Interagency Ecological Program for the San Francisco Estuary

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The Spring issue of the IEP Newsletter has traditionally been devoted to status and trends reporting, and that is the case for this issue as well. Due to staff vacancies, several articles normally part of this issue will be published in later issues, this includes Zooplankton Monitoring, Benthic Monitoring, and the San Francisco Bay Study contributions on estuarine and marine shrimps, crabs, and fishes.

In the first of the Status and Trends articles, **Dave Contreras** (CDFW), **Katherine Osborn** (CDFW), and **Randy Baxter** (CDFW) report on the 2013 abundance of upper estuary pelagic fishes. Many of the fish populations indexed by IEP surveys did poorly in 2013; notably, Delta Smelt indices declined between 2012 and 2013 in all surveys (20 mm, Summer Townet, and Fall Midwater Trawl). In addition, juvenile American Shad and Striped Bass indices declined to 2nd and 3rd lowest on record, respectively. A few increases did occur in 2013 from 2012 index levels for Threadfin Shad and Longfin Smelt, but these increases were modest and similar in scale to low values of recent years.

Within the Status and Trends section, **Geir Aasen** (CDFW) provides an update on fish salvage at the CVP's Tracy Fish Collection Facility (TFCF) and the SWP's Skinner Delta Fish Protective Facility (SDFPF) through the water year of 2013. Several species of management concern were provided individual consideration, including: Chinook Salmon, Steelhead, Striped Bass, Delta Smelt, Longfin Smelt, Splittail, and Threadfin Shad. The SWP exported 2.70 billion m^3 of water, which was a decrease from the export in WY 2012, and well below the record high export in WY 2011. The CVP exported 2.27 billion m^3 of water, which was comparable to exports in WYs 2008-2010 and 2012, but a decrease in exports from 2011 and WYs 2002-2007. Total fish salvage (all fish species combined) at the TFCF was low at 2,828,514. This was an increase from the record low of 475,082 in WY 2012, but well below the record high of 37,659,835 in WY 2006. Total fish salvage at the SDFPF was low at 3,042,176. This was an increase from WY 2012 (1,607,286) and comparable to WY 2011 (3,092,553). Relatively few Chinook Salmon, Steelhead, Delta Smelt, and Longfin Smelt were salvaged at the TFCF (4,032, 646, 300, and 241, respectively) and at the SDFPF (3,184, 861, 1,701, and 659, respectively), compared to total annual salvage.

In the final Status and Trends article, **Jason Azat** (CDFW) reports on Central Valley Chinook Salmon harvest and escapement in 2013. The estimated ocean harvest of Chinook Salmon was up 122% in 2013 from 2012 (337,663) and was the highest since 2004. The estimated harvest of Chinook Salmon in Central Valley waters in 2013 was 80% of the 2012 harvest, the majority of which was Sacramento fall-run (61,672). Harvest of late-fall run (2,164), spring-run (523), and San Joaquin River fall-run (2,196) were all down from the previous year. Escapement of Chinook Salmon to the Sacramento River system in 2013 was up from 2012 for all runs (late-fall, winter, spring, and fall), as was escapement of fall-run to the San Joaquin River system.

Our first contributed paper is a USGS fact sheet (2014–3090) provided by **Paul Buchanan** (USGS), **Maureen Downing-Kunz** (USGS), **David Schoellhamer** (USGS), **Greg Shellenbarger** (USGS), and **Kurt Weidich** (USGS), with the purpose to inform the public and resource managers of the availability of continuous water-quality and suspended-sediment data. The article, "Continuous Water-Quality and Suspended-Sediment Transport Monitoring in the San Francisco Bay, California, Water Years 2011–13," provides descriptions of the data types, locations of samples collected, and how the public may access the available data.

In a second contributed paper, **Trishelle Morris** (CDFW) reports on patterns of gelatinous zooplankton (jellyfish) collected by the CDFW Summer Townet Survey (STN) in June through August during 2007-2013. The STN began formally identifying and enumerating jellyfish catch in 2007. This article summarizes observations of *Maeotias marginata* medusae, which have dominated the catch of jellyfish since 2007. The summary found that *M. marginata* were caught regularly in low-flow, warm, brackish areas of the estuary; however, medusae were also present outside of these areas. Densities of jellyfish peaked in late summer most years.

In two contributed papers, the US Fish and Wildlife Service's (USFW) Delta Juvenile Fish Monitoring Program (DJFMP) utilize modeling to: 1) examine factors associated with incidental catch of Delta Smelt during monitoring at Chipps Island; and 2) estimate the rates of accurate fish identification by their program. **Joseph Kirsch** (USFWS) and **Denise Barnard** (USFWS) evaluate the influence of environmental conditions and trawl methodology on catches of both Delta Smelt and juvenile Chinook Salmon to determine if and how the monitoring of juvenile Chinook Salmon at Chipps Island might be modified to reduce the future incidental take of Delta Smelt. Their study provides evidence that targeting or avoiding certain environmental conditions to minimize the incidental take of Delta Smelt would likely negatively affect the integrity of long-term monitoring of juvenile Chinook Salmon at Chipps Island. **Joseph Kirsch** (USFWS), **Mike Marshall** (USFWS), and **Lori Smith** (USFWS) report on a laboratory experiment to estimate the fish identification accuracy rates among DJFMP observers and across fish species. In addition, they assess the factors that may influence an observer's identification accuracy of fishes found in the San Francisco Estuary. The modeling results suggest that average fish identification accuracy varied considerably among species, sizes, and observers. These papers provide examples of how IEP member agencies work toward improving fish monitoring programs operating within the San Francisco Estuary.

In our final contributed paper, **Marty Gingras** (CDFW) and **Jason DuBois** (CDFW) present an analysis of "bias" in the estimation of annual angler harvest rates for White Sturgeon. Angler harvest rates are calculated from the number of tags returned within 12 months of tagging in relation to the total number of tags applied at the start of the period. Angler willingness to report the capture of tagged sturgeon was investigated by comparing voluntary returns of tags with progressively higher reward values and similar record of tagged fish capture data reported on sturgeon report cards, which are mandatory to return. The investigation found that White Sturgeon harvest rates were biased low, and that this low bias was attributable to a lack of angler willingness to voluntarily contact CDFW to report capture of tagged fish. This information prompts a reinterpretation of the relative impact of fishing mortality on the population and suggests improvements to the population study harvest estimates could be made with additional public outreach and possibly increasing tag reward values.

Status and Trends

2013 Status and Trends Report for Pelagic Fishes of the Upper San Francisco Estuary

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Introduction

The 2013 Status and Trends report includes pelagic fish data from 4 of the Interagency Ecological Program's (IEP) long-term monitoring surveys, conducted in the upper San Francisco Estuary: 1) the 20 mm Survey, 2) the Summer Townet Survey, 3) the Fall Midwater Trawl Survey, and 4) the U.S. Fish and Wildlife Service (USFWS) Beach Seine Survey (See Honey et al. 2004 for additional information). We present the most recent abundance indices, long-term abundance trends, and distributional information phylogenetically for 7 species: American Shad (*Alosa sapidissima*), Threadfin Shad (*Dorosoma petenense*), Delta Smelt (*Hypomesus transpacificus*), Longfin Smelt (*Spirinchus thaleichthys*), Wakasagi (*H. nipponensis*), Splittail (*Pogonichthys macrolepidotus*), and age-0 Striped Bass (*Morone saxatilis*). Threadfin Shad, Delta Smelt, Longfin Smelt, and age-0 Striped Bass spawn and rear in the upper estuary, and have undergone severe declines in recent years (Sommer et al. 2007).

Methods

Sampling Background

The California Department of Fish and Wildlife (CDFW) 20 mm Survey monitors distribution and relative abundance of larval and juvenile Delta Smelt throughout its historical spring range (Figure 1). This includes

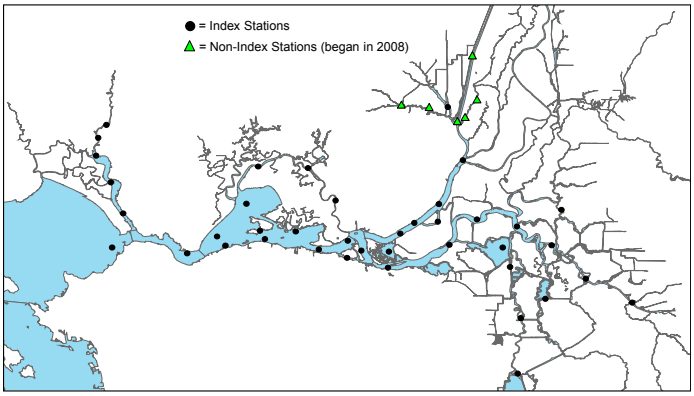

Figure 1 Map of 20 mm Survey stations. Index stations have been sampled since survey inception and their data is used for calculating indices. Non-Index stations were added to the survey at a later date to better assess the distribution of Delta Smelt and other pelagic fishes.

the entire Delta, downstream to eastern San Pablo Bay and the Napa River. Since 1995, the 20 mm Survey has conducted surveys on alternate weeks from early March through early July, completing 9 surveys per year since 2009. Three tows are conducted at each of the 47 stations (Figure 1) using a fixed-mouth, 1,600-µm mesh net (Dege and Brown 2004). The survey added 5 Napa River stations in 1996. In 2008, 2 stations each were added in Lindsey Slough, Miner Slough, and the Sacramento River Deep Water Ship Channel (SRDWSC). The survey name is from the size of Delta Smelt (20 mm) that the survey gear targets, which corresponds with the size at which Delta Smelt are readily identifiable and counted at the State Water Project and Central Valley Project fish salvage facilities. The 20 mm Survey uses fork length (FL) data (< 60 mm FL) from all stations sampled to determine the mean length of Delta Smelt for each survey. Not all stations are used for abundance index calculation, only the original 41 survey stations. The annual index is calculated using data from only 4 out of the 9 surveys conducted: the 2 surveys before and the 2 after the point where the mean length of Delta Smelt (< 60 mm FL) equals 20 mm. From this subset of surveys, the Delta Smelt catch-per-unit-effort (CPUE) is calculated for each of the 41 index stations. Station CPUE is the average CPUE from 3 tows conducted at each station, with tow CPUE calculated as catch divided by volume (m^3) sampled and multiplied by 10,000. One (1) is added to each station CPUE value and then a $\log_{10}$ transformation is performed. The $\log_{10}$ transformed station values are averaged within each survey to obtain one value. These values are back transformed (10^) and 1 is subtracted to return the mean to the original scale. The final 20 mm Survey Delta Smelt annual index is the summation of the 4 survey means.

The Summer Townet Survey (STN) began in 1959. Its data have been used to calculate age-0 Striped Bass indices for all years since except 1966, 1983, 1995 and 2002. Delta Smelt indices have also been calculated for the period of record, except for 1966 through 1968. Historically, STN conducted 2 to 5 surveys annually, depending on how quickly Striped Bass exceeded 38.1 mm in length, the length criterion used to establish the surveys used to calculate the Striped Bass abundance index (see below for more detail on index calculation). The STN field season currently begins in June and runs on alternate weeks through August. This sampling regime has been used since 2003 when CDFW standardized sampling to 6 surveys per year (Hieb et al. 2005). The survey samples 32 historic sites, 31 of which are used in index calculation. Sampling sites are distributed from eastern San Pablo Bay to Rio Vista on the Sacramento River and to Stockton on the San Joaquin River (Figure 2). In 2011, STN added 8 supplemental stations in the Cache Slough-SRDWSC region to increase spatial coverage and better describe Delta Smelt range and habitat (Figure 2). At least 2 tows are completed at historic stations, and a third is conducted if any fish are caught during the first 2 tows. At least 1 tow is completed at the new Cache Slough-SRDWSC stations. To reduce Delta Smelt take, STN only performs a

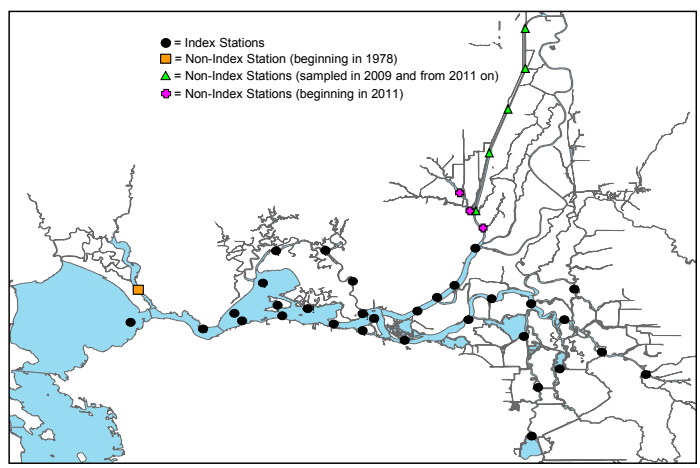

Figure 2 Map of the Summer Townet Survey stations. Index stations have been sampled since survey inception and their data is used for calculating indices. Non-index stations were added at a later date to better assess the distribution of Delta Smelt and other pelagic fishes.

second tow at these stations if the Delta Smelt catch from the first tow is less than ten. Abundance indices for both age-0 Striped Bass and Delta Smelt are calculated from monthly survey indices, but each is calculated differently. To calculate monthly survey indices, catch for a species is summed across tows at each station and multiplied by a weighting factor representing the volume of that station. These products are then summed across all 31 index stations to produce the survey abundance index. To calculate the annual age-0 Striped Bass abundance index, survey abundance is interpolated to the date that age-0 Striped Bass reach a mean FL of 38.1 mm using the abundance indices from the 2 surveys that bound the date (Chadwick 1964, Turner and Chadwick 1972). In contrast, the Delta Smelt annual index is the average of the first two survey abundance indices of each year.

The Fall Midwater Trawl Survey (FMWT) began in 1967. Surveys have been conducted in all years except 1974 and 1979. CDFW established the FMWT survey to examine age-0 Striped Bass relative abundance and distribution in the upper estuary (Stevens 1977). Later, FMWT developed abundance and distribution information for other upper-estuary pelagic fishes, including American Shad, Threadfin Shad, Delta Smelt, Longfin Smelt, and Splittail. The FMWT survey currently samples 122 stations monthly, from September to December. Trawl sampling ranges from San Pablo Bay to Hood on the Sacramento River, and from Sherman Lake to Stockton on the San Joaquin River (Figure 3). The annual abundance index calculation uses catch data from 100 of the 122 stations (see Stevens 1977). The remaining 22 stations were added in 1990, 1991, 2009, and 2010 to improve our understanding of Delta Smelt habitat use (Figure 3). To calculate survey abundance indices, we group the 100 index stations into 17 regions. Monthly indices are calculated by averaging index station catch in each region, multiplying these regional means by their respective weighting factors (i.e., a scalar based on water volume), and summing these products. Annual abundance indices are the sum of the 4 (September-December) monthly indices.

Since 1994, USFWS has conducted beach seine sampling weekly at approximately 40 stations in the Delta and the lower Sacramento and San Joaquin rivers (Brandes and McLain 2001) (Figure 4). Data from 33 stations are used to calculate the annual age-0 Splittail abundance index. These stations range from Sherman Lake to Ord Bend on the Sacramento River (not pictured), and to just downstream of the Tuolumne River confluence on the San Joaquin River (pictured). Hereafter, we refer to the confluence of the of the Sacramento and San Joaquin rivers at Sherman Lake as the Confluence, and the Tuolumne River confluence with the San Joaquin River as the Tuolumne confluence. All Splittail < 25 mm FL (measured individuals and proportions resulting from plus counts) are removed from calculations. The 33 "index" stations are grouped into 10 regions, and the annual index is calculated as the grand average of regional mean catch per m^3 for May and June sampling.

FMWT data were used to describe abundance trends and distribution patterns of all 7 species listed in the introduction. STN data described trends for 3 species, Delta Smelt, Wakasagi, and Striped Bass. Two studies only pro-

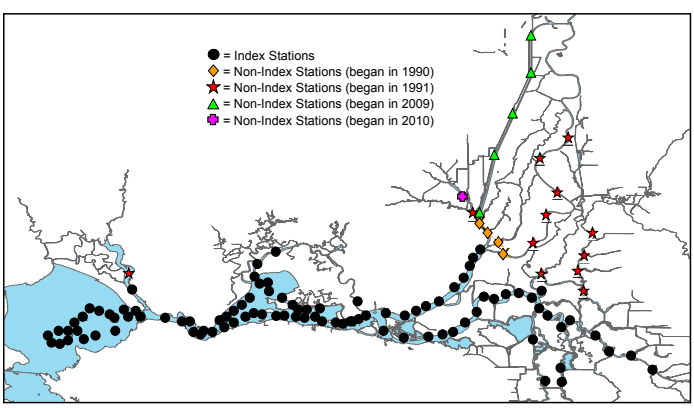

Figure 3 Map of the Fall Midwater Trawl Survey stations. Core stations have been sampled since survey inception and their data is used for indices. Non-index stations were added at a later date to better assess the distribution of Delta Smelt and other pelagic fishes.

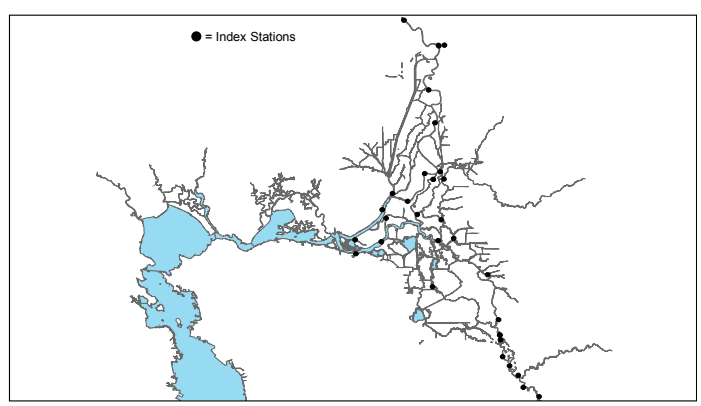

Figure 4 Map of USFWS beach seine survey stations. Data from all stations is used for indices. Locations for 8 stations on the Sacramento River are not pictured.

vided single species information: the 20 mm Survey for the combined index and distribution of larval and small juvenile Delta Smelt, and the USFWS beach seine data for age-0 Splittail index and distribution.

Results

American Shad

The American Shad was introduced into the Sacramento River in 1871 (Dill and Cordone 1997), and is now found throughout the estuary. This anadromous species spawns in rivers in late spring, rears in rivers and freshwater within the Delta from late May through summer, and migrates to the ocean in late summer and fall. Adults spend approximately 3 to 5 years maturing in the ocean before returning to freshwater to spawn. Most males reach maturity by 3 or 4 years of age, while females typically reach maturity at 4 to 5 years. Spawning occurs in the Sacramento, Feather, and American rivers from April through June; afterwards a large percentage of adults die (Stevens 1966). American Shad are planktivorous at all life stages.

The 2013 FMWT American Shad (all ages) index was 74% the 2012 index and the second lowest index on record (Figure 5). Indices have been below the study-period mean since 1998, except in 2003 and 2006, which had a record high index. American Shad were collected in all FMWT regions in 2013 (n = 414), but were most common (54%) during September through November from Cache Slough-SRDWSC downstream to the lower Sacramento River. American Shad distribution shifted during emigrated from the Estuary with 72% of December catch (n = 57) having occurred downstream of the Confluence.

Threadfin Shad

In the late 1950's, the Threadfin Shad was introduced to reservoirs that fed into the Sacramento and San Joaquin rivers. The species quickly moved downstream and established itself in the Delta. Although Threadfin Shad reside throughout the estuary, they more commonly inhabit fresh or oligohaline waters of dead-end sloughs and other slow-moving waterways (Wang 1986). They are planktivorous their entire life, and feed on zooplankton and algae (Holanov and Tash 1978). Threadfin Shad can reach maturity at the end of their first year, and may live up to 4 years. Spawning occurs from late spring throughout the summer, peaking from May to July (Wang 1986).

The 2013 FMWT Threadfin Shad (all ages) index was 6.8 times the 2012 index, but remained low (Figure 6). Since 2002, Threadfin Shad abundance has been below the study period mean, but showed a slight increasing trend through 2007 before dropping off precipitously. Abundance for 2013 remains among the lowest observed.

From September through November, the majority of Threadfin Shad were located in salinity < 1 ppt. However, a slight shift in distribution occurred in December, when approximately 19% of total catch from index stations occurred in salinity > 5 ppt (up to 28 ppt). As seen in previous years, Threadfin Shad were most common (n = 1437) in either Cache Slough or SRDWSC, versus the other regions (n = 213), during all survey months.

Delta Smelt

The Delta Smelt is a small (55-90 mm FL) osmerid endemic to the upper San Francisco Estuary. The Delta Smelt population declined dramatically in the 1980s, and in 1993 it was listed as a state and federal threatened species. The Delta Smelt is considered environmentally sen-

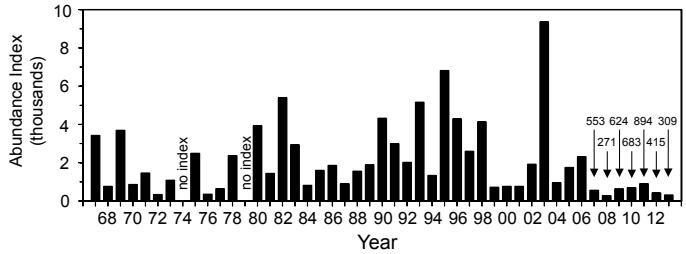

Figure 5 Annual abundance indices of American shad (all sizes) from the Fall Midwater Trawl Survey, 1967-2013.

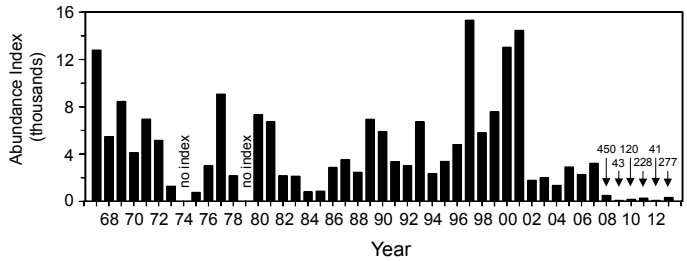

Figure 6 Annual abundance indices of Threadfin Shad (all sizes) from the Fall Midwater Trawl Survey, 1967-2013.

sitive because it typically lives for 1 year, resides within a limited range at the interface between salt and fresh water, and has low fecundity (females produce 1,200 to 2,600 eggs/batch; Moyle et al. 1992). Females may partially offset their low fecundity by producing multiple batches of eggs during the spawning season (Bennett 2005, California Department of Fish and Wildlife unpublished data).

The 2013 20 mm Survey Delta Smelt index was 70% the 2012 index (Figure 7A). Like the 2011 and 2012 indices, the 2013 index was more consistent with indices from the early 2000s than with the low indices from 2007-2010. In March, low Delta Smelt larval catches occurred from the Confluence upstream on the Sacramento River into Cache and Miner sloughs, but not the SRDWSC, and from the Confluence into the lower San Joaquin River and south Delta. In April, total larval catch increased sharply particularly in the Confluence, lower Sacramento River, Cache Slough, and the lower San Joaquin River; the first modest catches also occurred in Suisun Bay and the SRDWSC. By May, catch numbers declined at the Confluence, in Miner Slough, and in the south Delta, while catches peaked in SRDWSC and Suisun Bay. By June, catches declined in all areas except the lower Sacramento River, and by July Delta Smelt were only caught from Suisun Bay upstream into the lower Sacramento River.

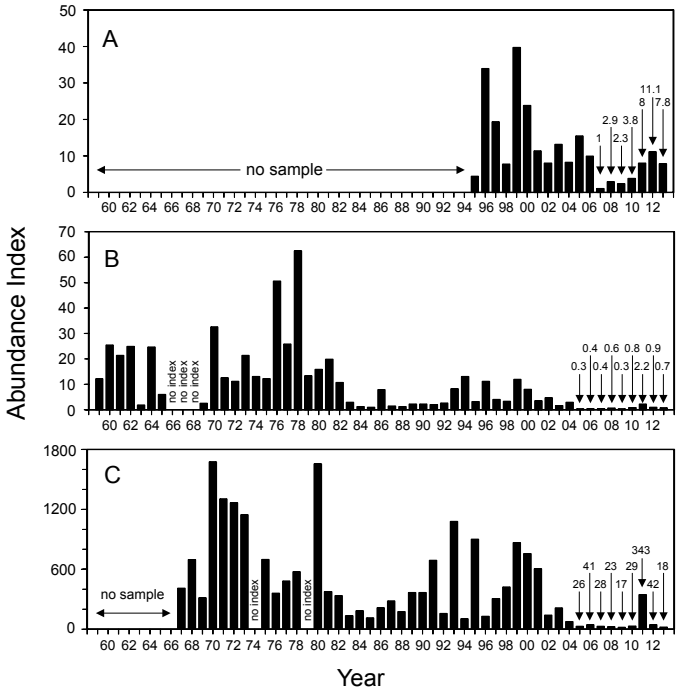

Figure 7 Annual abundance indices of Delta Smelt from: A) 20 mm Survey (larvae and juveniles; 1995-2013); B) Summer Townet Survey (juveniles; 1959-2013); C) Fall Midwater Trawl Survey (subadults; 1967-2013).

The 2013 STN age-0 Delta Smelt index was 78% the 2012 index (Figure 7B). The 2013 index was consistent with low STN indices since 2005, ranking as the sixth lowest in the study's history. The Delta Smelt catch declined throughout the sampling season, from 90 fish in June, to 64 fish in July, to 17 fish in August. In June, Delta Smelt were found in all regions except San Pablo Bay, the Napa River, and the south Delta: the largest catches came from Suisun Bay (n = 35) and the SRDWSC (n = 28). In July, catch decreased in all regions except Suisun Bay, where it increased (n = 54). By August, Delta Smelt catch was concentrated in Suisun Bay (n = 8) and the SRDWSC (n = 6). At index stations, Delta Smelt had the strongest presence in Suisun Bay. A single, large catch of 33 fish accounted for over half the Delta Smelt found in Suisun Bay in July.

The 2013 FMWT Delta Smelt index was 43% the 2012 index and the second lowest on record (Figure 7C). Less than 5 Delta Smelt were caught each month, except in December, when abundance and distribution increased slightly. In September, Delta Smelt (n = 4) were caught in Suisun Bay, the Confluence, and the lower Sacramento River. In October, Delta Smelt (n = 3) distribution was limited to the Confluence and the lower Sacramento River. In November, fish (n = 4) distribution expanded to include the SRDWSC. In December, Delta Smelt (n = 9) were found in Suisun Bay, the Confluence, and the lower Sacramento and San Joaquin rivers.

Longfin Smelt

The Longfin Smelt is a short-lived, anadromous fish that spawns in freshwater in winter and spring, and rears primarily in brackish water. Some age-0 and age-1 fish migrate to the ocean in summer and fall, often returning to the estuary in late fall or winter the same year as estuary water cools. A few Longfin Smelt mature at the end of their first year, but most mature at the end of their second year, and some live to spawn or spawn again at age-3 (Wang 1986).

The 2013 FMWT Longfin Smelt (all ages) index was 2.7 times the 2012 index (Figure 8). A few Longfin Smelt (n = 5) were caught in September in San Pablo and Suisun bays. In October, fish (n = 14) were found from Carquinez Strait to the lower Sacramento River. By November,

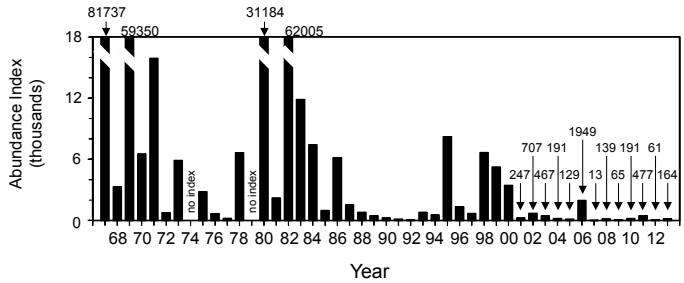

Figure 8 Annual abundance indices of Longfin Smelt (all sizes) from the Fall Midwater Trawl Survey, 1967-2013.

Longfin Smelt (n = 16) distribution had expanded back into San Pablo Bay and the SRDWSC. Catch increased in December (n = 53), but distribution contracted and no Longfin Smelt were caught in the SRDWSC. Similar to last year, 60% of the season's catch occurred in December, from the lower Sacramento River to San Pablo Bay.

Wakasagi

In 1959, the Wakasagi was intentionally introduced from Japan as a baitfish for California lakes and reservoirs (Wales 1962, Dill and Cordone 1997). The Wakasagi was first detected in the San Francisco Estuary in 1990, but may have escaped from California lakes and traveled to the estuary as early as 1974 (Moyle et al. 1992). Although generally found in freshwater, the Wakasagi has a higher salinity tolerance than the Delta Smelt (Swanson et al. 2000). Wakasagi and Delta Smelt are usually planktivorous and reach maturity within a year (Moyle et al. 1992). We report on Wakasagi to track their abundance and to see how Delta Smelt and Wakasagi distribution overlap.

Since 1995, STN has caught 42 Wakasagi, with more than half of them collected (n = 25) at the recently added Cache Slough-SRDWSC stations. Before 2009, STN rarely captured Wakasagi (n = 7). In 2013, 10 Wakasagi were collected, 8 in the SRDWSC and 2 in the lower Sacramento River (Table 1).

FMWT has caught a few more Wakasagi (n = 45; Table 2) than STN (c.f., Tables 1, 2). Like STN, Wakasagi were sporadically collected prior to 2009 (n = 10). These catches occurred in Suisun Bay, the Confluence, the lower Sacramento River, and Cache Slough. Since adding stations in the SRDWSC, this region has produced 61% of Wakasagi catch and both of the Wakasagi captured in 2013 (Table 2).

Table 1 Summer Townet Survey Wakasagi total annual catch by region, 1959 to 2013 (regions with positive catch of Wakasagi were reported).

Year	*Suisun Bay*	*Confluence*	*Lower Sac River*	*Cache Slough*	*SRDWSC*	*Lower San Joaquin River*	*South Delta*
1995	0	0	1	no sample	no sample	0	0
1996	0	0	1	no sample	no sample	0	1
1997	0	0	0	no sample	no sample	0	0
1998	2	0	0	no sample	no sample	0	0
1999	0	0	0	no sample	no sample	0	0
2000	0	0	1	no sample	no sample	0	1
2001	0	0	0	no sample	no sample	0	0
2002	0	0	0	no sample	no sample	0	0
2003	0	0	0	no sample	no sample	0	0
2004	0	0	0	no sample	no sample	0	0
2005	0	0	0	no sample	no sample	0	0
2006	0	0	0	no sample	no sample	0	0
2007	0	0	0	no sample	no sample	0	0
2008	0	0	0	no sample	no sample	0	0
2009	4	0	1	no sample	no sample	0	0
2010	0	0	0	no sample	no sample	0	0
2011	2	0	0	4	9	0	0
2012	0	0	0	0	4	1	0
2013	0	0	2	0	8	0	0

Splittail

The Splittail is endemic to the San Francisco Estuary and its watershed. Adults migrate upstream as river flows increase from late fall through early spring. They move from tidal brackish or freshwater habitats to inundated floodplains and river margins to forage and spawn (Sommer et al. 1997, Moyle et al. 2004). Such migrations occur in the Sacramento, San Joaquin, Cosumnes, Napa, and Petaluma rivers, as well as Butte Creek and other small tributaries. Most spawning takes place from March through May. Young disperse as either larvae or juveniles. Larvae migrate downstream when river levels drop, while juveniles move downstream in late spring and summer, when reduced flows diminish backwater and edge-water habitats. The timing and duration of floodplain inundation impacts year-class strength; springtime inundation periods of 30 days or more resulted in moderate to large Splittail year classes (Sommer et al. 1997, Moyle et al. 2004).

The 2013 Splittail age-0 beach seine index (U.S. Fish and Wildlife Service data) was 16% of the 2012 index, and was the second lowest on record (Figure 9A). This survey samples along the shoreline throughout the Delta and into the Sacramento and San Joaquin rivers (see methods), so it can detect recruitment in the rivers upstream of current trawl sampling areas. Splittail were caught with greatest consistency and in greatest numbers in the Delta, as compared to the Sacramento and San Joaquin rivers. The variability of the age-0 Splittail abundances likely reflects the variability in outflows and floodplain-terrace inundation in recent years.

The 2013 FMWT Splittail (all ages) index was the same as the 2012 index (Figure 9B), and represented 1 fish, as it did in 2012. Splittail young prefer shallow water, so FMWT may sample them ineffectively, since trawling occurs in open, moderately deep (> 2 m) water. FMWT sampling does not sensitively portray Splittail abundance and distribution when levels are low, but it does appear to effectively detect strong year classes.

Age-0 Striped Bass

The Striped Bass is a long-lived, anadromous fish first introduced to the San Francisco Estuary over 125 years ago (Dill and Cordone 1997). Adult Striped Bass forage in coastal bays and the near-shore ocean, and migrate up rivers to spawn in spring. Juveniles rear in the fresh and brackish waters of the estuary.

Age-0 Striped Bass abundance began declining by mid-1970s (Figure 10). STN and FMWT indices declined in the late 1990s and again early 2000s; indices have remained low since (Figure 10). Stevens et al. (1985) hypothesized that low Striped Bass recruitment was related to 4 factors: 1) the declining adult population; 2) reduced plankton food supply; 3) loss of large numbers of young Striped Bass to water diversions; and 4) population-level contaminant effects. Additionally, Sommer et al. (2011) suggest that age-0 Striped Bass have shifted from a broad distribution including channel and shoal habitats to almost exclusively shoal areas, which are under-sampled by CDFW trawl surveys. This distributional shift may have contributed to low indices in recent years.

The 2013 STN age-0 Striped Bass 38.1 mm index was 0.6, or 35% of the 2012

Table 2 Fall Midwater Trawl Survey Wakasagi total annual catch by region, 1967 to 2013 (regions with positive catch of Wakasagi were reported).

Year	Suisun Bay	Confluence	Lower Sac River	Cache Slough	SDWSC	South Delta
1995	0	0	3	0	no sample	0
1996	1	0	0	0	no sample	0
1997	1	0	0	0	no sample	0
1998	0	0	0	0	no sample	0
1999	0	0	0	0	no sample	0
2000	0	0	0	3	no sample	0
2001	0	0	1	0	no sample	0
2002	0	0	0	0	no sample	0
2003	0	0	0	0	no sample	0
2004	0	0	0	0	no sample	0
2005	0	0	0	0	no sample	0
2006	0	0	0	0	no sample	0
2007	0	0	0	0	no sample	0
2008	0	0	0	0	no sample	0
2009	1	0	1	0	6	0
2010	0	0	0	1	8	0
2011	3	1	0	3	9	0
2012	0	0	0	0	1	0
2013	0	0	0	0	2	0

index (Figure 10A). This index was the third lowest on record and consistent with low indices since 2003. Striped Bass juvenile catch peaked in early June during the first survey (n = 99) and dropped through the remaining field season. In June, STN captured Striped Bass in all regions except San Pablo Bay and the Napa River. The June catch (n = 158) was concentrated mostly in the lower Sacramento River and the south Delta, and to a lesser extent in the lower San Joaquin River and Suisun Bay. In July, STN caught 47 Striped Bass with most occurring in Suisun Bay (n = 20) and the lower Sacramento River (n = 15). By August, total catch was 10 fish, mainly from Suisun Bay (n = 8).

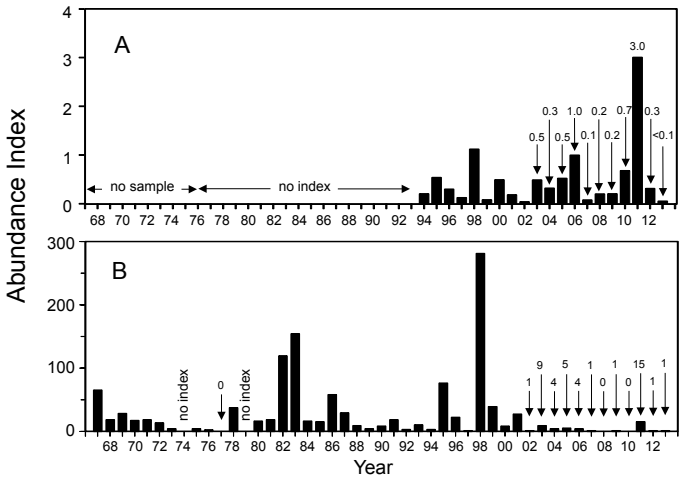

Figure 9 Annual abundance indices of Splittail from: A) USFWS beach seine (juveniles ≥ 25mm), 1994-2013; B) Fall Midwater Trawl Survey (all sizes), 1967-2013.

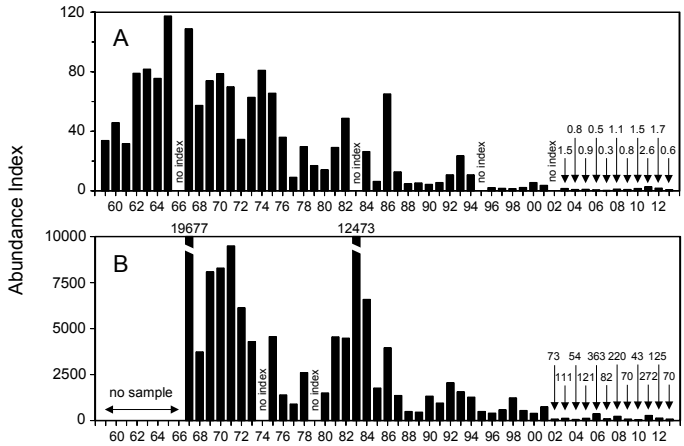

Figure 10 Annual abundance indices of age-0 Striped Bass from: A) Summer Townet, 1959-2013; B) Fall Midwater Trawl Survey, 1967-2013.

The 2013 FMWT age-0 Striped Bass index decreased to 56% of the 2012 index. It was tied for the third lowest index on record and was consistent with the low indices seen since 2002 (Figure 10B). During September, FMWT captured Striped Bass (n = 14) from Carquinez Strait through Suisun Bay, and into the lower Sacramento and San Joaquin rivers. In October, distribution (n = 5) shifted upstream of Carquinez Strait and extended into the SRDWSC. In November, Striped Bass (n = 6) were found from San Pablo to Suisun Bay and in the lower Sacramento River. Striped Bass catches peaked modestly in December (n = 31) and were limited to Suisun Bay, the Confluence, and the lower Sacramento River.

Conclusion

All of the 2013 pelagic fish abundance indices were consistent with the low index values of recent years. The year 2013 was also the second in a row of low freshwater Delta outflow. The Cache Slough-SRDWSC region continued to provide high catches during STN and FMWT sampling, while the southeast and northeast Delta continued to provide low catches.

Extending our sampling into the Cache Slough-SRDWSC region has provided valuable habitat and catch data for several pelagic species. For example, the region was home to 87% of Threadfin Shad across CDFW surveys. These non-index stations also produce relatively high catches of American Shad (up to 80+ per tow). The region provides valuable habitat for Delta Smelt: STN caught 34% of its total and FMWT caught 10% of its total Delta Smelt catch in Cache Slough-SRDWSC. These values are lower than previous years. Lastly, Wakasagi continue to show up in predominately in the SRDWSC as compared with other Delta regions.

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Fish Salvage at the State Water Project's and Central Valley Project's Fish Facilities during the 2013 Water Year

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Introduction

Two facilities mitigate fish losses associated with water export by the federal Central Valley Project (CVP) and California's State Water Project (SWP). The CVP's Tracy Fish Collection Facility (TFCF) and the SWP's Skinner Delta Fish Protective Facility (SDFPF) divert (salvage) fish from water exported from the southern end of the Sacramento-San Joaquin Delta. Both facilities use louver-bypass systems to divert fish from the exported water. The diverted fish are periodically loaded into tanker trucks and transported to fixed release sites in the western Delta. Operations began in 1957 at the TFCF and in 1968 at the SDFPF.

Methods

This report summarized the 2013 water year (10/1/2012-9/30/2013) salvage information from the TFCF and the SDFPF, and examined data from water years (WY) 1981 to 2013 for possible relevance to salvage trends in recent years. The following species were given individual consideration: Chinook Salmon (*Oncorhynchus tshawytscha*), Steelhead (*O. mykiss*), Striped Bass[1] (*Morone saxatilis*), Delta Smelt[1] (*Hypomesus transpacificus*), Longfin Smelt[1] (*Spirinchus thaleichthys*), Splittail (*Pogonichthys macrolepidotus*), and Threadfin Shad[1] (*Dorosoma petenense*).

Systematic sampling was used to estimate the numbers and species of fish salvaged at both facilities. Bypass flows into the fish-collection buildings were subsampled generally once every 1 or 2 hours for 1 to 60 minutes (= 27.1, sd = 7.4) at the SDFPF and once every 2 hours for 10 to 120 minutes (= 27.9, sd = 6.8) at the TFCF. Fish 20 mm fork length (FL) or larger were identified, numerated, and measured. These fish counts were expanded to estimate the total number of fish salvaged in each 1- to 2-hour period of water export. For example, a subsample duration of 30 minutes over a 120-minute export period equals an expansion factor of 4, which was multiplied by the number of fish per species collected from the fish count. These incremental salvage estimates were then summed across time to develop monthly and annual species-salvage totals for each facility.

Chinook Salmon loss estimates were presented because the loss model has been widely accepted by regulatory agencies and has undergone extensive review. Loss is the estimated number of Chinook Salmon entrained by the facility minus the number of Chinook Salmon that survive salvage operations (California Dept. of Fish and Game 2006). Salmon salvage and loss were summarized by origin (i.e., hatchery fish defined as adipose fin clipped or wild fish defined as non-adipose fin clipped) and race (fall, late-fall, winter, or spring). Race of Chinook Salmon was determined solely by the Delta criteria based on length at date of salvage (California Dept. of Fish and Wildlife 2014).

Larval fish were also collected and examined to determine the presence of Delta Smelt and Longfin Smelt < 20 mm FL. Larval sampling ran from March 6 through June 18 at the SDFPF and from March 11 through June 20 at the TFCF. Larval samples were collected once for every 6 hours of water export. Duration of larval samples was the same as the duration for counts. To retain these smaller fish, the fish screen used in the routine counts was lined with a 0.5 mm Nitex net. Larval fish from the TFCF were identified to species by TFCF personnel and larval fish from the SDFPF were identified to the lowest taxa possible by California Dept. of Fish and Wildlife personnel.

[1] Pelagic Organism Decline (POD) species

Water Exports

The SWP exported 2.70 billion m^3 of water, which was a decrease from the export in WY 2012 (3.25 billion m^3), and well below the record high export in WY 2011 (4.90 billion m^3) (Figure 1). The CVP exported 2.27 billion m^3 of water, which was comparable to exports in WYs 2008-2010 and 2012, but a decrease in exports from 2011 (3.13 billion m^3) and WYs 2002-2007, which ranged from 3.08 billion m^3 to 3.35 billion m^3.

Exports from the two water projects generally followed a similar seasonal pattern. Exports at the CVP peaked in October-November 2012 and again in July-August 2013 (Figure 2). During these periods, the CVP exported 1.15 billion m^3, which represented 50.6% of annual export. Exports at the SWP peaked in December 2012 and again in July-August 2013 (Figure 2). During this period, the SWP exported 1.15 billion m^3, which represented 42.5% of annual export. CVP monthly exports ranged from 33.36 to 297.35 million m^3. SWP monthly exports ranged from 67.56 to 438.79 million m^3.

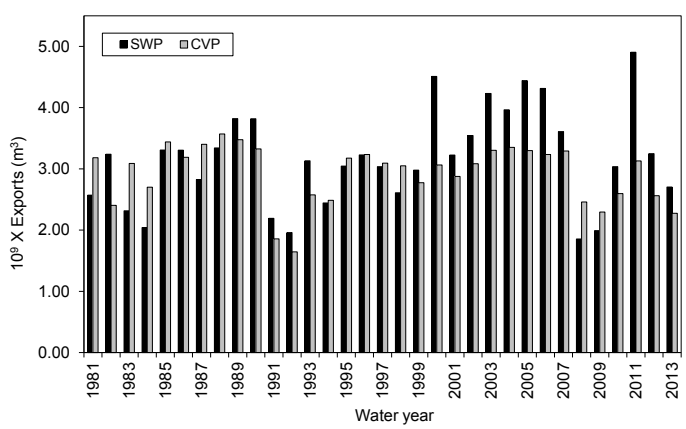

Figure 1 Annual water exports in billions of cubic meters for the SWP and the CVP, water years 1981 to 2013.

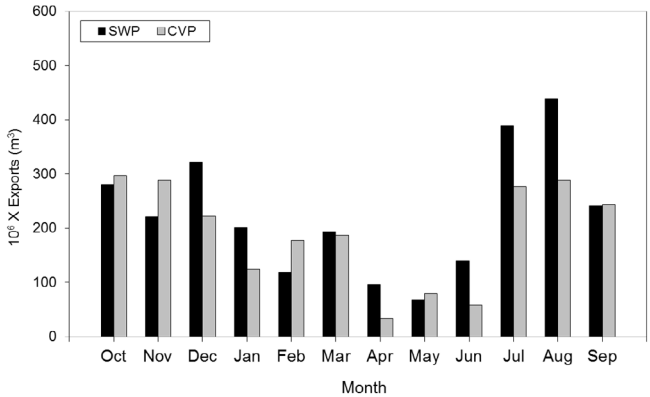

Figure 2 Monthly water exports in millions of cubic meters for the SWP and the CVP, water year 2013.

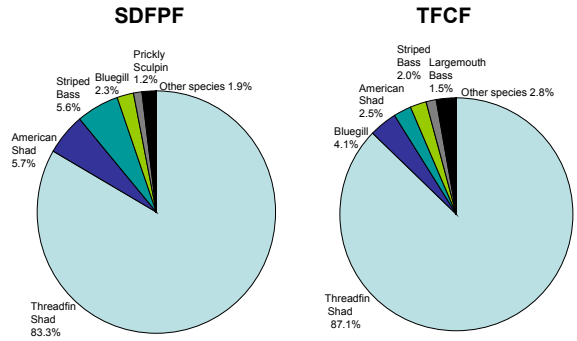

Figure 4 Percentages of annual salvage for the 5 most prevalent fish species and other fish species combined at the SDFPF and TFCF, WY 2013.

Total Salvage and Prevalent Species

Total fish salvage (all fish species combined) at the TFCF was low at 2,828,514 (Figure 3). This was an increase from the record low in WY 2012 (475,082) but well below the record high of 37,659,835 in WY 2006 (Figure 3). Total fish salvage at the SDFPF was low at 3,042,176. This was an increase from WY 2012 (1,607,286) and comparable to WY 2011 (3,092,553).

Threadfin Shad was the most-salvaged species at SDFPF and TFCF (Figure 4 and Table 1). American Shad (*Alosa sapidissima*) and Striped Bass were the 2nd and 3rd most-salvaged fishes at SDFPF. Bluegill (*Lepomis macrochirus*) and American Shad were the 2nd and 3rd most-salvaged fishes at TFCF. Native species comprised 1.4% of annual fish salvage at SDFPF and 0.5% of annual fish salvage at TFCF. Relatively few Chinook Salmon, Steelhead, Delta Smelt, and Longfin Smelt were salvaged at the SDFPF (< 0.3% combined of total annual salvage) and at the TFCF (< 0.2% combined of total annual salvage).

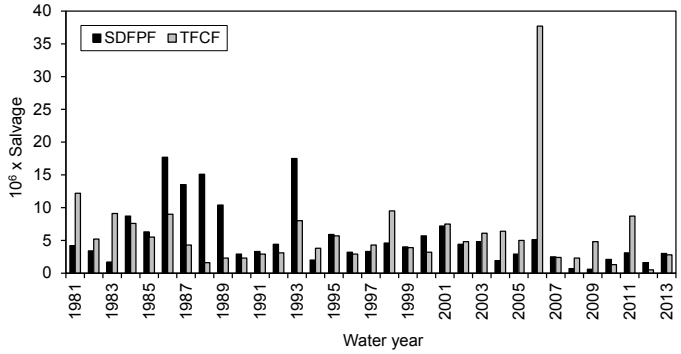

Figure 3 Annual salvages of all fish taxa combined at the SDFPF and the TFCF, water years 1981 to 2013.

Chinook Salmon

Salvages of Chinook Salmon (all races and origins combined) at both facilities continued the low salvage trend since WY 2001 (Figure 5). SDFPF salvage (3,184) increased from the record-low salvage in WY 2012 (1,579) but substantially decreased from WY 2011 levels (18,830). Mean WY 2001-2013 SDFPF salvage was about 10% of the mean salvages in the 1980s and the 1990s. Salvage of Chinook Salmon at the TFCF (4,032) increased from WY 2012 (1,965) but substantially decreased from WY 2011 (18,135). Mean WY 2001-2013 TFCF salvage was about 13% of the mean salvages in the 1980s and the 1990s.

Salvaged Chinook Salmon at the TFCF were primarily wild fall-run fish, which comprised 84.6% of wild fish (Table 2). Salvaged Chinook Salmon at the SDFPF were also primarily wild fall-run fish, which comprised 66.8% of wild fish. The majority of wild fall-run fish at the SDFPF and the TFCF were salvaged in May (Figure 6).

Loss of Chinook Salmon (all origins and races) was higher at the SDFPF (14,171) than at the TFCF (3,069) (Table 2). Greater entrainment loss at the SDFPF than at the TFCF was attributable to greater pre-screen loss (California Dept. of Fish and Game 2006).

Steelhead

Salvage of Steelhead (wild and hatchery origins combined) continued the pattern of low salvage observed since WY 2005 (Figure 7). Salvage at the SDFPF (861) was higher than in WY 2012 (443). Salvage at the TFCF (646) was also higher than in WY 2012 (493).

Table 1 Annual fish salvages and percentages of annual fish salvage (%) collected from the SDFPF and TFCF in water year 2013.

SDFPF			TFCF		
Species	***Salvage***	***%***	***Species***	***Salvage***	***%***
Threadfin Shad	2,535,117	83.3	Threadfin Shad	2,463,695	87.1
American Shad	173,480	5.7	Bluegill	115,796	4.1
Striped Bass	170,306	5.6	American Shad	71,619	2.5
Bluegill	70,616	2.3	Striped Bass	57,855	2.0
Prickly Sculpin	36,170	1.2	Largemouth Bass	43,291	1.5
Inland Silverside	15,445	0.5	Inland Silverside	28,747	1.0
Largemouth Bass	11,825	0.4	White Catfish	12,341	0.4
White Catfish	8,601	0.3	Rainwater Killifish	7,444	0.3
Black Crappie	3,881	0.1	Prickly Sculpin	6,621	0.2
Shimofuri Goby	3,539	0.1	Chinook Salmon	4,032	0.1
Chinook Salmon	3,184	0.1	Golden Shiner	3,920	0.1
Delta Smelt	1,701	0.1	Channel Catfish	3,150	0.1
Yellowfin Goby	1,690	0.1	Black Crappie	3,093	0.1
Channel Catfish	1,660	0.1	Yellowfin Goby	1,343	<0.1
Western Mosquitofish	888	<0.1	Lamprey Unknown	1,288	<0.1
Rainbow / Steelhead Trout	861	<0.1	Redear Sunfish	1,078	<0.1
Rainwater Killifish	848	<0.1	Rainbow / Steelhead Trout	646	<0.1
Longfin Smelt	659	<0.1	Shimofuri Goby	601	<0.1
Bigscale Logperch	643	<0.1	Delta Smelt	300	<0.1
Splittail	329	<0.1	Longfin Smelt	241	<0.1
Lamprey Unknown	161	<0.1	Tule Perch	232	<0.1
Golden Shiner	111	<0.1	Western Mosquitofish	222	<0.1
Redear Sunfish	92	<0.1	Black Bullhead	166	<0.1
Wakasagi	88	<0.1	Warmouth	159	<0.1
Tule Perch	71	<0.1	Bigscale Logperch	141	<0.1
Common Carp	66	<0.1	Splittail	125	<0.1
Starry Flounder	57	<0.1	Threespine Stickleback	97	<0.1
Threespine Stickleback	22	<0.1	Brown Bullhead	47	<0.1
White Sturgeon	12	<0.1	Green Sunfish	46	<0.1
Red Shiner	12	<0.1	Wakasagi	42	<0.1
Shokihaze Goby	10	<0.1	Common Carp	26	<0.1
Brown Bullhead	9	<0.1	Pacific Lamprey	24	<0.1
Riffle Sculpin	6	<0.1	White Crappie	23	<0.1
Goldfish	4	<0.1	Sacramento Blackfish	16	<0.1
Hitch	4	<0.1	Pacific Brook Lamprey	16	<0.1
Sacramento Sucker	4	<0.1	Shokihaze Goby	11	<0.1
Blue Catfish	4	<0.1	Pacific Staghorn Sculpin	8	<0.1
			White Sturgeon	4	<0.1
			Starry Flounder	4	<0.1
			Fathead Minnow	4	<0.1

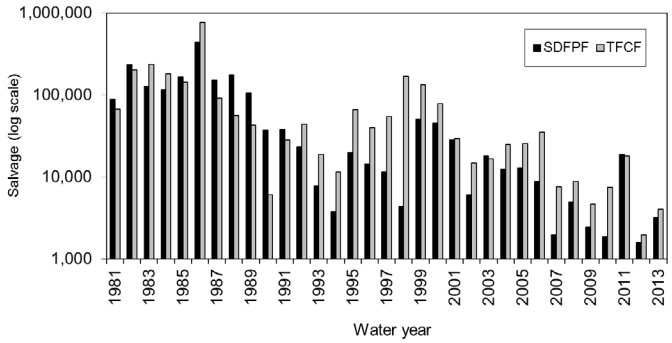

Figure 5 Annual salvages of Chinook Salmon (all races and wild and hatchery origins combined) at the SDFPF and the TFCF, water years 1981 to 2013. The logarithmic scale is log_{10}.

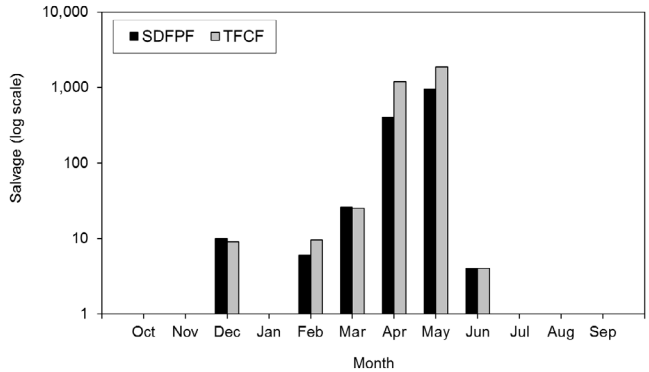

Figure 6 Monthly salvages of wild, fall-run Chinook Salmon at the SDFPF and the TFCF, WY 2013. The logarithmic scale is log_{10}.

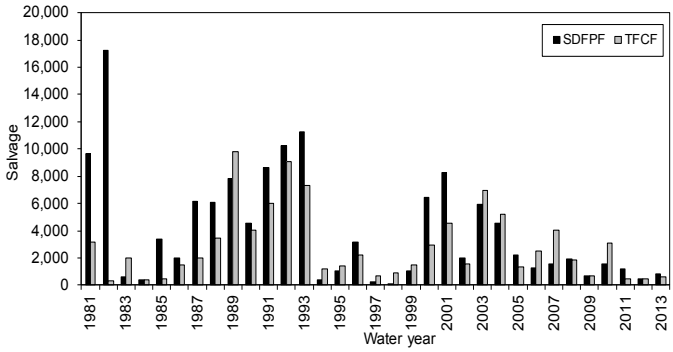

Figure 7 Annual salvages of Steelhead (wild and hatchery origins combined) at the SDFPF and the TFCF, water years 1981 to 2013.

Table 2 Chinook Salmon annual salvages, percentages of annual salvage, race and origin (wild or hatchery), and losses at the SDFPF and the TFCF, water year 2013.

Facility	*Origin*	*Race*	*Salvage*	*Percentage*	*Loss*
SDFPF					
	Wild				
		Fall	1,419	66.8	6,285
		Late-fall	57	2.7	259
		Spring	505	23.8	2,199
		Winter	142	6.7	633
	Total Wild		2,123		9,376
	Hatchery				
		Fall	322	30.3	1,460
		Late-fall	616	58.1	2,780
		Spring	3	0.3	13
		Winter	120	11.3	542
	Total Hatchery		1,061		4,795
	Grand Total		3,184		14,171
TFCF					
	Wild	Fall	3,134	84.6	2,416
		Late-fall	28	0.8	18
		Spring	404	10.9	297
		Winter	129	3.5	98
		Unknown Race	8	0.2	5
	Total Wild		3,703		2,834
	Hatchery				
		Fall	93	28.3	62
		Late-fall	165	50.1	118
		Spring	4	1.2	2
		Winter	67	20.4	53
	Total Hatchery		329		235
	Grand Total		4,032		3,069

The TFCF salvaged 320 hatchery Steelhead and 326 wild Steelhead. The SDFPF salvaged 389 hatchery Steelhead and 472 wild Steelhead.

Salvage of wild Steelhead at both facilities peaked around the middle of the water year (Figure 8). Wild Steelhead were salvaged most frequently in March at the TFCF and in April at the SDFPF.

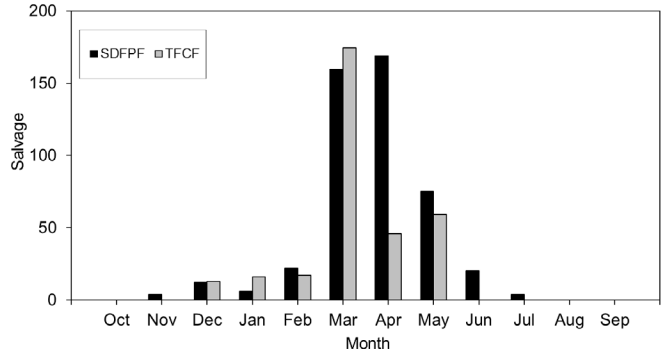

Figure 8 Monthly salvages of wild Steelhead at the SDFPF and the TFCF, WY 2013.

Striped Bass

Salvage at the TFCF (57,855) was a near-record low. Salvage at the TFCF and the SDFPF (170,306) continued the generally-low trend observed since the mid-1990s (Figure 9). Prior to WY 1995, annual Striped Bass salvages were generally above 1,000,000 fish.

Most Striped Bass salvage at the SDFPF occurred in December and from June-July. Most Striped Bass salvage at the TFCF occurred from May-July (Figure 10). At the SDFPF, December salvage (34,934), June salvage (39,733), and July salvage (45,532) accounted for 70.6% of annual salvage. At the TFCF, May salvage (14,445), June salvage (16,812), and July salvage (19,878) accounted for 88.4% of annual salvage. Striped Bass was salvaged every month at both facilities, with the lowest monthly salvage occurring in September at the SDFPF (307) and in April at the TFCF (56).

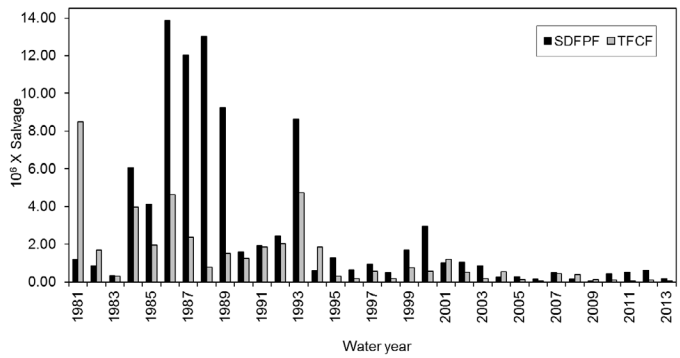

Figure 9 Annual salvages of Striped Bass at the SDFPF and the TFCF, water years 1981 to 2013.

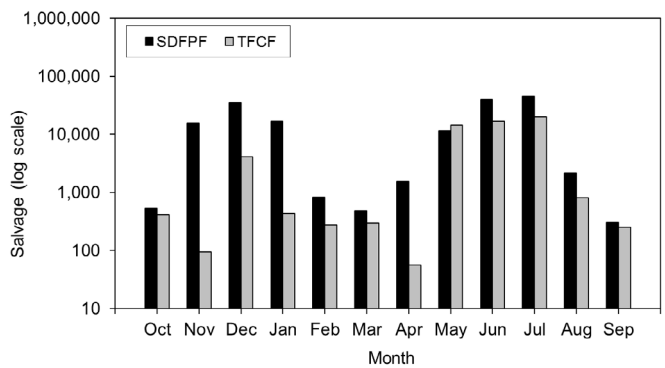

Figure 10 Monthly salvages of Striped Bass at the SDFPF and the TFCF, WY 2013. The logarithmic scale is $\log_{10}$.

Delta Smelt

Salvage of Delta Smelt continued the pattern of mostly low salvage observed since WY 2005 (Figure 11). Salvage at the SDFPF (1,701) decreased slightly from WY 2012 (1,999) and increased markedly from WY 2011 (0). Similarly, salvage at the TFCF (300) decreased slightly from WY 2012 (355) and increased noticeably from WY 2011 (51).

Salvage of Delta Smelt at both facilities occurred in the middle of the water year (Figure 12). Adult Delta Smelt were salvaged from December-March at the SDFPF. Juvenile Delta Smelt were salvaged from April-June, where June salvage (785) accounted for 46.1% of the total annual salvage. Adult Delta Smelt were salvaged from December-March at the TFCF. Juvenile Delta Smelt were salvaged from April-June, where May salvage (72) accounted for 24.0% of the total annual salvage.

Delta Smelt less than 20 mm FL were first detected at the SDFPF on April 4 and were observed on 14 days of monitoring (Table 3). The longest period of consecutive daily detections was April 21-24. April also recorded the most daily detections (12 days).

Delta Smelt less than 20 mm FL were first detected at the TFCF on March 18 and were observed on 8 days of monitoring (Table 3). The longest period of consecutive daily detections was March 22-23. March also recorded the most daily detections (4 days).

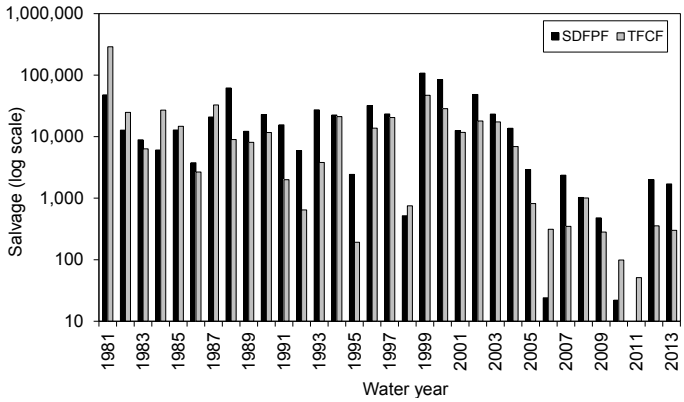

Figure 11 Annual salvages of Delta Smelt at the SDFPF and the TFCF, water years 1981 to 2013. The logarithmic scale is log_{10}.

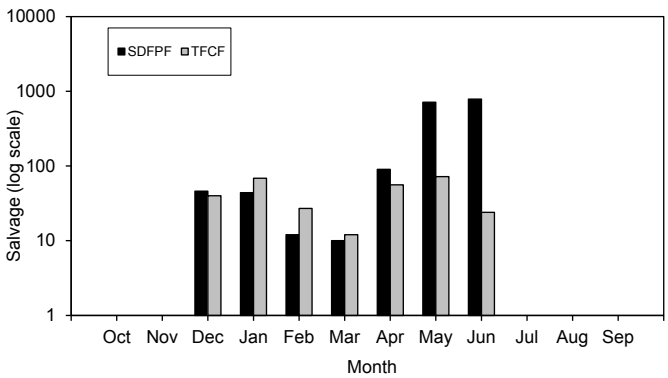

Figure 12 Monthly salvages of Delta Smelt at the SDFPF and the TFCF, WY 2013. The logarithmic scale is log_{10}.

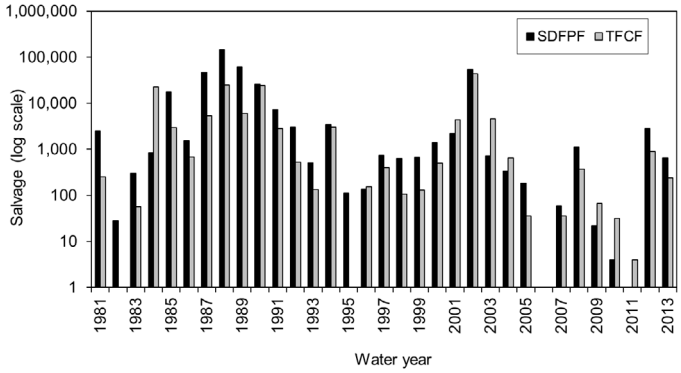

Figure 13 Annual salvages of Longfin Smelt at the SDFPF and the TFCF, water years 1981 to 2013. The logarithmic scale is log_{10}.

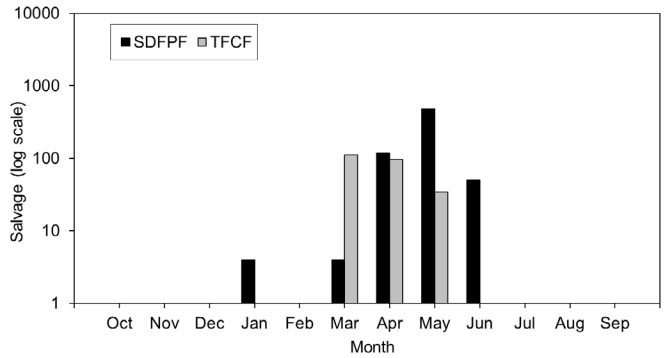

Figure 14 Monthly salvages of Longfin Smelt at the SDFPF and the TFCF, water year 2013. The logarithmic scale is log_{10}.

Longfin Smelt

Salvage at the SDFPF (659) decreased from WY 2012 (2,842) but increased from WY 2011 (0) (Figure 13). Salvage at the TFCF (241) also decreased from WY 2012 (898) but increased from WY 2011 (4).

Longfin Smelt was salvaged in January and from March-June at the SDFPF (Figure 14). May salvage (483) accounted for 73.3% of the total annual salvage. Longfin Smelt was salvaged from March-May at the TFCF. March salvage (111) accounted for 46.1% of the total annual salvage.

Longfin Smelt less than 20 mm FL were first detected at the SDFPF on March 9 and were observed on 13 days of monitoring (Table 3). The longest period of consecutive daily detections was from April 4-10. April also recorded the most daily detections (11 days).

Longfin Smelt less than 20 mm FL were first detected at the TFCF on March 13 and were observed on 17 days of monitoring (Table 3). The longest period of consecutive daily detections was from March 16-19. March also recorded the most daily detections (11 days).

Splittail

Annual salvages of Splittail at both facilities were lower than in WY 2012 (Figure 15). Salvage at the SDFPF (329) was much lower than in WY 2012 (4,057). Salvage at the TFCF was a record low (125), which was substantially lower than in WY 2012 (929) and the record-high in WY 2011 (7,660,024). Annual Splittail salvages have followed a boom-or-bust pattern, often varying year to year by several orders of magnitude.

Table 3 Smelt less than 20 mm fork length (FL) observed in larval samples collected from the SDFPF and the TFCF in WY 2013. Daily numbers of Delta Smelt and Longfin Smelt < 20 mm FL are recorded while an "N" indicates no detection and an "NS" indicates no sampling.

	SDFPF		*TFCF*	
DATE	*Delta Smelt larvae*	*Longfin Smelt larvae*	*Delta Smelt larvae*	*Longfin Smelt larvae*
3/9	N	1	NS	NS
3/13	N	N	N	4
3/14	N	N	N	5
3/16	N	N	N	3
3/17	N	N	N	5
3/18	N	N	1	6
3/19	N	N	N	3
3/22	N	N	1	1
3/23	N	N	2	N
3/24	N	N	N	1
3/25	N	2	N	1
3/26	N	N	1	N
3/28	N	N	N	1
3/31	N	N	N	4
4/4	1	1	NS*	NS*
4/5	2	3	NS*	NS*
4/6	N	1	NS*	NS*
4/7	N	3	NS*	NS*
4/8	2	3	NS*	NS*
4/9	N	2	NS*	NS*
4/10	N	1	NS*	NS*
4/12	1	N	NS*	NS*
4/13	1	N	NS*	NS*
4/15	N	N	N	1
4/16	N	N	N	1
4/18	1	1	N	1
4/20	N	N	N	1
4/21	16	3	N	N
4/22	7	3	N	1
4/23	8	N	N	N
4/24	1	N	N	N
4/29	1	1	N	N
4/30	1	N	1	N
5/7	NS**	NS**	N	1
5/13	N	N	1	N
5/19	N	N	1	N
5/21	N	N	1	N
6/1	1	N	N	N
6/3	1	N	N	N

NS* The TFCF was non-operational during April 2-14 due to installation of a new hoist trolley beam

NS** The SDFPF was non-operational due to inspections and replacing pumps

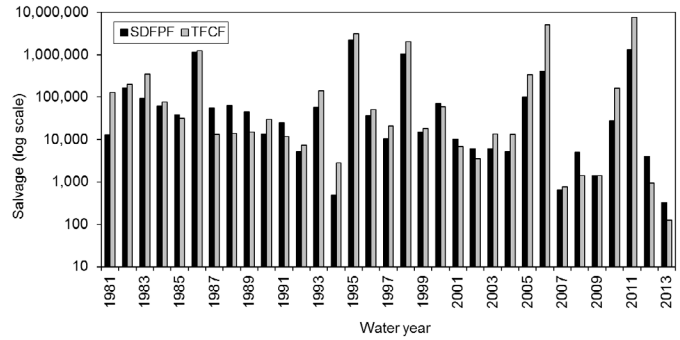

Figure 15 Annual salvages of Splittail at the SDFPF and the TFCF, water years 1981 to 2013. The logarithmic scale is $\log_{10}$.

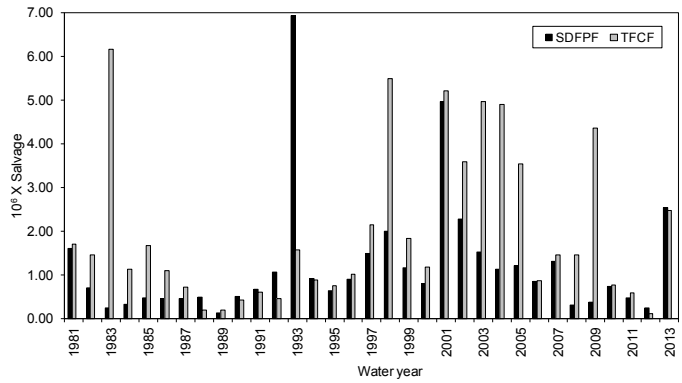

Figure 16 Annual salvages of Threadfin Shad at the SDFPF and the TFCF, water years 1981 to 2013.

Threadfin Shad

Annual salvage of Threadfin Shad at the SDFPF (2,535,117) was slightly higher than at the TFCF (2,463,695) (Figure 16). Salvage at the SDFPF was higher than in WY 2012 (238,135). Similarly, TFCF salvage was much higher than in WY 2012 (109,610), which was a record low. Similar to Splittail, annual salvages of Threadfin Shad have varied greatly through time.

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California Dept. of Fish and Wildlife. 2014. "Delta Model length at date table." Available at: ftp://ftp.dfg.ca.gov/salvage/.

Central Valley Chinook Salmon Harvest and Escapement

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This paper will present the available Chinook escapement and harvest estimates, with a focus on the California Central Valley escapement. The available estimates will be compared to estimates from earlier years and the data will be plotted. The data were collected from the Pacific Fishery Management Council (PFMC) Annual Fisheries Review, biologists throughout the Sacramento and San Joaquin River systems, and from the GrandTab database output.

California Ocean Harvest

The estimated harvest in California ocean waters was 410,687 Chinook Salmon in 2013, 122% of the 337,663 in 2012 (Pacific Fishery Management Council 2014). This is the highest since 2004, but 76% of the 40 year mean ocean harvest of 540,674 (Figure 1).

California Central Valley Harvest

The estimated harvest in Central Valley waters was 66,555 Chinook Salmon in 2013, 80% of the harvest of 83,145 in 2012. The harvest of late-fall run was 2,164 in 2013, 233% of the 930 in 2012. There was no winter run harvest in 2013, due to selective fishing regulations put into effect in 2010. The harvest of spring-run was 523 in 2013, 74% of the 708 in 2012. The harvest of Sacramento fall-run was 61,672 in 2013, 80% of the 76,628 in 2012. The harvest of San Joaquin fall-run was 2,196 in 2013, 45% of the 4,869 Chinook Salmon in 2012.

California Central Valley Escapement

The California Central Valley contains the Sacramento and San Joaquin river systems. The Sacramento River system is made up of the mainstem Sacramento River and the many tributaries that flow into it. Likewise, the San Joaquin River also has many tributaries. Each year, escapement estimates are made for Chinook Salmon that return to spawn in natural areas and for those that return to hatcheries within these river systems. These estimates are in addition to the inland harvest estimates.

In 2013, the escapement estimate for Chinook Salmon returning to hatcheries and natural areas of California's Central Valley was 492,423 fish (Azat 2014). This is the highest since 2005, and 156% of the 40 year mean of 315,591 (Figure 2). The late-fall-run escapement was 8,953, the winter-run escapement was 6,123, the spring-run escapement was 23,697, and the fall-run escapement was 453,650 Chinook Salmon. Escapement increased in 2013 compared to 2012 for all runs.

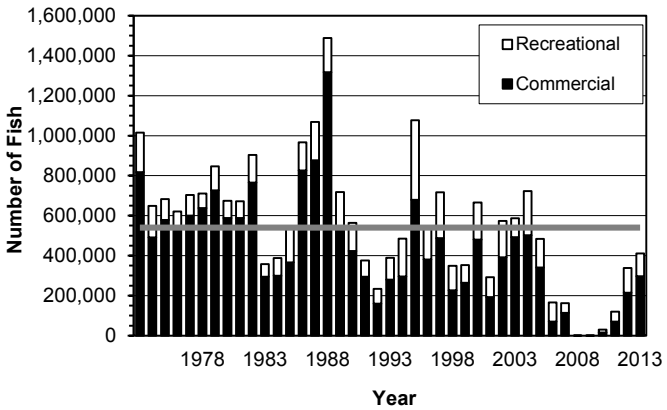

Figure 1 California commercial and recreational Chinook Salmon ocean catch from 1974 to 2013 and 40 year mean (gray line).

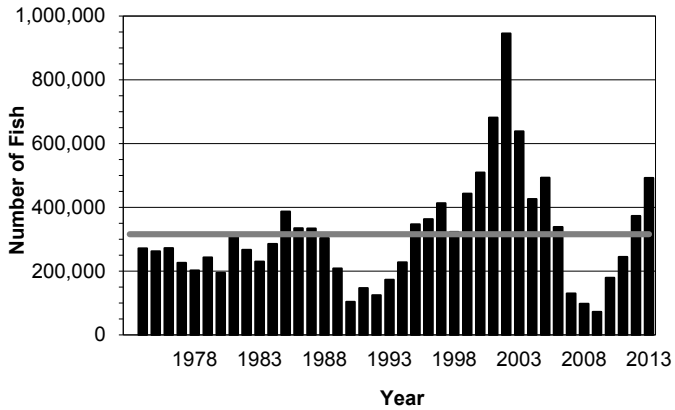

Figure 2 Annual Chinook Salmon escapement to the California Central Valley from 1974 to 2013 and 40 year mean (gray line).

Late-fall-run Escapement to the Sacramento River System

The estimated escapement of late-fall-run Chinook Salmon to the Sacramento River and its tributaries was 8,953 in 2013. This is 150% of the 5,969 of 2012, and is 75% of the 40 year mean of 11,984 (Figure 3). This marks the end of the continuous decline since 2007. Escapement to the Sacramento River was 5,257. Escapement to Battle Creek was 3,619. Most of the late-fall run in Battle Creek were counted at Coleman National Fish Hatchery, where the fish are propagated.

Winter-run Escapement to the Sacramento River

The estimated escapement of winter-run Chinook Salmon to the Sacramento River in 2013 was 6,123. This is 229% of last year's escapement of 2,674, and 91% of the 40 year mean of 6,699 (Figure 4).

Spring-run Escapement to the Sacramento River System

The estimated escapement of spring-run Chinook Salmon to the Sacramento River and its tributaries was 23,697 in 2013, 107% of the 2012 estimate of 22,249, and 163% of the 40 year mean of 14,514 (Figure 5). The majority of these fish were from Butte Creek with an estimate of 16,783 Chinook Salmon. It should be noted that carcass survey estimates have replaced the last 11 years of Butte Creek snorkel survey estimates in GrandTab as the official DFW estimate. Clint Garman's work on Butte Creek shows that the mean snorkel survey estimate has been roughly 54% of the mean carcass survey estimate over the last 11 years. The 2013 snorkel survey data indicated an estimate of 11,470 fish.

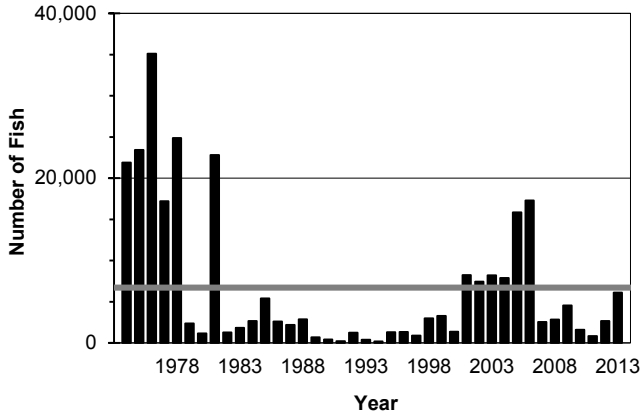

Figure 4 Annual winter-run Chinook Salmon escapement to the Sacramento River from 1974 to 2013 and 40 year mean (gray line).

Fall-run Escapement to the Sacramento River System

The estimated escapement of fall-run Chinook Salmon to the Sacramento River and its tributaries was 432,703 in 2013, the highest since 2003, and 164% of the 40 year mean of 263,462 (Figure 6).

Escapement to the Sacramento River and its tributaries upstream of Red Bluff Diversion Dam (RBDD) was 152,774, 141% of the 40 year mean of 108,732 Chinook

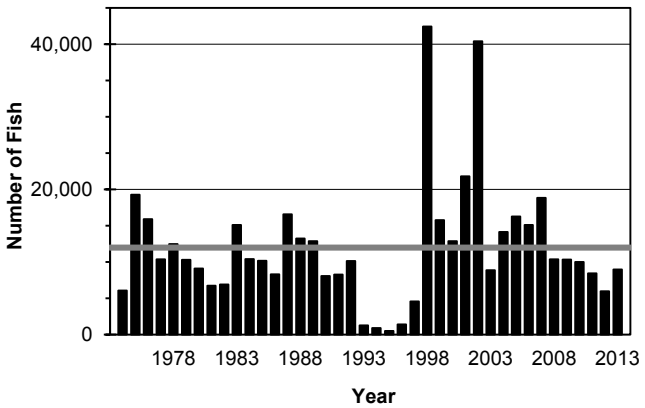

Figure 3 Annual late-fall-run Chinook Salmon escapement to the Sacramento River system from 1974 to 2013 and 40 year mean (gray line).

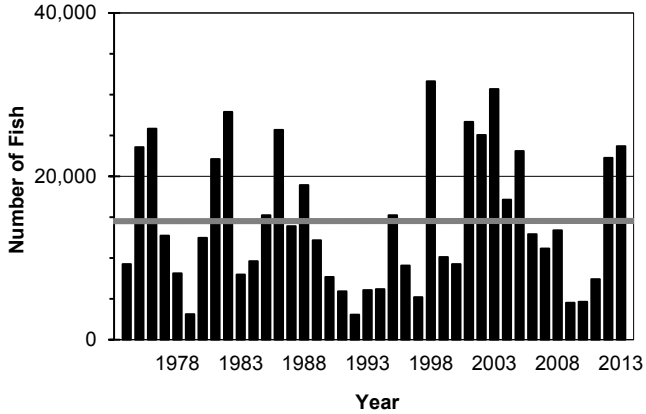

Figure 5 Annual spring-run Chinook Salmon escapement to Sacramento River tributaries from 1974 to 2013 and 40 year mean (gray line).

Salmon. Escapement to the Sacramento River and its tributaries between RBDD and Princeton Ferry was 10,792, 49% of the 40 year mean of 22,106 Chinook Salmon. Escapement to Sacramento River tributaries between Princeton Ferry and Sacramento was 269,137, 203% of the 40 year mean of 132,623 Chinook Salmon.

Fall-run Escapement to the San Joaquin River System

The estimated escapement of fall-run Chinook Salmon to the San Joaquin River and its tributaries was 20,947 in 2013. This estimate is 111% of the 40 year mean of 18,901 (Figure 7).

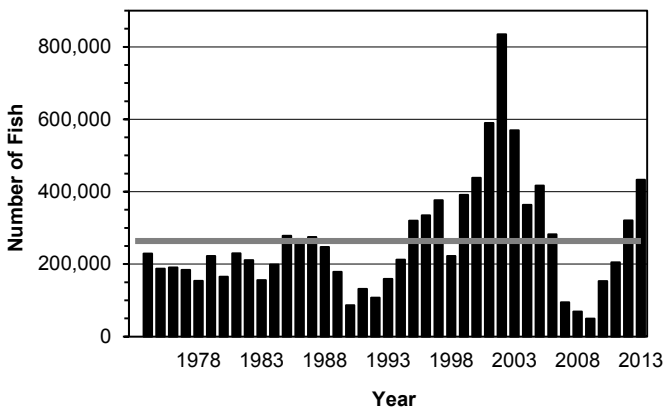

Figure 6 Annual fall-run Chinook Salmon escapement to the Sacramento River system from 1974 to 2013 and 40 year mean (gray line).

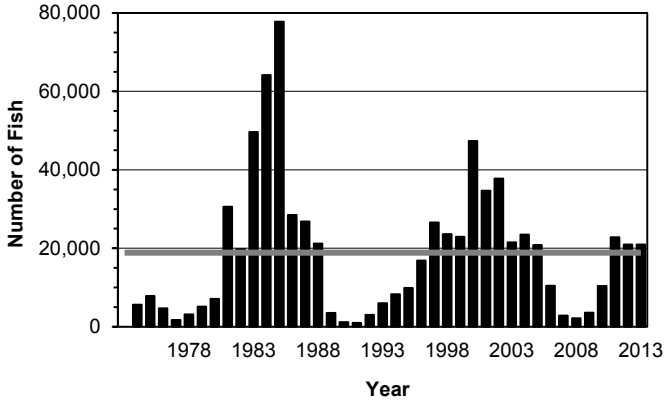

Figure 7 Annual fall-run Chinook Salmon escapement to the San Joaquin River system from 1974 to 2013 and 40 year mean (gray line).

The Chinook escapement and harvest estimates presented in this paper do not attempt to give a total Chinook population estimate, but rather the data from those areas where estimates are made are presented for comparison with previous years. The GrandTab database is a collection of these estimates in the Central Valley. The estimates are reviewed, recalculated, and finalized by the Department of Fish and Wildlife Fisheries Branch, and though they cannot give a complete accounting of the Chinook population, they present the best available data.

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Pacific Fishery Management Council. 2014. "Review of 2013 Ocean Salmon Fisheries." Prepared for the Council and its advisory entities, 7700 NE Ambassador Place, Suite 101, Portland, Oregon, 97220-1384. Available at http://www.pcouncil.org.

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Anderson, R. CDFW, North Coast Region (NCR), personal communication.

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Tsao, S. CDFW, San Joaquin Valley and Southern Sierra Region (SJVSSR), personal communication.

Contributed Papers

Continuous Water-Quality and Suspended-Sediment Transport Monitoring in the San Francisco Bay, California, Water Years 2011–13

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The U.S. Geological Survey (USGS) monitors water quality and suspended-sediment transport in the San Francisco Bay. The San Francisco Bay area is home to millions of people, and the bay teems with both resident and migratory wildlife, plants, and fish. Fresh water mixes with salt water in the bay, which is subject both to riverine and marine (tides, waves, influx of salt water) influences. To understand this environment, the USGS, along with its partners (see "Acknowledgements"), has been monitoring the bay's waters continuously since 1988. Several water-quality variables are of particular importance to State and Federal resource managers and are monitored at key locations throughout the bay (Figure 1). Salinity, which indicates the relative mixing of fresh and ocean waters in the bay, is derived from specific conductance measurements. Water temperature, along with salinity, affects the density of water, which causes gravity driven circulation patterns and vertical stratification in the water column. Turbidity is measured using light-scattering from suspended solids in water, and is used as a surrogate for suspended-sediment concentration (SSC). Suspended sediment often carries adsorbed contaminants; attenuates sunlight in the water column; deposits on tidal marsh and intertidal mudflats, which can help sustain these habitats as sea level rises; and deposits in ports and shipping channels, which can necessitate dredging. Dissolved oxygen, which is essential to a healthy ecosystem, is a fundamental indicator of water quality, and its concentration is affected by water temperature, salinity, ecosystem metabolism, tidal currents, and wind. Tidal currents in the bay reverse four times a day, and wind direction and intensity typically change on a daily cycle: consequently, salinity, water temperature, suspended-sediment concentration, and dissolved-oxygen concentration vary spatially and temporally throughout the bay, and continuous measurements are needed to observe these changes. The purpose of this fact sheet is to inform the public of the availability of these water-quality and suspended-sediment data.

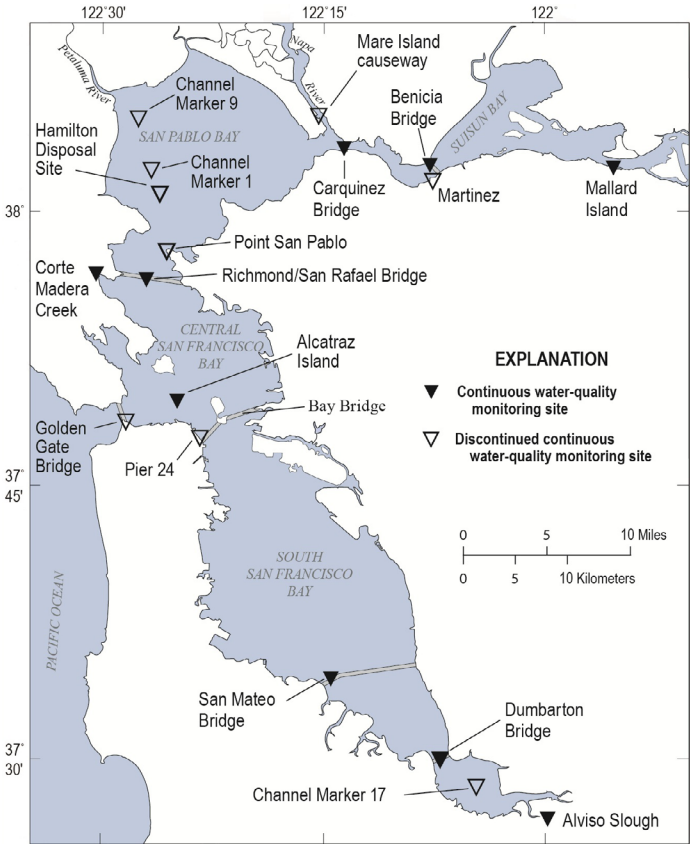

Figure 1 San Francisco Bay study area, California.

Program Overview

Continuous water-quality and suspended-sediment measurements are, or have been, collected at several monitoring stations in the bay (Figure 1, Table 1), as described at http://ca.water.usgs.gov/projects/baydelta/. Instruments usually are deployed in the water by suspension from a stainless-steel cable that is anchored to the bottom (Figure 2). Instruments are equipped with a variety of sensors (explained later), and data are recorded every 15 minutes. Data are retrieved by using cellular telemetry (these data are available on the internet within 1 hour of measurement) or, for stations without telemetry, are downloaded during periodic site visits (these data are available on the internet within 2 weeks of the site visit). Biological growth, which can affect sensor readings, usually increases with time, and the affected data need to be revised or deleted. Every 2–5 weeks (usually 3 weeks), each site is visited to clean and calibrate the instruments and, if needed, download data. Water samples are collected from the same depth as the sensor to calibrate the turbidity data to SSC (Figure 3). For stations that compute water discharge and cross-sectionally averaged SSC, water samples are collected periodically at points across the channel by using the equal-discharge-increment method and velocity is measured by using an acoustic Doppler current profiler (Figure 4). To calculate the suspended-sediment flux, in mass per unit time, the cross-sectionally averaged SSC is multiplied by the water discharge passing through the cross section. Data are reviewed and edited before final approval. Details about these methods are available at http://ca.water.usgs.gov/projects/baydelta/methods.html.

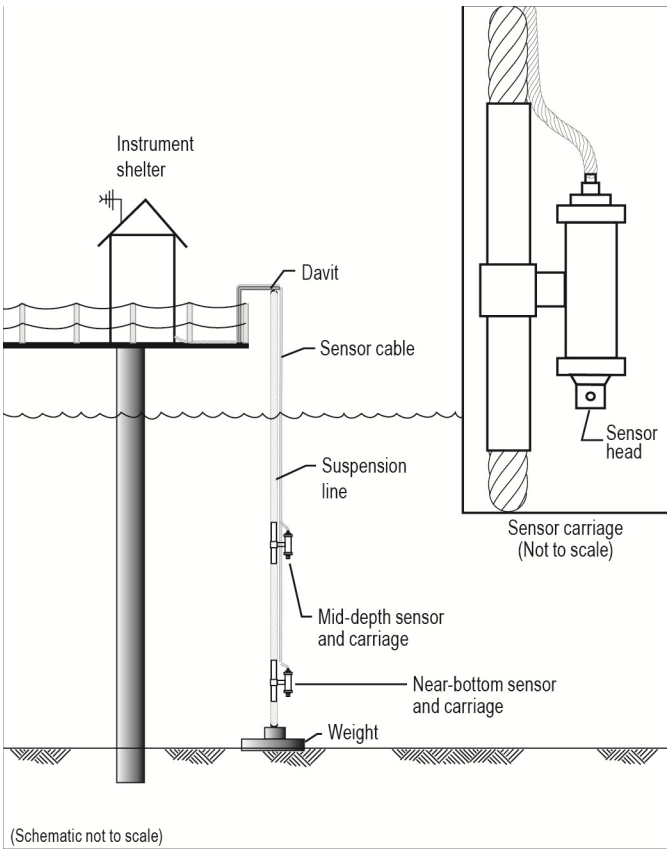

Figure 2 Typical monitoring installation, San Francisco Bay study.

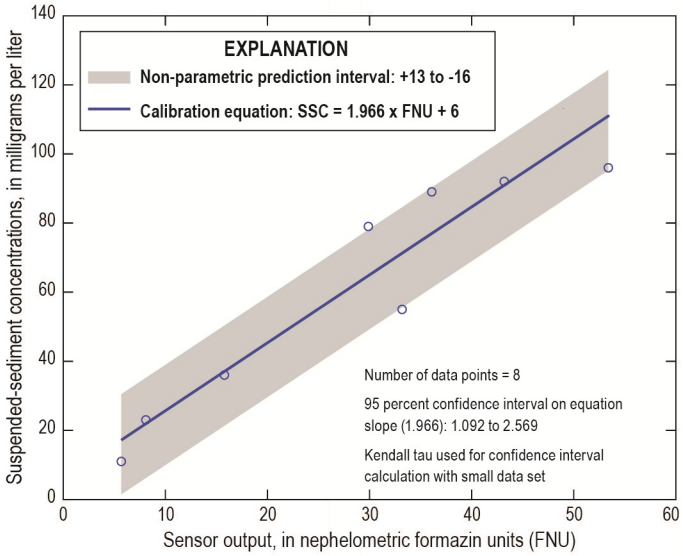

Figure 3 Example calibration curve that relates turbidity to suspended-sediment concentrations (SSC). Output of an optical turbidity sensor is related to the SSC measured in water samples.

Figure 4. U.S. Geological Survey scientists collecting velocity data by using an acoustic Doppler current profiler and suspended-sediment concentration samples by using a depth-integrated sampler.

Specific-conductance, water-temperature, and turbidity data are collected at two depths in the water column to help define the vertical variability. At the shallow water sites (Alviso Slough, Alcatraz Island, and Corte Madera Creek), however, data are collected only at one depth. At the sites where dissolved oxygen is monitored, data are collected at the near-bottom position only (Table 1).

Instrument Specifications

Specific conductance (reported in microsiemens per centimeter at 25 °Celsius) and water temperature (reported in degrees Celsius) have been measured by using a YSI, Inc., conductance/temperature sensor[2] (Figure 5). Two types of optical sensors have been used to monitor turbidity: the DTS-12, manufactured by Forest Technology Systems, and the model 6136, manufactured by YSI, Inc. Dissolved oxygen has been measured by using the optical model 6150, manufactured by YSI, Inc. Sensors manufactured by YSI, Inc., are installed on the 6920 multi-parameter water-quality logger.

In an environmental monitoring program, potential sources of introduced error include, but are not limited to, electronic drift, calibration errors, and biological fouling of sensors. Data corrections (necessary because of biological fouling or instrument electronic drift) have been applied to the affected periods of record following USGS guidelines (http://ca.water.usgs.gov/projects/baydelta/methods.html).

Continuous water-quality and suspended-sediment transport data collected during water years 2011–13 (October 2010 to September 2013) are archived in the USGS National Water Information System and are available to the public at http://waterdata.usgs.gov/ca/nwis/nwis.

For additional information:

http://ca.water.usgs.gov/projects/baydelta/
Contact: Paul Buchanan
California Water Science Center
6000 J Street, Placer Hall, Sacramento, CA 95819
buchanan@usgs.gov.

[2] The use of firm, trade, and brand names in this report is for identification purposes only and does not constitute endorsement by the U.S. Geological Survey.

Figure 5 Water-quality instrument ready for redeployment at Alcatraz Island.

Also published as USGS Fact Sheet: Buchanan, P.A., Downing-Kunz, M.A., Schoellhamer, D.H., Shellenbarger, G.G., and Weidich, K.W., 2014, Continuous water-quality and suspended-sediment transport monitoring in the San Francisco Bay, California: U.S. Geological Survey Fact Sheet 2014–3090, 4 p., http://dx.doi.org/10.3133/fs20143090.

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Table 1 Continuous water-quality monitoring stations, Suisun Bay, San Pablo Bay, and Central and South San Francisco Bays, California, water years 2011–13.

Water-quality parameter	*Measurement location*	*Period of record*	*Remarks*
Alviso Slough near Alviso, 11169750			
SpC, T, turbidity, SSC, DO	Near-bottom	2010–present	
SSC	Cross-section	2010–present	
Q, SSF	Cross-section	2010–present	Q and SSF data is not yet available from NWIS.
SF Bay at Dumbarton Bridge, 373015122071000			
Turbidity, SSC	Mid-depth	1993-present	Station temporarily discontinued because of bridge construction October 1, 2011 - March 16, 2003 Data is downloaded hourly by using cellular telemetry.
Turbidity, SSC, DO	Near-bottom	1993-present	
SSC	Cross-section	2009-present	
Q, SSF	Cross-section	2009-present	Q and SSF data is not yet available from NWIS.
SF Bay at San Mateo Bridge near Foster City, 11162765			
SpC, T	Near-bottom	1989-present	Turbidity and SSC collected from 1993 to 2005. Data is downloaded hourly by using cellular telemetry.
SpC, T, DO	Near-bottom	1989-present	
SF Bay at Alcatraz Island, 374938122251801			
SpC, T, turbidity, SSC	Mid-depth	2003-present	Data is downloaded hourly by using cellular telemetry.
Corte Madera Creek near Larkspur, 11460090			
SpC, T, turbidity, SSC, DO	Mid-depth	2010-2013	Station discontinued October 31, 2013.
SSC	Cross-section	2010-2013	
Q, SSF	Cross-section	2010-2013	Q and SSF data is not yet available from NWIS.
SF Bay at Richmond/San Rafael Bridge, 375607122264701			
SpC, T, turbidity, SSC	Mid-depth	2006-present	
SpC, T, turbidity, SSC, DO	Near-bottom	2006-present	
Carquinez Strait at Carquinez Bridge, 11455820			
SpC, T	Mid-depth	1998-present	Turbidity and SSC collected from 1998 to 2005.
SpC, T	Near-bottom	1998-present	
Suisun Bay at Benicia Bridge, 11455780			
SpC, T, turbidity, SSC	Near-surface	2001-present	
SpC, T, turbidity, SSC, DO	Near-bottom	2001-present	
Suisun Bay at Mallard Island, 11185185			
Turbidity, SSC	Near-surface	1994-present	Near-surface sensor attached to a float to maintain a constant depth below water surface of 3.3 feet. Data is downloaded hourly by using cellular telemetry.
Turbidity, SSC	Near-bottom	1994-present	

Abbreviations: SF, San Francisco; SpC, specific conductance; T, water temperature; SSC, suspended sediment concentration; DO, dissolved oxygen; Q, water discharge; SSF, suspended sediment flux; NWIS, National Water Inventory System; -, not applicable

Summer Townet 2007-2013 Gelatinous Zooplankton (jellyfish) Summary

By Trishelle Morris (CDFW), trishelle.morris@wildlife.ca.gov

Introduction

Jellyfish are an increasingly common sight in many aquatic environments around the world. Seasonally dense aggregations of jellyfish are a natural component of many healthy ecosystems, but the frequency and severity of such blooms seem to be on the rise (Richardson et al. 2009). Jellyfish blooms are associated with several negative ecosystem impacts: reduced or altered zooplankton populations, impeded fishing operations, and clogged water intake screens (Purcell et al. 2007). In the Black Sea, introduced jellyfish populations exploded as conditions became favorable to certain species, most notably in the late 1980s when populations of the scyphozoan jellyfish *Aurelia aurita* were consuming up to 62% of the annual zooplankton production (Zaitsev and Mamaev 1997). These correlated trends shifted the ecosystem from supporting a number of valuable fisheries to supporting high volumes of jellyfish, both native and invasive (Mills 2001). Several species of jellyfish have been introduced to the San Francisco Estuary, likely from the Black and Caspian seas. These invasive jellyfish appear to be numerous in the estuary, yet information regarding these species is limited.

The two morphologically different body forms of jellyfish – polyp and medusa – may contribute to their success as an invasive species. The free swimming medusa and the stationary polyp typically represent different life stages (Pechinik 2005). Within class hydrozoa, reproduction can occur asexually by budding from the polyp colony, or sexually when medusae release gametes (Pechinik 2005). These dual reproduction methods give hydromedusae the ability to reproduce rapidly while maintaining high genetic diversity, giving them an advantage in changing environments (Meek et al. 2013).

Four jellyfish species native to the Caspian and Black seas have been identified in the upper San Francisco estuary; *Maeotias marginata*, *Moerisia* spp., *Blackfordia virginica*, and *Cordylophora caspia* (Mills and Rees 2000). These species exploit a novel niche in the ecosystem, since no jellyfish are native to the low salinity zone (salinity 1-6 ppt) in the estuary. *M. marginata* is the largest of these species with a bell width up to 50 mm, followed by *B. virginica* with a bell width up to 20 mm (Rees and Kitting 2002). *M. marginata* may disrupt the food web by preying heavily on crustaceans and larval fish, and have no known predators (Rees and Gershwin 2000).

The Summer Townet Survey (STN) began formally identifying and enumerating jellyfish catch in 2007, but jellyfish presence has been noted on datasheets since the beginning of the study in 1959. The California Department of Fish and Wildlife (CDFW) initiated STN in order to determine distribution and relative abundance of young-of-the-year (age-0) Striped Bass (*Morone saxatilis*), and has since expanded the study to describe distribution of other fish species and the presence of many invertebrate species. The mesh size of the STN net is designed to target juvenile fish (25-50 mm), rendering it ineffective in catching small medusae or jellyfish polyps. As a result, the medusae form of *M. marginata* is the only jellyfish species regularly seen in STN catch. Other jellyfish are within the target range of the STN, but occur in higher salinity water mainly outside the sampling area. In this paper, I summarize observations of *M. marginata* medusae collected during 2007-2013 to depict average catch-per-unit-effort (CPUE) monthly across years, regionally within the estuary and across scales of temperature (°C), salinity (ppt), Secchi disk depth (cm), and water depths (ft).

Methods

The STN is conducted every two weeks, sampling 32 historic stations ranging from the San Pablo Bay to the east Delta, from June through August (Figure 1). In 2009, 5 supplemental stations were added in the Sacramento River Deep Water Ship Channel (SRDWSC), along with 3 more supplemental stations in Cache Slough (CS) in 2011 to more adequately describe Delta Smelt (*Hypomesus transpacificus*) distribution (Figure 1). The following summary of *M. marginata* catch pertains to the 32 historic STN stations sampled for the period 2007-2013; at the time of this article, jellyfish had not been observed at new SRDWSC or Cache Slough stations.

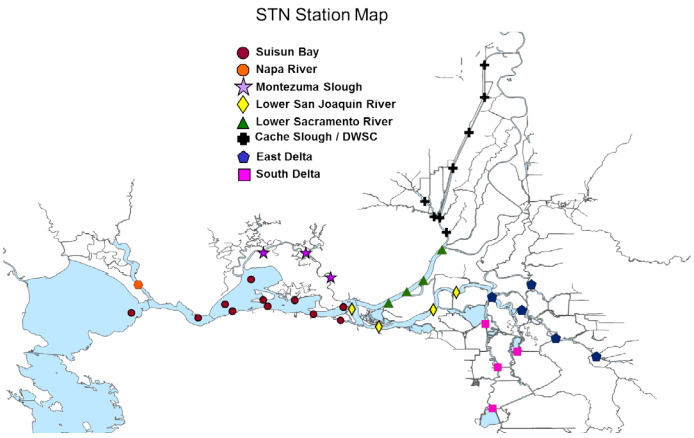

Figure 1 Summer Townet Survey stations by region in the upper San Francisco Estuary.

Jellyfish were retained in the codend portion of the townet, which is composed of stretch, knotless Nylon mesh with 0.25 cm (0.1 inch) diameter openings. For a full description of the townet gear, see Osborn and Avila (this issue). Two 10-minute stepped oblique tows were performed at each station; if fish were caught during these tows, a third tow was performed. In 2007, staff began identifying and enumerating all jellyfish caught per tow. Occasionally, jellyfish catch was estimated when catch was greater than 50 organisms.

Since *M. marginata* dominated STN jellyfish catch, I only touched on the presence of other taxa, summarized as total catch. The CPUE for *M. marginata* was calculated for each tow as: CPUEtow = (catch / volume (m^3) of water sampled) * 10,000. Next, I calculated the average CPUE per station as: CPUEstation = (sum CPUE per tow) / (number of tows conducted). *M. marginata* distribution was described by averaging CPUEstation for the following factors: time – by month and year; space – by embayment; and environment – by temperature, salinity, water depth, and Secchi depth. Environmental factors were summarized by intervals: 1°C for temperature, 1 ppt for salinity, 5 feet for water depth, and 10 cm for Secchi depth. Standard error was calculated for each of these intervals. To describe trends in density both spatially and temporally, I plotted the fraction of zero, low, medium, and high density observations for each station by month, averaged by survey and year. Low density was defined as 1-49 organisms collected per station in a single day, medium density was defined as 50-99 organisms collected, and high density was defined as 100+ organisms collected.

Results

From 2007 to 2013, six jellyfish taxa were observed, four hydromedusae: *M. marginata* (n=10,934), *Aequorea* spp. (n=36), *Polyorchis penicillatus* (n=3), *Scrippsia pacifica* (n=1); one scyphomedusae complex: *Aurelia* spp. (n=2); and one cydippid ctenophore: *Pleurobrachia bachei* (n=12, native). Of these taxa, *M. marginata* is commonly found the upper estuary while the other taxa are commonly found in San Francisco and San Pablo bays (Rees and Kitting 2002). Low catch of five taxa was likely due to the lack of STN stations in these bays; therefore the following analysis pertains only to the abundant *M. marginata*. In a prior issue, Osborn and Civiello (2013) provided an in-depth analysis of Fall Midwater Trawl (FMWT) data regarding observations of all jellyfish taxa caught in the FMWT survey, which sampled San Pablo Bay more extensively and offered a deeper look into some of the species found in the estuary but not regularly seen on STN.

M. marginata catch varied from year to year over the collection period. Average monthly CPUE of *M. marginata* numbers peaked in August 2007. Each year, abundance was highest in July or August (Figure 2). In 2011, observations fell to their lowest annual level since STN began recording jellyfish catch. *M. marginata* were caught at all stations west of the confluence, and once in the lower San Joaquin River at station 812 in 2009. Average embayment CPUE was highest in Montezuma Slough (Table 1). Medusae were seen across broad temperature and salinity

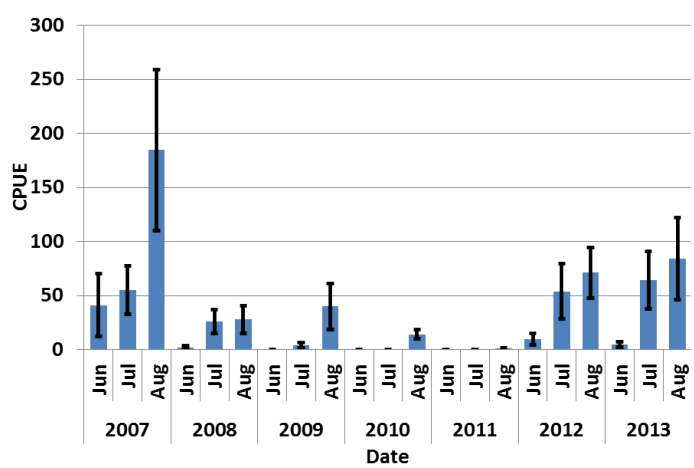

Figure 2 Average CPUE (±SE) of *Maeotias marginata* by month and year collected 2007 –2013 during the Summer Townet Survey.

Table 1 Average CPUE (±SE) of *Maeotias marginata* by region, collected 2007-2013 during the Summer Townet Survey.

Region	*CPUE*	*±SE*
Suisun Bay	13.13	2.02
Napa River	1.81	1.79
Montezuma Slough	294.71	48.34
Lower San Joaquin River	1.13	0.40
East Delta	0.00	0.00
South Delta	0.00	0.00
Lower Sacramento River	1.31	0.72

ranges, CPUE was highest between 20.0-22.9 °C (Figure 3) and 4.0-7.9 ppt (Figure 4). Highest CPUE was seen at water depths between 20 and 29 feet, but medusae were present at stations with depths up 60 feet (Figure 5). Lastly, medusae were caught within all Secchi depth intervals, up to 100 cm, but were not caught where Secchi values exceeded 100 cm. CPUE was highest between Secchi depths of 20-59 cm (Figure 6). On average, *M. marginata* catch increased in number and in density in an eastward fashion as the summer progressed (Figures 7A-7C).

Discussion

M. marginata catch fluctuated among years, but was highest in August each year, with the exception of 2008 when catches in July and August were comparable. This indicates that medusae presence may increase as the summer progresses due to continuous reproduction or increased net retention or both. Polyps can release additional medusae as long as favorable conditions last; Rees and Gershwin (2000) found newly released medusae as late as October. Medusae growth may also enhance catch records in STN in the late summer months, as larger medusae are more efficiently retained by the gear and more easily observed in the sample. The peaks observed in July and August of each year are likely the result of a combination of these two processes. Lastly, the higher catches later in the summer may be partially explained by salinity. As the summer progressed, average salinity typically increased in the eastern part of Suisun Bay to a range suitable for *M. marginata* medusae catch, which may have opened up new habitat in the latter part of the summer.

Inter-year trends partially mirrored Schroeter's (2008) findings. 2010 was a dry year and *M. marginata* catch was

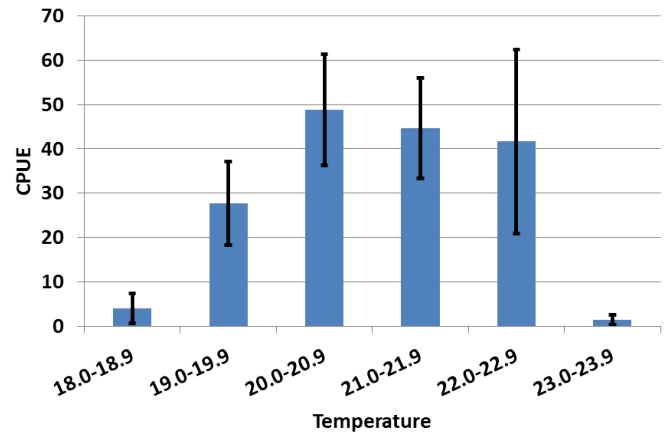

Figure 3 Average CPUE (±SE) of *Maeotias marginata* by water temperature intervals (1 °C), collected 2007-2013 during the Summer Townet Survey.

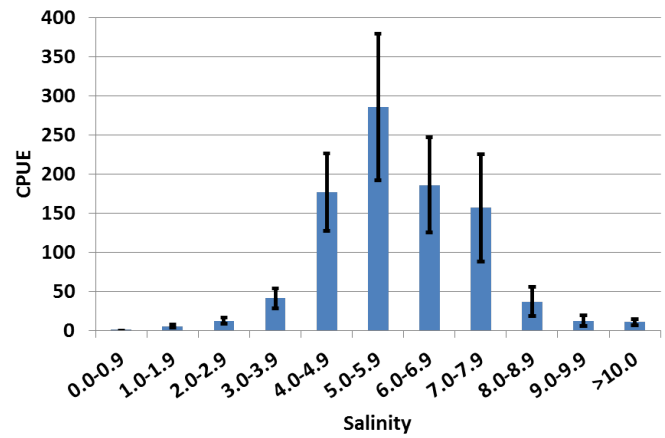

Figure 4 Average CPUE (±SE) of *Maeotias marginata* by 1 ppt salinity interval, collected 2007-2013 during the Summer Townet Survey.

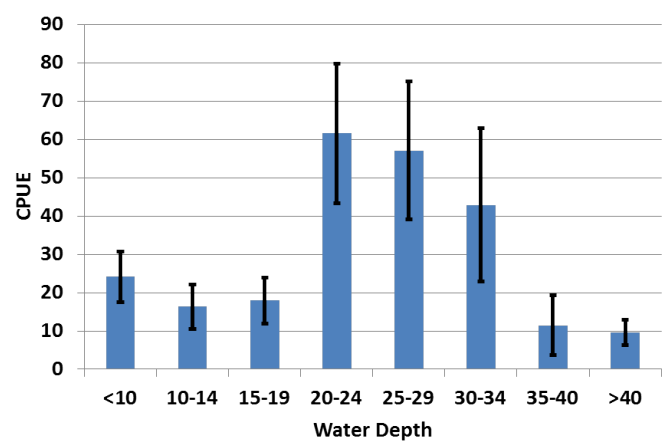

Figure 5 Average CPUE (±SE) of *Maeotias marginata* by water depth interval (5 ft) collected 2007-2013 during the Summer Townet Survey.

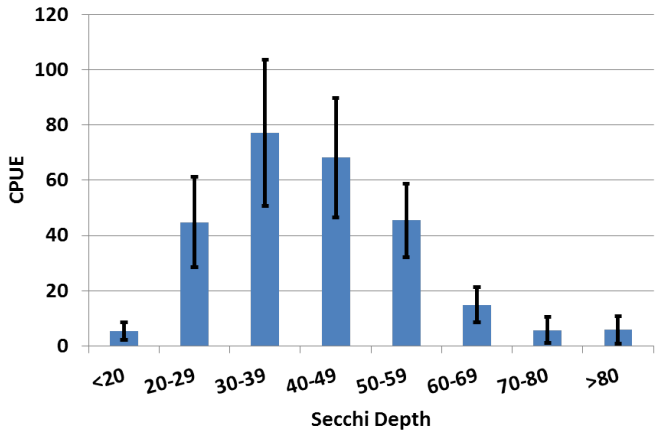

Figure 6 Average CPUE (±SE) of *Maeotias marginata* by Secchi depth interval (10 cm), collected 2007-2013 during the Summer Townet Survey.

low, an association seen in the past (Schroeter 2008). Low catch in dry years may stem from a variety of factors, including polyp mortality at high salinities (Schroeter 2008). Low catches were also recorded in the high flow year 2011, when catch was absent until August. However, FMWT showed high catches that fall (Osborn and Civiello 2013). Therefore, the high-flow, low-salinity conditions likely pushed bloom initiation later in the year, perhaps as late as August.

Given the differences between the STN and FMWT, we cannot infer medusae abundance trends from one survey to the other. However, following the low numbers seen in 2010 and 2011, *M. marginata* patterns in 2012 and 2013 returned back to "normal," resembling data from 2007-2009. It has been suggested that the medusae stage and polyp stage of *M. marginata* have different environmental tolerances (Schroeter 2008). Currently, there is very little information available on the *M. marginata* polyp, so their environmental requirements are unknown. However, the medusae population has repeatedly rebounded from low abundance years, suggesting polyps can survive conditions unfavorable for medusae development. Furthermore, polyps may be able to postpone medusae release until conditions are favorable (Schroeter 2008).

Montezuma Slough experienced high density catches of *M. marginata* in all years except 2011. Medusae were also caught regularly throughout Suisun Bay, although this is not well represented in average CPUE. *M. marginata* habitat overlaps that of many fish species, including the endangered Delta Smelt. This overlap may adversely affect these fishes, as diet preferences may also overlap. *M. marginata* medusae have been shown to have a varied diet including copepods, amphipods, and crab zoea (Mills and Sommer 1995), likely reducing food availability for larval fishes. Montezuma Slough appeared to provide very favorable habitat for *M. marginata* medusae each month of the study (June-August), high densities (> 49 organisms per tow) of medusae were observed regularly each of these months. Conditions in Suisun Bay typically appeared to become more favorable as the summer progressed. In this region, distribution tended to be more concentrated early in the summer and spread eastward later in the summer (Figures 7A-7C). Densities were typically lower in Suisun Bay than in Montezuma Slough. These patterns likely reflect the environmental suitability of these regions. The warm, brackish, low flow waters seen in Montezuma Slough may have provided good habitat regularly, whereas in Suisun Bay increasing salinity towards the east in late summer may have opened up new habitat, enabling *M. marginata* expansion.

STN data showed *M. marginata* can inhabit a wide range of environmental conditions, but were caught most often in warm, brackish water. Specifically, medusae presence was highest from 20.0-22.9 °C and 4.0-7.9 ppt. Schroeter (2008) posited that temperature may be the most influential factor in *M. marginata* bloom initiation. Rees and Gershwin (2000) found *M. marginata* reproduction occurred between 18-21°C in a laboratory setting. STN catch was highest within a slightly higher temperature range, which could reflect the time needed for the medusa to grow to be retained more effectively by the gear. High density catches (> 50 organisms per tow) were widely dispersed, from 18-23 °C. On FMWT, which immediately follows the STN survey, *M. marginata* CPUE peaked from 19-20 °C (Osborn and Civiello 2013). Given the widespread catch over temperature ranges collected during STN, temperature did not seem to be a largely limiting factor in the abundance of *M. marginata* in our data. This may be because STN begins sampling in June, when water is typically warm. By this time of the year, on average, temperatures may have already surpassed a potential threshold critical for bloom initiation.

Schroeter (2008) also found that salinity seemed to play a key role in bloom initiation. During a 1999 jellyfish survey in the San Francisco Estuary, small medusae were abundant at 4-6 ppt, suggesting that polyps may find this range suitable for releasing medusae (Rees and Kitting 2002). This range corresponds to that favored by *M.*

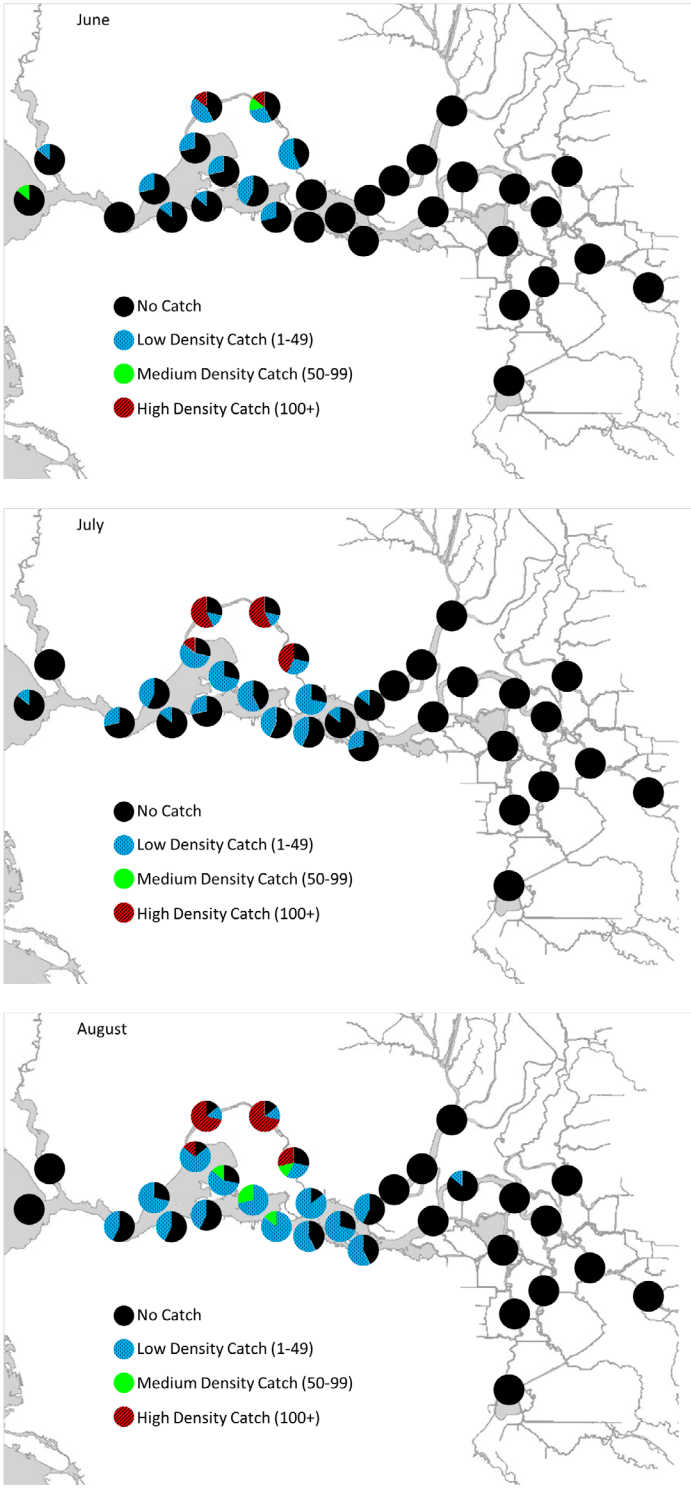

Figure 7 Map of the Summer Townet Survey stations within the known range of *Maeotias marginata*. *M. marginata* catch is represented at each station as the proportion of various catch densities: zero, low density (1-49 individuals), medium density (50-99 individuals), and high density (100+ individuals). Catch densities by station are shown for A) June, B) July, and C) August.

marginata in STN catch. Therefore, salinities experienced in summer in the estuary may facilitate *M. marginata* blooms.

M. marginata were caught regularly in low-flow, warm, brackish areas of the estuary; however, medusae were also present outside of these areas. STN observed medusae across a wide salinity gradient, but this may not indicate survivability. Prior work indicates that the medusa form of *M. marginata* are able to survive more extreme salinities (0-15 ppt), but not for long periods of time (Mills and Rees 2000). During STN, catch was low at salinities above 11 ppt; only one catch exceeded 50 organisms.

Water clarity and water depth also appear to play a role in *M. marginata* distribution. An increase in average CPUE of *M. marginata* medusae was observed within the Secchi range of 20-59 cm, which is in line with Schroeter's (2008) observations that bloom initiation occurred within 19-47 cm Secchi depths. Purcell et al. (2007) suggested that low light levels may inhibit medusae production by polyps, which may be a factor influencing STN catch in this range. Average CPUE of *M. marginata* medusae peaked in water depths between 20 and 29 ft. This depth characterizes two of the stations where medusae were often observed in greatest numbers, stations 606 and 609 in Montezuma Slough. High catch at these stations is likely due to the influence of other environmental conditions, namely: salinity, temperature, and Secchi depth, as supported by previous research.

Conclusion

M. marginata has been observed throughout the brackish waters of the upper San Francisco Estuary, with predominance during warmer months and in the region of Montezuma Slough. Given the dual life cycle and wide tolerance of environmental conditions, *M. marginata* may have the ability to quickly colonize new areas if environmental conditions shift to those favorable to the species.

STN catch provides an incomplete picture of *M. marginata* presence in the upper San Francisco Estuary. The gear is designed for fish, rendering it less effective for jellyfish. The dual life cycle of *M. marginata* further decreases catch and spatial representation. STN can only capture jellyfish species in the medusae phase, and is likely more efficient at targeting larger bell diameters. In addition, *M. marginata* is thought to spend a majority of

its time resting on the substrate, occasionally pulsing to the surface and drifting back down (Mills and Sommer 1995). As per protocol, the STN spends a short amount of time at the bottom of the water column, which may also lead to underrepresentation of medusae catch.

It is apparent that the invasive *M. marginata* has found a suitable niche in this system, likely in much higher abundances than can be observed in STN data. Increased occurrence and severity of *M. marginata* blooms in the estuary may occur in the future, as climate change and eutrophication can favor development (Purcell et al. 2007). Further studies are needed to better understand the abundance and distribution of this species in the estuary. Continued and improved monitoring may be pertinent in measuring the impact of this invasive species on native populations.

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Factors influencing the incidental take of Delta Smelt and Chinook Salmon catch while monitoring near Chipps Island within the San Francisco Estuary, CA

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Introduction

The US Fish and Wildlife Service's Delta Juvenile Fish Monitoring Program (DJFMP) began annual monitoring of juvenile Chinook Salmon (*Oncorhynchus tshawytscha*) at Chipps Island in 1976 (Figure 1; Kjelson et al. 1982). Monitoring was generally conducted seasonally until 1995 and year-round thereafter to better estimate the relative abundance, temporal distribution, and survival of juvenile Chinook Salmon emigrating from the Sacramento-San Joaquin River Delta into Suisun Bay. However, during water years (October–September) 2007, 2008, 2011, and 2012, the monitoring efforts at Chipps Island were temporarily modified or reduced due to relatively high incidental take of Delta Smelt (*Hypomesus transpacificus*).

The incidental take of fishes listed under the federal and state Endangered Species acts (ESA) is a growing concern to many long-term fish monitoring programs within the San Francisco Estuary. Despite ongoing efforts

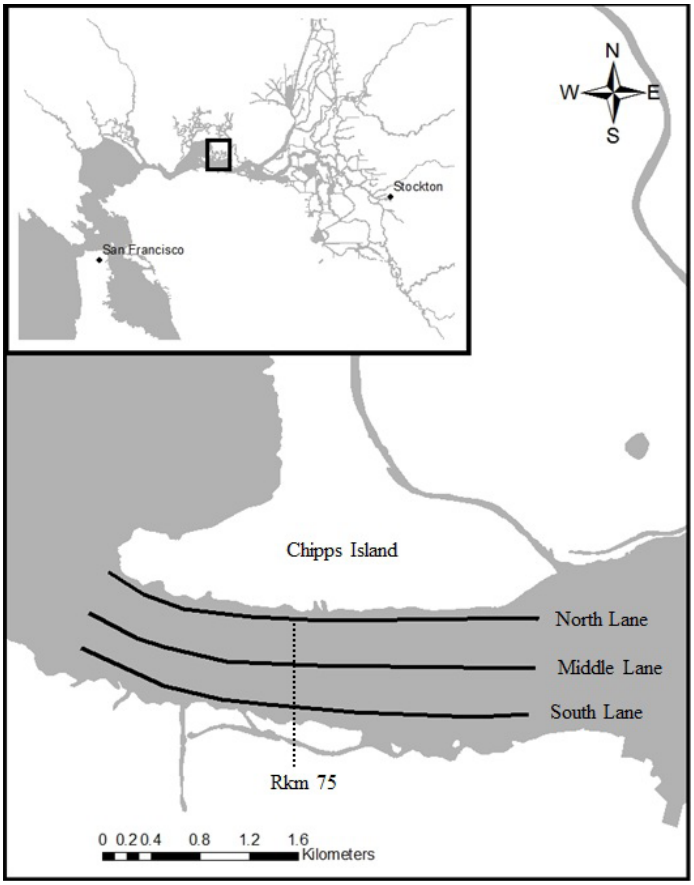

Figure 1 The location of the Chipps Island Trawl Site and the three trawl lanes where fishes are sampled within the San Francisco Estuary, CA, and river kilometers (Rkm) from the Golden Gate Bridge to Chipps Island.

by natural resource managers and monitoring programs, considerable uncertainty exists regarding the relative roles of habitat, abundance, seasonality, and sampling methodologies on the incidental capture of Delta Smelt while monitoring juvenile salmonids within the San Francisco Estuary. Here, we evaluated the relative importance of environmental characteristics and trawl methodology on catches of both Delta Smelt and juvenile Chinook Salmon to determine if and how the monitoring of juvenile Chinook Salmon can be modified at Chipps Island to reduce the future incidental take of Delta Smelt.

Methods

Data used for the analyses included trawl samples collected by the DJFMP at Chipps Island from July 2001 to December 2011. All fish samples were collected using surface trawl methodology and identical mid-water trawl nets with a 0.8 cm cod end mesh size (described in Speegle et al. 2013). In general, surface trawls were conducted for approximately 20 minutes, ten times per day, and three days per week throughout the sample period. However, the duration and frequency of sampling was modified for several weeks in the spring, summer, or fall during 2007, 2008, and 2011 to minimize the incidental take of Delta Smelt. Trawls were conducted facing both upstream and downstream in the north, south, and middle portions of the 4 km channel reach near Chipps Island (Figure 1).

All fishes captured were identified to species and enumerated for each sample. The distance traveled during each trawl relative to the water surface was recorded using a mechanical flow meter (General Oceanics, Model #2030). We estimated the volume of water as cubic meters sampled by multiplying the distance traveled by the mean net mouth area (18.58 m^2; U.S. Fish and Wildlife Service 1993). Weather, water visibility, and water temperature were recorded at the start of each trawl. Weather was recorded as clear, cloud, rain, or night. Water visibility was measured using a Secchi disc and was recorded to the nearest 0.1 meter. Water temperature was measured to the nearest 0.1 degree Celsius (ºC) using either a YSI 30 or 85 meter or to the nearest 0.5 ºC using a standard or digital thermometer. In addition to data collected by the DJFMP while sampling, we obtained predicted tidal stage (ebb, flood, slack high, and slack low) and tidal current data for each tow using the Tides and Current Pro software program (version 3.7). The tidal stage and current data represented the conditions at closest gauging station (i.e., Mallard Island) relative to Chipps Island. We also obtained daily estimated low salinity zone (2 ppt salinity isohaline) position data to the nearest kilometer (km) from DayFlow (DWR 2012a). Hourly local wind speed (m/s) measurements were garnered from the Antioch Pumping Station on the San Joaquin River (DWR 2012b). The Delta Smelt annual relative abundance index during years 2001–2011 was obtained from the California Department of Fish and Wildlife (Annual Fall Midwater Trawl Index Memorandum; Contreras 2011).

Statistical analysis.—The number of individuals captured while sampling is a function of the number of individuals present and the probability of detection (Bayley and Peterson 2001). As a result, we hypothesized a total of five categories of variables would affect the catch of Delta Smelt or juvenile Chinook Salmon by either influencing their occupancy or detection based on previous investiga-

tions (e.g., Stevens and Miller 1983; Sommer et al. 2011): water quality characteristics (temperature, Secchi depth, position of low salinity zone), tide (tidal stage, tidal current, tidal stage and current interaction), weather (rain, not clear, wind), methods (trawl direction relative to tidal current, trawl channel position, volume of water sampled), and the Delta Smelt annual relative abundance index. We evaluated the relative support for the influence of factors on Delta Smelt and juvenile Chinook Salmon catch by developing generalized linear mixed models using Program R (version 3.0.1) and assuming a Poisson distribution (Royle and Dorazio 2008; Zuur et al. 2009). We first developed a global model that contained all of the predictor variables that corresponded to our a priori hypotheses for Delta Smelt and juvenile Chinook Salmon. The Delta Smelt annual relative abundance index was only included within the Delta Smelt model. Prior to model fitting, we calculated Pearson correlation coefficients for all pairs of potential predictor variables to ensure that highly correlated variables ($r^2 > 0.5$) were not used in our analysis to avoid multicollinearity (Dormann et al. 2013). We standardized all continuous data with a mean of zero and a standard deviation of one to facilitate model-fitting and aid in the interpretation of parameter estimates. We also created binary indicator variables for tidal stage (flood, ebb, and slack), weather (rain and not clear), trawl direction relative to tidal current (upstream and downstream), and trawl channel position (north, middle, and south). During exploratory analyses, we determined the best model variance structure for each species by evaluating the goodness-of-fit of the global model using residual plots. The final model structure for both species included randomly varying intercepts that varied among year and month combinations and individual samples to account for temporal dependency and overdispersion, respectively. In addition, all water quality predictor variables contained a quadratic term to account for non-linearity.

We used an information-theoretic approach (Burnham and Anderson 2002) to evaluate the relative support for each species' candidate models that represented each possible combination of our hypothesis categories. We calculated Akaike Information Criteria with a small sample bias adjustment (AICc; Hurvich and Tsai 1989) to assess the fit of each candidate model. The relative fit of each candidate model was determined by calculating ratios of Akaike weights (w_i; Burnham and Anderson 2002). Akaike weights could range from zero to one, with the highest weight being the best fitting model. The precision of fixed effect parameter estimates was assessed by calculating 95% confidence intervals. Confidence intervals that contained zero were considered imprecise and represented a weak or inconclusive relationship and were not used for inference.

Results

A total of 20,131 trawl samples were collected during the study period. The data represented a wide range of environmental conditions and considerable variability in the catch of both juvenile Chinook Salmon and Delta Smelt (Table 1). The global candidate model was the most plausible for predicting Delta Smelt catch (Table 2). Akaike weights indicated that the global model was 6.14 times more supported than the next best candidate model. The best supported model for predicting juvenile Chinook Salmon catch was the candidate model using water quality, tide, and methods as predictors (Table 3). This model was 12.67 times more supported than the next best candidate model. We based all inferences on the best approximating models.

Table 1 Mean and standard deviation (in parentheses) of environmental and methodological variables used to predict Delta Smelt and juvenile Chinook Salmon catch from July 2001 to December 2011 at Chipps Island. The range was provided for continuous variables only.

Variable	*Mean (SD)*	*Range*
Delta Smelt Catch per Trawl	0.3 (0.9)	0–34
Juvenile Chinook Salmon Catch per Trawl	5.4 (14.9)	0 –547
Volume (m^3)	20479.2 (4516.0)	537– 254780
Temperature (°C)	15.57 (4.32)	5.3– 27.0
Secchi Depth (m)	0.47 (0.18)	0.07–1.21
Wind (m/s)	4.4 (2.8)	0–20
Tidal Current (m/s)	0.5 (0.3)	0–1.3
Trawl Moving with Net Tidal Current (proportion of samples)	0.2 (0.4)	0–1
Position of Low Salinty Zone (km)	73.28 (10.85)	43.77–91.84
Annual Fall Midwater Trawl Index of Delta Smelt	105.1 (125.2)	17–603
Ebbing Tide (proportion of samples)	0.5 (0.5)	0–1
Flooding Tide (proportion of samples)	0.4 (0.5)	0–1
Slack High Tide (proportion of samples)	<0.1 (0.2)	0–1
Rainy Weather (proportion of samples)	<0.1 (0.2)	0–1
Other Non-clear Weather (proportion of samples)	0.4 (0.5)	0–1

Table 2 Parameter estimates, standard error, and their upper and lower 95% confidence intervals for the best approximating model for predicting the catch of Delta Smelt at Chipps Island. Random effects are expressed as standard deviations. All water quality parameters contain an additional quadratic term (x2).

Parameter	*Estimate*	*SE*	*95% Lower CI*	*95% Upper CI*
Best Fitting Model (w = 0.86)				
Fixed effects				
Intercept	0.12	2.027	-3.851	4.093
Position of Low Salinity Zone	-0.48	0.075	-0.627	-0.332
Position of Low Salinity Zone²	-0.32	0.048	-0.410	-0.221
Temperature	0.08	0.061	-0.040	0.201
Temperature²	0.30	0.036	0.231	0.374
Secchi	0.11	0.035	0.046	0.184
Secchi²	-0.10	0.023	-0.145	-0.057
Ebbing Tide	-2.71	2.024	-6.675	1.259
Flooding Tide	-2.28	2.024	-6.245	1.690
Slack High Tide	-0.24	0.164	-0.559	0.084
Tidal Current	1.44	1.276	-1.056	3.944
Tidal Current*Ebbing Tide	-1.59	1.276	-4.086	0.915
Tidal Current*Flooding Tide	-1.34	1.276	-3.839	1.164
Rainy Weather	0.18	0.101	-0.021	0.376
Other Non-clear Weather	-0.02	0.047	-0.115	0.068
Wind	-0.12	0.024	-0.167	-0.073
Annual Fall Midwater Trawl Index	0.20	0.083	0.038	0.364
North Channel Position	0.28	0.042	0.202	0.366
South Channel Position	-0.38	0.050	-0.474	-0.279
Trawl Moving with Net Tidal Current	0.06	0.050	-0.042	0.153
Volume Sampled	0.13	0.017	0.092	0.160
Random effects				
Year*Month	1.07	1.035		
Samples	0.97	0.985		

Parameter estimates indicated that, on average, the catch of Delta Smelt increased by 0.2 individuals per trawl for every 125 increase in the annual Fall Midwater Trawl Delta Smelt Index (Table 2). The catch of both Delta Smelt and juvenile Chinook Salmon increased as the volume of water sampled increased. However, the relation between catch and most methodological and all environmental variables varied between the two species (Tables 2 and 3). For example, juvenile Chinook Salmon catch peaked differently within channel positions and among months relative to Delta Smelt catch (Figure 2). In addition, the peak catch of juvenile Chinook Salmon and Delta Smelt differed considerably among varying water quality conditions (Figure 3).

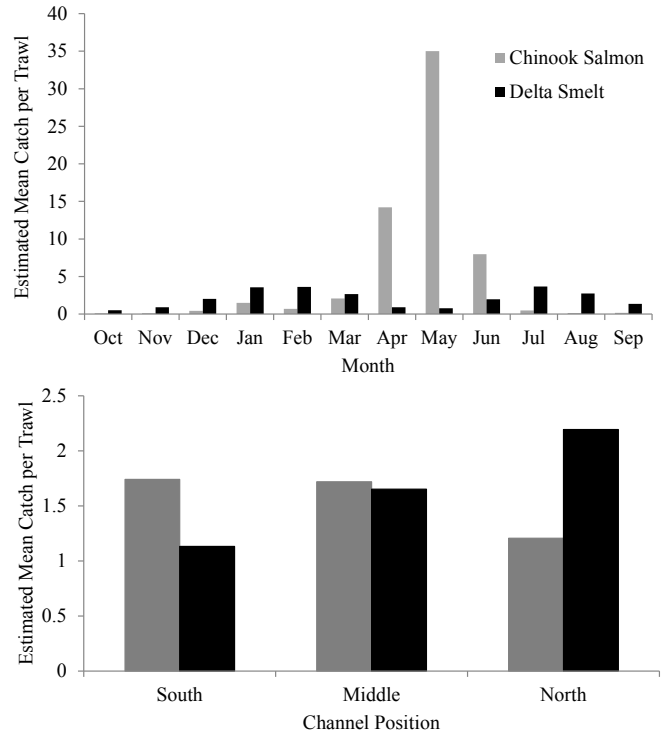

Figure 2 Estimated mean catch of juvenile Chinook Salmon (grey) and Delta Smelt (black) per trawl at Chipps Island among (A) months and (B) channel positions under average environmental conditions from July 2001 to December 2011. Estimates assumed the trawl occurred in the middle of the channel during a clear day, low tide, and sampled 20,000m3 of water.

Discussion

We observed that the catch of both Delta Smelt and juvenile Chinook Salmon at Chipps Island was influenced by a variety of variables including water quality, tide, abundance, and surface trawl methodology. In addition, modeling results indicated that the effect of all environmental variables on catch varied considerably among the species. As a result, our study provides evidence that the targeting or avoidance of particular environmental conditions to minimize the incidental take of Delta Smelt while sampling would likely affect the catch and therefore the integrity of the long-term monitoring of juvenile Chinook Salmon at Chipps Island.

Table 3 Parameter estimates, standard error, and their upper and lower 95% confidence intervals for the best approximating model for predicting the catch of juvenile Chinook Salmon at Chipps Island. Random effects are expressed as standard deviations. All water quality parameters contain an additional quadratic term (x2).

Parameter	*Estimate*	*SE*	*95% Lower CI*	*95% Upper CI*
Best Fitting Model (w = 0.93)				
Fixed effects				
Intercept	0.20	1.539	-2.814	3.221
Position of Low Salinity Zone	-1.93	0.069	-2.062	-1.793
Position of Low Salinity Zone²	-0.13	0.035	-0.193	-0.058
Temperature	0.30	0.043	0.217	0.385
Temperature²	-0.35	0.030	-0.404	-0.287
Secchi	0.24	0.023	0.192	0.283
Secchi²	-0.04	0.014	-0.070	-0.014
Ebbing Tide	-0.78	1.533	-3.784	2.225
Flooding Tide	-1.03	1.533	-4.039	1.971
Slack High Tide	-0.28	0.102	-0.481	-0.080
Tidal Current	0.44	0.964	-1.452	2.326
Tidal Current*Ebbing Tide	-0.46	0.964	-2.351	1.428
Tidal Current*Flooding Tide	-0.55	0.964	-2.443	1.338
North Channel Position	-0.35	0.027	-0.407	-0.302
South Channel Position	0.01	0.026	-0.040	0.064
Trawl Moving with Net Tidal Current	-0.03	0.031	-0.085	0.035
Volume Sampled	0.13	0.013	0.101	0.152
Random effects				
Year*Month	2.25	1.499		
Samples	0.98	0.988		

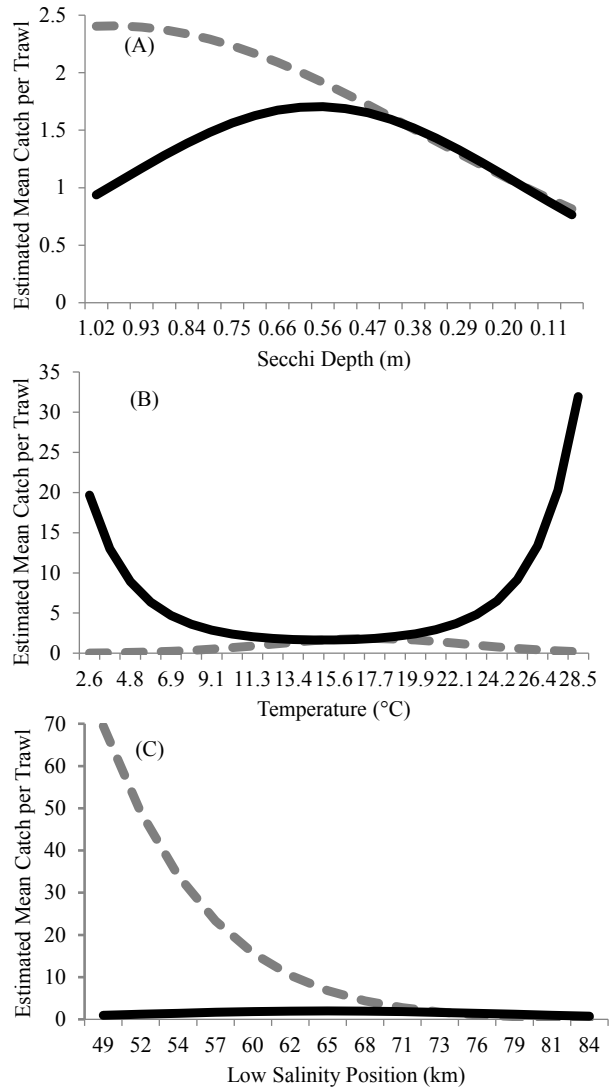

Figure 3 Estimated mean catch of juvenile Chinook Salmon (dashed grey) and Delta Smelt (solid black) per surface trawl at Chipps Island for varying (A) Secchi depths, (B) temperatures, and (C) low salinity positions under average environmental conditions. Estimates assumed the trawl occurred during a clear day, low tide, and sampled 20,000m³ of water.

We observed that the catch of Delta Smelt was highest from July through September and December through March. The two peaks potentially reflect the migration patterns of different life stages of Delta Smelt—the summer months displaying the migration of the new year class of juveniles into estuarine rearing habitat (Dege and Brown 2004), and the winter months displaying the spawning migration of adults upstream (Grimaldo et al. 2009; Sommer et al. 2011). Conversely, the catch of juvenile Chinook Salmon occurred primarily from March through June. Lastly, the catch of both species were positively influenced by increased sample volumes. Therefore, we recommend that the incidental take of Delta Smelt be reduced by decreasing the number or duration of tows during seasons when juvenile Chinook Salmon are not captured, or are captured in low numbers (i.e., summer and fall). The reduction in volume sampled on the long-term trends of juvenile Chinook Salmon indices is likely negligible. The DJFMP has historically reported the relative abundance of juvenile Chinook Salmon as a density metric where catch is standardized by volume to account for any variability in volume sampled (Speegle et

al. 2013). Because we observed patterns in catch among trawl lanes for each of the species, there should be equal reduction of trawls among each of the trawl lanes to ensure historical comparability of indices. Otherwise, inference from the Chipps Island monitoring data will need to be obtained from more complicated statistical approaches (e.g., mixed effects modeling; Zuur et al. 2009) to account for any spatial dependency while evaluating trends in Chinook Salmon catch over time.

It is well recognized that the incidental take of individuals can negatively influence the viability and probability of recovery of ESA listed species (McGowan and Ryan 2009). Our analysis provided evidence that the incidental take of Delta Smelt at Chipps Island is a function of their indexed abundance. Therefore, the DJFMP should further reduce the number or duration of trawl samples collected at Chipps Island particularly during years when Delta Smelt abundance indices are relatively low to further minimize the risk of jeopardy and promote recovery.

As the imperilment of our native fish species continues to rise in the San Francisco Estuary and its watershed, the continuity, comparability, and ecological soundness of long-term fish monitoring datasets will become more crucial. Our study provides an approach that natural resource managers and monitoring programs can employ when faced with the need to modify long-term fish monitoring sample designs to limit the incidental take of non-targeted ESA listed fishes. Future studies should evaluate the efficiency of the Chipps Island trawl to decipher the relative importance and effect of environmental variables on fish occupancy vs. detection.

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Fish identification accuracy and implications to monitoring within the San Francisco Estuary, CA

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Introduction

Fish monitoring programs rely on the collection, identification, and counting of individual fish over time to determine changes in fish distribution and abundance. Therefore, the usefulness of the data collected can be greatly affected by misidentification (Elphick 2008; Fitzpatrick et al. 2009). For example, monitoring data containing identification errors may confound true fish distributional patterns and suggest false ecological patterns (Shea et al. 2011). In general, fish monitoring programs operating within the San Francisco Estuary and its watershed have assumed 100% identification accuracy when making inferences using their data and informing management decisions. The accuracy of fish identification in the field can be influenced by the level of distinction of visible morphological traits among species (e.g., size, shape, color, etc.; Moyle 2002) coupled with observer bias (e.g., experience level; Fitzpatrick et al. 2009; Shea et al. 2011). Therefore, the assumption of accurate identification of all small juvenile fishes in the field within the San Francisco Estuary may be unwarranted.

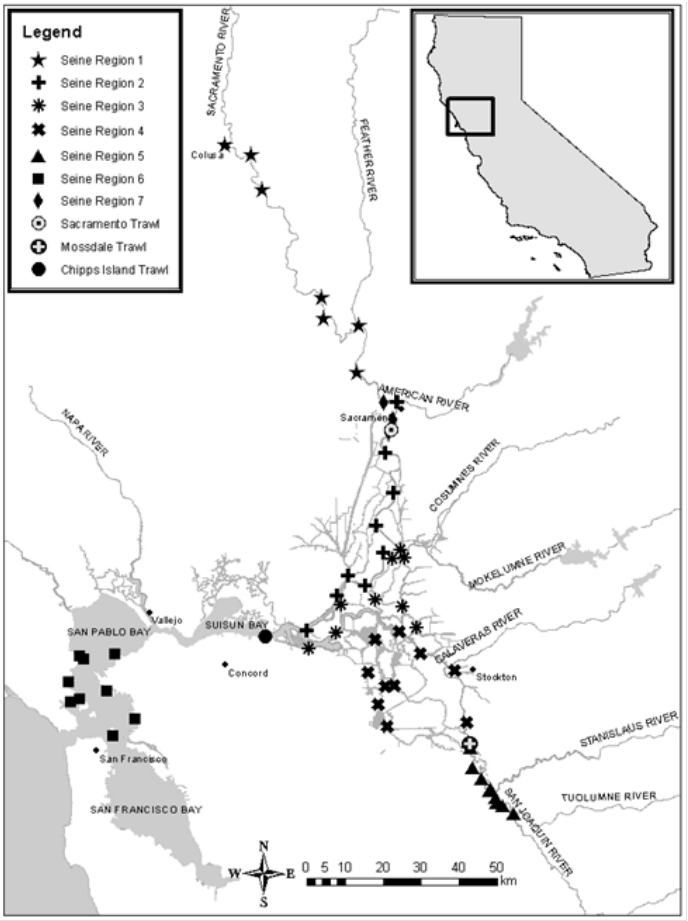
The US Fish and Wildlife Service's Delta Juvenile Fish Monitoring Program (DJFMP) has conducted sampling within the San Francisco Estuary since 1976 to primarily inform water operation decisions and assess the status of juvenile salmonids over time. Recognizing that misidentification could reduce the integrity of the DJFMP data and its inferences, numerous control measures have been implemented throughout the program's history to minimize bias from field identification errors. For example, all unidentifiable fish were brought back to the laboratory for positive identification, a reference collection was established, and informal training was given to inexperienced field observers. A minimum size criteria also was established for proper fish identification (e.g., fork length > 25 mm) after recognizing the difficulty of accurately identifying larval fish within the field. Furthermore, an extensive fish identification training program was created in response to a program review in 2000 that expressed concern regarding the identification accuracy of small non-salmonid resident fishes (Brandes et al. 2000). The fish identification training program was composed of one full-time fish identification biologist tasked with creating a formal training curriculum within the field and laboratory, developing accurate and effective fish identification keys, establishing a voucher collection, expanding the reference collection, and haphazard field identification validation (P. Cadrett and P. Brandes, US Fish and Wildlife Service, personal communication). Although the control measures implemented by the DJFMP have undoubtedly improved the identification accuracy of juvenile fishes, no robust assessment has been made to quantify fish identification accuracy. Therefore, the objectives of this study were to (1) estimate the identification accuracy rates among DJFMP observers and fish species and (2) assess the factors that may influence an observer's identification accuracy of fishes in the San Francisco Estuary.

Methods

Our evaluation of fish identification accuracy occurred under controlled conditions modified from Shea et al. (2011). A total of four fish identification exams were conducted at the Stockton Fish and Wildlife Office. The examinations occurred in the spring (March 8), summer (June 18) and fall (November 29) of 2012 and in the summer (May 22) of 2013. Prior to each exam, fish specimens greater than 25 mm in fork length were haphazardly collected at DJFMP monitoring locations to obtain representative samples among species and sizes throughout the San Francisco Estuary (Figure 1). Each specimen collected was immediately preserved by freezing in an attempt to maintain natural appearance (i.e., color). Fishes listed under the state or federal Endangered Species acts were not collected for this study, but specimens were obtained by preserving indirect mortalities from regular DJFMP monitoring.

Approximately 20-40 fish specimens were collected or obtained for each exam. Test specimens were selected after being defrosted during the morning of each exam.

Figure 1 The location of the DJFMP monitoring locations where test specimens were collected within the San Francisco Estuary, CA.



The staff member(s) who selected test specimens did not take the exam, but functioned as the exam's proctor. Test specimens that were damaged during the fish identification exams as a result of extensive handling were replaced, if possible, by another individual of the same species, size (+/- 5mm in fork length), and condition (e.g., color) during each exam. In general, three fish biologists (considered local experts) from the California Department of Fish and Wildlife verified the identification of each test specimen and their possible replacements. If the experts and the exam's proctor did not come to a consensus regarding the identification of a test specimen, the specimen was eliminated from the exam. Test specimens were randomly assigned to test stations numbered in sequential order. Both the species and fork length (measured to the nearest mm using a measuring board) were documented for each test specimen at each station.

During the spring and summer exams of 2012, observers were allowed to move freely among each exam station to identify test specimens with no time constraints. Conversely, during the fall of 2012 and summer of 2013, observers were given at least two minutes and up to three minutes to identify each test specimen before moving, in sequential order, to another station. The additional structure added to the latter two exams was an attempt to limit confounding results from observers having unequal time to identify test specimens. For all exams, observers were limited to individuals that had recently (e.g., within the last 60 days) identified fish on the behalf of the DJFMP either as an employee or volunteer. To simulate conditions within the field, observers were allowed to identify test specimens using their field keys. To prevent nomenclature errors during the exams, observers were asked to identify test specimens by common names and were provided sheets or keys that contain common and scientific names for all fishes occurring within the San Francisco Estuary. Observers also recorded their experience (in months) identifying juvenile or adult fishes within California's Central Valley during each exam.

Statistical Analysis

We evaluated fish identification accuracy using general linear hierarchical regression models assuming a logit distribution (Williams et al. 2002; Royle and Dorazio 2008; Shea et al. 2011). Individual test specimen identifications served as the response variable and were coded as 1 when the test specimen was identified correctly and as 0 when the test specimen was misidentified. We hypothesized that fish identification accuracy can be affected by observers, species, fork length, morphological characteristic groups (i.e., shad-like, bass-like, sculpin-like, minnow-like, pupfish-like, catfish-like, salmon-like; Table 1), Central Valley fish identification experience, and interactions between morphological characteristic groups and fork length. Secondarily, we were interested in evaluating the effect of different time limitations (i.e., test design) on test specimen identification among our four exams. To minimize the number of parameters, the effect of observers and species were modeled as randomly varying intercepts (Zuur et al. 2009). Prior to model fitting, we calculated Pearson correlation coefficients for all pairs of potential predictor variables and eliminated highly correlated variables ($r^2 > 0.5$) from our analysis to avoid

Table 1 Species, number of test observations, fork lengths, and morphological groupings of test specimens used evaluate the probability of identifying fish accurately. Species are listed alphabetically by morphological groups.

Species	*Observations*	*Fork Length (mm)*		*Morphological Group*
		Mean (SD)	*Range*	
Bluegill (*Lepomis macrochirus*)	48	46.4 (6.5)	35 - 52	Bass-like
Black Crappie (*Pomoxis nigromaculatus*)	29	97.8 (9.7)	88 - 107	Bass-like
Dwarf Surfperch (*Micrometrus minimus*)	35	86.4 (7.0)	78 - 92	Bass-like
Green Sunfish (*Lepomis cyanellus*)	42	60.1 (21.9)	27 - 82	Bass-like
Largemouth Bass (*Micropterus salmoides*)	72	74.7 (24.5)	38 - 96	Bass-like
Redear Sunfish (*Lepomis microlophus*)	76	61.1 (22.6)	42 - 102	Bass-like
Spotted Bass (*Micropterus punctulatus*)	35	71.2 (31.3)	46 - 109	Bass-like
Shiner Perch (*Cymatogaster aggregata*)	21	95 (0)	95 - 95	Bass-like
Striped Bass (*Morone saxatilis*)	21	36 (0)	36 - 36	Bass-like
Tule Perch (*Hysterocarpus traski*)	62	78.7 (44.4)	36 - 151	Bass-like
Warmouth (*Lepomis gulosus*)	35	46.4 (4.5)	41 - 50	Bass-like
White Crappie (*Pomoxis annularis*)	33	129.2 (26.4)	107 - 166	Bass-like
Brown Bullhead (*Ameiurus nebulosus*)	37	147.6 (49.2)	92 - 190	Catfish-like
Channel Catfish (*Ictalurus punctatus*)	27	59.7 (4.5)	56 - 65	Catfish-like
Common Carp (*Cyprinus carpio*)	12	126.4 (11.8)	123 - 164	Minnow-like
Fathead Minnow (*Pimephales promelas*)	62	48.3 (3.6)	46 - 56	Minnow-like
Goldfish (*Carassius auratus*)	16	196 (0)	196 - 196	Minnow-like
Golden Shiner (*Notemigonus crysoleucas*)	61	58.8 (18.2)	35 - 80	Minnow-like
Hitch (*Lavinia exilicauda*)	60	91.5 (24.4)	63 - 121	Minnow-like
Hardhead (*Mylopharodon conocephalus*)	70	99.0 (34.1)	53 - 149	Minnow-like
Red Shiner (*Cyprinella lutrensis*)	62	43.2 (4.1)	40 - 50	Minnow-like
Sacramento Pikeminnow (*Ptychocheilus grandes*)	78	83.1 (20.2)	45 - 102	Minnow-like
Sacramento Sucker (*Catostomus occidentalis*)	48	39.5 (3.2)	36 - 43	Minnow-like
Sacramento Blackfish (*Orthodon microlepidotus*)	37	146.7 (23.6)	120 - 167	Minnow-like
Splittail (*Pogonichthys macrolepidotus*)	78	81.6 (30.7)	48 - 134	Minnow-like
Western Mosquitofish (*Gambusia affinis*)	21	27 (0)	27 - 27	Pupfish-like
Rainwater Killifish (*Lucania parva*)	61	32.8 (3.3)	30 - 39	Pupfish-like
Chinook Salmon (*Oncorhynchus tshawytscha*)	97	80.8 (45.2)	50 - 180	Salmon-like
Steelhead Trout (*Oncorhynchus mykiss*)	99	84.0 (46.9)	46 - 175	Salmon-like
Bigscale Logperch (*Percina macrolepida*)	11	76 (0)	76 - 76	Sculpin-like
Prickly Sculpin (*Cottus asper*)	57	59.3 (15.3)	40 - 76	Sculpin-like

Shokihaze Goby (*Tridentiger barbatus*)	16	96 (0)	96 - 96	Sculpin-like
Shimofuri Goby (*Tridentiger bifasciatus*)	37	71.9 (4.5)	68 - 77	Sculpin-like
Yellowfin Goby (*Acanthogobius flavimanus*)	51	83.0 (37.6)	43 - 131	Sculpin-like
American Shad (*Alosa sapidissima*)	30	95.9 (6.6)	89 - 102	Shad-like
Delta Smelt (*Hypomesus transpacificus*)	76	69.1 (11.2)	56 - 85	Shad-like
Longfin Smelt (*Spirinchus thaleichthys*)	106	87.9 (13.2)	74 - 111	Shad-like
Inland Silverside (*Menidia beryllina*)	48	77.8 (7.9)	66 - 87	Shad-like
Pacific Herring (*Clupea pallasi*)	61	89.0 (43.7)	55 - 164	Shad-like
Topsmelt (*Atherinops affinis*)	16	74 (0)	74 - 74	Shad-like
Wakasagi (*Hypomesus nipponensis*)	55	90.3 (17.7)	69 - 115	Shad-like

multicollinearity (Dormann et al. 2013). To minimize the effects of heterogeneity, we transformed the Central Valley fish identification experience data using a natural logarithm. In addition, we standardized all continuous data with a mean of zero and a standard deviation of one to facilitate model fitting and simplify the interpretation of parameter estimates. Lastly, we created binary indicator variables for exams with a time constraint and morphological characteristic groups where the salmon-like group functioned as a baseline.

We quantified the relative importance of each predictor variable by comparing the relative fit of candidate models that represented all possible combinations of our a priori hypotheses using an information theoretic approach (Burnham and Anderson 2002). The relative fit of candidate models for fish identification was determined by calculating ratios of Akaike weights (w_i; Burnham and Anderson 2002) using each model's Akaike Information Criteria with the small sample bias adjustment (AICc; Hurvich and Tsai 1989). To account for model selection uncertainty, we constructed a confidence set of candidate models that included models with Akaike weights that are within 12% of the best approximating candidate model's Akaike weight (Royall 1997). To allow for ease of interpretation, an odds ratio (OR) was estimated for each fixed effect parameter in the composite models (Hosmer and Lemeshow 2000). An OR adjusts for the logistic element and is calculated by taking the exponent of the parameter estimate. An OR can range between zero and infinity. An OR that is less than one demonstrates that the response variable is less likely to occur and an OR that is greater than one demonstrates that the response variable is more likely to occur. The precision of fixed effect parameter estimates was assessed by calculating 95% credible intervals that are analogous to 95% confidence intervals (Congdon 2001). Credible intervals that contained one were considered imprecise. We based all inferences on the confidence set of models and only on precise parameter estimates.

Results

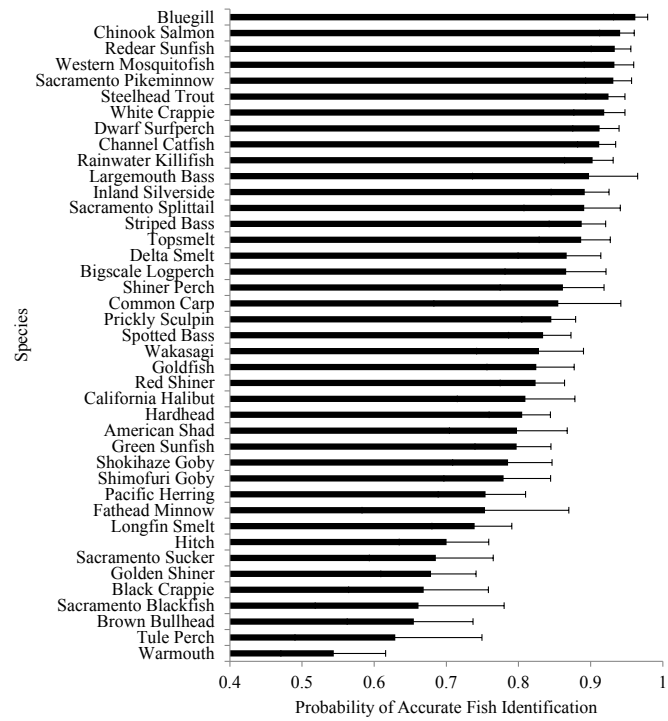
A total of 1,999 observations were recorded during our fish identification examinations. Data represented 41 species among 16 families (Table 1). Fish specimens represented a wide range of fork lengths (= 77.1 mm, SD = 36.6, range = 27 – 196 mm). A total of 29 observers participated in at least one of the fish identification examinations and observer Central Valley fish identification experience varied considerably (= 57.1 months, SD =50.6, range = 0 – 180 months).

Three candidate models were included in our confidence set of models (Table 2). The most plausible model for predicting the probability of accurately identifying fish was the model using fork length, Central Valley experience, species, and observers as predictors (Table 2). Akaike weights indicated that this model was 2.36 times more supported than the second best candidate model. The second best candidate model was identical to the best model but also contained an indicator variable for exams with a time constraint. The parameter estimate for the variable indicating exams with a time constraint was imprecise and was excluded from inference (Table 2). The best fitting model was 7.32 times more supported than the third best candidate model that was identical to the best

Table 2 Parameter estimates (standard error in parentheses), odds ratios (OR), and upper and lower 95% credible intervals for the confidence set of candidate models predicting the probability of identifying fish accurately. Random effects are expressed as standard deviations.

Parameter	*Estimate (SE)*	*Odds Ratio*	*95% Lower Credible Interval*	*95% Upper Credible Interval*
Best Fitting Model (w = 0.598)				
Fixed effects				
Intercept	1.68 (0.248)	5.369	3.3	8.736
Fork Length	0.23 (0.091)	1.26	1.055	1.505
CV Experience	0.7 (0.151)	2.01	1.496	2.701
Random effects				
Species	0.71 (0.842)			
Observers	0.99 (0.997)			
Second Best Fitting Model (w = 0.253)				
Fixed effects				
Intercept	1.64 (0.258)	5.155	3.109	8.546
Fork Length	0.23 (0.09)	1.259	1.055	1.503
CV Experience	0.69 (0.152)	1.986	1.474	2.674
Exam Time Constraint	0.11 (0.199)	1.114	0.753	1.647
Random effects				
Species	0.71 (0.84)			
Observers	0.98 (0.99)			
Third Best Fitting Model (w = 0.082)				
Fixed effects				
Intercept	1.67 (0.239)	5.336	3.342	8.519
CV Experience	0.7 (0.15)	2.008	1.497	2.693
Random effects				
Species	0.55 (0.741)			
Observers	0.98 (0.989)			

Figure 2 Estimated mean probability of accurate fish identification (95% Credible Intervals) among observers. These estimates assume that the fish has a fork length of 77mm and the observer possesses 57 months of Central Valley fish identification experience.



model but did not contain fork length as a predictor. Because parameter estimates were similar among the confidence set of candidate models, we based all inferences on the best approximating model.

The best fitting candidate model indicated that, on average, fish identification accuracy was 84.3% among our test specimens and observers. However, parameter estimates suggested that accurate fish identification varied considerably among observers and species (Table 2; Figure 2). For example, fish identification accuracy was, on average, greater than 92% for salmonids and less than 60% for Warmouth (*Lepomis gulosus*; Figure 2). Model estimates also indicated that fish identification accuracy increased as an observer's experience identifying Central Valley fishes increased and as fishes possessed larger fork lengths (Table 2; Figure 3). For every 36 mm increase in fork length, accurate fish identification was, on average, 1.26 times more likely (Table 2).

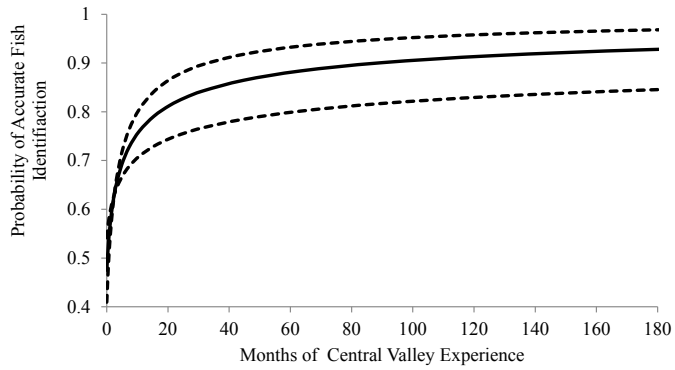
Discussion

Fish identification accuracy is an important aspect of any fish monitoring program (Elphick 2008). Misidentification can lead to inaccurate estimates of abundance or distribution and misinform vital management decisions (Shea et al. 2011). Tyre et al. (2003) demonstrated that false negative error rates (i.e., misidentification) as low as 20% can substantially bias occupancy estimation. Our modeling results suggested that average fish identification accuracy was 84% and varied considerably among species, sizes, and observers. We demonstrated that observer identification accuracy increased slightly with increasing fish size. As fish become larger, their definitive characteristics may become more apparent to observers (Moyle 2002). Our model estimates also indicated

that, on average, observers with more than 17 months of Central Valley fish identification experience generally had greater than 80% fish identification accuracy. However, considerable variability remained among observers after accounting for identification experience. Shea et al. (2011) reported similar findings between observer experience and the identification accuracy of freshwater mussels among observers. We surmise that other observer-based factors, in addition to Central Valley fish identification experience, influenced an observer's ability to accurately identify fish. Therefore, the assumption that highly experienced observers are adequate fish identifiers (i.e., accuracy > 80%) remains unsubstantiated and requires further assessment. A total of 26 species in our study, including salmonids, Sacramento Splittail (*Pogonichthys macrolepidotus*), and Delta Smelt (*Hypomesus transpacificus*), had an average identification accuracy greater than 80% among test observers. These species represent most fishes of management concern within the San Francisco Estuary and all the species commonly reported on by the DJFMP (Speegle et al. 2013). As a result, there is evidence that the fish metrics produced and reported by the DJFMP are not substantially biased from species-specific misidentification.

Although we presume that our results are representative of recent DJFMP field identification rates, there are several potential sources of uncertainty that may have biased our evaluation of fish identification accuracy. In general, our results were derived from laboratory examination, which may have caused stress among observers and thereby influenced their ability to accurately identify fish. We also did not include any spatial or temporal catch information for any of the test specimens, which can be used as supplemental identification criteria in the field (Moyle 2002). In addition, the use of preserved specimens rather than live fish may further confound our examination results relative to regular field observations. The DJFMP also allows and encourages its observers to bring specimens to the laboratory when they cannot be confidently and accurately identified in the field. However, this was not an option provided to observers who participated in our examinations. Furthermore, observers in the field often consult with one another when identifying ambiguous specimens whereas consultation between observers during our examinations was not allowed. Lastly, our analysis focused solely on assessing the implications of fish misidentification as it related to false negative errors. We did not assess or evaluate the implications of the observer's false positive error rates, which can further distort abundance or distributional trends (Royle and Link 2006). Future investigations should account for or address these potential sources of uncertainty when evaluating observer identification accuracy.

Figure 3 Estimated mean probability of accurate fish identification (95% Credible Intervals; dashed lines) versus the observer's Central Valley fish identification experience. These estimates assumed the fish has a fork length of 77mm.



It is evident that there is a need to continue the assessment and improvement of our fish identification accuracy within the DJFMP. Overall, we recommend that this study be expanded to include more observers and species across detectable size distributions, collect and assess the effect of additional observer factors (e.g., education, level of professional training, frequency of identifying fish in the field within a particular timeframe), evaluate fish identification accuracy in the field to substantiate our results or determine the effect of our laboratory examinations, and evaluate the prevalence and implications of both false presence and false absence errors among fish monitoring observers. Inferences drawn from these investigations can be used to inform hiring, training, work assignments, and analytical practices among fish monitoring programs operating within the San Francisco Estuary. Continuing our efforts to reduce and understand our misidentification of San Francisco estuarine fishes should reduce the biases associated with the data we collect and in turn improve our effectiveness and efficiency in informing management decisions throughout the system.

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Bias in Estimated Annual Harvest Rates for White Sturgeon of the San Francisco Estuary

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Introduction

Estimated annual White Sturgeon, *Acipenser transmontanus*, harvest rate is one of many metrics produced by the California Department of Fish and Wildlife (CDFW) sturgeon population study (the study). Defined as the fraction of a demographic (e.g., fish in a given size range) harvested in a given year, estimated annual harvest rates have been reported or alluded to at intervals since 1959 (Chadwick 1959; Skinner 1962; Miller 1972; Kohlhorst 1979; Kohlhorst 1980; Kohlhorst and others 1991; Schaffter and Kohlhorst 1999; Kohlhorst and Cech 2001; DuBois and Gingras 2011; DuBois and others 2012) and have been used to monitor the impact of fishing regulations, to estimate natural mortality rate, and to estimate abundance (DuBois and Gingras 2011). Calculated from tagging data and information provided voluntarily (for example, by mail) by sport anglers on their capture of tagged fish, the rates can be biased and — because White Sturgeon is (like most sturgeons are) particularly sensitive to harvest — it is important to understand the direction and magnitude of any bias.

The estimated annual harvest rates reported to date have not accounted for possible biases due to mixing of tagged fish with un-tagged fish, tag shedding, delayed mortality attributable to tagging, or angler willingness to voluntarily contact CDFW about capture of tagged fish. Of those issues, only the level of mixing might cause estimated rates to be biased high and we plan to look into the potential for bias attributable to the level of mixing. Miller

(1972) used a double-tagging study to assess tag shedding and characterized shedding as "negligible." Delayed mortality attributable to tagging has not been assessed, but survival of tagged fish is a top priority of the study and we suspect mortality is quite low. Angler willingness to voluntarily contact CDFW about capture of tagged fish (hereinafter we will call that "angler willingness") stands out as a potentially-substantial downward bias.

Angler willingness was first addressed in 1967 by placing a prominently-labeled $5 reward tag on each fish (Miller 1972) and was addressed in the 1980s by increasing the reward value to $20. Starting in 1998, angler willingness was further addressed — and the groundwork for an assessment was laid — by placing a $20, $50, or $100 reward tag on each fish. In an effort (in part) to better assess angler willingness, Sturgeon Fishing Report Cards (Sturgeon Cards) since 2010 have required anglers to record the harvest or release of fish tagged by the study. In the present investigation, we use reward value and Sturgeon Card data to briefly investigate angler willingness to voluntarily contact CDFW about capture of tagged fish in 1998, 2001, 2002, and 2005-2012.

Investigation

We hypothesized that anglers returning Sturgeon Cards since 2010 would report having caught approximately equal proportions of fish bearing $20, $50, or $100 reward tags, because (1) the study released approximately the same number of tags with each of the 3 rewards, (2) anglers who returned Sturgeon Cards have noted the annual capture of many White Sturgeon (approximately 3000-5000), and (3) documenting capture of tagged fish is required for Sturgeon Card holders. We also hypothesized that anglers returning Sturgeon Cards would report a substantially different proportion of fish bearing $20, $50, or $100 reward tags than would anglers reporting voluntarily, and that anglers would report voluntarily in rough proportion to reward value. To look into (if not test) those hypotheses, we did a few simple summaries of reported capture of tagged fish.

The proportion of tags reported by Sturgeon Cards during 2010-2012 was 45% at $20, 21% at $50, and 34% at $100, which is not equal proportions but is substantially different than the proportion of tags — 23% at $20, 23% at $50, and 54% at $100 — reported voluntarily during the same period. Having seen that the 2010-2012 proportions were a bit "noisy," we looked at the much larger 1998-2012 dataset on tags reported voluntarily and found the proportions to be 22% at $20, 33% at $50, and 45% at $100. These summaries demonstrate that reward value affected the willingness of many anglers to voluntarily contact CDFW about capture of tagged fish and suggest that estimated annual harvest rates calculated without consideration of reward value were likely biased low.

To get a sense of the magnitude of bias attributable to angler willingness, we estimated annual harvest rates using the below formula and the following permutations of data: (1) Recaptured fish without regard to the fish's reported fate or to reward value, which is the study's long-time "conventional" algorithm, (2) recaptured fish reported (voluntarily or by Sturgeon Card) as kept, but without regard to reward value, and (3) recaptured fish reported (voluntarily or by Sturgeon Card) as kept by reward value.

$$\mu_Y = \frac{\left(\sum[Tags\ Returned - \sum Tags\ Returned_{ra})]_{fy} + \left(\sum Tags\ Returned_c\right)_{fy}}{\sum Tags\ Released}$$

Where:
μ = Harvest Rate
Y = Year
fy = First-year (that is, tag returned within 365 days of being released)
ra = Fish reported by angler as being released alive
c = Tag return reported on Sturgeon Card only

We found that harvest rates calculated using only $100 tags were almost always substantially higher than harvest rates calculated otherwise and there was an increasing trend in harvest rate over time (Figure 1). These results strongly suggest that harvest rates calculated from $100 tags were closest to accurate but were likely still biased somewhat low.

Discussion

This brief investigation describes and (to a degree) quantifies a downward bias in estimated annual White Sturgeon harvest rates attributable to a lack of angler willingness to voluntarily contact CDFW about capture of tagged fish. As follows, this finding suggests that it is important to improve and continue assessing angler willingness, prompts a reinterpretation of the relative impact of fishing mortality on the population and fishery, and suggests that management actions may have contributed to an increase in annual harvest rate.

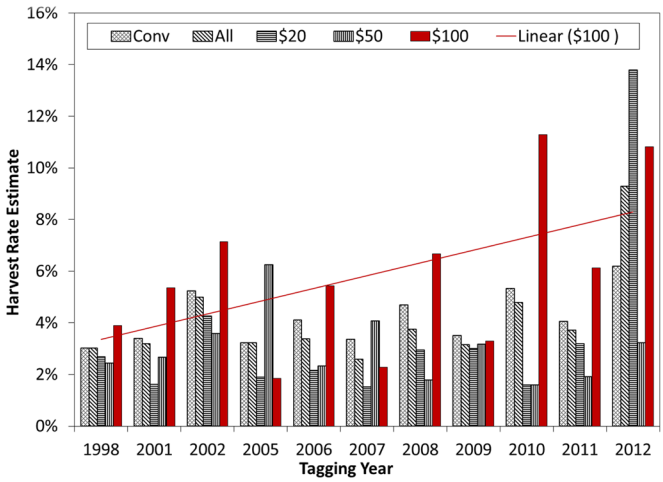

Figure 1 Estimated annual harvest rates (1998, 2001, 2002, and 2005-2012) for San Francisco Estuary White Sturgeon from the following permutations of data: (1) recaptured fish without regard to the fish's reported fate or tag reward value (Conv), (2) recaptured fish reported (voluntarily or by Report Card) as kept but without regard to tag reward value (All), and (3) recaptured fish reported (voluntarily or by Report Card) as kept by tag reward value ($20, $50, $100). Solid line indicates linear regression on harvest rates estimated using only data on $100 tags.

The downward bias is attributable to lack of angler willingness and is influenced by the reward paid by CDFW for contacting us. To improve and further assess angler willingness, we plan additional outreach and will modify aspects of study protocol. Additional outreach will include promoting the sturgeon population study through more-frequent distribution of informational fliers, use of social media and Press Releases, and (if possible) improved use of CDFW's web site. Study protocol modifications will include a minor reformatting of tags and the application of tags with rewards of $50, $100, and $150. We are also considering regulatory approaches to increase voluntary reporting of captured tagged fish.

Having seen from the present investigation that estimated annual harvest rates calculated without regard to reward value tended to be biased substantially low from 1998-2012 due to lack of angler willingness, we suspect that the rates calculated prior to that period — when no information on angler willingness was available — were also biased low. If so, harvest contributed more than previously thought to observed declines in the White Sturgeon fishery and population. Estimated annual harvest rates for fish ≥ 102 centimeters total length were approximately 7% in the late 1960s (Miller 1972) and increased to 11.5% in the 1980s (Kohlhorst and others 1991). Given that annual total mortality rates have rarely been more than 20% and have frequently been much less than that (Miller 1972; Kohlhorst 1979; Kohlhorst 1980; Kohlhorst and others 1991; Schaffter and Kohlhorst 1999; DuBois and others 2012), it is likely that harvest has typically been the largest part of total mortality.

White Sturgeon catch-per-unit-effort (CPUE) declined substantially from 1964 to 1974 (Kohlhorst 1980), while White Sturgeon CPUE (DuBois and others 2012) and estimated abundance collapsed through the 1980s (Schaffter and Kohlhorst 1999; DuBois and others 2012). In response to the 1980s collapses, in 1990 the California Fish and Game Commission (Commission) increased the minimum size limit and established a first-ever maximum size limit but made no change to the bag limit of 1 fish per day. The Commission also implemented a suite of fishing regulations in 2007 — including a 3-fish annual bag limit, a reduction in the maximum size limit, and establishment of the Sturgeon Card — that was expected to improve fishing for and the resiliency of White Sturgeon over time as well as provide useful information on the population and patterns in sturgeon fishing. White Sturgeon fishing effort and harvest from Commercial Passenger Fishing Vessels the decade before and the decade after implementation of the maximum size limit were similar, and fishing interest since 2007 has been very high (e.g., 41,000-112,000 Sturgeon Cards issued annually), which suggests that implementation of and later reduction in the maximum size limit focused substantial fishing effort on substantially fewer White Sturgeon cohorts.

We suspect the recent trend of increased estimated annual harvest rates was due the combination of: (1) The economic downturn, which increased angler willingness to contact CDFW to receive rewards for recapturing tagged fish, (2) additional outreach by the CDFW in support of the 2007-present fishing regulations, which increased angler interest about contacting CDFW to receive rewards for the capture of tagged fish, and (3) the 2007-present reduction in the maximum size of White Sturgeon that may be harvested legally, which focused substantial fishing effort on a narrower demographic.

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