Southern California steelhead monitoring in the Ventura River watershed 2018-2019

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August 2019

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ABSTRACT

This study is a continuation of monitoring efforts initiated by the California Department of Fish and Wildlife (CDFW) and the Pacific States Marine Fisheries Commission (PSMFC) in 2013 to ascertain the abundance and spatial structure of southern California steelhead (*Oncorhynchus mykiss*) via operation of a life cycle monitoring station (LCM). DIDSON and ARIS sonar cameras provided unbiased counts of anadromous adult *Oncorhynchus mykiss* while spatial structure and resident abundance were ascertained through comprehensive spawning ground surveys in the Ventura River, a watershed assigned a high priority for steelhead recovery action (Core 1; NMFS 2012).

Sonar cameras are a reliable method of obtaining adult counts in southern California systems because they are able to collect data at night and under dynamic stream conditions (Adams et al. 2011). Sonar footage was analyzed for observations of targets $\geq$ 40 cm. Every observation was measured for length and classified to species. In 2019, 143 days of footage were acquired through deployments in the Ventura River. Detection efficiency was consistently high across 2019. 274 observations of targets $\geq$ 40 cm were recorded at the Ventura River site. One *O. mykiss* and 273 Common Carp *Cyprinus carpio* were recorded was recorded in the Ventura River.

Common Carp data were further analyzed to explore means of differentiating between Carp and *O. mykiss*. This included examining differences in swimming motion, morphological features, and effects of environmental conditions (e.g. stream flow) on behavioral patterns. In contrast to previous years carp detections per hour were unimpacted by diel patterns and were loosely negatively associated with stream flow. As monitoring continues a stronger relationship between environmental factors and species identification may emerge. Overall, this project demonstrates the efficacy of sonar cameras for steelhead abundance monitoring in southern California streams. Continued refinement of deployment methods are recommended for maximizing DIDSON utility in southern California conditions.

Due to sampling concerns related to the low abundance of *O. mykiss* in southern California watersheds and the potential for patchy distribution, spawning surveys were conducted as a complete census of available habitat. During the 2018-2019 winter seasons, surveys were conducted from January through May on a bi-weekly basis on 18 survey reaches covering 41 stream miles when environmental conditions allowed.

Spawner surveys conducted during this season were hindered by a above average rainfall and flow events that limited site access and surveyor effectiveness due to poor water clarity. No *O. mykiss* or *O. mykiss* redds were observed over the 2018-2019 winter season. Multiyear comparisons were performed for visibility during surveys, redd life (i.e., the duration of time redds remained detectable), redd area, total redd count, total number of bankside observations and spatial distribution of *O. mykiss* in the Ventura River basin. Visibility, redd life, and redd area varied significantly across years included in the analyses. Post hoc tests were performed to determine which years varied significantly from others.

INTRODUCTION

Southern California steelhead trout (*Oncorhynchus mykiss*) populations have declined dramatically throughout their historic range. Consequently, steelhead trout inhabiting the area from the Santa Maria River to the U.S.-Mexico border have been listed as a federally endangered distinct population segment (DPS) (ESA; NMFS 2012). The ESA mandated recovery plan outlines goals to ensure the persistence of viable populations of anadromous *O. mykiss* across the DPS (NMFS 2012). The California Department of Fish and Wildlife has developed a framework to implement monitoring to track recovery progress (i.e., Fish Bulletin 180) (Adams et al. 2011). This framework is based on assessment of four viability metrics (i.e., Viable Salmonid Population parameters) comprised of abundance, productivity, spatial structure, and diversity (McElhany 2000, NMFS 2016).

To assess abundance for populations encompassed by the DPS, unbiased estimates of anadromous adults are required (NMFS 2016). Fish Bulletin 180 suggests the use of sonar cameras (i.e., DIDSON) to collect counts of anadromous adults in focal streams. Developed by Sound Metrics, DIDSON produces near video-quality imagery and allows for data collection night and during periods of high flow and turbidity (Sound Metrics 2018). These conditions are commonplace in southern California where flows are highly episodic resulting in dynamic hydrology. Additionally, DIDSON cameras allow for the passive collection of data and avoids alterations to steelhead behavior or inadvertent harm to a listed species (Pipal 2012).

To provide an index of effective population size (abundance) and an estimation of spatial structure Redd surveys were conducted as a complete census of available spawning habitat in the Ventura River watershed. Previous studies using redd surveys have demonstrated that steelhead and resident trout redds can be distinguished by redd size (Zimmerman and Reeves 2000). This information can provide insight into the complex interplay between resident and anadromous life history strategies in focal watersheds. Furthermore, pairing redd counts with abundance data provided by DIDSON allows calibration of a redd to spawner ratio to be used in nearby stream systems without counting stations thus fulfilling a critical role of life cycle monitoring stations (LCM). Finally, redd surveys can provide data on the timing and habitat for spawning activity that may guide management and restoration decisions.

This report summarizes the methodologies and findings of sonar deployment, redd surveys, data collection, and data analysis efforts along the Ventura River. The Ventura River is designated in the southern California steelhead recovery plan as being the first focus for recovery action (NMFS 2012). Findings will aid in the development southern California specific monitoring protocols, and will inform resource managers on the status of steelhead populations in these high priority systems.

Study Site - Ventura River

The Ventura River watershed consists of mountainous, high peak elevations that transition into a lower elevation coastal terrace before reaching the Pacific Ocean (NMFS 2012). It drains roughly 227 square miles and contains approximately 35 miles of anadromous water (NMFS 2012). Both the Casitas and Matilija dams act as total barriers to steelhead passage and prevent migration to spawning and rearing habitat in the upper watershed (NMFS 2012) (Figure 1). The Robles Diversion, located on the Ventura main stem 1.5 miles downstream of the confluence of Matilija and North Fork Matilija Creeks, diverts flow from the Ventura River to Lake Casitas and contains a fish-passage facility (NMFS 2012). The river flows through the cities of Ojai, Casitas Springs, and Ventura in Ventura County, California. The DIDSON deployment site is located five stream miles upstream of the Pacific Ocean (Figure 2).

Stream flows are highly dependent on rainfall and extensive sections of the main stem exhibit intermittent flows and drying over spring and summer. A seasonal sandbar prevents access to the watershed until the first large storm event of the season. Without consistent precipitation, flows drop quickly and access to the perennial upper watershed may be limited to short periods.

In 2018 the Thomas Fire burned 281,893 acres (InciWeb 2018, Figure 2) which included substantial portions of the Ventura River watershed (Klose 2018). A high intensity rain event following the fire lead to widespread flooding and debris flows resulting in road closures.

Areas of the Ventura River accessible to anadromous adult *O. mykiss* are inhabited by a number of native and invasive fish species. Native fish species consist of Threespine Stickleback (*Gasterosteus aculeatus*), Prickly Sculpin (*Cottus asper*), Pacific Lamprey (*Lampetra tridentate*) and Arroyo Chub (*Gila orcutti*) and Tidewater Goby (*Eucyclogobius newberryi*) (Walter 2015). Invasive fish species present consist of Common Carp (*Cyprinus carpio*), Black Bullhead (*Ameiurus melas*), Channel Catfish (*Ictalurus punctatus*), Green Sunfish (*Lepomis cyanellus*), Fathead Minnow (*Pimephales promelas*), and Largemouth Bass (*Micropterus salmoides*).

METHODS - DIDSON

Data Collection

This site's channel profile and substrate composition allow unobstructed views of the full stream channel. The site is located on property owned and operated by the Ojai Valley Sanitation District. The storage container, from which cameras are operated, is situated within a perimeter fence and behind two locked, electronic gates. Sonar cameras were deployed once unobstructed surface flow was established between the site and the ocean and remained deployed as long as conditions allowed for fish migration from the ocean to the monitoring site including periods of heavy rainfall and high flow. Additionally, cameras remained deployed following river mouth closures to allow time for any fish that had entered the system time to migrate to monitoring sites.

A standard DIDSON 300 m unit (Sound Metrics, Lake Forest Park, Washington) and an ARIS Explorer 3000 (Sound Metrics, Lake Forest Park, Washington), both operating in high frequency mode (1.8 MHz and 3.0 MHz respectively), were used for all deployments in the Ventura River from January 7th, 2019 through May 30th, 2019. Cameras were deployed in parallel to compare functionality under southern California stream conditions (Figure 3). The DIDSON was housed in a Sound Metrics manufactured silt box to prevent lens fouling and a custom aluminum box to prevent damage by floating debris and attached to the X2 pan and tilt rotator (Sound Metrics, Lake Forest Park, Washington) to aim the camera remotely. The camera and X2 were affixed to a steel, sled foot A- frame mount as described in Larson 2013. The ARIS was housed in a custom stainless-steel box to protect against damage by floating debris. This assembly was attached to an AR2 dual-axis pan/tilt rotator to aim the camera remotely. The AR2 was then affixed to an A-frame mount. Both A-frames were held in place by gravel bags placed on their sled feet, tethers running from the A-frames to adjacent t-posts, and by Duckbill Earth Anchors set into the substrate. To safeguard against theft and potential loss of equipment during peak flows, cameras were tethered to large nearby trees and to an earth anchor installed outside the stream channel (Figure 4). As an added layer of security, motion detecting trail cameras were installed. Deflection panels, consisting of aquaculture mesh fastened to PVC frames, were anchored upstream and downstream of the cameras on both banks. These panels both prevented fish from passing behind cameras and guided them to an optimal imaging range.

The DIDSON camera was connected to a topside control box via a 60 m DIDSON cable while the ARIS was connected to a command module via a 150 m ARIS cable. Both were connected to Dell Toughbook laptops running DIDSON and ARIS software respectively to interface with the camera and adjust record settings. Focus and frame rate were set automatically by the software based on camera settings. Gain was left at the default maximum value. DIDSON camera window length was set to either 10 m or 5 m depending on stream channel width. ARIS window length was kept at 5 m for all deployments to keep the camera operating in high frequency mode. Footage was captured in 20-minute files and written to external hard drives. Topside electronics and components were powered through uninterruptable power sources connected to permanent onsite power. Site visits were conducted on a daily basis to ensure proper camera operation. Prior to removing the cameras, walking surveys were conducted to verify that steelhead migration from the ocean to the camera location was no longer feasible.

Flow data for the Ventura River site were obtained from a U.S. Geological Survey (USGS) stream gauge located 0.7 miles upstream of the DIDSON site. The proximity, and lack of any substantial flow inputs between the DIDSON site and flow gage, make this value a reasonable approximation of flow at the deployment site.

Data Analysis

Sonar files were processed using the echogram function with background subtraction enabled in the Sound Metrics software. Echograms produce a visual representation of the entire file by compressing a given frame into a single pixel width along the full image range and background subtraction allows static objects to be filtered out (Sound Metrics 2012). These processes make moving objects easier to detect and expedites footage review.

All wildlife observations greater than 40 cm in length were recorded which was considered the minimum size needed to assign species and corresponds with the California Department of Fish and Wildlife's listed lower size limit for steelhead of 40 cm (California Department of Fish and Wildlife 2017).

Targets were measured using the box method per Pipal et al. (2010). The box method requires the reviewer to pause the footage, drag a box around the object seen in frame, and record the value for either the diagonal or width depending on the object's orientation relative to the camera (Pipal et al. 2010). For each observation, up to three measurements were taken from separate frames and then averaged as described in Pipal et al. 2010. Reviewers assigned species to fish observation based on behavioral and morphological cues. In instances where this was not feasible, observations were classified as "unidentified fish species". Observations of non-target aquatic species were designated as either "unidentified terrestrial species", "frog", "turtle", "waterfowl", "snake", "unidentified fish species" or "unknown". In instances where a reviewer was unsure of species designation, files were flagged for further review by a more experienced reviewer. For cases where this occurred in the Ventura River, footage from an ARIS camera or a DIDSON with an alternate view was consulted when available, before determination of species was finalized. For each observation, length, direction of travel, species, range from the camera to the target, timestamp, footage quality, confidence in species designation, and pertinent metadata (e.g. site location, date of recording, filename, reviewer name, and date viewed) were recorded.

Ten percent of footage analyzed by each staff member was randomly selected and checked for accuracy by an experienced biologist. Data were entered into an Access database where data rules were enforced to limit entry errors. Data proofing was completed using R software (R Core Team. 2016) to flag potential erroneous values, which were either corrected or omitted as appropriate.

Sonar detection efficiency (D_E) was calculated for each sonar deployment event. This was calculated by subtracting the amount of time the camera's range was limited (T_D) from the total deployment time (T_T) and then dividing by total deployment time.

$$D_E = (T_T - T_D) \ / \ T_T$$

For anadromous *O. mykiss* observations, net movement is calculated to estimate escapement. To calculate net movement (N) for focal species, the total number of downstream observations (D) were subtracted from the total number of upstream observations (U) as recommended by Xie et al. (2002) and put into practice by Larson (2013).

$$N = U - D$$

A net positive number would indicate net movement upstream and vice versa. Considerations for potential confounding of counts by downstream movement of post-spawning adults (i.e., kelts) would be addressed on a case-by-case basis.

Two species in the Ventura River (i.e., Common Carp and Pacific Lamprey) have the potential to be misidentified as steelhead due to the overlap in spatial distribution, temporal cycles and range in typical adult lengths. To learn more about these species, and how they may be differentiated from steelhead; additional analyses were done for Common Carp observations in the Ventura River. These observations were compared with synchronous flow data to explore the effect of flow on movement patterns. Additionally, Ventura River Common Carp observations were binned by hour of the day and classified as either "day" (i.e., the hours from 0600 to 1800) or "night" to characterize patterns in diurnal rhythms. The mean observed length and mean daily count of Common Carp > 40 cm were reviewed. All analyses were completed using R software.

RESULTS - DIDSON

Impacts of the Thomas Fire continued to influence data collection efforts in 2019 as the large amount of sediment mobilized by post-fire rain events continued to move through the system. This reduced sonar visibility during moderate flow events and deposited sediment in the camera housing during high flow events requiring removal from the water for cleaning.

One hundred and forty three days of sonar footage were recorded from January 7th, 2019 to May 30th, 2019. Detection efficiency remained high ($\geq$ 90%) in the Ventura River. A total of 274 observations of targets $\geq$ 40 cm in length were recorded. Of these observations, 1 was identified as *O.* mykiss and 273 were identified as Common Carp. The *O. mykiss*, measuring 62 cm, was observed migrating upstream on

February 28th at 21:53. This fish was observed during an increase in flow occurring earlier that day (Figure 5).

58% of Common Carp observations occurred under relatively low flow conditions ($\leq$ 40 ft^3s^{-1}) with the remaining observations split between moderate (40-100 ft^3s^{-1}) and high (>100 ft^3s^{-1}) flow conditions (Table 1).

For Common Carp $\geq$ 40 cm, the mean $\pm$ SE length was 55.28 $\pm$ 0.32 cm. This was not significantly larger than those carp observed in 2018 (46.84 $\pm$ 0.05 [mean $\pm$ SE]). (Welch Two Sample t-test results: $t = -19.108$, $df = 281.41$, p-value = 1). Carp observations in 2019 were not related to diurnal patters as an approximately equal number of observations occurred in each condition (day = 107, night = 101)

METHODS – REDD SURVEYS

Data Collection

Spawning ground surveys were conducted in accordance with standardized protocols developed by California Department of Fish and Wildlife scientists for southern California as part of the CMP (McLaughlin and Christianson 2016). Surveys were conducted from December through May of each year. In the Ventura River watershed, 18 reaches encompassing 41 stream miles of potential spawning habitat were surveyed (**Error! Reference source not found.**). Of these, 34.5 miles were accessible to anadromous adults, while 8.2 miles were above the Matilija Dam where resident populations are known to exist. Recent studies have demonstrated the genetic similarity between populations above and below fish passage barriers within the same system (Abadía-Cardoso et al. 2016; Clemento et al. 2009). We therefore considered these above-dam resident populations to be important to steelhead monitoring efforts.

Individual reach designations were determined by the sampling frame currently under development by CDFW CMP and Fisheries Branch biologists. Reaches began and ended at easily identifiable landmarks (e.g. bridges or stream confluences) and were designed to be completed in a single day. These reaches ranged in length from 1.3 to 4.5 stream miles with a mean of 2.5 $\pm$ 0.17 miles (mean ± SE).

Surveys were conducted every two weeks throughout the survey season when stream flows, and weather permitted. Two weeks is the minimum amount of time redds remain detectable in southern California stream systems (R. Bush, National Marine Fisheries Service, unpublished data). Teams of two to three surveyors walked reaches in an upstream direction recording observations on handheld data recorders. All fish observed were identified to species. For each *O. mykiss* observation, a total length estimate, location, and life history stage (when possible) were recorded.

When redds were first observed, measurements were taken for pot and tail spill dimensions. Pot length, width, and depth relative to the adjacent streambed were measures. For tail spill dimensions, we measured the tails spill length and width measurements were taken at 1/3 and 2/3 the distance from the upstream end of the tail spill. Dominant substrate size was also recorded for both the pot and tail spill. Redds were identified with a flag denoting the redd record number, distance and bearing of the redd from the flag location, date the redd was first identified, and redd age. Redd ages were categorized as the following: 1 - New since last survey, 2 - Previously identified and still measurable, 3 - No longer measurable but still visible, and 4 - No redd apparent. Redd ages were updated and recorded during subsequent observations. Redds were re-measured when pot and tail spill dimensions had noticeably changed following their initial observation.

Data Analysis

To examine survey frequency, we calculated the mean ± SE number of days between surveys. Water visibility was examined as a metric for redd detectability. Visibility measurements were classified as either "clear" (i.e., visibility = 100%) or "not clear" (visibility < 100%) for each survey. The mean ± SE number of surveys where visibility was 100% was calculated by reach and by watershed for each year.

Redd observations would have been mapped using ArcGIS 10.1 (ESRI, Redlands, California) and R Software (R Core Team. 2016). We would have calculated area and total redd length for each redd. Redd area would be calculated as the sum of pot and tail spill areas per Gallagher et al. (2007). Total redd length would be calculated as the sum of the pot and tail spill lengths. Redd dimensions and area would be used to compare the relative sizes of all redds observed and to indicate those produced by anadromous *O. mykiss*. Finally, we would have examined trout observation counts by size class and their spatial distribution by mapping observation locations.

To characterize any potential changes over time, we compared data collected from surveys conducted in the Ventura River in 2019 with data collected from 2013 through 2018. We examined redd counts, redd area, redd life (i.e., the duration of time redds remained detectable), O. *mykiss* bankside observation totals and visibility versus year for trends per Gallagher (2005). When evaluating redd and trout count data, comparisons were drawn from data where survey effort was consistent (i.e., years 2015 – 2019). Analyses of redd life by year were only performed for redds in years where a final status indicating that they were no longer visible (i.e., redd age 4) was recorded. All redds recorded from 2013 – 2019 were included in our examination of redd area versus year due to low sample size. Visibility comparisons were also drawn between all years from 2013 – 2018. All analyses were completed using R software.

RESULTS – REDD SURVEYS

Ninety-one redd surveys were conducted over 18 reaches from January 9th to May 20th, 2019 during the 2018–2019 spawning season in the Ventura River basin (Table 2). Above average rainfall impacted data collection efforts in the Ventura River basin in the form of access limitations and high turbidity. The number of days between surveys was typically greater than 14 with a mean ± SE of 18.28 ± 0.4 days (Table 2). This mean is influenced by the outlying survey frequency on reaches Ventura 1 and 2 where prolonged high flows limited access (37 days between surveys). With regards to water clarity, 83.5% of surveys were completed with 100% visibility (Figure 5).

No redds and no bankside observations of *O.mykiss* were observed in the Ventura River watershed during the 2018-2019 survey season (Table 3).

Multiyear Comparisons

The proportion of surveys where visibility was 100% varied significantly in the Ventura River watershed among years 2013 – 2019 ($\chi 2 = 46.23$; d.f. = 5; $p < 0.001$) (Figure 5). Post-hoc pairwise testing revealed that 2018 alone differed significantly from all other years ($p < 0.001$).

An analysis of variance showed there was a significant effect on estimated redd life by survey year from 2014 – 2017 in the Ventura River basin (ANOVA: f = 6.1; d.f. = 1, 30; $p < .05$) (Figure). A pairwise t-test conducted using Bonferroni adjusted alpha levels ($\alpha = .008$), showed that estimated redd life only varied significantly between years 2015 and 2017 ($p = 0.004$).

Redd area was found to vary significantly between years 2013 – 2018 in the Ventura River (ANOVA: $f = 9.26$; d.f. = 4, 122; $p < .01$) (Figure). Post hoc analysis indicated that 2017 was the only year where redd area was significantly different from all other years, with the exception of 2013 (Tukey multiple comparisons test, $p < 0.05$) (Table).

The mean $\pm$ SE total number of bankside observations for years 2015 – 2017 was 281 $\pm$ 39 trout in the Ventura River. The total number of *O. mykiss* bankside observations decreased by two orders of magnitude in 2018 ($n = 2$) and dropped further in 2019 ($n = 0$) (**Error! Reference source not found.**).

Spatial distribution of *O. mykiss* bankside observations across all years in the Ventura River (i.e., 2013 – 2019), show the majority of observations (i.e., 94%) were recorded in the upper watershed (i.e., upstream of the confluence of North Fork Matilija Creek and Upper Matilija Creek). Redd observations recorded during the same timeframe exhibited a similar distribution pattern with 88% of redds being observed in the upper watershed (**Error! Reference source not found.**).

DISCUSSION

DIDSON

This project demonstrates the efficacy of sonar cameras for steelhead abundance monitoring in southern California streams. One *O. mykiss* measuring 62 cm was observed moving upstream on February 28th at 21:53. Visual confirmation of a steelhead approximately the same length was provided by Scott Lewis (Fisheries Biologist – Casitas Municipal Water District) at the Robles Diversion during the following week (Lewis 2019). Subsequent walking surveys provided no additional observations.

Detection efficiency remained consistently high (~90%) despite challenging environmental conditions (e.g., high turbidity, high flow, dynamic channel morphology, and fine sediment). This suggests that our findings are reflective of true steelhead abundance in focal streams, rather than a function of methodological limitations. Given the scarcity of southern California steelhead in these watersheds, the use of passive data collection methods that do not alter or otherwise negatively influence potential spawning activity (i.e., DIDSON and ARIS), will likely remain at the forefront of preferred approaches to tracking adult abundance. Continued refinement of methods to differentiate between steelhead and non-target species and development of deployment methods that will allow sonar deployment during peak flows will be critical steps toward maximizing DIDSON and ARIS utility under southern California conditions.

Environmental Challenges

In previous years cameras were removed prior to large storm events to protect against loss or damage of equipment. Re-deployment had to be delayed until project personnel could safely work in and around the stream channel (approximately 400 cfs). This led to interruptions in data collection during high flows when steelhead may migrate (McEwan 2001). To limit any potential observational bias imposed by stream flows additional security anchors were installed and the cameras remained deployed through all flow events in the 2019 winter season. In theory this would allow continuous data collection during flow events outside an initial loss in sonar range caused by a pulse in turbidity associated with rapidly increasing flows. In practice the camera housing filled with extremely fine sediment following each flow event jamming the focus arm blocking the camera from transmitting. Cameras remained non-operational until staff were able to take apart and clean the camera housing. This created a similar interruption in collection as cleaning was delayed until flows receded to a safe working level. The presence of fine sediment and increase in sedimentation is most likely a temporary problem stemming from the Thomas Fire as opposed to a side effect of high flow deployments. Pre-fire deployments under moderate flows (~3,000 cfs) along the Ventura River and higher flow deployments in nearby unburned watersheds (Salsipuedes Creek ~7,000 cfs) experienced no fine sediment impairment (CDFW unpublished data).

Outside high flow events where fine sediment disabled camera operation entirely, elevated turbidity limited sonar camera effectiveness. When turbidity reached extreme levels (> 400 NTU) the sonar's range dropped to < 1 meter (Santa Barbara Channel Keeper 2018). Image range would then gradually increase as suspended sediments settled out of the water column. These periods were brief (i.e., < 48 hours) and followed peak flows. To the best of our knowledge, the limiting effects of turbidity on acoustic camera functionality are unavoidable. Further understanding of the relationship between turbidity and sonar effectiveness is needed. To assess this relationship, continuous turbidity readings, collected concurrently with sonar deployments, are needed. These data can be compared with sonar imagery to establish mathematical relationships between the two. This will allow for a quantifiable characterization of turbidity effects on sonar data collection and a better understanding of the implications for observational bias.

Operational Findings

The side-by-side operation of DIDSON and ARIS in the Ventura River site identified some key differences in performance under dynamic conditions. The DIDSON was more consistent in providing clear images during times of increased turbidity, while the ARIS' maximum viewing range was considerably decreased during these same periods. When turbidity was not a factor; however, ARIS imagery was noticeably superior to DIDSON. This was expected given that ARIS resolution is almost twice as high (1.8 and 3.0 MHz respectively). This increased resolution is the reason for ARIS' limitations regarding range and turbidity because higher frequency sound attenuates more rapidly with distance and increased suspended particulate concentration (Maxwell and Grove 2008). ARIS' increased resolution also has a direct effect on file size. Operating under similar settings ARIS files are typically three times larger than DIDSON files of the same length. This increased file size complicated file storage and added to the amount of time required to process files during analysis. For this reason, as well as the greater consistency in image quality and target detection, DIDSON footage was primarily used throughout the review process for the Ventura River site. ARIS files were reviewed in instances when a large fish was observed and species could not be confidently determined based on DIDSON footage alone. The dual deployment of DIDSON and ARIS cameras had the additional benefit of providing redundancy. In the event one of the cameras experienced technical difficulties, footage from the other camera was still available. This kept gaps in data collection to a minimum.

Species Determination

The most considerable and well documented challenge posed by DIDSON monitoring in project systems is species identification (Pipal et al. 2012, Burwen et al. 2007, Burwen et al. 2010). A number of methods have been suggested to address this problem, including the use of tail-beat frequencies (Mueller et al. 2010), acoustic shadows (Langkau et al. 2012), paired optical video system (Killam 2012), and trapping methods (Denton et al. 2015). The feasibility of each option is under review for southern California steelhead monitoring applications. The current means of differentiating between species relies on evaluating swimming behavior and body morphology on a case-by-case basis. This method is problematic because it is subjective and depends heavily on reviewer experience. In watersheds where species overlap in size with steelhead (e.g., *C. carpio* in the Ventura River) this project sought to explore additional means of differentiating between species by further examining observable characteristics such as swimming motion and morphological features. Relationships between stream conditions (i.e., flow), temporal distribution (i.e., time of day observed) and behavioral patterns were explored as suggested by Pipal et al. (2012). In contrast to previous years (2018 - 99% of carp observations occurring under 40 ft^3s^{-1}) Common Carp were observed under moderate to high flow conditions in 2019. While a higher proportion of carp are observed under low flow conditions it appears stream flow itself cannot be used as a means of distinguishing *O. mykiss* from *C. carpio*. Previous assessments of diel patterns in Carp movement in the Ventura River suggested a tendency towards nocturnal activity although data collected in 2019 does not support this assertion. As more data is collected a more defined relationship between the diel rhythms, stream flows, and *C. carpio* observations may emerge.

The large number of *C. carpio* observed in the Ventura River provided abundant opportunities to develop an understanding of Carp swimming behavior, morphological characteristics and behavioral responses to environmental conditions. Unfortunately, DIDSON footage of southern California *O. mykiss* is scarce. Project staff was able to obtain footage of large *O. mykiss* in non-study watersheds which has been helpful in providing a limited basis for comparison between the species but cannot substitute for imagery obtained of both species under identical conditions. Species designation will remain a problem until more steelhead imagery is obtained, more definite metrics for species determination can be evaluated, or methods of confirmation can be implemented (i.e. trap or optical system).

REDD SURVEYS

The 2018-2019 spawning season was characterized by above-average rainfall (Ventura Watershed Protection District), resulting in peak flows reaching 16,000 $ft^3\ s^{-1}$ in the Ventura River main (USGS 2018). Elevated flows led to persistent turbidity and increased depths preventing the completion of scheduled surveys during the months of February and March. This interruption in sampling coincided with what has historically been an active spawning period in the Ventura River watershed (**Error! Reference source not found.**) and may have influenced redd counts.

No redds were recorded in the Ventura River basin in 2018 – 2019 season. It is possible that the storm events obscured or destroyed redds during peak flows in February and March (Figure 4). Increased flow led to more connectivity in the lower watershed than in previous years. Migration corridors allowing anadromous individuals to reach spawning habitat in San Antonio Creek and North Fork Matilija Creek remained passable into late spring. While one anadromous fish was observed on DIDSON none were observed during walking surveys.

No *O. mykiss* were observed during spawning surveys conducted in the Ventura River watershed in the winter 2018-2019. While elevated turbidities were problematic in the 2017-2018 they may have been a secondary effect of the Thomas Fire due to the destabilization of stream hillslopes resulting from loss of vegetation (Klose 2018). The proportion of surveys conducted under sub-optimal conditions in 2019 returned to a pre-fire average (2013-2017)(.165, n = 15, sub-optimal conditions)(Figure 5). As banks revegetate in the years post fire we expect the 2017-2018 turbidity levels to remain an outlier. Limited visibility directly following flow events may have negatively biased redd counts due to missed observations. However, the absence of *O. mykiss* observations later into the year with more stable water conditions supports a lack of spawning activity.

The 95% CI [36.60, 50.90] of the mean estimated redd life (43.8 $\pm$ 3.51 [mean $\pm$ SE]) for all redds observed from 2014 – 2019 suggests a survey frequency of two weeks should allow for detection of new redds before they degrade to an extent where they are no longer visible (Figure 7). In practice; however, this is complicated by high flow events that may erase redds before they can be observed. In cases where high flow events were anticipated, we attempted to survey reaches where the risk for redds to be obscured was highest (i.e., reaches where stream gradient and wildfire burn severity were both relatively high). Continued tracking of redd life in subsequent years will further increase our understanding of redd longevity in southern California systems and its implications for survey methodology.

Mean redd areas observed in the Ventura River from 2013 to 2019 ranged from 1.54 ft^2 to 3.39 ft^2 (Figure 8). These sizes are much smaller than the areas of steelhead redd observed in northern California (e.g., 19.15 ft^2) (Gallagher and Gallagher 2005). Based on our findings, and with the support of evidence provided by a sonar count station operated in the lower Ventura River concurrently with redd survey efforts, we believe that all redds observed in the Ventura River from 2013 – 2019 were produced by resident rainbow trout.

When comparing historic data collected in Ventura River to the 2018-2019 spawning season there are considerable differences in *O. mykiss* redd and bankside observations. Zero bankside observations of trout were made during the 2018-2019 winter season. The 2018 – 2019 season represents the second time since redd surveys were initiated in the Ventura River by National Marine Fisheries staff in 2009 that zero redds were observed (R. Bush, National Marine Fisheries Service, unpublished data). These results are likely due to the combined effects of the Thomas Fire and persistent drought conditions. The negative synergistic effects of wildfire and drought have likely lead to the depressed number of *O. mykiss* in the Ventura River basin. Continued monitoring of these systems that have been assigned a high priority for recovery action via redd surveys will be a critical step toward better understanding of southern California steelhead recovery in response to extreme environmental conditions (NMFS 2012).

Figure 2. Ventura basin redd reaches and DIDSON monitoring sites located in Ventura County. UPM1-3 and UNF1 located above the Matilija Dam are surveyed due to genetic similarity between populations above and below fish passage barriers and the planned removal of said dam.

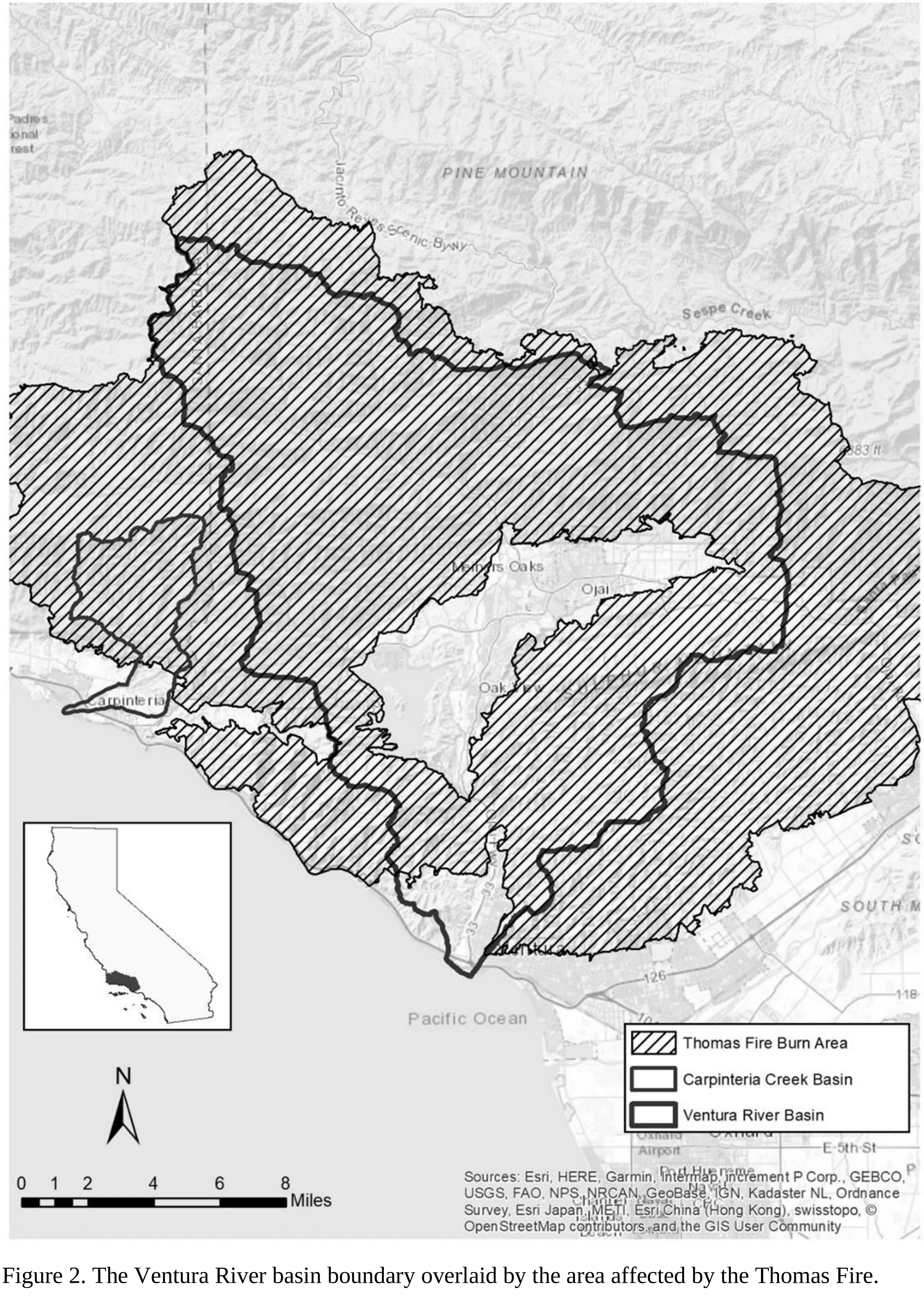

Figure 2. The Ventura River basin boundary overlaid by the area affected by the Thomas Fire.

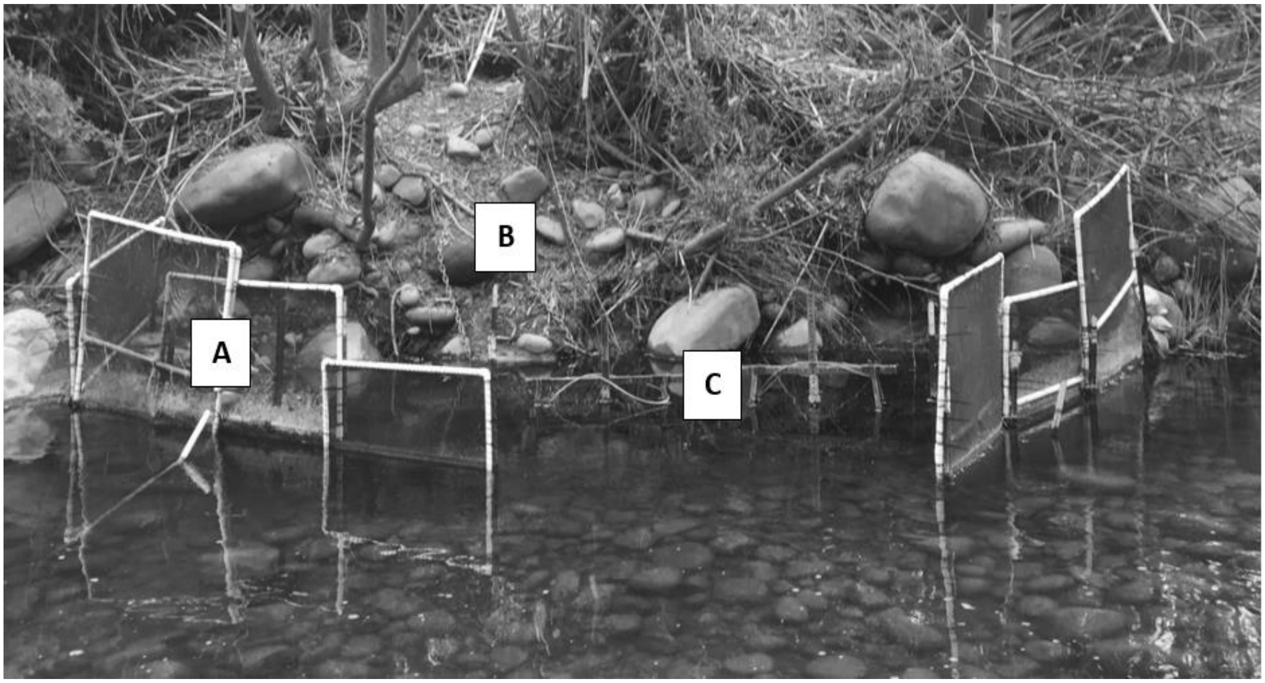

Figure 3. Ventura River monitoring site in 2019 with key features labeled. (A) Deflection panels; (B) security tether; (C) paired deployment of DIDSON and ARIS.

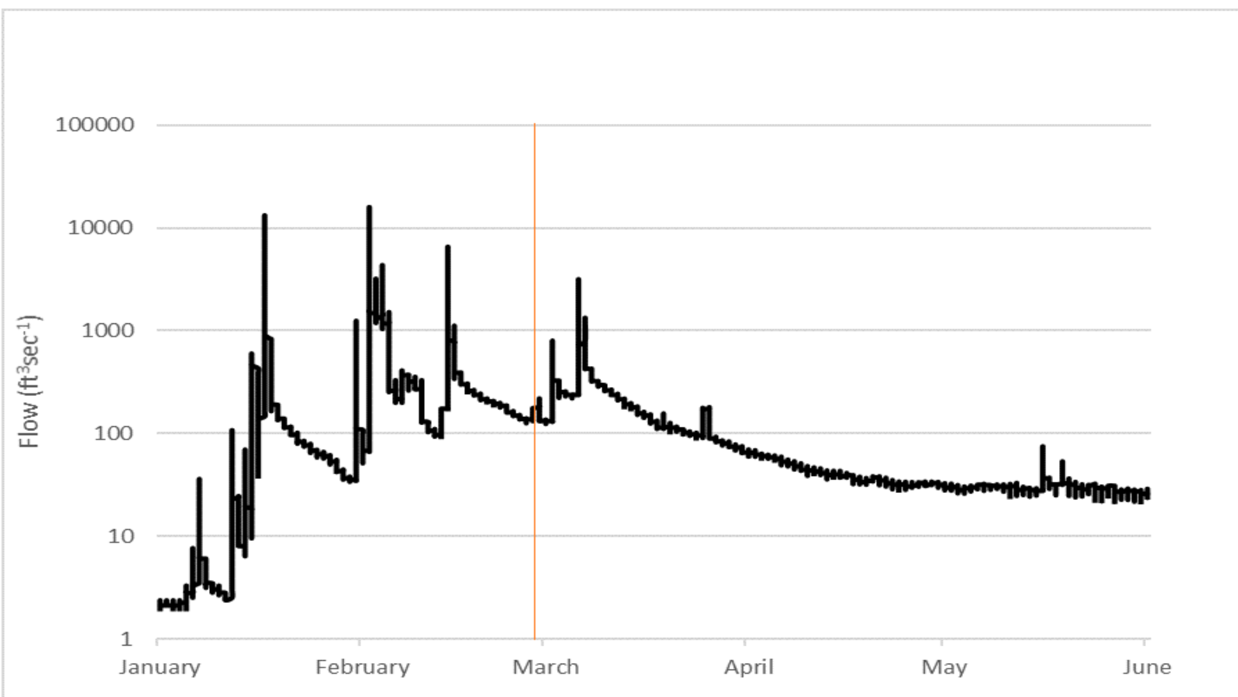

Figure 4. Ventura River stream flow plotted against time for 2019 on a $\log_{10}$ scale. Upstream migration of the only observed steelhead on February 28th indicated in orange.

Table 1. Ventura River Common Carp observations binned by increments of 20 ft3 s -1 for 2018.

Flow Bins (cfs)	*Percent of Observations*	*Cumulative Percent*	*n*
0-20	46.1	46.1	100
20-40	12	58.1	26
40-60	10.1	68.2	22
60-80	7.8	76	17
80-100	1.4	77.4	3
>100	22.6	100	49

Table 2. Redd survey frequency by reach in the Ventura River watershed for 2018 - 2019.

Reach	*N*	*Mean Survey Frequency*	*SE*
Bear Creek	6	16.6	1.08
Matilija Creek 1	4	15.67	2.19
Matilija Creek 2	3	17	3
Matilija Creek 3	3	14	5
North Fork Matilija 1	7	21.83	5.65
North Fork Matilija 2	7	21.83	5.53
Upper North Fork	3	19.5	0.5
Lion Creek	6	14	1.05
San Antonio Creek 1	6	14.6	0.75
San Antonio Creek 2	6	14.6	0.75
San Antonio Creek 3	6	14.6	0.75
Stewart Creek	6	13.8	1.16
Ventura River 1	5	36.75	22.09
Ventura River 2	5	37	22.34
Ventura River 3	5	14	1.29
Ventura River 4	5	14	1.29
Ventura River 5	4	14.67	0.88
Ventura River 5.1	4	14.67	0.88

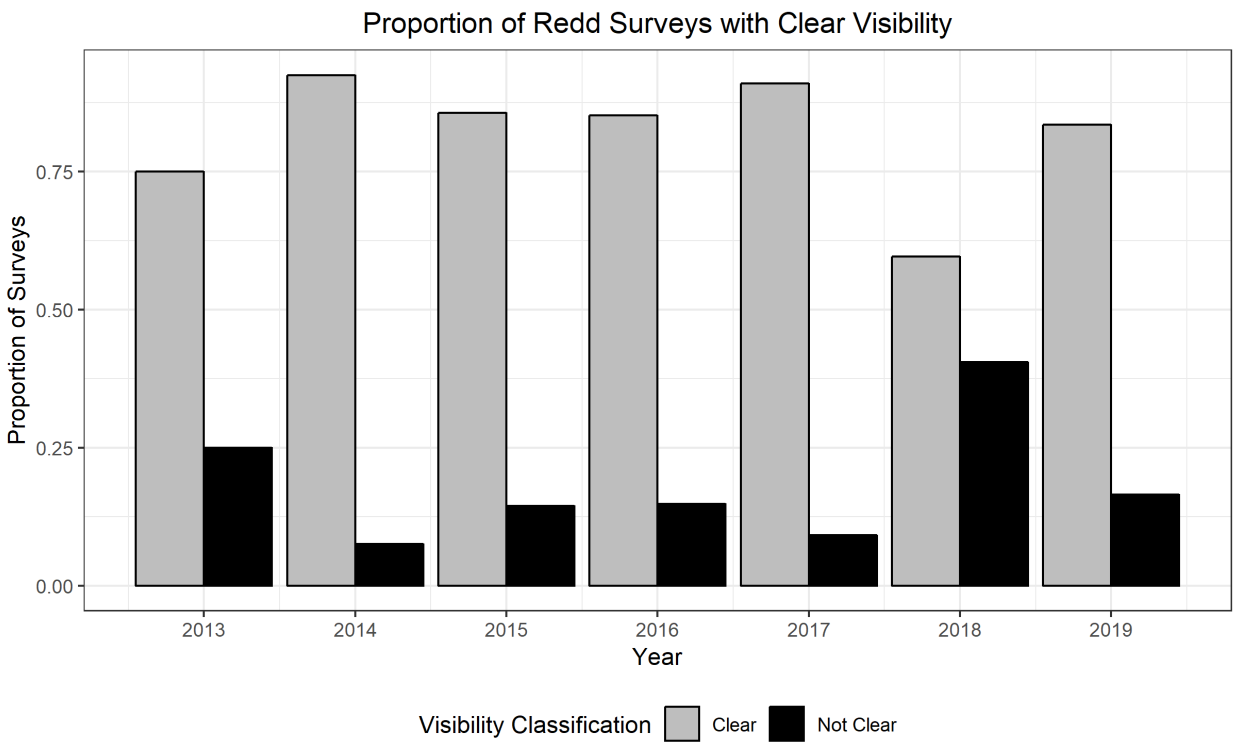

Figure 5. The proportion of surveys where visibility was 100% (i.e., Clear) compared with those where visibility was less than 100% (i.e., Not Clear) for years 2013 to 2019 in the Ventura River basin.

Table 3. Total *Oncorhynchus mykiss* redd counts and bankside observations for years 2015 to 2018 in the Ventura River watershed.

Year	*Redd Count*	*O. mykiss Observations*
2015	26	357
2016	24	227
2017	11	260
2018	0	2
2019	0	0

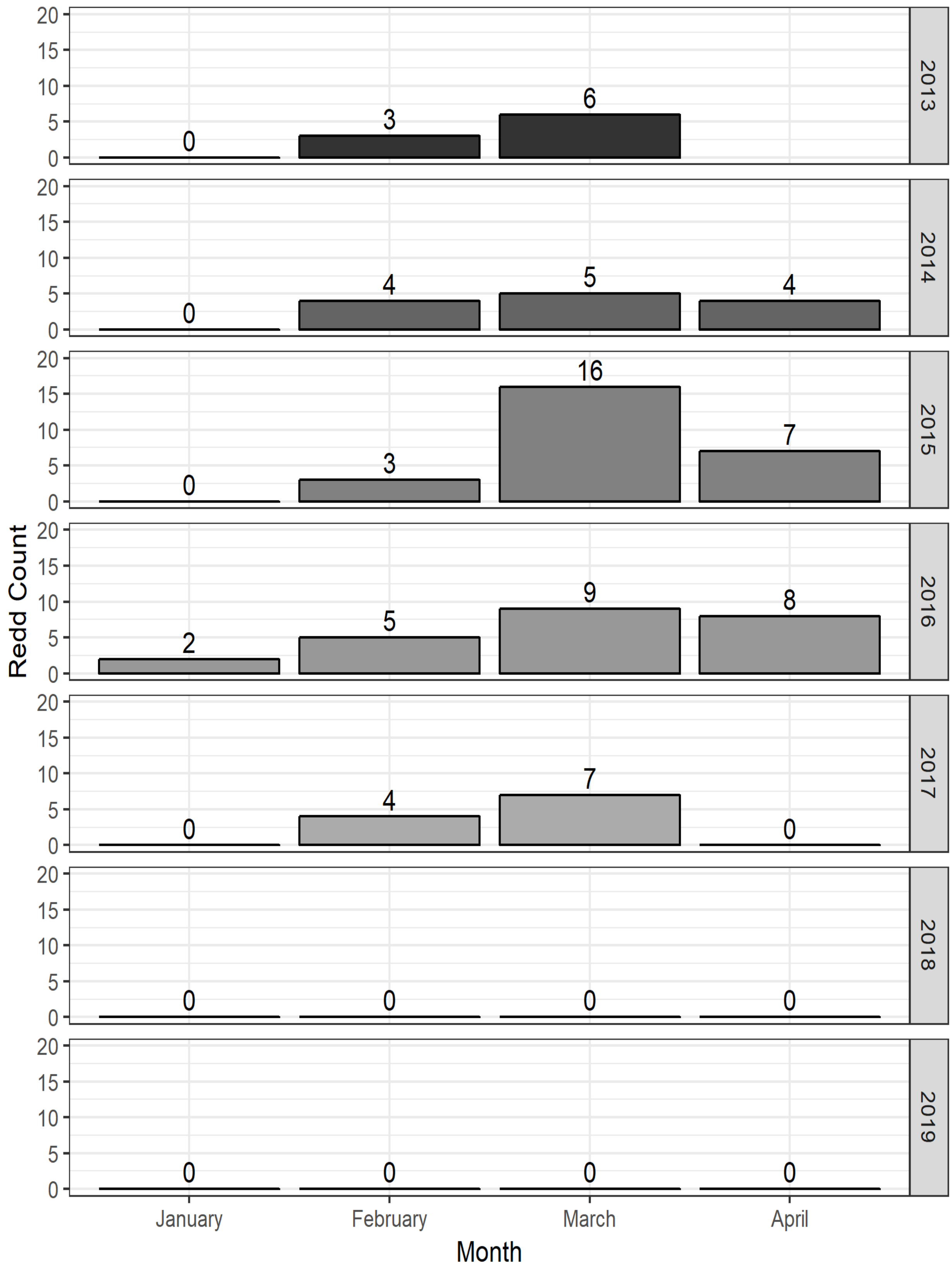

Figure 6. Monthly redd counts for years 2013 – 2019 in the Ventura River watershed.

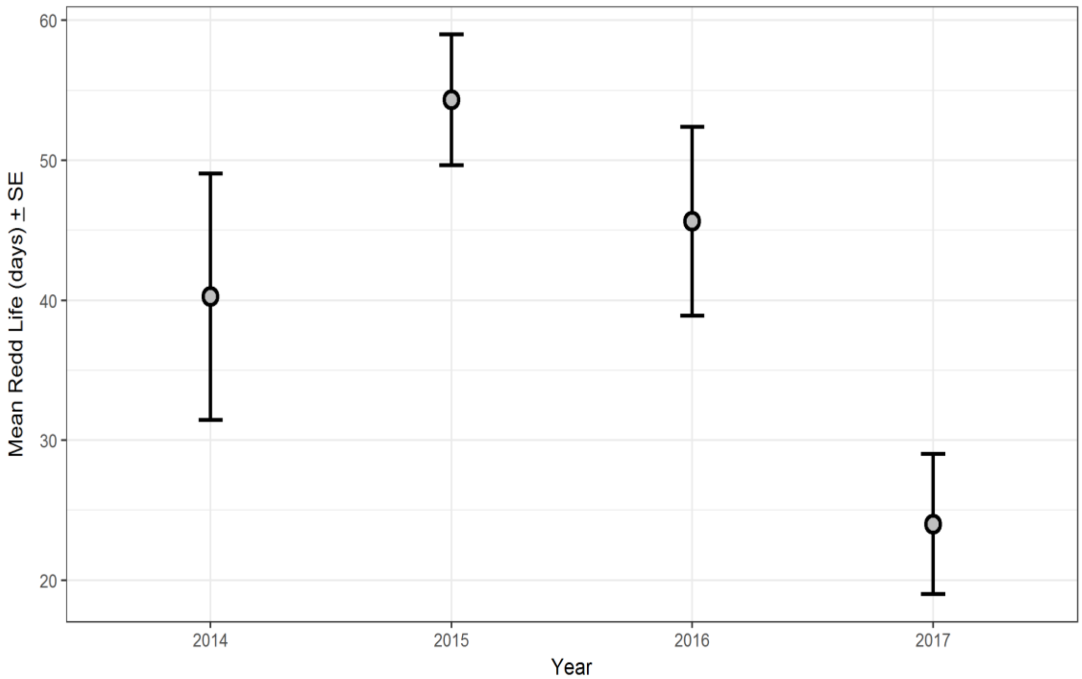

Figure 7. Mean estimated *Oncorhynchus mykiss* redd life in days by year in the Ventura River basin. Vertical bars indicate standard error. No redds were identified in 2018 or 2019 so they have been excluded from this figure.

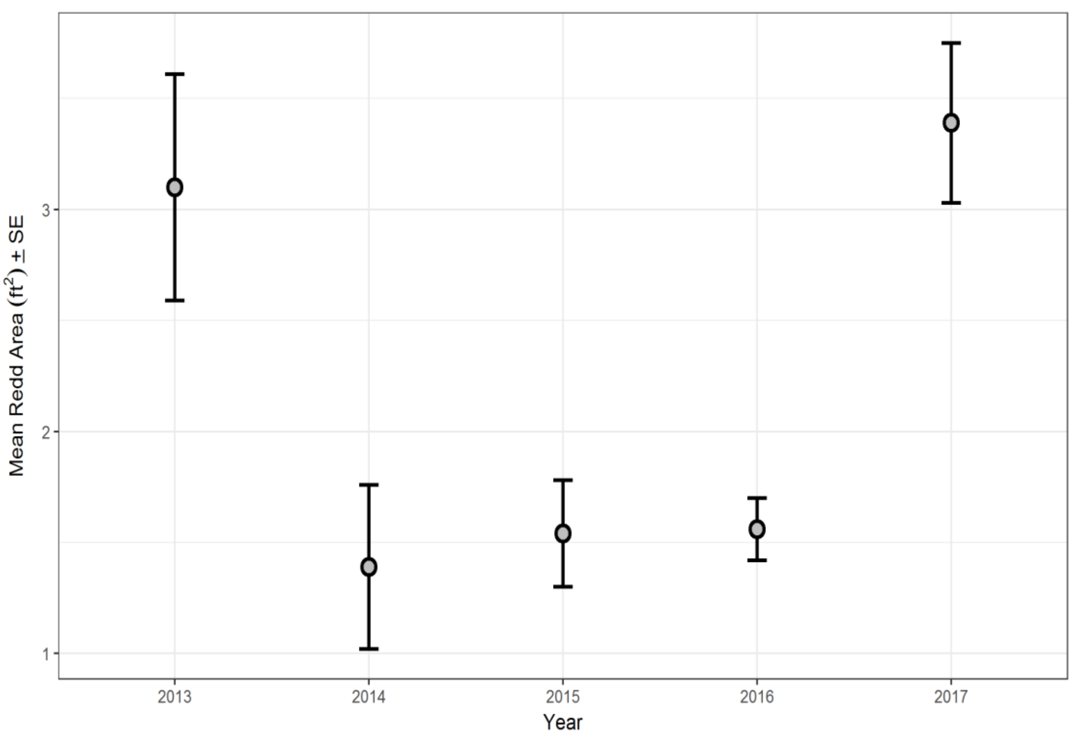

Figure 8. Mean *Oncorhynchus mykiss* redd area in ft^2 versus year in the Ventura River basin. Vertical bars indicate standard error. No redds were identified in 2018 or 2019 so these years have been excluded from the figure.

Table 4. Mean *O. mykiss* redd area for years 2013 – 2017 in the Ventura River watershed. Means followed by the same letter did not differ significantly (Tukey test, $p < 0.05$).

Year	*Mean Redd Area (ft^2)*	*SE*
2013	3.1_{AB}	0.51
2014	1.39_{B}	0.37
2015	1.54_{B}	0.24
2016	1.56_{B}	0.14
2017	3.39_{A}	0.36

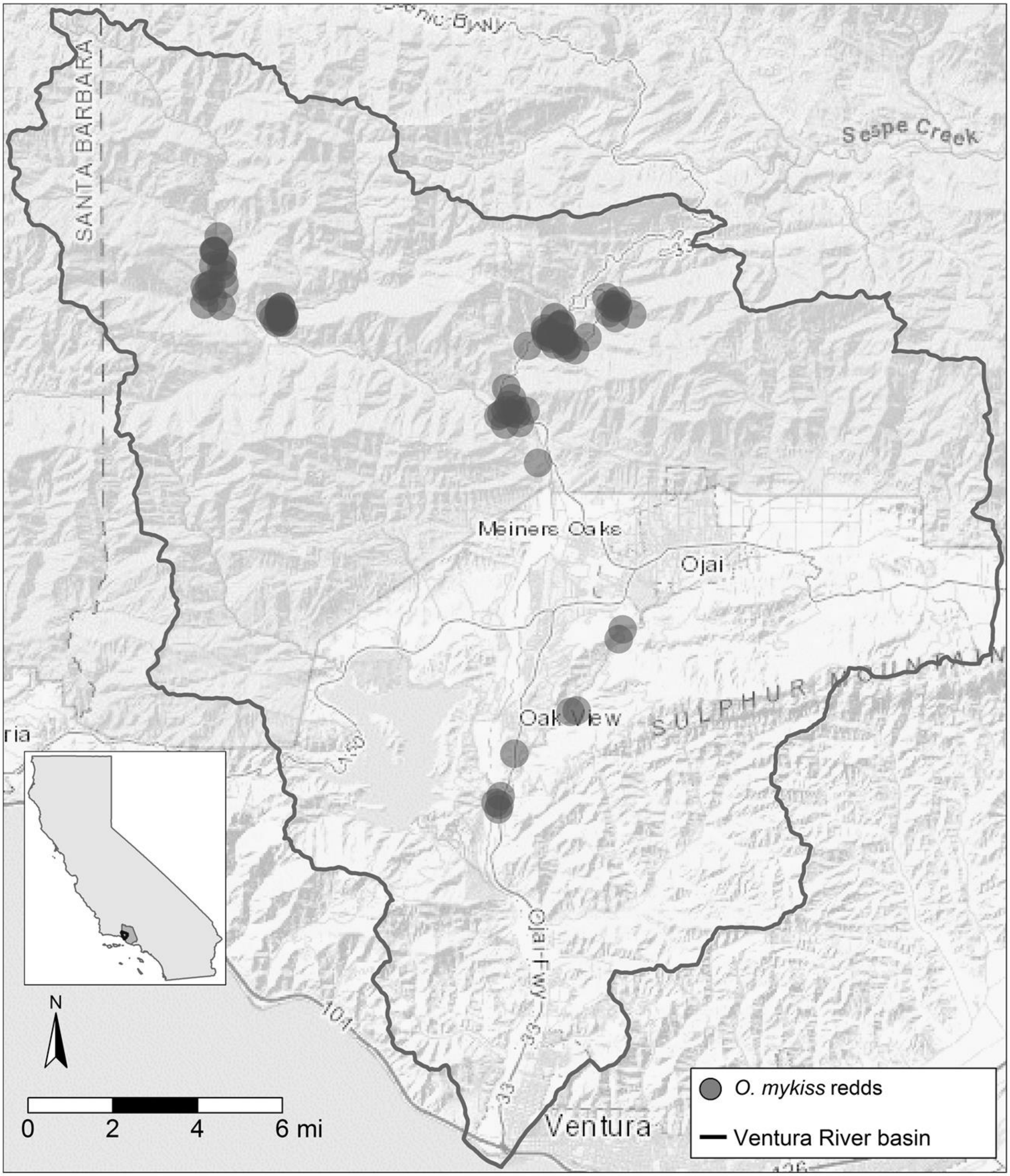

Figure 9. *Oncorhynchus mykiss* redd locations recorded from 2013 – 2019. Random noise has been added to observation positions to offset overlapping points.

ACKNOWLEDGEMENTS

We thank Kathryn Carmody for her assistance with data management and analysis. Data were collected through a cooperative effort by The California Department of Fish and Wildlife, The Pacific States Marine Fisheries Commission, The California Conservation Corps (CCC) Watershed Stewards Program, and the NOAA/CCC Veterans Corps Fishery Program. Field assistance was provided by Katie Carmody, Tanielle Redman, Elizabeth Martin, Danielle Fitts, Casey Horgan, Michael Morales, Zach Hogland, Elijah Mitrano, Kyle Maxwell, Dane St. George, Andre Briscoe, Paige Mathison, Janine Fischer, Dustin Geisen, Griffin Haverland, and Dana McCanne. We gratefully acknowledge the landowners, whose cooperation and willingness to work with our staff made this project possible.

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