6	2	15	36	87	3	14	33	36	93	0	-
			Variance (#/mi)	0.00	7.38	290.87	0	97	33	331	3	224	19	47	2,216	0	-
			95% C.I. (#/mi)	0.0	5.7	35.5	0	21	12	38	4	32	9	14	98	0	-
			Density (#/100ft^2)	0.00	0.011	0.053	0.00	0.01	0.05	0.056	0.00	0.02	0.02	0.03	0.10	0.00	-
			Variance (#/100ft^2)	0.0000	0.00000	0.0001	0.0000	0.0001	0.0001	0.0001	0.0000	0.0004	0.0000	0.0000	0.0026	0.0000	-
			95% C.I. (#/100ft^2)	0.00	0.003	0.02	0.00	0.02	0.02	0.024	0.00	0.04	0.01	0.01	0.11	0.00	-
2011	Juv+	Pools	# Units Sampled	7	7	8	8	8	7	8	8	8	8	8	7	8	0
	≥10cm		Abundance	2	43	495	6	8	20	21	44	62	25	68	365	49	-
			Variance	0	5	737	0	15	1	72	87	412	42	53	8754	128	-
			95% C.I.	0	5	64	0	9	2	20	22	48	15	17	229	27	-
			Density (#/mi)	7	214	1,334.0	47	80	325	149.2	235.4	327.7	156.8	430	1,605	279	-
			Variance (#/mi)	0	121	5,356.56	0	1,399	189	3,630.55	2,459.95	11,505.96	1,659.25	2,119	168,908	4,186	-
			95% C.I. (#/mi)	0	27	173.1	0	88	34	142.5	117.3	253.6	96.3	109	1,006	153	-
			Density (#/100ft^2)	0.00	0.09	0.44	0.02	0.07	0.40	0.08	0.22	0.41	0.10	0.31	1.37	0.43	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0006	0.0000	0.0011	0.0003	0.0011	0.0022	0.0179	0.00069	0.0011	0.1229	0.0101	-
			95% C.I. (#/100ft^2)	0.00	0.01	0.06	0.00	0.08	0.04	0.08	0.11	0.32	0.06	0.08	0.86	0.24	-
2011	Juv+	Flatwaters	# Units Sampled	8	8	8	8	8	8	8	8	8	8	8	8	7	0
	≥10cm		Abundance	7	10	15	13	17	12	15	81	36	39	73	135	25	-
			Variance	4	24	34	62	59	14	61	595	96	71	2399	1088	55	-
			95% C.I.	5	11	14	19	18	9	18	58	23	20	116	78	18	-
			Density (#/mi)	18.2	20.0	58.3	50	68	80	65	537	302	207.5	261	841	169	-
			Variance (#/mi)	30.84	103.96	516	905	910	605	1,133	26,517	6,743	1,963.45	30,742	41,962	2,602	-
			95% C.I. (#/mi)	13.1	24.1	53.7	71	71	58	80	385	194	104.8	415	484	125	-
			Density (#/100ft^2)	0.01	0.01	0.03	0.02	0.06	0.10	0.04	0.56	0.43	0.13	0.24	1.22	0.29	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0002	0.0002	0.0007	0.0009	0.0005	0.0290	0.0137	0.0008	0.0270	0.0891	0.0077	-
			95% C.I. (#/100ft^2)	0.01	0.01	0.03	0.03	0.06	0.07	0.05	0.40	0.28	0.07	0.39	0.71	0.21	-
2011	Juv+	Riffles	# Units Sampled	8	8	8	8	8	8	8	8	8	8	8	8	8	0
	≥10cm		Abundance	0	7	5	0	8	21	89	36	33	10	74	63	28	-
			Variance	0	7	4	0	8	37	52	27	94	4	135	375	34	-
			95% C.I.	0	6	5	0	6	14	17	12	23	5	28	46	14	-
			Density (#/mi)	0.0	21.9	21.9	0	43	106	771	574	349	69.6	475	544	167	-
			Variance (#/mi)	0.00	69.25	75.96	0	249	919	3,912	6,729	10,854	224.55	5,617	27,731	1,241	-
			95% C.I. (#/mi)	0.0	19.7	20.6	0	37	72	148	194	246	35.4	177	394	83	-
			Density (#/100ft^2)	0.00	0.01	0.01	0.00	0.04	0.14	0.60	0.57	0.49	0.05	0.39	0.75	0.26	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	0.0000	0.0003	0.0017	0.0023	0.0065	0.0214	0.0001	0.0037	0.0522	0.0030	-
			95% C.I. (#/100ft^2)	0.00	0.01	0.01	0.00	0.04	0.10	0.11	0.19	0.35	0.03	0.14	0.54	0.13	-

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2011	Juv+	All	# Units Sampled	23	23	24	24	24	23	24	24	24	24	24	23	23	0
	≥10cm	Habitats	Abundance	9	60	514	19	33	54	125	161	131	76	214	561	101	-
			Variance	4	36	774	62	81	52	166	709	602	121	2587	9524	217	-
			95% C.I.	4	12	58	16	19	16	27	58	53	23	106	204	31	-
			Density (#/mi)	9.5	59.4	604.8	35	119	204	256	401	462	155	362	1,111	208	-
			Variance (#/mi)	4.98	35.59	1,070.24	209	1,058	754	695	4,413	7,556	505	7,372	37,355	920	-
			95% C.I. (#/mi)	4.7	12.4	68.0	30	70	60	55	144	188	47	179	403	63	-
			Density (#/100ft^2)	0.01	0.026	0.259	0.02	0.11	0.27	0.164	0.40	0.60	0.10	0.30	1.21	0.33	-
			Variance (#/100ft^2)	0.0000	0.00001	0.0002	0.0000	0.0010	0.0013	0.0003	0.0043	0.0126	0.0002	0.0052	0.0446	0.0024	-
			95% C.I. (#/100ft^2)	0.00	0.005	0.03	0.01	0.07	0.08	0.035	0.14	0.24	0.03	0.15	0.44	0.10	-
2011	All *O. mykiss*	All Habitats	# Units Sampled	23	23	24	24	24	23	24	24	24	24	24	23	23	0
			Abundance	9	60	516	21	33	59	244	258	264	116	306	1076	455	-
			Variance	4	36	774	64	81	61	710	1049	890	142	2952	13626	1799	-
			95% C.I.	4	12	58	17	19	17	55	70	64	25	113	243	88	-
			Density (#/mi)	9.5	59	607	39	119	224	500	645	935	236	516	2,131	938	-
			Variance (#/mi)	5	36	1,070	215	1,058	896	2,978	6,529	11,175	591	8,410	53,448	7,631	-
			95% C.I. (#/mi)	5	12	68	31	70	65	113	175	228	51	191	482	182	-
			Density (#/100ft^2)	0.005	0.026	0.260	0.02	0.11	0.29	0.32	0.64	1.21	0.15	0.43	2.33	1.50	-
			Variance (#/100ft^2)	0.00000	0.00001	0.0002	0.0000	0.0010	0.0015	0.0012	0.0063	0.0187	0.0003	0.0059	0.0638	0.0195	-
			95% C.I. (#/100ft^2)	0.002	0.005	0.029	0.01	0.07	0.09	0.07	0.17	0.30	0.03	0.16	0.53	0.29	-
2010	Fry	Pools	# Units Sampled	6	0	6	5	8	8	8	8	8	8	11	0	8	0
	<10cm		Abundance	0	0	132	5	0	4	0	20	124	2	35	-	47	-
			Variance	0	0	30	0	0	0	0	264	394	1	18	-	246	-
			95% C.I.	0	0	14	1	0	0	0	38	47	2	9	-	37	-
			Density (#/mi)	0.0	0.0	511.0	51.0	0	70	0.0	89.0	586.0	19.0	265	-	285	-
			Variance (#/mi)	0.00	0.00	454.00	25.00	0	0	0.00	5,215.00	8,834.00	66.00	1,020	-	9,159	-
			95% C.I. (#/mi)	0.0	0.0	55.0	14.0	0	0	0.0	171.0	222.0	19.0	71	-	226	-
			Density (#/100ft^2)	0.00	0.00	0.31	0.02	0.00	0.10	0.00	0.10	1.06	0.01	0.17	-	0.59	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0002	0.0000	0.0000	0.0000	0.0000	0.0068	0.0287	0.00002	0.0004	-	0.0397	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.03	0.01	0.00	0.00	0.00	0.19	0.40	0.01	0.04	-	0.47	-
2010	Fry	Flatwaters	# Units Sampled	8	8	8	6	0	0	8	8	8	8	8	0	8	0
	<10cm		Abundance	0	0	143	0	-	-	101	55	142	15	216	-	81	-
			Variance	0	0	10417	0	-	-	997	151	431	69	1378	-	136	-
			95% C.I.	0	0	241	0	-	-	75	29	49	20	88	-	28	-
			Density (#/mi)	0.0	0.0	297.0	0.0	-	-	423	538	954	57.0	920	-	523	-
			Variance (#/mi)	0.00	0.00	44,838.00	0.00	-	-	17,568	14,320	19,380	984.00	25,057	-	5,608	-
			95% C.I. (#/mi)	0.0	0.0	501.0	0.0	-	-	313	283	329	74.0	374	-	177	-
			Density (#/100ft^2)	0.00	0.00	0.21	0.00	-	-	0.26	0.71	1.84	0.03	0.72	-	1.22	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0234	0.0000	-	-	0.0066	0.0248	0.0724	0.0003	0.0155	-	0.0307	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.36	0.00	-	-	0.19	0.37	0.64	0.04	0.29	-	0.41	-

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2010	Fry	Riffles	# Units Sampled	8	8	8	8	0	0	8	8	8	8	8	0	8	0
	<10cm		Abundance	0	0	56	5	-	-	111	84	76	11	109	-	58	-
			Variance	0	0	331	2	-	-	418	105	17	6	387	-	60	-
			95% C.I.	0	0	43	3	-	-	48	24	10	6	47	-	18	-
			Density (#/mi)	0.0	0.0	288.0	38.0	-	-	1,110	1,256	1,488	125.0	1,498	-	823	-
			Variance (#/mi)	0.00	0.00	8,825.00	113.00	-	-	42,131	23,545	6,639	780.00	73,614	-	12,118	-
			95% C.I. (#/mi)	0.0	0.0	222.0	25.0	-	-	485	363	193	66.0	642	-	260	-
			Density (#/100ft^2)	0.00	0.00	0.22	0.02	-	-	0.77	1.82	3.17	0.08	0.77	-	1.56	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0050	0.00003	-	-	0.0203	0.0492	0.0301	0.0003	0.0194	-	0.0435	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.17	0.01	-	-	0.30	0.52	0.41	0.04	0.33	-	0.49	-
2010	Fry	All	# Units Sampled	22	16	22	19	0	0	24	24	24	24	27	0	24	0
	<10cm	Habitats	Abundance	0	0	330	10	-	-	211	159	342	28	360	-	186	-
			Variance	0	0	10778	2	-	-	1415	520	842	76	1783	-	442	-
			95% C.I.	0	0	217	3	-	-	78	47	60	18	87	-	44	-
			Density (#/mi)	0.0	0.0	354.0	17.0	-	-	396	403	831	61	817	-	477	-
			Variance (#/mi)	0.00	0.00	12,388.00	6.00	-	-	4,973	3,341	4,983	345	9,211	-	2,907	-
			95% C.I. (#/mi)	0.0	0.0	233.0	5.0	-	-	147	120	147	39	198	-	112	-
			Density (#/100ft^2)	0.00	0.00	0.25	0.01	-	-	0.24	0.49	1.57	0.04	0.55	-	1.02	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0060	0.0000	-	-	0.0018	0.0050	0.0177	0.0001	0.0042	-	0.0133	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.16	0.003	-	-	0.09	0.15	0.28	0.02	0.13	-	0.24	-
2010	Juv+	Pools	# Units Sampled	6	0	6	5	8	8	8	8	8	8	11	0	8	0
	$\geq$10cm		Abundance	0	0	152	19	0	47	16	30	100	9	65	-	16	-
			Variance	0	0	12	1	0	11	30	110	292	9	134	-	34	-
			95% C.I.	0	0	9	2	0	8	13	25	40	7	26	-	14	-
			Density (#/mi)	8.3	0.0	589.0	196.0	0	822	82.0	133.0	472.0	77.0	488	-	95	-
			Variance (#/mi)	0.00	0.00	180.00	85.00	0	3,318	780.00	2,182.00	6,539.00	669.00	7,608	-	1,273	-
			95% C.I. (#/mi)	0.0	0.0	34.0	26.0	0	136	66.0	110.0	191.0	61.0	194	-	84	-
			Density (#/100ft^2)	0.00	0.00	0.36	0.08	0.00	1.19	0.04	0.15	0.85	0.05	0.31	-	0.20	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0001	0.0000	0.0000	0.0069	0.0002	0.0028	0.0213	0.00030	0.0030	-	0.0055	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.02	0.01	0.00	0.20	0.04	0.13	0.34	0.04	0.12	-	0.18	-
2010	Juv+	Flatwaters	# Units Sampled	8	8	8	6	0	0	8	8	8	8	8	0	8	0
	$\geq$10cm		Abundance	3	0	244	0	-	-	133	26	28	55	75	-	54	-
			Variance	7	0	4562	0	-	-	1231	12	49	657	394	-	114	-
			95% C.I.	6	0	160	0	-	-	83	8	17	61	47	-	25	-
			Density (#/mi)	7.0	0.0	506.0	0.0	-	-	559	253	191	207.0	319	-	349	-
			Variance (#/mi)	37.00	0.00	19,635.00	0.00	-	-	21,690	1,102	2,207	9,377.00	7,172	-	4,697	-
			95% C.I. (#/mi)	14.0	0.0	331.0	0.0	-	-	348	78	111	229.0	200	-	162	-
			Density (#/100ft^2)	0.01	0.00	0.37	0.00	-	-	0.34	0.33	0.37	0.12	0.25	-	0.82	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0103	0.0000	-	-	0.0081	0.0019	0.0082	0.0032	0.0044	-	0.0257	-
			95% C.I. (#/100ft^2)	0.01	0.00	0.24	0.00	-	-	0.21	0.10	0.21	0.13	0.16	-	0.38	-

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2010	Juv+	Riffles	# Units Sampled	8	8	8	8	0	0	8	8	8	8	8	0	8	0
	≥10cm		Abundance	2	15	165	0	-	-	101	29	12	19	21	-	7	-
			Variance	2	32	2396	0	-	-	342	23	0	0	10	-	1	-
			95% C.I.	3	13	116	0	-	-	44	11	2	0	8	-	3	-
			Density (#/mi)	11.0	59.0	854.0	0.0	-	-	1,009	431	240	217.0	288	-	106	-
			Variance (#/mi)	60.00	490.00	63,952.00	0.00	-	-	34,442	5,143	194	0.00	1,969	-	260	-
			95% C.I. (#/mi)	18.0	52.0	598.0	0.0	-	-	439	170	33	0.0	105	-	38	-
			Density (#/100ft^2)	0.01	0.04	0.65	0.00	-	-	0.70	0.62	0.51	0.14	0.15	-	0.20	-
			Variance (#/100ft^2)	0.0001	0.0002	0.0365	0.00000	-	-	0.0166	0.0107	0.0009	0.0000	0.0005	-	0.0009	-
			95% C.I. (#/100ft^2)	0.02	0.03	0.45	0.00	-	-	0.30	0.25	0.07	0.00	0.05	-	0.07	-
2010	Juv+	All	# Units Sampled	22	16	22	19	0	0	24	24	24	24	27	0	24	0
	≥10cm	Habitats	Abundance	5	15	561	19	-	-	250	85	140	83	161	-	77	-
			Variance	9	32	6970	1	-	-	1603	145	341	666	539	-	149	-
			95% C.I.	6	12	175	2	-	-	83	25	38	54	48	-	25	-
			Density (#/mi)	6.0	14.7	601.0	32.5	-	-	468	215	341	176	1,208	-	198	-
			Variance (#/mi)	13.00	30.00	8,010.80	2.30	-	-	5,631	932	2,020	3,031	30,499	-	982	-
			95% C.I. (#/mi)	7.6	11.8	187.3	3.2	-	-	156	63	93	114	360	-	65	-
			Density (#/100ft^2)	0.01	0.01	0.42	0.02	-	-	0.28	0.26	0.64	0.11	0.25	-	0.42	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0040	0.0000	-	-	0.0020	0.0014	0.0072	0.0011	0.0013	-	0.0045	-
			95% C.I. (#/100ft^2)	0.01	0.01	0.13	0.002	-	-	0.09	0.08	0.18	0.07	0.07	-	0.14	-
2010	All *O. mykiss*	All Habitats	# Units Sampled	22	16	22	19	-	-	24	24	24	24	27	0	24	0
			Abundance	5	15	891	29	-	-	461	244	482	111	520	-	263	-
			Variance	9	32	17748	3	-	-	3018	664	1183	742	2322	-	592	-
			95% C.I.	6	12	279	4	-	-	114	54	72	57	99	-	51	-
			Density (#/mi)	6.0	14.7	955.4	49.4	-	-	864	618	1,172	237	3,912	-	675	-
			Variance (#/mi)	13.00	30.00	20,398.40	8.60	-	-	10,604	4,274	7,002	3,376	131,360	-	3,890	-
			95% C.I. (#/mi)	7.6	11.8	298.9	6.2	-	-	214	136	174	121	748	-	130	-
			Density (#/100ft^2)	0.01	0.01	0.66	0.02	-	-	0.51	0.76	2.21	0.14	0.80	-	1.44	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0100	0.0000	-	-	0.0037	0.0064	0.0249	0.0012	0.0055	-	0.0178	-
			95% C.I. (#/100ft^2)	0.01	0.01	0.21	0.003	-	-	0.13	0.17	0.33	0.07	0.15	-	0.28	-
2009	Fry	Pools	# Units Sampled	0	0	6	0	0	0	8	7	8	0	0	0	0	0
	<10cm		Abundance	-	-	0	-	-	-	0	79	206	-	-	-	-	-
			Variance	-	-	0	-	-	-	0	682	1244	-	-	-	-	-
			95% C.I.	-	-	0	-	-	-	0	64	83	-	-	-	-	-
			Density (#/mi)	-	-	0.0	-	-	-	0.0	270.0	976.0	-	-	-	-	-
			Variance (#/mi)	-	-	0	-	-	-	0	8073	27902	-	-	-	-	-
			95% C.I. (#/mi)	-	-	0.0	-	-	-	0.0	220.0	395.0	-	-	-	-	-
			Density (#/100ft^2)	-	-	0.00	-	-	-	0.00	0.37	1.95	-	-	-	-	-
			Variance (#/100ft^2)	-	-	0.0000	-	-	-	0.0000	0.0148	0.1109	-	-	-	-	-
			95% C.I. (#/100ft^2)	-	-	0.00	-	-	-	0.00	0.30	0.79	-	-	-	-	-

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2009	Fry	Flatwaters	# Units Sampled	0	0	8	0	0	0	8	0	0	0	0	0	0	0
	<10cm		Abundance	-	-	0	-	-	-	0	-	-	-	-	-	-	-
			Variance	-	-	0	-	-	-	0	-	-	-	-	-	-	-
			95% C.I.	-	-	0	-	-	-	0	-	-	-	-	-	-	-
			Density (#/mi)	-	-	0.0	-	-	-	0.0	-	-	-	-	-	-	-
			Variance (#/mi)	-	-	0	-	-	-	0	-	-	-	-	-	-	-
			95% C.I. (#/mi)	-	-	0.0	-	-	-	0.0	-	-	-	-	-	-	-
			Density (#/100ft^2)	-	-	0.00	-	-	-	0.00	-	-	-	-	-	-	-
			Variance (#/100ft^2)	-	-	0.0000	-	-	-	0.0000	-	-	-	-	-	-	-
			95% C.I. (#/100ft^2)	-	-	0.00	-	-	-	0.00	-	-	-	-	-	-	-
2009	Juv+	Pools	# Units Sampled	0	0	6	0	0	0	8	7	8	0	0	0	0	0
	≥10cm		Abundance	-	-	134	-	-	-	0	60	88	-	-	-	-	-
			Variance	-	-	8	-	-	-	0	25	200	-	-	-	-	-
			95% C.I.	-	-	7	-	-	-	0	12	33	-	-	-	-	-
			Density (#/mi)	-	-	533.0	-	-	-	0.0	206	419	-	-	-	-	-
			Variance (#/mi)	-	-	130	-	-	-	0	292	4,483	-	-	-	-	-
			95% C.I. (#/mi)	-	-	29.0	-	-	-	0.0	42	158	-	-	-	-	-
			Density (#/100ft^2)	-	-	0.21	-	-	-	0.00	0.28	0.84	-	-	-	-	-
			Variance (#/100ft^2)	-	-	0.00002	-	-	-	0.0000	0.0005	0.0178	-	-	-	-	-
			95% C.I. (#/100ft^2)	-	-	0.01	-	-	-	0.00	0.06	0.32	-	-	-	-	-
2009	Juv+	Flatwaters	# Units Sampled	0	0	8	0	0	0	8	0	0	0	0	0	0	0
	≥10cm		Abundance	-	-	17	-	-	-	0	-	-	-	-	-	-	-
			Variance	-	-	197	-	-	-	0	-	-	-	-	-	-	-
			95% C.I.	-	-	33	-	-	-	0	-	-	-	-	-	-	-
			Density (#/mi)	-	-	36.0	-	-	-	0	-	-	-	-	-	-	-
			Variance (#/mi)	-	-	916	-	-	-	0	-	-	-	-	-	-	-
			95% C.I. (#/mi)	-	-	72.0	-	-	-	0	-	-	-	-	-	-	-
			Density (#/100ft^2)	-	-	0.03	-	-	-	0.00	-	-	-	-	-	-	-
			Variance (#/100ft^2)	-	-	0.0060	-	-	-	0.0000	-	-	-	-	-	-	-
			95% C.I. (#/100ft^2)	-	-	0.06	-	-	-	0.00	-	-	-	-	-	-	-
2009	All *O. mykiss*	Pools	# Units Sampled	0	0	6	0	0	0	8	7	8	0	0	0	0	0
			Abundance	-	-	134	-	-	-	0	139	295	-	-	-	-	-
			Variance	-	-	8	-	-	-	0	707	1444	-	-	-	-	-
			95% C.I.	-	-	7	-	-	-	0	65	90	-	-	-	-	-
			Density (#/mi)	-	-	533.0	-	-	-	0	477	1,395	-	-	-	-	-
			Variance (#/mi)	-	-	130	-	-	-	0	8,365	32,384	-	-	-	-	-
			95% C.I. (#/mi)	-	-	29.0	-	-	-	0	224	426	-	-	-	-	-
			Density (#/100ft^2)	-	-	0.21	-	-	-	0.00	0.64	2.78	-	-	-	-	-
			Variance (#/100ft^2)	-	-	0.00002	-	-	-	0.0000	0.0153	0.1287	-	-	-	-	-
			95% C.I. (#/100ft^2)	-	-	0.01	-	-	-	0.00	0.30	0.85	-	-	-	-	-

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2009	All *O. mykiss*	Flatwaters	# Units Sampled	0	0	8	0	0	0	8	0	0	0	0	0	0	0
			Abundance	-	-	17	-	-	-	0	-	-	-	-	-	-	-
			Variance	-	-	197	-	-	-	0	-	-	-	-	-	-	-
			95% C.I.	-	-	33	-	-	-	0	-	-	-	-	-	-	-
			Density (#/mi)	-	-	36.0	-	-	-	0	-	-	-	-	-	-	-
			Variance (#/mi)	-	-	916	-	-	-	0	-	-	-	-	-	-	-
			95% C.I. (#/mi)	-	-	72.0	-	-	-	0	-	-	-	-	-	-	-
			Density (#/100ft^2)	-	-	0.03	-	-	-	0.00	-	-	-	-	-	-	
			Variance (#/100ft^2)	-	-	0.0060	-	-	-	0.0000	-	-	-	-	-	-	-
			95% C.I. (#/100ft^2)	-	-	0.06	-	-	-	0.00	-	-	-	-	-	-	-
2008	Fry <10cm	Pools	# Units Sampled	7	6	5	0	0	0	8	8	8	8	10	9	8	0
			Abundance	0	0	28	-	-	-	8	69	70	0	1	38	149	-
			Variance	0	0	23	-	-	-	19	152	259	0	0	332	393	-
			95% C.I.	0	0	13	-	-	-	10	29	38	0	0	42	47	-
			Density (#/mi)	0	0	108.3	-	-	-	40.9	237.0	331.5	0	18	186	872	-
			Variance (#/mi)	0	0	350	-	-	-	502	1793	5813	0	0	7,814	13,423	-
			95% C.I. (#/mi)	0	0	52.0	-	-	-	53.0	100.1	180.3	0	0	204	274	-
			Density (#/100ft^2)	0.00	0.00	0.07	-	-	-	0.03	0.33	0.58	0.00	0.02	0.21	1.72	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0001	-	-	-	0.0002	0.0035	0.0179	0.0000	0.0000	0.0099	0.0522	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.03	-	-	-	0.04	0.14	0.32	0.00	0.00	0.23	0.54	-
2008	Fry <10cm	Flatwaters	# Units Sampled	8	8	8	0	0	0	8	0	0	0	0	0	0	0
			Abundance	0	0	54	-	-	-	33	-	-	-	-	-	-	-
			Variance	0	0	4104	-	-	-	829	-	-	-	-	-	-	-
			95% C.I.	0	0	151	-	-	-	68	-	-	-	-	-	-	-
			Density (#/mi)	0	0	116.5	-	-	-	136.4	-	-	-	-	-	-	-
			Variance (#/mi)	0	0	19108	-	-	-	14599	-	-	-	-	-	-	-
			95% C.I. (#/mi)	0	0	326.9	-	-	-	285.7	-	-	-	-	-	-	-
			Density (#/100ft^2)	0.00	0.00	0.11	-	-	-	0.11	-	-	-	-	-	-	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0158	-	-	-	0.0092	-	-	-	-	-	-	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.30	-	-	-	0.23	-	-	-	-	-	-	-
2008	Juv+ $\geq$10cm	Pools	# Units Sampled	7	6	5	0	0	0	8	8	8	8	10	9	8	0
			Abundance	0	0	806	-	-	-	97	100	208	5	12	151	52	-
			Variance	0	0	28459	-	-	-	769	1259	3013	1	4	3689	215	-
			95% C.I.	0	0	468	-	-	-	66	84	130	3	5	140	35	-
			Density (#/mi)	8	0	3,163.6	-	-	-	496.1	345	986	39	200	733	305	-
			Variance (#/mi)	0	0	438,569	-	-	-	20,090	14,897	67,571	83	1,085	86,871	7,337	-
			95% C.I. (#/mi)	0	0	1,838.7	-	-	-	335.2	289	615	22	75	680	203	-
			Density (#/100ft^2)	0.00	0.00	2.02	-	-	-	0.34	0.48	1.73	0.03	0.26	0.83	0.60	-
			Variance (#/100ft^2)	0.0000	0.0000	0.17834	-	-	-	0.0097	0.0293	0.2078	0.0000	0.0019	0.1104	0.0286	-
			95% C.I. (#/100ft^2)	0.00	0.00	1.17	-	-	-	0.23	0.40	1.08	0.02	0.10	0.77	0.40	-

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2008	Juv+	Flatwaters	# Units Sampled	8	8	8	0	0	0	8	n/a	n/a	n/a	n/a	n/a	n/a	0
	≥10cm		Abundance	0	0	414	-	-	-	143	-	-	-	-	-	-	-
			Variance	0	0	96404	-	-	-	3138	-	-	-	-	-	-	-
			95% C.I.	0	0	734	-	-	-	132	-	-	-	-	-	-	-
			Density (#/mi)	0	0	893.0	-	-	-	600	-	-	-	-	-	-	-
			Variance (#/mi)	0	0	448,846	-	-	-	55,271	-	-	-	-	-	-	-
			95% C.I. (#/mi)	0	0	1,584.2	-	-	-	556	-	-	-	-	-	-	-
			Density (#/100ft^2)	0.00	0.00	0.81	-	-	-	0.48	-	-	-	-	-	-	-
			Variance (#/100ft^2)	0.0000	0.0000	0.3721	-	-	-	0.0350	-	-	-	-	-	-	-
			95% C.I. (#/100ft^2)	0.00	0.00	1.44	-	-	-	0.44	-	-	-	-	-	-	-
2008	All *O. mykiss*	Pools	# Units Sampled	7	6	5	0	0	0	8	8	8	8	10	9	8	0
			Abundance	0	0	833	-	-	-	105	169	278	5	13	189	201	-
			Variance	0	0	28481	-	-	-	788	1411	3273	1	4	4020	609	-
			95% C.I.	0	0	469	-	-	-	66	89	135	3	5	146	58	-
			Density (#/mi)	0	0	3,271.9	-	-	-	537	582	1,318	39	218	920	1,176	-
			Variance (#/mi)	0	0	438,919	-	-	-	20,591	16,690	73,384	83	1,085	94,686	20,760	-
			95% C.I. (#/mi)	0	0	1,839.4	-	-	-	339	305	641	22	75	710	341	-
			Density (#/100ft^2)	0.00	0.00	2.09	-	-	-	0.37	0.82	2.31	0.03	0.28	1.04	2.32	-
			Variance (#/100ft^2)	0.0000	0.0000	0.17849	-	-	-	0.0099	0.0328	0.2257	0.0000	0.0019	0.1203	0.0808	-
			95% C.I. (#/100ft^2)	0.00	0.00	1.17	-	-	-	0.24	0.43	1.12	0.02	0.10	0.80	0.67	-
2008	All *O. mykiss*	Flatwaters	# Units Sampled	8	8	8	0	0	0	8	n/a	n/a	n/a	n/a	n/a	n/a	0
			Abundance	0	0	468	-	-	-	176	-	-	-	-	-	-	-
			Variance	0	0	100508	-	-	-	3966	-	-	-	-	-	-	-
			95% C.I.	0	0	750	-	-	-	149	-	-	-	-	-	-	-
			Density (#/mi)	0	0	1,009.5	-	-	-	737	-	-	-	-	-	-	-
			Variance (#/mi)	0	0	467,953	-	-	-	69,870	-	-	-	-	-	-	-
			95% C.I. (#/mi)	0	0	1,617.6	-	-	-	625	-	-	-	-	-	-	-
			Density (#/100ft^2)	0.00	0.00	0.92	-	-	-	0.59	-	-	-	-	-	-	-
			Variance (#/100ft^2)	0.0000	0.0000	0.3880	-	-	-	0.0442	-	-	-	-	-	-	-
			95% C.I. (#/100ft^2)	0.00	0.00	1.47	-	-	-	0.50	-	-	-	-	-	-	-
2007	Fry <10cm	Pools	# Units Sampled	7	6	6	0	0	0	8	8	8	8	7	8	8	0
			Abundance	0	0	0	-	-	-	7	111	214	0	37	106	84	-
			Variance	0	0	0	-	-	-	6	385	4117	0	24	85	650	-
			95% C.I.	0	0	0	-	-	-	6	46	152	0	12	22	60	-
			Density (#/mi)	0.0	0.0	0.0	-	-	-	34.3	382.4	1,012.0	0.0	292.8	514.8	488.6	-
			Variance (#/mi)	0.00	0.00	0.00	-	-	-	150.39	4,551.76	92,331.71	0.00	1,515.09	1,993.38	22,183.82	-
			95% C.I. (#/mi)	0.0	0.0	0.0	-	-	-	29.0	159.5	718.5	0.0	95.2	105.6	352.2	-
			Density (#/100ft^2)	0.00	0.00	0.00	-	-	-	0.02	0.41	1.71	0.00	0.32	0.52	0.97	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	-	-	-	0.0001	0.0053	0.2640	0.0000	0.0019	0.0020	0.0882	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	-	-	-	0.02	0.17	1.22	0.00	0.11	0.11	0.70	-

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2007	Fry	Flatwaters	# Units Sampled	8	8	8	0	0	0	8	8	8	8	6	7	8	0
	<10cm		Abundance	0	0	0	-	-	-	42	85	289	34	231	105	291	-
			Variance	0	0	0	-	-	-	234	88	1421	216	3085	366	1611	-
			95% C.I.	0	0	0	-	-	-	36	22	89	35	143	47	95	-
			Density (#/mi)	0.0	0.0	0.0	-	-	-	177	1,166	1,936	129.1	986	967	1,868	-
			Variance (#/mi)	0.00	0.00	0.00	-	-	-	4,122	16,524	63,944	3,084.55	56,210	30,900	66,463	-
			95% C.I. (#/mi)	0.0	0.0	0.0	-	-	-	152	304	598	131.3	609	430	610	-
			Density (#/100ft^2)	0.00	0.00	0.00	-	-	-	0.14	2.04	4.26	0.09	1.64	2.06	4.11	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	-	-	-	0.0027	0.0508	0.3101	0.0014	0.1551	0.1399	0.3223	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	-	-	-	0.12	0.53	1.32	0.09	1.01	0.92	1.34	-
2007	Fry	Riffles	# Units Sampled	8	8	8	0	0	0	8	8	8	8	4	7	8	0
	<10cm		Abundance	0	0	0	-	-	-	83	94	73	9	40	67	224	-
			Variance	0	0	0	-	-	-	365	48	22	0	0	61	433	-
			95% C.I.	0	0	0	-	-	-	44	16	11	0	0	19	49	-
			Density (#/mi)	0.0	0.0	0.0	-	-	-	866	2,121	1,438	108.2	542	1,047	3,184	-
			Variance (#/mi)	0.00	0.00	0.00	-	-	-	39,431	24,439	8,509	0.00	0	14,925	87,707	-
			95% C.I. (#/mi)	0.0	0.0	0.0	-	-	-	458	370	218	0.0	0	299	700	-
			Density (#/100ft^2)	0.00	0.00	0.00	-	-	-	1.00	2.83	3.13	0.09	1.60	1.93	7.63	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	-	-	-	0.0526	0.0435	0.0403	0.0000	0.0000	0.0505	0.5041	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	-	-	-	0.53	0.49	0.47	0.00	0.00	0.55	1.68	-
2007	Fry	All	# Units Sampled	23	22		0	0	0	24	24	24	24	17	22	24	0
	<10cm	Habitats	Abundance	0	0	0	-	-	-	132	290	575	44	308	278	598	-
			Variance	0	0	0	-	-	-	605	521	5560	216	3109	512	2694	-
			95% C.I.	0	0	0	-	-	-	51	47	155	31	120	47	108	-
			Density (#/mi)	0.0	0.0	0.0	-	-	-	249.5	711.3	1,399.7	93.1	1,001.0	734.4	1,506.1	-
			Variance (#/mi)	0.00	0.00	0.00	-	-	-	2,152.00	3,127.87	32,917.47	983.34	32,867.00	3,567.73	17,080.28	-
			95% C.I. (#/mi)	0.0	0.0	0.0	-	-	-	96.2	116.3	377.3	65.2	389.0	125.0	271.8	-
			Density (#/100ft^2)	0.00	0.00	0.00	-	-	-	0.19	0.84	2.66	0.07	1.10	0.96	3.22	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	-	-	-	0.0013	0.0044	0.1193	0.0005	0.0397	0.0061	0.0780	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	-	-	-	0.07	0.14	0.72	0.05	0.43	0.16	0.58	-
2007	Juv+	Pools	# Units Sampled	7	6	6	0	0	0	8	8	8	8	7	8	8	0
	≥10cm		Abundance	0	0	4	-	-	-	0	16	51	0	29	52	79	-
			Variance	0	0	0	-	-	-	0	48	205	0	0	127	849	-
			95% C.I.	0	0	0	-	-	-	0	16	34	0	0	27	69	-
			Density (#/mi)	8.3	0.0	15.7	-	-	-	0	55	239	0.0	230	254	462	-
			Variance (#/mi)	0.00	0.00	0.00	-	-	-	0	568	4,603	0.00	0	2,993	28,972	-
			95% C.I. (#/mi)	0.0	0.0	0.0	-	-	-	0	56	160	0.0	0	129	402	-
			Density (#/100ft^2)	0.00	0.00	0.01	-	-	-	0.00	0.06	0.40	0.00	0.25	0.26	0.92	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	-	-	-	0.0000	0.0007	0.0132	0.0000	0.0000	0.0030	0.1151	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	-	-	-	0.00	0.06	0.27	0.00	0.00	0.13	0.80	-

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2007	Juv+	Flatwaters	# Units Sampled	8	8	8	0	0	0	8	8	8	8	6	7	8	0
	≥10cm		Abundance	0	0	0	-	-	-	3	1	26	7	39	6	23	-
			Variance	0	0	0	-	-	-	8	0	39	25	425	6	11	-
			95% C.I.	0	0	0	-	-	-	6	1	15	12	53	6	8	-
			Density (#/mi)	0.0	0.0	0.0	-	-	-	14	17	176	25.8	166	54	149	-
			Variance (#/mi)	0.00	0.00	0.00	-	-	-	132	60	1,759	360.30	7,740	504	445	-
			95% C.I. (#/mi)	0.0	0.0	0.0	-	-	-	27	18	99	44.9	226	55	50	-
			Density (#/100ft^2)	0.00	0.00	0.00	-	-	-	0.01	0.03	0.39	0.02	0.28	0.11	0.33	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	-	-	-	0.0001	0.0002	0.0085	0.0002	0.0214	0.0023	0.0022	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	-	-	-	0.02	0.03	0.22	0.03	0.38	0.12	0.11	-
2007	Juv+	Riffles	# Units Sampled	8	8	8	0	0	0	8	8	8	8	4	7	8	0
	≥10cm		Abundance	0	0	0	-	-	-	7	6	14	0	1	0	11	-
			Variance	0	0	0	-	-	-	5	0	31	0	0	0	4	-
			95% C.I.	0	0	0	-	-	-	5	0	13	0	0	0	5	-
			Density (#/mi)	0.0	0.0	0.0	-	-	-	69	135	270	0.0	14	0	159	-
			Variance (#/mi)	0.00	0.00	0.00	-	-	-	567	0	12,040	0.00	0	0	740	-
			95% C.I. (#/mi)	0.0	0.0	0.0	-	-	-	55	0	259	0.0	0	0	64	-
			Density (#/100ft^2)	0.00	0.00	0.00	-	-	-	0.08	0.18	0.59	0.00	0.04	0.00	0.38	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	-	-	-	0.0008	0.0000	0.0571	0.0000	0.0000	0.0000	0.0043	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	-	-	-	0.06	0.00	0.56	0.00	0.00	0.00	0.15	-
2007	Juv+	All	# Units Sampled	23	22	22	0	0	0	24	24	24	24	17	22	24	0
	≥10cm	Habitats	Abundance	0	0	4	-	-	-	10	23	90	7	69	58	114	-
			Variance	0	0	0	-	-	-	13	48	275	25	425	133	864	-
			95% C.I.	0	0	0	-	-	-	7	14	35	10	44	24	61	-
			Density (#/mi)	0.0	0.0	4.3	-	-	-	19	57	220	14.6	159	154	286	-
			Variance (#/mi)	0.00	0.00	0.00	-	-	-	45	290	1,630	114.86	2,252	927	5,476	-
			95% C.I. (#/mi)	0.0	0.0	0.0	-	-	-	14	35	84	22.3	102	64	154	-
			Density (#/100ft^2)	0.00	0.00	0.00	-	-	-	0.01	0.07	0.42	0.01	0.25	0.20	0.61	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	-	-	-	0.0000	0.0004	0.0059	0.0001	0.0054	0.0016	0.0250	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	-	-	-	0.01	0.04	0.16	0.02	0.16	0.08	0.33	-
2007	All *O. mykiss*	All Habitats	# Units Sampled	23	22	22	0	0	0	24	24	24	24	17	22	24	0
			Abundance	0	0	4	-	-	-	142	313	666	51	377	336	712	-
			Variance	0	0	0	-	-	-	618	569	5835	242	3534	645	3558	-
			95% C.I.	0	0	0	-	-	-	52	50	159	32	128	53	124	-
			Density (#/mi)	0.0	0.0	4.3	-	-	-	268	768	1,620	107.7	868	888	1,792	-
			Variance (#/mi)	0.00	0.00	0.00	-	-	-	2,197	3,418	34,548	1,098.21	18,739	4,495	22,556	-
			95% C.I. (#/mi)	0.0	0.0	0.0	-	-	-	97	122	387	68.9	294	140	312	-
			Density (#/100ft^2)	0.00	0.00	0.00	-	-	-	0.21	0.91	3.08	0.08	1.35	1.16	3.83	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	-	-	-	0.0013	0.0048	0.1252	0.0005	0.0451	0.0076	0.1030	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	-	-	-	0.07	0.14	0.74	0.05	0.41	0.18	0.67	-

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2006	Fry	Pools	# Units Sampled	4	6	6	4	0	0	8	8	8	8	9	9	8	0
	<10 cm		Abundance	0	0	0	0	-	-	32	20	43	4	21	90	87	-
			Variance	0	0	0	0	-	-	329	205	223	5	72	262	273	-
			95% C.I.	0	0	0	0	-	-	43	34	35	6	20	37	39	-
			Density (#/mi)	0.0	0.0	0.0	0	-	-	165.6	111.7	202.6	36.3	169.1	437.6	751.0	-
			Variance (#/mi)	0.00	0.00	0.00	0	-	-	8,598.70	6,246.28	5,005.97	404.98	4,511.38	6,165.24	20,530.38	-
			95% C.I. (#/mi)	0.0	0.0	0.0	0	-	-	219.3	186.9	167.3	47.6	154.9	181.1	338.8	-
			Density (#/100ft^2)	0.00	0.00	0.00	0.00	-	-	0.09	0.14	0.32	0.02	0.10	0.41	1.19	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	0.0000	-	-	0.0027	0.0105	0.0127	0.0002	0.0016	0.0054	0.0511	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	0.00	-	-	0.12	0.24	0.27	0.03	0.09	0.17	0.53	-
2006	Fry	Flatw aters	# Units Sampled	4	8	8	4	0	0	8	8	8	8	8	8	8	0
	<10cm		Abundance	0	0	0	0	-	-	85	35	74	9	85	34	71	-
			Variance	0	0	0	0	-	-	758	99	168	25	301	39	212	-
			95% C.I.	0	0	0	0	-	-	65	23	31	12	41	15	34	-
			Density (#/mi)	0.0	0.0	0.0	0	-	-	355	241	499	32.6	361	317	639	-
			Variance (#/mi)	0.00	0.00	0.00	0	-	-	13,356	4,821	7,574	351.52	5,481	3,302	16,963	-
			95% C.I. (#/mi)	0.0	0.0	0.0	0	-	-	273	164	206	44.3	175	136	308	-
			Density (#/100ft^2)	0.00	0.00	0.00	0.00	-	-	0.23	0.31	0.85	0.02	0.27	0.38	1.27	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	0.0000	-	-	0.0055	0.0080	0.0221	0.0001	0.0032	0.0047	0.0674	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	0.00	-	-	0.18	0.21	0.35	0.02	0.13	0.16	0.61	-
2006	Fry	Riffles	# Units Sampled	4	8	8	4	0	0	8	8	8	8	8	8	8	0
	<10cm		Abundance	0	2	0	0	-	-	28	15	34	10	73	25	41	-
			Variance	0	4	0	0	-	-	107	0	35	0	125	0	43	-
			95% C.I.	0	4	0	0	-	-	24	0	14	0	26	0	15	-
			Density (#/mi)	0.0	9.5	0.0	0	-	-	290	360	672	114.0	982	392	572	-
			Variance (#/mi)	0.00	76.86	0.00	0	-	-	11,562	0	13,579	0.00	22,977	0	8,162	-
			95% C.I. (#/mi)	0.0	20.7	0.0	0	-	-	254	0	276	0.0	358	0	214	-
			Density (#/100ft^2)	0.00	0.01	0.00	0.00	-	-	0.22	0.51	1.11	0.06	0.48	0.39	1.19	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	0.0000	-	-	0.0069	0.0000	0.0368	0.0000	0.0056	0.0000	0.0354	-
			95% C.I. (#/100ft^2)	0.00	0.01	0.00	0.00	-	-	0.20	0.00	0.45	0.00	0.18	0.00	0.44	-
2006	Fry	All	# Units Sampled	12	22	22	12	0	0	24	24	24	24	25	25	24	0
	<10cm	Habitats	Abundance	0	2	0	0	-	-	145	70	151	23	178	150	199	-
			Variance	0	4	0	0	-	-	1194	304	426	30	498	301	528	-
			95% C.I.	0	4	0	0	-	-	72	36	43	11	46	36	48	-
			Density (#/mi)	0.0	2.1	0.0	0	-	-	273.1	190.6	367.9	48.7	410.6	395.1	665.8	-
			Variance (#/mi)	0.00	3.96	0.00	0	-	-	4,249.93	2,268.82	2,525.02	137.05	2,640.48	2,097.43	5,885.07	-
			95% C.I. (#/mi)	0.0	4.2	0.0	0	-	-	135.6	99.1	104.5	24.3	106.6	95.0	159.5	-
			Density (#/100ft^2)	0.00	0.00	0.00	0.00	-	-	0.17	0.25	0.60	0.03	0.27	0.40	1.22	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	0.0000	-	-	0.0017	0.0039	0.0068	0.0000	0.0011	0.0021	0.0196	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	0.00	-	-	0.09	0.13	0.17	0.01	0.07	0.10	0.29	-

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2006	Juv+	Pools	# Units Sampled	4	6	6	4	0	0	8	8	8	8	9	9	8	0
	≥10 cm		Abundance	0	0	6	0	-	-	75	53	74	34	39	118	112	-
			Variance	0	0	0	0	-	-	483	497	378	32	54	194	1001	-
			95% C.I.	0	0	0	0	-	-	52	53	46	13	17	32	75	-
			Density (#/mi)	0.0	0.0	23.6	0	-	-	384	294	352	287.6	311	573	967	-
			Variance (#/mi)	0.00	0.00	0.00	0	-	-	12,615	15,132	8,482	2,381.85	3,399	4,557	75,270	-
			95% C.I. (#/mi)	0.0	0.0	0.0	0	-	-	266	291	218	115.4	134	156	649	-
			Density (#/100ft^2)	0.00	0.00	0.01	0.00	-	-	0.21	0.38	0.56	0.19	0.19	0.53	1.53	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	0.0000	-	-	0.0039	0.0255	0.0215	0.0010	0.0012	0.0040	0.1875	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	0.00	-	-	0.15	0.38	0.35	0.08	0.08	0.15	1.02	-
2006	Juv+	Flatwaters	# Units Sampled	4	8	8	4	0	0	8	8	8	8	8	8	8	0
	≥10 cm		Abundance	0	0	0	0	-	-	94	42	68	48	179	24	26	-
			Variance	0	0	0	0	-	-	1665	153	140	358	2683	6	105	-
			95% C.I.	0	0	0	0	-	-	96	29	28	45	122	6	24	-
			Density (#/mi)	0.0	0.0	0.0	0	-	-	396	296	453	180.7	763	216	232	-
			Variance (#/mi)	0.00	0.00	0.00	0	-	-	29,332	7,493	6,318	5,105.76	48,876	478	8,388	-
			95% C.I. (#/mi)	0.0	0.0	0.0	0	-	-	405	205	188	169.0	523	52	217	-
			Density (#/100ft^2)	0.00	0.00	0.00	0.00	-	-	0.25	0.38	0.77	0.10	0.58	0.26	0.46	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	0.0000	-	-	0.0122	0.0124	0.0184	0.0016	0.0283	0.0007	0.0333	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	0.00	-	-	0.26	0.26	0.32	0.09	0.40	0.06	0.43	-
2006	Juv+	Riffles	# Units Sampled	4	8	8	4	0	0	8	8	8	8	8	8	8	0
	≥10cm		Abundance	0	2	0	0	-	-	34	3	28	13	54	11	11	-
			Variance	0	3	0	0	-	-	50	0	15	0	63	0	6	-
			95% C.I.	0	4	0	0	-	-	17	0	9	0	19	0	6	-
			Density (#/mi)	0.0	9.5	0.0	0	-	-	352	72	552	148.3	732	172	145	-
			Variance (#/mi)	0.00	75.25	0.00	0	-	-	5,420	0	5,733	0.00	11,604	0	1,146	-
			95% C.I. (#/mi)	0.0	20.5	0.0	0	-	-	174	0	179	0.0	255	0	80	-
			Density (#/100ft^2)	0.00	0.01	0.00	0.00	-	-	0.27	0.10	0.91	0.08	0.36	0.17	0.30	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	0.0000	-	-	0.0032	0.0000	0.0156	0.0000	0.0028	0.0000	0.0050	-
			95% C.I. (#/100ft^2)	0.00	0.01	0.00	0.00	-	-	0.13	0.00	0.29	0.00	0.13	0.00	0.17	-
2006	Juv+	All	# Units Sampled	12	22	22	12	0	0	24	24	24	24	25	25	24	0
	≥10cm	Habitats	Abundance	0	2	6	0	-	-	203	99	170	94	272	153	148	-
			Variance	0	3	0	0	-	-	2198	650	533	390	2800	199	1112	-
			95% C.I.	0	4	0	0	-	-	98	53	48	41	110	29	69	-
			Density (#/mi)	0.0	2.1	6.5	0	-	-	383	269	414	201.2	627	403	494	-
			Variance (#/mi)	0.00	3.88	0.00	0	-	-	7,822	4,857	3,158	1,774.65	14,846	1,388	12,403	-
			95% C.I. (#/mi)	0.0	4.1	0.0	0	-	-	184	145	117	87.6	253	77	232	-
			Density (#/100ft^2)	0.00	0.00	0.00	0.00	-	-	0.24	0.35	0.68	0.12	0.41	0.41	0.90	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	0.0000	-	-	0.0031	0.0083	0.0085	0.0006	0.0063	0.0014	0.0414	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	0.00	-	-	0.12	0.19	0.19	0.05	0.16	0.08	0.42	-

Year	Size Class	Habitat Type	Statistic	VEN 1	VEN 2	VEN 3	VEN 4	SAC mid	SAC up	VEN 5	LNF low/new	LNF mid	MAT 3	MAT 5	MAT 7/7b	UNF up/new	MUR 3
2006	All *O. mykiss*	All Habitats	# Units Sampled	12	22	22	12	0	0	24	24	24	24	25	25	24	0
			Abundance	0	4	6	0	-	-	348	168	321	117	450	302	347	-
			Variance	0	7	0	0	-	-	3392	954	960	420	3298	500	1640	-
			95% C.I.	0	6	0	0	-	-	121	64	64	43	119	46	84	-
			Density (#/mi)	0.0	4.3	6.5	0	-	-	657	460	781	249.9	1,037	798	1,160	-
			Variance (#/mi)	0.00	7.85	0.00	0	-	-	12,072	7,126	5,683	1,911.69	17,487	3,486	18,289	-
			95% C.I. (#/mi)	0.0	5.9	0.0	0	-	-	228	176	157	90.9	274	122	281	-
			Density (#/100ft^2)	0.00	0.00	0.00	0.00	-	-	0.41	0.60	1.28	0.14	0.68	0.81	2.12	-
			Variance (#/100ft^2)	0.0000	0.0000	0.0000	0.0000	-	-	0.0048	0.0122	0.0153	0.0006	0.0074	0.0036	0.0610	-
			95% C.I. (#/100ft^2)	0.00	0.00	0.00	0.00	-	-	0.14	0.23	0.26	0.05	0.18	0.12	0.51	-

Appendix E

Correlation Matrices between *O. mykiss* Abundance and Habitat Parameters According to Channel and Habitat Type, 2012

(see Table 5 for variable descriptions)

Mainstem Pools n= 28 critical r= 0.37

VAR	AvDep	ThalDep	MaxDep	D1	D2	D3	AvVel	V05	V1	CB	Turb	Bubble	IWBR	AQVeg	OHVeg	RipVeg	AllCov	Shelter	V05IW	V05OW	V1IW	V1OW
Fry	-0.10	-0.04	0.02	-0.02	0.02	-0.04	-0.06	0.03	0.02	0.01	0.05	-0.16	-0.14	-0.02	-0.20	-0.10	-0.11	-0.23	0.09	0.10	0.14	0.04
Juv	-0.02	0.14	0.16	0.08	0.04	0.05	-0.06	-0.13	0.04	0.10	0.07	0.13	-0.17	-0.03	-0.10	-0.08	0.01	0.08	-0.02	0.13	0.19	0.04
Adult	**0.49**	**0.49**	**0.54**	**0.41**	**0.44**	**0.46**	-0.20	-0.16	0.05	-0.17	0.01	-0.14	-0.01	-0.19	0.09	-0.05	-0.12	-0.17	-0.10	0.11	0.10	0.09
AvDep	1.00	0.97	0.94	0.84	0.94	0.88	-0.47	-0.48	-0.18	-0.21	-0.02	-0.03	0.11	-0.13	0.07	0.01	-0.15	0.16	-0.38	-0.11	-0.09	-0.06
ThalDep		1.00	0.96	0.79	0.92	0.87	-0.46	-0.49	-0.14	-0.11	0.03	0.04	0.06	-0.20	0.02	-0.04	-0.08	0.19	-0.36	-0.06	-0.02	-0.02
MaxDep			1.00	0.75	0.91	0.86	-0.52	-0.54	-0.17	-0.24	0.01	-0.04	0.12	-0.15	0.07	0.05	-0.17	0.15	-0.39	-0.08	-0.01	-0.03
D1				1.00	0.72	0.57	-0.42	-0.49	-0.09	-0.31	-0.02	0.02	0.20	0.03	0.25	0.18	-0.14	0.30	-0.42	0.02	-0.04	0.00
D2					1.00	0.87	-0.49	-0.48	-0.19	-0.10	0.05	-0.08	0.15	-0.22	0.03	-0.07	-0.04	0.16	-0.36	-0.07	-0.04	-0.02
D3						1.00	-0.45	-0.40	-0.28	-0.25	-0.08	-0.16	-0.02	-0.17	-0.09	-0.11	-0.29	-0.09	-0.37	-0.21	-0.20	-0.18
AvVel							1.00	0.94	0.60	0.55	0.27	0.44	-0.07	0.04	0.04	-0.12	0.55	0.28	0.86	0.34	0.36	0.34
V05								1.00	0.48	0.49	0.20	0.30	-0.07	-0.01	0.01	-0.13	0.47	0.15	0.86	0.27	0.25	0.26
V1									1.00	0.39	0.66	0.50	0.01	-0.17	0.19	-0.02	0.50	0.29	0.52	0.73	0.85	0.78
CB										1.00	0.40	0.54	-0.27	-0.36	-0.27	-0.43	0.80	0.28	0.69	0.22	0.37	0.31
Turb											1.00	0.46	-0.12	-0.19	-0.18	-0.21	0.33	0.24	0.39	0.75	0.82	0.87
Bubble												1.00	-0.15	-0.17	-0.16	-0.16	0.44	0.28	0.47	0.35	0.45	0.43
IWBR													1.00	-0.03	0.84	0.72	0.32	0.44	-0.16	0.20	-0.02	0.12
AQVeg														1.00	-0.01	0.38	-0.37	0.22	-0.10	-0.19	-0.19	-0.18
OHVeg															1.00	0.73	0.35	0.47	-0.13	0.28	0.08	0.16
RipVeg																1.00	0.05	0.42	-0.20	0.17	-0.03	0.07
AllCov																	1.00	0.58	0.60	0.42	0.44	0.44
Shelter																		1.00	0.24	0.34	0.32	0.34
V05IW																			1.00	0.41	0.47	0.45
V05OW																				1.00	0.83	0.95
V1IW																					1.00	0.90
V1OW																						1.00

Tributary Pools n= 42 critical r= 0.29

VAR	AvDep	ThalDep	MaxDep	D1	D2	D3	AvVel	V05	V1	CB	Turb	Bubble	IWBR	AQVeg	OHVeg	RipVeg	AllCov	Shelter	Fines	V05IW	V05OW	V1IW	V1OW
Fry	**-0.30**	**-0.31**	-0.19	-0.17	**-0.39**	-0.10	0.26	0.21	0.14	-0.05	0.20	-0.03	-0.17	-0.07	0.06	-0.11	-0.06	-0.02	-0.09	0.18	0.17	**0.29**	0.02
Juv	0.27	0.28	0.37	0.13	**0.36**	**0.50**	0.07	0.11	0.06	-0.04	0.08	0.24	-0.17	-0.14	0.17	0.25	0.01	0.22	-0.18	0.05	0.08	0.22	0.18
Adult	-0.09	-0.04	-0.03	-0.12	0.00	0.07	0.00	-0.07	-0.03	-0.10	-0.14	-0.15	0.04	-0.09	0.22	-0.05	0.05	0.14	0.19	-0.11	-0.14	-0.14	-0.12
AvDep	1.00	0.96	0.89	0.93	0.90	0.60	-0.29	-0.22	-0.17	-0.30	-0.06	0.21	-0.18	-0.15	0.00	0.31	-0.30	-0.08	0.09	-0.25	-0.12	-0.15	-0.13
ThalDep		1.00	0.91	0.84	0.91	0.70	-0.34	-0.27	-0.16	-0.24	-0.07	0.12	-0.11	-0.12	0.11	0.40	-0.18	0.09	0.10	-0.21	-0.12	-0.12	-0.12
MaxDep			1.00	0.75	0.89	0.71	-0.25	-0.17	-0.04	-0.34	-0.08	0.18	-0.06	-0.11	0.14	0.37	-0.22	0.11	0.17	-0.14	-0.11	-0.06	-0.06
D1				1.00	0.72	0.40	-0.25	-0.21	-0.20	-0.38	-0.07	0.25	-0.27	-0.07	-0.07	0.25	-0.44	-0.27	0.09	-0.32	-0.14	-0.19	-0.20
D2					1.00	0.65	-0.29	-0.20	-0.16	-0.21	-0.12	0.13	-0.08	-0.15	0.10	0.29	-0.15	0.08	0.12	-0.19	-0.15	-0.16	-0.07
D3						1.00	-0.18	-0.15	-0.13	-0.20	-0.07	-0.12	0.01	-0.07	0.26	0.47	-0.02	0.31	0.09	-0.04	-0.08	0.02	-0.05
AvVel							1.00	0.91	0.57	0.17	0.49	0.23	-0.27	0.05	-0.20	-0.14	-0.03	0.18	-0.37	0.65	0.45	0.53	0.54
V05								1.00	0.53	0.18	0.67	0.29	-0.27	0.02	-0.25	-0.19	-0.05	0.18	-0.39	0.75	0.64	0.66	0.75
V1									1.00	0.01	0.41	0.21	-0.31	0.00	-0.19	-0.11	-0.17	0.18	-0.29	0.32	0.38	0.43	0.51
CB										1.00	0.13	-0.09	-0.12	-0.13	-0.18	-0.31	0.70	0.43	-0.46	0.48	0.07	0.25	0.13
Turb											1.00	0.50	-0.19	-0.08	-0.11	-0.18	0.03	0.21	-0.31	0.52	0.93	0.71	0.87
Bubble												1.00	-0.15	-0.04	-0.04	-0.07	-0.12	-0.07	-0.22	0.03	0.32	0.13	0.34
IWBR													1.00	-0.01	0.40	-0.02	0.41	0.18	0.35	-0.07	-0.17	-0.17	-0.15
AQVeg														1.00	0.02	0.06	-0.10	-0.18	-0.17	-0.12	-0.07	-0.07	-0.06
OHVeg															1.00	0.45	0.52	0.51	0.21	-0.18	-0.02	-0.12	-0.12
RipVeg																1.00	-0.01	0.35	0.18	-0.10	-0.11	-0.05	-0.14
AllCov																	1.00	0.70	-0.17	0.30	0.03	0.12	0.04
Shelter																		1.00	-0.08	0.45	0.19	0.41	0.32
Fines																			1.00	-0.29	-0.26	-0.31	-0.25
V05IW																				1.00	0.55	0.74	0.63
V05OW																					1.00	0.74	0.87
V1IW																						1.00	0.82
V1OW																							1.00

Mainstem Flatwaters n= 28 critical r= 0.37

VAR	AvDep	ThalDep	MaxDep	D1	D2	AvVel	V05	V1	CB	Turb	Bubble	IWBR	AQVeg	OHVeg	RipVeg	AllCov	Shelter	Fines	V05IW	V05OW	V1IW	V1OW
Fry	-0.08	-0.16	-0.16	0.00	-0.20	**0.61**	**0.55**	**0.63**	-0.29	0.13	-0.25	0.35	**0.46**	**0.42**	0.31	-0.05	-0.11	**-0.48**	0.19	0.35	0.13	0.24
Juv	-0.12	-0.12	-0.12	-0.07	-0.22	**0.40**	**0.49**	0.32	0.17	0.15	-0.26	0.36	0.12	0.29	0.03	0.30	-0.03	**-0.41**	**0.55**	0.29	0.34	0.13
Adult	-0.21	-0.20	-0.21	-0.27	-0.10	-0.07	-0.05	-0.06	0.03	-0.10	-0.11	-0.14	-0.13	-0.21	-0.22	-0.05	-0.11	-0.13	-0.05	-0.13	-0.11	-0.11
AvDep	1.00	0.97	0.94	0.93	0.85	-0.20	-0.30	-0.18	0.55	-0.12	0.30	0.21	-0.07	0.27	0.23	0.52	0.65	-0.02	-0.05	-0.13	-0.05	-0.12
ThalDep		1.00	0.96	0.91	0.82	-0.20	-0.28	-0.17	0.58	-0.10	0.33	0.17	-0.15	0.22	0.22	0.53	0.68	-0.09	-0.03	-0.12	0.03	-0.10
MaxDep			1.00	0.88	0.82	-0.17	-0.25	-0.15	0.55	-0.04	0.41	0.10	-0.15	0.21	0.22	0.51	0.72	-0.06	0.00	-0.09	0.03	-0.05
D1				1.00	0.65	-0.10	-0.20	-0.06	0.60	-0.02	0.40	0.22	0.01	0.24	0.21	0.58	0.68	-0.15	0.10	-0.03	0.11	-0.01
D2					1.00	-0.23	-0.34	-0.17	0.38	-0.06	0.23	0.10	-0.21	0.15	0.08	0.32	0.50	0.16	-0.19	-0.12	-0.08	-0.07
AvVel						1.00	0.91	0.92	-0.03	0.58	0.14	0.30	0.16	0.42	0.25	0.23	0.14	-0.44	0.61	0.64	0.67	0.62
V05							1.00	0.81	-0.07	0.54	0.01	0.34	0.27	0.36	0.23	0.16	0.10	-0.51	0.75	0.63	0.64	0.57
V1								1.00	-0.04	0.66	0.18	0.39	0.09	0.45	0.24	0.24	0.14	-0.44	0.54	0.74	0.72	0.73
CB									1.00	0.24	0.42	0.16	-0.36	0.10	-0.05	0.92	0.68	-0.06	0.50	0.18	0.40	0.18
Turb										1.00	0.40	0.38	-0.19	0.34	0.15	0.44	0.27	-0.29	0.56	0.94	0.68	0.99
Bubble											1.00	-0.09	-0.24	-0.04	-0.11	0.44	0.69	-0.09	0.27	0.21	0.49	0.38
IWBR												1.00	0.04	0.58	0.22	0.34	0.14	-0.40	0.35	0.57	0.29	0.39
AQVeg													1.00	0.21	0.21	-0.26	-0.10	-0.16	0.07	-0.05	-0.14	-0.13
OHVeg														1.00	0.70	0.44	0.27	-0.12	0.24	0.49	0.28	0.35
RipVeg															1.00	0.19	0.25	-0.12	-0.01	0.24	0.00	0.20
AllCov																1.00	0.72	-0.14	0.61	0.42	0.56	0.39
Shelter																	1.00	-0.18	0.46	0.20	0.43	0.25
Fines																		1.00	-0.38	-0.37	-0.32	-0.34
V05IW																			1.00	0.60	0.79	0.53
V05OW																				1.00	0.66	0.95
V1IW																					1.00	0.63
V1OW																						1.00

Tributary Flatwaters n= 42 critical r= **0.29**

VAR	AvDep	ThalDep	MaxDep	D1	AvVel	V05	V1	CB	Turb	Bubble	IWBR	AQVeg	OHVeg	RipVeg	AllCov	Shelter	Fines	V05IW	V05OW	V1IW	V1OW
Fry	-0.08	-0.11	-0.14	-0.02	0.00	0.04	0.00	-0.05	-0.20	-0.21	0.28	-0.21	**-0.30**	**-0.29**	-0.14	-0.19	0.12	-0.15	-0.23	-0.05	-0.22
Juv	0.27	0.15	0.17	0.28	-0.17	-0.16	-0.10	**0.35**	0.05	0.22	0.07	-0.06	-0.10	-0.10	**0.35**	**0.32**	-0.03	0.13	0.02	0.04	0.03
Adult	-0.22	-0.10	-0.16	-0.12	0.22	0.21	0.14	-0.05	-0.09	-0.14	-0.15	-0.11	0.08	-0.13	-0.08	-0.10	-0.10	0.05	-0.02	0.16	-0.06
AvDep	1.00	0.90	0.81	0.86	-0.10	-0.21	0.09	0.04	0.16	0.32	-0.21	0.01	0.23	0.21	0.11	0.35	-0.32	-0.13	0.25	0.04	0.18
ThalDep		1.00	0.87	0.82	-0.14	-0.16	0.10	0.18	0.08	0.17	-0.26	0.03	0.14	0.08	0.18	0.34	-0.34	0.01	0.17	0.10	0.10
MaxDep			1.00	0.75	-0.10	-0.15	0.03	0.23	0.01	0.23	-0.16	0.05	0.24	0.28	0.31	0.45	-0.25	0.05	0.03	0.05	0.01
D1				1.00	-0.10	-0.12	0.11	0.05	0.14	0.35	-0.12	-0.11	0.17	0.15	0.12	0.27	-0.24	0.08	0.20	0.13	0.14
AvVel					1.00	0.82	0.69	-0.16	0.31	0.25	-0.15	0.18	0.16	0.26	-0.11	-0.10	-0.33	0.36	0.30	0.49	0.34
V05						1.00	0.76	-0.06	0.43	0.24	-0.18	0.21	0.06	0.19	-0.05	-0.07	-0.36	0.57	0.39	0.56	0.45
V1							1.00	-0.13	0.43	0.29	-0.14	0.38	0.07	0.20	-0.10	-0.06	-0.39	0.45	0.38	0.63	0.45
CB								1.00	-0.12	-0.01	-0.29	-0.05	-0.29	-0.19	0.80	0.44	-0.20	0.50	-0.12	0.36	-0.11
Turb									1.00	0.66	-0.16	0.71	0.27	0.36	0.07	0.19	-0.38	0.14	0.89	0.26	0.98
Bubble										1.00	-0.11	0.49	0.46	0.50	0.26	0.33	-0.44	0.07	0.58	0.28	0.64
IWBR											1.00	-0.07	-0.02	-0.08	0.03	0.01	0.48	-0.25	-0.19	-0.26	-0.18
AQVeg												1.00	0.32	0.43	0.17	0.31	-0.28	-0.02	0.58	0.12	0.70
OHVeg													1.00	0.82	0.25	0.40	-0.17	-0.20	0.42	-0.13	0.33
RipVeg														1.00	0.25	0.47	-0.30	-0.10	0.41	-0.12	0.39
AllCov															1.00	0.70	-0.17	0.36	0.12	0.25	0.10
Shelter																1.00	-0.33	0.20	0.26	0.09	0.23
Fines																	1.00	-0.24	-0.41	-0.28	-0.40
V05IW																		1.00	0.10	0.75	0.16
V05OW																			1.00	0.24	0.95
V1IW																				1.00	0.30
V1OW																					1.00

Mainstem Riffles n= 28 critical r= 0.37

VAR	AvDep	ThalDep	MaxDep	D1	AvVel	V05	V1	CB	Turb	Bubble	IWBR	AQVeg	OHVeg	RipVeg	AllCov	Shelter	Fines	V05IW	V05OW	V1IW	V1OW
Fry	0.10	0.09	0.21	-0.02	0.26	0.17	0.39	0.27	0.37	0.11	-0.19	-0.10	-0.27	-0.25	0.37	-0.11	-0.17	0.24	0.30	0.39	0.35
Juv	0.23	0.22	0.31	0.02	0.27	0.24	0.38	0.21	0.38	-0.10	-0.22	0.08	-0.28	-0.27	0.33	-0.09	-0.02	0.32	0.34	0.42	0.35
Adult	0.54	0.41	0.25	0.50	0.16	0.12	0.21	0.27	0.15	0.12	-0.09	-0.19	-0.22	-0.13	0.28	0.46	-0.07	0.31	0.09	0.37	0.13
AvDep	1.00	0.90	0.68	0.89	-0.14	-0.24	-0.08	0.53	0.07	-0.09	-0.34	-0.17	0.15	0.16	0.53	0.58	0.32	0.25	0.06	0.22	0.05
ThalDep		1.00	0.87	0.83	-0.26	-0.28	-0.16	0.56	0.04	-0.16	-0.34	-0.11	-0.02	0.22	0.46	0.64	0.25	0.26	0.02	0.22	0.00
MaxDep			1.00	0.60	-0.27	-0.28	-0.18	0.51	0.05	-0.21	-0.35	-0.13	-0.15	0.17	0.37	0.45	0.17	0.23	0.03	0.18	0.00
D1				1.00	-0.21	-0.30	-0.11	0.43	0.09	0.03	-0.25	-0.31	0.10	0.15	0.44	0.55	0.35	0.07	0.03	0.07	0.05
AvVel					1.00	0.82	0.93	-0.14	0.79	0.56	-0.03	0.06	0.18	0.04	0.34	-0.16	-0.58	0.24	0.73	0.44	0.80
V05						1.00	0.79	-0.17	0.67	0.37	0.19	0.20	0.03	-0.15	0.18	-0.08	-0.53	0.45	0.67	0.47	0.69
V1							1.00	-0.04	0.88	0.61	-0.15	0.00	0.11	0.04	0.44	-0.04	-0.57	0.30	0.81	0.51	0.90
CB								1.00	-0.02	0.06	-0.32	-0.38	-0.18	0.12	0.80	0.64	-0.10	0.69	-0.01	0.64	-0.01
Turb									1.00	0.60	-0.14	-0.05	0.16	0.09	0.52	0.06	-0.40	0.19	0.95	0.37	0.99
Bubble										1.00	-0.06	-0.31	0.19	0.33	0.40	0.13	-0.50	0.08	0.60	0.26	0.63
IWBR											1.00	0.01	-0.01	-0.14	-0.35	-0.26	0.08	-0.18	-0.15	-0.17	-0.15
AQVeg												1.00	0.02	-0.02	-0.34	-0.04	0.03	0.00	0.01	-0.02	-0.07
OHVeg													1.00	0.54	0.07	-0.10	0.21	-0.21	0.27	-0.17	0.22
RipVeg														1.00	0.20	0.25	0.04	-0.08	0.17	0.02	0.13
AllCov															1.00	0.55	-0.22	0.61	0.52	0.67	0.53
Shelter																1.00	-0.10	0.46	0.09	0.36	0.06
Fines																	1.00	-0.34	-0.35	-0.46	-0.42
V05IW																		1.00	0.24	0.90	0.21
V05OW																			1.00	0.35	0.96
V1IW																				1.00	0.37
V1OW																					1.00

Tributary Riffles n= 34 critical r= 0.34

VAR	AvDep	ThalDep	MaxDep	D1	AvVel	V05	V1	CB	Turb	Bubble	IWBR	AQVeg	OHVeg	RipVeg	AllCov	Shelter	Fines	V05IW	V05OW	V1IW	V1OW
Fry	0.00	0.00	0.18	0.14	-0.15	-0.10	-0.08	-0.10	**-0.49**	**-0.47**	-0.01	-0.14	0.01	-0.02	-0.20	-0.09	0.21	-0.23	**-0.36**	-0.21	**-0.41**
Juv	0.09	0.08	0.13	0.14	-0.20	-0.23	-0.21	**0.57**	0.09	0.22	0.18	0.00	-0.23	-0.12	**0.44**	0.14	-0.04	0.18	0.02	0.06	0.04
Adult	0.10	0.10	0.07	0.03	0.14	0.06	0.13	**0.37**	0.21	0.04	0.09	-0.05	0.02	**0.36**	0.21	**0.38**	0.10	**0.40**	0.35	**0.42**	0.33
AvDep	1.00	0.95	0.79	0.63	-0.15	-0.20	-0.19	0.14	0.28	-0.08	0.17	0.17	0.02	0.04	0.22	0.18	-0.08	-0.03	0.18	-0.02	0.20
ThalDep		1.00	0.83	0.58	-0.20	-0.26	-0.24	0.19	0.20	-0.15	0.13	0.20	0.00	0.04	0.21	0.24	0.00	-0.04	0.12	0.00	0.15
MaxDep			1.00	0.80	-0.35	-0.39	-0.34	0.13	0.00	-0.27	-0.03	0.00	-0.13	-0.10	0.06	0.00	0.03	-0.24	-0.06	-0.21	-0.05
D1				1.00	-0.17	-0.18	-0.14	-0.08	0.15	-0.06	-0.13	-0.11	-0.23	-0.12	-0.13	-0.14	-0.19	-0.22	0.05	-0.18	0.06
AvVel					1.00	0.95	0.95	0.07	0.67	0.60	-0.12	0.20	0.04	0.36	0.05	0.27	-0.31	0.75	0.67	0.76	0.72
V05						1.00	0.92	-0.02	0.58	0.51	-0.15	0.18	-0.03	0.22	-0.04	0.19	-0.26	0.68	0.54	0.65	0.58
V1							1.00	0.06	0.63	0.56	-0.09	0.16	0.01	0.32	0.04	0.26	-0.31	0.73	0.62	0.74	0.68
CB								1.00	0.17	0.09	0.37	0.23	-0.10	0.21	0.81	0.59	-0.06	0.56	0.20	0.48	0.24
Turb									1.00	0.61	0.03	0.24	0.12	0.41	0.29	0.43	-0.37	0.67	0.87	0.67	0.94
Bubble										1.00	-0.07	0.07	-0.13	0.08	0.07	0.04	-0.60	0.50	0.44	0.47	0.56
IWBR											1.00	0.12	0.12	0.08	0.53	0.24	0.13	0.13	0.06	0.19	0.05
AQVeg												1.00	0.13	0.04	0.31	0.52	-0.16	0.24	0.20	0.19	0.24
OHVeg													1.00	0.72	0.41	0.29	0.03	0.03	0.40	0.13	0.29
RipVeg														1.00	0.43	0.58	-0.10	0.50	0.70	0.60	0.63
AllCov															1.00	0.62	-0.07	0.46	0.36	0.41	0.37
Shelter																1.00	-0.01	0.60	0.49	0.59	0.51
Fines																	1.00	-0.30	-0.26	-0.24	-0.33
V05IW																		1.00	0.68	0.93	0.74
V05OW																			1.00	0.76	0.97
V1IW																				1.00	0.81
V1OW																					1.00

Appendix F

Monitoring Annual Trends in Abundance & Distribution of Steelhead Above and Below Matilija Dam, Ventura, California

Funding Allocations, Grant P0950018

Matilija Coalition/Surfrider Foundation-Ventura
Monitoring Annual Trends in Abundance & Distribution
of Steelhead Above and Below Matilija Dam, Ventura, California
Duration: June 7, 2010 to March 31, 2015

PERSONAL SERVICES	**Totals**
Administration (Matilija Coalition)	$4,400
Program Manager	$2,400
Grant Coordinator	$2,640
OPERATING EXPENSES	
Subcontractors	
TRPA Sr. Fishery Biologist 3	$142,923
Normandeau Sr. Fishery Biologist 3	$64,106
TRPA Sr. Fishery Biologist 2	$69,378
Normandeau Sr. Fishery Biologist 2	$38,220
TRPA Fishery Biologist 2	$24,706
2-TRPA Fishery Technician 2	$73,875
2-Normandeau Fishery Technician 2	$19,555
Materials & Supplies (rental)	
Backpack Electrofishers & WQ Meters	$17,136
Backpack Electrofishers & Nets	$4,840
WQ Meters	$540
Wetsuits/Waders (4)	$360
Wetsuits/Waders/Boots	$1,360
Rangefinders (2)	$1,080
GPS Units (2)	$580
Digital Cameras (2)	$1,218
50 ft. Seine	$60
Temperature Dataloggers (6)	$1,800
Temperature Dataloggers	$1,080
Hipchain	$210
Underwater Video Recorder	$60
Kayak	$100
Misc. (Lump Sum)	$200
Travel	
Per Diem	$22,240
Lodging	$41,700
Mileage	$23,280
Air Fare	$960
Air Fare	$1,650
Car Rental	$406
Car Rental	$680
Publication Costs (LS)	$4,840
Administrative Overhead @ 10%	$944
Total Cost:	**$569,527**

Source of Funds	Cash	In-Kind	Total
FRGP	$569,527	$0	$569,527
Other State Agency(ies)	$0	$0	$0
Federal - NOAA	$0	$43,200	$43,200
Applicant	$0	$0	$0
Others(s)	$0	$17,205	$17,205
Total Project Cost	$569,527	$60,405	$629,932