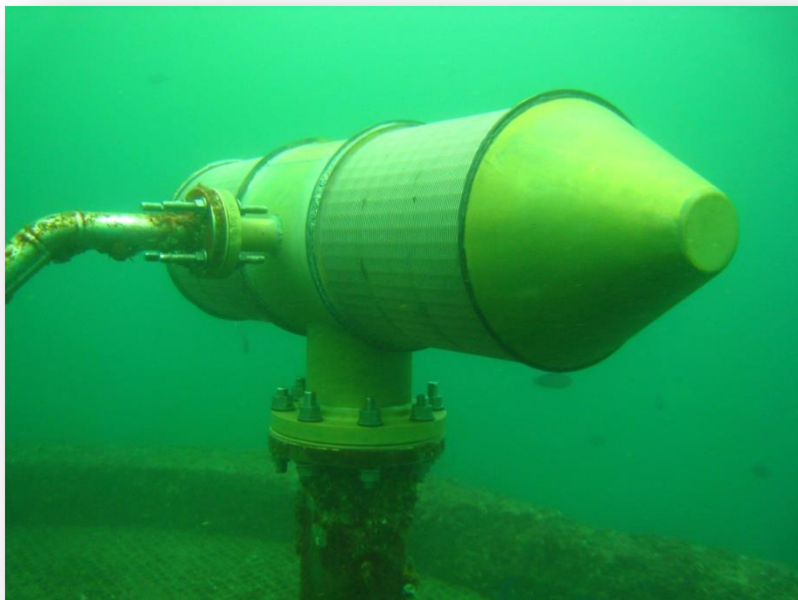
West Basin Municipal Water District
Desalination Demonstration Facility

Intake Effects Assessment Report



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Executive Summary

This report presents an assessment of potential impacts to marine life due to water withdrawals associated with the operation of open ocean intakes for a demonstration ocean water desalination facility constructed and operated by the West Basin Municipal Water District (West Basin) and an intake for a full-scale desalination plant being proposed by West Basin. The demonstration facility uses a wedgewire screen module (WWS) as part of the intake. WWS was designed to reduce impacts on marine organisms by reducing the number of organisms that are entrained, or drawn into the intake, and the numbers of organisms that become impinged on the screen surface. The two objectives of these studies were 1) to determine the potential effects on marine organisms due to the operation of the intakes for the demonstration and full-scale facilities, and 2) to determine the efficiency of the WWS at reducing the effects of entrainment and impingement. The study assessing the potential effects on marine organisms was completed as one phase of the project, while additional studies were conducted to evaluate the efficiency of the screening system at reducing impacts. The combined results from all studies provide an overall assessment of the impacts of the demonstration facility, the proposed full-scale facility, and of the potential reductions in impacts due to the use of WWS.

The purpose of the small-scale temporary Desalination Demonstration Facility (WBDDF) constructed and operated by West Basin at the SEA Lab in Redondo Beach, California was to conduct research and testing on full-scale desalination equipment, including the WWS intake system designed to reduce impacts on marine organisms. When water is withdrawn from a source water body for industrial or municipal purposes, organisms within the water body where the intake is located may be entrained through pumps or impinged on intake screens. The operation of such intake systems can affect biological populations in the source water through removal of larvae that are entrained in water flows and of larger life stages that may be impinged on the intake screens. The results from the studies of the intake system at the WBDDF could be used in the design of a larger facility being proposed by West Basin.

Intake Assessment Study

The first objective of the studies at the WBDDF was to conduct an assessment of the potential effects on marine organisms due to the operation of the intakes. This involved collecting monthly plankton samples using towed plankton nets over a 12-month study period to provide a baseline characterization of the potentially entrainable fish larvae, fish eggs, and selected invertebrate larvae at the proposed intake location and in the surrounding source water (**Figure ES-1**). Sampling for the 12-month baseline characterization study was initiated on March 31, 2011 and completed on March 5, 2012.

Over the year-long study there were 47 plankton samples collected and processed from the intake station (SWE) and 69 samples collected from the three source water stations (SW1–3) (**Figure**

ES-1). A total of 831 fish larvae in 44 taxonomic groups (including unidentified and/or damaged larvae) was collected at the SWE. Ten taxa comprised over 80 percent of the total mean concentration of fish larvae with the most abundant being jacksmelt (*Atherinopsis californiensis*), white croaker (*Genyonemus lineatus*), unidentified larval/post larval fishes, herrings and anchovies (Clupeiformes), combtooth blennies (*Hypsoblennius* spp.), roughcheek sculpin (*Ruscarius creaseri*) and garibaldi (*Hypsypops rubicundus*). Jacksmelt comprised approximately one fourth of the total number of larval fishes collected at SWE. An estimated total of 78,759 fish eggs (adjusted for subsampling) was collected from the intake station during the study period. Of the specimens that could be identified to a lower category, turbot, sanddab, herring, and sand flounder eggs were the most numerous. Fish larvae were generally collected in greater abundance at night than during the day. Target invertebrate larvae included Cancer crab megalops, market squid paralarvae (recently hatched), and California spiny lobster phyllosomes. There were 462 invertebrate specimens collected from seven taxonomic groupings. Cancer crabs representing at least four species were the most abundant of the target invertebrate larvae collected.

Figure ES-1. Location of intake (SWE) and source water stations for plankton sampling.

A total of 1,397 fish larvae in 59 taxonomic categories was collected at the source water stations (SW1, SW2, and SW3) during the 12 monthly surveys. Jacksmelt, anchovies, and white croaker were the three most abundant taxa overall. The peak in abundance of larval fish at the source water stations occurred during February 2012 and lowest concentrations occurred in June 2011. A total of 106,022 fish eggs (adjusted for subsampling) was collected from the three source water stations during the study period. The eggs were classified into 18 taxa, however, the majority of specimens (73 percent) were in early developmental stages and remained unidentified due to an absence of definitive identification characteristics. The peak in abundance of fish eggs at the source water stations was in August 2011 with approximately 35,000/1,000 m^3 and lowest concentrations occurred in January 2012 (2,600/1,000 m^3). A total of 457 target invertebrate larvae was collected at the source water stations during the study period including 359 Cancer crab megalopae from five taxonomic groupings. Cancer crab megalopae comprised 78.5 percent of the target larvae collected.

The average estimated concentrations of each taxon per survey were extrapolated over each survey period and then the survey period estimates added to provide an estimate of total annual entrainment. Based on a proposed maximum feedwater pumping rate of 170,722 m^3 per day (45.1 mgd) for a full-scale facility, it was estimated that 10.2 million larval fishes would be entrained annually through an unscreened intake system (**Table ES-1**). The WBDDF project had substantially lower pumping rates (1,309 m^3 per day [0.346 mgd] average), and 1,935 m^3 per day ([0.511 mgd] design), that entrained an estimated 77,939 and 115,208 fish larvae per year, respectively. The maximum annual entrainment estimate for fish eggs was approximately 834.5 million annually based on the 45.1 mgd flows, and 6.4 million based on the average demonstration plant flows of 0.346 mgd. An estimated 3.9 million target invertebrate larvae would be entrained at the maximum rate for the full-scale facility and 30,184 at the minimum calculated flow rate.

Seasonally, the highest overall concentrations of larval fishes at Station SWE occurred in January and February 2012 with smaller peaks in September and October 2011, while the lowest concentrations occurred in March and November 2011 (**Figure ES-2**). Peak concentrations of fish eggs occurred in August 2011 and February 2012.

Detailed modeling of potential impacts using the Empirical Transport Model (ETM) was done for the larvae of four fishes (silversides Family Atherinopsidae, white croaker *Genyonemus lineatus*, northern anchovy *Engraulis mordax*, and kelpfishes *Gibbonsia* spp.) and one group of invertebrates, Cancer crabs. These were selected for analysis because they were collected in relatively high abundance during the studies and also were collected during a majority of the 12 surveys. Other fishes, such as garibaldi, were high in overall abundance (**Table ES-1**) but were only collected during two of the surveys at the intake station and their occurrence at the source water stations only coincided with one of those surveys. As a result, there was only one estimate of entrainment relative to source water stations that could be calculated for garibaldi. The estimate of proportional entrainment is the primary input variable to the ETM and the data for garibaldi would have provided a single estimate for the modeling. As a result, while the ETM estimate of the potential effects of the intake on garibaldi larvae, or the annual proportion of the source water population of larvae entrained by the intake, could be calculated, the reliability of the estimate would be in question.

Table ES-1. Estimated annual entrainment of fish larvae, fish eggs, and target invertebrate larvae through an unscreened intake based on data collected with towed plankton nets at the intake station (SWE) from March 2011 through March 2012 for three intake volumes.

	Taxon	Common Name	Estimated Annual Entrainment 1,309 m³ (0.346 mgd)	1,935 m³ (0.511 mgd)	170,722 m³ (45.1 mgd)
Fish Larvae					
1	Atherinopsidae	silversides	18,983	28,061	2,475,663
2	larval/post-larval fish	larval fishes	7,508	11,099	979,167
3	*Genyonemus lineatus*	white croaker	7,251	10,718	945,568
4	*Hypsypops rubicundus*	garibaldi	5,371	7,939	700,450
5	Clupeiformes	herrings and anchovies	5,156	7,622	672,421
6	*Hypsoblennius* spp.	combtooth blennies	5,114	7,560	666,944
7	Engraulidae	anchovies	4,548	6,723	593,133
8	*Ruscarius creaseri*	roughcheek sculpin	4,545	6,718	592,713
9	*Gibbonsia* spp.	kelpfishes	2,580	3,814	336,495
10	*Citharichthys* spp.	sanddabs	2,468	3,649	321,907
11	*Paralichthys californicus*	California halibut	1,391	2,056	181,368
12	*Pleuronichthys* spp.	turbots	1,341	1,982	174,897
13	CIQ goby complex	gobies	1,160	1,715	151,294
14	Syngnathidae	pipefishes	1,152	1,703	150,258
15	*Parophrys vetulus*	English sole	893	1,320	116,477
16	*Neoclinus* spp.	fringeheads	848	1,253	110,540
17	*Sebastes* spp. V	rockfishes	704	1,041	91,798
18	*Sebastes* spp. V_	rockfishes	701	1,036	91,359
19	*Zaniolepis frenata*	shortspine combfish	670	990	87,358
20	*Paralabrax* spp.	sea basses	564	834	73,547
		24 other taxa	4,991	7,375	650,760
			77,939	**115,208**	**10,164,117**
Fish Eggs					
1	fish eggs unid. (early stage)	unidentified fish eggs	4,661,669	6,890,838	607,936,680
2	*Pleuronichthys* spp.	turbot eggs	507,156	749,673	66,139,109
3	*Citharichthys* spp.	sanddab eggs	470,298	695,190	61,332,411
4	Clupeidae	herring eggs	223,302	330,083	29,121,253
5	Paralichthyidae	sand flounder eggs	197,609	292,103	25,770,499
		15 other taxa	338,853	500,892	44,190,542
			6,398,887	**9,458,779**	**834,490,494**
Target Invertebrate Larvae					
1	*Romaleon antennarius / Metacarcinus gracilis*	Cancer crabs	22,971	33,955	2,995,675
2	*Metacarcinus anthonyi* (megalops)	yellow crab megalops	2,949	4,360	384,615
3	*Doryteuthis opalescens*	market squid	1,616	2,388	210,697
4	*Panulirus interruptus* (phyllosome)	Calif. spiny lobster (larval)	1,614	2,386	210,475
	Cancer crab megalops	Cancer crab megalops	1,034	1,529	134,916
			30,184	**44,618**	**3,936,378**

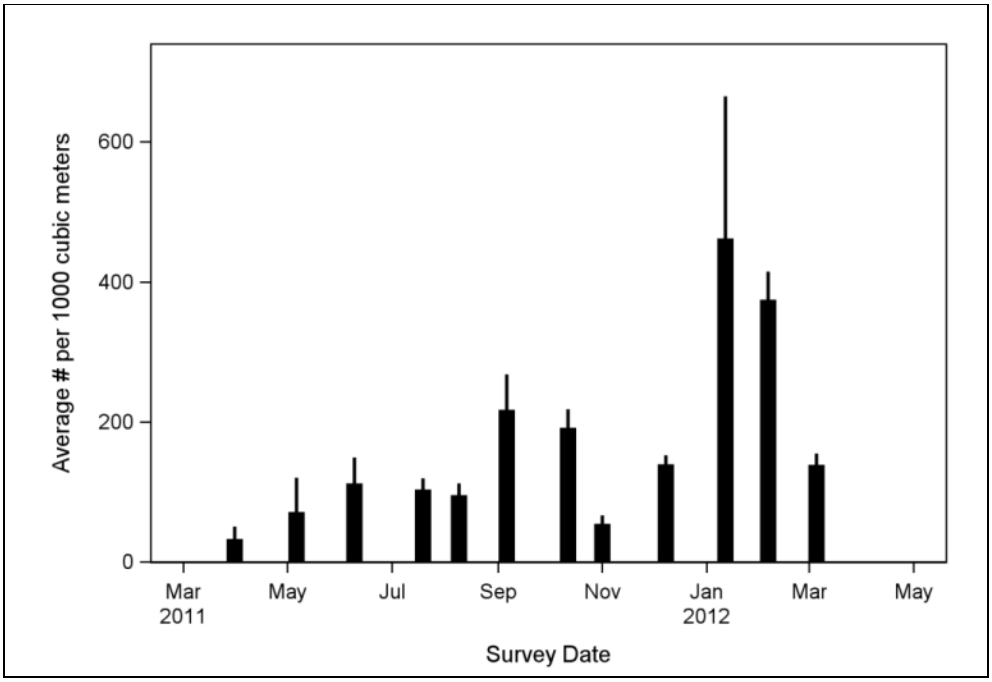

Figure ES-2. Mean concentration (#/1,000 m^3) and standard error for all larval fishes collected with towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

The ETM analyses of the four fishes and Cancer crab larvae for the WBDDF intake volumes of 1,309 m^3 (0.346 mgd) and 1,935 m^3 (0.511 mgd) resulted in estimated impacts on the populations of larvae for fishes and crabs in the extrapolated source water of a hundredth, or thousandths, of a percent (**Table ES-2**). The estimated impacts were higher for the proposed full-scale facility because the intake volume of 170,722 m^3 (45.1 mgd) was approximately two orders of magnitude larger than the intake volumes for the estimates calculated for the WBDDF intake. An annual percentage mortality of 7.9 percent for silversides was calculated using the larger intake volume of the full-scale facility, which was the largest for any of the taxa analyzed.

It is important to understand the many factors that can affect the ETM when interpreting results for silversides and other taxa. Seasonal changes in current speed and direction affected the estimated size of the source water during each survey period, and this also affected the results. For example, the estimated impacts for silversides were greater than the estimates for white croaker even though both fishes had very similar estimates of the number of days the larvae were exposed to entrainment. The ETM estimate for silversides was heavily weighted by one survey when a large proportion of the larvae were collected but the currents estimated an alongshore source water extent of only 2.1 km (1.3 mi). This results in concentrating the impacts into a smaller source water volume and results in increased estimated mortality for that survey.

Table ES-2. Estimated annual proportional (percentage mortality in parentheses) mortality to source water population due to entrainment by an unscreened intake using ETM. ETM results for each taxa are based on estimated upcoast and downcoast source water populations based on back-projections limited to depths of 30 m (98 ft), 100 m (328 ft), and 300 m (984 ft). ETM estimates calculated for three unscreened intake volumes: a) actual flow during the year for the WBDDF of 1,309 m^3 (0.346 mgd), b) design flow for the WBDDF of 1,935 m^3 (0.511 mgd), and c) a projected flow for the proposed facility of 170,722 m^3 (45.1 mgd).

Taxon	CODAR Depth Limit	a) ETM Estimate WBDDF Actual Flow	b) ETM Estimate WBDDF Design Flow	c) ETM Estimate WBDDF Full Scale Flow
silversides	30 m Depth	0.00067 (0.067)	0.00098 (0.098)	0.07878 (7.878)
	100 m Depth	0.00006 (0.006)	0.00009 (0.009)	0.00744 (0.744)
white croaker	30 m Depth	0.00011 (0.011)	0.00016 (0.016)	0.01351 (1.351)
	100 m Depth	0.00003 (0.003)	0.00004 (0.004)	0.00366 (0.366)
northern anchovy	100 m Depth	0.00003 (0.003)	0.00004 (0.004)	0.00323 (0.323)
	300 m Depth	0.00002 (0.002)	0.00003 (0.003)	0.00267 (0.267)
kelpfishes	30 m Depth	0.00018 (0.018)	0.00026 (0.026)	0.02234 (2.234)
	100 m Depth	0.00006 (0.006)	0.00009 (0.009)	0.00741 (0.741)
Cancer crabs	30 m Depth	0.00016 (0.016)	0.00023 (0.023)	0.01965 (1.965)
	100 m Depth	0.00007 (0.007)	0.00010 (0.010)	0.00857 (0.857)

As a result of the multiple factors that can affect the estimates from the ETM, the results, and most importantly the estimates of entrainment used in ETM calculations, must necessarily be placed into context. One approach is to compare the absolute numbers of larvae entrained to the actual reproductive capacity for a species. Although the ETM results may indicate a large percentage loss to the source water population of larvae, the actual impacts due to entrainment may be negligible since the actual number of larvae entrained is very small relative to the reproductive capacity. For example, the total entrainment estimates for white croaker and California halibut larvae for the proposed full-scale project were 945,568 and 181,368 per year, respectively. These annual entrainment estimates represent the annual production of a few females for white croaker to perhaps only one female for California halibut.

Another approach for placing ETM results into context is referred to as 'area of production foregone' (APF), which scales the ETM results based on the available adult spawning habitat. The APF places the ETM results into context by estimating the area of adult spawning habitat that would be necessary to replace the larvae lost due to entrainment. This requires that an actual habitat type can be identified that is associated with spawning by a species. For example, fishes such as garibaldi and kelpfishes occupy rocky reef areas that they also use for attaching eggs into nests that are guarded by these fishes. The APF would also be useful for placing the ETM results for silversides into context. Although silversides can occur in deeper water on the open coast they are most commonly associated with bay and estuarine habitats where the females attach their eggs to subtidal vegetation, such as kelp and eelgrass, and also other structures. The most prevalent habitat for egg attachment in the vicinity of Redondo Beach and King Harbor are small kelp beds in areas exposed to ocean water, shallow areas where eelgrass may occur, and rock jetties and pier pilings covered with marine algae. To calculate the APF for silversides, an estimate of these types of habitats would need to be calculated and then the ETM estimate of proportional mortality, 7.9 percent, applied to that estimate (**Table ES-2**). Although it is unlikely that the removal of 7.9 percent of the larval production from the area would affect the local population of silversides, the APF estimate could be used to determine appropriate mitigation that would completely compensate for the entrainment losses.

Intake Screen Efficiency Studies

The second objective of the Intake Effects Assessment studies was to determine the efficiency of the cylindrical narrow-slot WWS intake modules at reducing the effects of entrainment and impingement. This aspect of the study involved three components: 1) sampling at the offshore intake location to determine the effectiveness of entrainment reduction for WWS intake modules with two slot openings (1 mm [0.04 in.] and 2 mm [0.08 in.] slot widths), 2) analysis of length-head capsule data to predict entrainment probabilities for different taxa, larval size classes, and different screen slot dimensions, and, 3) collection of videographic data to assess overall impingement effects and to compare impingement performance between the two WWS modules.

Wedgewire Screen Efficiency Study

This study was designed to provide an alternate estimate of entrainment and to determine the effectiveness of the WWS screens at reducing entrainment of later stage fish and invertebrate larvae. The first part of the study involved collection of samples through the year in conjunction with the sampling for the impact assessment. The second part of the WWS efficiency study was designed to reduce temporal/seasonal variability in the data collected by conducting an intensive sampling effort over a short period of time during a time of high fish larvae abundance. During the special screen efficiency study, a large number of samples were planned to be collected simultaneously over a three-day period from an unscreemed intake and one of the WWS modules and then during a second three day period from the unscreened intake and the other WWS module. This sampling effort was intended to provide a large enough sample size for a statistically valid estimate of the percentage entrainment reduction for both WWS screens.

This component of the studies was not completed due to problems with both sampling efforts. The problems with the first sampling effort included issues resulting from the installation and maintenance of the WWS modules, and concerns regarding the integrity of the samples due to biofouling within the intake lines and the need to sample from a boat to provide the unscreened samples for comparison. These problems resulted in the collection of only a limited number of samples that did not provide the necessary data for detecting any differences between concentrations of larvae from the WWS modules and the unscreened intake. The inability to detect any differences between the screened and unscreened intake data was further complicated by the low abundances of fish larvae during the study period. Only 136 fish larvae were collected during the sampling and samples were not collected due to problems with the WWS modules during the only survey when fish larvae were in high abundances. Therefore, the presence of fish larvae in the samples almost represented random events resulting in data which could not be analyzed. Fish eggs and invertebrate larvae were more abundant and showed patterns which were consistent with the expectation of reduced entrainment through the 1 and 2 mm WWS modules. As a result of the concerns regarding the integrity of all the samples, none of these data, including the data for fish eggs and invertebrate larvae, were analyzed for this report, but the data are provided in the appendices.

The potential for high variability in the data from the first sampling effort was anticipated during the design of the study since it was expected that variation in abundances through the year may make it difficult to detect any differences due in entrainment through the WWS modules. Therefore, the design included a more intensive sampling effort designed to collect data during the times of the year when abundances of larvae where highest. This sampling effort was planned to occur during the spring and summer of 2012 following the regular monthly sampling when fish larvae were expected to be in highest abundance. Monitoring of larval fish abundances was conducted through the spring months of 2013, but suitable conditions in the marine waters off King Harbor/Manhattan Beach for conducting the study were not observed and as a result the study was curtailed.

Another component of the intake screen efficiency studies was an analysis that provided estimates of the probability of entrainment based on the notochord length and head capsule dimensions of the larval fishes found to be most susceptible to entrainment from the intake assessment. The probability of entrainment was calculated for the five most abundantly collected fish taxa for three different WWS slot widths, 0.5 mm (0.02 in.), 1 mm (0.04 in.), and 2 mm (0.08 in.). The length specific estimated reductions in entrainment for the three WWS slot openings for the five most abundantly collected taxa during the study were based on data from a recent study (Tenera 2011). The Tenera (2011) study estimated probabilities of entrainment for larvae of various lengths based on the relationship between length and head capsule dimension (width and depth). The head capsule of fish larvae is the only part of the body that is not soft and compressible and therefore was used as the limiting dimension for determining which larvae could be entrained. The calculated probabilities for each length for the five taxa were used to assess the reductions in entrainment based on the size composition of larvae collected from previous intake studies in southern California. Data from several studies were composited to provide a more complete data set for the analysis.

The results showed that the reductions in entrainment for the larger 1 mm (0.04 in.) and 2 mm (0.08 in.) WWS were generally very small with the largest reductions being estimated for the 1 mm (0.04 in.) slot width for silversides and northern anchovy at 41 and 28 percent, respectively. The levels of entrainment reductions were small (12 to <0.1 percent) for all three slot widths for combtooth blennies, which were estimated to be subject to entrainment over a very limited size range from 2–6 mm (0.08–0.24 in.). These estimates are likely very conservative as the approach assumes that all the larvae with head capsule dimensions smaller than the WWS slot opening would be entrained, that the larvae would be oriented in a position that would allow them to be entrained, and that there would be no benefit from hydrodynamic flow along the screen face due to ambient currents that would tend to carry the larvae away from the screen.

The results of the length-specific entrainment estimates showed the importance of considering the size composition of the population in considering the effects of the screens at reducing the effects of entrainment. The probabilities across the size range of entrainable larvae were used to assess reductions in entrainment mortality when using a particular screen dimension and the overall effects on population mortality. This analysis required the assumptions that the larvae show linear growth over time, and a constant exponential natural mortality rate. The average reductions in mortality shown in **Table ES-5** would need to be adjusted for the composition and size structure of the fish larvae for a specific location and sample year, but otherwise provide an estimate of population-level mortality identical to an adult equivalent model using constant growth and survival rates to the length or age when the fish are no longer subject to entrainment.

Table ES-5. Estimated percentage reductions (two standard errors in parentheses) in mortality (relative to an open intake) to the population surviving past the size where they would be subject to entrainment, based on probabilities of screen entrainment for larvae from five taxonomic categories of fishes.

Taxon	Size Range	Percentage Reduction in Mortality by Slot Opening Width		
		0.5 mm	1 mm	2 mm
kelpfishes	2–25 mm	82.2 (2.4)	64.6 (2.4)	24.9 (2.4)
combtooth blennies	2–10 mm	88.2 (4.3)	62.1 (4.1)	15.4 (2.7)
anchovies	2–25 mm	66.8 (2.3)	45.1 (2.3)	5.5 (1.6)
croakers	1–15 mm	85.3 (3.8)	66.5 (4.0)	28.1 (3.9)
silversides	2–25 mm	84.2 (2.6)	68.5 (2.5)	34.8 (2.4)
Average Percentage Reduction		**80.0**	**57.2**	**19.0**

These estimates were substantially greater than the estimated numbers of larvae protected from entrainment that were also estimated for each of the five taxa, and provide a more realistic estimate of the actual benefits of the WWS.

Wedgewire Screen Impingement Study

Videographic data were collected to determine the potential impingement effects on marine organisms due to the operation of the intakes for the demonstration and full-scale desalination facilities, and to assess the efficiency of WWS intake modules with different screen slot widths at reducing impingement effects. The video data was collected using a housed digital video camera focused on sections of the screen surface of each of the WWS intake modules.

A total of 12 WWS impingement surveys was conducted involving 21 video recording events, 11 for the 1 mm (0.04 in.) WWS module and ten for the 2 mm (0.08 in.) WWS module. A total of

67 hours and 45 minutes of digital video (135 files) was recorded of the intake screens while they were in operation during the WWS impingement study. Twenty one percent of the recorded video (30 files; 14 hours and 10 minutes) was randomly selected for review. All direct interactions with the screen were recorded for most of the randomly selected recordings. However, the numbers of invertebrates swimming near the screen were so great during some periods that impingement events could not be recorded individually.

A total of 717 events was observed and recorded onto video log sheets. Of the recorded observations, 339 were of invertebrates, 226 were of fishes, and 152 were of debris (**Table ES-6**). A description of each event was recorded on video log sheets and tallied by the type of organism/object (fish, invertebrate, or debris) and whether it came in contact with the screen or was seen in the vicinity of the screen. The events in which a fish was observed to contact the screen were generally classified as "bumps" or other brief "contacts" with the screen, and at no time was a fish observed that appeared to be impinged or otherwise held on either the 1 mm (0.04 in.) or 2 mm (0.08 in.) screen by the intake flows. The results support data from previous studies showing that impingement impacts should be substantially reduced or eliminated through the use of WWS.

Table ES-6. Summary of information collected from randomly chosen video files taken during the WWS impingement study from March 30, 2011 through January 18, 2012.

Date	Cycle	Sample Length (min)	Amount Reviewed (min)	Percent Reviewed	Fishes		Invertebrates		Debris	
					Near	Contact	Near	Contact	Near	Contact
03/30/11*	Day	–	–	–	–	–	–	–	–	–
	Night	113	113	100	0	2	0	17	0	0
05/04/11	Day	57	57	100	19	0	0	5	1	1
	Night	85	85	100	5	2	135	54	69	23
07/28/11	Day	–	–	–	–	–	–	–	–	–
	Night	28	28	100	0	0	1	13	4	4
08/08/11	Day	–	–	–	–	–	–	–	–	–
	Night	28	28	100	0	0	0	0	0	0
09/22/11	Day	113	113	100	67	5	25	9	8	10
	Night	57	57	100	2	0	0	3	0	0
10/26/11	Day	–	–	–	–	–	–	–	–	–
	Night	113	113	100	1	0	3	27	3	1
12/07/11*	Day	–	–	–	–	–	–	–	–	–
	Night	28	28	100	13	5	0	1	0	0
01/18/12*	Day	113	113	100	9	1	5	2	11	16
	Night	113	113	100	82	13	1	38	0	1
	Totals	**14 hrs: 10 min**	**14 hrs: 10 min**	**100**	**198**	**28**	**170**	**169**	**96**	**56**

* Numbers of invertebrates in constant proximity/contact with the screen too great to record individually.

Summary

The results of the studies presented in this report indicate that the operation of the WBDDF and the proposed full-scale project with a projected intake volume of 170,722 m^3 (45.1 mgd) will result in very low levels of impact to populations of fishes and invertebrates that were the focus of the study. The detailed impact assessment used the same modeling approach, ETM, that has been used in evaluating the impacts of ocean intakes at most of the coastal power plants in California. The results of the modeling showed that the estimated impacts due to the operation of the WBDDF, with a maximum daily intake volume of 1,935 m^3 (0.511 mgd), resulted in estimated losses to the populations of larvae for fishes and crabs in the extrapolated source water of only a hundredth, or thousandths, of a percent (**Table ES-2**). While the estimated impacts were higher for the proposed full-scale facility, which were based on an intake volume of 170,722 m^3 (45.1 mgd), the losses due to entrainment represented only 1–2 percent of the estimated source water populations for all of the taxa analyzed except silversides, which were estimated at 7.9 percent.

The integrated estimate of ETM is dependent on the comparison of the estimates of entrained larvae to the estimate of the number of larvae in the source water and therefore can be affected by local conditions that may result in greater abundances of larvae at the intake or source water locations. In the case of silversides, the higher ETM estimate for that fish reflects the greater concentrations of silverside larvae at the intake location relative to the source water stations, which was likely due to the proximity of the intake to the rock jetties around Redondo Beach and King Harbor where there are small kelp beds in areas exposed to ocean water, shallow areas where eelgrass may occur, and also rock jetties and pier pilings covered with marine algae, which jacksmelt may use as spawning habitat. This type of habitat also occurs at the base of the concrete structure where the WBDDF intakes were located.

Although silverside larvae were collected in high numbers using the net sampling used for the impact assessment, they were collected in very low numbers from the pumped samples for both the unscreened and WWS intakes. While the low numbers did not allow for an assessment of the effectiveness of the WWS at reducing entrainment, the large average length of the silverside larvae collected from the net samples (9.87 mm [0.39 in.]) indicates that the planned use of an intake with a WWS would reduce or nearly eliminate entrainment of silverside larvae, since the large size of the larvae would prevent the entrainment of a large percentage of the larvae. For example, the modeling indicates that a 0.5 mm (0.02 in.) WWS module would reduce the numbers of silverside larvae entrained by approximately 84 percent (**Table ES-5**). Depending on the final selection of WWS technology and slot opening, the 7.9 percent ETM estimate of population-level losses would need to be adjusted by the level of reduction obtained through the WWS. For example, the adjusted ETM estimate using a 0.5 mm (0.02 in.) WWS module would be 0.2 percent.

As the results of the screen effectiveness study showed, intakes fitted with WWS will significantly reduce or eliminate any effects due to impingement, and modeling results also show the potential for reduction in entrainment. The evaluation of the effectiveness of WWS needs to consider the sizes of the larvae entrained. The life history characteristics of most marine fishes and invertebrates involve the production of a very large number of larvae, which experience very high rates of natural mortality. The effectiveness of WWS is measured by eliminating entrainment of larger, older larvae which have a much greater probability of surviving to the adult stage and shown in the results of the screen comparison study where the average length of fishes collected from the 2 mm (0.08 in.) WWS module was larger than the average length from the unscreened intake. The proven effectiveness of WWS at reducing or eliminating impingement and the potential for reductions in entrainment indicate that the use of WWS modules at the planned West Basin facility and the low intake volume represents the best technology currently available for reducing the impacts of ocean intakes.

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1.0 Introduction

1.1 Overview and Background

This report presents an assessment of potential impacts to marine life from water withdrawals associated with the operation of an open ocean intake for a small, temporary ocean water desalination demonstration facility by the West Basin Municipal Water District (West Basin), and a full-scale ocean water desalination facility proposed by West Basin. The intake for the demonstration facility uses a passive screening system that was designed to reduce impacts to populations of marine organisms by reducing the number of organisms that are entrained, or drawn into the intake, and by reducing the numbers of organisms that become impinged on the screen surface. The two objectives of these studies are 1) to determine the potential effects on marine organisms due to the operation of the intakes for the demonstration and full-scale facilities, and 2) to determine the efficiency of the wedgewire screens (WWS) at reducing entrainment and impingement effects on local populations of marine fishes and invertebrates. One study was completed using towed plankton nets to collect data for assessment of the potential effects of feedwater withdrawls on marine organisms . Additional studies were designed to assess the efficiency of the screening system in reducing impacts. The combined results from the two sets of studies provide an overall assessment of the impacts of the demonstration facility, the proposed full-scale facility, and of the potential reductions in impacts due to the WWS.

West Basin is a public agency that provides drinking and recycled water to a 185-square mile service area in southwest Los Angeles County that has been testing the feasibility of ocean water desalination since 2002. After six years of study at a pilot facility in El Segundo, West Basin constructed and is operating a temporary ocean water Desalination Demonstration Facility (WBDDF) at the SEA Lab in Redondo Beach, California to conduct research and testing using full-scale equipment. The WBDDF draws in approximately 1,892 m^3 (500,000 gallons per day [gpd]) from a 1 mm (0.04 inch [in.]) and/or 2 mm (0.08 in.) wedgewire intake screen mounted atop of an existing decommissioned intake riser for the adjacent AES Redondo Beach Generating Station (RBGS). The RBGS is a fossil fuel electric generating station that uses a once-through seawater cooling system with a maximum pumping capacity of over 800 million gallons per day (mgd). Previous environmental studies have described the impacts of RBGS operation on the nearshore marine environment.

When water is withdrawn from a source water body for industrial or municipal purposes, organisms within the water body may be entrained through pumps or impinged on intake screens. Water intake systems can affect source water populations by uncompensated removal of larvae that are entrained in water flows and removal of larger life stages that may be impinged on the intake screens. Saline water for a desalination facility feedwater can be supplied from either offshore open-water intakes or, in some cases, a subsurface infiltration system that withdraws saline or brackish water from beneath shallow sediments along the coastline.

As part of the WBDDF, this study was conducted to evaluate potential impacts to marine organisms from the operation of the demonstration facility and to help evaluate the potential impacts from a larger, full-scale facility. In addition, the intake system for the WBDDF was designed to allow for the evaluation of passive screening intake systems to determine their effectiveness at reducing impacts to marine life due to entrainment and impingement.

1.2 Regulatory Setting

This study was undertaken in consideration of State Water Code Section 13142.5(d), which states that, "…independent baseline studies of the existing marine system should be conducted in the area that could be affected by a new or expanded industrial facility using seawater, in advance of the carrying out of the development." The WBDDF feedwater withdrawal would not be subject to regulation under the Federal Clean Water Act (CWA) Section 316(b) because it does not include a cooling water intake structure (CWIS). However, the State Water Quality Control Board recommends that 316(b)-type studies be conducted for open water intakes. Section 316(b) requires that the design and operation of a CWIS minimize adverse environmental effects due to impingement and entrainment of aquatic life. Impingement of larger organisms occurs when they are trapped against the screening systems commonly used at CWIS entrances and entrainment occurs when organisms pass through any screening system into the CWIS where mortality occurs. This study addressed the potential of entrainment and examined effects of larval impingement since the intake system will be designed to reduce intake velocities to levels that should eliminate any concerns regarding impingement of larger organisms. The entrainment study was conducted using sampling and analysis methods consistent with Section 316(b) studies done throughout California over the past several years.

1.3 Study Objectives

This study had two objectives. The first objective was to estimate the potential effects on marine organisms of the WBDDF intake and the intake for a proposed larger facility. The study plan for this part of the overall study was based on a survey of available background literature, results of a recent study of intake effects at the RBGS, and intake system studies at other power plants and proposed desalination facilities in California. The overall approach was to collect data on the concentrations of fish eggs, fish larvae, and target invertebrate larvae in the source water at the intake using towed plankton nets, the standard sampling method for these organisms. Once the average concentrations of larvae were quantified over a period of a year, potential impacts to populations of fishes and invertebrates can be assessed based on the fractional removal of larvae caused by a nearshore unscreened intake and knowledge of the individual species life histories. Typically the assessment is only done for the most abundant organisms in the samples to ensure that adequate data exist to ensure reasonable levels of confidence in the abundance estimates.

Entrainment effects from coastal power plants and desalination pilot facilities in California have been assessed using the Empirical Transport Model (ETM), as recommended and approved by the California Energy Commission (CEC), California Coastal Commission (CCC), Regional Water Quality Control Boards and other regulatory and resources agencies (Steinbeck et al. 2007). This model estimates the proportional loss to the standing stock of larvae in the source water due to entrainment through an unscreened intake using an estimate of mortality calculated as the ratio of the number of larvae entrained to the number estimated in the source water. The source water is defined as the area or volume of water from which larvae could be subject to entrainment. Source water volumes, intake volumes, larval concentrations, and larval durations are the variables used in the ETM calculations, which are done for each taxon due to differences in distribution, larval biology, and seasonality of spawning.

The second objective of the study was to determine the effectiveness of WWS intakes at reducing the effects of entrainment and impingement. The study plan for this part of the overall study included the collection of similar data on entrainment and impingement from a small-scale pump and screening system at the WBDDF. The system was comprised of two cylindrical intake screen modules, one constructed with narrow-slot 1 mm (0.04 in.) WWS material and the other with narrow-slot 2 mm (0.08 in.) WWS, and two feedwater pumps located at the onshore WBDDF. The intake modules were installed on the decommissioned discharge structure for Unit 1 (retired) of the RBGS, located offshore from the northwest corner of the King Harbor breakwater in a depth of 10.2 m (33.5 ft) Mean Lower Low Water (MLLW)**.** Feedwater drawn through each of the two screened intake modules was sampled using plankton nets suspended in tanks at the onshore WBDDF. Screening efficiency was evaluated by comparing larval concentrations from samples collected simultaneously from the two differently-sized screened intakes and from an unscreened intake.

The final goal of the study was to combine the results from the two study components to evaluate potential entrainment impacts by adjusting the proportional loss estimate from the ETM based on the estimated reduction of entrainment using the WWS intake.

1.3.1 Target Organisms Selected for Study

This section provides the rationale for the selection of the specific subset of planktonic organisms that were the focus of these studies. This rationale is based on regulatory guidance on §316(b) provided by the U.S. Environmental Protection Agency (USEPA) (USEPA 1977), as well as practical considerations regarding intake assessment studies and the methods used to analyze the data. Another important consideration in the selection of organisms is the actual assessment on the effects of the intake, and the use of the results in scaling mitigation that could be used to compensate for any impacts.

Planktonic organisms that are smaller than the screen mesh size used at the entrance to a power plant's CWIS are susceptible to entrainment. These organisms include species that complete their entire life cycle as planktonic forms (holoplankton), such as diatoms, and those that only spend a portion of their life cycle in the plankton as eggs or larvae (meroplankton). All of the §316(b)-like entrainment studies in California have been designed to assess effects on meroplanktonic species including fish eggs and larvae, and advanced larval stages of several invertebrate species such as crabs, squid, and California spiny lobster (*Panulirus interruptus*). The studies have not included holoplankton because, as recognized in the original §316(b) guidance (USEPA 1977), the potential for any impacts on holoplankton due to entrainment is extremely low, especially in the ocean environment. The reasons include the following:

1. Holoplankton have extremely short generation times; on the order of a few hours to a few days for phytoplankton and a few days to a few weeks for zooplankton;
2. Both phyto- and zooplankton have the capability to reproduce continually depending on environmental conditions; and
3. Many of the most abundant phyto- and zooplankton species along the California coast have populations that span the entire Pacific or, in some cases, all of the world's oceans. For example, *Acartia tonsa*, one of the common copepod (zooplankton) species, found in the nearshore areas of California, is distributed along the Atlantic and Pacific coasts of North and South America and the Indian Ocean.

Relative to the large abundances of phyto- and zooplankton, larval fishes make up a small fraction of the total numbers of the planktonic organisms present in seawater. The USEPA guidance correctly focused on potential impacts on fishes and shellfishes because they are more susceptible to entrainment effects for the following reasons:

1. They have much shorter spawning seasons relative to phyto- and zooplankton. In many species of fishes, spawning occurs only once during the year;
2. Unlike phyto- and zooplankton that may be distributed over large oceanic areas, most fishes are restricted to the narrow shelf along the coast, and in some cases have specific habitat requirements that further restrict their distribution;
3. Unlike many phyto- and zooplankton, there is a greater likelihood of mortality due to entrainment in larval fishes. This is because larval fishes are soft bodied and many phyto- and zooplankton are not soft bodied. The phyto- and zooplankton are better able to tolerate passage through intake systems, and as a result it is likely that some phyto- and zooplankton species survive entrainment; and
4. The process of entrainment probably has very little effect on the value of phyto- and zooplankton species as food sources for higher trophic levels, whereas the value of fish larvae lies in their potential to contribute to the adult populations of those species.

The entrainment sampling for all the intake studies in California have focused on fish and shellfish larvae as recommended in USEPA guidance. The USEPA guidance also provides latitude for focusing on the set of species that could be accurately quantified and would provide the necessary detail to support development of the assessment. The specific taxa (species or group of species) that are included in the intake assessments are typically limited to taxa that are sufficiently abundant to provide reasonable assessments of impacts. For most studies, the taxa analyzed in the assessments were limited to the most abundant and most commonly occurring taxa, because they provided the most robust and reliable estimates for the purpose of assessing impacts. Since the most abundant organisms may not necessarily be the organisms that experience the greatest effects on the population level, the data were also examined to determine if additional taxa should be included in the assessment. For example, this might include commercially or recreationally important taxa, taxa with limited habitats, and any threatened or endangered fish or shellfish species.

The USEPA lists several criteria for selecting appropriate target organisms for assessment including the following:

1. representative, in terms of their biological requirements, of a balanced, indigenous community of fish, shellfish, and wildlife;
2. commercially or recreationally valuable (i.e., among the top ten species landed—by dollar value);
3. threatened or endangered;
4. critical to the structure and function of the ecological system (e.g., habitat formers);
5. potentially capable of becoming localized nuisance species;
6. necessary, in the food chain, for the well-being of species determined in 1–4; and
7. meeting criteria 1–6 with potential susceptibility to entrapment/impingement and/or entrainment.

In addition to these USEPA criteria, there are certain practical considerations that limit the selection of target organisms, such as the following:

1. identifiable to the species level;
2. collected in sufficient abundance to allow for impact assessment (i.e., allowing the model(s) constraints to be met and confidence intervals to be calculated); and
3. having local adult and larval populations (i.e., source not sink species). For example, certain species that may be relatively abundant as entrained larvae may actually occur offshore or in deep water as adults.

Another important consideration in identifying organisms to include in an assessment is the ability to identify the habitat associated with a species. During reproduction, habitat associations may become even more defined as certain fishes use a specific habitat type for spawning, and may even tend a nest of eggs associated with a specific habitat. Results from our studies have shown that the distribution of larvae in an area can be strongly dependent on the variation in habitats or water depths.

The recognition of the importance of these habitat dependencies is one of the reasons for focusing these intake assessments on fish larvae. One of the important issues associated with these assessments is determining whether specific habitat types, and the associated species, are disproportionately affected by entrainment. Although this has not been shown in any of the intake assessments in California, it is conceivable that a small reef outcropping in an area could be disproportionately impacted due to an intake located in close proximity.

These habitat dependencies are not an issue with many mid-water and pelagic fishes such as anchovies, mackerel, barracuda, and some of the croakers. These fishes release eggs directly into the water column and, in the absence of any large spawning aggregations, should result in relatively uniform dispersal of eggs and larvae as we have seen in studies in southern California (ref?). In contrast with nearshore rocky reef fishes, which may have larvae that tend to occur close to shore, the larvae for these fishes tend to be distributed over large areas of coastline, reducing the potential for intake effects.

There are also no habitat dependencies for almost all holoplankton. Even in areas where the bottom habitat is not homogeneous, holoplankton would still, on average, tend to be distributed uniformly. Therefore, a simple ETM approach where the concentration of the organisms is assumed to be uniform in the intake and source water volumes can be used to provide a reasonable approximation for any impacts to holoplankton and also many fish and shellfish larvae (meroplankton).

Another critical component of the ETM approach is an estimate of the number of days the organisms are exposed to entrainment. This estimate is typically calculated based on the size of the fish larvae and estimates of larval growth rates from the scientific literature. The estimates can vary considerably among different fishes, with estimates exceeding 30 or 40 days for some taxa. As the number of days increases, the estimate of entrainment mortality from the ETM increases exponentially. The turnover rate of many holoplankton may be as short as a few days; therefore, as recognized in the original EPA guidance (USEPA 1977), the potential for impacts to these populations are considerably less than the potential for impacts to fish and shellfish larvae.

Therefore, the target taxa for this study include egg and larval stages of fishes that were found to be most abundant in the entrainment samples, later stage megalopal larvae of all species of Cancer crabs (family Cancridae, which includes the edible species of rock crabs), later stage phyllosome larvae of California spiny lobster, and market squid paralarvae. These are the same organisms included in recent entrainment studies at power plants along the southern California coast.

1.4 Report Organization

Section 2.0 provides a description of the proposed desalination project and the environmental setting. Section 3.0 presents the results of the source water and intake study and the modeling results for the most abundant fish and target invertebrates collected during the surveys. The results of a screen efficiency study and an underwater videographic study on impingement is provided in Section 4.0. An integrated discussion of impacts based on intake modeling results and screening efficiency studies is presented in Section 5.0, and the literature cited in the report is listed in Section 6.0. Appendix A provides the impact assessment modeling formulation, Appendix B provides ocean surface current information, Appendix C present the WBDDF intake and source water data by survey, Appendix D presents the pump sampling data by survey, Appendix E presents the length-head capsule regression plots for the length specific entrainment analysis, and Appendix F presents the field and lab sampling procedures.

2.0 Description of the Proposed Project and Environmental Setting

West Basin serves approximately one million customers within its southern California service area. The bulk of the water resources managed by West Basin are currently imported from the Colorado River and Northern California or produced locally through water recycling programs. The ocean water desalination technologies currently under development by West Basin are intended to offer a new source of water that will reduce dependence on imported water sources. West Basin's goal for the ocean water desalination program is for the production of desalinated ocean water to comprise 10 percent of the drinking water supplied to customers by 2020.

After preliminary studies at a pilot facility in El Segundo, West Basin is operating the temporary WBDDF at the SEA Lab in Redondo Beach (**Figure 2-1**) to conduct research and testing on full-scale desalination equipment. Approximately 378 m^3 per day (100,000 gpd) from the WBDDF intake flow is pretreated via ultrafiltration (Zeeweed 1000®), desalinated using a first pass reverse osmosis treatment system and partial second pass reverse osmosis system, and post-stabilized to evaluate water quality objectives.

2.1 Description of the Project

This intake effects study was conducted to provide an estimate of the entrainment effects of the WBDDF water intake system if an unscreened intake were being used, and through extrapolation the anticipated entrainment effects of a proposed full-scale desalination facility on selected marine biological resources. The study included two main components:

1) The first component involved monthly sampling using towed plankton nets at four locations in the source water and around the location of the WBDDF intake. The primary objective of the sampling was to collect sufficient data on the composition and concentrations of planktonic organisms in the source water for the estimation of annual entrainment effects for an unscreened intake using the ETM.

2) The second component of the study involved collecting plankton samples from the two WWS intake modules and an unscreened intake to provide data on concentrations of planktonic organisms that will be used to evaluate the efficiency of the different sized (slot size) wedgewire intake screens in reducing entrainment. This task also involved collecting plankton samples from the Substrate Infiltration Bed Intake (SIBI) feed water and the filtered water exiting the bed. These data were collected to document and quantify the potential entrainment impact of a potential subsurface intake.

All plankton samples collected were processed at Tenera Environmental's laboratory in San Luis Obispo, California. The fish larvae and target invertebrate larvae (Cancer crab megalops, California spiny lobster phyllosoma, and market squid paralarvae) were removed from the samples and identified to the lowest taxonomic level practicable, although some samples with large quantities of material were divided prior to processing. Fish eggs were also counted and identified to the lowest practical taxonomic level. Samples with large quantities of fish eggs were subsampled.

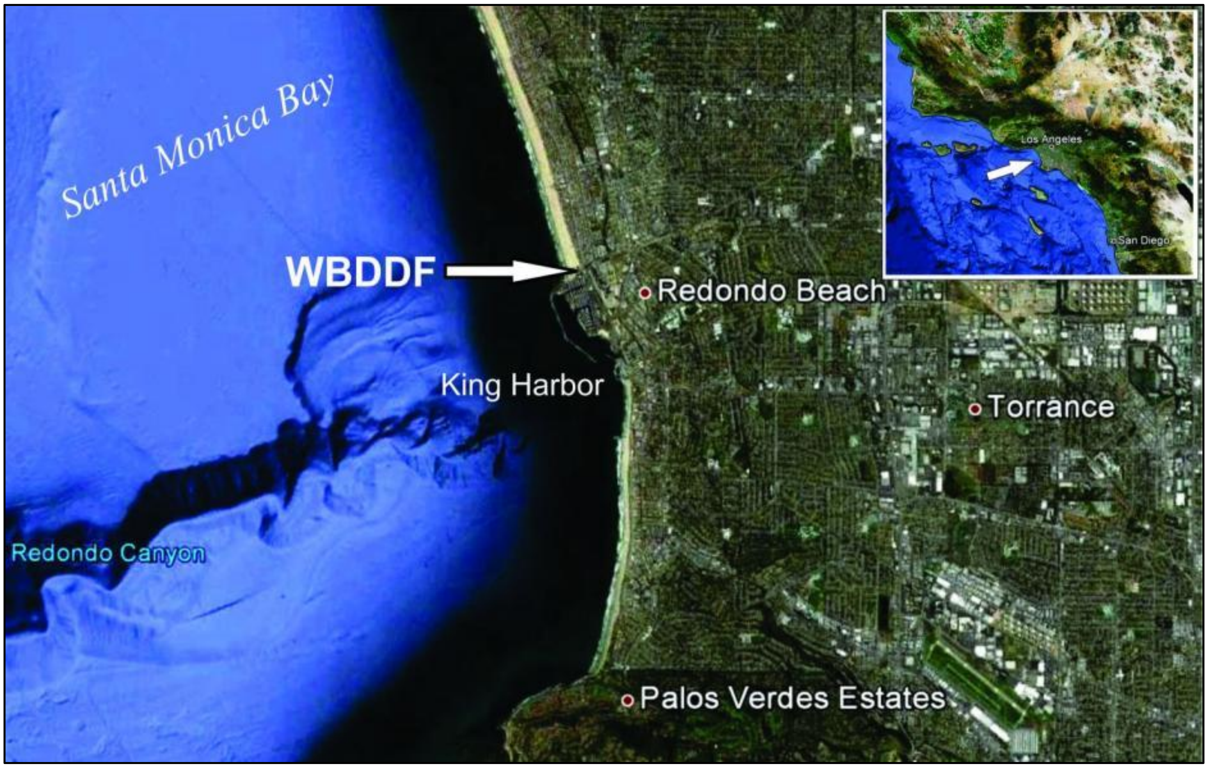


Figure 2-1. Location of West Basin Desalination Demonstration Facility in Redondo Beach, California.

2.2 Environmental Setting

Descriptions of Santa Monica Bay's physical and biological characteristics are provided in Sections 2.2.1 and 2.2.2, respectively.

2.2.1 Physical Description

2.2.1.1 Geographic Setting

Southern California lies in a climatic regime defined as Mediterranean, characterized by mild winters and warm, dry summers. In Santa Monica Bay, coolest temperatures generally occur from December through March, with warmest temperatures in July through September. Average monthly air temperatures in Santa Monica Bay in 2011 ranged from 12.4 to 14.2°C (54.3 to 57.5°F) from December through March to about 17.9°C (64°F) in July (NDBC 2011). Average annual precipitation in the coastal regions ranges between 25 and 38 cm (10 and 15 in.), with most precipitation occurring from October through April.

A subtropical high-pressure system offshore the Southern California Bight produces a net weak southerly/onshore flow in the area (Dailey et al. 1993). Wind speeds are usually moderate, and are on the order of 10 km/hr (6.2 mph). Wind speeds diminish with proximity to the coast, averaging about one-half the speeds offshore. Coastal winds in southern California are about one-half those found off central and northern California. However, strong winds occasionally accompany the passage of a storm. A diurnal land breeze is typical, particularly during summer, when a thermal low forms over the deserts to the east of the Los Angeles area. On occasion, a high-pressure area develops over the Great Basin, reversing the surface pressure gradient and resulting in strong, dry, gusty offshore winds in the coastal areas. These Santa Ana winds are most common in late summer, but can occur any time of year.

2.2.1.2 Physical Features

Santa Monica Bay is an open embayment approximately 43 km (27 mi) across and delineated by Point Dume, which is located approximately 42 km (26 mi) to the northwest of Redondo Beach and Palos Verdes Point, which is located approximately 9 km (6 mi) to the south (**Figures 2-1** and **2-2**). The surface area of the Bay is approximately 428 km^2 (266 mi^2). The Bay is characterized by a gently sloping continental shelf which extends seaward to the shelf break at water depths of approximately 80 m (265 ft) (Terry et al. 1956). Natural rocky outcrops are confined to the northern and southern portions of the Bay from Point Dume to the Malibu coast area to the north, and the Palos Verdes Point area to the south, respectively (Claisse et al. 2012; **Figure 2-2**). Sediments off the King Harbor area are primarily composed of sand, with lesser amounts of gravel, silt, and clay.

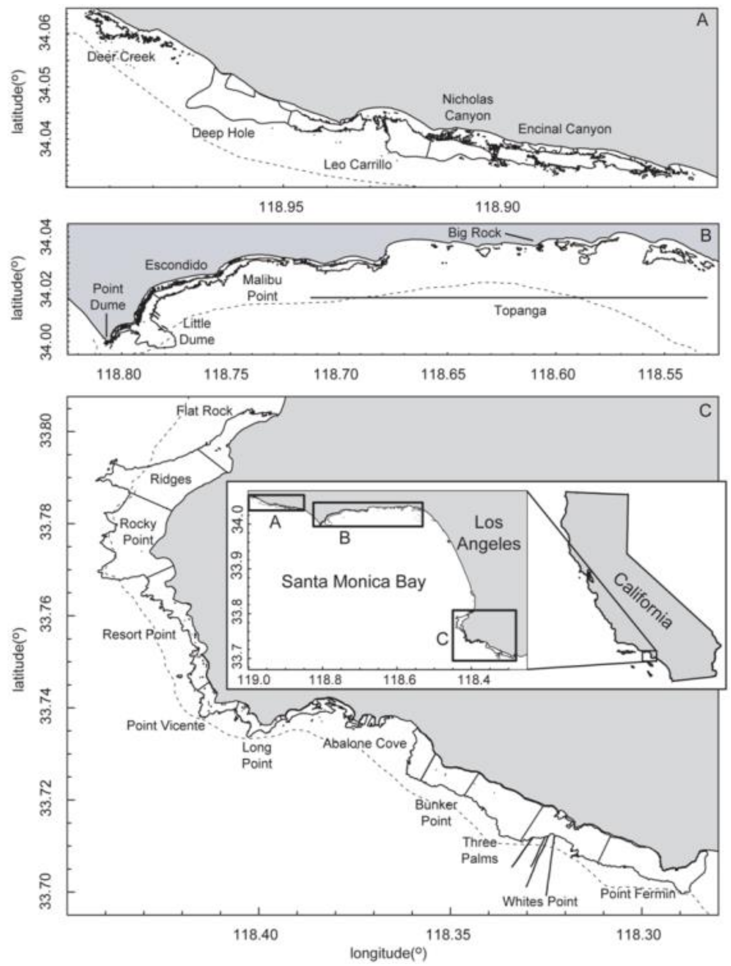


Figure 2-2. Extent of nearshore rocky reefs in Santa Monica Bay, California for (A) western Malibu, (B) eastern Malibu and (C) Palos Verdes Peninsula. The 30 m depth contour appears as a dotted line (Source: Claisse et al. 2012).

There are two submarine canyons in central and southern Santa Monica Bay: Redondo Canyon (off King Harbor, Redondo Beach, California) and Santa Monica Canyon, which is in the central portion of Santa Monica Bay upcoast from King Harbor (**Figure 2-3**). Santa Monica Canyon heads at a depth of about 55 m (180 ft) at a location about 5.6 km (3.5 mi) offshore, and the average gradient along the canyon axis is 3 percent (Terry et al. 1956). The head of Redondo Canyon is much closer to shore, and the gradient is much steeper at the head (8 percent). However, the average gradient throughout the rest of the canyon (4 percent) is similar to that of Santa Monica Canyon.

Wastewater from the City of Los Angeles is discharged into Santa Monica Bay from an ocean discharge that extends 8 km (5 mi) offshore from the Hyperion Treatment Plant (HTP), which is approximately 9 km (6 mi) upcoast from the WBDDF. The HTP has a design capacity of 1.7 million m^3 per day (450 mgd) of secondary-treated effluent. Up until the 1980s, the HTP

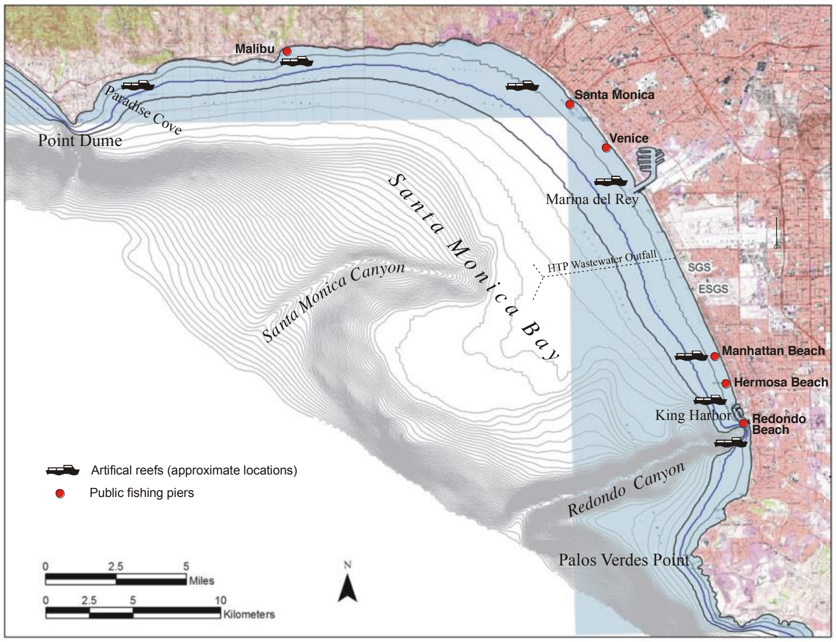


Figure 2-3. Seafloor bathymetry of Santa Monica Bay with the location of public fishing piers and artificial reefs.

discharged sludge through another discharge that extends 11 km (7 mi) from shore. That outfall is still in place but not used. A third sewage outfall extends 2 km (1 mi) from shore immediately upcoast from the Los Angeles Department of Water and Power Scattergood Generating Station (SGS), but is only used for emergency purposes.

Three coastal generating stations withdraw ocean water from Santa Monica Bay for cooling water purposes. The AES Redondo Beach Generating Station is located immediately adjacent and south of the WBDDF. It operates four natural gas units (Units 5, 6, 7, & 8). Plant 2 (Units 5 & 6) and Plant 3 (Units 7 & 8) utilize once-through cooling water systems with offshore intake and discharge systems (MBC and Tenera 2007). The total Plant 2 flow with all pumps operating is 0.8 million m^3 per day (216.5 mgd). The total Plant 3 flow with all pumps operating is 2.5 million m^3 per day (674.7 mgd). The El Segundo Generating Station, located approximately 7 km (4 mi) upcoast from the RBGS, operates two cooling water systems with a maximum permitted volume of 2.3 million m^3 per day (607 mgd) (Tenera and MBC 2008). The SGS, located approximately 8 km (5 mi) upcoast from the RBGS, withdraws up to 1.9 million m^3 per day (495 mgd) of cooling water from Santa Monica Bay through a single cooling water intake structure (MBC et al. 2008). A Chevron refinery also discharges about 22,710–26,495 m^3 (6–7 mgd) of treated effluent to Santa Monica Bay just downcoast from the SGS.

Two small-vessel harbors serve Santa Monica Bay: Marina del Rey and King Harbor. Fourteen artificial reefs designed to enhance marine life and provide sport fishing opportunities were installed off Malibu, Paradise Cove, Santa Monica, Marina del Rey, Manhattan Beach, Hermosa Beach, and Redondo Beach beginning in 1958; at least nine of these reefs remain (MBC 1993). Public piers are located at Malibu, Santa Monica, Venice, Manhattan Beach, Hermosa Beach, and Redondo Beach.

2.2.1.3 Temperature and Salinity

The salinity in the surface waters of the Southern California Bight is relatively constant (isohaline). According to Dailey et al. (1993) salinities in the nearshore peak in July at around 33.6 parts per thousand (ppt) and decrease in late winter and early spring to 33.4–33.5 ppt. Tides and temperatures are recorded at the NOAA station (Station ID: 9410840) located on the Santa Monica Pier 11.8 km (7.4 mi) northwest of the WBDDF (34° 0.5' N, 118° 30.0' W). In 2011, the sea temperatures ranged from a low of 12.0°C (53.6°F) in early March to 20.5°C (68.9°F) in late July (**Figure 2-4**). No data were available from this station from mid-April through mid-June due to equipment problems.

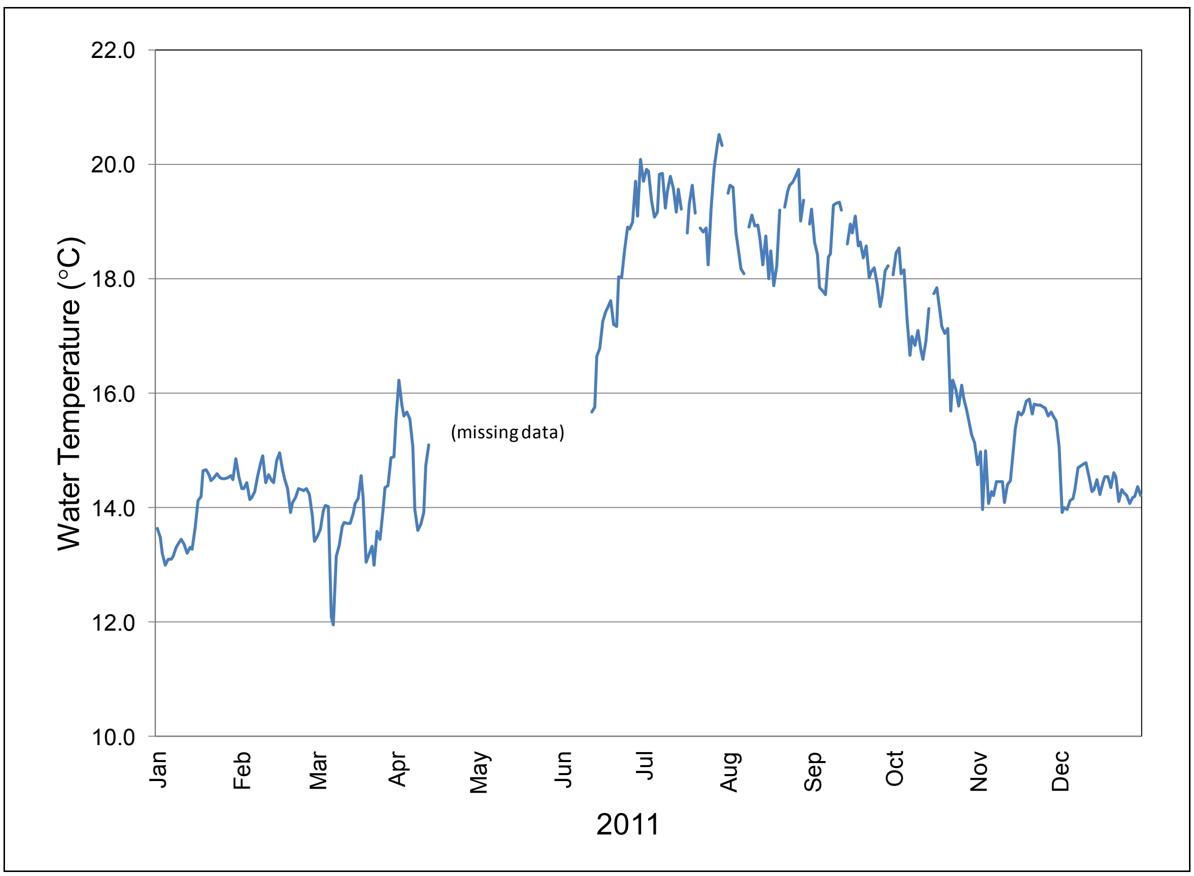


Figure 2-4. Average surface water temperatures at NOAA Station 9410840 at Santa Monica Pier, California from January through December, 2011.

In King Harbor, temperatures are normally slightly higher than in the surrounding Santa Monica Bay since the harbor is more isolated from nearshore currents and surf-induced turbulence. However, the volume and temperature of waters discharged from Units 7&8 have a strong influence on the temperature of the receiving waters within King Harbor (MBC and Tenera 2007). In winter and summer, water temperatures within the harbor can be similar to those in the nearshore waters of Santa Monica Bay if Units 7&8 are not operating. Strong thermoclines can develop in summer due to solar warming and the relatively shallow waters of King Harbor. Salinity is usually between 32.5 and 34.5 ppt in King Harbor, with values increasing with depth (MBC and Tenera 2007).

2.2.1.4 Currents and Tides

The prevailing current direction in the shallow, nearshore areas of Santa Monica Bay is downcoast (equatorward) suggesting an eddy-type circulation pattern resulting from the upcoast (poleward) currents outside of the Bay (Hendricks 1980). This description is supported by more extensive studies by Hickey (1992) that also showed downcoast currents on the shelf within the Bay and prevailing upcoast (poleward) currents at the edge of the shelf at the outer boundary of Santa Monica Bay. The circulation pattern within the Bay results from the presence of the Southern California Countercurrent in the outer coastal waters of the Southern California Bight. Hickey et al. (2003) found that subtidal currents in Santa Monica Bay are dominated by relatively long time scales (10–25 days), large alongshore scales, and significant offshore propagation. Large-scale remote forcing initially pushes water into the Bay as part of a throughflow, later becoming an eddy that produces counterflow in a typically southeastern direction along the Santa Monica Bay shoreline. However, currents shift in relation to upwelling events and other large-scale hydrographic processes along the coast (**Figure 2-5**) resulting in flow regimes that differ seasonally (**Figure 2-6**). Locally, water circulation near the WBDDF is affected by the presence of the constructed breakwater protecting King Harbor.

Hickey (1992) described the residence time of water within the Santa Monica and San Pedro basins using drifters and found that the residence time is both spatially and temporally variable as some drifters barely moved at all whereas others nearby moved large distances in the same period. Drifters deployed in January 1990 had drifted out of the Bay to the west in about a week. In July, residence times were only 3–5 days for drifters deployed anywhere over Santa Monica Basin. Drifters caught up by the Santa Monica Canyon eddy escaped the basin in less than one week, and most of the other drifters that were not cast ashore escaped the Bight in the ~2-week deployment period, roughly half passing north into the Santa Barbara Channel and half passing south of the Channel Islands.

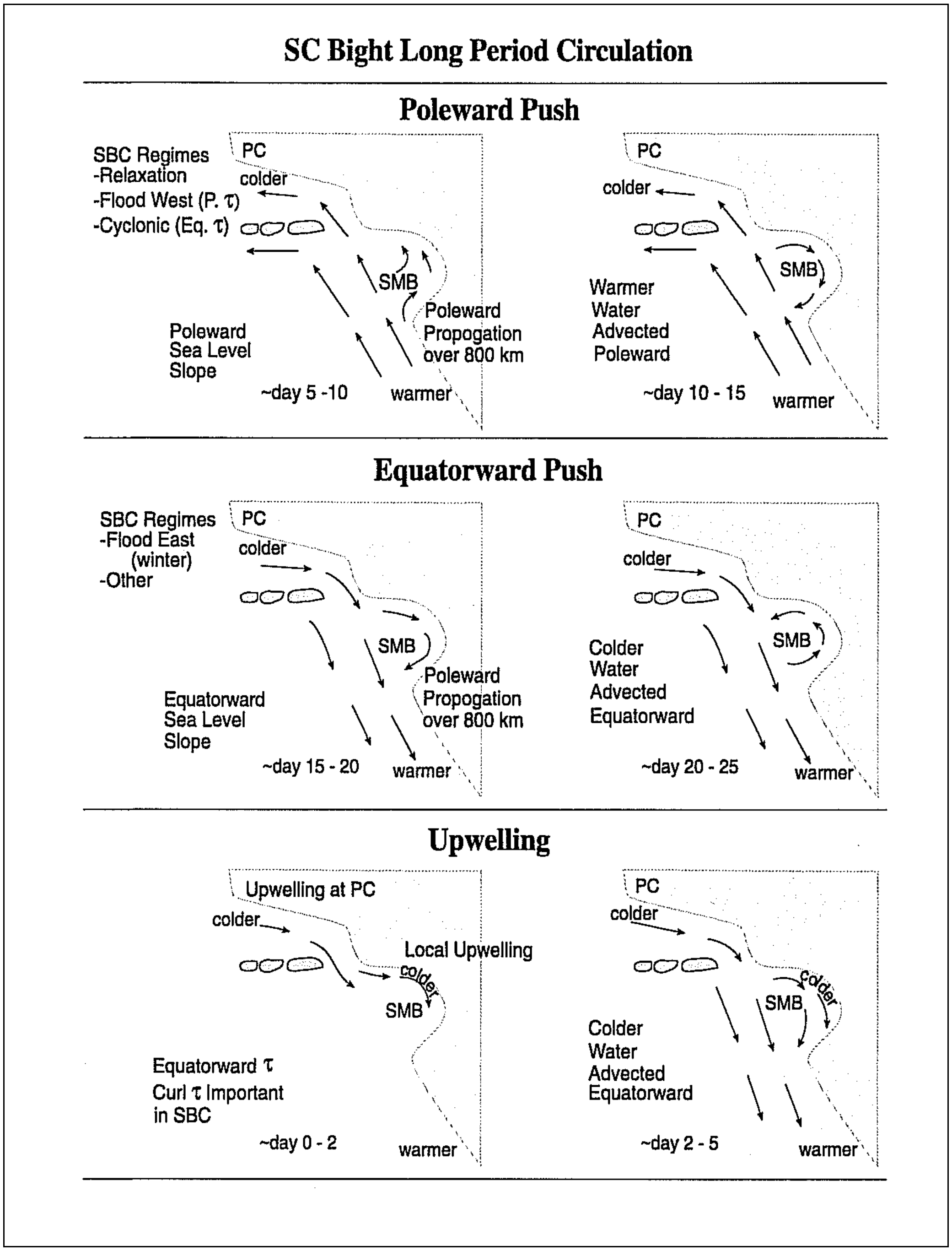


Figure 2-5. Schematic showing processes affecting long-period circulation and water properties in the Southern California Bight (from Hickey et al. 2003).

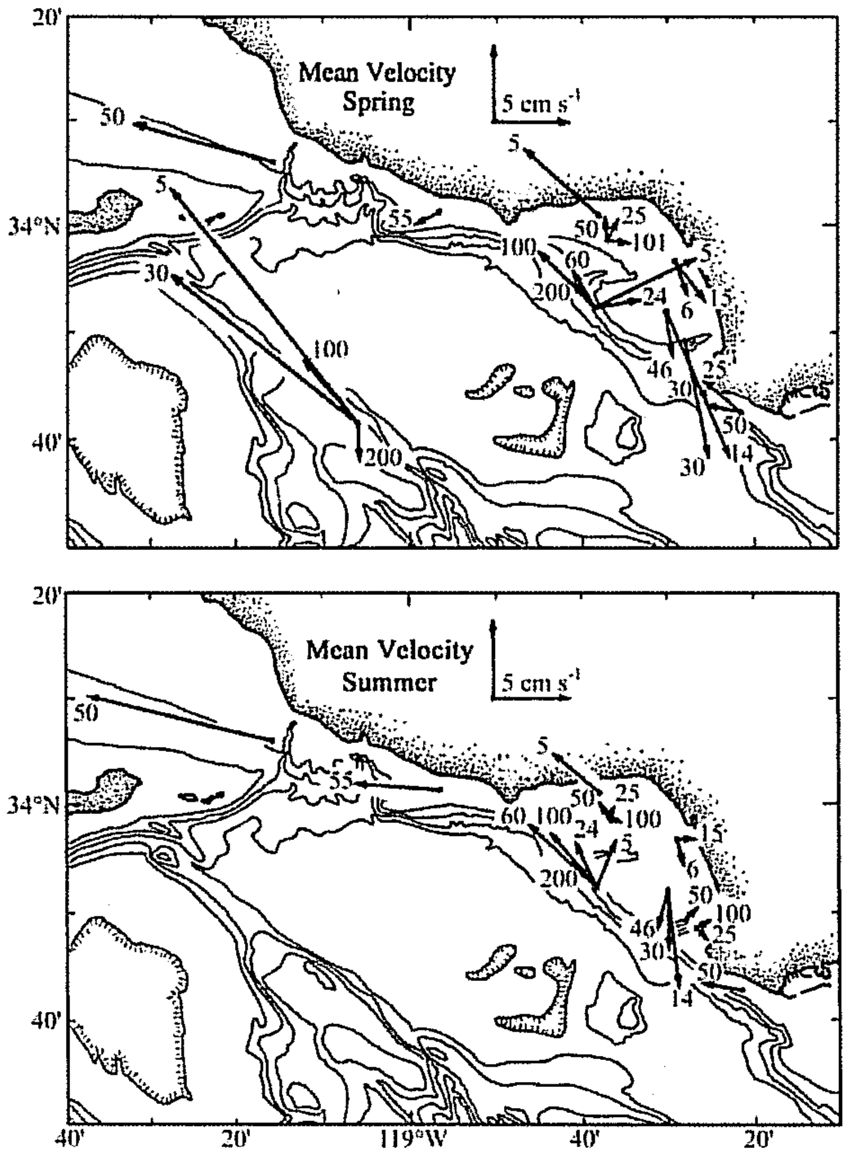


Figure 2-6. Selected mean currents in the central Southern California Bight for spring and summer. Measurement depth in meters is given near the tip of each arrow (from Hickey et al. 2003).

The CROSS oceanographic study deployed current meters in the Santa Monica Basin over bottoms as shallow as 30 m (100 ft) in Santa Monica Bay from October 1985 through January 1986 (Hickey 1992). Monthly mean velocities from three depths at the station closest to RBGS are presented in **Table 2-1**.

Table 2-1. Mean velocities (cm/s) in the vicinity of the WBDDF from Hickey (1992) from October 1985 through January 1986 at Station C1 at 30 m (100 ft) bottom depth.

	October		November		December		January		ALL	
Depth (m)	u*	v	u	v	u	v	u	v	u	v
5	-1.2	-8.8	-1.1	-7.9	-0.3	-1.7	-0.4	-1.3	-0.7	-4.5
10	1.3	-6.4	1.2	-5.2	1.0	-0.9	0.7	-0.5	1.0	-2.9
20	-0.4	-3.2	-0.5	-2.1	0.3	1.9	-0.1	0.6	-0.1	-0.3

*Note: u=across basin, v=along basin.

Recent efforts to assess currents and transport in the Southern California Bight include modeling and the use of high frequency (HF) radar observations of currents. Dong et al. (2009) presented results from an 8-year integration of the Regional Ocean Modeling System (ROMS) from 1996 to 2003 against a number of observational datasets including Hickey et al. (2003) and Noble et al. (2009). The model was able to simulate a realistic ocean environment on multi-year, interannual, seasonal, and eddy time scales (**Figure 2-7,** Idica 2010).

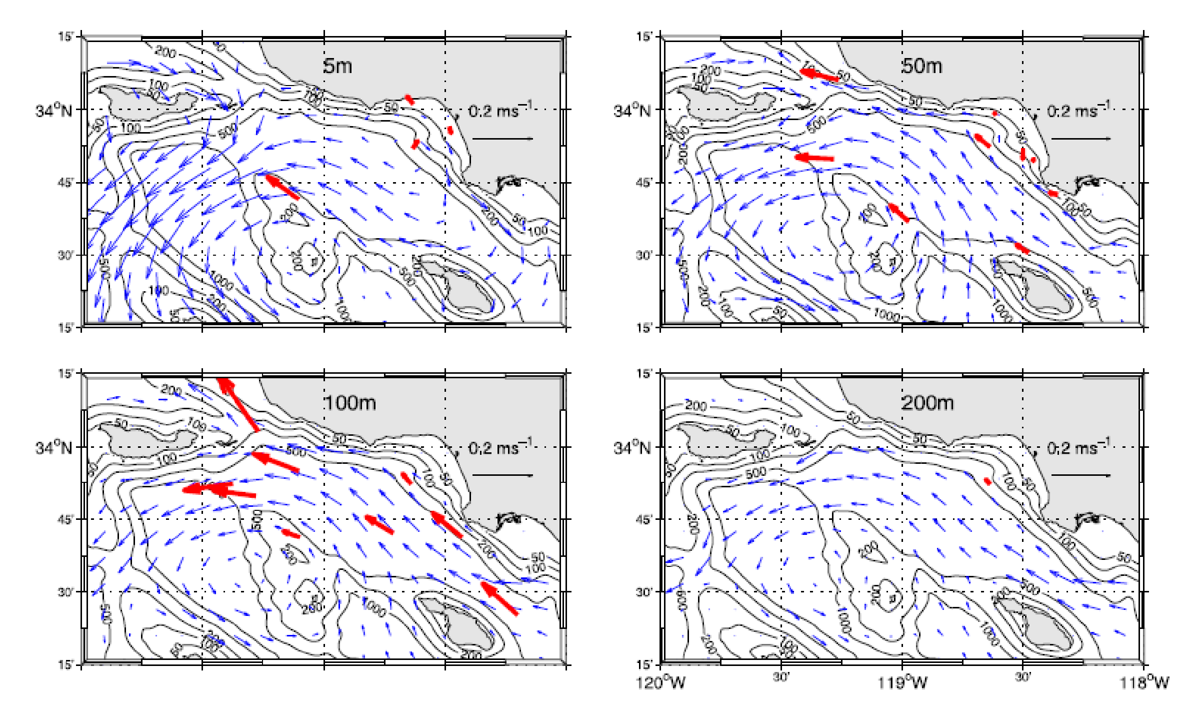


Figure 2-7. Comparison of observed and modeled mean-summer currents in the central Southern California Bight at four depths. The thinner and thicker arrows are the ROMS and observed vectors analyzed by Hickey et al. (2003), respectively. (Figure from Dong et al. 2009).

For the past few years the surface currents in the Southern California Bight have been measured on 2 km (1.2 mi) and 6 km (3.7 mi) resolutions using HF radar. The HF radar measurements of the speed and direction of the surface ocean waters were integrated with currents measured by a subsurface Acoustic Doppler Current Profiler (ADCP) in Santa Monica Bay, California to provide data for modeling the extent of the source water potentially entrained through the WBDDF seawater intake structure at Redondo Beach. Surface currents were measured hourly off the entire coastline of southern California by a network of Coastal Ocean Dynamics Applications Radar (CODAR) ocean sensors, and SeaSonde® HF radars available through the Southern California Coastal Ocean Observing System (SCCOOS) using the network of instruments deployed by the State of California's Coastal Ocean Currents Monitoring Program (COCMP). Nearshore sub-surface currents were measured by the ADCP deployed on the Santa Monica Bay Observatory oceanographic mooring (MUCLA), managed by University of California, Los Angeles' Institute of the Environment & Sustainability. The depth averaged currents were used in adjusting surface currents to the upper waters limited by 22 m (72.2 ft) depth. Currents at the MUCLA station were generally downcoast punctuated by eddy movements over June 2007 to March 2008 and November 2008 to April 2009 periods (**Figure 2-9**).

Tides in southern California are classified as mixed, semi-diurnal, with two unequal high tides (high water and higher high water) and two unequal low tides (low water and lower low water) each lunar day (approximately 24.5 hr). Predicted tides in 2011 at King Harbor, in the southern part of Santa Monica Bay, ranged from a high of 2.1 m (6.9 ft) MLLW to a low of -0.5 m (-1.6 ft) MLLW with a mean tide level of 0.84 m (2.74 ft) referenced to MLLW.[1]

[1] http://tidesandcurrents.noaa.gov

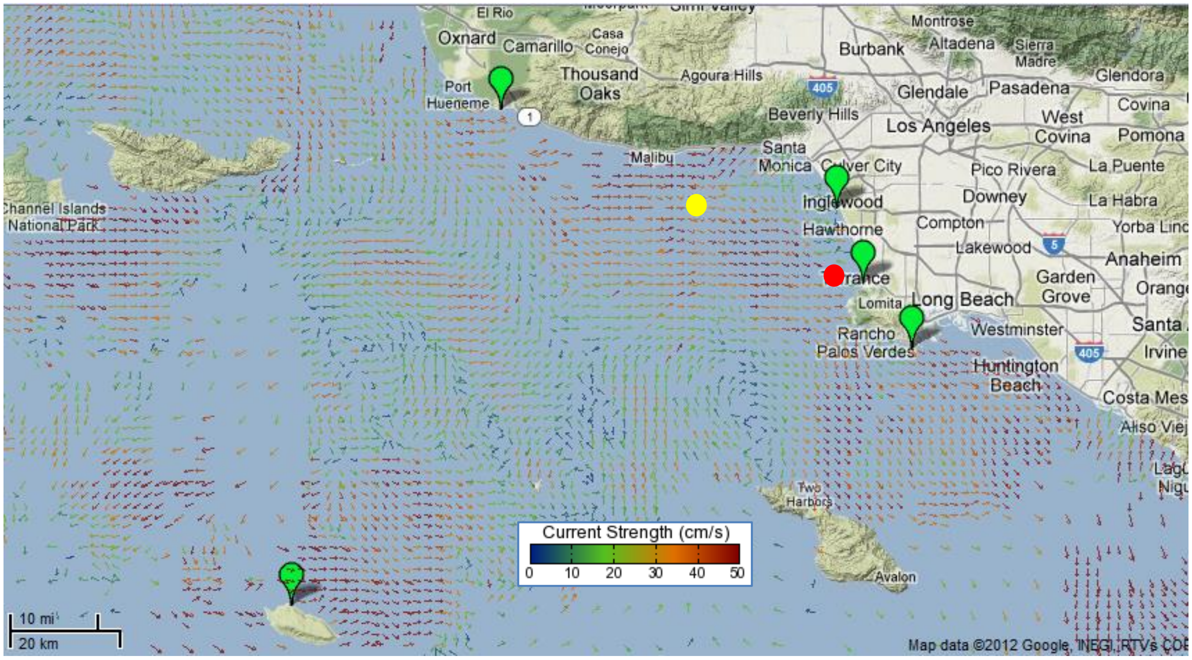


Figure 2-8. Ocean surface current vectors representative of data coverage throughout the study period; measured on October 1, 2011 at 0000 UTC in Santa Monica Bay, California. (Zalenke 2012). *Symbols*: CODAR SeaSonde stations (green markers), MUCLA ADCP (yellow marker), WBDDF intake structure (red marker). Vectors shaded according to their speed per the color-bar.

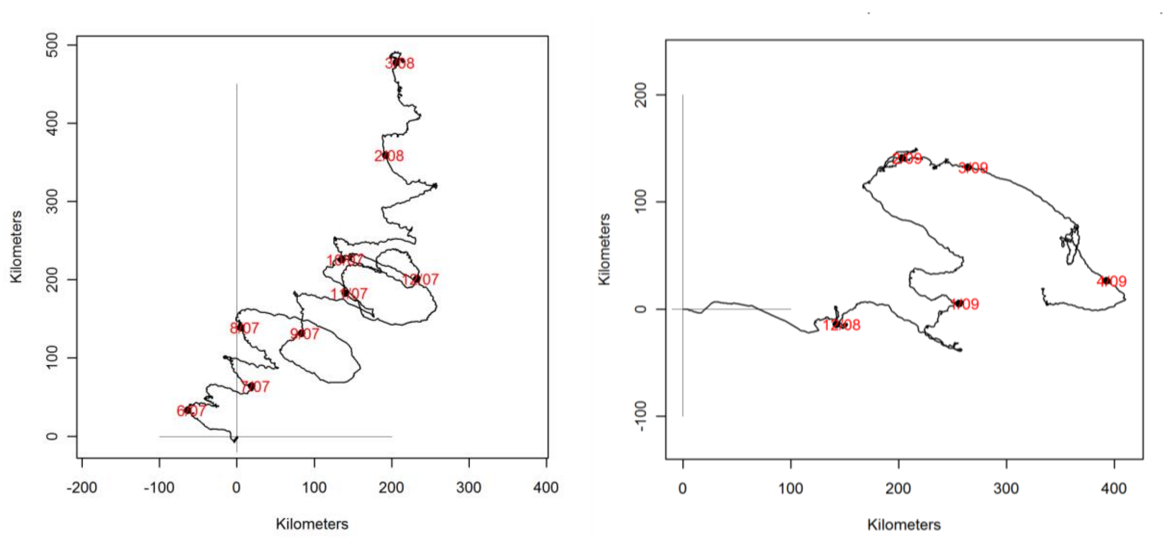


Figure 2-9. Progressive currents measured by an Acoustic Doppler Current Profiler (ADCP) at station MUCLA and averaged over a 6–22 m (19.7–72.2 ft) depth range. The ADCP hourly vectors were used to adjust surface currents measured by CODAR.

2.2.2 Biological Communities

The following sections describe the aquatic biological habitats and communities in the vicinity of the WBDDF, including both invertebrate and fish communities. Santa Monica Bay is the submerged portion of the Los Angeles Coastal Plain, and includes several types of marine habitat that support more than 5,000 species of plants and animals (SMBRC 2004). Most marine organisms within Santa Monica Bay and its watershed are temperate species with geographic ranges extending far beyond the immediate area. Most species are members of the San Diegan Province, which extends from Point Conception south to Magdalena Bay, Baja California Sur (Horn and Allen 1978). Fewer species belong to the Oregonian Province, which ranges from southern Canada to northern Baja California.

2.2.2.1 Habitat Variation

The pelagic habitat of Santa Monica Bay includes the entire water column within the Bay, a volume of approximately 6,840 billion gallons (MBC 1993). Organisms found in this habitat include a myriad of planktonic organisms (phytoplankton, zooplankton, and ichthyoplankton) that have little or no swimming ability to resist ocean currents, and nektonic organisms, such as fishes and sharks that are freely mobile in local and oceanic currents. The pelagic habitat also supports large numbers of pinnipeds (including Pacific harbor seal [*Phoca vitulina richardsi*] and California sea lion [*Zalophus californianus*]), cetaceans (such as gray whale [*Eschrichtius robustus*], bottlenose dolphin [*Tursiops truncatus*], and common dolphin [*Delphinus delphis*]), and birds, including California brown pelican (*Pelecanus occidentalis californicus*), terns, and gulls (MBC 1988).

Intertidal habitat within Santa Monica Bay includes both sandy and rocky habitats. Rocky intertidal habitat is comprised of both natural and artificial rocky substrate, such as the breakwaters at Marina del Rey and King Harbor. Natural rocky intertidal substrate occurs along the Malibu coast from Point Dume to Paradise Cove, along occasional patches from Paradise Cove to Big Rock Beach, and south along the Palos Verdes Peninsula.

Giant kelp beds occur on submerged rocky reefs in depths of about 6–21 m (20–70 ft). At present, kelp beds are limited to locations on the Palos Verdes Shelf, approximately 6 km (3.7 mi) south of King Harbor, and along Leo Carillo State Beach and the Malibu coast, approximately 30 km (18.6 mi) to the northwest (SMBRC 2004). Current canopy coverage is relatively low compared to historic coverage, but the extent of kelp is considered stable at Palos Verdes. The kelp beds in the Malibu area have increased in recent years, due in part to recent restoration efforts, improved water quality, and favorable oceanic conditions.

Most of the seafloor in Santa Monica Bay and King Harbor consists of unconsolidated (soft) sediments comprised of sand, silt, and clay. Most of the input to this habitat is in the form of detrital fallout and phytoplankton from the pelagic habitat, although detritus from surface runoff and discharged sewage may also be important (MBC 1988). A high proportion of soft-bottom benthos live most of their lives permanently in the sediments and are termed infauna; those which live on the surface of the seafloor are called epifauna. The soft-bottom habitat also supports several species of algae, macrofauna/megafauna (including crabs, snails, sea stars, urchins, and sea cucumbers), and fishes, including California halibut (*Paralichthys californicus*).

Along the shore of Santa Monica Bay are ten brackish wetlands of various sizes and conditions that contribute larval and adult forms of marsh fish and invertebrates, and vegetative organic production. The marshes range from small, seasonally-inundated river mouths (Zuma Beach west of Point Dume) to the larger Ballona Wetlands Complex at Marina del Rey. Historically, the Los Angeles River occasionally emptied into Santa Monica Bay at Ballona Creek instead of at its present-day mouth at Long Beach. The course of the River changed during unusually heavy storms from 1815–1825 and again in 1862 and 1884 (Terry et al. 1956). The area between Ballona Creek and present-day Beverly Hills was often a vast swamp. In 1868, the Ballona Wetlands comprised 8.5 km^2 (2,100 acres). Development of Marina del Rey, the Venice Canal system, residential and commercial properties, and the channelization of Ballona Creek reduced this area to less than 0.6 km^2 (160 acres) of wetland habitat (MBC 1993).

The wetlands at Ballona Creek, 14 km (8.7 mi) north of King Harbor, support a number of transient fish species but only nine residents (Swift and Frantz 1981). Dominant species include arrow goby (*Clevelandia ios*), mosquitofish (*Gambusia affinis*; a freshwater species), and topsmelt (*Atherinops affinis*). Numerous shorebirds, water fowl, and terrestrial birds are known to occur at Ballona wetlands, Marina del Rey, and Malibu Lagoon (MBC 1988).

There are no major freshwater rivers that empty into Santa Monica Bay, though there are some smaller streams. Small freshwater marshes occur at Malibu Lagoon and at Ballona Creek (MBC 1988). These marshes are home to numerous insects, amphibians, reptiles, and birds that live among the tules, cattails, and pond weeds (Jaeger and Smith 1966). Fresh water introduced by storm water and urban runoff has attracted increased attention in recent years. Control of pollutants from runoff has proven difficult due to the ubiquitous nature of the sources and current storm water regulations rely on compliance with best management practices instead of clearly defined effluent limits (SMBRC 2004).

Some of the important human uses of Santa Monica Bay that have directly and indirectly affected its ecology include sport and commercial fishing, industrial uses, and coastal development. Approximately 48 percent of Santa Monica Bay's watershed is characterized as developed (SMBRC 2004). Most of the remaining undeveloped area is located in the Santa Monica Mountains National Recreation Area. Since the 1950s, Santa Monica Bay has been closed to commercial fishing activities that use trawls, drag nets, gill nets, and traps, except for a small live-bait fishery that uses lampara nets (Schroeder and Love 2002). Sport fishing is allowed throughout Santa Monica Bay, and landings are currently operated out of Marina del Rey and King Harbor. Recreational fish are also caught by private boaters, from shore, and from piers. Several artificial reefs were constructed in Santa Monica Bay beginning in 1958 to enhance marine life and fishing opportunities (MBC 1988).

Industrial uses of Santa Monica Bay include cooling water supply, transport and refinery of oil/gas products, and waste disposal. Both the Joint Water Pollution Control Plant (JWPCP) and the HTP discharge treated wastewater to Santa Monica Bay. These facilities achieved full secondary treatment in 1998 for HTP and late 2002 for JWPCP (SMBRC 2004). Since 1971 there has been a steady decrease of contaminant inputs from these facilities to the Bay. Still, Santa Monica Bay is listed as a Section 303(d) impaired waterbody mainly due to sediment contamination (PCBs, DDT, sediment toxicity) resulting from the historic discharge of wastewater and sludge.[2]

2.2.2.2 Nursery Grounds

It is unknown to what extent Santa Monica Bay serves as a nursery for fish and invertebrate species; however, it can be assumed that the variety of habitat types within the Bay are likely used by numerous species for such purposes. On the open coast, recruitment to the mainland shelf occurs year-round, but is greatest from winter to spring (Cross and Allen 1993). The rocky intertidal zone is a turbulent and dynamic environment, and in southern California there are only a handful of fish species considered residents of this habitat, including some sculpins (Cottidae) and pricklebacks (Stichaeidae). Most resident intertidal fishes lay demersal rather than planktonic eggs, and parental care is relatively high (Horn and Martin 2006). The larvae of most intertidal fishes spend about one to two months in the plankton, but disperse only short distances and tend to stay within the area in which they were hatched.

Reefs and kelp beds provide habitat for a wide variety of fishes and invertebrates. The nearshore is the primary starting and ending point for larval dispersal of coastal species, and one where cross-shelf and along-shelf flow change rapidly over short distances owing to strong interactions of frictional forces of coastal topography, stratified water columns, tidal forces, wind, buoyancy, surface waves, and turbulence (Largier 2003). In other species, larvae produced on a reef may have behavioral mechanisms to retard drift processes, keeping them in the local area for settlement (Stephens et al. 2006). Pondella et al. (2002) determined densities of adult black perch (*Embiotoca jacksoni*) and pile perch (*Rhacochilus vacca*) remained stable during the period 1974–1998, indicative of a mature reef community. However, during this same time period, density of sub-adults for both species decreased at King Harbor and nearby Rancho Palos Verdes, while density of juveniles was relatively constant. Overall, surfperch production at King Harbor (an artificial reef) was higher than that at Rancho Palos Verdes (a natural reference reef). From 1974–1993, the number of young-of-the-year (YOY) fish of all species that recruited annually to King Harbor and Palos Verdes was highly correlated with the annual biomass of macrozooplankton in the California Current, which is an indicator of Bight-wide productivity (Holbrook et al. 1997). Furthermore, the relationships between macrozooplankton biomass and annual recruitment did not differ between surfperches (live-bearers) and fishes that disperse larvae during spawning.

[2] http://www.waterboards.ca.gov/rwqcb4/water_issues/programs/regional_program/wmi/ws_santamonica.shtml.

On the soft-bottom substrata of the southern California mainland shelf, Allen (1982) found that 45 percent of the 40 major fish community members had pelagic eggs and larvae, 18 percent (all rockfishes) were ovoviviparous with pelagic larvae, 15 percent had demersal eggs and pelagic larvae (such as combfishes, sculpins, and poachers), 12 percent were viviparous (bearing live young—all surfperches), and 10 percent had demersal eggs and larvae (including midshipman and eelpouts). Southern California is located at the edge of the geographic range of many cool- and warm-water fish species, and recruitment of juveniles is episodic and species dependent (Allen 2006). As a result, coastal settlement is more variable than in bays, and interannual variation is probably primarily due to oceanic conditions that affect transport and survival of larvae, along with spawning success and availability of suitable benthic habitat for settling juveniles. In 1989, Allen and Herbinson (1991) surveyed bay, open coast, and protected coastal habitats in southern California with fine-mesh beam trawls. In general, fish densities were higher in bays than on the open coast, densities decreased with increasing depth, and highest densities were recorded in spring (May). On the inner shelf (6–15 m, [20–49 ft]), speckled sanddab (*Citharichthys stigmaeus*) was the most frequent juvenile fish taxa encountered, but queenfish (*Seriphus politus*) was most abundant.

2.2.2.3 Fish Diversity (all Life Stages)

From 1991 through 2006, at least 103 distinct fish species were impinged during normal operations and heat treatments at the RBGS (MBC and Tenera 2007). The most abundant fishes collected were Pacific sardine (S*ardinops sagax*), blacksmith (*Chromis punctipinnis*), and queenfish. These three species combined accounted for 62 percent of annual impingement abundance. Nearly 93 percent of the Pacific sardine were impinged during 1992 and 1993, and their abundance declined dramatically thereafter. An average of about 49 fish species are impinged annually. In 2005, the most abundant species were shiner perch (*Cymatogaster aggregata*), queenfish, and California scorpionfish (*Scorpaena guttata*).

McGowen (1978) recorded 46 egg types from King Harbor and surrounding waters in 1974 and 1975, although 81 to 87 percent of those were unidentifiable. In winter, the most abundant larvae were from sculpins and croakers (Sciaenidae), while gobies (Gobiidae) were dominant in spring. In summer, the most abundant larvae were those of combtooth blennies (*Hypsoblennius* spp.), clinid kelpfishes (Clinidae), gobies, and croakers. Highest densities of combtooth blennies, unidentified gobies, and snubnose pipefish (formerly *Bryx arctus*, now *Cosmocampus arctus*) larvae occurred at stations furthest inside King Harbor, while the percentage of larvae which hatched from pelagic eggs increased with distance from the inner reaches of the harbor. In total, more than 90 percent of the larval fishes of the surface waters of King Harbor and nearby Santa Monica Bay were found to belong to just five fish families: Blenniidae (blennies), Clinidae (kelpfishes), Gobiidae (gobies), Engraulidae (anchovies), and Sciaenidae (croakers).

2.2.2.4 Shellfish Diversity (all Life Stages)

In 2005, 27 macroinvertebrate taxa were collected in impingement samples at the RBGS (MBC 2006). The most abundant taxa were unidentified moon jelly (*Aurelia* sp.), purple-striped jelly (*Chrysaora colorata*), and red rock shrimp (*Lysmata californica*). Impingement abundance was slightly higher at Units 7&8 (59 percent) than at Units 5&6 (41 percent). However, biomass was substantially greater at Units 7&8 (198 kg; 436.5 lb) than at Units 5&6 (52 kg; 114.6 lb), due primarily to the higher number of larger individuals of purple-striped jelly, California spiny lobster (*Panulirus interruptus*), and sheep crab (*Loxorhynchus grandis*).

2.2.2.5 Protected Species

No federal/state threatened or endangered fish/shellfish species have been identified in entrainment and impingement samples collected at the nearby RBGS. This is consistent with past entrainment and impingement sampling conducted at the RBGS.

Off southern California, the species with designated Essential Fish Habitat (EFH) are listed in the Coastal Pelagics Fishery Management Plan (FMP) and Pacific Groundfish FMP. The goals of the management plans include, but are not limited to: the promotion of an efficient and profitable fishery, achievement of optimal yield, provision of adequate forage for dependent species, prevention of overfishing, and development of long-term research plans. There are four fish and one invertebrate species covered under the Coastal Pelagics FMP: northern anchovy, Pacific sardine, jack mackerel (*Trachurus symmetricus*), Pacific chub mackerel (*Scomber japonicus*), and market squid (*Doryteuthis opalescens*). There are 89 fish species covered under the Pacific Groundfish FMP: three species of sharks, three species of skates, six species of roundfish, 62 species of scorpionfishes and thornyheads, 12 species of flatfishes, and one species each of ratfish, Morids, and grenadiers. For both the Coastal Pelagics and Pacific Groundfish, EFH includes all the waters off southern California offshore to the Exclusive Economic Zone. A list of species covered under the two FMPs that occurred during entrainment and impingement sampling at the RBGS is provided in **Table 2-2**.

Some fish and invertebrate species (abalone) in southern California are protected under California Department of Fish and Wildlife (CDFW) (formally known as California Department of Fish and Game) regulations although few marine species are listed as either threatened or endangered. Special-status species that occur in the vicinity of the RBGS and that have planktonic larvae potentially at risk of entrainment include garibaldi (*Hypsypops rubicundus*), giant sea bass (*Stereolepis gigas*), tidewater goby (*Eucyclogobius newberryi*), and California grunion (*Leuresthes tenuis*).

Table 2-2. Fish and shellfish species with designated EFH or CDFW special-status species entrained and/or impinged at the RBGS in 2006.

Species	Common Name	Management Group
Engraulis mordax	northern anchovy	Coastal Pelagics
Doryteuthis opalescens	market squid	Coastal Pelagics
Sardinops sagax	Pacific sardine	Coastal Pelagics
Hypsypops rubicundus	garibaldi	CDFW
Leuresthes tenuis	California grunion	CDFW
Citharichthys sordidus	Pacific sanddab	Pacific Groundfish
Merluccius productus	Pacific hake	Pacific Groundfish
Microstomus pacificus	Dover sole	Pacific Groundfish
Parophrys vetulus	English sole	Pacific Groundfish
Platichthys stellatus	starry flounder	Pacific Groundfish
Scorpaena guttata	Calif. scorpionfish	Pacific Groundfish
Scorpaenichthys marmoratus	cabezon	Pacific Groundfish
Sebastes spp.	rockfishes	Pacific Groundfish
Sebastes auriculatus	brown rockfish	Pacific Groundfish
Sebastes miniatus	vermilion rockfish	Pacific Groundfish

The garibaldi, designated as the California state marine fish, is a bright orange shallow-water species that is relatively common around natural and artificial rock reefs in southern California. Because of its territorial behavior it is an easy target for fishers and could be significantly depleted if not protected. Garibaldi spawn from March through October, and the female deposits demersal adhesive eggs in a nest that may contain up to 190,000 eggs deposited by several females (Fitch and Lavenberg 1975). Larval duration ranges from 18–22 days (mean of 20 days) based on daily incremental marks on otoliths in recently settled individuals (Wellington and Victor 1989). The larvae are susceptible to entrainment, particularly in summer months when spawning is at its peak.

The giant sea bass is a long-lived species that can grow to over 2.1 m (7 ft) in length and weigh over 227 kg (500 lb) (Love 2011). Giant sea bass were once a relatively common inhabitant of southern California waters, yet in the 1980s it was facing the threat of local extinction off the California coast due to overfishing. Actions were taken by CDFW, resulting in protection from commercial and sport fishing that went into effect in 1982. Although the larvae are potentially susceptible to entrainment from coastally-sited power plants in southern California, no giant sea bass larvae have been identified from entrainment samples.

The tidewater goby is a fish species endemic to California and is listed as federally endangered. The tidewater goby is threatened by modification and loss of habitat resulting primarily from coastal development. It appears to spend all life stages in lagoons, estuaries, and river mouths (Swift et al. 1989) but may enter marine environments when flushed out of these preferred habitats during storm events. Adults or larvae may not survive for long periods in the marine environment but larval transport over short distances may be a natural mechanism for local dispersal.

California grunion is a special-status species not because the population is threatened or endangered, but because their spring-summer spawning activity on southern California beaches puts them at risk of overharvesting, and CDFW actively manages the fishery to ensure sustainability. Spawning occurs only three or four nights following each full or new moon, and then only for 1–3 hours immediately after the high tide, from late February to early September (Love 2011). The female swims onto the beach, digs tail-first into the wet sand, and releases eggs that are then fertilized by the male. After the eggs hatch in two weeks, the larvae are carried offshore and can be susceptible to entrainment for approximately 30 days as they develop in the plankton.

3.0 Offshore Intake and Source Water Study

3.1 Introduction

The purpose of the sampling associated with this part of the study was to determine the extent of potential impacts from entrainment of fish, fish eggs, and selected invertebrate larvae (target organisms) due to the operation of an open ocean unscreened intake. Entrainment refers to the withdrawal of aquatic organisms from the source water into and through the feedwater intake system of the desalination facility. To determine the potential effects of the proposed intake on these stages of the target organisms, data from the 2011–2012 study were analyzed using three intake flow volumes; the average daily flow during the study period (1,309 m^3 [0.35 mgd]), the design flow for the demonstration project (1,935 m^3 [0.511 mgd]), and the proposed flow for the full-scale plant (170,722 m^3 [45.1 mgd]).

3.1.1 Species to be Analyzed

A diverse array of planktonic organisms are susceptible to entrainment. The intent of this study was to estimate entrainment effects on several types of organisms: fish eggs and larvae, and late-stage larvae of target invertebrates (Cancer crab megalops, California spiny lobster phyllosoma, and market squid paralarvae). All of the specimens from these groups were quantified and then extrapolated to numbers representing potential losses from the volumes of seawater withdrawn for the desalination facility feedwater. A more detailed population-level assessment of potential entrainment effects was limited to the most abundant and most frequently occurring larval fish taxa in the samples, as well as Cancer crab megalops larvae.

3.2 Methods

3.2.1 Field Sampling for the Baseline Characterization Study

Plankton samples were collected monthly over a 12-month study period to provide a baseline characterization of the potentially entrainable fish larvae, fish eggs, and selected invertebrate larvae at the proposed intake location and in the surrounding source water. Plankton net sample collection methods were similar to those developed and used by the California Cooperative Oceanic and Fisheries Investigation (CalCOFI) in their long-term studies of larval fishes in the California Current (Smith and Richardson 1977) and subsequently used in nearshore source water studies at the RBGS (MBC and Tenera 2007) and at several other coastal locations in California. Sampling for the 12-month baseline characterization study was initiated on March 31, 2011 and completed on March 5, 2012.

Towed plankton net samples were collected from a 22-ft vessel at four stations in the vicinity of the proposed intake (**Figure 3-1**). Weather and sea state conditions affected the timing of the start of monthly sampling, so the intervals between surveys varied. The ‘entrainment’ station

(SWE) was located at the site of the WBDDF intake, and three 'source water' stations were located downcoast (SW1), upcoast (SW2) and offshore (SW3) from Station SWE. Station , SW2 was 0.51 km (0.32 mi) to the north, and SW3 was 0.65 km (0.40 mi) to the west of Station SWE. The water depth at stations SWE, SW1, and SW2 was approximately 10.2 m (33.5 ft) MLLW, and SW3 was approximately 18.3 m (60 ft) MLLW. The stations were located to provide a characterization of the habitats in the vicinity of the WBDDF intake. Station SW1, 0.49 km (0.30 mi) downcoast to the south, was positioned to characterize the larvae potentially produced in the rocky habitat along the King Harbor breakwaters. Station SW2, 0.51 km (0.32 mi) upcoast to the north, was positioned to characterize the larvae potentially produced along the sandy beach areas characteristic of other areas upcoast in Santa Monica Bay. Station SW3, 0.65 km (0.40 mi) offshore to the west of Station SWE, was positioned to characterize larvae produced from fishes in slightly deeper waters that might be transported onshore to the intake.

Figure 3-1. Location of intake and source water stations for plankton sampling. Depth contours are at 10 m (33 ft) MLLW intervals.

The samples were collected monthly at each of the four stations (**Figure 3-1**). Weather and sea state conditions affected the timing of the start of monthly sampling, so the intervals between surveys varied. Samples were collected from each station twice per survey, once during the day (day cycle) and once at night (night cycle), to characterize potential diel variation in larval abundances. The day cycle typically started between 1200 and 1500 PST and sampling for the night cycle started between 2300 and 0100 PST. The exact time when sampling was initiated during a cycle varied during the study period due to changing diurnal patterns and the scheduling of other sampling tasks. Five towed plankton net samples were collected during each cycle; two replicate samples at the intake location (SWE), and one sample at each of the three source water stations (SW1, SW2, and SW3).

Samples were collected by towing a bongo frame with two 0.71 m (2.3 ft) diameter openings, each equipped with a 335-µm (0.13 in.) mesh plankton net, a codend at the end of the net, and flowmeter (**Figure 3-2**). The water volume filtered through the nets was measured using a calibrated flowmeter (General Oceanics Model 2030R) mounted in the center of the opening of each net frame. The target volume for each sample was at least 40 m^3 (10,567 gal) of seawater per net. Samples were collected using oblique tows, where the nets were lowered from the surface to a depth within approximately 1 m (3.3 ft) from the bottom and towed back to the surface at a speed of 0.5−1.0 m/sec (1.6–3.3 ft/sec) using a hydraulic winch. From two and six repetitions were required (depending on water depth) to reach or exceed the target volume (based on the flowmeter readings). Prior to, and after, each tow, the flowmeter counter values were recorded on sequenced waterproof datasheets to allow a calculation of the volume of water filtered by each net.

Figure 3-2. Bongo frame and plankton nets used to collect fish and shellfish larvae at the intake and in the source water.

At the completion of each tow, the frame and nets were retrieved from the water and all of the collected material was rinsed into the codend collection container. The contents of each net were combined into a single sample, transferred into a sample jar with internal and external labeling, and preserved in a 5−10 percent solution of buffered formalin. When a large volume of filtered material was collected, the content of the sample was divided into two or more jars that were labeled as sample fractions. Each sample was labeled with information on the survey, station, cycle, sample, and number of jars in the sample. The labeling allowed each sample to be tracked through laboratory processing, data analysis, and reporting.

Field data were recorded on preprinted data sheets formatted for entry into a computer database for analysis and archiving. All field data were recorded on sequenced data sheets and were entered into a Microsoft Access® computer database that was verified for accuracy against the original data sheets.

A Quality Assurance/Quality Control (QA/QC) program was implemented for the field component of the study. The field survey procedures were reviewed with all personnel prior to the start of the study, and all field staff were given printed copies of the procedures. In addition to ongoing training and periodic review of sampling procedures, quality control assessments were completed during the study to ensure that the field sampling continued to be conducted properly.

3.2.2 Laboratory Analysis

Samples from the field were returned to the laboratory, and after at least 72 hours, transferred into a solution of 70–80 percent ethanol preservative. Processing consisted of examining the collected material under a dissecting microscope and removing and counting fish larvae and fish eggs, market squid paralarvae, the megalopal stage of Cancer crabs, and the phyllosome stage of California spiny lobster. These target taxa conformed to the taxa enumerated in similar studies completed at other locations in California. The organisms were placed in labeled vials and then identified to the lowest practical taxonomic level. The developmental stage of fish larvae (yolk-sac, preflexion, flexion, postflexion, transformation) was also recorded.

When samples were particularly dense, a Folsom Plankton Splitter was used to divide samples into smaller, more manageable subsamples. Aliquot portions of ½ and ¼ of the original sample were obtained and sorted. When original samples were estimated to contain greater than 500 fish eggs a 10 percent aliquot was removed from the sample and the eggs were removed from that portion. Fish specimens that could not be identified to the species level were identified to the lowest taxonomic classification possible (e.g., genus and species are lower levels of classification than order or family). Crab megalops were generally identified to combination categories because of overlapping size ranges between species. Myomere and pigmentation patterns were used to identify the larval fishes, however, this can be problematic for some species. For example, several species of the Gobiidae family of fishes share similar characteristics during early life stages, making identification to the species level uncertain (Moser 1996).

Notochord length was determined for a representative number of individuals of larval fish taxa. These measurements were determined and recorded for up to 50 specimens from each taxon during each survey using a video capture system and image analysis software. These data were used to determine the duration at risk of entrainment for the larvae and for screening efficiency studies. Individuals longer than 30 mm (1.18 in.) in length were considered non-entrainable because they would not normally be susceptible to entrainment due to their size. Fish that were considered non-entrainable were not used in the determination of the annual entrainment estimate.

Laboratory data were recorded on preprinted data sheets formatted for entry into a computer database for analysis and archiving. All laboratory data were recorded on sequenced data sheets entered into an Access® computer database, then verified for accuracy against the original data sheets.

3.2.3 Quality Control Program

A detailed QA/QC program was also applied to all laboratory processing. Sample sorting procedures were reviewed with all personnel prior to the start of the study and all personnel were given printed copies of the procedures. The first ten samples sorted by an individual were re-sorted by a designated QC sorter. In order to pass a QC re-sort, a sorter was allowed to miss only one fish larva in a sample when the total number of larvae was less than 20. For samples with 20 or more larvae, the sorter was required to maintain a sorting accuracy of 90 percent. After a sorter sorted ten consecutive samples with greater than 90 percent accuracy, the sorter had one of their next ten samples randomly selected for a QC check. If the sorter failed to achieve an accuracy level of 90 percent, their samples were re-sorted by the QC sorter until ten consecutive samples met the required level of accuracy. If the sorter maintained the required level of accuracy, one of their next ten samples was re-sorted by QC personnel.

A similar QA/QC program was conducted for the taxonomists identifying the samples. The first ten samples of fish identified by an individual taxonomist were re-identified by a second taxonomist. Taxonomic precision was calculated as Percent Taxonomic Disagreement (PTD) and served to quantify the rates of error (percent disagreement) in assignment of nomenclature to individual specimens in the sample. The error rate was quantified as the proportion of individual specimens in the sample identified differently (PTD) by the two taxonomists calculated as:

$$PTD = \left[\frac{1-comp_{pos}}{N}\right] \times 100,$$

where: $comp_{pos}$ is the number of agreements and N is the total number of organisms in the larger of the two counts. A PTD goal of less than or equal to 10 percent was targeted. The lower the PTD value, the greater the overall taxonomic precision indicating relative consistency in sample treatment. If the PTD goal of 10 percent was not attained, taxonomist interaction was used to determine problem areas, identify consistent disagreements, and define corrective actions. Additionally, the next ten consecutive samples the original taxonomist identified were checked by the second taxonomist for accuracy. Samples were re-identified until ten consecutive samples met the PTD criterion. Identifications were verified with taxonomic voucher collections maintained by Tenera.

3.2.4 Data Analysis

The following sections describe how the data were processed and analyzed.

3.2.4.1 Entrainment Estimates

Entrainment estimates were calculated using the daily average larval concentrations from the field samples and the average daily flow during the study period (1,309 m^3/day [0.348 mgd]), the design flow for the pilot project (1,935 m^3/day [0.511 mgd]), and the proposed flow for the full-scale plant (170,722 m^3/day [45.1 mgd]) for the desalination intake. The estimates of the daily average concentrations and associated variance were calculated based on a stratified random design with two cycles (day and night) and two replicates per cycle. The estimates of larval entrainment at the WBDDF offshore intake were based on monthly sampling where E_T is the estimate of total entrainment for the study period and E_i is the monthly entrainment estimate calculated from the average concentration calculated for the day sampled within each survey period. The average concentration and variance calculated for the day was extrapolated across the days within each sampling period. Estimates of proportional mortality using the ETM were also calculated using daily entrainment estimates calculated using daily unscreened intake flows of 1,309 m^3/day [0.346 mgd], 1,935 m^3/day [0.511 mgd], and 170,722 m^3/day [45.1 mgd].

3.2.4.2 Estimates from Source Water Stations

Estimates of the population of larvae withinthe source water were calculated using larval concentrations from field samples collected at the WBDDF intake location (SWE) and the three source water stations (SW1–SW3). Estimates of the average number of larvae in each of the source water areas during the day that sampling occurred were calculated from the monthly sampling using the average concentrations from the samples collected during the two temporal cycles at each station. The associated variance for the survey was calculated using data from the two cycles. An estimate of the larval concentration in the adjacent areas: 1) source water offshore northwest (SWONW) and 2) source water offshore southwest (SWOSW) were interpolated using the average of the concentrations at the three other adjoining stations (SW1–SW3) (**Figure 3-3**). This was done to allow for a rectangular-shaped source water area that could be extrapolated using alongshore current displacement, otherwise the layout of the sampling locations would have required separate source water estimates for the offshore (SWONW and SWOSW) station areas. The estimates of the daily concentration for the stations were multiplied by the volume of each station area (**Table 3-1**), calculated from bathymetric data, to calculate the population for the day. The estimates from the source water stations were combined to provide an estimate of the total number in the source water sampling area that was then extrapolated to estimate the entire source population at risk to entrainment.

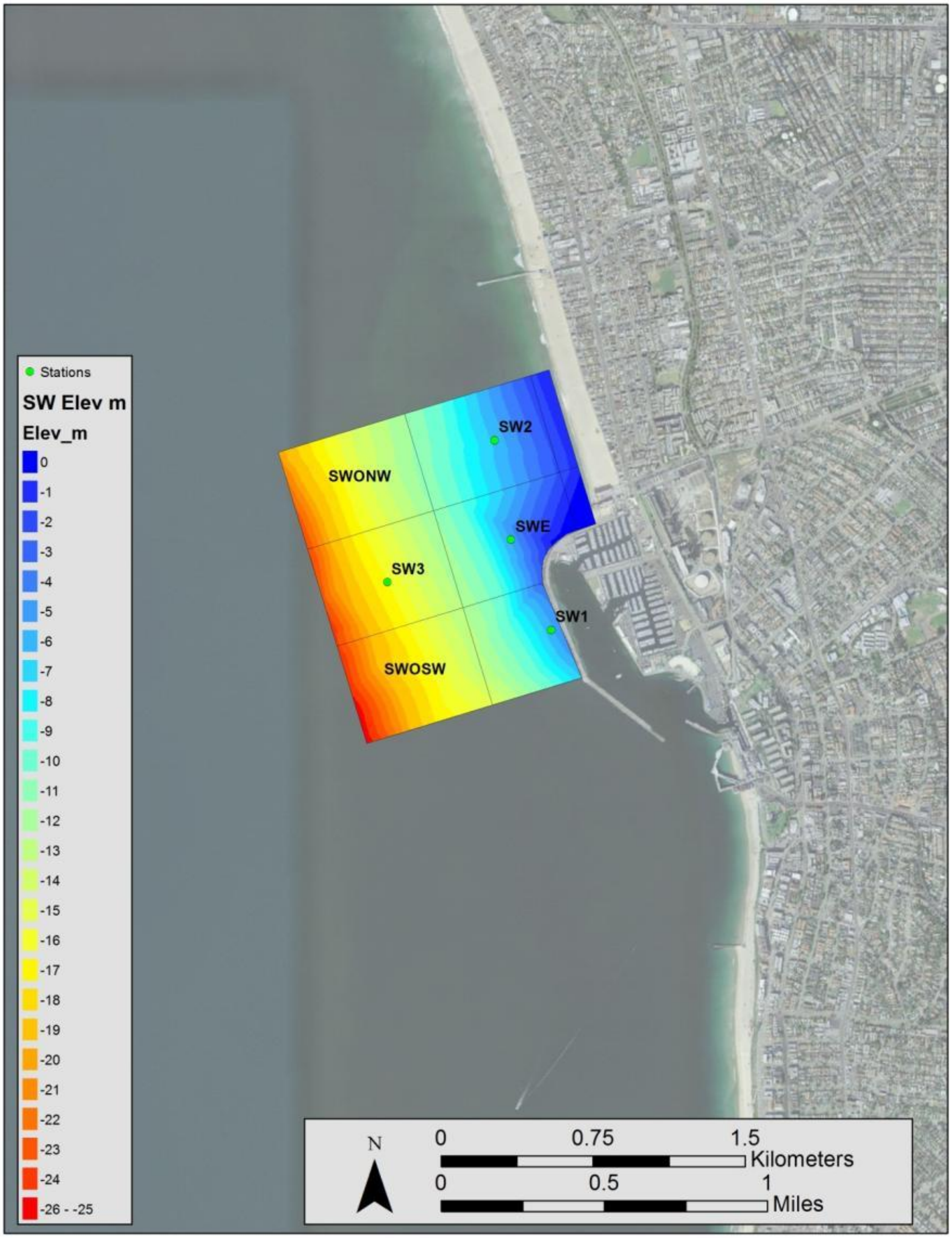


Figure 3-3. Bathymetry and areas used in calculating source water volumes for ETM calculations.

Table 3-1. Area, volume, and average depth of WBDDF sampling locations, including the values for the two extrapolated source water areas, SWONW and SWOSW.

Station	Area (m^2)	Volume (m^3)	Average Depth (m)
SWE	316,210	2,076,142	6.76
SW1	218,525	2,160,975	9.89
SW2	371,683	2,518,368	6.74
SW3	324,916	5,617,687	17.26
SWONW	324,173	5,293,865	16.34
SWOSW	325,697	6,031,979	18.53

3.2.4.3 Entrainment Impact Assessment

The primary method used in assessing the effects of ocean water intake systems for power plants and desalination plants is a modeling approach that uses data on target taxa abundances from sampling of the entrained larvae and potential source populations of larvae to calculate estimates of proportional entrainment (*PE*), which is an estimate of the daily conditional mortality due to entrainment. The *PE* estimates and other information on the source population of larvae are used to estimate the total probability of mortality (P_M) due to entrainment using the Empirical Transport Model (ETM) (Boreman et al. 1978; Boreman et al. 1981). The ETM has been used in the assessment of the effects of ocean intakes due to entrainment for all of the power plants and desalination facilities in California using ocean or estuarine waters as a source for the intake. The source document for conducting these studies has been Steinbeck et al. (2007) which was a report prepared for the California Energy Commission to provide guidance on impact assessments associated with certification of new applications for power plants in California.

Assessment of entrainment effects using ETM were limited to the most abundant fish and invertebrate taxa collected during the sampling. The fishes also had to be collected during several of the surveys at both the entrainment and source water stations to provide an adequate number of *PE* estimates to support the calculation of a meaningful estimate of P_M due to entrainment using the ETM. If data were only available from a few surveys then the reliability of the estimate of P_M would be in question due to the small sample size.

The four taxonomic groups (taxa) of fishes with data that met these criteria were: silversides, white croaker, kelpfishes, and anchovies (Engraulidae). Together these four taxa comprised approximately 42 percent of the total estimated fish larvae sampled at Station SWE during the March 2011–March 2012 study period. Data for the target taxa from sampling at the intake and source water stations were used to calculate estimates of *PE* which were used to estimate the probability of mortality P_M due to entrainment using the ETM. Detailed mathematical formulation of the ETM model is presented in **Appendix A**.

3.2.4.3.1 Larval Lengths

To represent the distribution of the lengths of the entrained larvae, a random sample of 200 measurements from all of the measured larvae for a taxa was drawn with replacement (bootstrap) and proportionally allocated among the surveys based on the abundances of larvae in those surveys. The 200 bootstrap measurements for each taxon were output as box plot histograms using SAS Graph® (SAS Institute 2008). An explanation of the legend accompanying the histograms is shown in **Figure 3-4**, and may be referred to for interpreting the length frequency dispersion statistics for selected taxa that are presented in Section 3.3.4–*Analysis of Individual Taxa*. The tick marks below the histogram represent the data from the bootstrap sampling. The statistics accompanying each figure represent the values computed for the measurements presented in the figure, not the statistics used in calculating the average age at entrainment and period of exposure. Those statistics were calculated from 100 random draws of 100 samples, drawn with replacement.

The average age at entrainment was calculated by dividing the difference between a computed size at hatching and the average length of the larvae by a larval growth rate obtained directly or derived from information available from scientific reports and journal articles. The period of time that the larvae were exposed to entrainment was calculated by dividing the difference between the size at hatching and the size at the 95th percentile by a larval growth rate obtained from the literature. The duration of the egg stage was added to this value for species with planktonic eggs. The 95th percentile value was used to eliminate outliers from the calculations. The size at hatching was estimated as follows:

$$\textit{Hatch Length} = (\textit{Median Length} + 1^{st}\ \textit{Percentile Length})/2$$

This calculated value was used because of the large variation in size among larvae smaller than the average length, and approximates the value of the 25th percentile used in other studies as the hatch length. This calculation assumes that the length frequency distribution is skewed towards smaller-sized larvae and usually resulted in a value close to the hatch size reported in the literature. The length frequency distributions for several of the fishes did not follow this pattern and the length of the 10th percentile of the distribution was used as the hatch length for these taxa to eliminate outlier values.

The ETM requires an estimate of the age of the larvae being entrained that is then used to estimate the period of time that the larvae are exposed to entrainment. This estimate was obtained by measuring a representative number of larvae of each of the target taxa from the intake samples and using published larval growth rates. The number of larvae collected and measured from intake samples varied by species among surveys, so the statistics used in calculating the average age at entrainment and total larval duration were standardized by drawing 100 random samples of 100 measurements from the pool of measured larvae with the samples proportionally allocated among the surveys, based on the abundances of larvae in those surveys. The samples were drawn with replacement because the number of larvae measured from each survey may have been less than the number needed to proportionally allocate the measurements among the surveys. Statistics and percentile values from each of the 100 samples were computed and the average of those values was used in calculating the period of time that the larvae were exposed to entrainment.

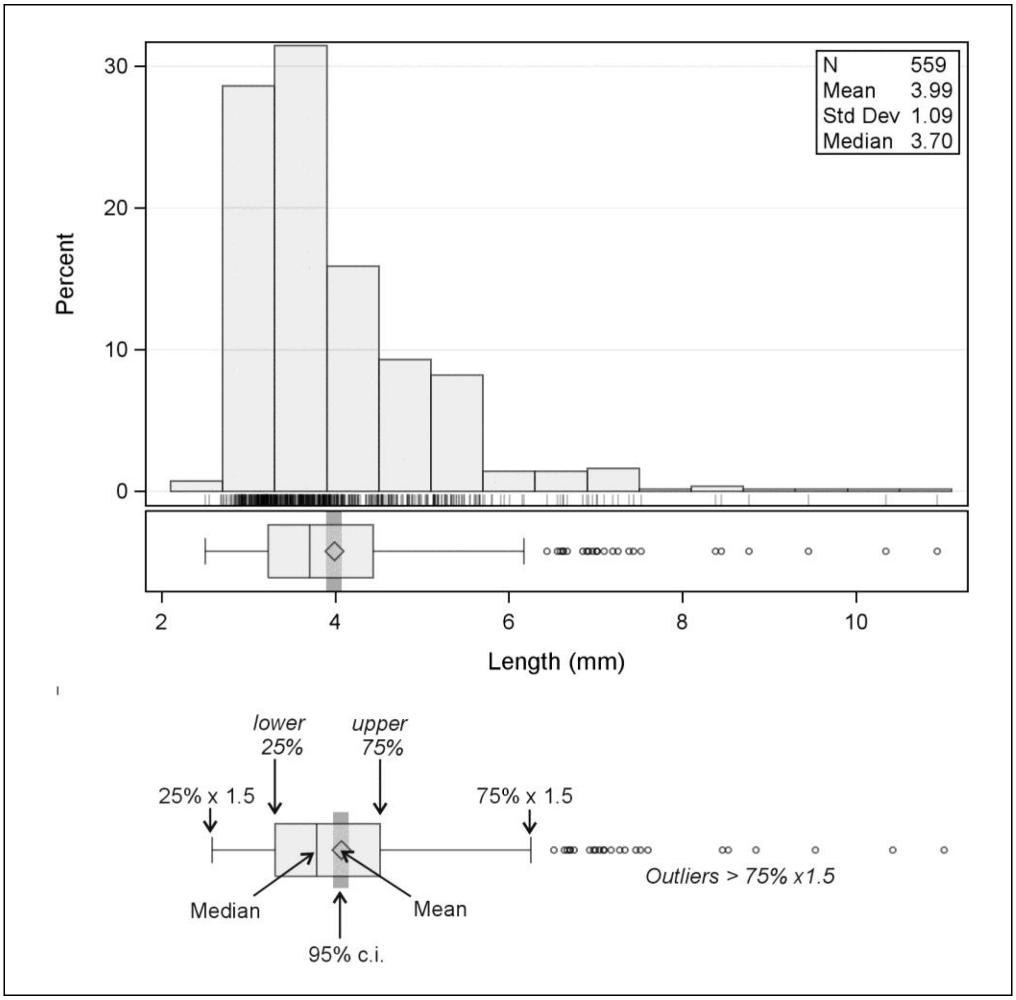


Figure 3-4. Explanation of dispersion statistic symbols for length frequency histograms.

3.2.4.3.2 ETM Model

The ETM was proposed by the U.S. Fish and Wildlife Service to estimate mortality rates resulting from cooling water withdrawals by power plants (Boreman et al. 1978, and subsequently in Boreman et al. 1981). The ETM provides an estimate of incremental mortality (a conditional estimate of entrainment mortality in absence of other mortality) (Ricker 1975) based on estimates of the fractional loss to the source water population represented by entrainment. The conditional mortality is represented as estimates of *PE* that are calculated for each survey and then expanded to predict regional effects on populations using ETM, as described below. Variations of this model have been discussed in MacCall et al. (1983) and have been used to assess impacts in the previous studies at California power plants (MacCall et al. 1983; Parker and DeMartini 1989; Steinbeck et al. 2007).

The estimate of *PE* is the central feature of the ETM (Boreman et al. 1981; MacCall et al. 1983). Estimates of *PE* are calculated for each taxon for each survey as the ratio of the estimated numbers of larvae entrained per day to the larval population estimates within specific volumes of the source water as follows:

$$PE_i = \frac{N_{E_i}}{N_{S_i}} = \frac{\overline{\rho}_{E_i} V_{E_i}}{\overline{\rho}_{S_i} V_{S_i}}, \qquad (1)$$

where N_{E_i} and N_{S_i} are the estimated numbers of larvae entrained and in the sampled source water per day in survey period *i*, $\overline{\rho}_{E_i}$ and $\overline{\rho}_{S_i}$ are the average concentrations of larvae from the intake and source water sampling, respectively, per day in survey period *i*, and V_{E_i} and V_{S_i} are the estimated volumes of the feedwater flow and sampled source water per day in survey period *i*. While a reasonably accurate estimate of the volume of the intake flow can be obtained, estimating the extent of the source water is more difficult and will vary depending upon oceanographic conditions and the period of time that the taxon being analyzed is in the plankton and exposed to entrainment. The *PE* estimated using Equation 1 is then adjusted based on the proportion of the sampled source water population to the total source population (P_S) (Steinbeck et al. 2007). Other intake assessments in California calculated *PE* using Equation 1 and then adjusted the estimate of *PE* using the proportion of the sampled source water population to the total source population (P_S) (Steinbeck et al. 2007). The *PE* for this assessment was calculated directly from the extrapolated source water populations as described below.

The extrapolated source water areas used in the *PE* estimates were calculated for each survey period and were calculated over the period of time that the larvae were estimated to be exposed to entrainment. This period of time was estimated using length data from the larvae measured from the entrainment samples for each taxon. The maximum age was calculated as the upper 95th percentile value of the lengths measured from the samples. The maximum age at entrainment was calculated by dividing the difference between the upper 95th percentile values of the lengths and the estimated hatch length or 10th percentile value of the lengths, depending on the taxa, by an estimated larval growth rate.

The source water populations for each survey were estimated using CODAR data on surface currents as described in **Appendix B**. The CODAR data were used to estimate potential sources of the larvae by back-projecting their transit over their estimated duration of exposure to entrainment. For each fish taxon, there were 30 back-projections calculated for each survey by randomly selecting an hour to start the back-projection within the 120-hour period centered on the survey date. The larval duration was used to determine the number of hour steps to include in the back-projections for each taxon. The maximum extents, upcoast and downcoast from the WBDDF intake, for each of the 30 back-projections were identified by determining the closest point on the coastline to the final location for the back-projection and determining the distance upcoast or downcoast from the intake from that point on the coast. The average of these maximum upcoast and downcoast distances was used to define the coastal extent of the source water. All points on the back-projections outside of depth limits defined for each taxon were eliminated, but points that returned inside these limits were included in determining maximum upcoast and downcoast distances.

The depth at the outer edge of the sampled source water (SW3; **Figure 3-3**), about 1,200 m (3,940 ft) offshore, was 22–24 m [72–79 ft]. The four fish taxa analyzed for this assessment varied in their depth ranges. As described above, these depth distributions were used to limit the points used in determining the upcoast and downcoast limits of the back-projections. Due to the poor coverage of CODAR close to shore, the shallowest depth used in limiting the back-projections was 30 m (98 ft), which was beyond the depths of the sampled source water stations, but helped ensure that all of the back-projections that were not along the shore were not eliminated from the analysis. A second set of back-projections was analyzed using a depth of 100 m (330 ft) for these nearshore species, which allowed a larger number of back-projection points to be included in the analysis. Depths of 100 m (330 ft) and 300 m (990 ft) were used for northern anchovy which has a broader depth and offshore distribution than the other modeled species.

The total estimated upcoast and downcoast extent from the back-projections was used to scale the estimate of the source water population (N_{Si}) using the ratio of the sampled source water area (1,500 m [4,921 ft]) by the estimated coastal extent of the source water. The PE_i calculated from the sampled source water data was then recalculated using this ratio and the P_M for each taxa was calculated as follows:

$$P_M = 1 - \sum_{i=1}^{12} f_i (1 - PE_i)^d, \qquad (2)$$

where f_i = the fraction of the source water population from the year present during survey i, and d = the period of exposure in days that the larvae are exposed to entrainment mortality represented by the PE_i. Using this approach, the two estimates of PE_i used for the ETM calculations for each survey were proportional to the difference between the upcoast and downcoast extents from the back-projections for that survey. Since the difference was not consistent among surveys, the average difference may not have reflected the differences in the resulting estimates of P_M.

The data and calculations necessary for ETM are provided in this example using data from this study for silverside larvae (Table 3-2). The lengths of the silverside larvae collected were evaluated to estimate that the larvae are susceptible to entrainment for 13.7 d. The following describes the information in each column of the table:

1. Survey Date
2. Daily Entrainment Estimate – the estimate of the total number of silverside larvae entrained per day based on an intake volume of 170,722 m^3/day 45.1 mgd);
3. Daily Sampled SW Population Estimate – the estimate of the total number of silverside larvae in the source water (SW) area sampled (**see Figure 3-3**) during the day the survey was conducted;
4. Sampled SW *PE* Estimate – the estimate of *PE* based on the entrainment and sampled SW estimates – not used in the calculations;
5. Sampled SW Survey Period Estimate – the estimate of the total number of silverside larvae in the SW during the survey period. This is the estimate from Column 3 multiplied by the number of days in the survey period. The sum of the estimates is used as an estimate of the total SW population;
6. Survey Fraction of Total SW (f_i) – the estimate of the fraction of the total source water population available during each survey period calculated from the survey estimates and total in Column 5;
7. Estimate of Coastal Extent of the SW (m) – the estimate from the CODAR extrapolations of the distance along the coast that larvae could have traveled over 13.7 d;
8. P_S Adjustment for SW – the ratio of the coastal distance of the sampled SW are (1,500 m) divided by the estimate of the coastal distance in Column 7;
9. Extrapolated SW Population Estimate – the estimate of the total number of silverside larvae in the extrapolated SW along the distance of coast from Column 7;
10. Extrapolated SW *PE* Estimate – the estimate of *PE* based on the entrainment (Column 2) and extrapolated SW (Column 8) estimates;
11. Intermediate ETM calculation – the estimate of the proportional survival for each survey period weighted by the fraction of the SW present during the survey (Column 6);

The final ETM estimate of P_M in the table is calculated by summing all the estimates of survival in Column 10 and subtracting the total from one to convert the survivals to an estimate of mortality.

Two estimates of variance were calculated for the ETM that are not presented in **Table 3-2**. One estimate was based on combining the variance components for estimates comprising the ETM, including the entrainment (N_{Ei}) and source water (N_{Si}) estimates, and estimate of the weight (f_i). The variance associated with the duration (d) and the scaling factor used to adjust the source water estimate were based on the coastal back-projections and were treated as constants. An alternative approach to estimating the variance for the ETM was calculated using the weighted-average coefficient of variation from the *PE*s from all 12 surveys.

Table 3-2. Example ETM calculations using data for silversides collected from this study. The numbers for each column correspond to descriptions in the text.

1. Survey Date	2. Daily Entrainment Estimate	3. Daily Sampled SW Estimate	4. Sampled SW *PE* Estimate	5. Sampled SW Survey Period Estimate	6. Survey Fraction of Total SW (f_i)	7. Estimate of Coastal Extent of SW (m)	8. P_S Adjustment for SW	9. Extrapolated SW Population	10. *PE* based on Extrapolated SW	11. $=1-f_i*(1-PE)^{\wedge d}$
31-Mar-11	882	115,021	0.007668	3,565,660	0.01603	7,546	0.19878	578,624	0.001524	0.01570
6-May-11	-	-	0	-	0	15,421	-	-	0	0.00000
9-Jun-11	-	-	0	-	0	11,486	-	-	0	0.00000
19-Jul-11	-	-	0	-	0	10,797	-	-	0	0.00000
9-Aug-11	-	-	0	-	0	46,352	-	-	0	0.00000
6-Sep-11	-	-	0	-	0	26,166	-	-	0	0.00000
12-Oct-11	-	-	0	-	0	8,079	-	-	0	0.00000
1-Nov-11	-	68,236	0	1,978,837	0.00890	18,829	-	856,528	0	0.00890
8-Dec-11	5,824	657,125	0.008863	23,656,517	0.10634	15,378	0.29633	6,736,770	0.000865	0.10509
12-Jan-12	59,021	2,349,954	0.025116	70,498,626	0.31690	2,131	1.00037	3,337,779	0.017683	0.24823
6-Feb-12	12,712	3,536,696	0.003594	91,954,090	0.41335	10,620	1.14162	25,040,156	0.000508	0.41048
5-Mar-12	5,092	1,141,078	0.004462	30,809,111	0.13849	2,197	1.82435	1,671,324	0.003047	0.13282
			Sum =	222,462,842					**ETM Estimate of PM =**	**0.07878**

Assumptions associated with the estimation of P_M include the following:

- The samples from each survey period represent a new and independent cohort of larvae;
- The estimates of larval abundance for each survey represent the proportion of total annual larval production during that survey;
- The conditional probability of entrainment, PE_i, is constant within survey periods;
- The conditional probability of entrainment, PE_i, is constant within each of the size classes of larvae present during each survey period;
- The conditional probability of entrainment, PE_i, is constant between egg and larval stages for species with a planktonic egg stage;
- The concentrations of larvae in the sampled source water are representative of the concentrations in the extrapolated source water; and
- Lengths and applied growth rates of larvae accurately estimate larval duration.

3.3 Results

3.3.1 Ocean Currents

Surface currents were measured hourly off the entire coastline of the State of California using high frequency (HF) radar by a network of CODAR Ocean Sensors, Ltd. SeaSondes® operated by the member institutions of the Central & Northern California Ocean Observing System (CeNCOOS) and SCCOOS consortia of the COCMP. **Appendix B** shows the source water back-projections from the WBDDF intake with the end date centered on the date of each source water survey. These data were used to estimate source water areas for the ETM modeling.

3.3.2 Larval Abundances at Proposed Offshore Intake Location

The following section presents larval fish and target invertebrate concentrations collected at the intake and source water stations from March 31, 2011 through March 5, 2012. Estimates of entrainment were derived from samples collected near the proposed offshore intake location (SWE). Source water larval concentration estimates were derived from samples collected at three source water stations (SW1, SW2, and SW3) located upcoast, downcoast, and offshore from the intake (**Figure 3-1**). Larval data by survey are presented in **Appendix C**. Over the year-long study there were 47 plankton samples collected and processed from the intake station (SWE) (**Table 3-2**). During the June 2011 survey only one of the two scheduled samples during the night cycle could be collected because of an equipment malfunction that could not be repaired in the field.

There were 831 fish larvae in 44 taxonomic groups (including unidentified and/or damaged larvae) collected at the intake station (SWE) with towed plankton nets during the 12 monthly surveys conducted from March 2011 through March 2012 (**Table 3-3**). Two of the fishes that were collected were post-larval specimens that were considered too large to be entrained: one vermilion rockfish (*Sebastes miniatus*) and one pipefish (*Syngnathus* spp.). These two fishes were not listed in **Table 3-3** and were not included in estimates of annual entrainment. Ten taxa comprised over 80 percent of the total mean concentration of fish larvae collected at SWE during the study period with the most abundant being silversides (Atherinopsidae), white croaker (*Genyonemus lineatus*), unidentified larval/post larval fishes, herrings and anchovies (Clupeiformes), combtooth blennies, roughcheek sculpin, and garibaldi (*Hypsypops rubicundus*). The silversides were all identified as jacksmelt (*Atherinopsis californiensis*), but are listed as silversides (Family Atherinopsidae) because another species of silversides, California grunion (*Leurestes tenuis*) was collected in the source water samples. Jacksmelt (silversides) comprised approximately one fourth of the total number of larval fishes collected at SWE.

Fish eggs were enumerated and identified from all samples collected at SWE during the study period. Subsampling (consisting of a 10 percent aliquot) of eggs was conducted if more than 500 eggs were estimated to occur in the entire sample, based on preliminary examination of the sample under a microscope. Thirty five of the 47 samples (74.5 percent) were subsampled for fish eggs. An estimated total of 78,759 fish eggs (adjusted for subsampling) was collected with towed plankton nets from the intake station (SWE) during the study period. Because of difficulties in positively identifying early developmental stages of most fish eggs, only one-fourth of the specimens could be identified to the family (or multiple-family) level or lower, with the remainder unidentified. Of the specimens that could be identified to a lower category, turbot, sanddab, herring, and sand flounder eggs were the most numerous. (**Table 3-3**).

Table 3-3. Summary of the number of towed plankton net samples collected at intake (SWE) and source water stations (SW1–SW3).

		Number of samples collected by station			
Survey	**Date**	**SWE**	**SW1**	**SW2**	**SW3**
WBN01	03/31/11	4	2	2	2
WBN03	05/05/11	4	2	2	2
WBN05*	06/09/11	3	1	1	1
WBN07	07/21/11	4	2	2	2
WBN09	08/09/11	4	2	2	2
WBN11	09/06/11	4	2	2	2
WBN13	10/12/11	4	2	2	2
WBN15	11/01/11	4	2	2	2
WBN17	12/08/11	4	2	2	2
WBN19	01/12/12	4	2	2	2
WBN21	02/06/12	4	2	2	2
WBN23	03/05/12	4	2	2	2
	Total	**47**	**23**	**23**	**23**

*During Survey 5 the night sampling at Stations SW1, SW2, and SW3 could not be completed due to equipment malfunction.

Target invertebrate larvae included Cancer crab megalops, market squid paralarvae (recently hatched), and California spiny lobster phyllosomes. There were 462 specimens collected from 7 taxonomic groupings (**Table 3-4**). Cancer crabs, representing at least four species, were the most abundant target invertebrate larvae with *Romaleon antennarius*/*Metacarcinus gracilis* (Pacific rock crab/slender crab) comprising 75.5 percent of the total. Yellow crab megalops were the next most abundant category (9.4 percent), followed by market squid paralarvae (6.0 percent) and California spiny lobster phyllosomes (5.3 percent).

The concentrations of each taxon per survey were extrapolated over one year to estimate total annual entrainment. Based on a proposed maximum feedwater pumping rate of 170,722 m^3 per day (45.1 mgd) for a full-scale facility, it was estimated that there would be 10,164,117 larval fishes entrained annually through an unscreened intake system (**Table 3-5**). The WBDDF project has substantially lower pumping rates (1,309 m^3 per day [0.346 mgd] average, and 1,935 m^3 per day ([0.511 mgd] design), that would entrain an estimated 77,939 and 115,208 fish larvae per year, respectively, through an unscreened intake. The maximum estimate for fish eggs entrained through an unscreened intake was 834.5 million annually based on the 45.1 mgd flows, and 6.4 million based on the average demonstration plant flows of 0.346 mgd. An estimated 3,936,378 target invertebrate larvae would be entrained through an unscreened intake at the maximum rate and 30,184 at the minimum calculated rate. The ordering of taxa based on estimated annual entrainment differed somewhat from the ordering based on average concentration per survey due to the varying time periods between surveys over which the concentrations were extrapolated.

Seasonally, the highest overall concentrations of larval fishes at Station SWE occurred in January−February 2012 with a smaller peak in September−October 2011, while the lowest concentrations occurred in March 2011 and November 2011 (**Figure 3-5**). The peak concentration occurred in January at 462/1,000 m^3. The peak concentration of fish eggs (32,554/1,000 m^3) occurred in February 2012, however, fish egg abundance was also high in August 2011 (29,725/1,000 m^3) (**Figure 3-6**).

Larval fish concentrations were substantially higher in samples collected at night than those collected during the day (**Figure 3-7**). The only exception was during December 2011 when the concentrations were slightly higher during the day. Diel differences in fish egg concentrations were more variable than those of fish larvae, but still generally exhibited a tendency for higher concentration at night during most monthly samples (**Figure 3-8**).

Table 3-4. Counts and concentrations of fish larvae, fish eggs, and target invertebrate larvae collected with towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

	Taxon	Common Name	Total # Collected	Mean Conc. (#/1,000 m^3)	Percent of Total Mean Conc.	Cumul. Percent
Fish Larvae						
1	Atherinopsidae	silversides	193	40.77	24.50	24.50
2	*Genyonemus lineatus*	white croaker	80	16.47	9.90	34.40
3	larval/post-larval fish unid.	unidentified larval fishes	75	15.06	9.05	43.45
4	Clupeiformes	herrings and anchovies	57	12.58	7.56	51.01
5	*Hypsoblennius* spp.	combtooth blennies	54	11.49	6.90	57.91
6	*Ruscarius creaseri*	roughcheek sculpin	48	10.25	6.16	64.07
7	*Hypsypops rubicundus*	garibaldi	53	10.10	6.07	70.14
8	Engraulidae	anchovies	46	9.53	5.73	75.87
9	*Gibbonsia* spp.	kelpfishes	33	5.31	3.19	79.06
10	*Citharichthys* spp.	sanddabs	22	4.56	2.74	81.80
11	*Paralichthys californicus*	California halibut	15	3.38	2.03	83.83
12	*Pleuronichthys* spp.	turbots	14	2.99	1.80	85.63
13	CIQ goby complex	gobies	13	2.63	1.58	87.21
14	Syngnathidae	pipefishes	13	2.43	1.46	88.67
15	*Neoclinus* spp.	fringeheads	8	1.67	1.00	89.67
16	*Parophrys vetulus*	English sole	16	1.62	0.97	90.64
17	*Sebastes* spp. *V*	rockfishes	9	1.55	0.93	91.58
18	*Sebastes* spp. *V_*	rockfishes	6	1.31	0.79	92.36
19	*Zaniolepis frenata*	shortspine combfish	12	1.22	0.73	93.10
20	*Paralabrax* spp.	sea basses	5	1.12	0.67	93.77
21	*Oxylebius pictus*	painted greenling	4	0.98	0.59	94.36
22	*Gobiesocidae*	clingfishes	5	0.86	0.52	94.87
23	*Leptocottus armatus*	Pacific staghorn sculpin	4	0.85	0.51	95.38
24	Labrisomidae	labrisomid blennies	3	0.67	0.40	95.79
25	Myctophidae	lanternfishes	5	0.63	0.38	96.17
26	*Menticirrhus undulatus*	California corbina	3	0.60	0.36	96.53
27	*Seriphus politus*	queenfish	3	0.60	0.36	96.89
28	Pleuronectiformes	flatfishes	5	0.50	0.30	97.19
29	*Artedius* spp.	sculpins	2	0.49	0.29	97.48
30	*Cheilotrema saturnum*	black croaker	2	0.46	0.28	97.76
31	*Peprilus simillimus*	Pacific butterfish	2	0.42	0.25	98.01
32	Blennioidei	blennies	2	0.40	0.24	98.25
33	*Chitonotus/Icelinus* spp.	sculpins	4	0.39	0.23	98.49
34	*Heterostichus rostratus*	giant kelpfish	2	0.36	0.22	98.70
35	Cottidae	sculpins	4	0.34	0.20	98.91
36	*Atractoscion nobilis*	white seabass	1	0.25	0.15	99.06
37	*Orthonopias triacis*	snubnose sculpin	1	0.24	0.14	99.20
38	*Rhinogobiops nicholsi*	blackeye goby	1	0.22	0.13	99.33
39	*Opisthonema* spp.	thread herrings	1	0.20	0.12	99.45
40	*Pleuronichthys verticalis*	hornyhead turbot	1	0.20	0.12	99.57
41	*Sardinops sagax*	Pacific sardine	1	0.20	0.12	99.69
42	*Xystreurys liolepis*	fantail sole	1	0.20	0.12	99.81

(table continued)

Table 3-4 (continued). Counts and concentrations of fish larvae, fish eggs, and target invertebrate larvae collected with towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

	Taxon	Common Name	Total # Collected	Mean Conc. (#/1,000 m³)	Percent of Total Mean Conc.	Cumul. Percent
43	*Scorpaena guttata*	California scorpionfish	1	0.16	0.10	99.91
44	*Trachurus symmetricus*	jack mackerel	1	0.15	0.09	100.00
			831	**166.41**	**100.00**	
Fish Eggs						
1	fish eggs (early development stage)	fish eggs	57,560	9,957.89	72.18	72.18
2	*Pleuronichthys* spp.	turbot eggs	6,256	1,095.41	7.94	80.11
3	*Citharichthys* spp.	sanddab eggs	4,938	1,066.92	7.73	87.85
4	Clupeidae	herring eggs	2,384	546.73	3.96	91.81
5	Paralichthyidae	sand flounder eggs	3,272	422.22	3.06	94.87
6	Sciaenidae	croaker eggs	1,062	186.09	1.35	96.22
7	Sciaenidae/Paralichthyidae/ Labridae	fish eggs	1,371	181.58	1.32	97.54
8	Engraulidae	anchovy eggs	751	166.18	1.20	98.74
9	Labridae	wrasse eggs	422	73.54	0.53	99.27
10	fish eggs (damaged)	damaged fish eggs unid.	356	31.95	0.23	99.50
11	fish eggs (later development stage)	fish eggs	129	24.83	0.18	99.68
12	Sciaenidae/Paralichthyidae	fish eggs	47	11.49	0.08	99.77
13	Pleuronectidae	righteye flounder eggs	88	9.68	0.07	99.84
14	*Genyonemus lineatus*	white croaker eggs	20	5.00	0.04	99.87
15	Labridae/Paralichthyidae	fish eggs	20	4.73	0.03	99.91
16	*Engraulis mordax*	northern anchovy eggs	40	3.45	0.03	99.93
17	Haemulidae	grunt eggs	13	2.61	0.02	99.95
18	Haemulidae/Paralichthyidae	fish eggs	10	2.53	0.02	99.97
19	*Sphyraena argentea*	Pacific barracuda eggs	10	2.02	0.01	99.99
20	*Trachurus symmetricus*	jack mackerel eggs	10	2.02	0.01	100.00
			78,759	**13,796.85**	**100.00**	
Target Invertebrate Larvae						
1	*Romaleon anten./Metacarcinus grac.* (megalops)	Pacific rock crab/slender crab	370	45.99	75.53	75.53
2	*Metacarcinus anthonyi* (megalops)	yellow crab megalops	43	5.74	9.43	84.96
3	*Doryteuthis opalescens*	market squid	18	3.65	5.99	90.95
4	*Panulirus interruptus* (phyllosome)	Calif. spiny lobster (larval)	16	3.21	5.27	96.22
5	Cancridae (megalops)	Cancer crabs megalops	12	1.86	3.05	99.28
6	*Cancer productus/Romaleon* spp. (megalops)	rock crab megalops	2	0.23	0.38	99.66
7	Cancridae (megalops, damaged)	Cancer crabs meg., damaged	1	0.21	0.34	100.00
			462	**60.89**	**100.00**	

Table 3-5. Estimated annual entrainment of fish larvae, fish eggs, and target invertebrate larvae through an unscreened intake based on data collected with towed plankton nets at the intake station (SWE) from March 2011 through March 2012 for three intake volumes.

	Taxon	Common Name	Estimated Annual Entrainment 1,309 m³ (0.346 mgd)	1,935 m³ (0.511 mgd)	170,722 m³ (45.1 mgd)
Fish Larvae					
1	Atherinopsidae	silversides	18,983	28,061	2,475,663
2	larval/post-larval fish	larval fishes	7,508	11,099	979,167
3	*Genyonemus lineatus*	white croaker	7,251	10,718	945,568
4	*Hypsypops rubicundus*	garibaldi	5,371	7,939	700,450
5	Clupeiformes	herrings and anchovies	5,156	7,622	672,421
6	*Hypsoblennius* spp.	combtooth blennies	5,114	7,560	666,944
7	Engraulidae	anchovies	4,548	6,723	593,133
8	*Ruscarius creaseri*	roughcheek sculpin	4,545	6,718	592,713
9	*Gibbonsia* spp.	kelpfishes	2,580	3,814	336,495
10	*Citharichthys* spp.	sanddabs	2,468	3,649	321,907
11	*Paralichthys californicus*	California halibut	1,391	2,056	181,368
12	*Pleuronichthys* spp.	turbots	1,341	1,982	174,897
13	CIQ goby complex	gobies	1,160	1,715	151,294
14	Syngnathidae	pipefishes	1,152	1,703	150,258
15	*Parophrys vetulus*	English sole	893	1,320	116,477
16	*Neoclinus* spp.	fringeheads	848	1,253	110,540
17	*Sebastes* spp. V	rockfishes	704	1,041	91,798
18	*Sebastes* spp. V_	rockfishes	701	1,036	91,359
19	*Zaniolepis frenata*	shortspine combfish	670	990	87,358
20	*Paralabrax* spp.	sea basses	564	834	73,547
21	*Leptocottus armatus*	Pacific staghorn sculpin	438	648	57,171
22	*Oxylebius pictus*	painted greenling	415	613	54,089
23	Gobiesocidae	clingfishes	392	580	51,152
24	Myctophidae	lanternfishes	318	471	41,512
25	*Menticirrhus undulatus*	California corbina	303	449	39,572
26	Labrisomidae	labrisomid blennies	297	439	38,739
27	*Seriphus politus*	queenfish	276	407	35,930
28	Pleuronectiformes	flatfishes	269	398	35,136
29	*Cheilotrema saturnum*	black croaker	229	338	29,839
30	*Chitonotus*/*Icelinus* spp.	sculpins	216	319	28,165
31	*Peprilus simillimus*	Pacific butterfish	210	310	27,347
32	*Artedius* spp.	sculpins	207	307	27,045
33	Cottidae	sculpins	190	280	24,717
34	Blennioidei	blennies	187	276	24,339
35	*Heterostichus rostratus*	giant kelpfish	174	257	22,688
36	*Atractoscion nobilis*	white seabass	127	188	16,607
37	*Rhinogobiops nicholsi*	blackeye goby	126	186	16,431
38	*Orthonopias triacis*	snubnose sculpin	104	153	13,522
39	*Sardinops sagax*	Pacific sardine	101	150	13,232
40	*Opisthonema* spp.	thread herrings	101	149	13,109
41	*Pleuronichthys verticalis*	hornyhead turbot	89	132	11,609
42	*Xystreurys liolepis*	fantail sole	86	127	11,231

(table continued)

Table 3-5 (continued). Estimated annual entrainment of fish larvae, fish eggs, and target invertebrate larvae through an unscreened intake based on data collected with towed plankton nets at the intake station (SWE) from March 2011 through March 2012 for three intake volumes.

			Estimated Annual Entrainment		
	Taxon	Common Name	1,309 m³ (0.346 mgd)	1,935 m³ (0.511 mgd)	170,722 m³ (45.1 mgd)
43	*Scorpaena guttata*	California scorpionfish	78	116	10,236
44	*Trachurus symmetricus*	jack mackerel	56	83	7,347
			77,939	**115,208**	**10,164,117**
	Fish Eggs				
1	fish eggs unid. (early stage)	unidentified fish eggs	4,661,669	6,890,838	607,936,680
2	*Pleuronichthys* spp.	turbot eggs	507,156	749,673	66,139,109
3	*Citharichthys* spp.	sanddab eggs	470,298	695,190	61,332,411
4	Clupeidae	herring eggs	223,302	330,083	29,121,253
5	Paralichthyidae	sand flounder eggs	197,609	292,103	25,770,499
6	Sciaenidae/Paralichthyidae/ Labridae	fish eggs	91,525	135,291	11,935,931
7	Sciaenidae	croaker eggs	85,675	126,644	11,173,052
8	Engraulidae	anchovy eggs	69,448	102,658	9,056,856
9	Labridae	wrasse eggs	38,926	57,540	5,076,414
10	fish eggs (damaged)	damaged fish eggs unid.	17,331	25,618	2,260,153
11	fish eggs unid. (later stage)	unidentified fish eggs	13,626	20,142	1,776,991
12	Sciaenidae/Paralichthyidae	fish eggs	5,228	7,729	681,858
13	Pleuronectidae	righteye flounder eggs	5,123	7,573	668,096
14	*Genyonemus lineatus*	white croaker eggs	2,829	4,182	368,952
15	Labridae/Paralichthyidae	fish eggs	2,675	3,953	348,790
16	*Engraulis mordax*	northern anchovy eggs	1,895	2,802	247,167
17	Haemulidae/Paralichthyidae	fish eggs	1,273	1,882	166,072
18	Haemulidae	grunt eggs	1,270	1,877	165,577
19	*Sphyraena argentea*	Pacific barracuda eggs	1,015	1,500	132,317
20	*Trachurus symmetricus*	jack mackerel eggs	1,015	1,500	132,317
			6,398,887	**9,458,779**	**834,490,494**
Target Invertebrate Larvae					
1	*Romaleon antennarius / Metacarcinus gracilis* (megalops)	Pacific rock crab/slender crab	22,971	33,955	2,995,675
2	*Metacarcinus anthonyi* (megalops)	yellow crab megalops	2,949	4,360	384,615
3	*Doryteuthis opalescens*	market squid	1,616	2,388	210,697
4	*Panulirus interruptus* (phyllosome)	California spiny lobster (larval)	1,614	2,386	210,475
5	Cancridae (megalops)	Cancer crabs megalops	829	1,225	108,087
6	*Cancer productus / Romaleon* spp. (megalops)	rock crab megalops	110	162	14,296
7	Cancridae (megalops, damaged)	damaged Cancer crab megalops	96	142	12,533
			30,184	**44,618**	**3,936,378**

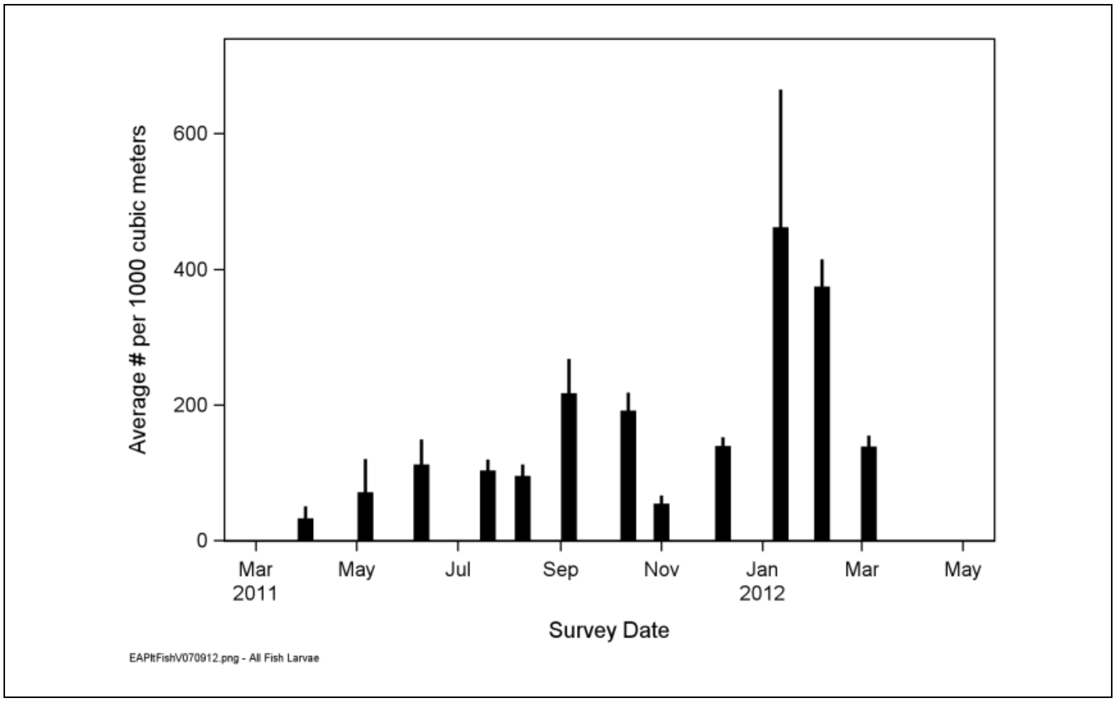


Figure 3-5. Mean concentration (#/1,000 m^3) and standard error for all larval fishes collected with towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

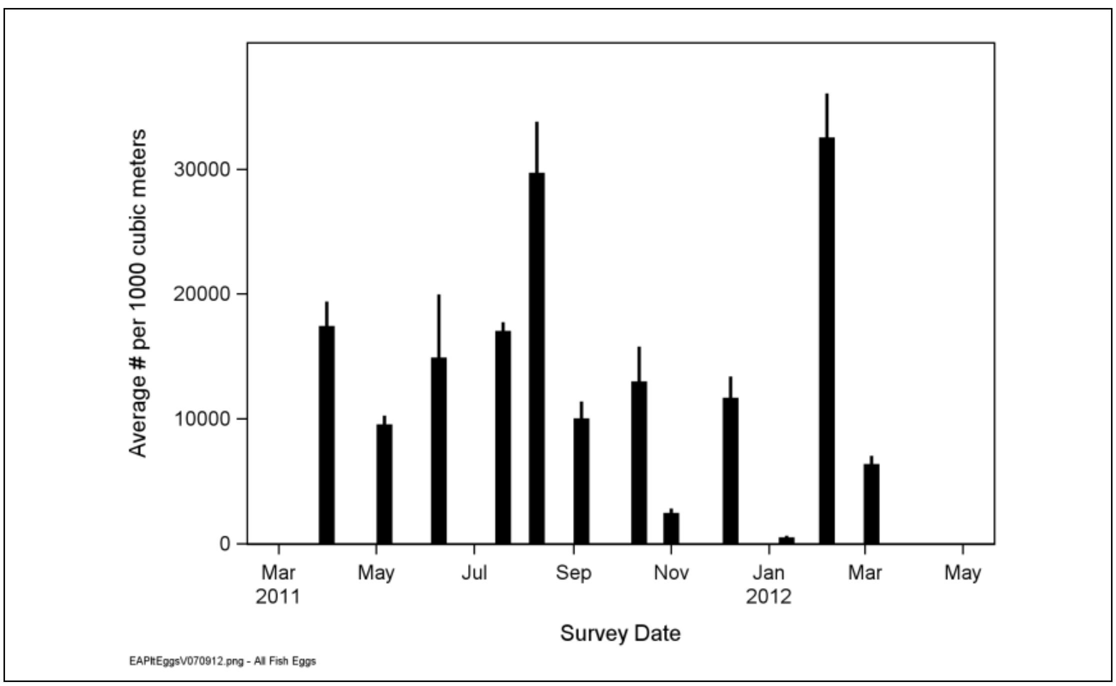


Figure 3-6. Mean concentration (#/1,000 m^3) and standard error for fish eggs collected with towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

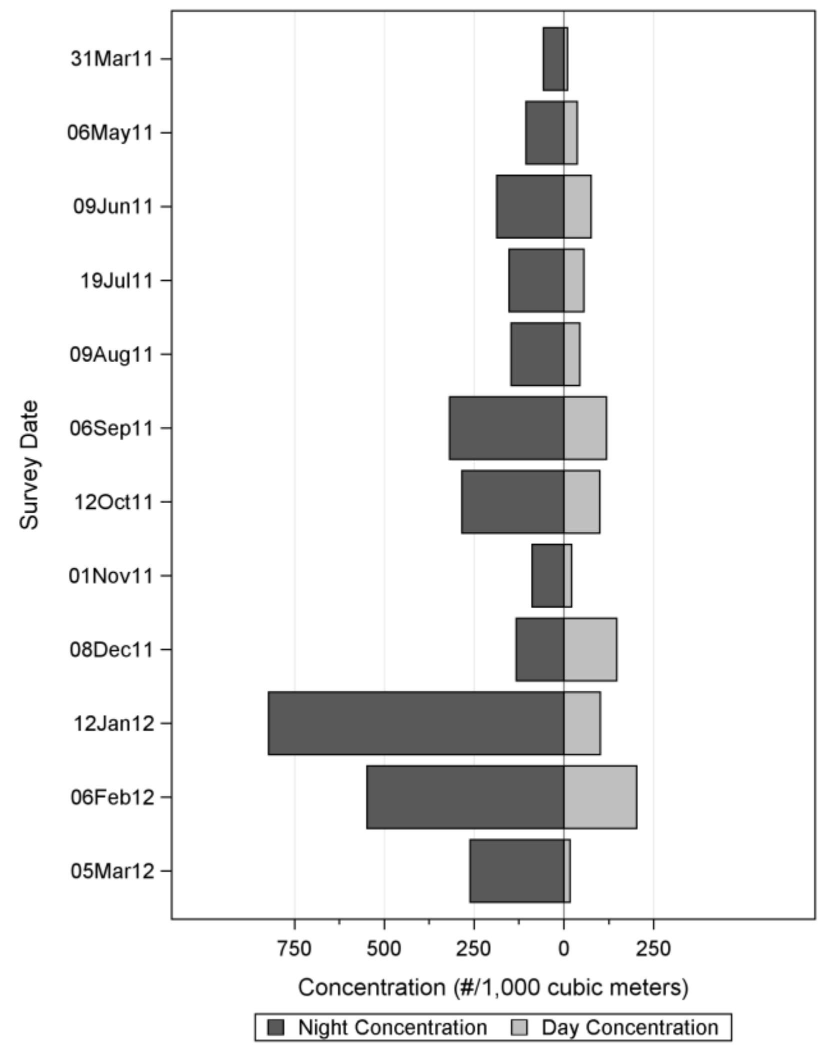


Figure 3-7. Mean concentration (#/1,000 m^3) in night and day samples for all larval fishes collected with towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

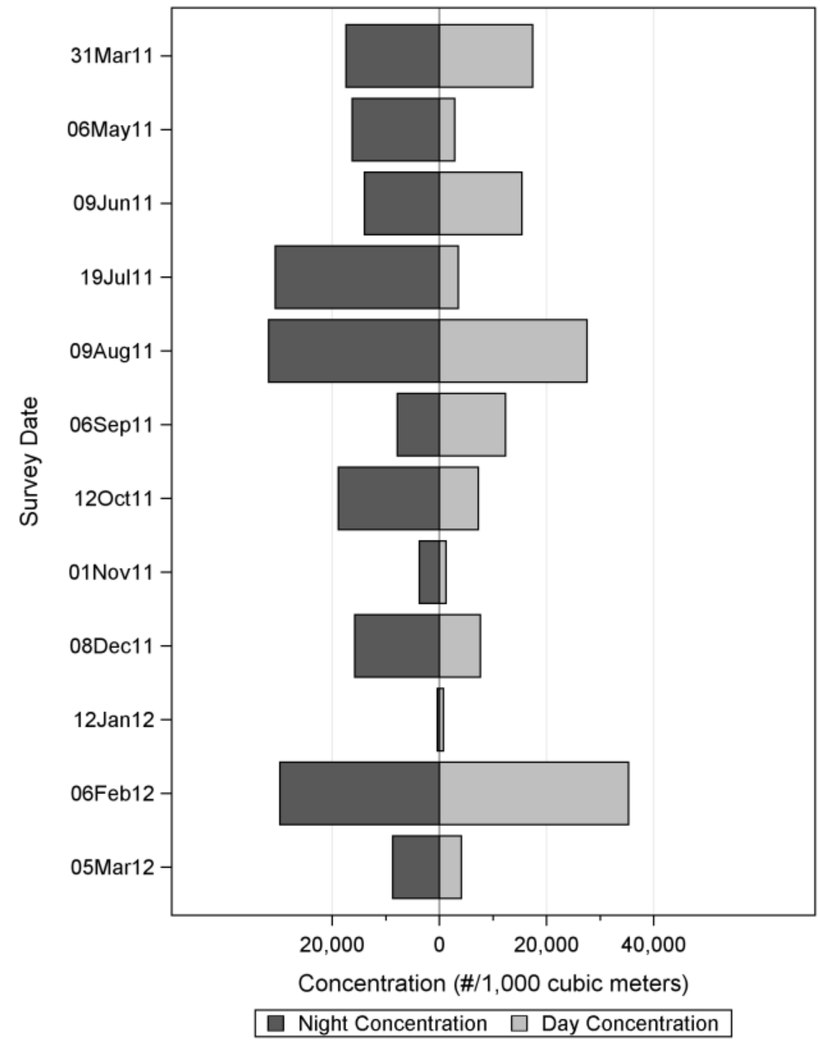


Figure 3-8. Mean concentration (#/1,000 m^3) in night and day samples for fish eggs collected with towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

3.3.3 Source Water Summary

A total of 1,397 fish larvae in 59 taxonomic categories was collected using towed plankton nets at the source water stations (SW1, SW2, and SW3) during the 12 monthly surveys conducted from March 2011 through March 2012 (**Table 3-6**). Of the larval fishes collected, approximately 11 percent could not be identified because they were either too small to exhibit definitive identification characteristics or they had been damaged. Silversides (98 percent jacksmelt and 2 percent California grunion), anchovies, and white croaker were the three most abundant taxa overall. The peak in abundance of larval fish at the source water stations occurred during February 2012 and lowest concentrations occurred in June 2011 (**Figure 3-9**). Higher overall concentrations were also seen in September and October 2011. Complete data by survey are presented in **Appendix C**.

Fish eggs were enumerated and identified from all source water (SW1, SW2, and SW3) samples collected using towed plankton nets during the study period. As with the entrainment samples, subsampling of eggs was conducted if more than 500 eggs were estimated to be present in a sample. A total of 106,022 fish eggs (adjusted for subsampling) was collected using towed plankton nets from the three source water stations during the study period (**Table 3-6**). The eggs were classified into 18 taxa, with the majority of specimens (73 percent) remaining unidentified due to their early developmental stages that lacked definitive identification characteristics. The next most abundant fish egg groups were sanddabs, turbots, and sand flounders. The peak in abundance of fish eggs at the source water stations was in August 2011 with approximately 35,000/1,000 m^3, and lowest concentrations occurred in January 2012 (approximately 2,500/1,000 m^3)(**Figure 3-10**).

A total of 457 target invertebrate larvae was collected using towed plankton nets from the source water stations (SW1, SW2, and SW3) during the study period (**Table 3-6**). These included a total of 359 Cancer crab megalopae from five taxonomic groupings, comprising 72.5 percent of the target larvae collected. Cancer crab megalops are identified by size and shape, however, some of them overlap in their dimensions making it unreliable to visually separate them to the species level. The remainder of the target invertebrate larvae were market squid and California spiny lobster.

As with the intake station samples, fish larvae were generally more abundant at night than during the day (**Figure 3-11**). Fish egg concentration was also generally higher during the night sampling except during July and August 2011 (**Figure 3-12**).

Table 3-6. Counts and concentrations of larval fishes, fish eggs, and target invertebrate larvae collected using towed plankton nets at three source water stations (SW1, SW2, and SW3) from March 2011 through March 2012.

	Taxon	Common Name	Total # Collected	Mean Conc. (#/1,000 m³)	Percent of Total Mean Conc.	Cumul. Percent
Fish Larvae						
1	Atherinopsidae	silversides	182	26.30	13.94	13.94
2	Engraulidae	anchovies	181	24.39	12.93	26.87
3	*Genyonemus lineatus*	white croaker	140	20.85	11.05	37.92
4	larval fish unidentified	unidentified larval fishes	144	20.43	10.83	48.75
5	Clupeiformes	herrings and anchovies	93	15.13	8.02	56.77
6	*Hypsoblennius* spp.	combtooth blennies	98	12.20	6.47	63.24
7	*Citharichthys* spp.	sanddabs	65	8.70	4.61	67.85
8	*Pleuronichthys* spp.	turbots	46	6.32	3.35	71.20
9	*Hypsypops rubicundus*	garibaldi	42	5.06	2.68	73.88
10	*Gibbonsia* spp.	kelpfishes	40	4.85	2.57	76.45
11	*Ruscarius creaseri*	roughcheek sculpin	23	3.59	1.90	78.36
12	*Paralichthys californicus*	California halibut	22	3.38	1.79	80.15
13	*Opisthonema* spp.	thread herrings	24	3.34	1.77	81.92
14	CIQ goby complex	gobies	20	2.75	1.46	83.38
15	*Paralabrax* spp.	sea basses	17	2.38	1.26	84.64
16	*Parophrys vetulus*	English sole	36	2.36	1.25	85.89
17	Myctophidae	lanternfishes	21	2.22	1.18	87.07
18	*Oxylebius pictus*	painted greenling	14	2.15	1.14	88.21
19	*Sardinops sagax*	Pacific sardine	14	2.08	1.10	89.31
20	Pleuronectiformes	flatfishes	14	1.65	0.87	90.18
21	Syngnathidae	pipefishes	11	1.59	0.84	91.03
22	*Sebastes* spp. V	rockfishes	11	1.46	0.77	91.80
23	Sciaenidae	croakers	10	1.30	0.69	92.49
24	*Leptocottus armatus*	Pacific staghorn sculpin	6	0.90	0.48	92.97
25	*Sebastes* spp. V_	rockfishes	6	0.86	0.46	93.42
26	*Pleuronichthys ritteri*	spotted turbot	6	0.79	0.42	93.84
27	Labrisomidae	labrisomid blennies	6	0.78	0.41	94.25
28	*Gobiesox* spp.	clingfishes	12	0.76	0.40	94.66
29	*Peprilus simillimus*	Pacific butterfish	5	0.76	0.40	95.06
30	*Zaniolepis frenata*	shortspine combfish	10	0.74	0.39	95.45
31	*Artedius* spp.	sculpins	7	0.71	0.38	95.83
32	*Pleuronichthys verticalis*	hornyhead turbot	7	0.63	0.33	96.16
33	*Rhinogobiops nicholsi*	blackeye goby	6	0.61	0.32	96.49
34	*Xystreurys liolepis*	fantail sole	4	0.57	0.30	96.79
35	*Symphurus atricaudus*	California tonguefish	4	0.56	0.30	97.08
36	Cottidae	sculpins	7	0.54	0.29	97.37
37	Chaenopsidae/Labrisomidae	tube/labrisomid blennies	4	0.51	0.27	97.64
38	*Pleuronichthys guttulatus*	diamond turbot	3	0.43	0.23	97.87
39	*Chitonotus*/*Icelinus* spp.	sculpins	3	0.40	0.21	98.08
40	*Chromis punctipinnis*	blacksmith	5	0.33	0.17	98.26
41	*Scorpaenichthys marmoratus*	cabezon	3	0.30	0.16	98.42
42	*Neoclinus* spp.	fringeheads	2	0.28	0.15	98.56
43	*Umbrina roncador*	yellowfin croaker	2	0.26	0.14	98.70
44	Blennioidei	blennies	2	0.25	0.13	98.83

(table continued)

Table 3-6 (continued). Counts and concentrations of larval fishes, fish eggs, and target invertebrate larvae collected using towed plankton nets at three source water stations (SW1, SW2, and SW3) from March 2011 through March 2012.

	Taxon	Common Name	Total # Collected	Mean Conc. (#/1,000 m^3)	Percent of Total Mean Conc.	Cumul. Percent
45	*Liparis* spp.	snailfishes	4	0.22	0.11	98.95
46	*Lepidopsetta bilineata*	rock sole	2	0.17	0.09	99.04
47	*Alloclinus holderi*	island kelpfish	1	0.17	0.09	99.13
48	*Cheilotrema saturnum*	black croaker	1	0.17	0.09	99.22
49	*Menticirrhus undulatus*	California corbina	1	0.17	0.09	99.31
50	*Platichthys stellatus*	starry flounder	1	0.16	0.08	99.40
51	*Synodus lucioceps*	California lizardfish	1	0.16	0.08	99.48
52	*Scomber japonicus*	Pacific mackerel	1	0.15	0.08	99.56
53	*Girella nigricans*	opaleye	1	0.14	0.07	99.63
54	*Seriphus politus*	queenfish	1	0.13	0.07	99.70
55	*Chilara taylori*	spotted cusk-eel	1	0.12	0.06	99.77
56	Chaenopsidae	tube blennies	1	0.12	0.06	99.83
57	*Heterostichus rostratus*	giant kelpfish	1	0.11	0.06	99.89
58	Clinidae	kelpfishes	1	0.11	0.06	99.95
59	*Atractoscion nobilis*	white seabass	1	0.10	0.05	100.00
			1,397	**188.65**	**100.00**	
Fish Eggs						
1	fish eggs (early stage)	fish eggs	79,199	9,490.69	73.07	73.07
2	*Citharichthys* spp.	sanddab eggs	10,009	1,389.59	10.70	83.77
3	*Pleuronichthys* spp.	turbot eggs	7,540	958.87	7.38	91.15
4	Paralichthyidae	sand flounder eggs	3,422	348.35	2.68	93.83
5	Clupeidae	herring eggs	2,156	308.63	2.38	96.21
6	Engraulidae	anchovy eggs	1,309	190.53	1.47	97.68
7	Sciaenidae	croaker eggs	539	85.80	0.66	98.34
8	fish eggs	fish eggs	626	63.23	0.49	98.82
9	Sciaenidae/Paralichthy/Labridae	fish eggs	450	61.44	0.47	99.30
10	Labridae/Paralichthyidae	fish eggs	238	35.08	0.27	99.57
11	Sciaenidae/Paralichthyidae	fish eggs	121	18.83	0.14	99.71
12	Pleuronectidae	righteye flounder eggs	219	16.74	0.13	99.84
13	Labridae	wrasse eggs	129	12.26	0.09	99.93
14	*Opisthonema* spp.	thread herring eggs	20	2.84	0.02	99.96
15	fish eggs (damaged)	damaged fish eggs unid.	20	1.81	0.01	99.97
16	Haemulidae/Paralichthyidae	fish eggs	10	1.72	0.01	99.98
17	*Paralabrax* spp.	sand bass eggs	10	1.50	0.01	100.00
18	*Vinciguerria lucetia*	Panama lightfish eggs	4	0.61	<0.01	100.00
			106,022	**12,988.52**	**100.00**	
Target Invertebrate Larvae						
1	*Romaleon anten./Metacarc. grac.* (megalops [meg.])	Pacific rock crab/slender crab	287	28.84	55.41	55.41
2	*Doryteuthis opalescens*	market squid	75	11.15	21.42	76.83
3	*Metacarcinus anthonyi* (meg.)	yellow crab megalops	59	7.15	13.74	90.57
4	*Panulirus interruptus* (phyllosome)	Calif. spiny lobster	23	3.18	6.11	96.68
5	Cancridae damaged (meg.)	Cancer crab megalops	6	0.83	1.59	98.27
6	Cancridae (meg.)	Cancer crab megalops	6	0.77	1.48	99.75
7	*Cancer prod./Romaleon* spp.(meg.)	rock crab megalops	1	0.13	0.25	100.00
			457	**52.05**	**100.00**	

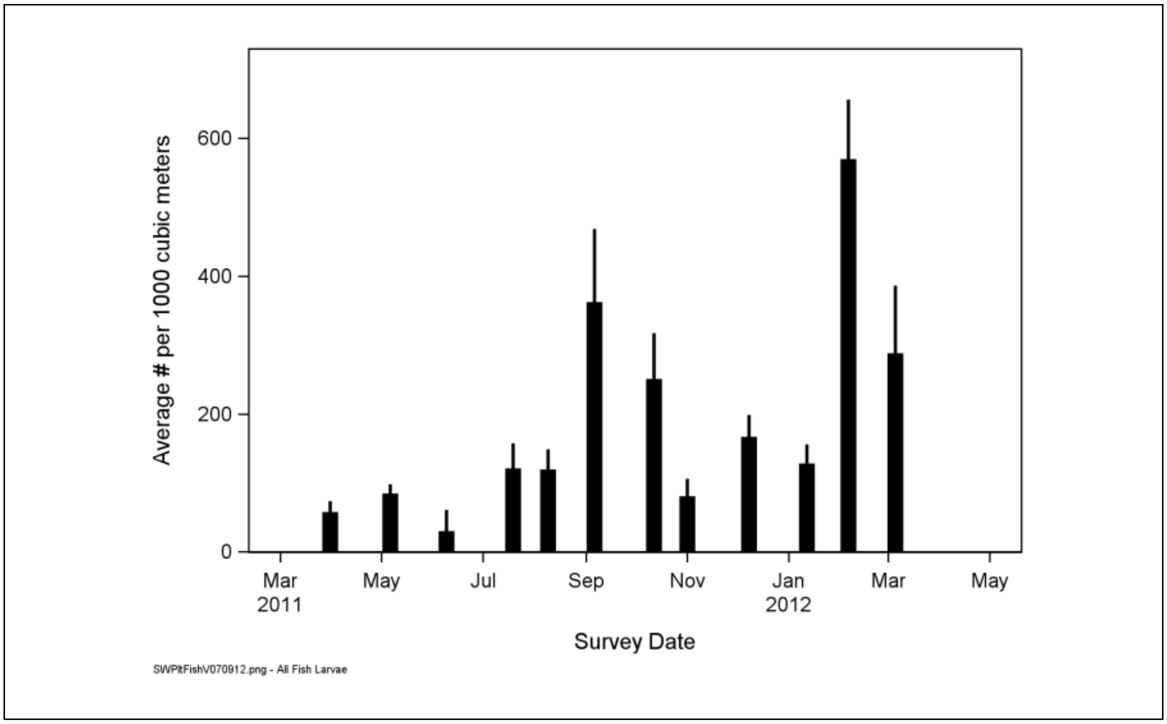


Figure 3-9. Mean concentration (#/1,000 m^3) and standard error for all larval fishes collected using towed plankton nets at source water stations (SW1, SW2, SW3) from March 2011 through March 2012.

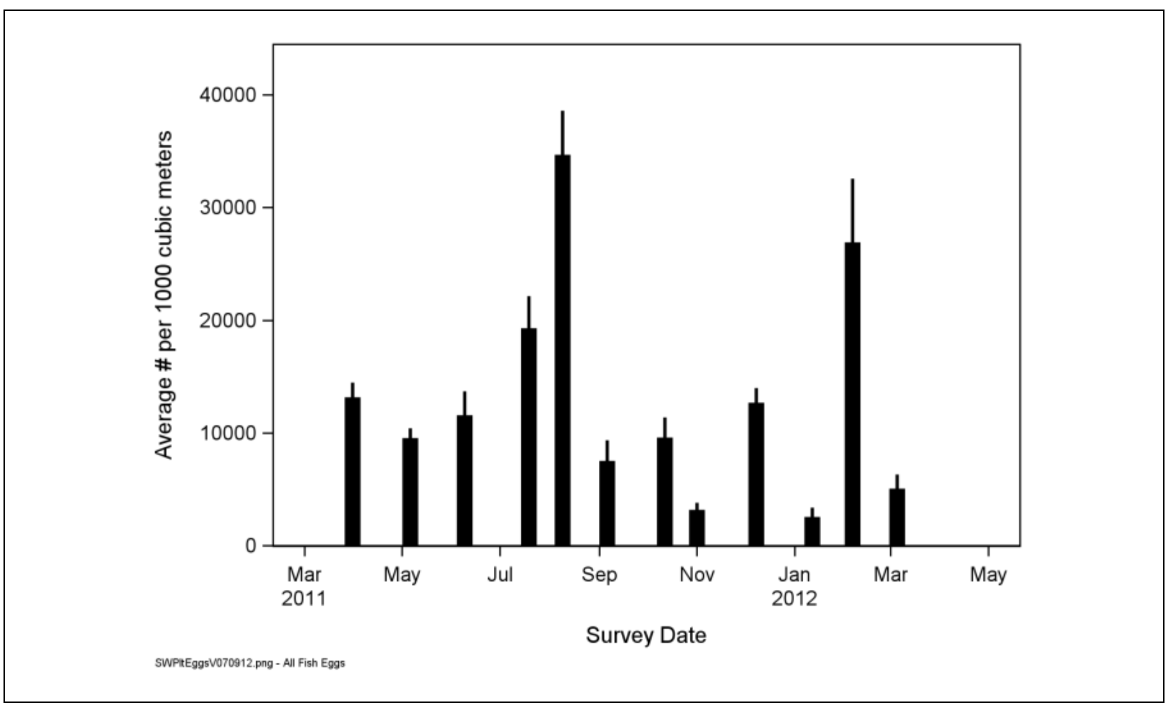


Figure 3-10. Mean concentration (#/1,000 m^3) and standard error for fish eggs collected using towed plankton nets at source water stations (SW1, SW2, SW3) from March 2011 through March 2012.

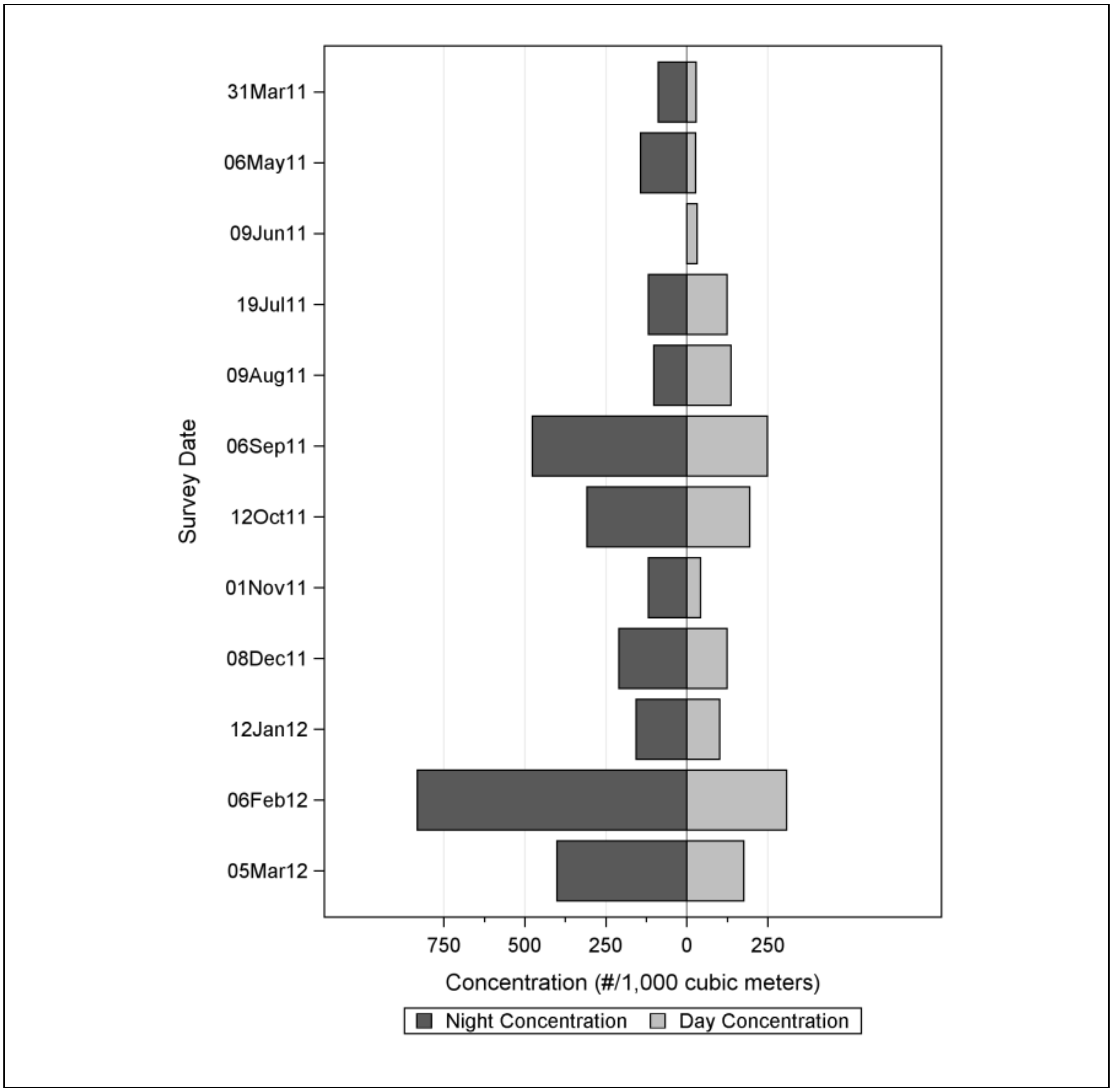


Figure 3-11. Mean concentration (#/1,000 m^3) in night and day samples for fish larvae collected using towed plankton nets at source water stations (SW1, SW2, SW3) from March 2011 through March 2012.

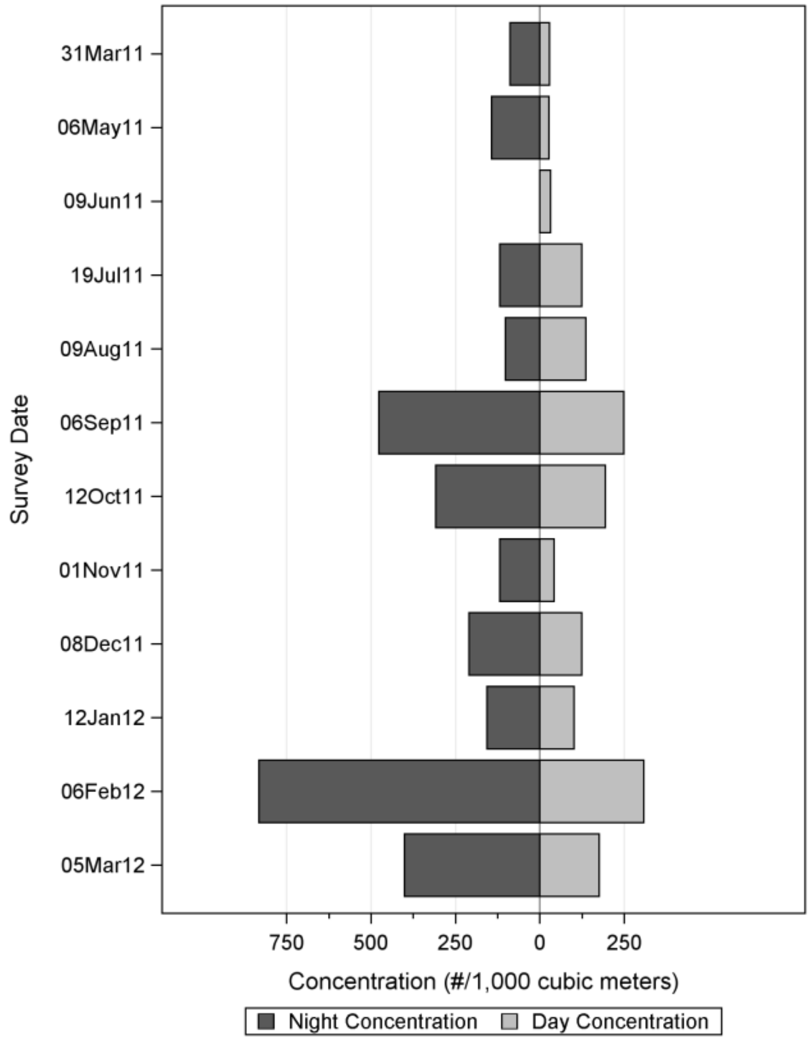


Figure 3-12. Mean concentration (#/1,000 m^3) in night and day samples for fish eggs collected using towed plankton nets at source water stations (SW1, SW2, SW3) from March 2011 through March 2012.

3.3.4 Analysis of Individual Taxa

Six larval fish taxa that comprised over half of the larval fish population at the location of the proposed intake were selected for detailed impact analysis. Impact analysis was also done for Cancer crab megalops larvae. The selection criteria for the fishes were their high estimated average concentrations in both the entrainment and source water samples, high frequency of occurrence among surveys, or fishery importance. These taxa included jacksmelt (Atherinidae), white croaker, northern anchovy, kelpfishes, California halibut, garibaldi, and Cancer crabs. California halibut and garibaldi were not abundant enough among surveys to analyze using the ETM modeling approach.

Detailed modeling of potential impacts using the ETM was only done for the larvae of four fishes (silversides Atherinopsidae, white croaker *Genyonemus lineatus*, northern anchovy *Engraulis mordax*, and kelpfishes *Gibbonsia* spp.) and Cancer crabs. These five taxa were selected for analysis because they were collected in relatively high abundance during the studies, but also were collected during a large number of the 12 surveys. Garibaldi was high in overall abundance (**Table 3-3**), but was only collected during two of the surveys at the intake station, and its occurrence at the source water stations only coincided with one of those surveys. As a result, there was only one estimate of entrainment relative to source water stations that could be calculated for garibaldi. This estimate of proportional entrainment is the primary input variable to the ETM and the data for garibaldi would have only provided a single estimate for the modeling. As a result, while the ETM estimate of the potential effects of the intake on garibaldi larvae could be calculated, or the annual proportion of the source water population of larvae entrained by the intake estimated, the reliability of the estimates would be in question.

Similarly, while combtooth blennies (*Hypsoblennius* spp.) were more abundant overall in the samples at the intake station, they were only collected during four of the surveys and only three surveys when combtooth blennies were also collected at the source water stations. As a result, data for kelpfishes (*Gibbonsia* spp.) were selected for analysis as the larvae for this fish were collected during most of the surveys at both the intake and source water stations resulting in a more reliable estimate for the ETM. Kelpfishes also occur in similar habitats as combtooth blennies, and therefore provide results that could be used to estimate impacts to other fishes that occupy a similar habitat.

3.3.4.1 Jacksmelt (*Atherinopsis californiensis*)

Jacksmelt (*Atherinopsis californiensis*) occurs in California ocean waters in estuaries and coastal marine environments from Yaquina Bay, Oregon to the Gulf of California (Eschmeyer et al. 1983; Robertson and Allen 2002). Although two other silverside species (topsmelt, *Atherinops affinis*, and California grunion, *Leuresthes tenuis*) occur in the project vicinity, neither of these were collected at the intake location, while topsmelt was absent and grunion was rare in the source water samples. Therefore, only jacksmelt is profiled as the main representative of the silverside family, although in the summary tables in the previous section the data are listed as silversides.

3.3.4.1.1 Life History and Ecology

Jacksmelt are small, slim schooling fishes characteristically having a broad silvery band along each side. They are greenish blue above, silver below, with a metallic stripe bordered with blue extending the length of the body. Jacksmelt is the largest member of the three species of the silversides that occur in California, with adults reaching a maximum length of 44 cm (17 in.) (Miller and Lea 1972). The fish reach maturity after two years at a size range of 18 to 20 cm (7.0 to7.8 in.) standard length (SL), and can live to a maximum age of 9–10 years (Clark 1929).

Bays, estuaries, and soft bottom sediments in the surf zone are the primary habitats where jacksmelt are typically most abundant within southern California (Allen et al. 2006; Allen and Pondella 2006). Jacksmelt are found in California bays and ocean waters throughout the year and have been observed at depths of 29 m (95 ft) but are most common in 1.5–15.2 m (5–50 ft) depths. Jacksmelt form larger and denser schools than topsmelt in nearshore areas (Gregory 2001).

The spawning season for jacksmelt is from October through March (Clark 1929), with peak activity from January through March (Allen et al. 1983). Individuals may spawn multiple times during the reproductive season and reproductive females have eggs of various sizes and maturities present in the ovary (Clark 1929). Fecundity has not been well documented but is possibly over 2,000 eggs per female (Emmett et al. 1991). Females lay eggs on marine plants and other floating objects where fertilization by males occurs (Love 2011). Hatch length for jacksmelt ranges from 6 to 9 mm (0.24 to 0.35 in.) (typically 7.5 to 8.5 mm [0.30 to 0.33 in.]) (Moser 1996). Larval growth rate averages approximately 0.37 mm/d (0.01 in./d) for both species based on data from Middaugh et al. (1990). Plankton sampling conducted in Alamitos Bay during a 316(b) study (IRC 1981) found that nearly all silverside larvae were collected in surface samples indicating a strong behavioral tendency for these larvae to actively maintain their position in surface strata, possibly through a phototatic response.

Summary of jacksmelt distribution and life history attributes.

Range: Yaquina Bay, Oregon to Gulf of California

Life History:

- **Size**: up to 44 cm (17 in.)
- **Age at maturity**: 2 yr
- **Life span**: 9–10 yr
- **Spawning**: October through March, with average fecundity of 2,000 eggs

Habitat: Bays, estuaries, rocky intertidal, kelp beds, nearshore waters

- Observed at depths of 29 m (95 ft); most common in depths of 1.5 to 15.2 m (5 to 50 ft)

Fishery: Incidental commercial and limited recreational take on hook and line or with nets

3.3.4.1.2 Population Trends and Fishery

A limited fishery exists for jacksmelt in which they are marketed fresh for human consumption or for bait (Gregory 2001). The commercial fishery for jacksmelt has been conducted with a variety of gears including gillnets, lampara nets, and round haul nets. Historically, set-lines were used in San Francisco Bay for jacksmelt, and during the 1920s beach nets were used at Newport Beach (Gregory 2001). Commercial catches of jacksmelt have varied sharply in California over the past 80 years, fluctuating from more than 2 million pounds in 1945 to only 2,530 pounds in 1998 and 1999. Jacksmelt are not listed in the commercial landings records in the PacFIN database, although they may be caught incidentally in purse seines along with other baitfish species and are therefore not reported as an individual species.

Both topsmelt and jacksmelt make up a significant portion of the recreational fishing catch from piers and along shores. Jacksmelt shore landings declined by over 75 percent in the 1990s compared to the 1980s (Jarvis et al. 2004). Recent statewide catch estimates have fluctuated from a combined high of 234,610 fish in 2004 to lows of less than 100,000 in 2007, 2010, and 2011, with an average of 147,276 caught annually during this time period. Catches of jacksmelt by recreational anglers in southern California from 2004 to 2011 ranged from a high of 153,093 in 2004 to a low of 55,303 in 2011 (**Table 3-7**).

The commercial and recreational catch statistics for jacksmelt and topsmelt, in general, represent incidental catch as they are not targeted by any fishery. Therefore, the large fluctuations in the catch records reflect demand rather than relative abundances.

Jacksmelt are not federally or state listed as threatened or endangered.

Table 3-7. Annual recreational fishery landings for jacksmelt in the Southern California region (Los Angeles to San Diego) based on RecFIN data (values are estimated numbers of fish landed based on sampling).

Year	Jacksmelt
2004	153,093
2005	113,035
2006	131,842
2007	69,919
2008	123,678
2009	118,301
2010	56,399
2011	55,303
Average	**102,696**

3.3.4.1.3 Sampling Results

Jacksmelt (silversides) was the most abundant larval fish taxon collected using towed plankton nets at the intake station (SWE) with a mean concentration of 40.8 per 1,000 m^3 (**Table 3-4**), and also the most abundant taxon at the source water stations with a mean concentration of 26.3 per 1,000 m^3 (**Table 3-6**). The larvae occurred mainly from December through March with peak abundances in January (345 per 1,000 m^3) (**Figure 3-13**). They were absent in samples collected from May through November. Monthly source water concentrations peaked in February (**Figure 3-14**) and overall were lower than concurrent entrainment samples. Jacksmelt larvae were significantly more abundant in nighttime samples than daytime samples (**Figure 3-15**). The length frequency distribution of measured silverside larvae had a unimodal distribution with most larvae in the range of 8–10 mm (0.31–0.39 in.) notochord length (NL) (**Figure 3-16**). The lengths of the larvae collected using towed plankton nets from the intake station samples ranged from 7.3 to 18.0 mm (0.29–0.71 in.) with a mean of 9.9 mm (0.39 in.).

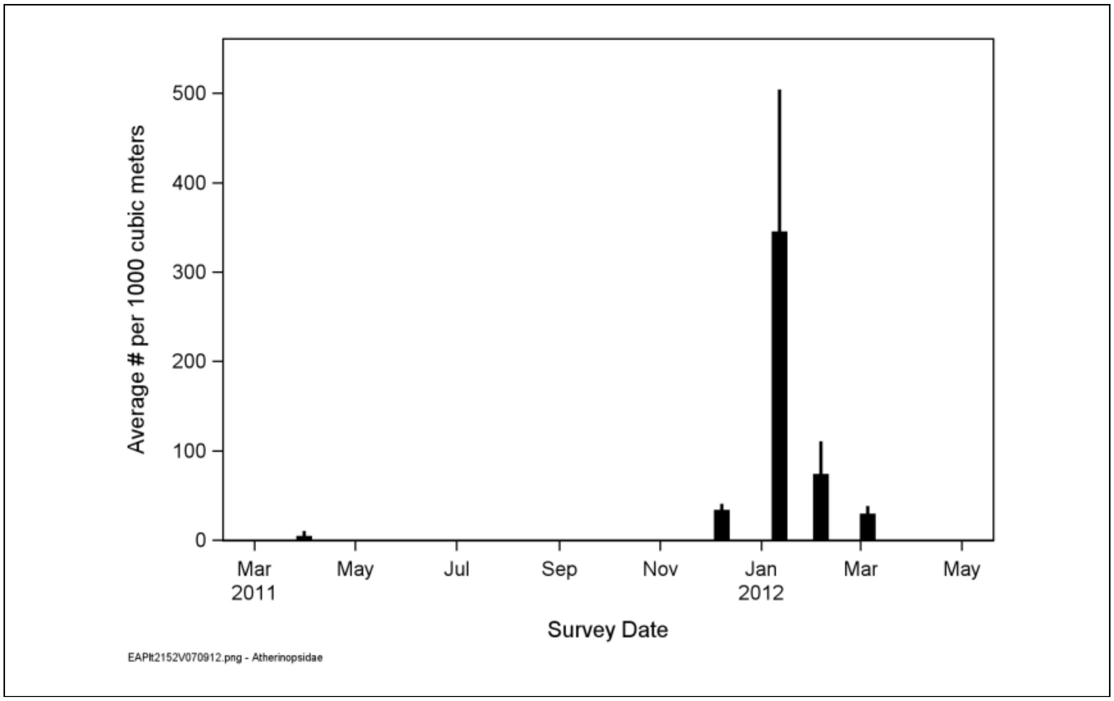


Figure 3-13. Mean concentration (#/1,000 m^3) and standard error for jacksmelt larvae collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

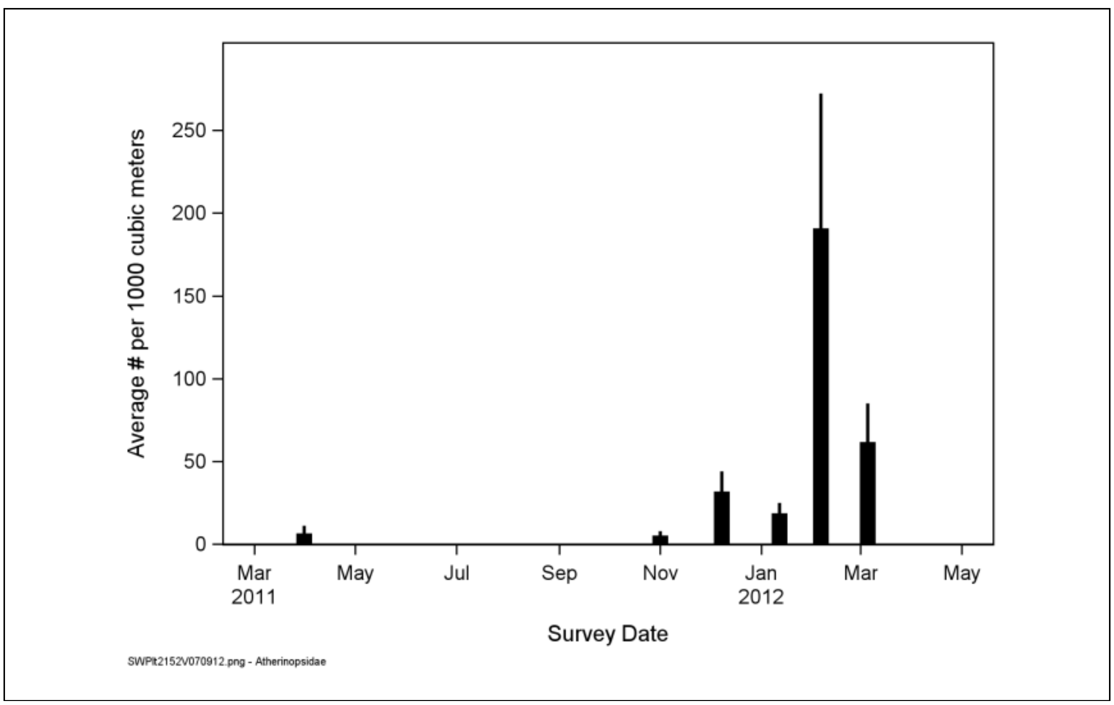


Figure 3-14. Mean concentration (#/1,000 m^3) and standard error for jacksmelt larvae collected using towed plankton nets at source water stations (SW1, SW2, SW3) from March 2011 through March 2012.

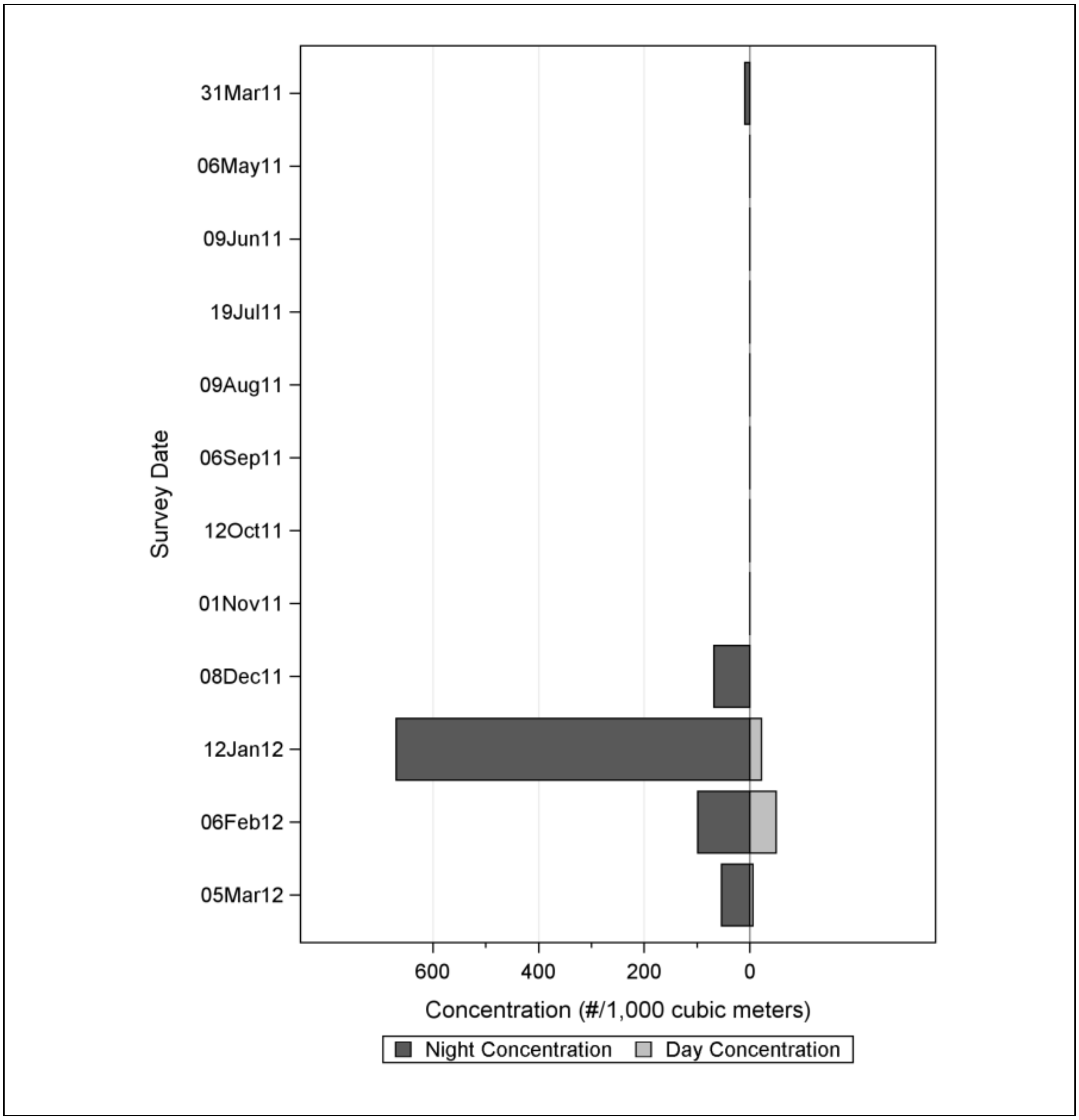


Figure 3-15. Mean concentration (#/1,000 m^3) in night and day samples for jacksmelt larvae collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

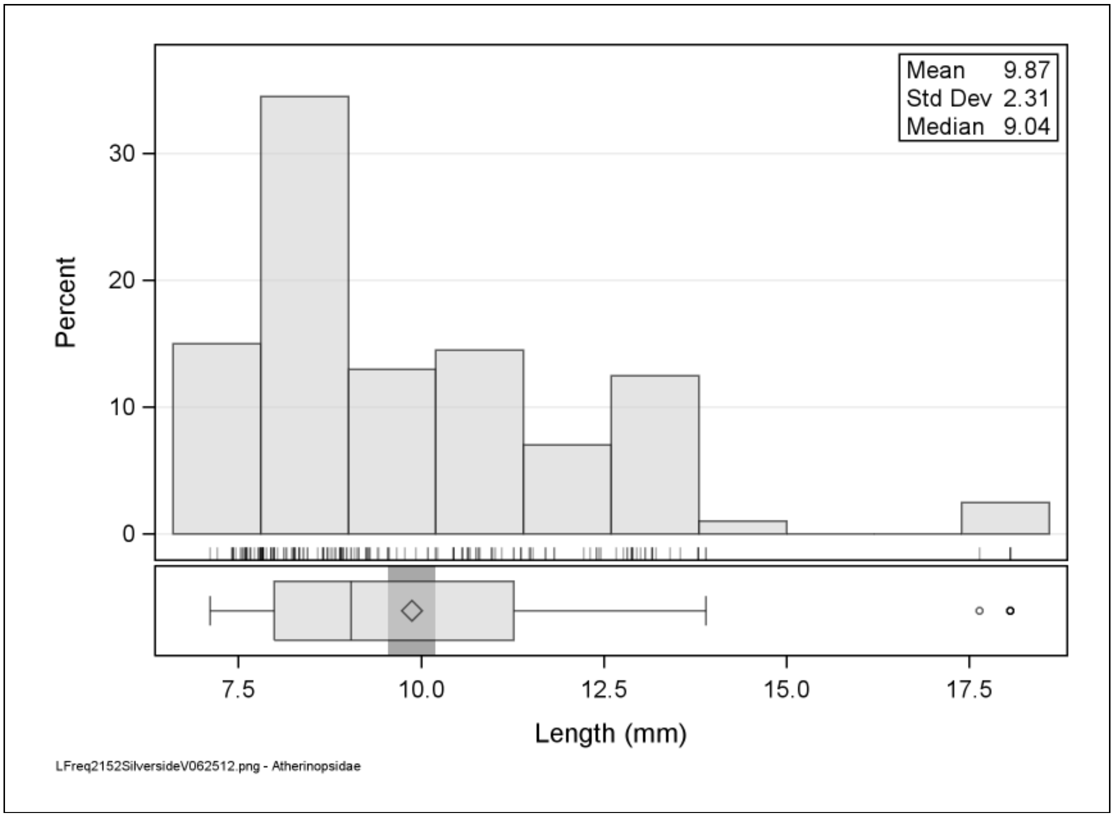


Figure 3-16. Length frequency for jacksmelt larvae collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

Refer to Figure 3.2-4 for symbol explanation.

3.3.4.1.4 Impact Modeling Results

Estimates of 18,983, 28,061, and 2,457,663 jacksmelt larvae would have been entrained through an unscreened intake during the March 2011 to March 2012 study period based, respectively, on daily intake flows of 1,309 m^3 (0.346 mgd) and 1,935 m^3 (0.511 mgd) for the demonstration facility, and a projected flow of 170,722 m^3 (45.1 mgd) for the final proposed facility (**Table 3-5**). All of the silverside larvae collected using towed plankton nets at the intake station (SWE) were jacksmelt, while both jacksmelt and a small number of California grunion were collected at the source water stations. As a result, data for the two species were combined for the impact modeling.

Empirical Transport Model (*ETM*)

The period of time that silverside larvae were vulnerable to entrainment at the WBDDF was estimated from the average age at entrainment, which was based on measurements of 172 larvae. The calculated hatch length from the data resulted in an estimate of 8.2 mm (0.32 in.), which exceeded the 25^{th} percentile estimate, so the 10^{th} percentile estimate, 7.6 mm (0.30 in.) was used as the estimated hatch length. This is consistent with published hatch length estimates for jacksmelt (Moser 1996). The difference in length between the bootstrap estimate of the 95th percentile of 13.62 mm (0.53 in.) and the estimated hatch length, divided by an estimated growth rate of 0.44 mm/d (0.02 in./d) from Middaugh (1990), resulted in an estimated larval exposure to entrainment of 13.7 days.[3] Silversides do not have planktonic eggs so no adjustment to the larval duration was made to account for an egg stage that would be vulnerable to entrainment.

The data used to calculate the ETM estimates (f_i values) for silversides are shown in **Tables 3-8** and **3-9**. Data collected during the February survey period produced the largest estimate of the source water population, while the largest estimated entrainment occurred during the January survey period (**Table 3-8**). Assuming the use of an unscreened intake, the total estimated annual entrainment for the projected flow of 170,722 m^3 (45.1 mgd) for the proposed facility represents less than one-quarter of one percent of the extrapolated source water population potentially subject to entrainment.

The source water estimates in **Table 3-8** were used in calculating the weights (f_i) used in the ETM calculations, which showed that over 70 percent of the larvae collected using towed plankton nets at the source water stations were collected during the January and February surveys (**Table 3-9**). Jacksmelt are typically associated with shallow nearshore areas but have been reported to depths of 29 m (95 ft) (Love 2011). Based on the nearshore distribution of jacksmelt and other silversides, the results of the ETM analysis based on the back-projections that are limited to depths of 30 m (98 ft) are likely to be more representative of actual impacts on the population, but the second set of results may provide a more realistic estimate of the potential source area where the larvae could have been hatched. The average alongshore displacement for the source water population was 7.58 km (4.71 mi) and 30.95 km (19.23 mi) for the back-projections limited to depths of 30 m (98 ft) and 100 m (328 ft), respectively.

The ETM estimates attributable to entrainment through an unscreened intake for silversides ranged from 0.00006 (0.006 percent) to 0.00098 (0.098 percent) for the WBDDF intake volumes, to a high of 0.079 (7.9 percent) for the intake volume at the proposed facility (**Table 3-10**). The high estimate for the proposed facility is due to higher intake volumes, but also that concentrations of silverside larvae were higher at the intake location (SWE) relative to the source water stations. Silversides attach eggs to hard substrates and submerged vegetation such as occurs at the breakwater and rock armoring adjacent to the intake structure, and therefore, it is not surprising that the larval concentrations were highest at the intake location.

[3] The entrainment exposure duration was calculated from size and growth values with greater decimal precision than those shown, and differs slightly from the duration calculated using these rounded values.

Table 3-8. Daily and survey period estimates for silversides for entrainment, sampled source water, and extrapolated source water. The entrainment estimate was calculated based on the projected flow for the proposed facility of 170,722 m^3 (45.1 mgd) through an unscreened intake. Extrapolated source water population calculated based on a depth of 30 m (98 ft).

Survey Date	Survey Days	Daily Entrainment Estimate (45.1 mgd)	Survey Entrainment Estimate (45.1 mgd)	Daily Sampled SW Estimate	Sampled SW Survey Period Estimate	Extrapolated SW Survey Population
31 Mar 11	31	882	27,341	115,021	3,565,660	17,937,351
06 May 11	35	-	-	-	-	-
09 Jun 11	37	-	-	-	-	-
19 Jul 11	31	-	-	-	-	-
09 Aug 11	24	-	-	-	-	-
06 Sep 11	32	-	-	-	-	-
12 Oct 11	28	-	-	-	-	-
01 Nov 11	29	-	-	68,236	1,978,837	24,839,303
08 Dec 11	36	5,824	209,676	657,125	23,656,517	242,523,736
12 Jan 12	30	59,021	1,770,639	2,349,954	70,498,626	100,133,360
06 Feb 12	26	12,712	330,524	3,536,696	91,954,090	651,044,065
05 Mar 12	27	5,092	137,481	1,141,078	30,809,111	45,125,758
	Annual Totals		**2,475,661**		**222,462,842**	**1,081,603,573**

Table 3-9. Data used in ETM calculations for silversides. The estimates of PE_i for each survey period i, represent the unadjusted source water estimates and were calculated using only towed plankton net data from the intake and source water stations. The *PE*s were calculated based on three unscreened intake volumes: a) actual flow during the year for the WBDDF of 240 gpm or 1,309 m^3 (0.346 mgd), b) design flow for the WBDDF of 355 gpm or 1,935 m^3 (0.511 mgd), and c) a projected flow for the proposed facility of 170,722 m^3 (45.1 mgd).

	a) Actual Average Demonstration Plant Flow = 240 gpm		b) Demonstration Plant Design Flow = 355 gpm		c) Final Desalination Plant Intake Flow = 45.1 mgd			Source Water Extrapolation Alongshore (km)	
Survey Date	PE_i	PE_i Std. Error	PE_i	PE_i Std. Error	PE_i	PE_i Std. Error	Weight (f_i)	Depth (m) = 30	Depth (m) = 100
31-Mar-11	0.000059	0.000065	0.000087	0.000096	0.007668	0.008475	0.016	7.55	26.95
6-May-11	0	0	0	0	0	0	0.000	15.42	61.91
9-Jun-11	0	0	0	0	0	0	0.000	11.49	19.34
19-Jul-11	0	0	0	0	0	0	0.000	10.80	23.22
9-Aug-11	0	0	0	0	0	0	0.000	46.35	50.04
6-Sep-11	0	0	0	0	0	0	0.000	26.17	30.12
12-Oct-11	0	0	0	0	0	0	0.000	8.08	34.86
1-Nov-11	0	0	0	0	0	0	0.009	18.83	21.12
8-Dec-11	0.000068	0.000030	0.000100	0.000045	0.008863	0.003958	0.106	15.38	33.36
12-Jan-12	0.000193	0.000111	0.000285	0.000163	0.025116	0.014413	0.317	2.13	31.48
6-Feb-12	0.000028	0.000017	0.000041	0.000025	0.003594	0.002228	0.413	10.62	23.55
5-Mar-12	0.000034	0.000013	0.000051	0.000019	0.004462	0.001706	0.139	2.20	39.39
Average (n=12)	**0.000032**		**0.000047**		**0.004142**			**14.59**	**32.95**
Average (*PE*>0)	**0.000076**		**0.000113**		**0.009941**			**7.58**	**30.95**

Table 3-10. ETM estimates of P_M for silversides. ETM estimates of P_M based on estimated upcoast and downcoast source water populations based on back-projections limited to depths of 30 m (98 ft) and 100 m (328 ft). ETM estimates calculated for three unscreened intake volumes: a) actual flow during the year for the WBDDF of 1,309 m3 (0.346 mgd), b) design flow for the WBDDF of 1,935 m3 (0.511 mgd), and c) a projected flow for the proposed facility of 170,722 m3 (45.1 mgd). Two estimates of standard error (Std. Error) are presented: one based on combining the variance components from the estimates comprising ETM, and the second using the weighted average coefficient of variation across the PEs for the twelve surveys.

Flow	CODAR Depth Limit	P_M Estimate	P_M Std. Error	P_M + Std. Error	P_M - Std. Error	CV Based Std. Error	P_M + CV Std. Error	P_M - CV Std. Error
a) WBDDF Actual Flow	30 m Depth	0.000666	0.151555	0.152221	-0.150888	0.000371	0.001037	0.000296
	100 m Depth	0.000057	0.151655	0.151713	-0.151598	0.000032	0.000089	0.000025
b) WBDDF Design Flow	30 m Depth	0.000984	0.151502	0.152487	-0.150518	0.000548	0.001532	0.000437
	100 m Depth	0.000085	0.151651	0.151736	-0.151566	0.000047	0.000132	0.000038
c) Proposed Design Flow	30 m Depth	0.078782	0.143839	0.222621	-0.065057	0.043819	0.122601	0.034964
	100 m Depth	0.007440	0.150482	0.157922	-0.143041	0.004138	0.011579	0.003302

3.3.4.2 White Croaker (*Genyonemus lineatus*)

White croaker (*Genyonemus lineatus*) range from Todos Santos Bay, Baja California, Mexico north to Barkley Sound in British Columbia, Canada (Miller and Lea 1972). They are one of eight species of croakers (Family Sciaenidae) found off California. The other croakers include: white seabass (*Atractoscion nobilis*), black croaker (*Cheilotrema saturnum*), queenfish (*Seriphus politus*), California corbina (*Menticirrhus undulatus*), spotfin croaker (*Roncador stearnsii*), yellowfin croaker (*Umbrina roncador*), and shortfin corvina (*Cynoscion parvipinnis*).

3.3.4.2.1 Life History and Ecology

The white croaker is a deep-bodied elongate fish with a bluntly rounded head and small barbels on its lower jaw. Its color is incandescent brownish to yellowish on the back becoming silvery below. White croaker are found in shallow-water areas both in the ocean and in larger bays. The white croaker is an abundant nearshore species that prefers the sandy bottoms of bays and estuaries and the area just outside the surf zone. White croaker may move offshore into deeper water during winter months (Allen and DeMartini 1983) however, this pattern is apparent only south of Redondo Beach (Herbinson et al. 2001). The reported depth range of white croaker is from near the surface to depths of 238 m (781 ft) (Love et al. 2005) however, in southern California, Allen (1982) found white croaker over soft bottoms between 10 and 130 m (32.8 and 426.5 ft), and it was most frequently collected at 10 m (32.8 ft).

White croaker are oviparous broadcast spawners. They mature between about 130 and 190 mm (5.1 and 7.5 in.) total length (TL), somewhere between the first and fourth year. About one-half of males mature by 140 mm (5.5 in.) TL, and one-half of females by 150 mm (5.9 in.) TL, and all fishes are mature by 190 mm (7.5 in.) TL in their third to fourth year (Love et al. 1984). Off Long Beach, California, white croaker spawn primarily from November through August, with peak spawning from January through March (Love et al. 1984). However, in central California some spawning can occur year-round. Moser (1996) states that white croaker spawning in the CalCOFI area is most abundant from December to April with peak spawning occurring in March. In southern California, white croaker larvae were generally more abundant in the upper 30 m (98 ft) of the water column at their inshore stations (Moser and Pommeranz 1999). Batch fecundities ranged from about 800 eggs in a 155 mm (6.1 in.) female to about 37,200 eggs in a 260 mm (10.5 in.) female, with spawning taking place as often as every five days (Love et al. 1984). In their first and second year, females spawn for three months for a total of about 18 times per season. Older individuals spawn for about four months and about 24 times per season (Love et al. 1984). Some older fish may spawn for seven months. The nearshore waters from Redondo Beach (Santa Monica Bay, California) to Laguna Beach, California, are considered an important spawning center for this species (Love et al. 1984). A smaller spawning center occurs off Ventura.

Newly hatched white croaker larvae are 1–2 mm (0.04–0.08 in.) SL and are not well developed (Watson 1982). Larvae are principally located within 4 km (2.5 mi) from shore, and as they develop tend to move shoreward and into the epibenthos (Schlotterbeck and Connally 1982). Murdoch et al. (1989) estimated a daily larval growth rate of 0.20 mm/d (0.008 in./d). Using otolith analysis, Miller et al. (2011) aged field-collected larvae and found that the smallest individual was 1.84 mm (0.07 in.) with an estimated age of 10 days, while the largest specimen was 10.18 mm (0.40 in.) with an estimated age of 38 days. White croaker grow at a fairly constant rate throughout their lives, though females increase in size more rapidly than males from age one (Moore 2001). Maximum reported size is 41.4 cm (16.3 in.) (Miller and Lea 1972), with a life span of 12 to 15 years (Frey 1971, Love et al. 1984). No mortality estimates are available for any of the life stages of this species.

White croaker are primarily nocturnal benthic feeders, although juveniles may feed in the water column during the day (Allen 1982). Important prey items include polychaetes, amphipods, shrimps, and chaetognaths. In Outer Los Angeles Harbor, Ware (1979) found that important prey items included polychaetes, benthic crustaceans, free-living nematodes, and zooplankton. Younger individuals feed on holoplanktonic crustaceans and polychaete larvae.

Summary of white croaker distribution and life history attributes.

Range: Todos Santos Bay, Baja California, Mexico north to Barkley Sound, Vancouver Island, B.C., Canada
Life History:

- **Maximum size**: 414 mm (16.3 in.)
- **Age at maturity**: between about 130 and 190 mm (5.1 and 7.5 in.), somewhere between the first and fourth year
- **Life span**: 12–15 yr
- **Spawning**: Off Long Beach- primarily from November through August, with peak spawning from January through March. In CalCOFI area- from December to April with peak spawning occurring in March. Spawning can occur year-round in central California.

Habitat: Shallow-water areas both in the ocean and in larger bays

Fishery: Commercial and recreational

3.3.4.2.2 Population Trends and Fishery

White croaker is an important constituent of commercial and recreational fisheries in California. Prior to 1980, most commercial catches of white croaker were taken by otter trawl, round haul net (lampara), gill net, and hook and line in southern California, but after 1980 most commercial catches were taken primarily by trawl and hook and line (Love 2011). Also, since then the majority of the commercial fishery shifted to central California near Monterey mainly due to the increased demand for this species from the developing fishery by Southeast Asian refugees

(Moore and Wild 2001). Most of the recreational catch still occurs in southern California from piers, breakwaters, and private and sport boats.

Before 1980, state-wide white croaker landings averaged 685,000 pounds annually, exceeding 1 million pounds for several years (Moore and Wild 2001). High landings in 1952 probably occurred due to the collapse of the Pacific sardine fishery. Since 1991, landings averaged 461,000 pounds and steadily declined to an all-time low of 142,500 pounds in 1998. Although considered a minor commercial fishery species, annual landings in Los Angeles County since 2004 have declined dramatically with approximately 20,000 pounds landed in 2004, 2005, and 2008, but none landed in 2011 (PacFIN 2012).

Landings by recreational fishermen aboard commercial passenger fishing vessels (CPFVs) averaged about 12,000 fish per year from 1990 to 1998, with most of the catch coming from southern California. Sport fishery annual catch estimates of white croaker in the southern California region ranged from 22,450–131,864 fish from 2004–2011, with an average of 66,017 fish caught annually (RecFIN 2012) (**Table 3-11**).

Table 3-11. Annual landings and revenue for white croaker based on RecFIN data (Los Angeles–San Diego region) and PacFIN data (Los Angeles County), 2004–2011.

	Recreational Fishery	Commercial Fishery	
Year	Estimated Numbers Landed	Landed Weight (lb)	Revenue ($)
2004	92,636	19,608	$14,653
2005	131,864	24,652	$17,531
2006	44,232	15,020	$11,079
2007	35,859	11,977	$4,000
2008	40,184	20,235	$2,083
2009	126,845	2,546	$1,150
2010	34,068	3,108	$242
2011	22,450	-	-
Average	**66,017**	**13,878**	**$7,248**

Although recreational and commercial catches of white croaker showed a declining trend from 1980–2006, fishery-independent data from trawls did not indicate a similar trend, suggesting the recreational fishery trend is due to changes in fishing practices since 1980 (D. Pondella, Vantuna Research Group, pers. comm.). In the Ocean Resources Enhancement and Hatchery Program (OREHP) monitoring program, the catch per sampling period increased over the sample period. National Pollutant Discharge Elimination System (NPDES) trawl data suggested a similar pattern with catches of white croaker from 1978–2006 oscillating, but without a significant trend over the study period. Furthermore, these catches were not correlated with any oceanographic parameters, such as sea surface temperature.

There are other studies, however, that suggest declining populations in southern California. Mean larval concentrations of white croaker near the RBGS intakes in King Harbor were approximately ten times greater in 1979–1980 (SCE 1983) than during similar studies conducted in 2006 (MBC and Tenera 2007). Annual relative abundance of white croaker in impingement samples at southern California power plants showed decreases particularly during the strong El Niño events of 1982–83, 1986–87, and 1997-98 as compared with non-El Niño years (Herbinson et al. 2001).

3.3.4.2.3 Sampling Results

White croaker was the second most abundant larval fish taxon collected using towed plankton nets at the intake station (SWE) with a mean concentration of 16.5 per 1,000 m^3 (**Table 3-4**), and was the third most abundant taxon at the source water stations with a mean concentration of 20.9 per 1,000 m^3 (**Table 3-6**). The larvae occurred mainly from September through February with peak abundances in October at both the entrainment and source water stations (**Figures 3-17** and **3-18**). Source water concentrations were generally higher than concurrent entrainment samples. The larvae were generally more abundant in nighttime samples than daytime samples except during the July 2011 and February 2012 surveys (**Figure 3-19**). Most white croaker larvae were recently hatched as the mean size of 2.44 mm (0.10 in.) was only slightly larger than the reported hatch length of 1–2 mm (0.04–0.08 in.) (**Figure 3-20**). The lengths of the larvae from the intake station samples ranged from approximately 1.2 to 4.8 mm (0.05–0.19 in.) NL.

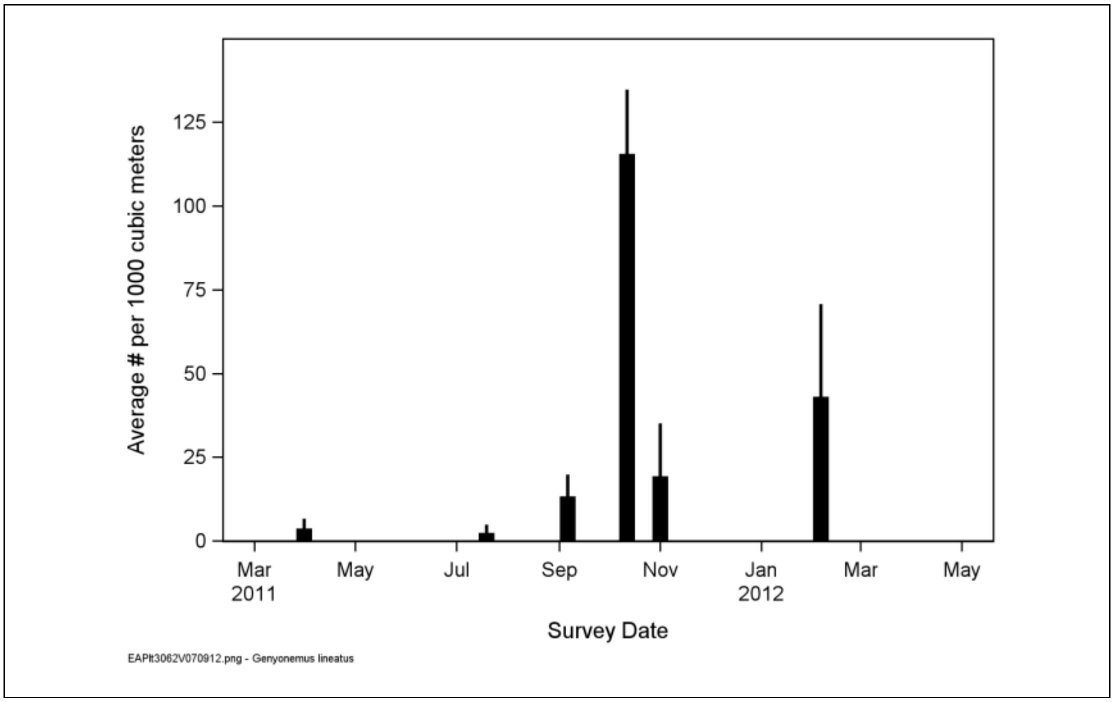


Figure 3-17. Mean concentration (#/1,000 m^3) and standard error for white croaker larvae collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

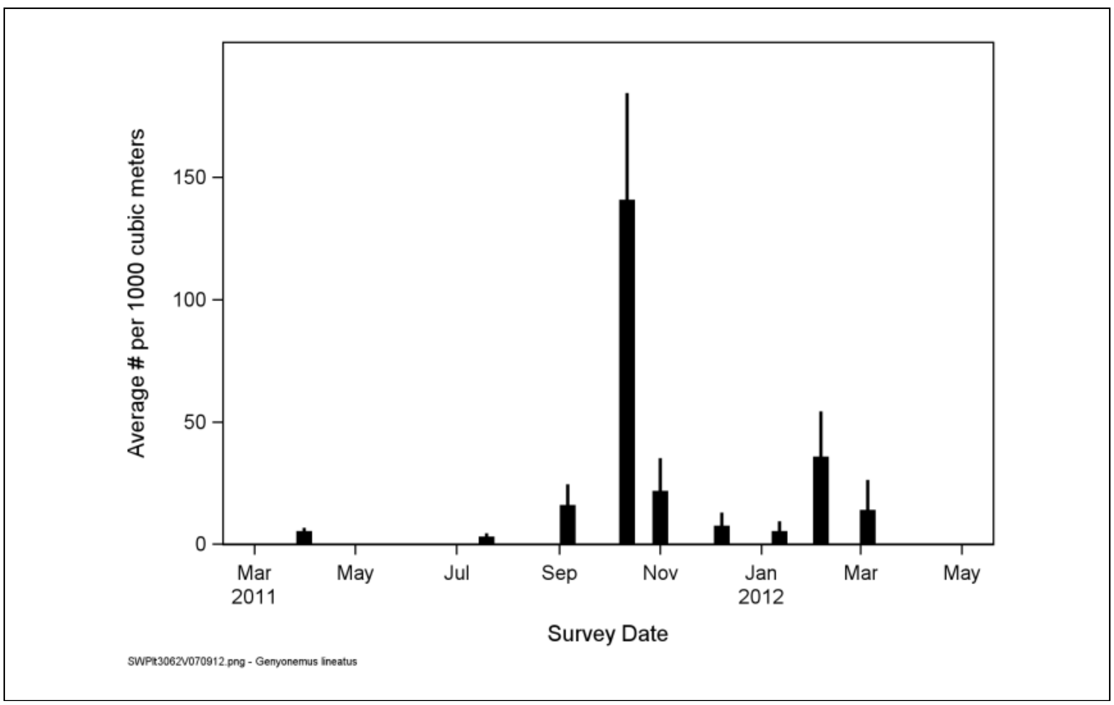


Figure 3-18. Mean concentration (#/1,000 m^3) and standard error for white croaker larvae collected using towed plankton nets at source water stations (SW1, SW2, SW3) from March 2011 through March 2012.

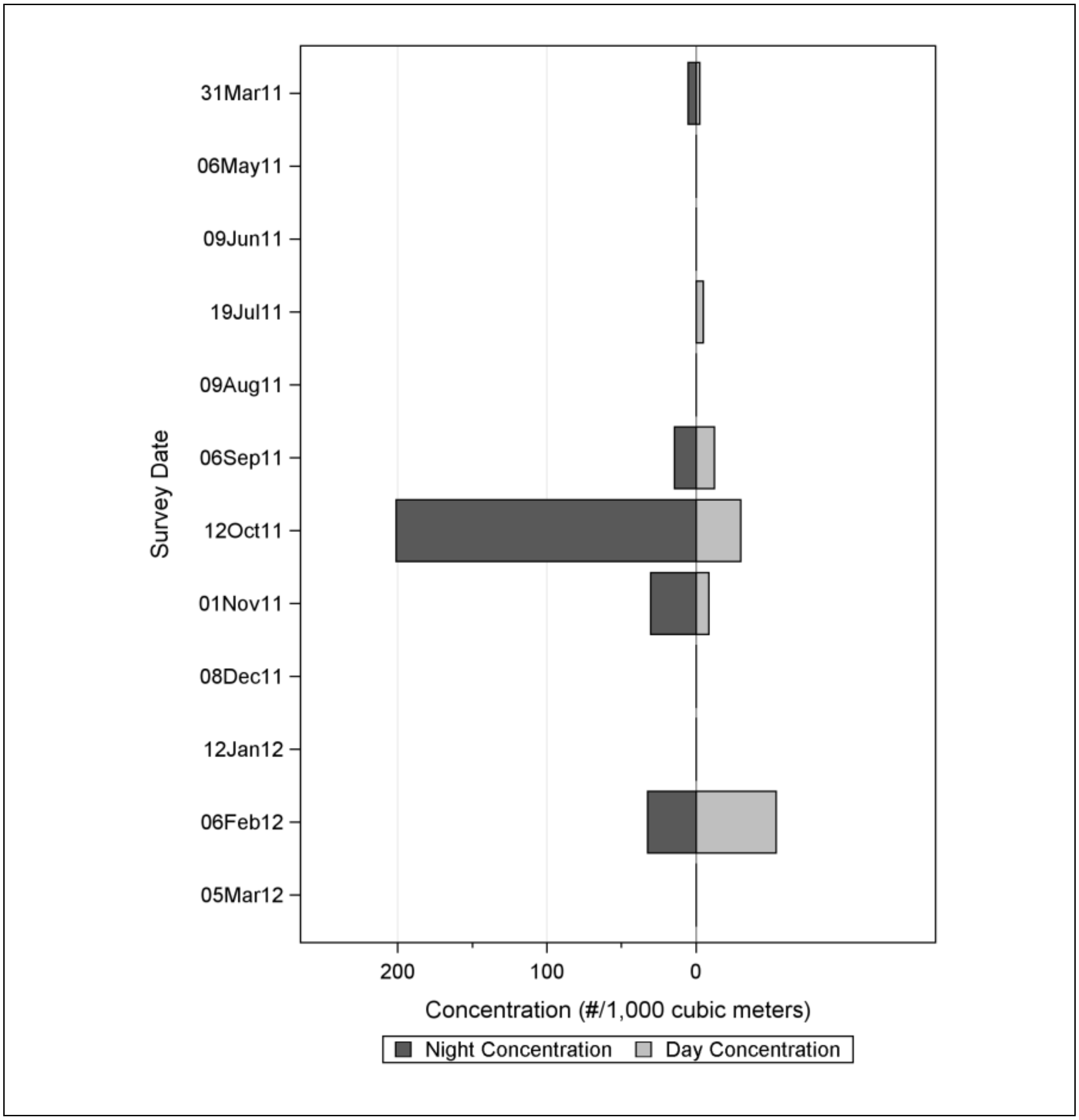


Figure 3-19. Mean concentration (#/1,000 m^3) in night and day samples for white croaker larvae collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

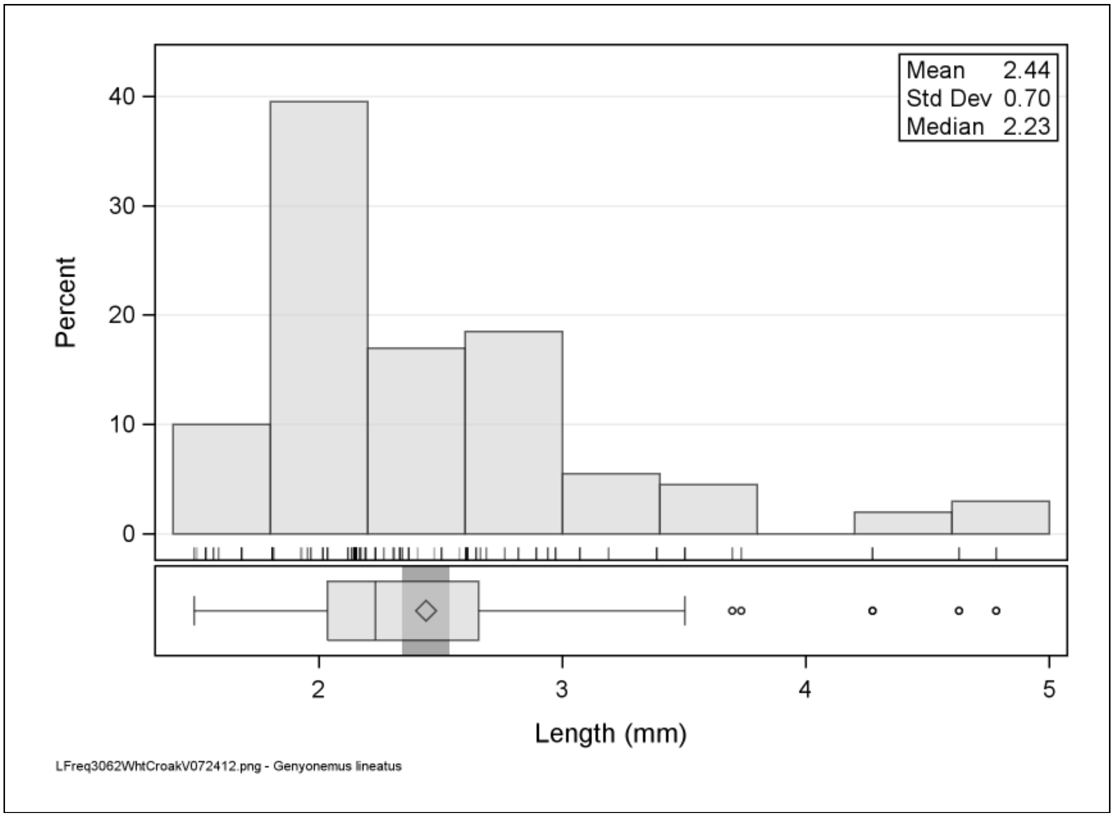


Figure 3-20. Length frequency for white croaker larvae collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

Refer to Figure 3.2-4 for symbol explanation.

3.3.4.2.4 Impact Modeling Results

Estimates of 7,251, 10,718, and 945,568 white croaker larvae would have been entrained through an unscreened intake during the March 2011 to March 2012 study period based, respectively, on daily intake flows of 1,309 m^3 (0.346 mgd) and 1,935 m^3 (0.511 mgd) for the demonstration facility, and a projected flow of 170,722 m^3 (45.1 mgd) for the final proposed facility (**Table 3-5**).

Empirical Transport Model (*ETM*)

The period of time that white croaker larvae were vulnerable to entrainment at the WBDDF was estimated from the average age at entrainment, which was based on measurements of 60 larvae. The calculated hatch length from the data resulted in an estimate of 1.79 mm (0.07 in.), which exceeded the 25th percentile estimate, so the 10th percentile estimate, 1.83 mm (0.07 in.) was used as the estimated hatch length. This is consistent with published hatch length estimates for white croaker (Moser 1996). The difference in length between the bootstrap estimate of the 95th percentile of 4.38 mm (0.17 in.) and the estimated hatch length, divided by an estimated growth rate of 0.25 mm/d (0.01 in./d) from Moser (1996), after adding the estimated egg duration of 2.2 d, resulted in an estimated larval exposure to entrainment of 12.4 days.[4]

The data used to calculate the ETM estimates (f_i values) for white croaker are shown in **Tables 3-12** and **3-13**. Data collected during the October survey period produced the largest estimate of the source water population (**Table 3-12**). The largest estimated entrainment also occurred during the October survey period (**Table 3-12**). The total estimated annual entrainment through an unscreened intake for the projected flow of 170,722 m^3 (45.1 mgd) for the proposed facility represents less than 0.03 percent of the extrapolated source water population potentially subject to entrainment.

The source water estimates in **Table 3-12** were used in calculating the weights (f_i) used in the ETM calculations, which showed that over 60 percent of the larvae collected using towed plankton nets at the source water stations were collected during the October and November surveys (**Table 3-13**).

The data used to calculate the ETM estimates (f_i values) for white croaker showed that over 50 percent of the larvae were collected from the source water stations during the October 2011 survey (**Table 3-13**). White croaker are typically associated with nearshore areas along sandy beaches but have been reported to depths of 101 m (330 ft) (Miller and Lea 1972). Based on the primarily nearshore distribution of white croaker, ETM analyses were done using depths of both 30 m (98 ft) and 100 m (330 ft) as limits on the back-projections. The average alongshore displacement for the source water population was 12.41 km (7.71 mi) and 24.67 km (15.33 mi) for the back-projections limited to depths of 30 m (98 ft) and 100 m (328 ft), respectively.

The ETM estimates attributable to entrainment through an unscreened intake for white croaker ranged from 0.00003 (0.003 percent) to 0.00015 (0.015 percent) for the WBDDF intake volumes, to a high of 0.014 (1.4 percent) for the intake volume at the proposed facility (**Table 3-14**). The higher estimate for the proposed facility is due to higher intake volume.

[4] The entrainment exposure duration was calculated from size and growth values with greater decimal precision than those shown, and differs slightly from the duration calculated using these rounded values.

Table 3-12. Daily and survey period estimates for white croaker for entrainment, sampled source water, and extrapolated source water. The entrainment estimate was calculated based on the projected flow for the proposed facility of 170,722 m^3 (45.1 mgd) through an unscreened intake. Extrapolated source water population calculated based on a depth of 100 m (328 ft).

Survey Date	Survey Days	Daily Entrainment Estimate (45.1 mgd)	Survey Entrainment Estimate (45.1 mgd)	Daily Sampled SW Estimate	Sampled SW Survey Period Estimate	Extrapolated SW Survey Population
31 Mar 11	31	655	20,312	123,367	3,824,370	55,081,873
06 May 11	35	-	-	-	-	-
09 Jun 11	37	-	-	-	-	-
19 Jul 11	31	413	12,788	76,395	2,368,236	36,344,244
09 Aug 11	24	-	-	-	-	-
06 Sep 11	32	2,271	72,663	466,676	14,933,640	298,905,373
12 Oct 11	28	19,726	552,321	3,428,609	96,001,043	2,231,174,961
01 Nov 11	29	3,317	96,189	669,814	19,424,598	267,363,273
08 Dec 11	36	-	-	149,326	5,375,752	115,737,687
12 Jan 12	30	-	-	122,769	3,683,077	75,129,975
06 Feb 12	26	7,357	191,293	891,491	23,178,773	276,314,774
05 Mar 12	27	-	-	367,741	9,929,017	260,725,071
	Annual Totals		**945,567**		**178,718,507**	**3,616,777,232**

Table 3-13. Data used in ETM estimates of P_M for white croaker. The estimates of PE_i for each survey period i, represent the unadjusted source water estimates and were calculated using only towed plankton net data from the intake and source water stations. The *PE*s were calculated based on three unscreened intake volumes: a) actual flow during the year for the WBDDF of 240 gpm or 1,309 m^3 (0.346 mgd), b) design flow for the WBDDF of 355 gpm or 1,935 m^3 (0.511 mgd), and c) a projected flow for the proposed facility of 170,722 m^3 (45.1 mgd).

	a) Actual Average Demonstration Plant Flow = 240 gpm		b) Demonstration Plant Design Flow = 355 gpm		c) Final Desalination Plant Intake Flow = 45.1 mgd			Source Water Extrapolation Alongshore (km)	
Survey Date	PE_i	PE_i Std. Error	PE_i	PE_i Std. Error	PE_i	PE_i Std. Error	Weight (f_i)	Depth (m) = 30	Depth (m) = 100
31-Mar-11	0.000041	0.000036	0.000060	0.000054	0.005311	0.004733	0.021	7.55	21.60
6-May-11	0	0	0	0	0	0	0.000	6.61	60.46
9-Jun-11	0	0	0	0	0	0	0.000	11.34	18.32
19-Jul-11	0.000041	0.000047	0.000061	0.000069	0.005400	0.006081	0.013	10.60	23.02
9-Aug-11	0	0	0	0	0	0	0.000	45.60	48.84
6-Sep-11	0.000037	0.000020	0.000055	0.000029	0.004866	0.002560	0.084	26.07	30.02
12-Oct-11	0.000044	0.000011	0.000065	0.000016	0.005753	0.001428	0.537	8.08	34.86
1-Nov-11	0.000038	0.000034	0.000056	0.000050	0.004952	0.004411	0.109	18.36	20.65
8-Dec-11	0	0	0	0	0	0	0.030	14.31	32.29
12-Jan-12	0	0	0	0	0	0	0.021	2.04	30.60
6-Feb-12	0.000063	0.000046	0.000094	0.000068	0.008253	0.005988	0.130	3.79	17.88
5-Mar-12	0	0	0	0	0	0	0.056	2.20	39.39
Average (n=12)	**0.000022**		**0.000033**		**0.002878**			**13.05**	**31.49**
Average (*PE*>0)	**0.000044**		**0.000065**		**0.005756**			**12.41**	**24.67**

Table 3-14. ETM estimates of PM for white croaker. Estimates of PM based on estimated upcoast and downcoast source water populations based on back-projections limited to depths of 30 m (98 ft) and 100 m (328 ft). ETM estimates calculated for three unscreened intake volumes: a) actual flow during the year for the WBDDF of 1,309 m3 (0.346 mgd), b) design flow for the WBDDF of 1,935 m3 (0.511 mgd), and c) a projected flow for the proposed facility of 170,722 m3 (45.1 mgd). Two estimates of standard error (Std. Error) are presented: one based on combining the variance components from the estimates comprising ETM, and the second using the weighted average coefficient of variation across the PEs for the twelve surveys.

Flow	CODAR Depth Limit	P_M Estimate	P_M Std. Error	P_M + Std. Error	P_M - Std. Error	CV Based Std. Error	P_M + CV Std. Error	P_M - CV Std. Error
a) WBDDF Actual Flow	30 m Depth	0.000105	0.086163	0.086268	-0.086059	0.000042	0.000147	0.000063
	100 m Depth	0.000028	0.086172	0.086200	-0.086144	0.000011	0.000039	0.000017
b) WBDDF Design Flow	30 m Depth	0.000155	0.086158	0.086312	-0.086003	0.000062	0.000217	0.000092
	100 m Depth	0.000042	0.086170	0.086212	-0.086129	0.000017	0.000058	0.000025
c) Proposed Design Flow	30 m Depth	0.013505	0.084797	0.098302	-0.071292	0.005432	0.018938	0.008073
	100 m Depth	0.003657	0.085783	0.089440	-0.082126	0.001471	0.005128	0.002186

3.3.4.3 Northern Anchovy (*Engraulis mordax*)

The northern anchovy (*Engraulis mordax*) is a clupeoid fish (anchovy, sardine, herring, and pilchards) belonging to the family Engraulidae (the anchovies). Engraulidae contains 139 species of anchovies that occur throughout the world (Moyle and Cech 1988). Three species of anchovy inhabit nearshore areas of southern California: northern anchovy (*Engraulis mordax*), deepbody anchovy (*Anchoa compressa*) and slough anchovy (*Anchoa delicatissima*). This analysis of entrainment effects on anchovies will concentrate on life history aspects of the northern anchovy because all of the Engraulid larvae collected that were large enough to be positively identified were northern anchovy.

Northern anchovy range from the Queen Charlotte Islands, British Columbia, Canada to Magdalena Bay, Baja California, Mexico and anchovy have recently colonized the Gulf of California. Three genetically distinct subpopulations are recognized for northern anchovy: (1) Northern subpopulation, from northern California to British Columbia; (2) Central subpopulation, off southern California and northern Baja California; and (3) Southern subpopulation, entirely within Mexican waters along the southern coast of Baja California (Emmett et al. 1991).

3.3.4.3.1 Life History and Ecology

Northern anchovy are small elongate fishes with blue or green backs and silver bellies. This pelagic marine species occurs from surface waters down to depths of 305 m (1,000 ft) (Love 2011). They can be found in ocean waters from just outside the surf zone to more than 483 km (300 mi) offshore, however they are most common within 161 km (100 mi) of the shore. Northern anchovy make extensive seasonal migrations up and down the coast, as well as offshore and inshore (Love 2011). There is a great deal of regional variation in age composition and size with older and larger individuals further offshore and to the north (Parrish et al. 1985). These patterns are accentuated during warm years such as El Niño and when abundance is high (Methot 1989). Densities of juvenile anchovy in nearshore areas have been shown to be about ten times higher than in other habitat areas. Methot (1989) concluded that nearshore habitats supported at least 70 percent of the juvenile anchovy population. Juveniles and adults move offshore in winter and return toward shore in spring. They are found in dense schools within bays and estuaries in the spring, summer, and fall.

Northern anchovy spawn throughout the year off southern California, with peak spawning between February and May (Brewer 1978). Moser and Smith (1993) analyzed the CalCOFI larval fish data from 1951 to 1984 and found that northern anchovy generally spawned from January to April with peak spawning in March. A fall spawning stock may occur in central

California and the offshore areas of the Southern California Bight. Most spawning takes place within 100 km (62.1 mi) from shore (MBC 1987). On average, female anchovy off Los Angeles spawn every 7 to 10 days during peak spawning periods, approximately 20 times per year (Hunter and Macewicz 1980; MBC 1987). Northern anchovy off southern and central California can reach sexual maturity by the end of their first year of life, with all individuals being mature by four years of age (Clark and Phillips 1952). The maturation rate of younger individuals is dependent on water temperature (Bergen and Jacobsen 2001). Love (2011) reported that they release 2,700 to 16,000 eggs per batch, with an annual fecundity of up to 130,000 eggs per year in southern California. Parrish et al. (1986) and Butler et al. (1993) stated that the total annual fecundity for one-year-old females was 20,000 to 30,000 eggs, while a five-year-old could release up to 320,000 eggs per year.

Eggs are found from the surface to 50 m (164 ft) and larvae are found from the surface to 75 m (246 ft) in epipelagic and nearshore waters (Garrison and Miller 1982). The eggs hatch in 2–4 d and larvae begin their pelagic phase at approximately 3 mm (0.12 in.) TL (Wang 1986). After hatching, larvae are inactive and float motionless in the water except during short bursts of intense swimming at about one-minute intervals. Moser and Pommeranz (1999) found about 90 percent of the northern anchovy larvae occurred in the upper 30 m (98 ft) of the water column. Northern anchovy larvae feed on dinoflagellates, rotifers, and copepods (MBC 1987). The larval phase lasts for approximately 70 days. Larvae begin schooling at 11–12 mm (0.43–0.47 in.) and transform into juveniles at 35–40 mm (1.38 to 1.57 in.) (Hart 1973; Hunter and Coyne 1982).

Growth in length is most rapid during the first four months and growth in weight is most rapid during the first year (Hunter and Macewicz 1980). Collins (1969) presented age at length and weight at length regressions based on data from the southern California reduction fishery from which an average age-1 fish was estimated as 115 mm (4.5 in.) and 14.9 g (0.5 oz). Northern anchovy reach 102 mm (4 in.) in their first year and 119 mm (4.7 in.) in their second (Sakagawa and Kimura 1976). They have been reported to reach a size of 229 mm (9 in.) but rarely exceed 178 mm (7 in.) (Miller and Lea 1972). All anchovy are mature by the time they reach four years in age; however, anchovy in the central sub-population are all sexually mature at age two while some anchovy may mature within their first year in the southern parts of their range (Love 2011). Maximum age is about seven years, though most live less than four years (Hart 1973).

Northern anchovy are facultative filter-feeding planktivores. They primarily feed on zooplankton including copepods and other crustaceans (various life stages), but also consume phytoplankton and the pelagic eggs and larvae of other fishes. Northern anchovy feed largely during the night, though they were previously thought to feed mainly during the day (Allen and DeMartini 1983). Northern anchovy change from filter feeding to particulate feeding on the basis of prey size; they filter small prey and bite larger ones (Leong and O'Connell 1969). Northern anchovy, during all life stages, are an important part of the food chain for other species, including other fishes, sea birds, and marine mammals.

Summary of northern anchovy distribution and life history attributes.

Range: Queen Charlotte Islands, British Columbia, Canada to Magdalena Bay, Baja California, Mexico

Life History:

- **Maximum size**: about 230 mm (9 in.)
- **Age at maturity:** 1–2 yr
- **Life span**: 7 yr
- **Spawning**: Year round, peaks in February–April; may spawn every 7–10 days, releasing from 2,700 to16,000 eggs per batch

Habitat: Bays, estuaries, offshore pelagic

Fishery: Commercial for fish meal reduction, fish oil, and live/frozen fishing bait

3.3.4.3.2 Population Trends and Fishery

Northern anchovy are fished commercially for reduction (e.g., fish meal, oil, and paste) and live bait (Bergen and Jacobsen 2001). This species is the most important bait fish in southern California, and is also used in Oregon and Washington as bait for sturgeon (*Acipenser* spp.), salmonids (*Oncorhynchus* spp.), and other species (Emmett et al. 1991). Northern anchovy populations increased dramatically during the collapse of the Pacific sardine fishery, suggesting competition between these two species (Smith 1972).

Historically, estimates of the central subpopulation averaged about 325,700 metric tons (MT) (359,000 tons) from 1963 through 1972, increasing to over 1,542,200 MT (1.7 million tons) in 1974, then declining to 325,700 MT (359,000 tons) in 1978 (Bergen and Jacobsen 2001). Anchovy biomass in 1994 was estimated at 391,900 MT (432,000 tons). The stock is thought to be stable, and the size of the anchovy resource is largely dependent on natural influences such as ocean temperature. Landings in Los Angeles County since 2004 have fluctuated from a high of over 4 million pounds landed in 2005 and 2008, to a low of 212,444 pounds in 2010 (PacFIN 2012) (**Table 3-15**).

Northern anchovy are not federally or state listed as threatened or endangered.

Table 3-15. Annual landings and revenue for northern anchovy in Los Angeles County based on PacFIN data, 2004–2011.

Year	Northern Anchovy	
	Landed weight (lb)	Revenue ($)
2004	324,087	$35,699
2005	4,365,130	$185,579
2006	1,909,139	$75,104
2007	2,044,713	$81,953
2008	4,338,137	$196,660
2009	2,687,325	$140,143
2010	212,444	$10,280
2011	918,429	$42,015
Average	**2,099,926**	**$95,929**

3.3.4.3.3 Sampling Results

Northern anchovy (Engraulidae) was the eighth most abundant larval fish taxon collected using towed plankton nets at the intake station (SWE) with a mean concentration of 9.5 per 1,000 m^3 (**Table 3-4**), but was the second most abundant taxon at the source water stations with a mean concentration of 24.4 per 1,000 m^3 (**Table 3-6**). The larvae occurred during most months of the year but were most abundant during the months of September 2011 and January 2012 at the intake station (**Figure 3-21**) and September 2011 and March 2012 at the source water stations (**Figure 3-22**). Source water concentrations were substantially higher than the entrainment station (SWE) sample concentrations reflecting the mainly offshore distribution of the anchovies. The larvae were generally more abundant in nighttime samples than daytime samples except during the December 2011 survey (**Figure 3-23**). The length frequency distribution of measured northern anchovy larvae had a bimodal distribution with peaks in the range of 2.5–5.0 mm (0.10–0.20 in.) NL and 10.0–12.5 mm (0.39–0.50 in.) NL (**Figure 3-24**). Measured larval lengths from the intake station samples ranged from approximately 1.8–22.0 mm (0.07–0.87 in.) with a mean of 8.9 mm (0.35 in.).

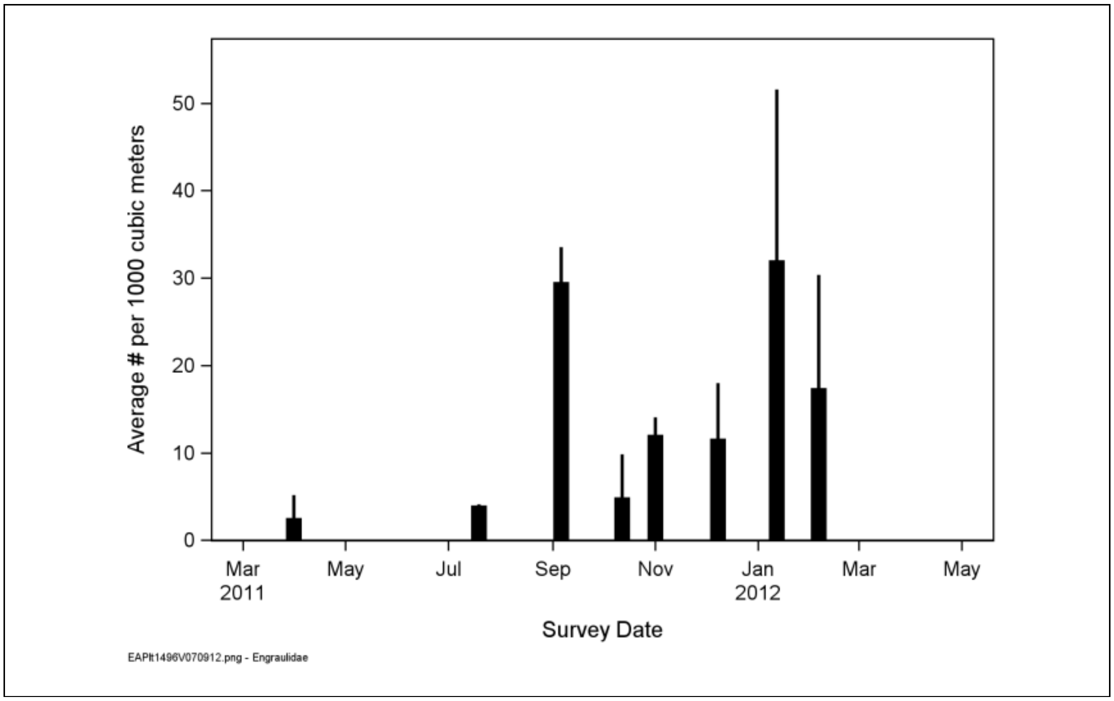


Figure 3-21. Mean concentration (#/1,000 m^3) and standard error for northern anchovy larvae collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

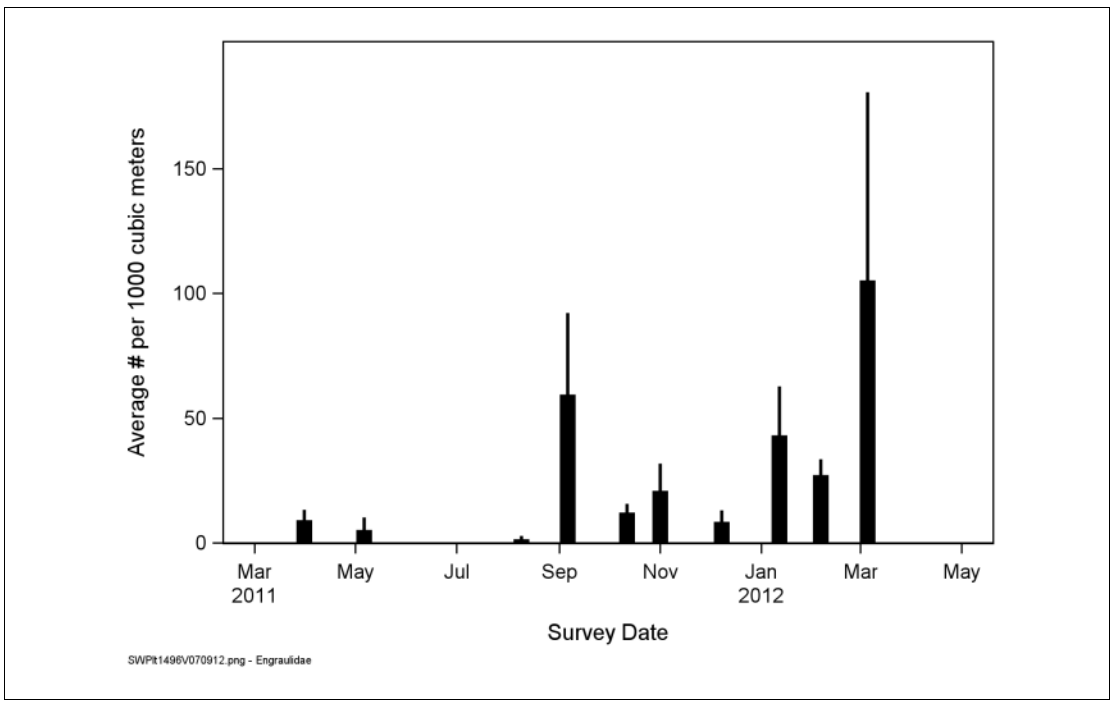


Figure 3-22. Mean concentration (#/1,000 m^3) and standard error for northern anchovy larvae collected using towed plankton nets at source water stations (SW1, SW2, SW3) from March 2011 through March 2012.

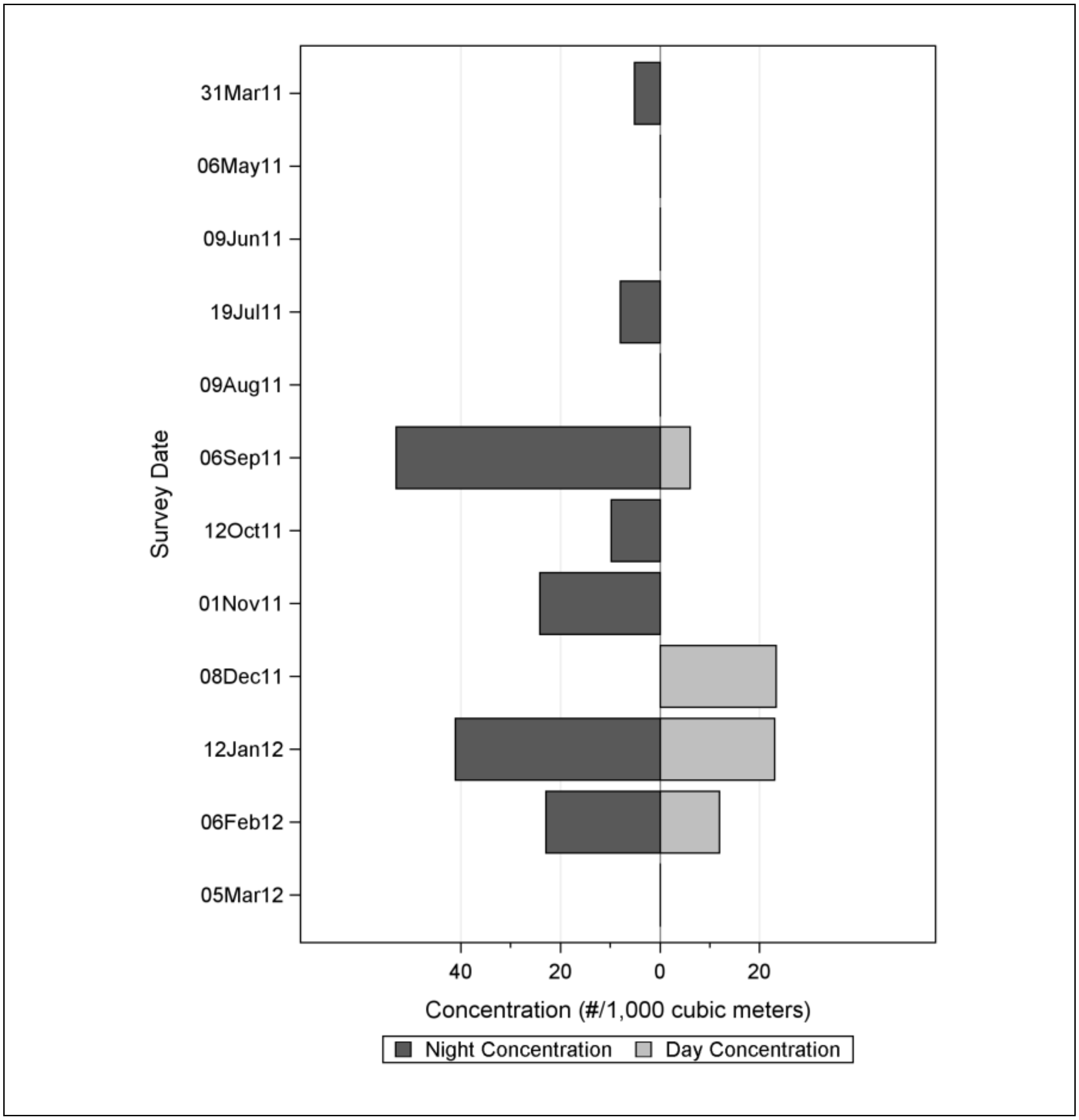


Figure 3-23. Mean concentration (#/1,000 m^3) in night and day samples for northern anchovy larvae collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

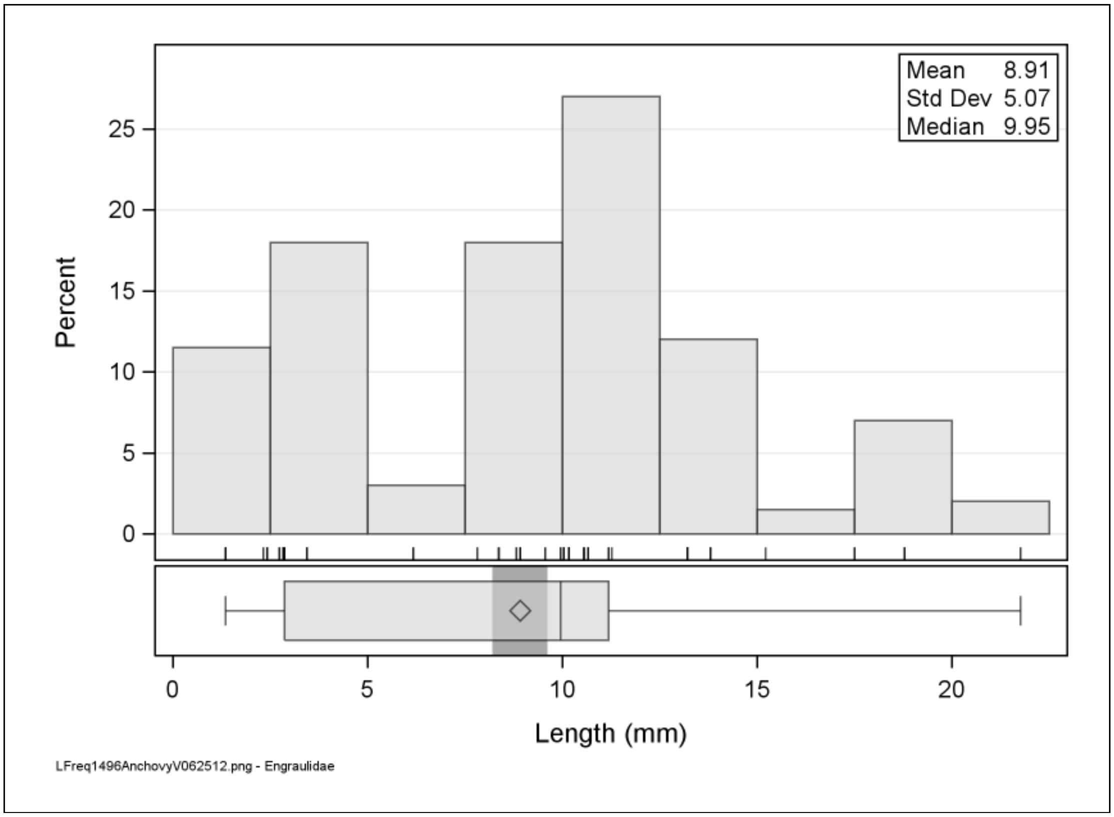


Figure 3-24. Length frequency for northern anchovy larvae collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

Refer to Figure 3.2-4 for symbol explanation.

3.3.4.3.4 Impact Modeling Results

Estimates of 4,548, 6,723, and 593,133 northern anchovy larvae and 69,448, 102,658, and 9,056,856 eggs were entrained during the March 2011 to March 2012 study period based, respectively, on daily intake flows through an unscreened intake of 1,309 m^3 (0.346 mgd) and 1,935 m^3 (0.511 mgd) for the demonstration facility, and a projected flow of 170,722 m^3 (45.1 mgd) for the final proposed facility (**Table 3-5**).

Empirical Transport Model (*ETM*)

The period of time that northern anchovy larvae were vulnerable to entrainment at the WBDDF was estimated from the average age at entrainment, which was based on measurements of 29 larvae. The calculated hatch length from the data resulted in an estimate of 5.58 mm (0.22 in.), which exceeded the 25th percentile estimate, so the 10th percentile estimate, 2.33 mm (0.09 in.) was used as the estimated hatch length. This estimate is only slightly smaller than the published hatch length of 2.5–3.0 mm (0.10–0.12 in.) for northern anchovy (Moser 1996). The difference in length between the bootstrap estimate of the 95th percentile of 17.98 mm (0.71 in.) and the estimated hatch length, divided by an estimated growth rate of 0.41 mm/d (0.02 in./d) from

Methot and Kramer (1979), after adding the estimated egg duration of 2.9 d, resulted in an estimated larval exposure to entrainment of 41.6 days.[5] Although the modeling inputs are based on the data from the larvae, the total estimated impacts to both eggs and larvae can be accounted for by adding the egg duration to the estimated period of exposure to entrainment.

The data used to calculate the ETM estimates (f_i values) for northern anchovy are shown in **Tables 3-16** and **3-17**. Data collected during the January 2012 survey period produced the largest estimate of the source water population although the estimated source water population in March 2012 was nearly as large (**Table 3-16**). The largest estimated entrainment also occurred during the January 2012 survey period, however, the estimated entrainment during September 2011 was nearly as high. The total estimated annual entrainment through an unscreened intake for the projected flow of 170,722 m^3 (45.1 mgd) for the proposed facility represents less than 0.006 percent of the extrapolated source water population potentially subject to entrainment.

The source water estimates in **Table 3-16** were used in calculating the weights (f_i) used in the ETM calculations, which showed that the largest percentages of larvae collected using towed plankton nets at the source water stations occurred during the September 2011 and March 2012 surveys. However, larvae were only collected at the intake station during the September survey (**Table 3-17**). As a result no *PE* estimate was calculated for the March survey when over 35 percent of the larvae were collected (**Table 3-17**). Northern anchovy larvae can occur throughout the Southern California Bight, but are usually associated with surface water down to a depth of 100 m (330 ft). Due to the broad distribution of this species ETM analyses were done using depths of both 100 m (328 ft) and 300 m (984 ft) as limits on the back-projections. The average alongshore displacement for the source water population was 68.85 km (42.78 mi) and 84.97 km (52.80 mi) for the back-projections limited to depths of 100 m (328 ft) and 300 m (984 ft), respectively.

The ETM estimates for northern anchovy attributable to entrainment through an unscreened intake ranged from 0.00002 (0.002 percent) to 0.00004 (0.004 percent) for the WBDDF intake volumes, to a high of 0.0032 (0.3 percent) for the intake volume at the proposed facility (**Table 3-18**). The higher estimate for the proposed facility is due to higher intake volume. All of these estimates are very low, reflecting the small intake volumes relative to the potential source water for this wide-ranging species.

[5] The entrainment exposure duration was calculated from size and growth values with greater decimal precision than those shown, and differs slightly from the duration calculated using these rounded values.

Table 3-16. Daily and survey period estimates for northern anchovy for entrainment, sampled source water, and extrapolated source water. The entrainment estimate was calculated based on the projected flow of 170,722 m^3 (45.1 mgd) for the proposed facility through an unscreened intake. Extrapolated source water population calculated based on a depth of 300 m (984 ft).

Survey Date	Survey Days	Daily Entrainment Estimate (45.1 mgd)	Survey Entrainment Estimate (45.1 mgd)	Daily Sampled SW Estimate	Sampled SW Survey Period Estimate	Extrapolated SW Survey Population
31 Mar 11	31	441	13,670	213,236	6,610,315	321,969,557
06 May 11	35	-	-	146,166	5,115,797	459,852,008
09 Jun 11	37	-	-	-	-	-
19 Jul 11	31	682	21,149	23,383	724,871	29,861,864
09 Aug 11	24	-	-	19,022	456,528	22,252,685
06 Sep 11	32	5,048	161,540	1,622,071	51,906,257	1,105,208,786
12 Oct 11	28	839	23,488	274,242	7,678,784	627,786,493
01 Nov 11	29	2,065	59,892	556,901	16,150,142	1,345,482,845
08 Dec 11	36	1,994	71,774	229,092	8,247,329	359,129,899
12 Jan 12	30	5,474	164,222	1,059,899	31,796,978	2,780,704,639
06 Feb 12	26	2,977	77,397	670,554	17,434,392	800,823,705
05 Mar 12	27	-	-	2,964,550	80,042,843	2,259,359,205
	Annual Totals		**593,132**		**226,164,237**	**10,112,431,685**

Table 3-17. Data used in ETM calculations for northern anchovy. The estimates of PE_i for each survey period i, represent the unadjusted source water estimates and were calculated using only towed plankton net data from the intake and source water stations. The *PE*s were calculated based on three unscreened intake volumes: a) actual flow during the year for the WBDDF of 240 gpm or 1,309 m^3 (0.346 mgd), b) design flow for the WBDDF of 355 gpm or 1,935 m^3 (0.511 mgd), and c) a projected flow for the proposed facility of 170,722 m^3 (45.1 mgd).

Survey Date	a) Actual Average Demonstration Plant Flow = 240 gpm		b) Demonstration Plant Design Flow = 355 gpm		c) Final Desalination Plant Intake Flow = 45.1 mgd		Weight (f_i)	Source Water Extrapolation Alongshore (km)	
	PE_i	PE_i Std. Error	PE_i	PE_i Std. Error	PE_i	PE_i Std. Error		Depth (m) = 100	Depth (m) = 300
31-Mar-11	0.000016	0.000017	0.000023	0.000025	0.002068	0.002222	0.029	73.06	73.06
6-May-11	0	0	0	0	0	0	0.023	134.82	134.83
9-Jun-11	0	0	0	0	0	0	0.000	30.54	40.66
19-Jul-11	0.000224	0.000009	0.000331	0.000014	0.029176	0.001237	0.003	56.27	61.79
9-Aug-11	0	0	0	0	0	0	0.002	72.75	73.12
6-Sep-11	0.000024	0.000010	0.000035	0.000015	0.003112	0.001350	0.230	31.90	31.94
12-Oct-11	0.000023	0.000025	0.000035	0.000036	0.003059	0.003204	0.034	109.90	122.63
1-Nov-11	0.000028	0.000013	0.000042	0.000019	0.003708	0.001707	0.071	26.18	124.97
8-Dec-11	0.000067	0.000047	0.000099	0.000069	0.008703	0.006131	0.037	64.04	65.32
12-Jan-12	0.000040	0.000030	0.000059	0.000045	0.005165	0.003971	0.141	122.21	131.18
6-Feb-12	0.000034	0.000026	0.000050	0.000039	0.004439	0.003421	0.077	67.21	68.90
5-Mar-12	0	0	0	0	0	0	0.354	39.39	42.34
Average (n=12)	**0.000038**		**0.000056**		**0.004953**			**69.02**	**80.90**
Average (*PE*>0)	**0.000057**		**0.000084**		**0.007429**			**68.85**	**84.97**

Table 3-18. ETM estimates of P_M for northern anchovy. Estimates of P_M based on estimated upcoast and downcoast source water populations based on back-projections limited to depths of 100 m (328 ft) and 300 m (984 ft). ETM estimates calculated for three unscreened intake volumes: a) actual flow during the year for the WBDDF of 1,309 m^3 (0.346 mgd), b) design flow for the WBDDF of 1,935 m^3 (0.511 mgd), and c) a projected flow for the proposed facility of 170,722 m^3 (45.1 mgd). Two estimates of standard error (Std. Error) are presented: one based on combining the variance components from the estimates comprising ETM, and the second using the weighted average coefficient of variation across the *PE*s for the twelve surveys.

Flow	CODAR Depth Limit	P_M Estimate	P_M Std. Error	P_M + Std. Error	P_M - Std. Error	CV Based Std. Error	P_M + CV Std. Error	P_M - CV Std. Error
a) WBDDF Actual Flow	100 m Depth	0.000025	0.104851	0.104876	-0.104826	0.000010	0.000035	0.000015
	300 m Depth	0.000021	0.104852	0.104872	-0.104831	0.000008	0.000029	0.000012
b) WBDDF Design Flow	100 m Depth	0.000037	0.104849	0.104886	-0.104813	0.000014	0.000051	0.000022
	300 m Depth	0.000030	0.104850	0.104880	-0.104820	0.000012	0.000042	0.000018
c) Proposed Design Flow	100 m Depth	0.003226	0.104324	0.107550	-0.101097	0.001267	0.004493	0.001959
	300 m Depth	0.002674	0.104390	0.107064	-0.101716	0.001050	0.003725	0.001624

3.3.4.4 Kelpfishes (*Gibbonsia* spp.)

Kelpfishes or kelp blennies (family Clinidae) of the genus *Gibbonsia* are represented in southern California waters by three species: *G. montereyensis, G. metzi*, and *G. elegans*. The first two species, *G. montereyensis* (crevice kelpfish) and *G. metzi* (striped kelpfish) range from British Columbia to central Baja California while *G. elegans* (spotted kelpfish) ranges from Piedras Blancas Point in central California to southern Baja California, Mexico (Love 2011). All three species are similar in appearance and are differentiated mainly by fin ray counts and the presence or absence of scales on the caudal fin (Miller and Lea 1972).

3.3.4.4.1 Life History and Ecology

Kelpfishes are small, cryptic fishes generally found living in nearshore rocky reefs among kelp and seaweeds from the intertidal zone to depths of 56 m (185 ft) but is not common below about 15 m (50 ft) (Lamb and Edgell 1986; Fitch and Lavenberg 1975). Kelpfishes are known to spawn year-round (Williams 1954) though exhibit a peak in their spawning between February and April (Watson 1996). Each species of *Gibbonsia* is oviparous, spawning demersal eggs which are adhesive and are attached to algal nests (Fitch and Lavenberg 1975; Moser 1996). *Gibbonsia elegans* is reported to have a fecundity of 2,300 eggs/female (Bane and Bane 1971). Kelpfishes first spawn at two years of age and may spawn several times per year (Fitch and Lavenberg 1975). Larval growth was estimated by Stepien (1986) for the closely-related giant kelpfish, *Heterostichus rostratus*, at 0.25 mm/day ± 0.013 (0.98 in.). The larval yolk-sac stage ranges in size from 4.6-4.8 mm (0.18-0.19 in.), preflexion from 4.6-6.4 mm (0.18-0.25 in.), flexion from 6.6-8.0 mm (0.26-0.31 in.), and postflexion from 8.4-20.0 mm (0.33-0.79 in.) (Watson 1996). Kelpfishes may live to about seven years (Fitch and Lavenberg 1975) and attain a length of about 160 mm (6 in.). There are no catch data for these species because they are not caught commercially and only captured occasionally for aquarium display.

Summary of kelpfish distribution and life history attributes.

Range: British Columbia to central Baja California

Life History:

- **Maximum size**: about 160 mm (6 in.)
- **Age at maturity:** 2 yr
- **Life span**: 7 yr
- **Spawning**: Demersal eggs; may spawn year round, peaking in spring; mean fecundity of 2,250 eggs

Habitat: Rocky reefs, shallow subtidal and intertidal among kelp and seaweeds

Fishery: None

3.3.4.4.2 Population Trends and Fishery

Concentrations of larval kelpfishes (*Gibbonsia* spp.) in King Harbor have increased substantially since 1990 (**Figure 3-25**) (Vantuna Research Group, unpublished data). This taxon was collected sporadically and in low concentrations during the first 17 years of the study, but densities increased in the following years. Highest concentrations were recorded in 1994, and from 2001 through 2006. Kelpfishes are not federally or state listed as threatened or endangered. There is no fishery for kelpfishes.

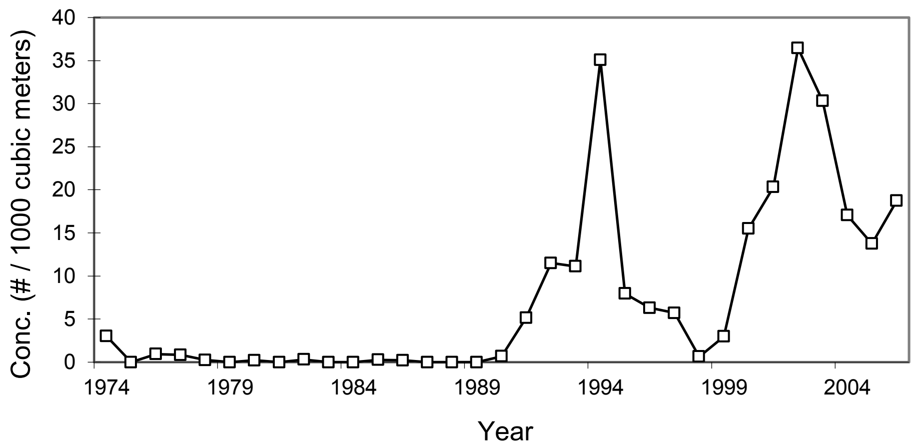


Figure 3-25. Mean concentration (#/1,000 m^3) of kelpfish larvae collected from King Harbor, 1974–2006. Source: Vantuna Research Group, unpublished data.

3.3.4.4.3 Sampling Results

Kelpfish (*Gibbonsia* spp.) was the ninth most abundant larval fish taxon collected using towed plankton nets at the intake station (SWE) with a mean concentration of 5.3 per 1,000 m^3 (**Table 3-4**), and the tenth most abundant taxon at the source water stations with a mean concentration of 4.9 per 1,000 m^3 (**Table 3-6**). The larvae occurred during most months of the year with peaks during the months of March 2011 and October 2011 at the intake station (**Figure 3-26**), and July 2011 and January 2012 at the source water stations (**Figure 3-27**). Source water concentrations overall were similar to the entrainment sample concentrations. The larvae were variable in abundance between the nighttime and daytime samples with no clear trends (**Figure 3-28**). The length frequency distribution of measured kelpfish larvae was strongly skewed to the smallest size class with most specimens in the range of 4.0–4.5 mm (0.16–0.18 in.) NL (**Figure 3-29**). Measured larval lengths from the intake station samples ranged from approximately 4.2 to 8.7 mm (0.16–0.87 in.) with a mean of 4.7 mm (0.19 in.).

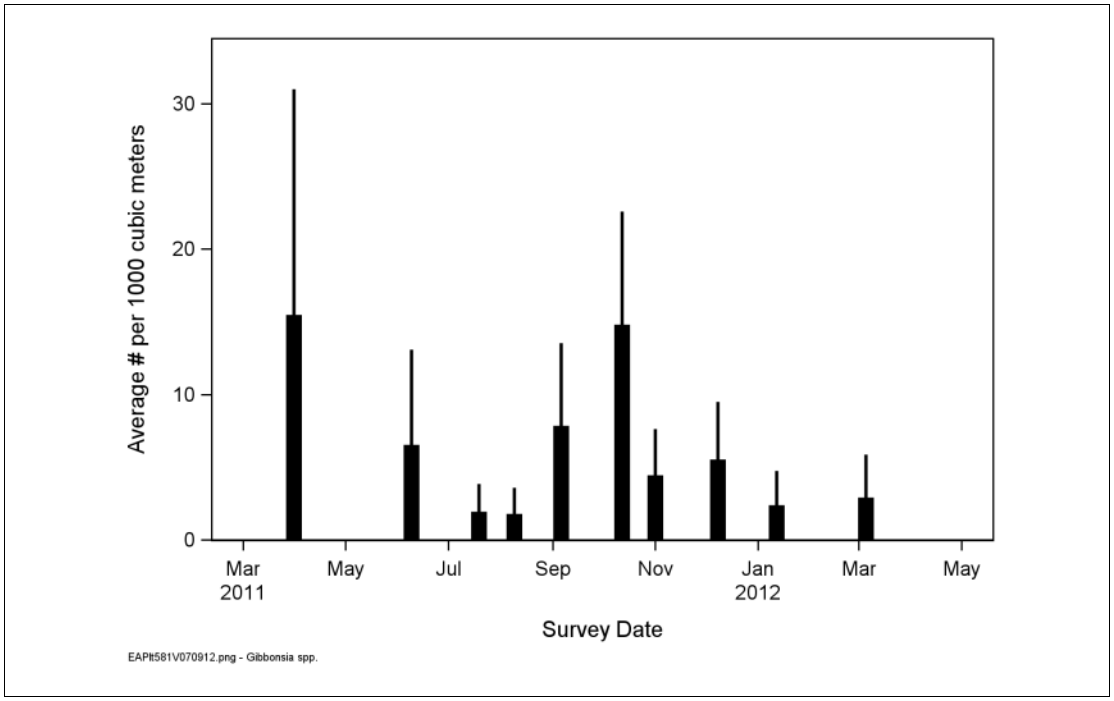


Figure 3-26. Mean concentration (#/1,000 m^3) and standard error for kelpfish larvae collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

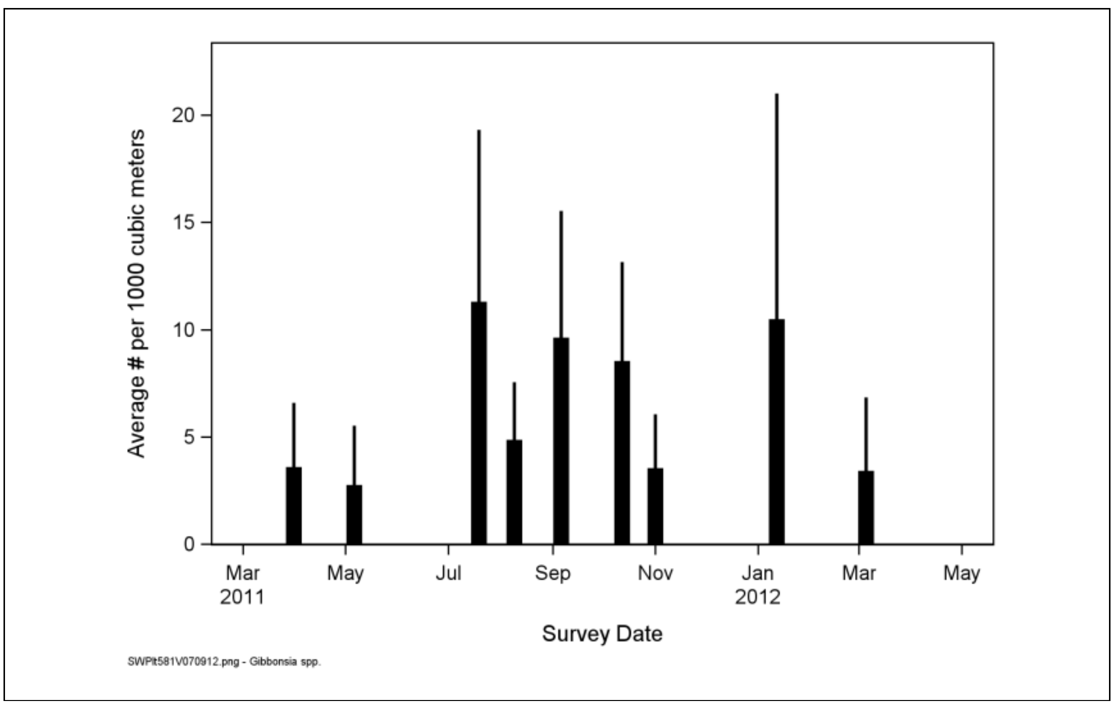


Figure 3-27. Mean concentration (#/1,000 m^3) and standard error for kelpfish larvae collected using towed plankton nets at source water stations (SW1, SW2, SW3) from March 2011 through March 2012.

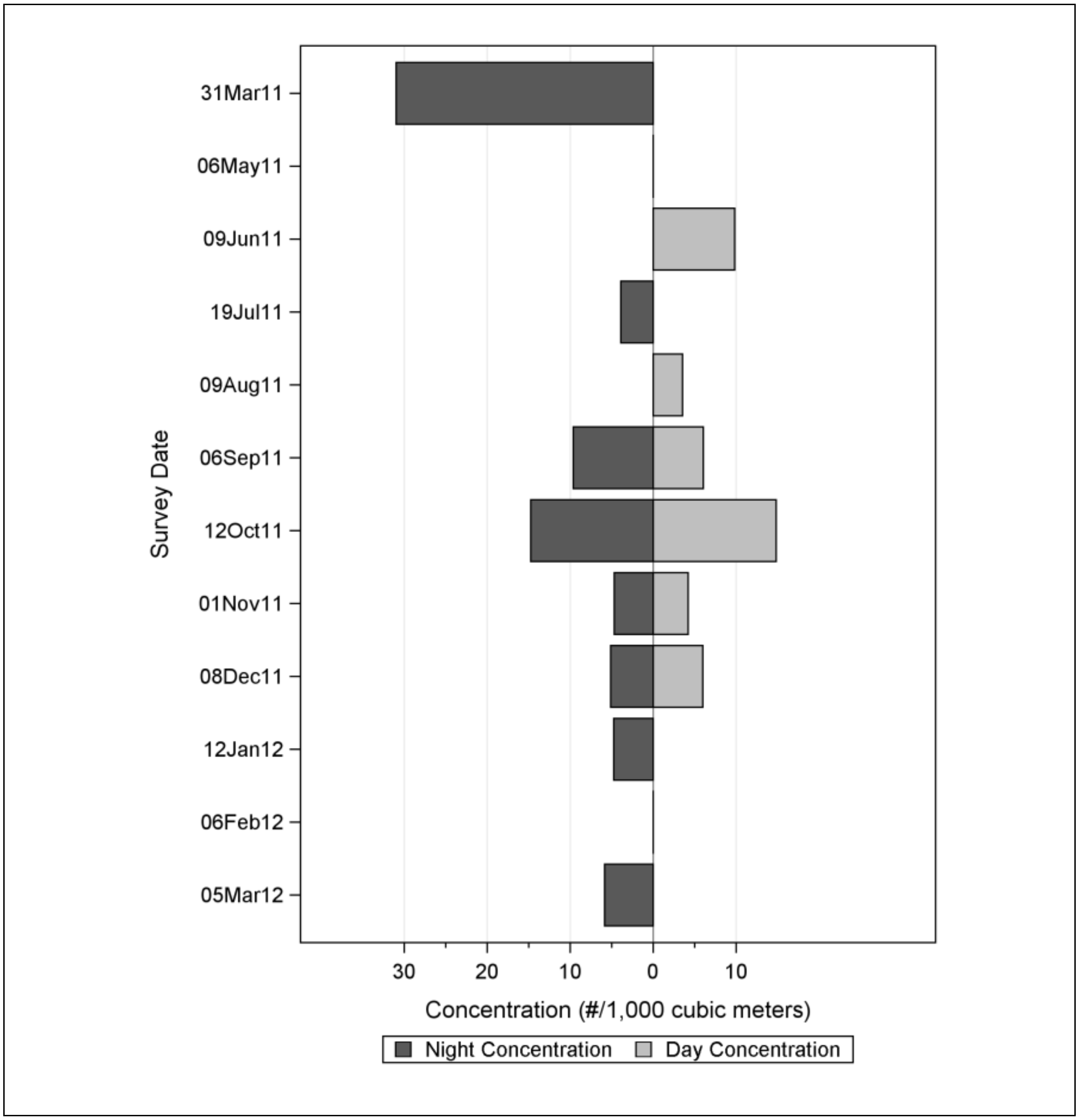


Figure 3-28. Mean concentration (#/1,000 m^3) in night and day samples for kelpfish larvae collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

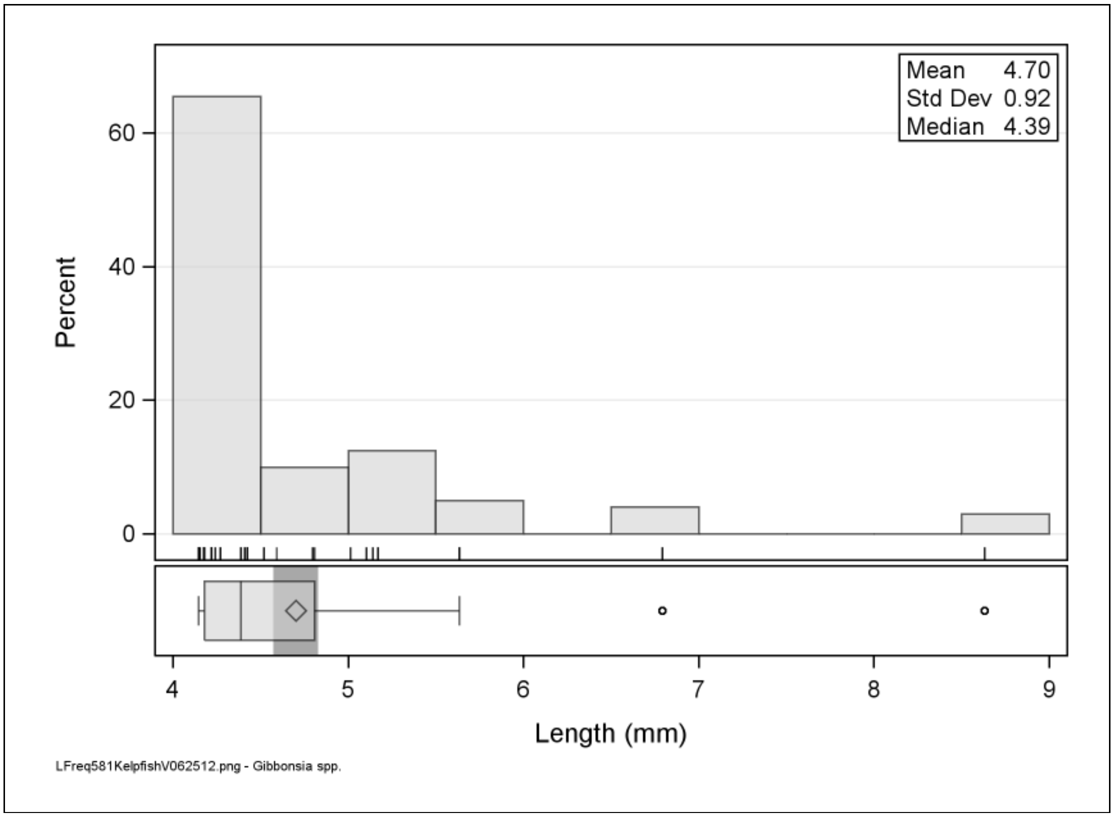


Figure 3-29. Length frequency for kelpfish larvae collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

Refer to Figure 3.2-4 for symbol explanation.

3.3.4.4.4 Impact Modeling Results

Estimates of 2,580, 3,814, and 336,495 kelpfish larvae were entrained during the March 2011 to March 2012 study period based respectively on daily unscreened intake flows of 1,309 m^3 (0.346 mgd) and 1,935 m^3 (0.511 mgd) for the demonstration facility, and a projected flow of 170,722 m^3 (45.1 mgd) for the final proposed facility (**Table 3-5**).

Empirical Transport Model (*ETM*)

The period of time that kelpfish larvae were vulnerable to entrainment at the WBDDF was estimated from the average age at entrainment, which was based on measurements of 21 larvae. The calculated hatch length from the data resulted in an estimate of 4.27 mm (0.17 in.), which exceeded the 25th percentile estimate, so the 10th percentile estimate, 4.15 mm (0.16 in.) was used as the estimated hatch length. This estimate is only slightly smaller than the published hatch length of 4.5 mm (0.18 in.) for *Gibbonsia elegans* (Moser 1996). The difference in length between the bootstrap estimate of the 95th percentile of 7.07 mm (0.28 in.) and the estimated hatch length, divided by an estimated growth rate of 0.30 mm/d (0.01 in./d) was used to calculate an estimated larval exposure to entrainment of 9.8 days.[6] There were no published larval growth rates for kelpfish so the growth rate used in the calculations was based on data from Moser (1996) and a total estimated time of 60 days to the transformation length of 22 mm (0.09 in.). Kelpfishes do not have a planktonic egg stage so no adjustment was made to the duration.

The data used to calculate the ETM estimates (f_i values) for kelpfishes are shown in **Tables 3-19** and **3-20**. The only survey when kelpfish larvae were not collected in both the entrainment and source water samples was the February 2012 survey. Data collected during the September survey period produced the largest estimate of the source water population (**Table 3-19**), and over 32 percent of the total annual source water population was estimated to occur during the September and October 2011 surveys (**Table 3-20**). The largest estimated entrainment occurred during the October survey period (**Table 3-19**). The total estimated annual entrainment for the projected flow through an unscreened intake of 170,722 m^3 (45.1 mgd) for the proposed facility represented 0.15 percent of the extrapolated source water population potentially subject to entrainment.

The source water estimates in **Table 3-19** were used in calculating the weights (f_i) used in the ETM calculations, which showed that approximately 32 percent of the larvae collected using towed plankton nets at the source water stations were collected during the September and October surveys (**Table 3-20**).

Kelpfishes are typically associated with shallow nearshore areas and are not common below about 15 m (50 ft) (Fitch and Lavenberg 1975) although some species may occur to depths of 56 m (185 ft) (Love 2011). Based on the nearshore distribution of kelpfishes, the results of the ETM analysis based on the back-projections that are limited to depths of 30 m (98 ft) are likely to be more representative of actual impacts on the population, but the second set of results using a depth limit of 100 m (328 ft) results in a larger range of coastal area where the larvae could have been hatched. The average alongshore displacement for the source water population was 10.98 km (6.82 mi) and 25.25 km (15.69 mi) for the back-projections limited to depths of 30 m (98 ft) and 100 m (328 ft), respectively.

The ETM estimates attributable to entrainment through an unscreened intake for kelpfish ranged from 0.00006 (0.006 percent) to 0.00026 (0.026 percent) for the WBDDF intake volumes, to a high of 0.0223 (2.2 percent) for the intake volume at the proposed facility (**Table 3-21**). The higher estimate for the proposed facility is due to higher intake volume.

[6] The entrainment exposure duration was calculated from size and growth values with greater decimal precision than those shown, and differs slightly from the duration calculated using these rounded values.

Table 3-19. Daily and survey period estimates for kelpfishes for entrainment, sampled source water, and extrapolated source water. The entrainment estimate was calculated based on the projected flow for the proposed facility of 170,722 m^3 (45.1 mgd) through an unscreened intake. Extrapolated source water population calculated based on a depth of 30 m (98 ft).

Survey Date	Survey Days	Daily Entrainment Estimate (45.1 mgd)	Survey Entrainment Estimate (45.1 mgd)	Daily Sampled SW Estimate	Sampled SW Survey Period Estimate	Extrapolated SW Survey Population
31 Mar 11	31	2,646	82,023	135,712	4,207,061	19,979,393
06 May 11	35	-	-	34,542	1,208,969	5,329,847
09 Jun 11	37	1,119	41,389	38,340	1,418,597	4,229,693
19 Jul 11	31	330	10,236	153,007	4,743,207	29,788,399
09 Aug 11	24	306	7,347	71,923	1,726,155	41,755,071
06 Sep 11	32	1,338	42,824	168,935	5,405,908	54,265,293
12 Oct 11	28	2,527	70,746	194,159	5,436,451	29,280,179
01 Nov 11	29	762	22,105	71,056	2,060,633	23,644,673
08 Dec 11	36	947	34,105	32,471	1,168,946	6,265,793
12 Jan 12	30	407	12,197	145,473	4,364,179	5,666,293
06 Feb 12	26	-	-	-	-	-
05 Mar 12	27	501	13,522	59,952	1,618,694	2,370,879
	Annual Totals		**336,495**		**33,358,800**	**222,575,514**

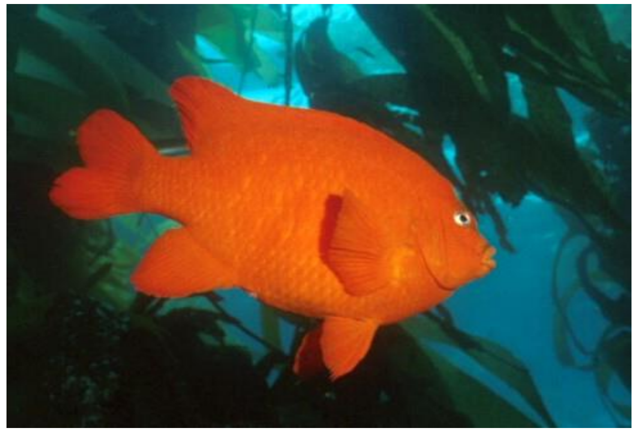
Table 3-20. Data used in ETM calculations for kelpfishes. The estimates of PE_i for each survey period i, represent the unadjusted source water estimates and were calculated using only towed plankton net data from the intake and source water stations. The *PE*s were calculated based on three unscreened intake volumes: a) actual flow during the year for the WBDDF of 240 gpm or 1,309 m^3 (0.346 mgd), b) design flow for the WBDDF of 355 gpm or 1,935 m^3 (0.511 mgd), and c) a projected flow for the proposed facility of 170,722 m^3 (45.1 mgd).

Survey Date	a) Actual Average Demonstration Plant Flow = 240 gpm		b) Demonstration Plant Design Flow = 355 gpm		c) Final Desalination Plant Intake Flow = 45.1 mgd		Weight (f_i)	Source Water Extrapolation Alongshore (km)	
	PE_i	PE_i Std. Error	PE_i	PE_i Std. Error	PE_i	PE_i Std. Error		Depth (m) = 30	Depth (m) = 100
31-Mar-11	0.000149	0.000175	0.000221	0.000259	0.019496	0.022888	0.126	7.12	14.95
6-May-11	0	0	0	0	0	0	0.036	6.61	50.06
9-Jun-11	0.000224	0.000296	0.000331	0.000438	0.029176	0.038629	0.043	4.47	12.59
19-Jul-11	0.000017	0.000017	0.000024	0.000025	0.002158	0.002163	0.142	9.42	15.54
9-Aug-11	0.000033	0.000038	0.000048	0.000057	0.004256	0.005003	0.052	36.29	45.10
6-Sep-11	0.000061	0.000048	0.000090	0.000072	0.007922	0.006325	0.162	15.06	26.72
12-Oct-11	0.000100	0.000057	0.000147	0.000085	0.013013	0.007461	0.163	8.08	29.21
1-Nov-11	0.000082	0.000071	0.000122	0.000104	0.010727	0.009196	0.062	17.21	19.03
8-Dec-11	0.000224	0.000210	0.000331	0.000311	0.029176	0.027406	0.035	8.04	27.31
12-Jan-12	0.000021	0.000029	0.000032	0.000042	0.002795	0.003745	0.131	1.95	22.62
6-Feb-12	0	0	0	0	0	0	0.000	2.08	16.79
5-Mar-12	0.000064	0.000080	0.000095	0.000118	0.008354	0.010412	0.049	2.20	39.39
Average (n=12)	**0.000081**		**0.000120**		**0.010589**			**9.88**	**26.61**
Average (*PE*>0)	**0.000098**		**0.000144**		**0.012707**			**10.98**	**25.25**

Table 3-21. ETM estimates of P_M for kelpfishes. Estimates of P_M based on estimated upcoast and downcoast source water populations based on back-projections limited to depths of 100 m (328 ft) and 300 m (984 ft). ETM estimates calculated for three unscreened intake volumes: a) actual flow during the year for the WBDDF of 1,309 m^3 (0.346 mgd), b) design flow for the WBDDF of 1,935 m^3 (0.511 mgd), and c) a projected flow for the proposed facility of 170,722 m^3 (45.1 mgd). Two estimates of standard error (Std. Error) are presented: one based on combining the variance components from the estimates comprising ETM, and the second using the weighted average coefficient of variation across the *PE*s for the twelve surveys.

Flow	CODAR Depth Limit	ETM Estimate	ETM Std. Error	ETM + Std. Error	ETM - Std. Error	CV Based Std. Error	ETM + CV Std. Error	ETM - CV Std. Error
a) WBDDF Actual Flow	30 m Depth	0.000175	0.160224	0.160399	-0.160050	0.000166	0.000341	0.000008
	100 m Depth	0.000057	0.160249	0.160306	-0.160192	0.000055	0.000112	0.000003
b) WBDDF Design Flow	30 m Depth	0.000258	0.160208	0.160466	-0.159950	0.000246	0.000504	0.000012
	100 m Depth	0.000085	0.160244	0.160329	-0.160160	0.000081	0.000165	0.000004
c) Proposed Design Flow	30 m Depth	0.022335	0.156179	0.178514	-0.133844	0.021268	0.043603	0.001067
	100 m Depth	0.007414	0.159040	0.166454	-0.151626	0.007060	0.014474	0.000354

3.3.4.5 Garibaldi (Hypsypops rubicundus)



Garibaldi (*Hypsypops rubicundus*) ranges from Monterey Bay, California to southern Baja California and Guadalupe Island (off northern central Baja California) in Mexico, but is not abundant north of Santa Barbara (Fitch and Lavenberg 1975). They are one of two common species of damselfishes (Family Pomacentridae) found off southern California, the other being the blacksmith. Garibaldi is the California state marine fish and is fully protected by law.

3.3.4.5.1 Life History and Ecology

Garibaldi occurs over rocky bottoms in clear water, often near crevices and small caves, from the intertidal zone (as juveniles) to depths of 29 m (95 ft). They occur on the outer coast, around islands, and in protected bays and harbors (Fitch and Lavenberg 1975), typically as individuals (adults defend a territory all year) but occasionally in loose aggregations. They attain a maximum length up to 38.1 cm (15 in.) TL, although few are larger than 30.5 cm (12 in.). Males are larger than females at a given age (Limbaugh 1964). Males begin to mature at about 3 years but females may not reproduce until age 5 to 6 years. Garibaldi may attain a maximum age of 18 years.

Garibaldi spawn from March through October, and the female deposits demersal adhesive eggs in a nest that the male has prepared by clearing off all growth except calcareous tubes and filamentous red algae (Love 2011). Males defend algal nests within permanent territories (10–15 m^2 [33–49 ft^2]) on which females deposit eggs (Clarke 1970). Males that guard nesting areas with sparse algal cover tend to be less likely to court passing females (Sikkel 1995). DeMartini et al. (1994) measured mean batch fecundity at 12,546 eggs with an average of 35 eggs per gram of body weight. Some nests may contain up to 190,000 eggs deposited by several females (Fitch and Lavenberg 1975). In southern California female garibaldi were estimated to spawn about 24 times during their 144-day spawning season (DeMartini et al. 1994). Females preferentially approach nests with eggs in the early stages of development prior to or in the absence of male courtship and are more likely to spawn in such nests than in empty nests or nests with only eggs in the advanced stages of development (Sikkel 1989). Eggs in the early stages of development are bright yellow and turn gray as development proceeds. Eggs hatch in 12−23 days (Sikkel 1989) depending on temperature. Larvae are primarily neustonic, initially approximately 2.2 mm (0.09 in.) in length and attain flexion at approximately 3.5 mm (0.14 in.) (Moser 1996). Transformation occurs at a length of approximately 5−10 mm (0.20–0.39 in.) and settlement has been noted to occur at approximately 20 mm (0.79 in.) SL. Larval duration ranges from 18−22 days (mean of 20 days) based on daily incremental marks on otoliths in recently settled individuals (Wellington and Victor 1989). High sedimentation and increased turbidity are factors that affect recruitment success of juveniles (Pondella and Stephens 1994), while increasing water temperatures during El Niño years have a negative effect on larval populations because adults may abandon their nests and retreat to deeper, cooler waters (Stephens et al. 1994).

Juveniles garibaldi feed on planktonic crustaceans such as copepods, amphipods, and isopods (Clarke 1970). As adults they are typically carnivorous, feeding on a variety of invertebrates including sponges, sea anemones, bryozoans, worms, crustaceans, clams and mussels, snail eggs, and their own eggs. Field observations and experiments during the mating phase show that brood-guarding males usually cannibalize older clutches if the older eggs are exposed to empty nest space (Sikkel 1994a). Males nearly always cannibalize the entire brood when they receive only a single clutch, and the probability of cannibalism of last clutches increases with brood age (Sikkel 1994b). Garibaldi are only active during the day and shelter in holes in the reef at night (Clarke 1970). Juvenile garibaldi are preyed upon by larger fishes such as kelp bass, and adult garibaldi are preyed upon by sharks, giant sea bass, moray eels, and sea lions.

Summary of garibaldi distribution and life history attributes.

Range: Monterey Bay, California to southern Baja California, Mexico

Life History:

- **Maximum size**: 38.1 cm (15 in.) TL
- **Age at maturity:** Males 3 yr, females 5–6 yr
- **Life span**: 18 yr
- **Spawning**: Demersal eggs; spawn mainly late spring through summer; multiple spawning with a mean batch fecundity of approximately 12,500 eggs

Habitat: Rocky reefs in shallow subtidal zone, kelp forests and breakwaters

Fishery: None (protected species)

3.3.4.5.2 Population Trends and Fishery

Long-term trends in population abundance of garibaldi have been extensively studied at a few sites within southern California including King Harbor, Palos Verdes, and along the Channel Islands (Stephens et al. 1984, 1994; Pondella and Stephens 1994; Davis et al. 1997). There were notable increases in fish density from diver transect surveys during the late 1970s to early 1980s with a slight decline by the mid 1980s at Palos Verdes and King Harbor. Since then the population has remained fairly stable and constant at both sites. Garibaldi showed some year-to-year variation at Channel Island sites since 1985, but there were no apparent long-term trends in abundance (Davis et al. 1997; Tenera Environmental 2006). Concentrations of garibaldi larvae, as measured in King Harbor as part of the Occidental College – Vantuna Research Group's long-term studies, have fluctuated substantially since 1974 (**Figure 3-30**). This species was absent during the first three years of the study, but appeared in increasing numbers in subsequent years. Highest concentrations were recorded in 1990, 1997, and 2004.

Although some individuals may be caught incidentally, there is no legal fishery for garibaldi because it is a protected species. This species is considered widespread in the Eastern Pacific, and it is common in many parts of its range. There are no major threats for this species and no current indication of population decline.

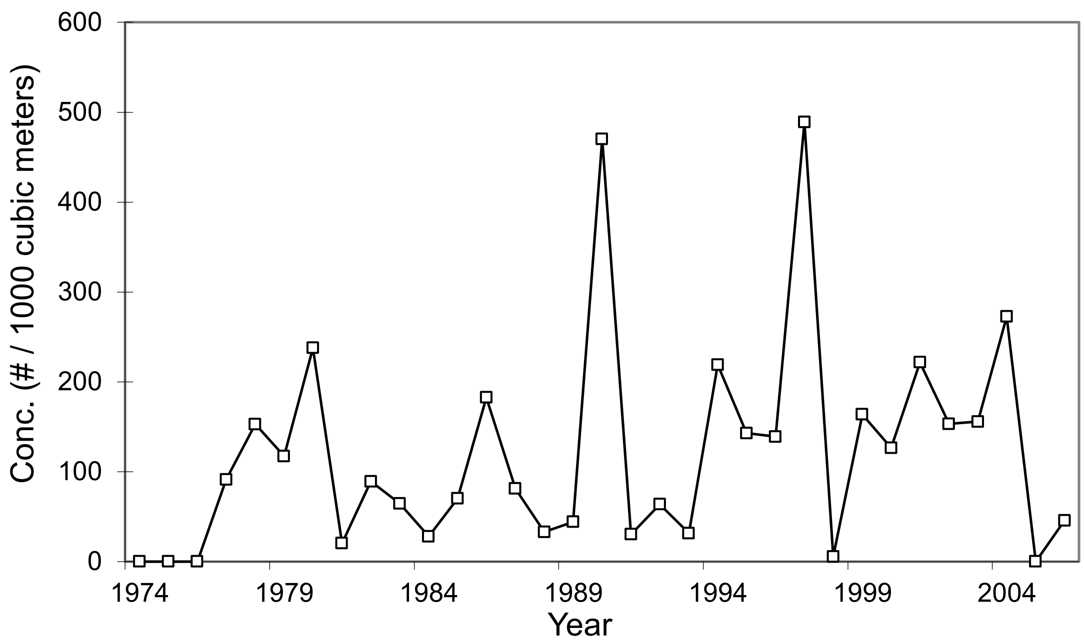


Figure 3-30. Mean concentration (# / 1,000 m^3) of garibaldi larvae collected from King Harbor, 1974-2006. Source: Vantuna Research Group.

3.3.4.5.3 Sampling Results

Garibaldi was the seventh most abundant larval fish taxon collected using towed plankton nets at the intake station (SWE) with a mean concentration of 10.1 per 1,000 m^3 (**Table 3-4**), and the ninth most abundant taxon at the source water stations with a mean concentration of 5.1 per 1,000 m^3 (**Table 3-6**). Garibaldi are exclusively summer spawners, and this was reflected in the narrow temporal distribution of larvae to only two (source water) or three (entrainment) months from June through September (**Figures 3-31** and **3-32**). Source water concentrations were lower than the entrainment station sample concentrations, except in August when no garibaldi were collected at the intake station. Larvae were predominantly collected in the nighttime samples (**Figure 3-33**). The length frequency distribution of measured garibaldi larvae were within a very narrow size range that was actually smaller than the hatch length of 2.9–3.5 mm (0.11–0.14 in.) reported by Moser (2006) (**Figure 3-34**). The mean length was 2.5 mm (0.10 in.).

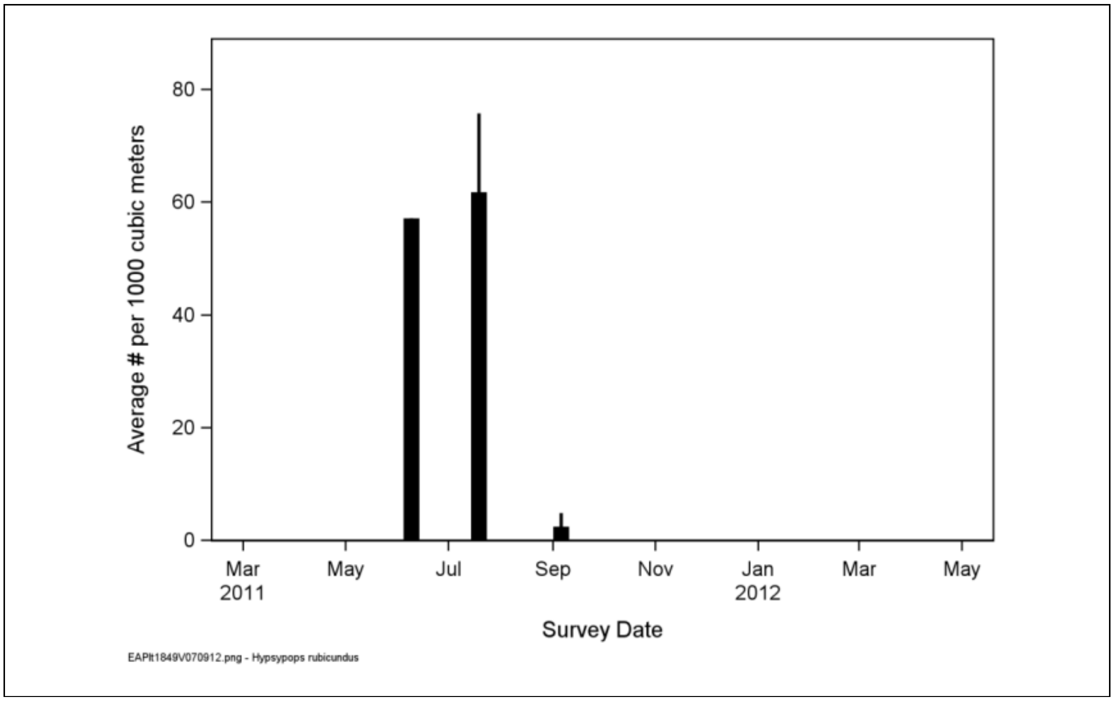


Figure 3-31. Mean concentration (#/1,000 m^3) and standard error for garibaldi larvae collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

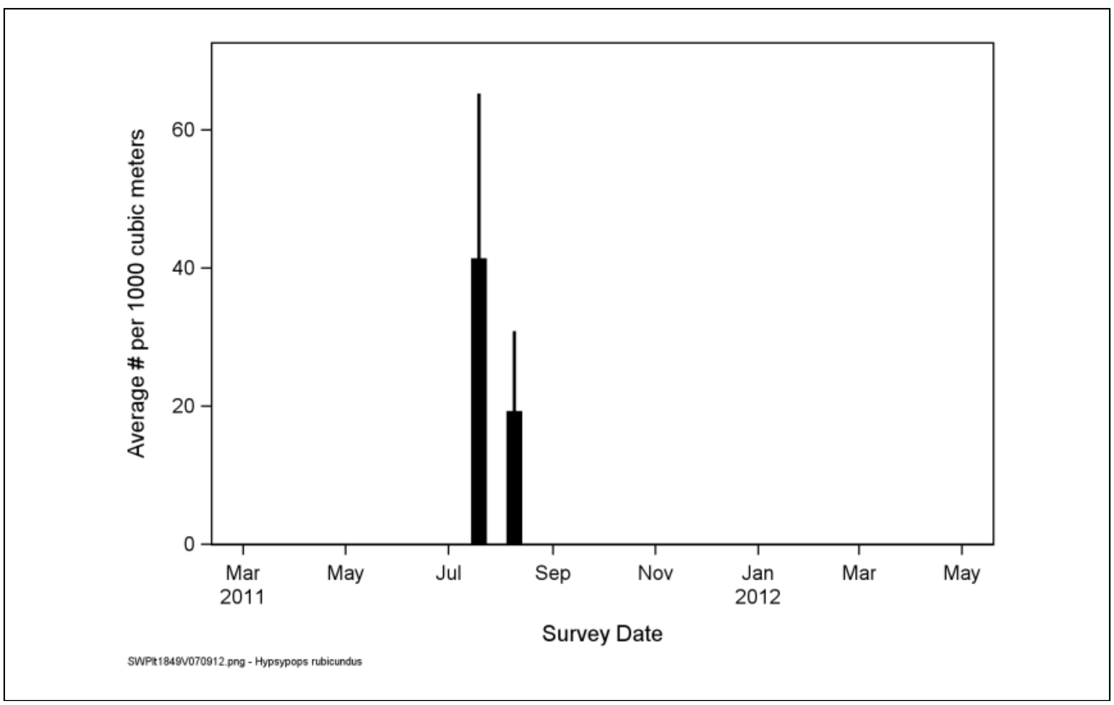


Figure 3-32. Mean concentration (#/1,000 m^3) and standard error for garibaldi larvae collected using towed plankton nets at source water stations (SW1, SW2, SW3) from March 2011 through March 2012.

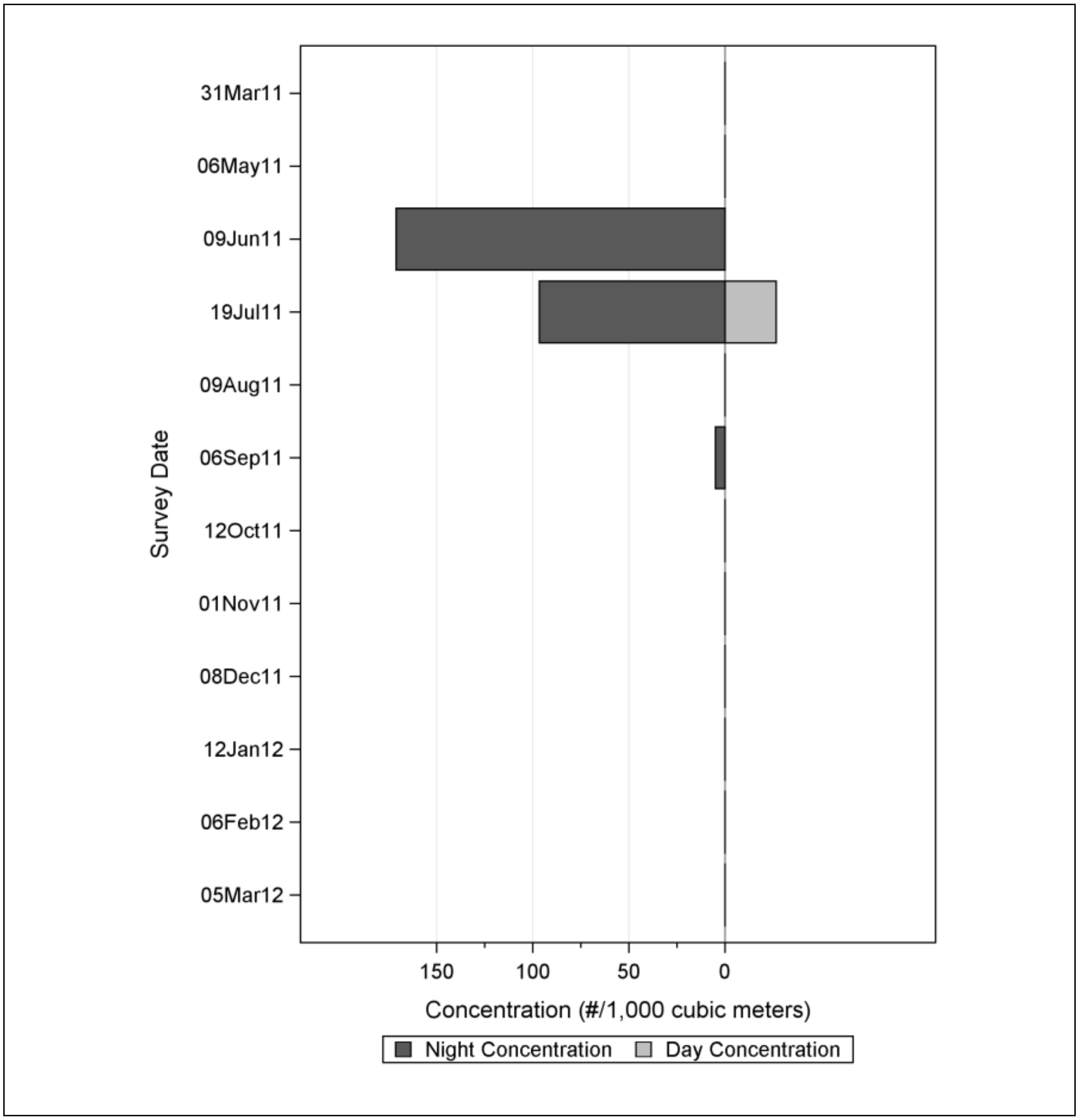


Figure 3-33. Mean concentration (#/1,000 m^3) in night and day samples for garibaldi larvae collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

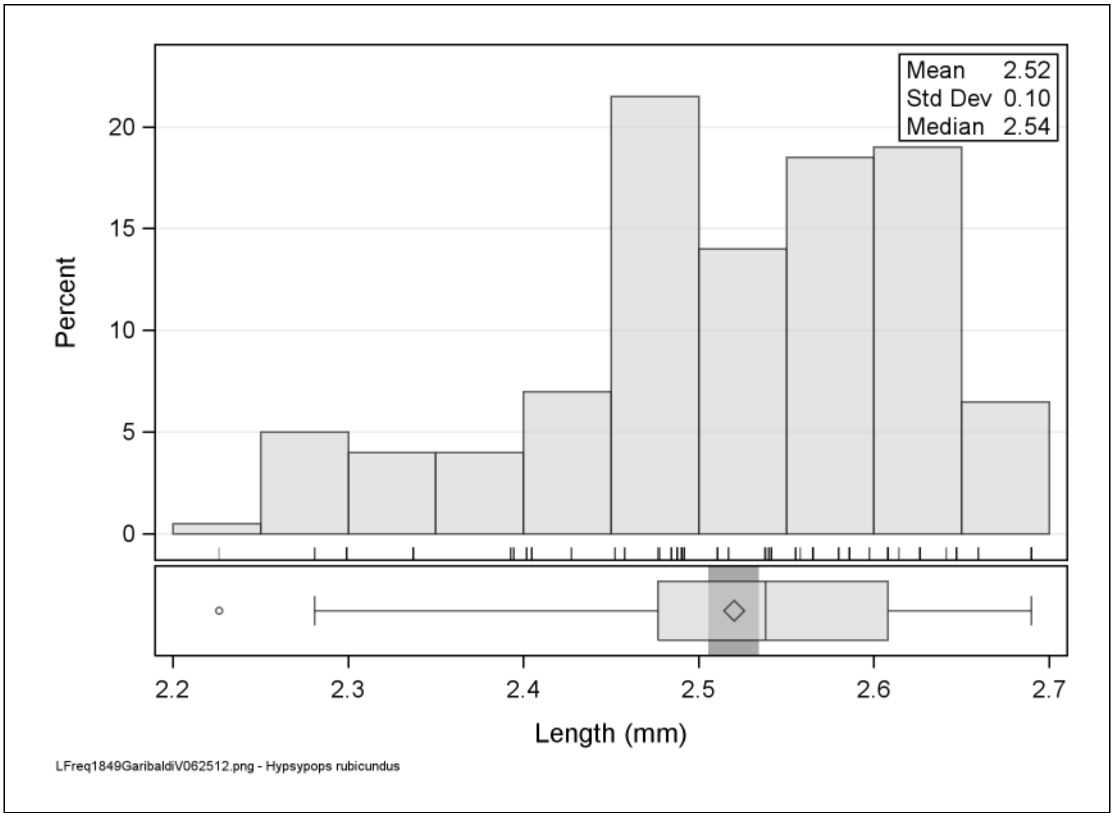


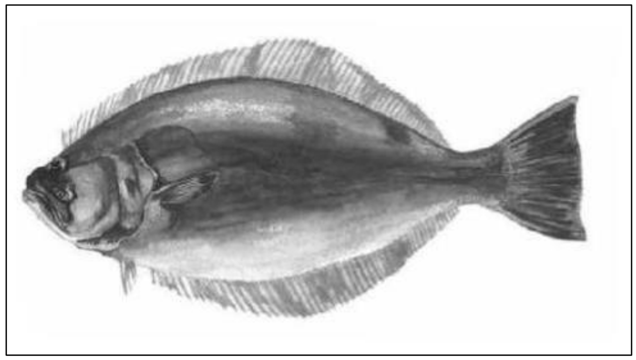
Figure 3-34. Length frequency for garibaldi larvae collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

Refer to Figure 3.2-4 for symbol explanation.

3.3.4.5.4 Impact Modeling Results

Garibaldi larvae were only collected at the entrainment location during three of the monthly surveys (**Figure 3-31**) and at the source water stations during only two of the monthly surveys (**Figure 3-32**). Because of the low frequency of occurrence of garibaldi larvae in monthly samples, the data for garibaldi larvae were not analyzed using the ETM.

3.3.4.6 California Halibut (*Paralichthys californicus*)



California halibut (*Paralichthys californicus*) is an important part of California's commercial and recreational fisheries (Starr et al. 1998; Kramer and Sunada 2001). It ranges from northern Washington to Bahia Magdalena, southern Baja California and is found from very shallow nearshore waters in bay nursery grounds to depths of at least 185 m (607 ft) (Miller and Lea 1972; Haaker 1975). Juveniles and adults typically occur on sandy sediments at depths less than 30 m (98.5 ft) but sometimes concentrate near rocks, algae, or Pacific sand dollar (*Dendraster excentricus*) beds (Feder et al. 1974). As with other flatfishes, they frequently lie buried or partially buried in the sediment. Newly settled and juvenile halibut often occur in non-vegetated shallow embayments and occasionally on the outer coast, suggesting that bays are an important nursery habitat for this species (Kramer and Sunada 2001).

3.3.4.6.1 Life History and Ecology

California halibut is a broadcast spawner with eggs being fertilized externally. The spawning season is generally thought to extend from February to August with most spawning occurring in May (Frey 1971), although some fall spawning may also occur. The average number of eggs per spawn is 313,000–589,000 with an average reproductive output of approximately 5.5 million eggs per spawning season (Caddell et al. 1990). During spawning season females may release eggs every seven days and the largest individuals may produce in excess of 50 million eggs per year. Captive specimens were observed to spawn at least 13 times per season. Halibut eggs are 0.7–0.8 mm (0.027–0.031 in.) in diameter (Ahlstrom et al. 1984) and are most abundant in the water column at depths less than 75 m (246 ft) and within 6.5 km (4.0 mi) from shore (Kramer and Sunada 2001).

Upon hatching, the larvae (1.6–2.1 mm [0.06–0.08 in.]) NL [Moser 1996]) are pelagic (Frey 1971) and most abundant between Santa Barbara, California, and Punta Eugenia, Baja California Sur (Ahlstrom and Moser 1975) from January through April and June through August (Moser 1996). A summary of the 1951–1984 CalCOFI data showed that halibut larvae were in the water column from February–March and July–August with the peaks in abundance occurring in February and August (Moser and Smith 1993). Moser and Pommeranz (1999) found the majority of the halibut larvae that they caught were in their 2.3 mm (0.09 in.) size class. California halibut has a relatively short pelagic larval stage, from 20–29 days (Gadomski et al. 1990). Larval transformation occurs at a length of about 7.5–9.4 mm (0.3–0.4 in.) SL (Moser 1996) at which time the young fish settle to the bottom, generally in bays but also occasionally in shallow substrates along the open coast (Haugen 1990). Kramer (1991) found that 6–10 mm (0.2–0.4 in.) California halibut larvae grew <0.3 mm/day (0.012 in/day), while larger 70–120 mm (2.8–4.7

in.) halibut grew about 1.0 mm/day (0.04 in./day). In a laboratory study, California halibut held at 16°C (60.8°F) grew to a length of 11.1 mm ± 2.61 (0.44 in. ± 0.1) (SD) in two months from an initial hatch length of 1.9 mm (0.07 in.) (Gadomski et al. 1990). After settling in the bays, the juveniles may remain there for about two years until they emigrate to the outer coast. Males mature at 2–3 years and 20–23 cm (7.9–9.0 in.) SL; females mature at 4–5 years and 38–43 cm (14.9–16.9 in.) SL (Fitch and Lavenberg 1971; Haaker 1975). Males emigrate out of the bays when they mature (i.e., at ~20 cm [7.9 in.]) but females migrate out as subadults at a length of about 25 cm (9.8 in.) (Haugen 1990). Subadults remain nearshore at depths of 6–20 m (19.7–65.6 ft) (Clark 1930; Haaker 1975). California halibut may reach 152 cm (60 in.) and 33 kg (73 lb) (Eschmeyer et al. 1983). Individuals may live as long as 30 years (Frey 1971).

Summary of California halibut distribution and life history attributes.

Range: Northern Washington to Bahia Magdalena, southern Baja California

Life History:
- **Maximum size**: 152 cm (60 in.) and 33 kg (73 lb)
- **Age at maturity:** Males 2–3 yr, females 4–5 yr
- **Life span**: 30 yr
- **Spawning**: Broadcast eggs from February to August with peak in May; multiple spawning with an average seasonal output of approximately 5.5 million eggs

Habitat: Sand bottoms and soft substrates along shallow parts of the continental shelf

Fishery: Important commercial and recreational fisheries

3.3.4.6.2 Population Trends and Fishery

California halibut have a high commercial and recreational fishery value. The population for this species appears to be stable in the majority of its range off the coast of California (Haugen 1990). California halibut has shown a historical decline in commercial landings, mainly due to overfishing. Maximum landing of 5,000,000 pounds in 1919 were reduced to a historical low of 257,000 pounds in 1970. In the late 1950s and 1960s, there was a slight increase in landings following warmer waters during El Niño events. The fishery for California halibut was reviewed by Kramer and Sunada (2001) and recent catch statistics are available through the Pacific States Marine Fisheries Commission (PSMFC) PacFIN (commercial) and RecFIN (recreational) databases. Historically, halibut have been commercially harvested by three principal gear types: otter trawl, set gill and trammel nets, and hook and line. Presently there are numerous gear, area, and seasonal restrictions that have been imposed on the commercial halibut fishery for management purposes. In southern California, the average annual recreational catch during 2004–2011 was 16,914 fish, while commercial landings over the same time period averaged approximately 54,706 pounds with an average annual value of $262,226 (**Table 3-22**). During this period, the commercial landings have steadily declined in Los Angeles County from a high of 112,481 pounds in 2004 to less than 40,000 pounds in 2009–2011.

California halibut populations are thought to be limited by the amount of available nursery habitat because juvenile halibut appear to be dependent on shallow water embayments as nursery areas. The historical declines in California halibut landings are considered to correspond to a decline in shallow water habitats in southern California associated with dredging and filling of bays and wetlands. It is a popular species for sport and commercial fishing but the fisheries are well managed and existing levels of fishing are not thought to pose any significant threat.

Table 3-22. Annual landings and revenue for California halibut based on RecFIN data (Los Angeles–San Diego region) and PacFIN data (Los Angeles County), 2004–2011.

	Recreational Fishery	Commercial Fishery	
Year	Estimated Numbers Landed	Landed Weight (lb)	Revenue ($)
2004	20,539	112,418	$487,046
2005	22,737	62,080	$296,200
2006	28,224	56,007	$272,572
2007	15,940	46,640	$244,362
2008	14,188	57,061	$291,877
2009	9,732	25,467	$118,977
2010	16,701	38,362	$184,470
2011	7,250	39,612	$202,303
Average	**16,914**	**54,706**	**$262,226**

3.3.4.6.3 Sampling Results

California halibut was the eleventh most abundant larval fish taxon collected at the intake station (SWE) using towed plankton nets with a mean concentration of 3.4 per 1,000 m^3 (**Table 3-4**), and the twelfth most abundant taxon collected at the source water stations, also with a mean concentration of 3.4 per 1,000 m^3 (**Table 3-6**). California halibut were only collected at the intake station during two months (January and February 2012, [**Figure 3-35**]), but were present in five of the monthly source water surveys from July 2011 to February 2012 (**Figure 3-36**). Although the sample size was small, larvae were more abundant in the nighttime samples (**Figure 3-37**). Only seven specimens were measured from the intake station and they ranged in size from 1.5 to 2.3 mm (0.06–0.10 in.) with a mean size of 2.0 mm (0.08 in.) (**Figure 3-38**).

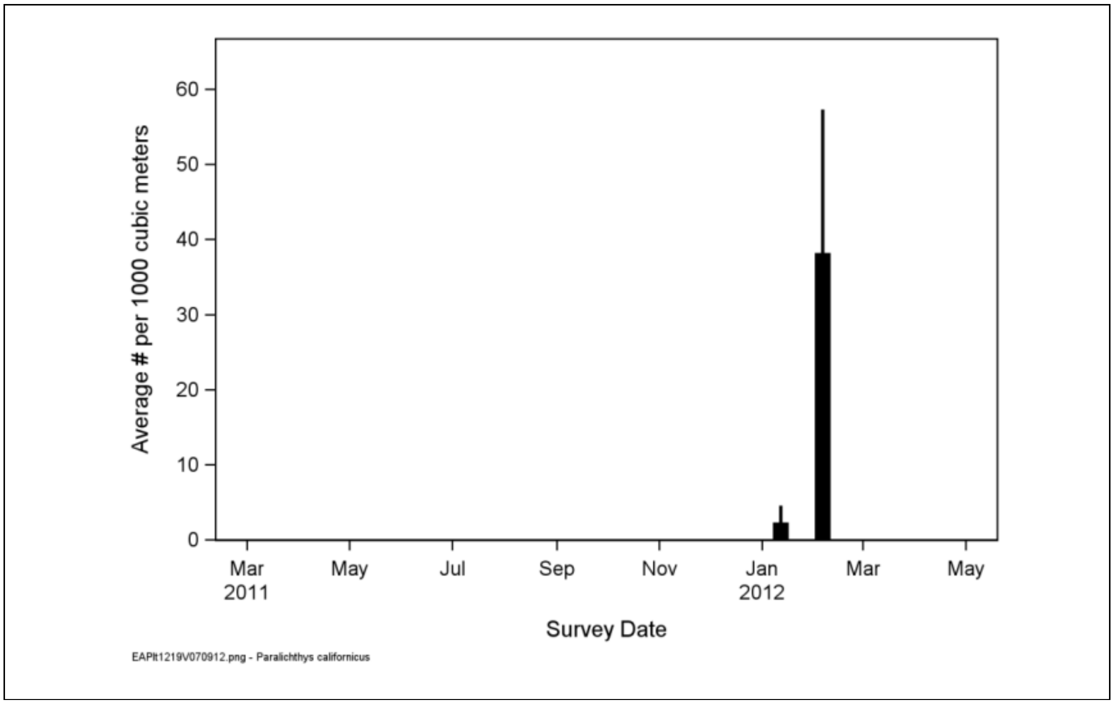


Figure 3-35. Mean concentration (#/1,000 m^3) and standard error for California halibut larvae collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

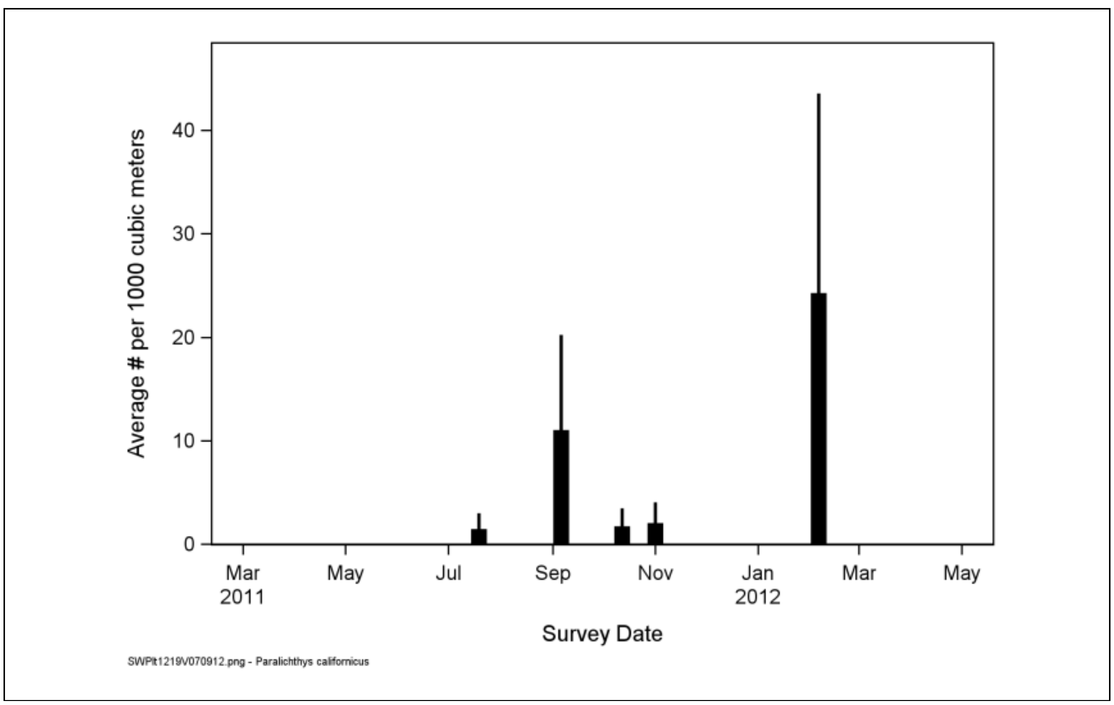


Figure 3-36. Mean concentration (#/1,000 m^3) and standard error for California halibut larvae collected using towed plankton nets at source water stations (SW1, SW2, SW3) from March 2011 through March 2012.

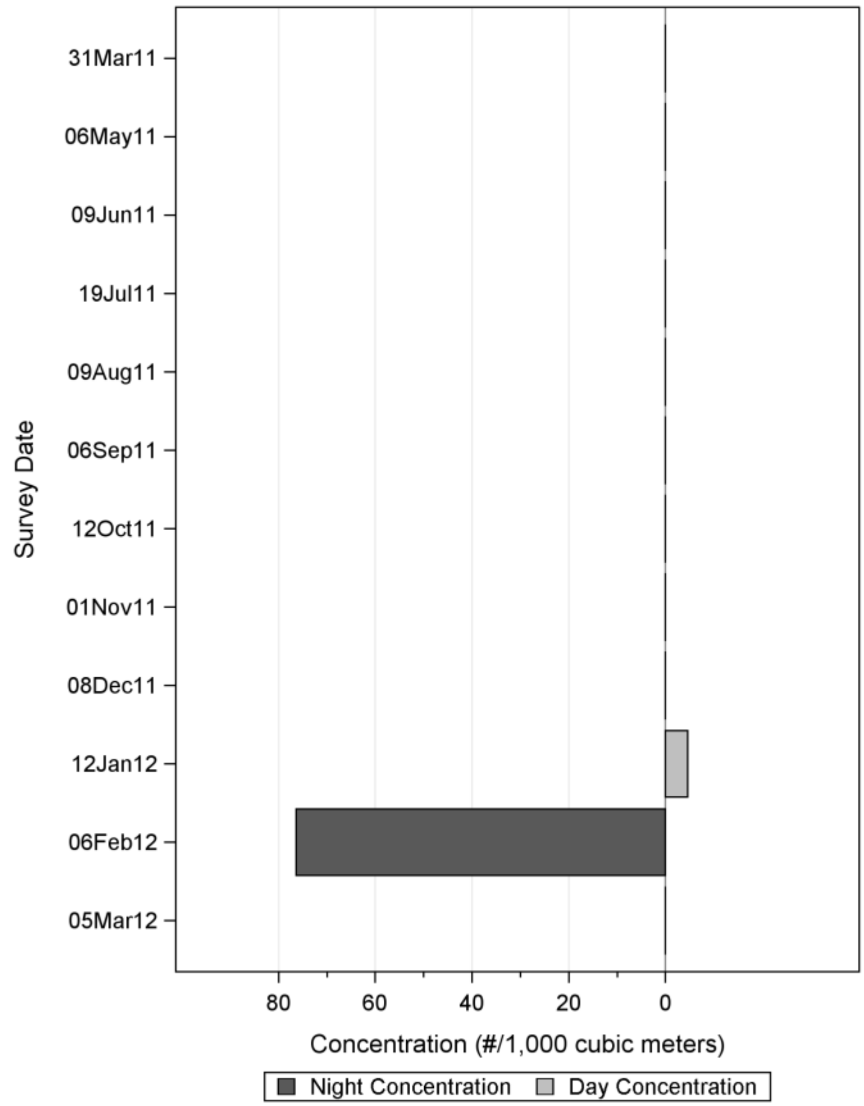


Figure 3-37. Mean concentration (#/1,000 m^3) in night and day samples for California halibut larvae collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

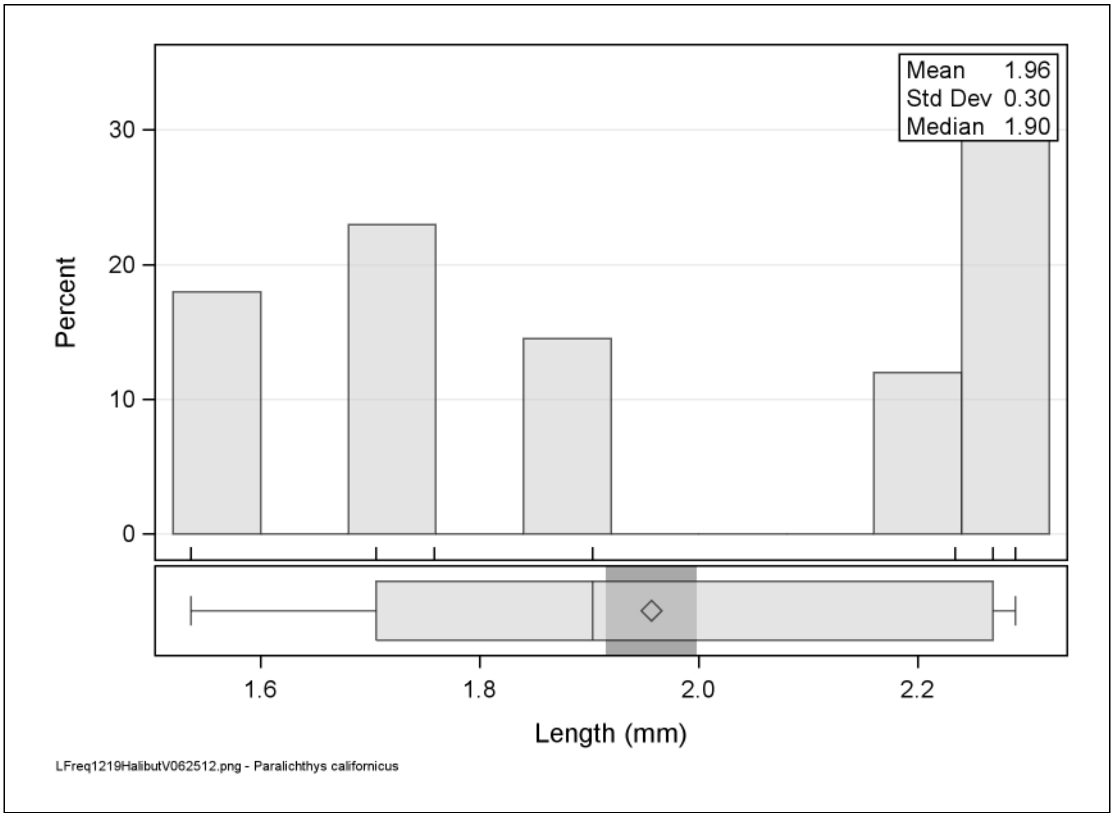


Figure 3-38. Length frequency for California halibut larvae collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

Refer to Figure 3.2-4 for symbol explanation.

3.3.4.6.4 Impact Modeling Results

California halibut larvae were only collected during two of the monthly surveys at the intake location (**Figure 3-35**). Because of the low frequency of occurrence and overall low abundance of California halibut larvae the data were not analyzed using the ETM.

3.3.4.7 Rock Crabs (Cancridae)

Crabs of the family Cancridae are widely distributed in coastal waters of the west coast of North America (Nations 1975). They occur in intertidal and shallow subtidal habitats on both rock and sand substrate. All of the nine species known to occur in the northeast Pacific were formerly classified into a single genus, *Cancer*, but a taxonomic revision of the family by Schweitzer and Feldmann (2000) based on molecular, fossil, and morphological evidence resulted in dividing the genus into four genera: *Glebocarcinus*, *Romaleon*, *Metacarcinus* and *Cancer*. The following five species of cancrid crabs (Cancer crabs) megalops may occur in the vicinity of the WBDDF, but due to overlapping ranges in sizes, and similarities in morphology, the megalops larvae could not always be reliably identified to the level of species.

Common name(s)	Scientific Name
Red rock crab	*Cancer productus*
Yellow crab	*Metacarcinus anthonyi*
Pacific (brown) rock crab [pictured]	*Romaleon antennarius*
Slender crab	*Metacarcinus gracilis*
Hairy rock crab	*Romaleon jordani*

Each species has characteristic differences in distribution, preferred habitat, growth rates, and demographic parameters (Winn 1985). For example, the yellow crab is the largest of these species, lives primarily on silty sand bottoms, and can grow to a carapace width (CW) exceeding 178 mm (7.01 in.). The Pacific rock crab is also a relatively large species (CW >155 mm [6.10 in.]) but lives primarily at sand/rock interfaces, among kelp forests, and also in bays on sand and shell debris. The slender crab is a smaller species (CW >130 mm [5.12 in.]) associated with mixed rock-sand substrates in shallow outer coast habitats. Maximum clutch sizes in Cancer crabs can range from as many as 5,000,000 eggs in yellow crab to approximately 50,000 in *G. oregonensis,* one of the smaller species (Hines 1991). These types of differences imply that specific information on life history parameters cannot readily be generalized among cancrid species.

3.3.4.7.1 Life History and Ecology

All species of Cancer crabs share certain fundamental life history traits. Eggs are extruded from the ovaries through an oviduct and are carried in a sponge-like mass beneath the abdominal flap of the adult female. After a development period of several weeks, the eggs hatch and a pre-zoea larva emerges, beginning the planktonic life history phase. As in all crustaceans, growth progresses through a series of molts. The planktonic larvae advance through six stages of successive increases in size: five zoea (not including the brief pre-zoea stage) followed by one megalops stage. After several weeks as planktonic larvae, the crabs metamorphose into the first crab stage (first instar) and settle out to begin their benthic life history phase. Maturity is generally attained within 1–2 years. Mature females mate while in the soft shell molt condition and extrude fertilized eggs onto the abdominal pleopods. Females generally produce one or two batches per year, typically in winter or spring. Fecundity per batch increases significantly with female body size (Hines 1991). While the longevity of rock crabs is not well known, many crabs may reach 5–6 years of age (Leet et al. 2001).

Pacific rock crab females can extrude between approximately 156,000 and 5 million eggs per batch (Hines 1991). Females on average produce a single batch per year; however, due to occasional multiple spawnings, the average number of batches per year may be greater than one (Carroll 1982). Eggs require a development time of approximately 7–8 weeks from extrusion to hatching (Carroll 1982). Larval development in the Pacific rock crab was described by Roesijadi (1976). Eggs hatch into pre-zoea larvae that molt to first stage zoea in less than 1 hour. Average larval development time (from hatching through completion of the fifth stage) was 36 days at 13.8°C (57°F). Although some crabs molted to the megalops stage, none molted to the first crab instar stage, so the actual duration of the megalops stage is unknown. A reasonable estimate can be derived from studies of slender crab by Ally (1975), who found an average duration of megalops stage of 14.6 days. An average estimated length of time from hatching to settling is approximately 45 days.

During their planktonic existence, Cancer crab larvae can become widely distributed in nearshore waters. In a study in Monterey Bay, Graham (1989) found that Pacific rock crab stage-1 zoea are most abundant close to shore and that subsequent zoeal stages tend to remain within a few kilometers of the coastline. The adult population primarily resides in relatively shallow rocky areas, and the nearshore retention of larvae in Graham’s study (1989) was related to the formation of an oceanographic frontal zone in northern Monterey Bay that prevented substantial offshore transport during upwelling periods.

The nearshore distribution of Cancer crab larvae depends upon developmental stage. Shanks (1985) presented evidence that early stage larvae of Cancer crabs (probably yellow crab in his southern California study) generally occur near the bottom, in depths up to 80 m (262 ft). Late stage larvae, however, were more abundant near the surface. He found that a combination of physical factors (primarily including wind-generated surface currents and tidally forced internal waves) caused megalopae to be transported shoreward. Late stage larvae (megalops) generally begin to recruit to the nearshore habitat in spring (Winn 1985).

Summary of Cancer crab distribution and life history attributes.

Range: Yellow crab–Humboldt Bay to southern Baja California; Pacific rock crab–Queen Charlotte Sound, British Columbia to Cabo San Lucas, Baja California; Slender crab–Alaska to Sebastian Vizcaino Bay, Baja California.

Life History:

- **Maximum size**: Yellow crab–176 mm (7.0 in.) carapace width; Pacific rock crab–178 mm (7.0 in.) carapace width; Slender crab–89 mm (3.5 in.) carapace width.
- **Age at maturity:** 1-2 yr
- **Life span**: 5–6 yr
- **Spawning**: Female carries eggs until hatching; spawn year-round but mainly winter and spring; multiple spawning with a mean batch fecundity of up to several million (yellow crab).

Habitat: Rocky reefs in shallow subtidal zone and at sand-rock interface (Pacific rock crab); soft substrates on shallow shelf, or bays and estuaries (yellow crab and slender crab).

Fishery: Sport and commercial fisheries (yellow crab and Pacific rock crab).

3.3.4.7.2 Population Trends and Fishery

In southern California, the three largest species of Cancer crabs (Pacific rock crab, red rock crab, and yellow crab) contribute to economically significant fisheries. There is no commercial fishery for the slender crab. Recreational crabbing is popular in many areas and is often conducted in conjunction with other fishing activities. There is a 102 mm (4.0 in.) minimum carapace width regulation and a personal bag limit of 35 crabs per day. The commercial harvest has been difficult to assess on a species-by-species basis because the fishery statistics are combined into the general "rock crab" category. Commercial fishing regulations currently specify a minimum harvest size of 108 mm (4.3 in.) carapace width for rock crabs. Recent catch statistics from the PSMFC PacFIN (commercial) database for 2004–2011 from Los Angeles County show that the average annual commercial catch and ex-vessel revenue from rock crab during this period was 91,316 pounds and $130,222, respectively (**Table 3-23**). The year 2007 had the highest landings and revenue total, followed by steady declines in subsequent years to a low of 51,442 pounds and $69,797 in 2011.

Table 3-23. Annual landings and revenue for rock crab in Los Angeles County based on PacFIN data, 2004–2011.

	Commercial Fishery	
Year	Landed Weight (lb)	Revenue ($)
2004	75,638	$109,536
2005	71,181	$105,941
2006	75,118	$112,994
2007	186,966	$253,217
2008	133,042	$194,491
2009	81,093	$118,092
2010	56,050	$77,708
2011	51,442	$69,797
Average	**91,316**	**$130,222**

3.3.4.7.3 Sampling Results

There were five taxa of Cancer crab megalops identified at both the intake and source water stations, with yellow crab the only taxon that could be reliably identified to the species level. Pacific rock crab/slender crab megalops was the most abundant target invertebrate taxon collected at the intake station using towed plankton nets, with a mean concentration of 46.0 per 1,000 m^3 (**Table 3-4**), and also the most abundant taxon at the source water stations with a mean concentration of 28.8 per 1,000 m^3 (**Table 3-6**). Cancer crab megalops (all taxa combined) occurred in all surveys with peak average abundances in March 2011 at the intake station (380 per 1,000 m^3) (**Figure 3-39),** and the source water stations (169 per 1,000 m^3) (**Figure 3-40**). They were almost exclusively collected in nighttime samples during all months (**Figure 3-41**).

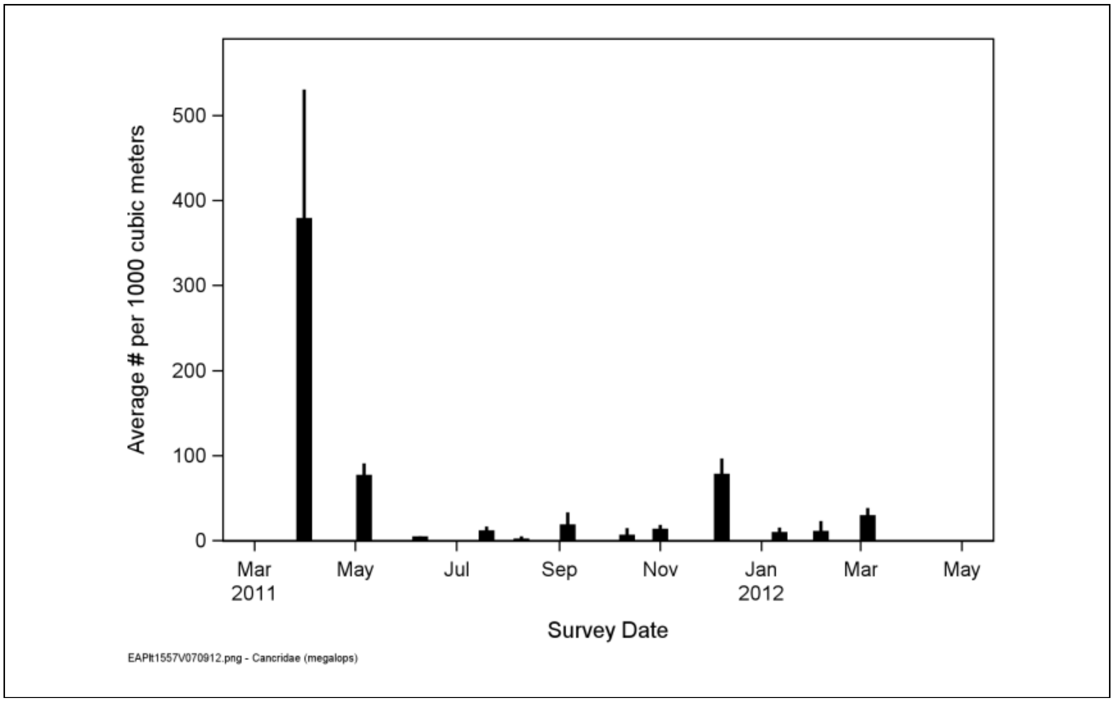


Figure 3-39. Mean concentration (#/1,000 m^3) and standard error for Cancer crab megalops collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

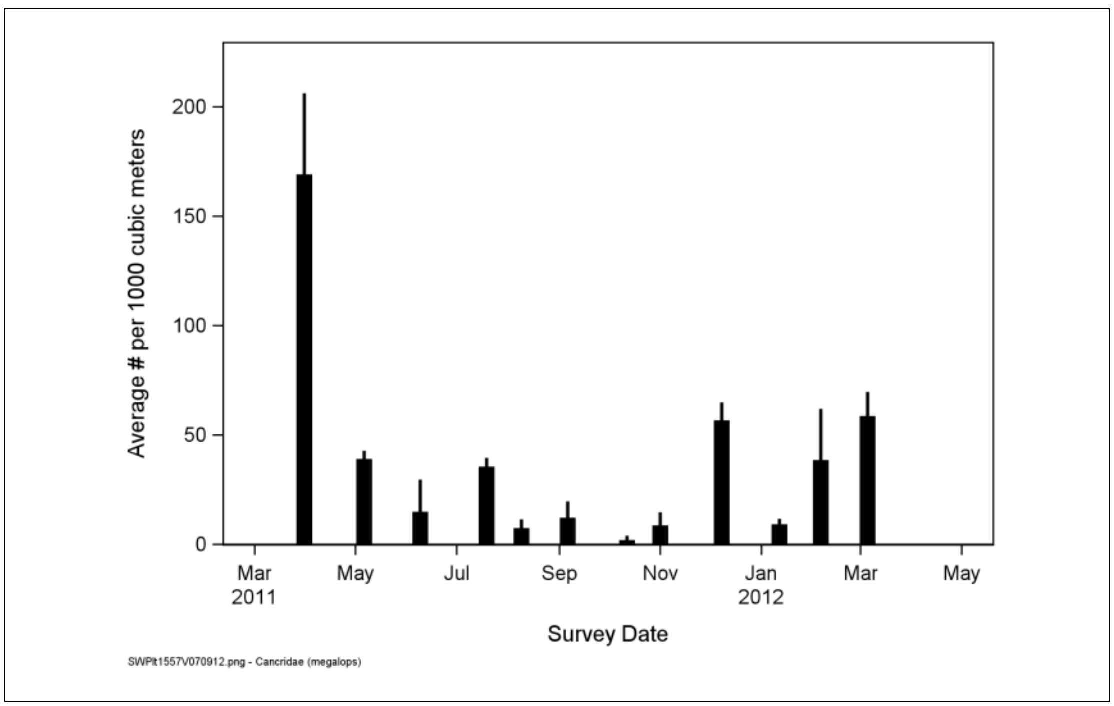


Figure 3-40. Mean concentration (#/1,000 m^3) and standard error for Cancer crab megalops collected using towed plankton nets at source water stations (SW1, SW2, SW3) from March 2011 through March 2012.

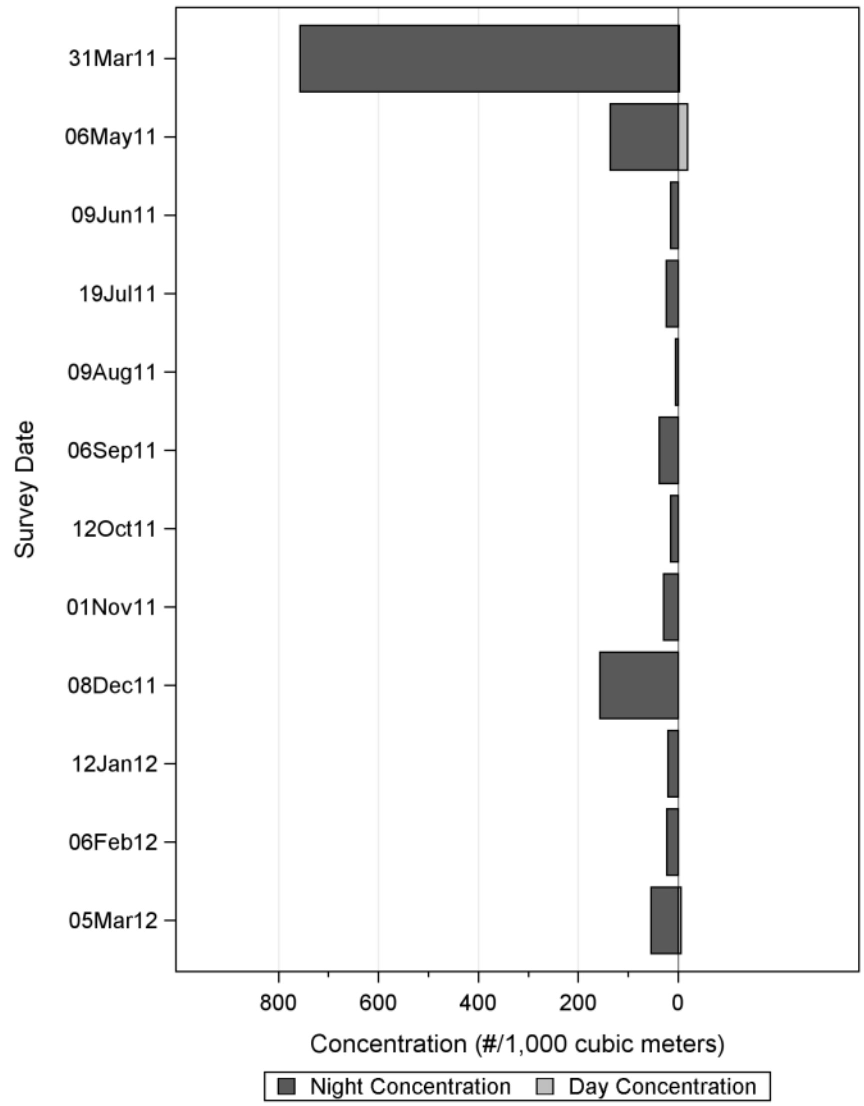


Figure 3-41. Mean concentration (#/1,000 m^3) in night and day samples for Cancer crab megalops collected using towed plankton nets at the intake station (SWE) from March 2011 through March 2012.

3.3.4.7.4 Impact Modeling Results

Estimates of 26,955, 39,844, and 3,515,206 Cancer crab megalops larvae would have been entrained during the March 2011 through March 2012 study period based, respectively, on daily unscreened intake flows of 1,309 m^3 (0.346 mgd) and 1,935 m^3 (0.511 mgd) for the demonstration facility, and a projected flow of 170,722 m^3 (45.1 mgd) for the final proposed facility (**Table 3-5**).

Empirical Transport Model (*ETM*)

The larval duration for Cancer crab larvae through the megalops stage was assumed to be 45 days based on larval duration estimates for slender crab (Ally 1975) and Pacific rock crab (Roesijadi 1976). The data used to calculate the ETM estimates (f_i values) for cancer crab megalops are shown in **Tables 3-24** and **3-25**. Data collected during the March 2011 survey period produced the largest estimate of the source water population and the largest entrainment estimates (**Table 3-24**). The total estimated annual entrainment through an unscreened intake for the projected flow of 170,722 m^3 (45.1 mgd) for the proposed facility represents approximately 0.02 percent of the extrapolated source water population potentially subject to entrainment.

The source water estimates in **Table 3-24** were used in calculating the weights (f_i) used in the ETM calculations which showed that over 44 percent of the larvae collected using towed plankton nets at the source water stations were collected during the March 2011 surveys (**Table 3-25**). Although Cancer crabs are generally associated with shallower nearshore habitats, they can occur in deeper water. Therefore, ETM analyses were done using depths of both 30 m (98 ft) and 100 m (330 ft) as limits on the back-projections. The average alongshore displacement for the source water population was 56.52 km (35.12 mi) and 75.94 km (47.19 mi) for the back-projections limited to depths of 30 m (98 ft) and 100 m (328 ft), respectively.

The ETM estimates attributable to entrainment through an unscreened intake for Cancer crabs ranged from 0.00007 (0.007 percent) to 0.00023 (0.023 percent) for the WBDDF intake volumes, to a high of 0.020 (2.0 percent) for the intake volume at the proposed facility (**Table 3-26**). The higher estimate for the proposed facility is due to higher intake volume.

Table 3-24. Daily and survey period estimates for Cancer crab megalops for entrainment, sampled source water, and extrapolated source water. The entrainment estimate was calculated based on the projected flow for the proposed facility of 170,722 m^3 (45.1 mgd) through an unscreened intake. Extrapolated source water population calculated based on a depth of 30 m (98 ft).

Survey Date	Survey Days	Daily Entrainment Estimate (45.1 mgd)	Survey Entrainment Estimate (45.1 mgd)	Daily Sampled SW Estimate	Sampled SW Survey Period Estimate	Extrapolated SW Survey Population
31 Mar 11	31	64,791	2,008,526	5,562,149	172,426,613	7,314,888,958
06 May 11	35	13,209	462,302	1,151,064	40,287,251	3,491,734,106
09 Jun 11	37	886	32,782	448,762	16,604,196	353,902,309
19 Jul 11	31	2,069	64,124	709,713	22,001,113	1,228,595,381
09 Aug 11	24	474	11,382	182,329	4,375,899	168,403,717
06 Sep 11	32	3,281	104,991	290,820	9,306,226	184,799,384
12 Oct 11	28	1,258	35,232	101,396	2,839,079	84,164,300
01 Nov 11	29	2,466	71,501	195,667	5,674,343	71,352,931
08 Dec 11	36	13,422	483,208	1,391,994	50,111,773	1,946,086,528
12 Jan 12	30	1,737	52,118	262,167	7,865,006	503,054,106
06 Feb 12	26	1,958	50,903	985,143	25,613,722	1,052,459,135
05 Mar 12	27	5,116	138,136	1,224,745	33,068,124	48,434,508
	Annual Totals		**3,515,203**		**390,173,344**	**16,447,875,364**

Table 3-25. Data used in ETM calculations for Cancer crab megalops. The estimates of PE_i for each survey period i, represent the unadjusted source water estimates and were calculated using only towed plankton net data from the intake and source water stations. The *PE*s were calculated based on three unscreened intake volumes: a) actual flow during the year for the WBDDF of 240 gpm or 1,309 m^3 (0.346 mgd), b) design flow for the WBDDF of 355 gpm or 1,935 m^3 (0.511 mgd), and c) a projected flow for the proposed facility of 170,722 m^3 (45.1 mgd).

Survey Date	a) Actual Average Demonstration Plant Flow = 240 gpm		b) Demonstration Plant Design Flow = 355 gpm		c) Final Desalination Plant Intake Flow = 45.1 mgd		Weight	Source Water Extrapolation Alongshore (km)	
	PE_i	PE_i Std. Error	PE_i	PE_i Std. Error	PE_i	PE_i Std. Error	Weight (f_i)	Depth (m) = 30	Depth (m) = 100
31-Mar-11	0.000089	0.000048	0.000132	0.000071	0.011649	0.006272	0.442	63.63	76.15
6-May-11	0.000088	0.000032	0.000130	0.000048	0.011475	0.004205	0.103	130.01	137.36
9-Jun-11	0.000015	0.000012	0.000022	0.000017	0.001974	0.001526	0.043	31.97	37.09
19-Jul-11	0.000022	0.000013	0.000033	0.000020	0.002915	0.001727	0.056	83.76	122.55
9-Aug-11	0.000020	0.000023	0.000029	0.000034	0.002601	0.003026	0.011	57.73	72.75
6-Sep-11	0.000087	0.000077	0.000128	0.000113	0.011282	0.009996	0.024	29.79	32.32
12-Oct-11	0.000095	0.000110	0.000141	0.000162	0.012409	0.014337	0.007	44.47	109.90
1-Nov-11	0.000097	0.000052	0.000143	0.000077	0.012601	0.006762	0.015	18.86	26.18
8-Dec-11	0.000074	0.000031	0.000109	0.000046	0.009643	0.004027	0.128	58.25	64.18
12-Jan-12	0.000051	0.000030	0.000075	0.000044	0.006627	0.003849	0.020	95.94	123.42
6-Feb-12	0.000015	0.000017	0.000023	0.000025	0.001987	0.002171	0.066	61.63	69.96
5-Mar-12	0.000032	0.000013	0.000047	0.000019	0.004177	0.001645	0.085	2.20	39.39
Average (n=12)	**0.000057**		**0.000084**		**0.007445**			**56.52**	**75.94**
Average (*PE*>0)	**0.000057**		**0.000084**		**0.007445**			**56.52**	**75.94**

Table 3-26. ETM estimates of P_M for Cancer crab megalops. Estimates of P_M based on estimated upcoast and downcoast source water populations based on back-projections limited to depths of 30 m (98 ft) and 100 m (328 ft). ETM estimates calculated for three unscreened intake volumes: a) actual flow during the year for the WBDDF of 1,309 m^3 (0.346 mgd), b) design flow for the WBDDF of 1,935 m^3 (0.511 mgd), and c) a projected flow for the proposed facility of 170,722 m^3 (45.1 mgd). Two estimates of standard error (Std. Error) are presented: one based on combining the variance components from the estimates comprising ETM, and the second using the weighted average coefficient of variation across the *PE*s for the twelve surveys.

Flow	CODAR Depth Limit	P_M Estimate	P_M Std. Error	P_M + Std. Error	P_M - Std. Error	CV Based Std. Error	P_M + CV Std. Error	P_M - CV Std. Error
a) WBDDF Actual Flow	30 m Depth	0.000156	0.126809	0.126965	-0.126652	0.000088	0.000244	0.000068
	100 m Depth	0.000066	0.126815	0.126881	-0.126749	0.000037	0.000103	0.000029
b) WBDDF Design Flow	30 m Depth	0.000231	0.126801	0.127032	-0.126571	0.000130	0.000361	0.000101
	100 m Depth	0.000098	0.126811	0.126908	-0.126713	0.000055	0.000153	0.000043
c) Proposed Design Flow	30 m Depth	0.019649	0.125069	0.144719	-0.105420	0.011062	0.030711	0.008588
	100 m Depth	0.008566	0.125756	0.134322	-0.117190	0.004822	0.013388	0.003744

3.4 Impact Modeling Summary

Impact analyses of the effects of larval entrainment by the WBDDF and a proposed full-scale facility were completed using the ETM approach for four taxonomic groups of fishes and Cancer crab megalopae. Impact estimates were based on intake flow volumes drawn through unscreened intakes. The narrow slot WWS modules present on the WBDDF intakes would presumably result in reduced impact estimates. The small intake volumes of 1,309 m^3 (0.346 mgd) and 1,935 m^3 (0.511 mgd) for the analyses of the WBDDF effects resulted in estimated impacts on the populations of larvae for fishes and crabs in the extrapolated source water of a hundredth, or thousandths, of a percent (**Table 3-27**). The estimated impacts were higher for the proposed full-scale facility, which were based on an intake volume of 170,722 m^3 (45.1 mgd) that was approximately two orders of magnitude larger than the WBDDF intake flows.

While ETM results will be affected by the volume of the intake relative to the source water, they can also be affected by other factors such as the number of days the larvae are estimated to be subject to entrainment. This affects both the overall level of mortality and the defined size of the source water for the larvae. Although the size of the extrapolated source water for the larvae may increase with the number of days the larvae are exposed to entrainment, the increase may not always be proportional. This is due to the use of CODAR data in estimating the larval transport over those periods. If the CODAR data indicate that the currents are predominantly alongshore, then the source water estimates will increase proportionally with the estimated period of exposure. However, if the current data indicate strong onshore transport, or recirculation, over a survey period then the estimated impacts for that period could be higher.

Seasonal changes in current speed and direction will also affect both the estimated size of the source water during each survey period and the ETM results. For example, the extrapolated source water for silversides based on a period of exposure of 13.7 days and within a depth limit of 30 m (98 ft), ranged from 2.1 to 15.4 km (1.3 to 9.6 mi) for the surveys when silverside larvae were collected at both the intake and source water stations (**Table 3-9**). During the November 2011 survey silverside larvae were only collected at the source water stations so no estimate of PE_i was calculated, however, the extrapolated source water extended up to a distance of 18.8 km (11.7 mi) alongshore during the survey. The extrapolated distances alongshore for other surveys when silverside larvae were not collected were even greater, thus demonstrating the effect that seasonal differences in currents can have on ETM results.

These differences are apparent when comparing the ETM results for silversides with white croaker, which had a very similar estimate of exposure to entrainment of 12.4 days. Although the average *PE* for silversides was only 73 percent higher than the average for white croaker, the ETM estimates for silversides were almost 600 percent higher than the estimates for white croaker (**Tables 3-9** and **3-113**). This seemingly disproportionate increase is due to the highest PE_i for silversides occurring during the survey when the currents estimated an alongshore source water extent of only 2.1 km (1.3 mi). The result is increased mortality to a population which is now concentrated into a smaller source water area.

Multiple factors affect ETM results so it is necessary to place the estimates of entrainment used in the ETM calculations into context, particularly with regard to the actual reproductive capacity for a species. Although the ETM results may indicate a large percentage loss to the source water population of larvae, the actual impacts due to entrainment may be negligible since the actual number of larvae entrained is very small relative to the reproductive capacity of the particular species. Although this can be done using adult equivalent modeling approaches, this may be unnecessary when the absolute levels of entrainment are very low. For example, the total entrainment estimates for white croaker and California halibut larvae for the proposed full-scale project were 945,568 and 181,368 per year, respectively. These annual entrainment estimates represent the annual production of a few females for white croaker (based on an average batch fecundity of 19,000 eggs and an average of 19 batch spawnings per year [Love et al. 1994]) to perhaps only one female for California halibut (based on an average batch fecundity of 522,000 eggs and an average of 12 batch spawnings per year [Caddell et al. 1990]).

Another way to place ETM results into context is to use an approach referred to as 'area of production foregone' (APF), which scales the ETM results based on the available adult spawning habitat. The APF places the ETM results into context by estimating the area of adult spawning habitat that would be necessary to replace the larvae lost due to entrainment. This requires that an actual habitat type can be identified that is associated with spawning by a species. For some fishes this is relatively easy. Fishes such as garibaldi and kelpfishes occupy rocky reef areas that they also use for attaching eggs into nests that are guarded by these fishes. The APF also requires that there be estimates of these habitat areas inside the extrapolated source water for the larvae. Therefore, even though the source water for the larvae may extend over many kilometers of coastline due to larval transport from the habitats where the larvae are spawned, the actual area of critical habitat for spawning within the source water may be much smaller. In the case of rocky reef fishes such as garibaldi and kelpfishes, this type of habitat is limited in Santa Monica Bay and is likely restricted to the areas around Redondo Beach and to rocky areas at the north and south ends of the Bay. The APF estimate is calculated by multiplying the estimate of spawning habitat within the source water by the proportional mortality (P_M) from the ETM.

In addition to fishes such as garibaldi and kelpfishes mentioned above, the APF would also be useful for placing the results for silversides (mostly jacksmelt) into context. Although jacksmelt can occur in deeper water on the open coast they are also commonly associated with bay and estuarine habitats (Love 2011). Jacksmelt females attach their eggs to subtidal vegetation, such as kelp and eelgrass, and also other structures (Love 2011). This type of habitat is most prevalent around Redondo Beach and King Harbor where there are small kelp beds in areas exposed to ocean water, shallow areas where eelgrass may occur, and also rock jetties and pier pilings covered with marine algae. The average alongshore distance for the extrapolated source water for the larvae was only 7.6 km (4.7 mi) (**Table 3-9**), indicating that the larvae collected in the entrainment samples were likely spawned in areas relatively close to the intake. To calculate the APF for jacksmelt, an estimate of these types of habitats would need to be calculated and then the estimated P_M of 7.9 percent applied to that estimate. Although it is unlikely that the removal of 7.9 percent of the larval production from the area would affect the local population of jacksmelt, the APF estimate could be used to determine appropriate mitigation that would completely compensate for the entrainment losses.

Table 3-27. Estimates of proportional mortality (percentages in parentheses) on source populations for taxa analyzed using ETM. ETM estimates of additional proportional mortality resulting from the WBDDF intake for each taxa are based on estimated upcoast and downcoast source water populations based on back-projections limited to depths of 30 m (98 ft) and 100 m (328 ft); and 100 m (328 ft) and 300 m (984 ft) for northern anchovy. ETM estimates calculated for three unscreened intake volumes: a) actual flow during the year for the WBDDF of 1,309 m^3 (0.346 mgd), b) design flow for the WBDDF of 1,935 m^3 (0.511 mgd), and c) a projected flow for the proposed facility of 170,722 m^3 (45.1 mgd).

Taxon	CODAR Depth Limit	a) ETM Estimate WBDDF Actual Flow	b) ETM Estimate WBDDF Design Flow	c) ETM Estimate WBDDF Full Scale Flow
silversides	30 m Depth	0.00067 (0.067)	0.00098 (0.098)	0.07878 (7.878)
	100 m Depth	0.00006 (0.006)	0.00009 (0.009)	0.00744 (0.744)
white croaker	30 m Depth	0.00011 (0.011)	0.00016 (0.016)	0.01351 (1.351)
	100 m Depth	0.00003 (0.003)	0.00004 (0.004)	0.00366 (0.366)
northern anchovy	100 m Depth	0.00003 (0.003)	0.00004 (0.004)	0.00323 (0.323)
	300 m Depth	0.00002 (0.002)	0.00003 (0.003)	0.00267 (0.267)
kelpfishes	30 m Depth	0.00018 (0.018)	0.00026 (0.026)	0.02234 (2.234)
	100 m Depth	0.00006 (0.006)	0.00009 (0.009)	0.00741 (0.741)
Cancer crabs	30 m Depth	0.00016 (0.016)	0.00023 (0.023)	0.01965 (1.965)
	100 m Depth	0.00007 (0.007)	0.00010 (0.010)	0.00857 (0.857)

4.0 Wedgewire Screen (WWS) Efficiency Study

The second objective of the Intake Effects Assessment studies was to determine the efficiency of the cylindrical narrow-slot WWS intake modules at reducing the effects of entrainment and impingement. This aspect of the study involved three components: 1) sampling to provide a measure of entrainment reduction through two WWS intake modules with different size slot openings (1 mm [0.04 in.] and 2 mm [0.08 in.] slot widths), 2) analysis of length-head capsule data to predict entrainment probabilities for different taxa, larval size classes, and different screen slot dimensions, and, 3) collection of videographic data to assess overall impingement effects and to compare impingement performance between the two WWS modules.

The intake screening system for the WBDDF feedwater system is comprised of two cylindrical intake modules installed on the abandoned offshore discharge structure for RBGS Units 1&2, located 0.15 km (0.09 mile) offshore from the northwest corner of the King Harbor breakwater at the approximate location of Station SWE (**Figure 3-1**). The water depth surrounding the abandoned discharge structure is approximately 10.2 m (33.5 ft) MLLW and the top of the riser extends 2.4 m (8.0 ft) above the sand seabed and protective rock armoring surrounding the riser. The two intake modules are 3.7 m (12 ft) apart and located atop riser pipes that extend above the abandoned discharge riser to a depth of approximately 7.0-8.2 m (23-27 ft) beneath the sea surface (depending on tide stage) (**Figure 4-1**). The WWS intake modules were constructed with two slot opening widths: 1 mm (0.04 in.) and 2 mm (0.08 in.). The dimensions of the 1 mm (0.04 in.) and 2 mm (0.08 in.) WWS modules were sized to ensure a maximum through-screen velocity of 0.1 m/sec (0.33 ft/sec). An example WWS module is shown in **Figure 4-2**. Seawater pumped through the WWS modules flows through a system of 20-cm (8-inch) pipes that extend approximately 445 m (1,460 ft) though the abandoned discharge conduit to the onshore pilot project facility at SEA Lab.

4.1 Screen Entrainment Comparison Study

This section presents the first component of the WWS efficiency studies, which involved the collection of samples through the two WWS modules for comparison with samples collected by drawing seawater through an intake with a larger mesh screen that would only exclude larger non-entrainable organisms (unscreened intake). The data collected through the unscreened intake were intended to provide estimates of the concentrations of larvae in the source water that would be entrained through an intake with a standard sized intake screen. Screening efficiency was then to be evaluated by comparing larval concentrations in samples collected simultaneously from the two differently sized WWS intake modules and from the unscreened intake.

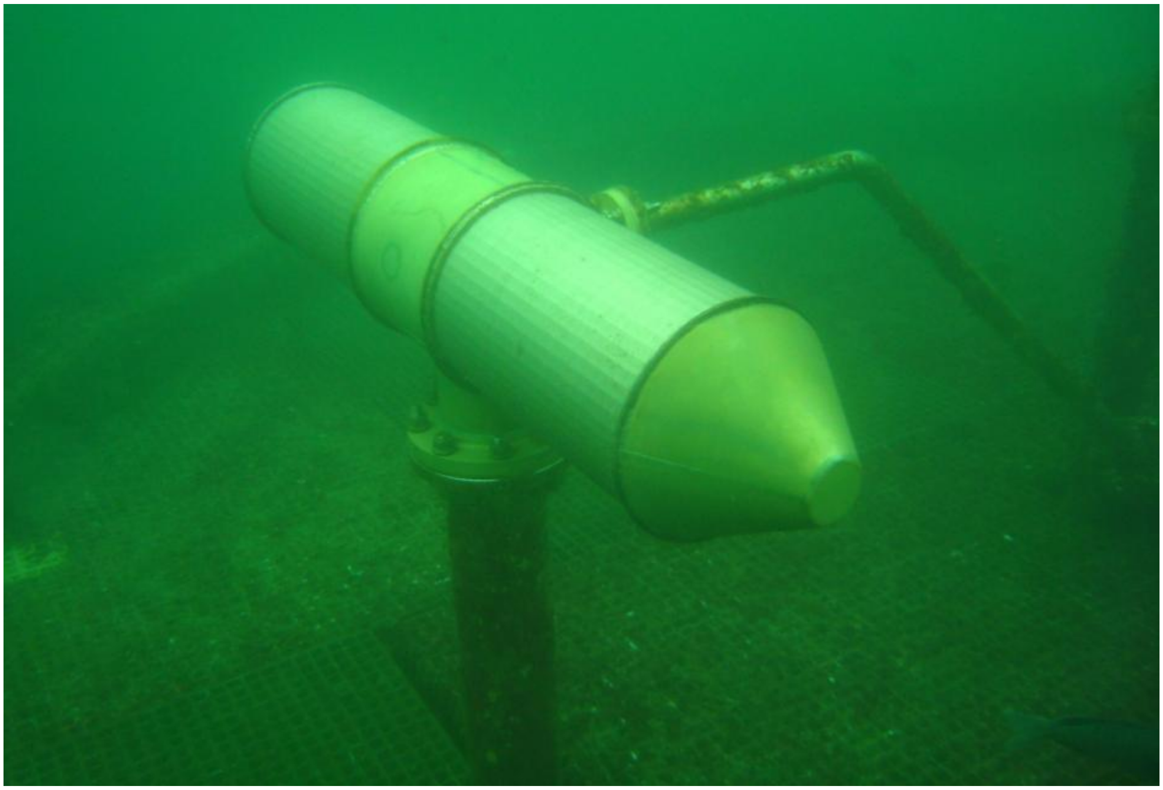


Figure 4-1. Wedgewire screen intake module installed on abandoned offshore discharge structure.

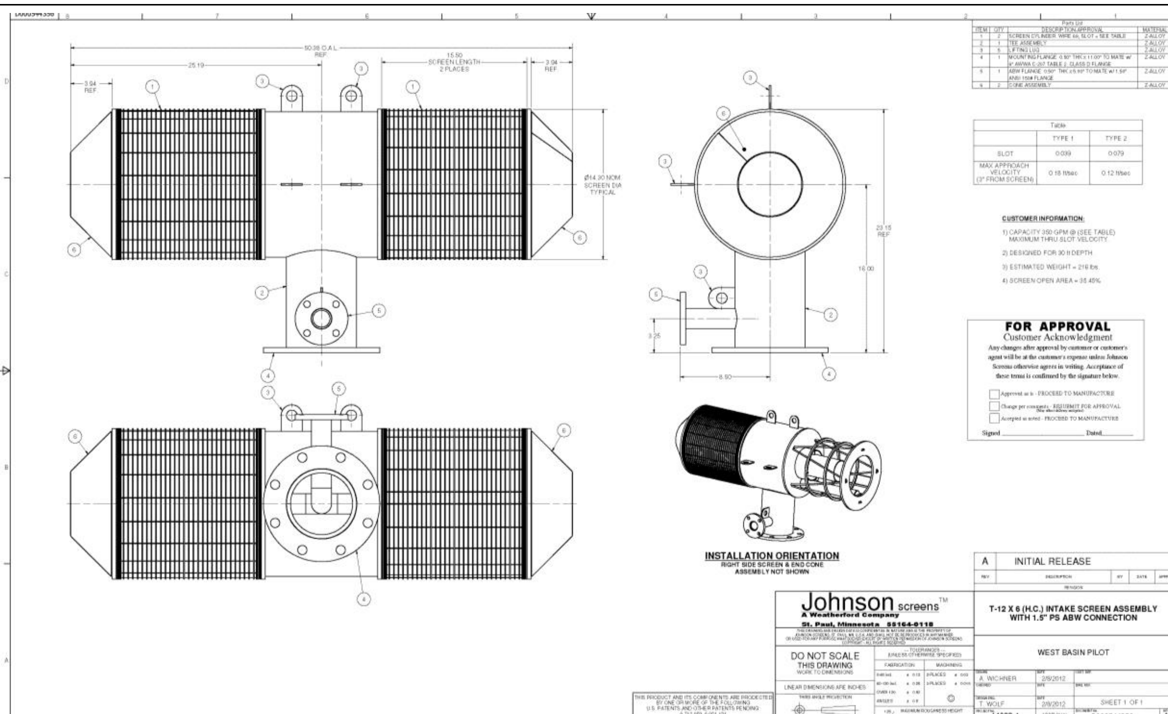


Figure 4-2. Diagram showing detail of wedgewire screen intake module.

The study of screen efficiency was designed to include two sampling efforts. The first sampling effort involved sampling through the two WWS modules for comparison with samples collected through an intake with a larger mesh screen that would only exclude larger non-entrainable organisms (unscreened intake). This sampling was done to determine the effectiveness of the WWS screens at reducing entrainment of later stage fish and invertebrate larvae. This component of the study involved collection of samples through the year in conjunction with the sampling for the impact assessment. The second sampling effort was designed to reduce the variability inherent in data collected throughout the year by collecting a large number of samples (n~90) over a short time period (three days) during a time of high larval fish abundance. The samples were to be collected simultaneously from the unscreened intake and, alternately, from each WWS screened intake. The 1 mm WWS module was to be tested during the first three-day sampling event and the 2 mm WWS module would be tested during a second sampling event. The data collected were expected to provide an adequate sample size for a statistically valid estimate of the percentage in entrainment reduction specific to each WWS screen slot width.

The objective of the screen entrainment comparison study was to determine the effectiveness of the WWS screens of different slot widths at reducing the entrainment of later stage fish larvae, fish eggs, and Cancer crab megalpoae. Differences detected in the number and sizes of the organisms entrained between the unscreened intake and two intakes with WWS screens was expected to provide data for the calculation of entrainment reductions associated with the two WWS slot widths. A variety of issues, both natural and mechanical, occurred during the study period that limited the usefulness of the data for this intended purpose. The problems with the first sampling effort included issues resulting from the installation and maintenance of the WWS modules, and concerns regarding the integrity of the samples due to biofouling within the intake lines and the need to sample from a boat to provide the unscreened samples for comparison. For example, the samples collected from October 2011 through February 2012 could not be used for the analysis due to bias introduced by predation on entrained organisms from biofouling organisms within the WWS screen modules. These problems resulted in the collection of only a limited number of samples that did not provide the necessary sample size for detecting any differences between concentrations of larvae from the WWS modules and the unscreened intake.

In addition to a large sample size, the ability to detect fine-scale differences in entrainment between the screens required high densities of the target organisms within the source water and a large number of entrained organisms. Data collected during the study period showed suitably high densities of fish eggs and Cancer crab megalops but very low densities of fish larvae. Only 136 fish larvae were collected during the sampling. Therefore, the presence of fish larvae in the samples almost represented random events resulting in data which could not be analyzed. Fish eggs and invertebrate larvae were more abundant and showed patterns which were consistent with the expectation of reduced entrainment through the 1 and 2 mm WWS modules. As a result of the concerns regarding the integrity of all the samples, none of these data, including the data for fish eggs and invertebrate larvae, were analyzed for this report, but the data are provided in the appendices.

The potential for high variability in the data from the first sampling effort was anticipated during the design of the study since it was expected that variation in abundances through the year may make it difficult to detect any differences due in entrainment through the WWS modules. Therefore, the design included the more intensive sampling effort described above. This sampling effort was planned to occur during the spring and summer of 2012 following the regular monthly sampling when fish larvae were expected to be in highest abundance. Monitoring of larval fish abundances was conducted from the spring of 2012 through June of 2013, but suitable conditions in the marine waters off King Harbor/Manhattan Beach for conducting the study were not observed and as a result the study was curtailed. A sampling effort was initiated during the evening hours of 5 July 2012, however, the effort was terminated because too few larval fish were collected in the first few samples to continue sampling.

Due to the problems with the sampling and data collected for this component of the study, the results are not presented. All of the data collected are presented in Appendix D, but should not be used due to the sampling issues described above.

4.2 Length Specific Entrainment Analysis

This section presents the second component of the WWS effectiveness studies, which is an analysis that provided estimates of the expected effectiveness of WWS at reducing entrainment of fish larvae via exclusion. The estimates are based on the length of the larvae and the corresponding width and depth of the head capsule (**Figure 4-3**). Although most of the body parts of fish larvae are soft and easily compressible at the early stages of development when they are susceptible to entrainment, the head capsule has harder cartilage and bone that is not compressible. In theory, individuals with head capsules larger than the screen mesh size would be excluded from entrainment, even if the approach vector was perpendicular (head-on) to the screen. Therefore, the smallest dimension (width or depth) of the head capsule can be used to predict the lengths of the range of larval fish sizes that could pass through a rectangular mesh or WWS slot opening.

A study on the relationship between head capsule (head depth [height] and width) and notochord length (NL) of larval fishes was conducted by Tenera (2011). The larvae used in the study were collected during sampling near the intakes of eight power plants in central and southern California (**Table 4-1**). Larval length and head capsule dimension measurements were compiled for 15 abundantly collected taxa, which represented a small subset of the larval fishes collected during larval entrainment studies at the eight plants, but all of the fish taxa selected for detailed analysis in this study. The entrainment studies represented in the analysis were among the studies that contributed to the data provided in Appendices E and F of the State Policy Final Substitute Environmental Document (California plant once-through cooling entrainment and impingement estimates). The Tenera (2011) study used a nonlinear allometric regression analysis where head capsule dimension is a power function of notochord length to estimate the proportion of larvae of each length entrained. This type of regression model is used to describe proportional changes in body shape with growth (e.g., Fuiman 1983, Gisbert et al. 2002, and Pena and Dumas 2009).

Screen entrainment probabilities were calculated for six screen slot widths using larval size class data for each of the selected taxa up to a size where the larvae are no longer considered to be subject to entrainment. The probabilities across the size range of entrainable larvae for a taxon were also used to assess reductions in entrainment when using a particular screen dimension and the overall effects on population mortality. The estimated reductions in population mortality would need to be adjusted for the composition and size structure of the fish larvae for a specific location and sample year, but otherwise provide an estimate of population-level mortality identical to an adult equivalent model using constant growth and survival rates to the length or age when the fish are no longer subject to entrainment.

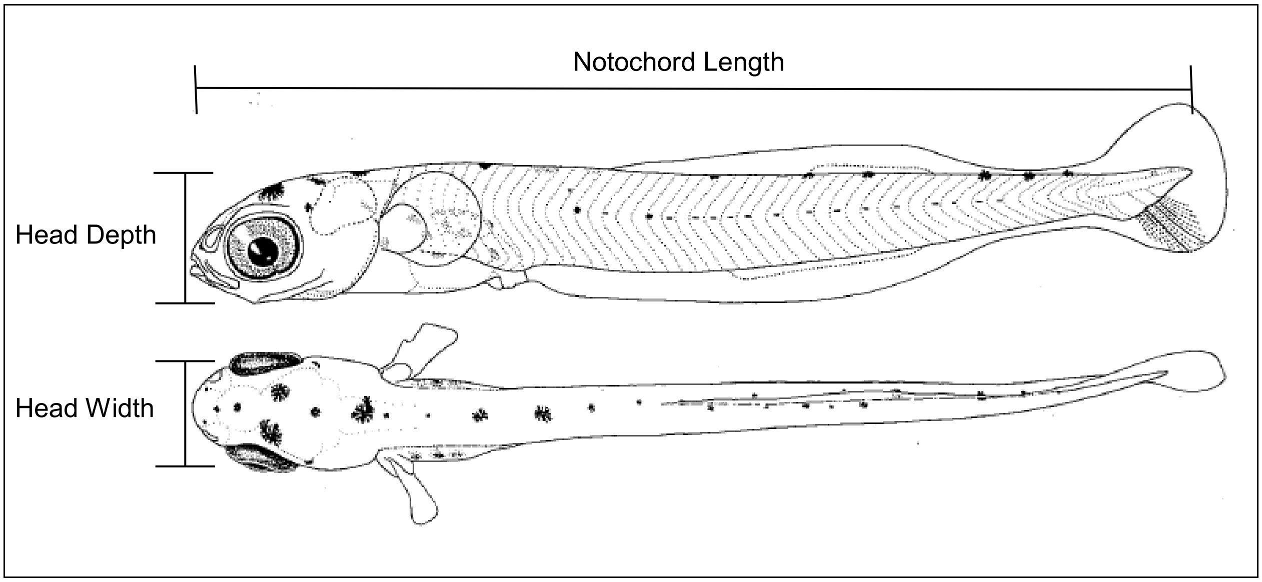


Figure 4-3. Illustration of the measurement locations for notochord length and head depth (height) and width of a preflexion stage larval fish. Larval fish is a jacksmelt from Moser (1996).

Table 4-1. Location of power plants and the years during which larval fish were collected.

Power Plant	Owner (present)	Intake Latitude	Intake Longitude	Sample Period
Moss Landing	Dynegy Inc.	36° 48.292' N	121° 47.130' W	1999–2000
Diablo Canyon	Pacific Gas and Electric Co.	35° 12.456' N	120° 51.407' W	1996–1999
Scattergood	LADWP	33° 54.985' N	118° 26.106' W	2006–2007
El Segundo	El Segundo Power, LLC	33° 54.433' N	118° 26.031' W	2006–2007
Redondo	AES Southland, LLC	33° 50.409' N	118° 23.718' W	2006–2007
Haynes	LADWP	33° 45.121' N	118° 06.556' W	2006–2007
Harbor	LADWP	33° 45.932' N	118° 15.790' W	2006–2007
South Bay	Dynegy Inc.	32° 36.869' N	117° 05.942' W	2001–2003

4.2.1 Methods

The relationships between overall notochord length of the larvae and the parameters of head capsule width and depth based on the allometric regression models in Tenera (2011) were used in this report to estimate the length specific probabilities of entrainment for some of the taxa of larval fishes that were collected in high abundance during sampling for the West Basin DDF intake study and the entrainment studies at coastal power plants in central and southern California (**Table 4-1**). The estimated probabilities were calculated for WWS slot openings of 0.5 mm (0.02 in), 1 mm (0.04 in), and 2 mm (0.08 in) for the five most abundantly collected taxa during the West Basin IEA study. The species groups used for the analysis were kelpfishes (*Gibbonsia* spp.), combfish blennies (*Hysonblennius* spp.), anchovies (Engraulidae), croakers (Sciaenidae) and silversides (Atherinopsidae).

Probability estimates were calculated over a size range that approximately corresponded to the range of the lengths of larvae that would be potentially entrainable. The minimum lengths for the taxa were based on the smallest larvae measured from the studies (**Table 4-2**). The maximum was set at either 20 or 25 mm (0.79 or 0.98 in) depending on the fish taxon. Fishes larger than 20–25 mm (0.79 or 0.98 in) generally have characteristics (e.g., presence of head and opercular spines) that would likely bias entrainment probabilities based only on larval head capsule measurements. Fishes at this size also have swimming abilities that allow them to potentially avoid entrainment, especially at reduced intake velocities that could be used at plants retrofitting with fine mesh or WWS. Percent survival or the viability of larvae following possible screen impingement are not assessed.

The parameters for the regression models of the relationship between larval length and head capsule dimensions from Tenera (2011) for the five taxa are shown in **Table 4-2**. The number of specimens included per taxa ranged from a high of 282 for anchovies to a low of 42 for combtooth blennies. Although the numbers measured were roughly proportional to the relative abundances of the target taxa in the selected entrainment samples, the range of lengths shown in **Table 4-2** do not necessarily correspond to the complete size range collected during the studies.

A summary of the data on all of the larvae measured from studies conducted in southern California (excluding Diablo Canyon Power Plant) including data from the San Onofre Generating Station are shown in **Table 4-3**. All of the taxa in Tenera (2011), using the data summarized in **Table 4-2,** were first analyzed with a single model using all of the measured individuals. However, kelpfishes (*Gibbonsia* spp.), anchovies (Engraulidae), and silversides (Atherinopsidae) showed a discontinuity in the growth relationship at lengths that corresponded approximately to the larval transformation phase, or slightly smaller in the case of anchovies, when the larvae start developing into a juvenile and might begin to take on some adult characteristics (Moser 1996). Separate regression models were presented in Tenera (2011) for the two different stages of larval development for these three taxa. For example, separate models were developed for silverside larvae smaller than 15 mm (0.59 in.) notochord length (NL), and those larger than that size, which approximately corresponds to the length of transformation. The same approach was used by Gisbert et al. (2002) and Pena and Dumas (2009) in their analyses of allometric growth patterns in California halibut and spotted sand bass larvae, respectively.

Screen entrainment probabilities were calculated for the three WWS slot widths (0.5 mm [0.02 in], 1 mm [0.04 in], 2 mm [0.08 in]) using estimates of the variability around the allometric regressions. The variability corresponding to the allometric regression estimates were calculated by using the standard errors of the two parameters of the regression (**Table 4-4**). To describe the effects of this variation on head capsule dimensions, 10,000 estimates of head width and head depth for each millimeter size class of notochord length (from a minimum up to a maximum length determined for the taxon) were computer-generated using the estimated standard errors for each regression parameter. Errors were assumed to be normally distributed. For each set of 10,000 values, a length-specific probability of entrainment was calculated for head widths and depths. The probability of entrainment for each notochord length was determined as the larger value of either the head width entrainment probability or the head depth entrainment probability. The 10,000 estimates were calculated 1,000 times using randomly selected values within ±0.5 mm (0.02 in) of each length. The average proportion and standard error were calculated from the 1,000 estimates calculated for each 1 mm (0.04 in) length increment.

The probabilities across the size range of entrainable larvae were used to assess the effects of the reductions in entrainment on population mortality. The following two assumptions were necessary to calculate the reductions in population mortality: 1) linear growth over time, and 2) constant exponential natural mortality. These assumptions were reasonable because they are only being applied to the short period of time that the larvae are subject to entrainment, which may only be a few days for fishes that are only subject to entrainment over a narrow size range, but would likely never extend beyond one or two months. By assuming linear growth, length becomes directly proportional to age. As a larval cohort progresses through consecutive length classes it follows an exponential decrease in numbers over time due to natural mortality. Under these assumptions, each length (or age) would result in an identical number of adult equivalents or fishes at the length (or age) where they are not subject to entrainment. As a result, the effects on the population due to the reduction for each screen mesh dimension can be made by summing the length-specific entrainment probabilities, and dividing by the number of probability estimates. The subtraction of this value from one determines the reduction of mortality for the total cohort of larvae that would survive to the length or age when they are no longer subject to entrainment. The population-level estimates for these five taxa were adjusted to the range of lengths collected from studies at ocean intakes throughout southern California (**Table 4-3**).

Actual entrainment estimates for each length were also calculated using the probabilities based on the head capsule measurements. This additional analysis required data on the size composition of the larvae for these taxa. The results from this study did not provide adequate data across all the size classes for this analysis so data for these five taxa from previous intake studies in southern California (**Table 4-3**) were used to provide a more complete data set for the analysis. The estimates were only calculated for an intake volume of 170,722 m^3 (45.1 mgd), but the percentage reductions of entrainment would apply to any intake volume assuming that the size composition of the population subject to entrainment is the same as the results presented for each taxa.

Table 4-2. Summary statistics on length and head capsule dimensions for the fishes from each taxon used in the head capsule analysis.

Common Name	N	Length (mm) Mean	Max	Min	Median	Std.Dev.	Head Depth (mm) Mean	Max	Min	Median	Std.Dev.	Head Width (mm) Mean	Max	Min	Median	Std.Dev.
kelpfishes	75	10.40	25.91	3.46	10.22	4.93	1.18	4.36	0.47	1.03	0.68	1.09	3.23	0.45	0.98	0.51
combtooth blennies	42	2.54	4.31	1.87	2.25	0.66	0.49	1.10	0.35	0.44	0.14	0.42	0.89	0.32	0.38	0.12
anchovies	282	14.10	31.01	1.51	14.23	8.20	1.15	3.49	0.15	0.95	0.82	1.16	3.10	0.19	1.12	0.67
croakers	167	5.18	14.87	1.23	4.18	3.59	1.29	4.31	0.15	0.89	1.03	0.94	3.21	0.20	0.73	0.69
silversides	221	12.28	31.07	3.63	11.01	5.77	1.54	4.37	0.34	1.14	0.95	1.42	3.70	0.35	1.15	0.71

Table 4-3. Summary statistics on lengths of larvae collected from intake studies in southern California (excluding Diablo Canyon Power Plant) listed in **Table 4-1** including data from the study at the San Onofre Nuclear Generating Station.

Common Name	N	Length (mm) Mean	Std.Dev	Max	Min	Median
kelpfishes	817	6.14	2.04	20.63	2.46	5.57
combtooth blennies	3,269	2.35	0.47	13.07	1.56	2.28
anchovies	2,427	10.04	6.91	31.31	1.05	9.63
croakers	2,240	2.89	1.51	14.00	0.92	2.37
silversides	1,933	9.09	2.66	24.40	2.50	8.50

4.2.2 Results

The statistics and parameters resulting from the allometric regressions are shown in **Table 4-4**, and dispersion plots of the data for each taxon are shown in Appendix E. The results for kelpfishes (Appendix E Figure 1), anchovies (Appendix E Figure 8), and silversides (Appendix E Figure 13) showed discontinuities in the relationship that corresponded approximately to the larval transformation phase for kelpfishes and silversides (Moser 1996). Moser (1996) gives transformation sizes of 15 mm (0.59 in.) for silversides and 21 mm (0.83 in.) for kelpfishes. Anchovies (Engraulids) appear to have a growth inflection at about 19 mm (0.75 in.) which is less than the reported northern anchovy transformation size (Moser 1996). Separate calculations for both growth phases (smaller and larger sized groups) were calculated for those taxa and are shown in Appendix E following the model for each taxa using all the lengths.

Parameters of allometric regressions and their standard errors describing head capsule dimensions as a function of notochord length were used to predict the proportion of the five selected larval taxa that could be susceptible to entrainment through the three WWS slot sizes chosen for the analysis. **Tables 4-5** to **4-14** show the estimated length specific entrainment probabilities for the larval taxa as a function of slot dimension. Tables of entrainment probabilities for larval kelpfishes less than 21 mm (0.83 in), anchovies less than and greater than

19 mm (0.75 in), and silversides less than and greater than 15 mm (0.59 in) follow the tables that present the result based on all the length data for those taxa.

The results of the analyses by length presented in **Table 4-5** to **4-14** were used to calculate actual reductions in larval entrainment for each length class subject to entrainment (**Tables 4-15** to **4-29**). The results show that the reductions in entrainment for the 2 mm (0.08 in.) slot width were very small for all five taxa with the largest reduction being estimated for northern anchovy at 3.1 percent (**Table 4-17**). The levels of entrainment reductions were small for all three slot widths for combtooth blennies, which, as the length frequency data show, are only subject to entrainment over a very limited size range (**Table 4-16**). The results show the importance of the size composition of the population in considering the function of the screens at reducing the effects of entrainment.

4.2.2.1 Extrapolated Population-Level Efficiency

The probabilities in **Tables 4-5** to **4-14** were used to assess the effects of reducing the entrainment of larvae on population-level mortality for the three WWS slot widths after omitting the size-specific estimates for kelpfishes, anchovies, and silversides (**Table 4-20**). The population-level mortality reductions were calculated to a length where data from **Table 4-3** indicate that the larvae were no longer vulnerable to entrainment. This was maintained at 25 mm (0.98 in.) for kelpfishes, anchovies, and silversides, and adjusted to 15 mm (0.59 in.) for combtooth blennies and white croakers based on the data shown in **Table 4-3**. The estimates of population-level mortality are generally much larger than the estimates of entrainment reduction shown in **Table 19-19**. Exceptions occur for taxa that continue to be entrained over a large number of length classes such as the results for the 2 mm (0.08 in.) slot width for northern anchovy which only increased from 3.1 to 5.5 (**Table 4-17**).

Table 4-4. Allometric regression parameter statistics ($y = ax^b$) and standard errors describing the sample composition of each taxon used in the analysis, where x = notochord length (mm). All stages (sizes) were used unless noted.

Taxon	Y Variable: Head Depth (Height)				Stage	Y Variable: Head Width			
	a	SE(a)	b	SE(b)		a	SE(a)	b	SE(b)
kelpfishes	0.0541	0.0079	1.2856	0.0533	all	0.0998	0.0091	1.0137	0.0344
	0.1175	0.0132	0.9680	0.0441	≤ 21 mm	0.1492	0.0103	0.8436	0.0274
combtooth blennies	0.1833	0.0160	1.0427	0.0814		0.1777	0.0166	0.9231	0.0884
anchovies	0.0215	0.0023	1.4524	0.0342	all	0.0776	0.0046	1.0167	0.0195
	0.0964	0.0062	0.8739	0.0247	≤ 19 mm	0.1202	0.0054	0.8461	0.0173
	0.0104	0.0035	1.6831	0.1037	≥ 19 mm	0.0216	0.0054	1.4184	0.0784
croakers	0.2094	0.0129	1.0979	0.0276		0.1894	0.0148	0.9783	0.0356
silversides	0.0588	0.0035	1.2880	0.0206	all	0.1006	0.0038	1.0531	0.0135
	0.0908	0.0060	1.0730	0.0280	≤ 15 mm	0.1328	0.0073	0.9219	0.0236
	0.1400	0.0220	1.0089	0.0520	≥ 15 mm	0.1394	0.0171	0.9490	0.0406

Table 4-5. Estimated proportions (standard error) of kelpfish larvae entrained through three different size screen slot openings based on head capsule allometric regressions on notochord lengths to 25 mm.

Length (mm)	Screen Slot Dimension (mm)		
	0.5	1	2
2	1 (0)	1 (0)	1 (0)
3	1 (0)	1 (0)	1 (0)
4	0.995 (0.007)	1 (0)	1 (0)
5	0.817 (0.117)	1 (0)	1 (0)
6	0.354 (0.118)	1 (0)	1 (0)
7	0.087 (0.038)	0.998 (0.002)	1 (0)
8	0.016 (0.008)	0.954 (0.030)	1 (0)
9	0.003 (0.001)	0.747 (0.087)	1 (0)
10	0.001 (0.000)	0.426 (0.081)	1 (0)
11	0 (0)	0.213 (0.046)	1 (0)
12	0 (0)	0.098 (0.024)	1 (0)
13	0 (0)	0.041 (0.010)	0.999 (0.000)
14	0 (0)	0.017 (0.004)	0.995 (0.003)
15	0 (0)	0.007 (0.002)	0.976 (0.009)
16	0 (0)	0.003 (0.001)	0.927 (0.021)
17	0 (0)	0.001 (0.000)	0.831 (0.034)
18	0 (0)	0.001 (0.000)	0.698 (0.043)
19	0 (0)	0 (0)	0.543 (0.046)
20	0 (0)	0 (0)	0.397 (0.040)
21	0 (0)	0 (0)	0.271 (0.032)
22	0 (0)	0 (0)	0.175 (0.023)
23	0 (0)	0 (0)	0.108 (0.016)
24	0 (0)	0 (0)	0.065 (0.010)
25	0 (0)	0 (0)	0.037 (0.006)

Table 4-6. Estimated proportions (standard error) of kelpfish larvae entrained through three different size screen slot openings based on head capsule allometric regressions on notochord lengths less than or equal to 21 mm.

Length (mm)	Screen Slot Dimension (mm)		
	0.5	1	2
2	1 (0)	1 (0)	1 (0)
3	0.998 (0.004)	1 (0)	1 (0)
4	0.781 (0.155)	1 (0)	1 (0)
5	0.237 (0.117)	1 (0)	1 (0)
6	0.032 (0.020)	1 (0)	1 (0)
7	0.004 (0.002)	0.998 (0.002)	1 (0)
8	0 (0)	0.946 (0.037)	1 (0)
9	0 (0)	0.708 (0.100)	1 (0)
10	0 (0)	0.351 (0.094)	1 (0)
11	0 (0)	0.136 (0.033)	1 (0)
12	0 (0)	0.058 (0.015)	1 (0)
13	0 (0)	0.022 (0.006)	1 (0)
14	0 (0)	0.009 (0.002)	1 (0)
15	0 (0)	0.003 (0.001)	0.999 (<.001)
16	0 (0)	0.001 (<.001)	0.994 (0.003)
17	0 (0)	0.001 (<.001)	0.977 (0.008)
18	0 (0)	0 (0)	0.935 (0.017)
19	0 (0)	0 (0)	0.858 (0.028)
20	0 (0)	0 (0)	0.743 (0.037)
21	0 (0)	0 (0)	0.605 (0.041)

Table 4-7. Estimated proportions (standard error) of combtooth blenny larvae entrained through three different size screen slot openings based on head capsule allometric regressions on notochord lengths to 20 mm.

Length (mm)	Screen Slot Dimension (mm)		
	0.5	1	2
2	0.993 (0.013)	1 (0)	1 (0)
3	0.573 (0.233)	1 (0)	1 (0)
4	0.076 (0.051)	0.997 (0.004)	1 (0)
5	0.007 (0.004)	0.917 (0.050)	1 (0)
6	0.001 (0.000)	0.659 (0.088)	1 (0)
7	0 (0)	0.381 (0.071)	0.999 (<.001)
8	0 (0)	0.183 (0.040)	0.992 (0.004)
9	0 (0)	0.087 (0.020)	0.966 (0.012)
10	0 (0)	0.041 (0.009)	0.908 (0.022)
11	0 (0)	0.020 (0.004)	0.817 (0.029)
12	0 (0)	0.010 (0.002)	0.707 (0.034)
13	0 (0)	0.005 (0.001)	0.593 (0.033)
14	0 (0)	0.003 (<.001)	0.481 (0.031)
15	0 (0)	0.002 (<.001)	0.385 (0.026)
16	0 (0)	0.001 (<.001)	0.304 (0.022)
17	0 (0)	0.001 (<.001)	0.239 (0.017)
18	0 (0)	0 (0)	0.186 (0.014)
19	0 (0)	0 (0)	0.145 (0.011)
20	0 (0)	0 (0)	0.113 (0.009)

Table 4-8. Estimated proportions (standard error) of anchovy larvae entrained through three different size screen slot openings based on head capsule allometric regressions on notochord lengths to 25 mm.

Length (mm)	Screen Slot Dimension (mm)		
	0.5	1	2
1	1 (0)	1 (0)	1 (0)
2	1 (0)	1 (0)	1 (0)
3	1 (0)	1 (0)	1 (0)
4	1 (0)	1 (0)	1 (0)
5	1 (0)	1 (0)	1 (0)
6	1 (0)	1 (0)	1 (0)
7	0.993 (0.009)	1 (0)	1 (0)
8	0.830 (0.098)	1 (0)	1 (0)
9	0.382 (0.131)	1 (0)	1 (0)
10	0.093 (0.045)	1 (0)	1 (0)
11	0.013 (0.007)	0.997 (0.003)	1 (0)
12	0.001 (0.001)	0.956 (0.024)	1 (0)
13	0 (0)	0.803 (0.067)	1 (0)
14	0 (0)	0.530 (0.084)	1 (0)
15	0 (0)	0.268 (0.061)	1 (0)
16	0 (0)	0.109 (0.031)	1 (0)
17	0 (0)	0.037 (0.012)	1 (0)
18	0 (0)	0.011 (0.004)	1 (0)
19	0 (0)	0.003 (0.001)	0.999 (<.001)
20	0 (0)	0.001 (<.001)	0.993 (0.004)
21	0 (0)	0 (0)	0.968 (0.012)
22	0 (0)	0 (0)	0.900 (0.028)
23	0 (0)	0 (0)	0.765 (0.047)
24	0 (0)	0 (0)	0.590 (0.057)
25	0 (0)	0 (0)	0.401 (0.052)

Table 4-9. Estimated proportions (standard error) of anchovy larvae entrained through three different size screen slot openings based on head capsule allometric regressions on notochord lengths less than or equal to 19 mm.

Length (mm)	Screen Slot Dimension (mm)		
	0.5	1	2
1	1 (0)	1 (0)	1 (0)
2	1 (0)	1 (0)	1 (0)
3	1 (0)	1 (0)	1 (0)
4	1 (0)	1 (0)	1 (0)
5	0.998 (0.004)	1 (0)	1 (0)
6	0.821 (0.128)	1 (0)	1 (0)
7	0.277 (0.139)	1 (0)	1 (0)
8	0.030 (0.023)	1 (0)	1 (0)
9	0.002 (0.001)	1 (0)	1 (0)
10	0 (0)	1 (0)	1 (0)
11	0 (0)	0.997 (0.002)	1 (0)
12	0 (0)	0.969 (0.017)	1 (0)
13	0 (0)	0.861 (0.050)	1 (0)
14	0 (0)	0.642 (0.073)	1 (0)
15	0 (0)	0.392 (0.068)	1 (0)
16	0 (0)	0.196 (0.045)	1 (0)
17	0 (0)	0.084 (0.023)	1 (0)
18	0 (0)	0.031 (0.010)	1 (0)
19	0 (0)	0.010 (0.003)	1 (0)

Table 4-10. Estimated proportions (standard error) of anchovy larvae entrained through three different size screen slot openings based on head capsule allometric regressions on notochord lengths between 19 and 25 mm.

Length (mm)	Screen Slot Dimension (mm)		
	0.5	1	2
19	0.035 (0.003)	0.230 (0.014)	0.868 (0.016)
20	0.029 (0.002)	0.186 (0.012)	0.810 (0.018)
21	0.024 (0.002)	0.151 (0.009)	0.746 (0.020)
22	0.020 (0.002)	0.124 (0.008)	0.679 (0.021)
23	0.017 (0.001)	0.102 (0.006)	0.609 (0.020)
24	0.015 (0.001)	0.085 (0.005)	0.541 (0.020)
25	0.013 (0.001)	0.071 (0.005)	0.477 (0.019)

Table 4-11. Estimated proportions (standard error) of croaker larvae entrained through three different size screen slot openings based on head capsule allometric regressions on notochord lengths to 20 mm.

Length (mm)	Screen Slot Dimension (mm)		
	0.5	1	2
1	1 (0)	1 (0)	1 (0)
2	0.980 (0.040)	1 (0)	1 (0)
3	0.230 (0.243)	1 (0)	1 (0)
4	0.001 (0.001)	0.998 (0.004)	1 (0)
5	0 (0)	0.780 (0.155)	1 (0)
6	0 (0)	0.224 (0.123)	1 (0)
7	0 (0)	0.019 (0.015)	1 (0)
8	0 (0)	0.001 (0.001)	0.998 (0.002)
9	0 (0)	0 (0)	0.969 (0.020)
10	0 (0)	0 (0)	0.826 (0.067)
11	0 (0)	0 (0)	0.546 (0.086)
12	0 (0)	0 (0)	0.280 (0.064)
13	0 (0)	0 (0)	0.113 (0.032)
14	0 (0)	0 (0)	0.039 (0.013)
15	0 (0)	0 (0)	0.012 (0.004)
16	0 (0)	0 (0)	0.003 (0.001)
17	0 (0)	0 (0)	0.001 (<.001)
18	0 (0)	0 (0)	0 (0)
19	0 (0)	0 (0)	0 (0)
20	0 (0)	0 (0)	0 (0)

Table 4-12. Estimated proportions (standard error) of silverside larvae entrained through three different size screen slot openings based on head capsule allometric regressions on notochord lengths to 25 mm.

Length (mm)	Screen Slot Dimension (mm)		
	0.5	1	2
2	1 (0)	1 (0)	1 (0)
3	1 (0)	1 (0)	1 (0)
4	1 (0)	1 (0)	1 (0)
5	0.748 (0.251)	1 (0)	1 (0)
6	0.043 (0.058)	1 (0)	1 (0)
7	0 (0)	1 (0)	1 (0)
8	0 (0)	0.969 (0.040)	1 (0)
9	0 (0)	0.528 (0.202)	1 (0)
10	0 (0)	0.065 (0.055)	1 (0)
11	0 (0)	0.002 (0.002)	1 (0)
12	0 (0)	0 (0)	1 (0)
13	0 (0)	0 (0)	1 (0)
14	0 (0)	0 (0)	1 (0)
15	0 (0)	0 (0)	0.994 (0.006)
16	0 (0)	0 (0)	0.896 (0.062)
17	0 (0)	0 (0)	0.545 (0.127)
18	0 (0)	0 (0)	0.175 (0.076)
19	0 (0)	0 (0)	0.027 (0.017)
20	0 (0)	0 (0)	0.002 (0.002)
21	0 (0)	0 (0)	0 (0)
22	0 (0)	0 (0)	0 (0)
23	0 (0)	0 (0)	0 (0)
24	0 (0)	0 (0)	0 (0)
25	0 (0)	0 (0)	0 (0)

Table 4-13. Estimated proportions (standard error) of silverside larvae entrained through three different size screen slot openings based on head capsule allometric regressions on notochord lengths from 2 to 15 mm.

Length (mm)	Screen Slot Dimension (mm)		
	0.5	1	2
2	1 (0)	1 (0)	1 (0)
3	1 (0)	1 (0)	1 (0)
4	0.983 (0.029)	1 (0)	1 (0)
5	0.435 (0.259)	1 (0)	1 (0)
6	0.015 (0.018)	1 (0)	1 (0)
7	0 (0)	1 (0)	1 (0)
8	0 (0)	0.963 (0.033)	1 (0)
9	0 (0)	0.679 (0.132)	1 (0)
10	0 (0)	0.238 (0.099)	1 (0)
11	0 (0)	0.042 (0.024)	1 (0)
12	0 (0)	0.005 (0.003)	1 (0)
13	0 (0)	0 (0)	1 (0)
14	0 (0)	0 (0)	1 (0)
15	0 (0)	0 (0)	0.995 (0.003)

Table 4-14. Estimated proportions (standard error) of silverside larvae entrained through three different size screen slot openings based on head capsule allometric regressions on notochord lengths from 15 to 25 mm.

Length (mm)	Screen Slot Dimension (mm)		
	0.5	1	2
15	0 (0)	0.001 (<.001)	0.724 (0.039)
16	0 (0)	0.001 (<.001)	0.585 (0.041)
17	0 (0)	0 (0)	0.449 (0.038)
18	0 (0)	0 (0)	0.330 (0.031)
19	0 (0)	0 (0)	0.235 (0.025)
20	0 (0)	0 (0)	0.161 (0.018)
21	0 (0)	0 (0)	0.110 (0.013)
22	0 (0)	0 (0)	0.073 (0.009)
23	0 (0)	0 (0)	0.048 (0.006)
24	0 (0)	0 (0)	0.031 (0.004)
25	0 (0)	0 (0)	0.020 (0.003)

Table 4-15. Estimated annual entrainment by length for kelpfishes based on intake volume of 170,722 m3 (45.1 mgd) and total percentage reduction from annual estimated entrainment by length class based on probabilities in **Table 4-9**.

Length (mm)	Length			Entrainment by Screen Slot Dimension (mm)		
	Count	Percent	Entrainment	0.5	1	2
2	1	0.12	412	412	412	412
3	5	0.61	2,059	2,059	2,059	2,059
4	136	16.65	56,014	55,734	56,014	56,014
5	250	30.60	102,967	84,124	102,967	102,967
6	152	18.60	62,604	22,162	62,604	62,604
7	107	13.10	44,070	3,834	43,982	44,070
8	68	8.32	28,007	448	26,719	28,007
9	40	4.90	16,475	49	12,307	16,475
10	23	2.82	9,473	9	4,036	9,473
11	18	2.20	7,414	0	1,579	7,414
12	6	0.73	2,471	0	242	2,471
13	7	0.86	2,883	0	118	2,880
14	1	0.12	412	0	7	410
15	1	0.12	412	0	3	402
16	0	0.00	0	0	0	0
17	1	0.12	412	0	0	342
18	0	0.00	0	0	0	0
19	0	0.00	0	0	0	0
20	0	0.00	0	0	0	0
21	1	0.12	412	0	0	112
Total			336,496	168,832	313,048	336,111
Total Percentage Reduction				49.83%	6.97%	0.11%

Table 4-16. Estimated annual entrainment by length for combtooth blennies based on intake volume of 170,722 m^3 (45.1 mgd) and total percentage reduction from annual estimated entrainment by length class based on probabilities in **Table 4-7**.

Length (mm)	Length			Entrainment by Screen Slot Dimension (mm)		
	Count	Percent	Entrainment	0.5	1	2
2	2,470	75.56	503,932	500,404	503,932	503,932
3	756	23.13	154,239	88,379	154,239	154,239
4	34	1.04	6,937	527	6,916	6,937
5	5	0.15	1,020	7	936	1,020
6	1	0.03	204	0	134	204
7	0	0.00	0	0	0	0
8	0	0.00	0	0	0	0
9	0	0.00	0	0	0	0
10	0	0.00	0	0	0	0
11	1	0.03	204	0	4	167
12	0	0.00	0	0	0	0
13	2	0.06	408	0	2	242
Total			666,945	589,318	666,163	666,741
Total Percentage Reduction				11.64%	0.12%	0.03%

Table 4-17. Estimated annual entrainment by length for anchovies based on intake volume of 170,722 m^3 (45.1 mgd) and total percentage reduction from annual estimated entrainment by length class based on probabilities in **Table 4-8**.

Length (mm)	Length			Entrainment by Screen Slot Dimension (mm)		
	Count	Percent	Entrainment	0.5	1	2
1	6	0.25	1,470	1,470	1,470	1,470
2	285	11.77	69,824	69,824	69,824	69,824
3	440	18.17	107,798	107,798	107,798	107,798
4	108	4.46	26,459	26,459	26,459	26,459
5	69	2.85	16,905	16,905	16,905	16,905
6	35	1.45	8,575	8,575	8,575	8,575
7	58	2.40	14,210	14,110	14,210	14,210
8	98	4.05	24,010	19,928	24,010	24,010
9	101	4.17	24,744	9,452	24,744	24,744
10	114	4.71	27,929	2,597	27,929	27,929
11	118	4.87	28,909	376	28,823	28,909
12	122	5.04	29,889	30	28,574	29,889
13	114	4.71	27,929	0	22,427	27,929
14	104	4.30	25,479	0	13,504	25,479
15	85	3.51	20,825	0	5,581	20,825
16	108	4.46	26,459	0	2,884	26,459
17	89	3.68	21,805	0	807	21,805
18	71	2.93	17,395	0	191	17,395
19	57	2.35	13,965	0	42	13,951
20	46	1.90	11,270	0	11	11,191
21	42	1.73	10,290	0	0	9,961
22	40	1.65	9,800	0	0	8,820
23	22	0.91	5,390	0	0	4,123
24	15	0.62	3,675	0	0	2,168
25	21	0.87	5,145	0	0	2,063
26	18	0.74	4,410	0	0	1,054
27	16	0.66	3,920	0	0	510
28	5	0.21	1,225	0	0	77
29	8	0.33	1,960	0	0	55
30	6	0.25	1,470	0	0	18
Total			593,133	277,524	424,768	574,604
Total Percentage Reduction				53.21%	28.39%	3.12%

Table 4-18. Estimated annual entrainment by length for white croakers based on intake volume of 170,722 m^3 (45.1 mgd) and total percentage reduction from annual estimated entrainment by length class based on probabilities in **Table 4-11**.

Length (mm)	Length			Entrainment by Screen Slot Dimension (mm)		
	Count	Percent	Entrainment	0.5	1	2
1	150	6.70	63,319	63,319	63,319	63,319
2	1100	49.11	464,341	455,054	464,341	464,341
3	491	21.92	207,265	47,671	207,265	207,265
4	201	8.97	84,848	85	84,678	84,848
5	118	5.27	49,812	0	38,853	49,812
6	96	4.29	40,524	0	9,077	40,524
7	47	2.10	19,840	0	377	19,840
8	23	1.03	9,709	0	10	9,690
9	7	0.31	2,955	0	0	2,863
10	2	0.09	844	0	0	697
11	1	0.04	422	0	0	230
12	3	0.13	1,266	0	0	355
13	0	0.00	0	0	0	0
14	1	0.04	422	0	0	16
Total			945,566	566,129	867,920	943,800
Total Percentage Reduction				40.13%	8.21%	0.19%

Table 4-19. Estimated annual entrainment by length for silversides based on intake volume of 170,722 m^3 (45.1 mgd) and total percentage reduction from annual estimated entrainment by length class based on probabilities in **Table 4-12**.

Length (mm)	Length			Entrainment by Screen Slot Dimension (mm)		
	Count	Percent	Entrainment	0.5	1	2
3	6	0.31	7,684	7,684	7,684	7,684
4	13	0.67	16,649	16,649	16,649	16,649
5	52	2.69	66,598	49,815	66,598	66,598
6	157	8.12	201,076	8,646	201,076	201,076
7	274	14.17	350,923	0	350,923	350,923
8	464	24.00	594,261	0	575,839	594,261
9	335	17.33	429,047	0	226,537	429,047
10	175	9.05	224,129	0	14,568	224,129
11	128	6.62	163,933	0	328	163,933
12	106	5.48	135,758	0	0	135,758
13	91	4.71	116,547	0	0	116,547
14	49	2.53	62,756	0	0	62,756
15	35	1.81	44,827	0	0	44,558
16	19	0.98	24,333	0	0	21,803
17	9	0.47	11,527	0	0	6,282
18	6	0.31	7,684	0	0	1,345
19	5	0.26	6,405	0	0	173
20	2	0.10	2,562	0	0	5
21	3	0.16	3,842	0	0	0
22	0	0.00	0	0	0	0
23	1	0.05	1,280	0	0	0
24	3	0.16	3,842	0	0	0
Total			2,475,663	82,795	1,460,201	2,443,526
Total Percentage Reduction				96.66%	41.02%	1.30%

Table 4-20. Estimated percentage reductions (two standard errors in parentheses) in mortality (relative to an open intake) to the population surviving past the size where they would be subject to entrainment, based on probabilities of screen entrainment for larvae from five taxonomic categories of fishes.

Taxon	Size Range	Percentage Reduction in Mortality by Slot Opening Width		
		0.5 mm	1 mm	2 mm
Kelpfishes	2–25 mm	82.2 (2.4)	64.6 (2.4)	24.9 (2.4)
combtooth blennies	2–10 mm	88.2 (4.3)	62.1 (4.1)	15.4 (2.7)
Anchovies	2–25 mm	66.8 (2.3)	45.1 (2.3)	5.5 (1.6)
Croakers	1–15 mm	85.3 (3.8)	66.5 (4.0)	28.1 (3.9)
Silversides	2–25 mm	84.2 (2.6)	68.5 (2.5)	34.8 (2.4)
Average Percentage Reduction		**80.0**	**57.2**	**19.0**

4.3 Wedgewire Screen Impingement Study

This section presents the third component of the WWS efficiency studies, which involved collection of data on the potential effects of impingement on marine organisms due to the operation of the intakes for the demonstration and full-scale facilities, and to assess the efficiency of the different WWS screen slot widths at reducing impingement effects. Two WWS modules were used for the study, one constructed with 1.0 mm (0.04 in.) slot widths and one with 2.0 mm (0.08 in.) slot widths. Each WWS module was sized differently to ensure a maximum through-screen velocity of 0.1 m/sec (0.33 ft/sec).

4.3.1 Methods

To study the possibility of fish and other organisms being pulled and held (impinged) on the WWS, housed video cameras were installed on camera stands next to each of the WWS modules. The cameras were positioned to the outboard side and slightly above each intake screen module (**Figure 4-4a** and **Figure 4-4b**) at a distance of approximately 30 cm (11.8 in.) in order to image a surface area of the screen of approximately 30 cm x 22 cm (11.8 in. x 8.7 in.) (600 cm^2 [102.7 in.^{2}]) (**Figure 4-4c** and **Figure 4-4d**). Each camera was oriented to videograph an oblique view of the screen's surface. Underwater lights were attached to each camera stand and positioned to provide suitable illumination but also minimize particle backscatter interference with video images. The cameras and lights were self contained and required no cables or other linkage to the surface.

The camera stands, cameras, and underwater lights were installed by divers at the beginning of each survey and retrieved the next morning before sampling for the impact assessment began. This ensured that the lights did not attract organisms to the screens that could bias the sampling.

Videos were recorded to a SD card onboard the camera that was capable of capturing approximately 7 hours and 18 minutes of high resolution digital video. The cameras were installed during the afternoon hours so that approximately equal video footage would be captured during daylight hours and after dark. During most surveys only one of the WBDDF pumps was running at a time so pump operation (1 mm [0.04 in.] or 2 mm [0.08 in.]) was coordinated with West Basin staff to capture approximately equal footage of each screen during both daytime and nighttime hours. Video data from the SD card were transferred to files in a folder on the Tenera server denoting the date and the WWS size. Files were then viewed on a computer screen with Adobe® Premier® Elements video editing software.

Impingement recordings were reviewed and observations logged of events where organisms or debris occurred in close proximity or contacted the screens. The observations logged only included events involving organisms and debris that were generally too large to be entrained into the DDF feedwater system. Logged events included quantitative observations of the number of times fish, impingeable invertebrates, or impingeable debris were observed near, or in contact with, the WWS, and qualitative observations such as when a fish appeared to resist the flow.

A total of 30 of the recordings made when the pumps were in operation were reviewed. Each of the reviewed recordings was randomly selected from four categories: 1 mm (0.04 in.) screen-day cycle, 2 mm (0.08 in.) screen-day cycle, 1 mm (0.04 in.) screen-night cycle, 2 mm (0.08 in.) screen- night cycle. Five video files from each of the two day cycle categories and ten video files from each of the two night cycle categories were randomly selected for review. All of the selected clips were watched in their entirety. The recordings were reviewed by a second biologist to ensure consistency with the logging of observations.

4.3.2 Results

A total of 12 WWS impingement surveys was conducted involving 21 filming events, 11 for the 1 mm (0.04 in.) WWS module and ten for the 2 mm (0.08 in.) WWS module (**Table 4-21**). Fewer impingement surveys were conducted for the 2 mm (0.08 in.) screen module because damaged grating prevented the installation of the camera stand during the first two surveys. New camera stands, designed and constructed to anchor to the riser pipe instead of the grating, were installed for the first time during the third survey on May 25, 2012. Despite considerable damage to the fiberglass-reinforced grating during swell events in 2011, no additional problems were encountered with anchoring the cameras until February 2012. Winter storm swells had swept away most of the remaining grating by February 2012 and only the camera/light on the 2 mm (0.08 in.) screen module could be installed. However, the camera malfunctioned and did not record any useable video in February.

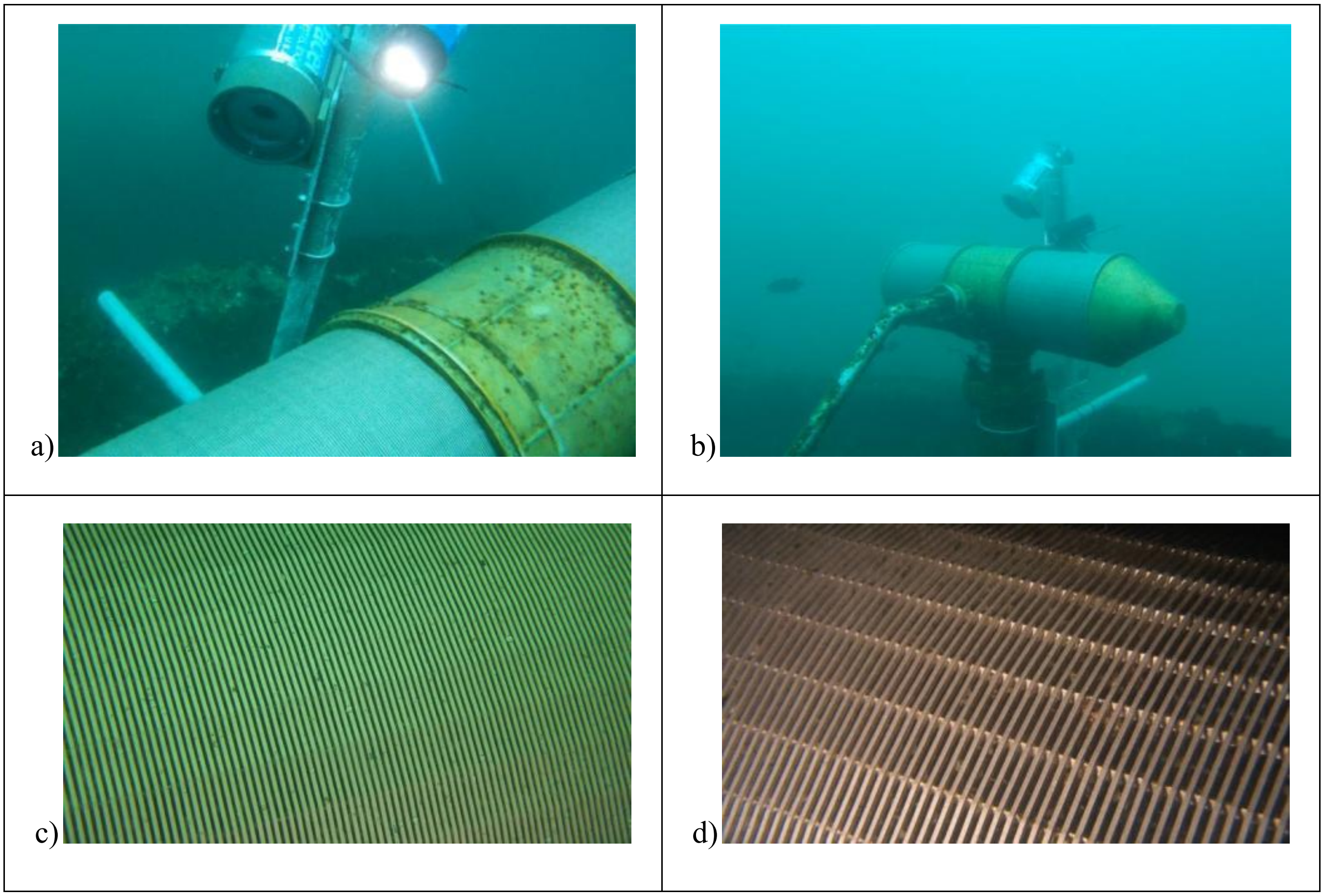


Figure 4-4. Photographs of: a) camera and light in filming position (upper left) on 1 mm (0.04 in) screen, b) camera and light in filming position (top center) on 2 mm (0.08 in) screen, c) screenshot from 1 mm (0.04 in) camera during the day, and d) screenshot from 2 mm (0.08 in) camera at night.

Impingement surveys were initially one of the sampling tasks scheduled during the first sampling event of the month but were found to be more readily accomplished during the second sampling event. When conducted during the first sampling event, the impingement cameras were installed by divers during the afternoon hours on the day before baseline characterization and WWS efficiency surveys. The impingement surveys and pump sampling for the screen efficiency surveys could not be conducted concurrently because the video lights for the impingement study could bias the results of the pump sampling by attracting fish and other organisms that may not otherwise have been in the area around the screens. Impingement surveys were conducted during the afternoon and evening hours on the night before baseline characterization and WWS efficiency surveys from March-May 2011, and in August and December 2011. In November 2011 and January 2012 the impingement survey was conducted concurrently with the WWS efficiency surveys, however, samples were not processed during those months due to issues with biofouling inside of the screen modules in November and damage to the temporary replacement 1 mm (0.04 in.) screen module in January. Because sample data from these two surveys were not included in the data set used to analyze screen efficiency in reducing entrainment, the presence of the video lights during the surveys did not introduce bias into the results.

Table 4-21. Summary of the filming activities during the wedgewire screen impingement surveys from March 30, 2011 through February 22, 2012. Footage times shown are only for footage recorded when the pump for the particular screen was in operation and the camera was functioning properly.

Screen Efficiency Survey	Date	Day 1 mm Camera	Day 2 mm Camera	Night 1 mm Camera	Night 2 mm Camera	Total
1	March 30, 2011	113 min	0 min	324 min	0 min	7 hrs:17 min
2	May 4, 2011	113 min	0 min	325 min	0 min	7 hrs:18 min
3	May 25, 2011	–	–	–	–	–
4	June 21, 2011	–	–	–	–	–
5	July 28, 2011	28 min	113 min	170 min	126 min	7 hrs:17 min
6	August 8, 2011	170 min	0 min	98 min	170 min	7 hrs:18 min
7	September 22, 2011	142 min	85 min	70 min	142 min	7 hrs 19 min
8	October 26, 2011	57 min	57 min	189 min	142 min	7 hrs:25 min
9	November 1, 2011	85 min	28 min	155 min	170 min	7 hrs:18 min
10	December 7, 2011	0 min	0 min	117 min	0 min	1 hr:57 min
11	January 18, 2012	198 min	198 min	240 min	240 min	14 hrs:36 min
12	February 22, 2012	–	–	–	–	0 min
	Total	**15 hrs: 7 min**	**8 hrs: 2 min**	**28 hrs: 7 min**	**16 hrs: 29 min**	**67 hrs: 45 min**

The remainder of the impingement surveys were conducted in conjunction with the second survey of the month. During these surveys, the impingement cameras were installed either prior to the initiation of pump sampling or during the break between the collection of day cycle sample replicates. Two WWS impingement surveys were conducted in May 2011, the first included only the installation of a camera on the 1 mm (0.04 in.) screen because damage to the grating around the 2 mm (0.08 in.) screen module prevented camera installation. Impingement cameras were installed on both screen modules for the first time during the survey on 25 May 2011 following the modification of the camera stands.

Recordings made during several surveys were not used in the impingement analysis due to uncertainty regarding the operation of the DDS feedwater pumps or cameras. The impingement study was initiated assuming that both pumps were operating when the survey was being conducted. Video recordings made during the May and June 2011 surveys were not used in the impingement analysis because the operational status of the pumps could not be determined. Filming and pump operation was coordinated with West Basin staff starting with the July 2011 survey to record approximately equal video of the two WWS modules in operation during both the day and night cycle. Camera malfunctions limited the amount of useable video recorded during the November 2011 and December 2011 surveys, and also resulted in no useable video being recorded during the February 2012 survey.

A total of 67 hours and 45 minutes of digital video (135 files) was recorded of the intake screens while they were in operation during the WWS impingement study (**Table 4-21**). Of these, 23 hours and 9 minutes were recorded during the day and 44 hours and 36 minutes were recorded at night. The shortened diurnal period during the early spring, late fall, and winter compressed the sampling schedule and resulted in a reduction in the amount of video footage collected during the day cycle. A total of 15 hours and 7 minutes was recorded of the 1 mm (0.04 in.) screen in operation during the day and 28 hours and 7 minutes at night. A total of 8 hours and 2 minutes was recorded of the 2 mm (0.08 in.) screen in operation during the day and 16 hours and 29 minutes at night.

The 30 video files randomly selected for review totaled 14 hours and 10 minutes, or 21 percent of the recorded video. A total of 4 hours and 43 minutes of the reviewed video was recorded during the day and 9 hours 27 minutes was recorded during the night (**Table 4-22).** A total of 717 events was observed and recorded. **Figure 4-5** provides photographs of some of the events captured by the video cameras. All direct interactions with the screen were recorded for most of the randomly selected recordings. However, the numbers of invertebrates swimming near the screen were so great in the recordings made during the nighttime hours of the March 2011, December 2011, and January 2012 surveys that impingement events were not recorded individually. During these recordings, large numbers of mysid shrimp and other small invertebrates attracted by the video lights were either in close proximity to, or in contact with, the screens at all times. Since these organisms were entrainable and generally too small to be impinged, their presence near, or contact with, the screens was not recorded as individual events. Of the recorded observations, 339 were of invertebrates, 226 were of fishes, and 152 were of debris. A description of each event was recorded on filming log sheets and events were tallied by the type of object (fish, invertebrate, or debris) and whether they came in contact with the screen or were seen in the vicinity of the screen.

Of the 226 interactions with fish recorded during the logging of the WWS impingement study, 198 occurrences were observed near one of the screens but not in contact with it, while during the other 28 logged occurrences fishes were observed contacting one of the screens. Contact behaviors were classified according to the duration and speed of contact, or activity such as picking food items off of the screen surface (**Table 4-23**). Early post larval fishes were observed frequently among the swarms of mysid shrimp occurring near the screens in recordings made at night during the January 18, 2012 survey. Many of the fishes observed were small enough to pass through the screen slots and be entrained, and therefore were too small to meet the event criterion for logging on the filming data sheet. Additionally, the smaller fish were difficult to distinguish from the great numbers of mysid shrimp and other small invertebrates constantly swirling past the screens, so the actual number of fish occurring near the screen during the survey was likely much greater than logged. Interestingly, observations show that these small, entrainable, early post larval fishes were able to swim away from the screen if they drifted too close or made screen contact, thereby avoiding entrainment or impingement.

Table 4-22. Summary of information collected from randomly chosen video files taken during the wedgewire screen impingement study from March 30, 2011 through February 22, 2012.

Date	Cycle	Sample Length (min)	Amount Reviewed (min)	Percent Reviewed	Fishes		Invertebrates		Debris	
					Near	Contact	Near	Contact	Near	Contact
03/30/11*	Day	–	–	–	–	–	–	–	–	–
	Night	113	113	100	0	2	0	17	0	0
05/04/11	Day	57	57	100	19	0	0	5	1	1
	Night	85	85	100	5	2	135	54	69	23
07/28/11	Day	–	–	–	–	–	–	–	–	–
	Night	28	28	100	0	0	1	13	4	4
08/08/11	Day	–	–	–	–	–	–	–	–	–
	Night	28	28	100	0	0	0	0	0	0
09/22/11	Day	113	113	100	67	5	25	9	8	10
	Night	57	57	100	2	0	0	3	0	0
10/26/11	Day	–	–	–	–	–	–	–	–	–
	Night	113	113	100	1	0	3	27	3	1
12/07/11*	Day	–	–	–	–	–	–	–	–	–
	Night	28	28	100	13	5	0	1	0	0
01/18/12*	Day	113	113	100	9	1	5	2	11	16
	Night	113	113	100	82	13	1	38	0	1
Totals		**14 hrs: 10 min**	**14 hrs: 10 min**	**100**	**198**	**28**	**170**	**169**	**96**	**56**

* Numbers of invertebrates in constant proximity/contact with the screen too great to record individually.

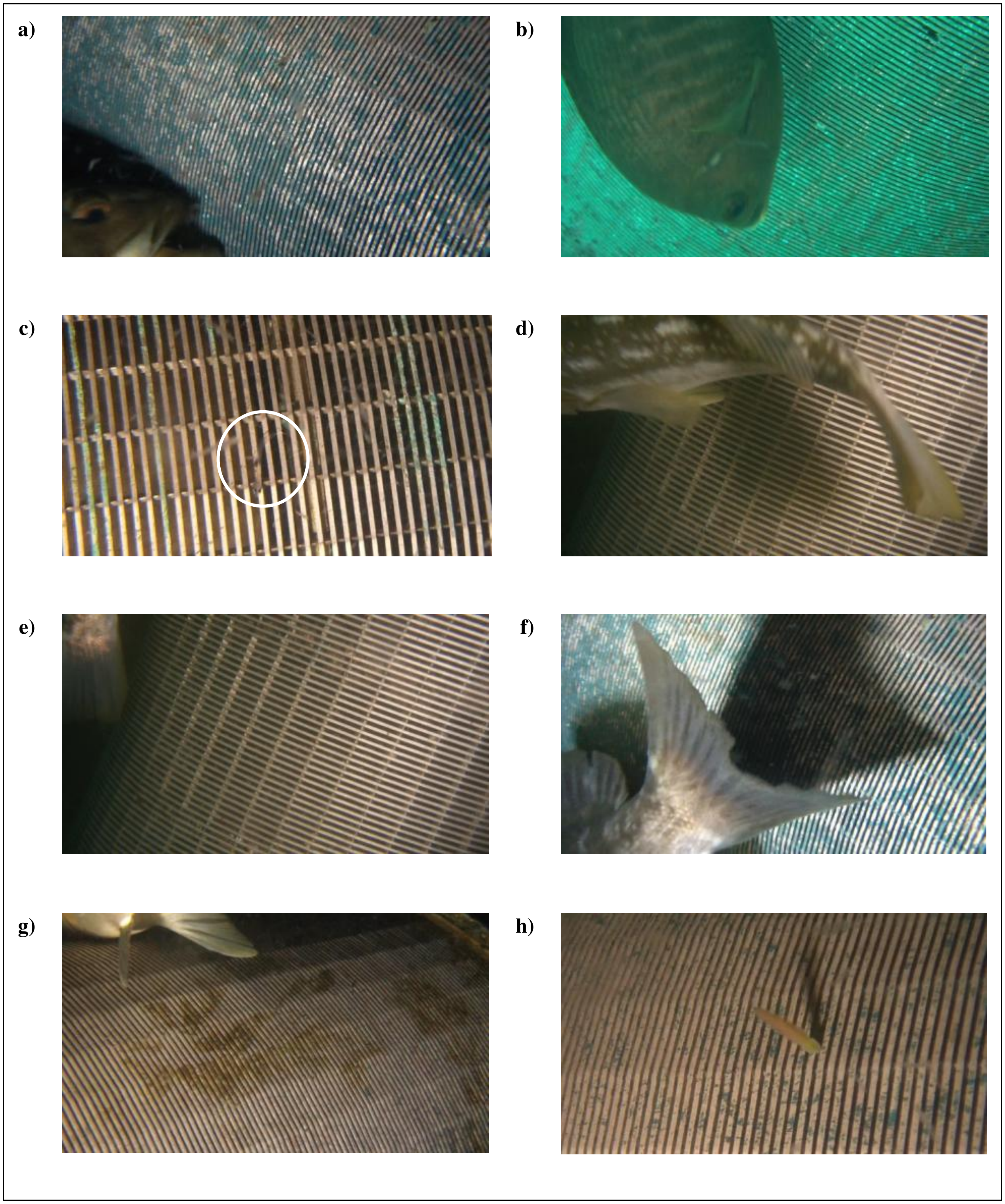


Figure 4-5. Video frame grabs taken during wedgewire screen efficiency study with pump operating: a) kelp bass feeding on invertebrates near screen; b) black perch swimming close to screen; c) small fish (possibly a goby) resting on screen (in white circle); d) kelp bass contacting screen; e) and f) kelp bass tail sweep on screen; g) kelp bass bumping screen; and h) polychaete trying to enter 1 mm (0.04 in.) screen.

Table 4-23. Description of behavior categories used to classify fish interactions with wedgewire screens during operational flow periods.

Behavior Category	Description
bump	Anterior or lateral parts of the body briefly (<1 sec) contact the screen; usually induced by wave surge
sweep	Tail only brushes against screen as fish swims by once
scrape	Fish contacts screen and is drawn transversely across screen
contact	Controlled contact with screen, then active swimming movements away
eat	Fish picks food item off of screen (e.g., polychaete, amphipod)
sit	Resting in place on screen
swim	Fish swimming near screen with several brief periods of contact

The majority of fish interaction events occurred during the surveys conducted from September 2011–January 2012. Of the 28 events in which a fish was observed to contact the screen, the majority could be classified as “bumps” or other brief “contacts” with the screen, which were usually followed by “sweeps” of the tail fin as a fish swam past the screen (**Table 4-24**). The most frequently observed fishes were kelp bass (*Paralabrax clathratus*) and many of the logged contact events were of kelp bass feeding on mysids which were likely attracted to the screen by the video lights. Unidentified post larval fishes were the second most frequently observed group making contact with the screens. During most logged events these small fishes would swim into the camera view and then drift or be drawn down toward the screen only to dart up and away from the screen after contact (**Figure 4-6a- Figure 4-6h**). At no time was a fish observed that appeared to be impinged or otherwise held on either the 1 mm (0.04 in.) or 2 mm (0.08 in.) screen by intake flows.

Table 4-24. Summary of recorded fish contacts with the wedgewire screens during video surveys conducted from March 30, 2011 through January 18, 2012.

	Kelp Bass		Unid. Goby		Unid. Post-Larval Fish		
Behavior	n	avg. sec.	n	avg. sec.	n	avg. sec.	Total No. of Interactions
bump	10	<1.0	–	–	–	–	**10**
sweep	5	1.0	–	–	–	–	**5**
scrape	–	–	–	–	–	–	–
contact	1	<1.0	–	–	10	<1.0	**11**
eat	–	–	–	–	1	<1.0	**1**
sit	–	–	1	20.0	–	–	**1**
swim	–	–	–	–	–	–	–
	16		**1**		**11**		**28**

The 339 invertebrate interactions logged during review of the WWS impingement study video files represent only a small fraction of the invertebrates observed near or in contact with the screens. Most of the invertebrates that were observed but not logged were too small to be impinged and therefore did not meet the criterion for a logged event. These included small caprellids and other amphipods, as well as other entrainable invertebrates, swimming around and into and out of, or crawling on the screen. In addition to these entrainable invertebrates, the numbers of both entrainable and potentially impingeable invertebrates swimming near the screen were so great in the recordings made during the nighttime hours of the March 2011, May 2011, December 2011, and January 2012 surveys that most proximity and contact events could not be recorded individually. In these recordings, debris and invertebrates were moving past the cameras (swimming or being swept by through wave action) in such numbers and/or at such velocity that individual events could not be logged even when viewed frame by frame. As a result, the number of logged events is considerably lower than the number of invertebrates observed near or in contact with the screens. The primary omission from the logged events was the great numbers of mysid shrimp swimming constantly near the screen. However, numerous larger caprellids and other amphipods crawling on, and moving in and out of, the screen were also not logged independently because it could not be determined if the contact by the organism had been continuous or if it was even the same individual re-emerging from the screen.

The 170 observations of invertebrates in the vicinity of the screen consisted almost entirely of amphipods, isopods, polychaetes, and mysid shrimp. The 169 observations logged where invertebrates contacted the screen consisted primarily of caprellids and other amphipods crawling on the screen, and shrimps bumping into it. During the night recordings for the March 2011, survey two polychaete worms tried to enter the 1 mm (0.04 in) screen but were unsuccessful and swam away **Figure 4-5h**. Many apparent entrainment events were observed during review of the video recordings, however, often the entrained organism would swim back out of the screen. During a few logged events small isopods and unidentified worms appeared to be held against the screen briefly by either wave surge or intake flows. In several such events the organism was pulled into the screen (entrained) but in other events the organism moved off the screen either by swimming or being washed off by wave action. Observations during the impingement study do not indicate that mysid shrimp are vulnerable to impingement from the DDF intake flows. Caprellids and other amphipods appeared to be using the screens for feeding and shelter in the reviewed video and were also not impinged by DDF intake flows.

The 56 interactions of debris contacting the screen primarily consisted of organic matter being pulled onto the screen, where it was typically pushed off by wave action after brief contact, sometimes after rolling across the screen’s surface. During a few logged events debris was pulled into the screen (entrained).

Video data recorded and reviewed during the study do not show correlations between screen slot size and the number of organisms occurring near or contacting the screen (**Table 4-25**). Higher numbers of fishes and invertebrates were logged near, and in contact with, the screens at night than during the day. Much of the variation between the day and night cycles can be attributed to organisms being attracted to the video lights. Differences in the number of logged events between each screen do not appear to be the result of intake flows but instead reflect the conditions specific to each of the reviewed video files such as strong wave action or little wave action and large aggregations of invertebrates or few invertebrates. Independent events could not be logged during several of the reviewed video files due to the great numbers of mysid shrimp and other invertebrates in constant view near a screen. One important finding of the impingement study was that no observations were logged that provide any clear examples of an organism encountering the screen that was held on or appeared to struggle to move away from the surface of either the 1 mm (0.04 in) or 2 mm (0.08 in) WWSs.

Table 4-25. Comparison of information by screen size and cycle for events logged during review of 30 randomly chosen video files recorded during the wedgewire screen impingement study from March 30, 2011 through February 22, 2012.

Screen Size	Cycle	Fishes		Invertebrates		Debris		Impingement Events	Total
		Near	Contact	Near	Contact	Near	Contact		
1 mm	Day	26	0	0	5	9	17	0	**57**
	Night	66	8	139	97	72	23	0	**405**
2 mm	Day	69	8	30	9	11	10	0	**137**
	Night	37	12	1	58	4	6	0	**118**
	Totals	**198**	**28**	**170**	**169**	**96**	**56**	0	**717**

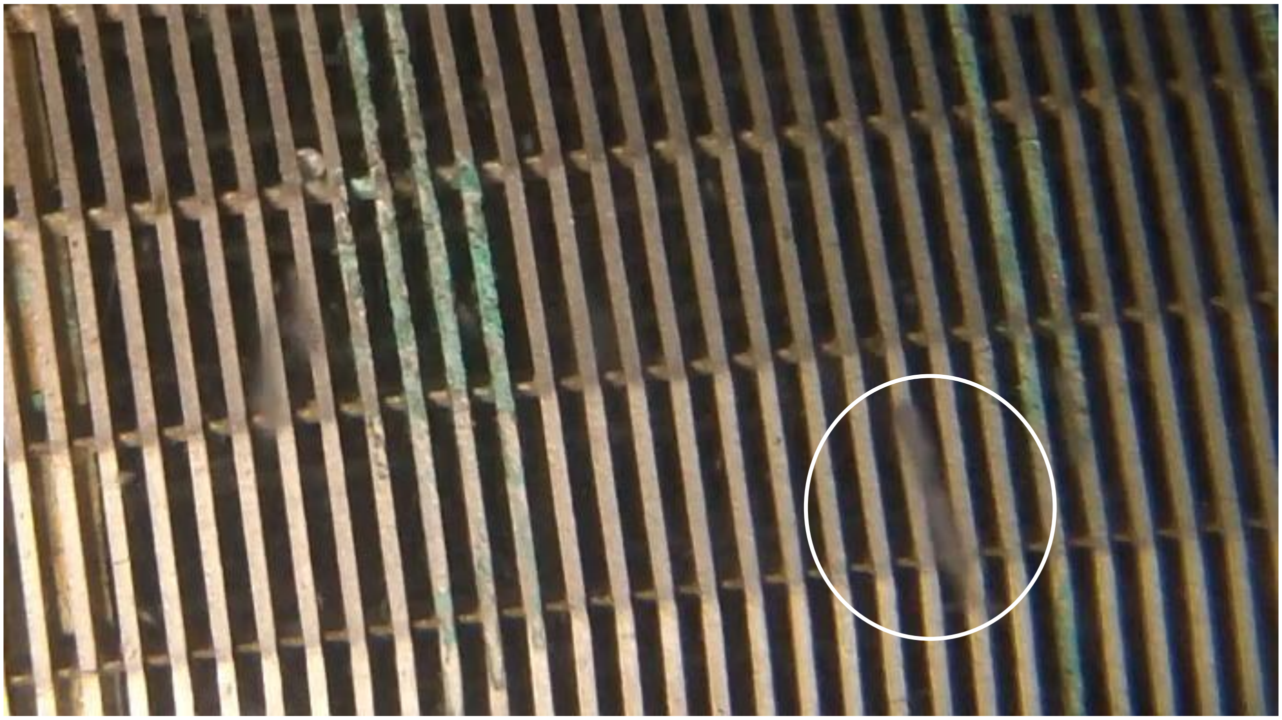


Figure 4-6a. Video frame grab of the 2 mm screen taken in January 2012 during the wedgewire screen efficiency study with the pump operating. Frame shows an early post-larval fish (est. 16 mm in length) swimming into view above screen.

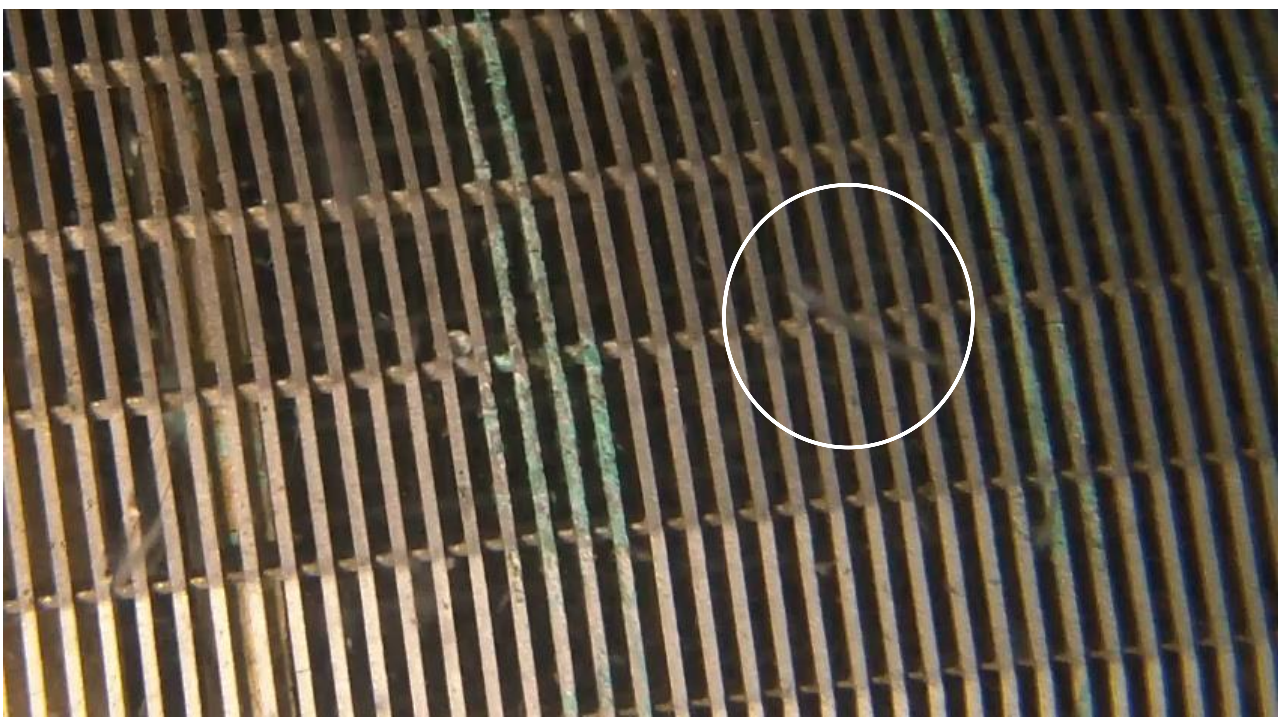


Figure 4-6b. Video frame grab of the 2 mm screen taken in January 2012 during wedgewire screen efficiency study with the pump operating. Frame shows the early post-larval fish swimming along horizontal to the screen.

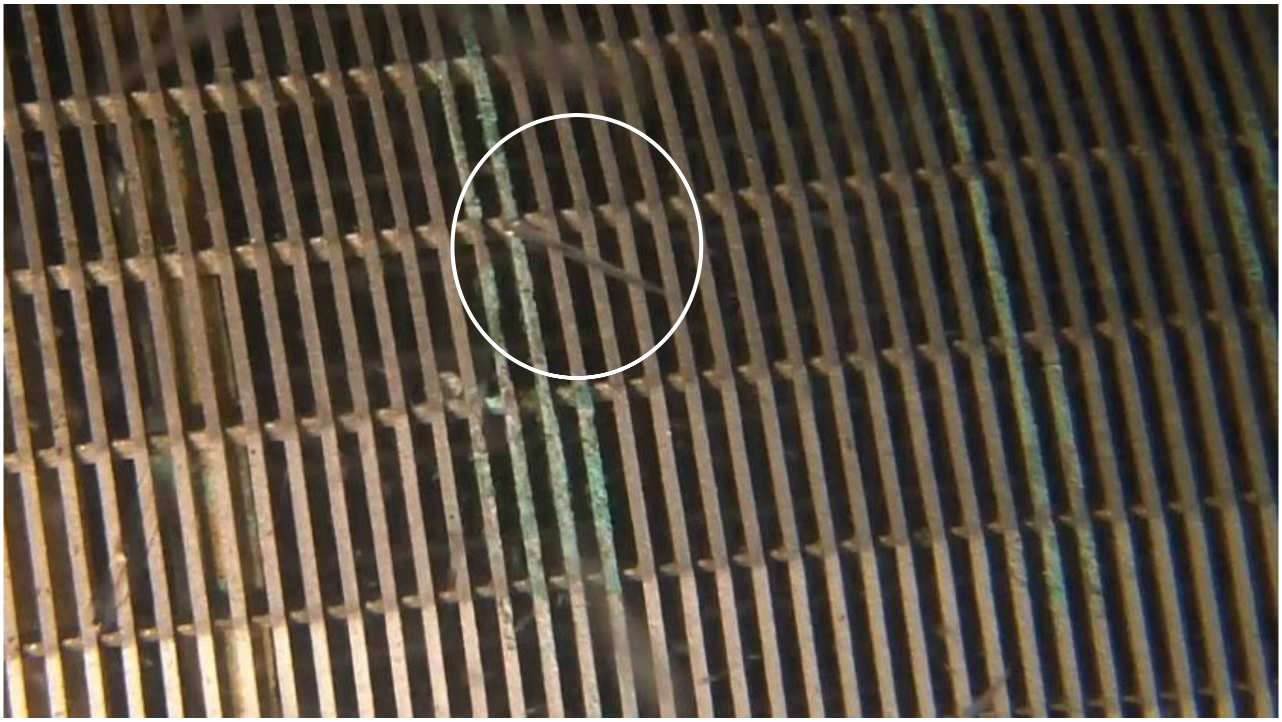


Figure 4-6c. Video frame grabs of the 2 mm screen taken in January 2012 during the wedgewire screen efficiency study with the pump operating. Frame shows the fish swimming horizontally above but slightly closer to the screen.

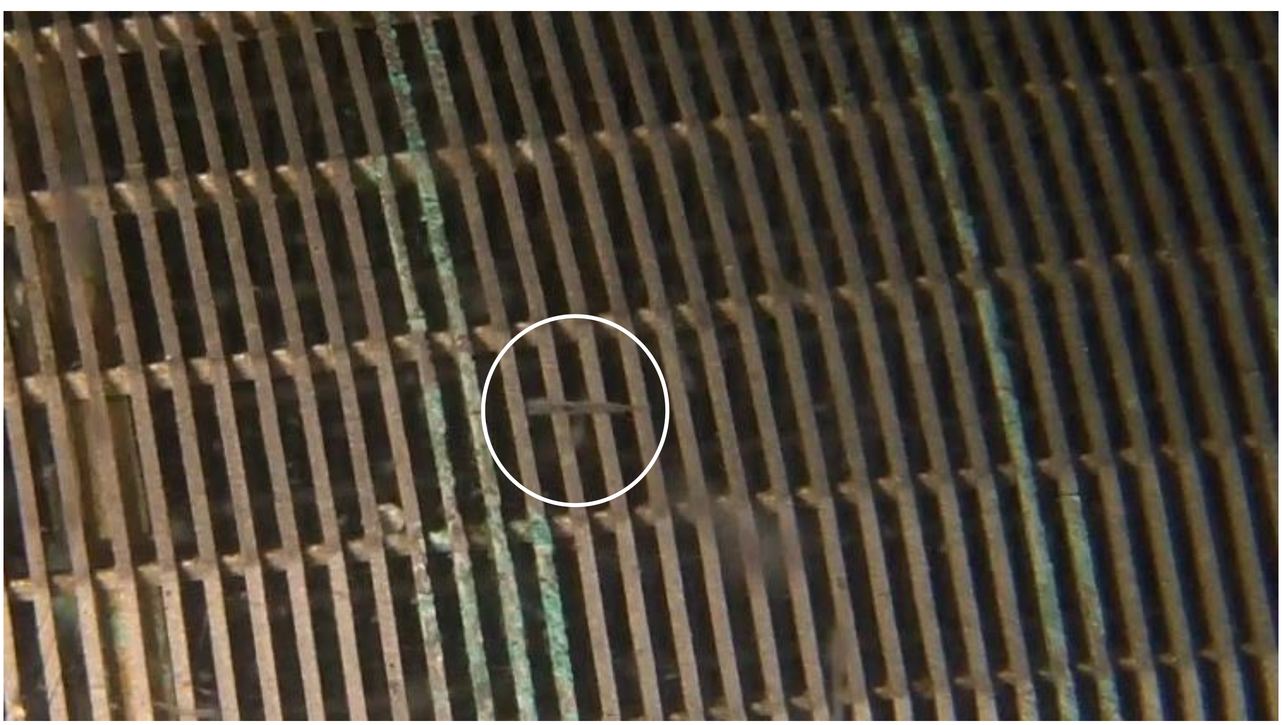


Figure 4-6d. Video frame grab of the 2 mm screen taken in January 2012 during the wedgewire screen efficiency study with the pump operating. Frame shows the fish as it drifts down head first toward the screen.

Figure 4-6e. Video frame grab of the 2 mm screen taken in January 2012 during the wedgewire screen efficiency study with the pump operating. Frame shows the fish possibly contacting the screen.

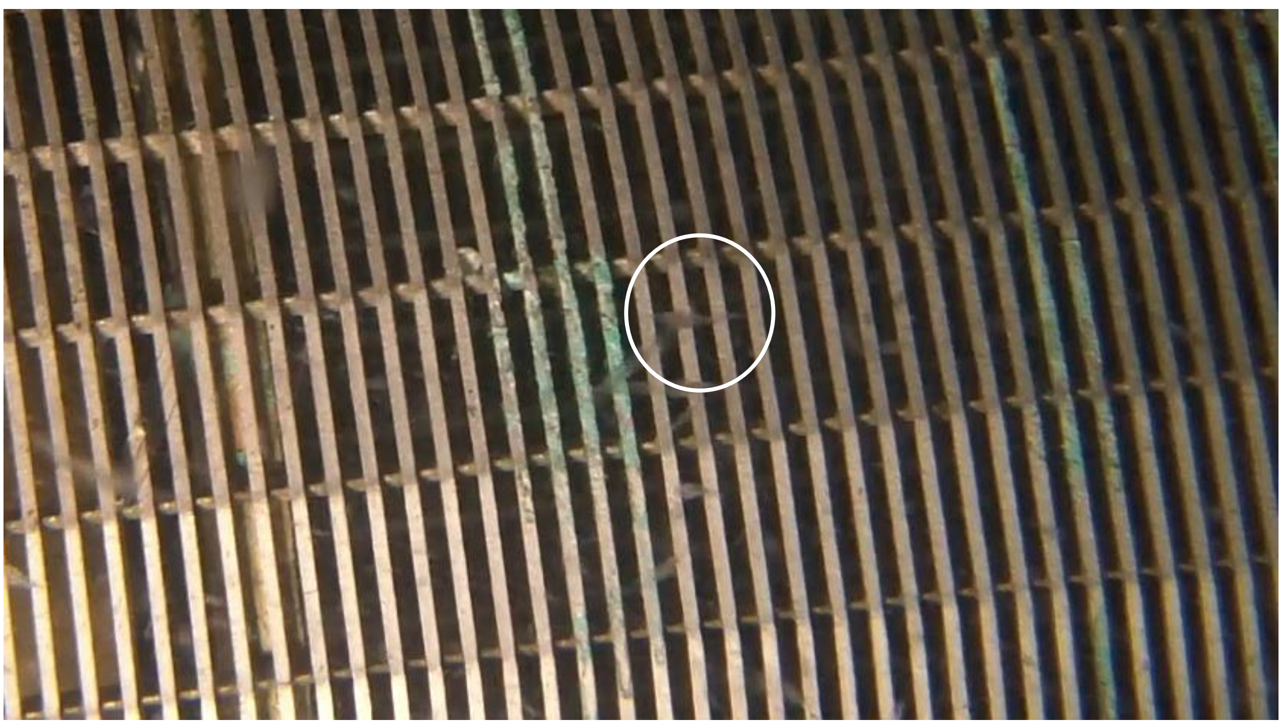


Figure 4-6f. Video frame grab sequence of the 2 mm screen taken in January 2012 during the wedgewire screen efficiency study with the pump operating. Frame shows the fish as it initiates rapid tail beats and begins movement away from the screen.

Figure 4-6g. Video frame grab of the 2 mm screen taken in January 2012 during the wedgewire screen efficiency study with pump operating. Frame shows the fish as it moves rapidly away from the screen.

Figure 4-6h. Video frame grab of the 2 mm screen taken in January 2012 during the wedgewire screen efficiency study with the pump operating. Frame shows the fish as it swims away from the screen and the frequency of its tail beats slows.

4.3.3 Screen Entrainment Comparison Study

The sampling done from March 2011 through April 2012 to determine the potential effectiveness of WWS on reducing the effects of entrainment by the WBDDF demonstrates the difficulty of detecting small differences in abundances in a highly variable environment. The WWS modules are expected to provide only a small percentage decrease in entrainment by only excluding larger organisms. Detecting differences in abundance for the 1 mm (0.04 in.) and 2 mm (0.08 in.) WWS slot openings that were 10 percent or less for most taxa as shown in **Tables 4-15** to **4-19** requires very large sample sizes with low and controlled levels of variation. While the expected reductions may be larger for some taxa, for example anchovies and silversides, any valid statistical analysis would still require a large sample size over a wide range of lengths. Even in studies that were conducted during periods with much higher larval fish concentrations, there were generally not enough larvae collected over a wide range of sizes to conduct a valid analysis of screen performance. As a result, to obtain a more representative sample over a range of lengths, data from studies throughout the southern California coastal region were combined for the analyses presented in this report. Therefore, while the low abundances of larval fish over the sampling period would have made it extremely difficult to successfully complete this additional sampling effort, the modeling approach presented here may be the most practical approach to WWS screen evaluation. The results of the modeling are also likely to be conservative as they assume that all larvae with head capsule dimensions smaller than the WWS slot opening would be entrained and does not consider the potential orientation of the larvae or hydrodynamic forces that may improve screen performance.

While the overall assessment of WWS performance was based largely on modeling, the results of the sampling provided supporting evidence that the WWS screens would be effective at reducing entrainment. For example, the results showed reduced entrainment of fish eggs and invertebrate larvae in the WWS modules relative to the unscreened intake. Although the results were not significant, the differences in average entrainment were consistent with the expectation that, on average, entrainment would be highest in the unscreened intake samples and lowest in the 1 mm (0.04 in.) WWS intake samples. The uncertainty in interpreting the results for fish larvae were reduced for fish eggs and invertebrate larvae, which were collected in much higher numbers. Also, the results for fish eggs may indicate that there is some benefit to the low through-slot velocity of the screens and the hydrodynamic forces around the screen as the diameters of the egg stage for most marine fishes in California is smaller than the 1 mm (0.04 in.) slot opening of the smallest WWS screen (Steinbeck 2011). The hydrodynamic forces may increase cross screen velocities and carry fish eggs and larvae along the screen face at a velocity exceeding the through-slot velocity resulting in entrainment reductions that exceed the levels expected based on egg diameter. Although hydrodynamic forces may be important in increasing the performance of the screens, the size of the larvae that could be entrained is still limited by the dimensions of the head capsule relative to the WWS slot width.

The length (and corresponding age) of the larvae entrained is important because the life history characteristics of most marine fishes and invertebrates involves the production of a very large number of larvae. These larvae experience very high rates of natural mortality due to numerous factors such as predation, ocean conditions, and absence of suitable habitat. For example, the expected survival to an age of one year for northern anchovy, which includes a 2–3 day planktonic egg stage, is 0.003 percent (based on stage specific survival rates from Butler et al. 1993). Therefore, the main function of WWS is to reduce or eliminate entrainment of larger, older larvae which have a much greater probability of surviving to the adult stage. This is shown by comparing the entrainment reduction for the five target fish taxa in **Tables 4-15** to **4-19** with the estimated population-level reductions for those fishes in **Table 4-20**. The population-level reductions represent the percentage reductions in late stage larvae that are no longer susceptible to entrainment. Since the results of the assessment also indicates that there would be no added mortality to later stage larvae and juveniles due to impingement, the population-level reductions would represent the total mortality to all life stages.

4.3.4 Wedgewire Screen Impingement Assessment Summary

The objectives of the WWS impingement study were to determine the potential impingement effects on marine organisms due to the operation of the intakes for the demonstration and full-scale facilities, and to compare the efficiency of intake modules with differently-sized WWS material at reducing impingement. Videographic methods were used to measure impingement, or demonstrate its absence at the two free-standing, open-ocean intake screens used in the study. Twenty one percent of the video recorded during the study was reviewed to assess overall impingement effects and to compare impingement rates between the two screen modules tested. No observations were logged from the video files reviewed that provide any clear examples of an organism encountering the screen that was held on, or appeared to struggle to move away from, the surface of either the 1 mm (0.04 in.) or 2 mm (0.08 in.) WWS. These results showing absence of any impingement effects, are consistent with previous studies in which similar videographic methodology was used to determine the potential impingement effects on marine organisms from use of passive cylindrical narrow-slot screens for an open ocean desalination facility intake.

The results of this study were also intended to assess possible reductions in impingement that could be attained by reducing the WWS mesh size from 2 mm (0.08 in.) to 1 mm (0.04 in.). However, the data showed that impingement did not occur on either the 1 mm (0.04 in) or 2 mm (0.08 in) narrow slot intake screens tested during the study, and either slot width would effectively eliminate impingement of sea life of any kind at the desalination seawater intakes. It is highly likely that the screen's ability to reduce or eliminate impingement at full scale would be equally effective.

The performance of a cylindrical narrow-slot screen in an open ocean setting is affected by specific-site ocean conditions, particularly the strength and frequency of ambient wave surge and the depth of the intake screen and possibly by the orientation of the screen. The WBDDF intakes are in relatively shallow water and the screens are oriented perpendicular to the shoreline. Strong wave surge was noted in many of the video files reviewed for the impingement assessment, however, the surge was also relatively weak in some of the reviewed files. It is reasonable to expect that some organisms might remain on the surface of the screen for longer periods of time when wave surge is of low strength and frequency. Screen transit time would also be affected by the length and diameter of the intake screen. During the study the screen transit times were a matter of seconds or fractions of a second. The intake screens were also situated in a location with high variability in the strength and direction of ocean currents. The wave surge, currents, and orientation at the WBDDF intake site may function to enhance performance of the intake screens. Therefore, the conclusions regarding the effectiveness of the screens at reducing impingement may not apply to other locations.

The observation of the digital video images and results from the analysis of the videographic data demonstrate that the impingement of sea life from desalination plant intakes can be effectively eliminated by use of a passive cylindrical narrow-slot screen of a design, construction, and flow rate (0.1 m/sec [0.33 ft/sec]) similar to either the 1 mm (0.04 in.) and 2 mm (0.08 in.) WWS tested during this study. The video files recorded provide unambiguous evidence of marine fishes and invertebrates as small as a few millimeters that are able to encounter the surface of the screen and move away at will. Qualitative impressions from watching video recordings of these encounters suggest that the ability of these small organisms to move off of the screen’s surface is aided by the continually changing directions and velocities of currents on the surface of the screen that are created by its interaction with the nearly constant to and fro of wave surge. In any case, in viewing the video recordings we found no clear example of an organism being impinged or appearing to have difficulty moving away from the screen’s surface.

5.0 Integrated Discussion

The results of the studies presented in this report indicate that the operation of the WBDDF and the proposed full-scale project with a projected intake volume of 170,722 m^3 (45.1 mgd) will result in very low levels of impact to populations of fishes and invertebrates that were the focus of the study. The detailed impact assessment used the same modeling approach, ETM, used in evaluating the impacts of ocean intakes at most of the coastal power plants in California. The results of the modeling showed that the estimated impacts due to the operation of the WBDDF, with a maximum daily intake volume of 1,935 m^3 (0.511 mgd), resulted in estimated losses to the populations of larvae for fishes and crabs in the extrapolated source water of only a hundredth, or thousandths, of a percent (**Table 3-21**). While the estimated impacts were higher for the proposed full-scale facility, which were based on an intake volume of 170,722 m^3 (45.1 mgd), the losses due to entrainment represented only 1–2 percent of the estimated source water populations for all of the taxa analyzed except silversides.

The estimated losses due to entrainment on the population of silverside larvae in the source water for the full-scale facility with a daily intake volume of 170,722 m^3 (45.1 mgd) were estimated at 7.9 percent. The integrated estimate of ETM is dependent on the estimates of proportional entrainment (*PE*) which represent the ratio of the estimate of entrained larvae to the estimate of the number of larvae in the source water and therefore reflects the greater concentrations of silverside larvae collected at the intake location (Station SWE – 40.8 per 1,000 m^3) relative to the average concentration at the source water stations (26.3 per 1,000 m^3). The concentrations were likely greater at the intake location since jacksmelt females attach their eggs to subtidal vegetation, such as kelp and eelgrass, and also other structures (Love 2011). This type of habitat is prevalent around Redondo Beach and King Harbor where there are small kelp beds in areas exposed to ocean water, shallow areas where eelgrass may occur, and also rock jetties and pier pilings covered with marine algae. This habitat also occurs at the base of the concrete structure where the WBDDF intakes were located.

Although silverside larvae were collected in high numbers using the net sampling used for the impact assessment, they were collected in very low numbers from the pumped samples for both the unscreened and WWS intakes. While the low numbers did not allow for an assessment of the effectiveness of the WWS at reducing entrainment, the large average length of the silverside larvae collected from the net samples (9.87 mm [0.39 in.]) indicates that the planned use of an intake with a WWS would reduce or nearly eliminate entrainment of silverside larvae, since the large size of the larvae would prevent the entrainment of a large percentage of the larvae. For example, the modeling indicates that a 0.5 mm (0.02 in.) WWS module would reduce the numbers of silverside larvae entrained by almost 97 percent (**Table 4-19**). Depending on the final selection of WWS technology and slot opening, the 7.9 percent ETM estimate of population-level losses would need to be adjusted by the level of reduction obtained through the

WWS. For example, the adjusted ETM estimate using a 0.5 mm (0.02 in.) WWS module would be 0.2 percent.

The estimates of impacts from the ETM need to be considered in context with the levels of entrainment since the actual number of larvae entrained may be very small relative to the reproductive capacity of the particular species. Although this can be done using adult equivalent modeling approaches, this is not necessary when the absolute levels of entrainment are very low as was the case in this study for all the taxa analyzed with the exception of silversides. For example, the total entrainment estimates for white croaker and California halibut larvae for the proposed full-scale project were 945,568 and 181,368 per year, respectively. These annual entrainment estimates represent the annual production of a few females for white croaker (based on an average batch fecundity of 19,000 eggs and an average of 19 batch spawnings per year [Love et al. 1994]) to perhaps only one female for California halibut (based on an average batch fecundity of 522,000 eggs and an average of 12 batch spawnings per year [Caddell et al. 1990]).

The reproductive capacity of Pacific rock crab is also very large relative to the total estimated annual entrainment of crab megalops, but more importantly the results of the comparison of the WWS modules indicated the potential for reduction in entrainment of cancer crab megalops, which, while not as large as silverside larvae, are larger than the WWS slot widths (ca. 5 mm). Also, crab megalops have a hardened carapace that is less compressible than the soft tissues found in fish larvae.

The low abundances of larval fish during the sampling period made it extremely difficult to successfully complete the evaluation of the effectiveness of the WWS at reducing entrainment. As a result, a modeling approach was used that may be the most practical approach to WWS screen evaluation, due to the difficulty in designing studies to provide adequate estimates of WWS entrainment reduction. The results of the modeling are also likely to be conservative as they assume that all larvae with head capsule dimensions smaller than the WWS slot opening would be entrained and does not consider the potential orientation of the larvae or hydrodynamic forces that may improve screen performance.

The results of the sampling provided evidence that the WWS screens may be more effective at reducing entrainment than the results of the modeling indicate. For example, the results showed reduced entrainment of fish eggs and invertebrate larvae in the WWS modules relative to the unscreened intake. The reduced entrainment of fish eggs through the WWS modules may indicate that there is some benefit to the low through-slot velocity of the screens and the hydrodynamic forces around the screen as the diameters of the egg stage for most marine fishes in California is smaller than the 1 mm (0.04 in.) slot opening of the smallest WWS screen (Steinbeck 2011). The hydrodynamic forces may increase cross screen velocities and carry fish eggs and larvae along the screen face at a velocity exceeding the through-slot velocity resulting in entrainment reductions that exceed the levels expected based on egg diameter or head capsule dimension. Although hydrodynamic forces may be important in increasing the performance of the screens, the size of the larvae that could be entrained is still limited by the dimensions of the head capsule relative to the WWS slot width.

The evaluation of impingement using underwater videography showed that the WWS modules are very effective at reducing or eliminating impingement. Reducing or eliminating impacts on adult fishes is critical to maintaining sustainable populations. Unlike losses to larvae due to entrainment which has never been shown to directly result in any impacts to adult populations at coastal power plants in California with much larger intake volumes, the loss of adults directly affects the reproductive capacity of the population. The importance of protecting reproductive adults through the reduction or elimination of impingement is most likely the rationale behind the most recent draft regulations for Section 316(b), which sets strict guidelines for reducing impingement mortality, but provides considerable flexibility in addressing entrainment.[7]

As the results of the screen effectiveness study showed, intakes fitted with WWS will significantly reduce or eliminate any effects due to impingement, and modeling results also show the potential for reduction in entrainment. As explained in Section 4.4, the evaluation of the effectiveness of WWS needs to consider the sizes of the larvae entrained. The life history characteristics of most marine fishes and invertebrates involves the production of a very large number of larvae, which experience very high rates of natural mortality. The effectiveness of WWS is measured by eliminating entrainment of larger, older larvae which have a much greater probability of surviving to the adult stage. The proven effectiveness of WWS at reducing or eliminating impingement and the potential for reductions in entrainment indicate that the use of WWS modules at the planned West Basin facility and the low intake volume represents the best technology currently available for reducing the impacts of ocean intakes.

[7] U.S. Environmental Protection Agency. 2011. Proposed Rules National Pollutant Discharge Elimination System—Cooling Water Intake Structures at Existing Facilities and Phase I Facilities. Federal Register Vol. 76, No. 76, April 20, 2011: 22174–22288.

6.0 Literature Cited

Ahlstrom, E. H. and H. G. Moser. 1975. Distributional atlas of fish larvae in the California Current region: flatfishes, 1955 through 1960. CalCOFI Atlas No. 23. 207 p.

Ahlstrom, E. H., K. Amaoka, D. A. Hensley, H. G. Moser, and B. Y. Sumida. 1984. Pleuronectiformes: development. Pp. 640–670 *In* H. G. Moser, W. J. Richards, D. M. Cohen, M. P. Fahay, A. W. Kendall, Jr., and S. L. Richardson, eds. Ontogeny and systematics of fishes. Amer. Soc. Ichthyol. and Herpetol., Spec. Publ. No. 1. 760 p.

Allen, G., R. Robertson, and B. Lea. 2010. *Hypsypops rubicundus*. *In:* IUCN 2012. IUCN Red List of Threatened Species. Version 2012.1. www.iucnredlist.org.

Allen, L. G. 1982. Seasonal abundance, composition, and productivity of the littoral fish assemblage in upper Newport Bay, California. Fish. Bull. 80: 769−790.

Allen, L. G. and D. J. Pondella II. 2006. Surf zone, coastal pelagic zone, and harbors. pp. 149−166 *in* The Ecology of Marine Fishes: California and Adjacent Waters. L. G. Allen, D. J. Pondella, and M. H. Horn (eds.). University of California Press, Berkeley, 670 p.

Allen, L. G. and E. E. DeMartini. 1983. Temporal and spatial patterns of nearshore distribution and abundance of the pelagic fishes off San Onofre–Oceanside, CA. Fish. Bull. U.S. 81(3): 569−586.

Allen, L. G., M. H. Horn, F. A. Edmands II, and C. Usui. 1983. Structure and seasonal dynamics of the fish assemblage in the Cabrillo Beach area of Los Angeles Harbor, California. Bull. So. Cal. Acad. Sci. 82(2):47–70.

Allen, L. G., M. M. Yoklavich, G. M. Cailliet, and M. H. Horn. 2006. Bays and estuaries. Pp. 119−148 *In* The ecology of marine fishes: California and adjacent waters. L. G. Allen, D. J. Pondella II, and M. H. Horn, eds. Univ. Calif. Press., Los Angeles, CA. 660 p.

Allen, M. J. 1982. Functional structure of soft-bottom fish communities of the southern California shelf. Ph.D. dissertation, Univ. Calif., San Diego, La Jolla, CA. 577 p.

Allen, M. J. 2006. Continental shelf and upper slope. Pp. 167−202 *in* The ecology of marine fishes: California and adjacent waters. L. G. Allen, D. J. Pondella II, and M. H. Horn, eds. Univ. Calif. Press, Los Angeles, CA. 660 p.

Allen, M. J. and K. T. Herbinson. 1991. Beam–trawl survey of bay and nearshore fishes of the soft–bottom habitat of southern California in 1989. CalCOFI Report 32: 112−127.

Ally, J. R. R. 1975. A description of the laboratory-reared larvae of *Cancer gracilis* Dana, 1852 (Decapoda, Brachyura). Crustaceana 23: 231−246.

Bane, G. W. and A. W. Bane. 1971. Bay fishes of northern California. Mariscos Publications, Hampton Bays, N. Y. 143 p.

Bergen, D. R. and L. D. Jacobsen. 2001. Northern anchovy. Pp. 303–305 *in* W. S. Leet, C. M. Dewees, R. Klingbeil, and E. J. Larson (eds.). California's living marine resources: A status report. Calif. Dept. Fish and Game. 592 p.

Boreman, J., C. P. Goodyear, and S. W. Christensen. 1978. An empirical transport model for evaluating entrainment of aquatic organism by power plants. US Fish Wildl. Ser. FWS/OBS–78/90, Ann Arbor, MI.

Boreman, J., C. P. Goodyear, and S. W. Christensen. 1981. An empirical methodology for estimating entrainment losses at power plant sites on estuaries. Trans. Amer. Fish Society 110: 253-260.

Brewer, G. D. 1978. Reproduction and spawning of northern anchovy, *Engraulis mordax*, in San Pedro Bay, CA. Calif. Fish. Game 64(3): 175–184.

Butler, J. L., P. E. Smith, and N. C. H. Lo. 1993. The effect of natural variability of life-history parameters on anchovy and sardine population growth. California Cooperative Oceanic Fishery Investigations Report 34:104−111.

Caddell, S. M., D. M. Gadomski, and L. R. Abbott. 1990. Induced spawning of the California halibut, *Paralichthys californicus*, under artificial and natural conditions. Calif. Dept Fish Game, Fish Bull. 174:175−197.

Carroll, J. C. 1982. Seasonal abundance, size composition, and growth of rock crab, *Cancer antennarius*, off central California. J. Crustacean Biol. 2(4): 549–561.

Claisse, J. T., D. J. Pondella II, J. P. Williams, and J. Sadd. 2012. Using GIS mapping of the extent of nearshore rocky reefs to estimate the abundance and reproductive output of important fishery species. PLoS ONE 7(1): e30290.

Clark, F. N. 1929. The life history of the California jacksmelt, *Atherinopsis californiensis*. Calif. Div. Fish Game, Fish Bull. 16. 22 p.

Clark, F. N. and J. B. Phillips. 1952. The northern anchovy (*Engraulis mordax*) in the California fishery. Calif. Dept. Fish and Game No. 38(2): 189–208.

Clark, G. H. 1930. California halibut. Calif. Fish Game 16:315–317.

Clarke, T. A. 1970. Territorial behavior and population dynamics of a pomacentrid fish, the garibaldi, *Hypsypops rubicunda*. Ecol. Mongr. 40:189−212.

Collins, R. A. 1969. Size and age composition of northern anchovies (*Engraulis mordax*) in the California anchovy reduction fishery for the 1965–66, 1966–67, and 1967–68 seasons. California Department of Fish and Game, Fishery Bulletin 14: 756–774.

Cross, J. N. and L. G. Allen. 1993. Fishes. Ch. 9 *in*: Dailey, M.D., D.J. Reish, and J.W. Anderson (eds.). Ecology of the Southern California Bight: A synthesis and interpretation. Univ. Calif. Press, Los Angeles, CA. 926 p.

Dailey, M. D., J. W. Anderson, D. J. Reish, and D. S. Gorsline. 1993. The Southern California Bight: Background and setting. Ch. 1 in Dailey, M.D., D.J Reish, and J.W. Anderson, eds.

Ecology of the Southern California Bight: A synthesis and interpretation. Univ. Calif. Press, Los Angeles, CA. 926 p.

Davis, G. E., D. J. Kushner, J. M. Mondragon, J. E. Mondragon, D. Lerma, and D. V. Richards. 1997. Kelp Forest Monitoring Handbook. Volume 1: Sampling Protocol Channel Islands National Park. National Park Service. Ventura, California.

DeMartini, E. E., A. M. Barnett, T. D. Johnson and R. F. Ambrose. 1994. Growth and production estimates for biomass-dominant fishes on a southern California artificial reef. Bull. Mar. Sci. 55(2–3):484–500.

Dong, C., Idica, E. Y., and J.C. McWilliams. 2009. Circulation and multiple-scale variability in the Southern California Bight. Progress in Oceanography 82: 168–190.

Emmett, R. L., S. A. Hinton, S. L. Stone, and M. E. Monaco. 1991. Distribution and abundance of fishes and invertebrates in west coast estuaries, Volume II: Species life history summaries. ELMR Report No 8. NOAA/NOS Strategic Environmental Assessments Division. Rockville, MD, 329 p.

Eschmeyer, W. N., and E. S. Herald. 1983. A field guide to Pacific coast fishes of North America. Houghton Mifflin Company, Boston. 336 p.

Feder, H. M., C. H. Turner, and C. Limbaugh. 1974. Observations on fishes associated with kelp beds in southern California. Calif. Dept. Fish and Game, Fish Bull. 160. 138 p.

Fitch, J. E. and R. J. Lavenberg. 1975. Tidepool and nearshore fishes of California. University of California Press, Berkeley, CA. 156 p.

Frey, H. W. (ed.) 1971. California's living marine resources and their utilization. California Department Fish Game. 148 p.

Fuiman, L. A. 1983. Growth gradients in fish larvae. J. Fish. Biol. 23:117-123.

Gadomski, D. M., S. M. Caddell, L. R. Abbott, and T. C. Caro. 1990. Growth and development of larval and juvenile California halibut *Paralichthys californicus*, reared in the laboratory. California Department of Fish and Game, Fish Bulletin 174: 85–98.

Garrison, K. J. and B. S. Miller. 1982. Review of the early life history of Puget Sound fishes. Fish. Res. Inst., Univ. Wash., Seattle, WA. FRI-UW-8216. 729 p.

Gisbert, E., G. Merino, J. B. Muguet, D. Bush, R. H. Piedrahita, and D. E. Conklin. 2002. Morphological development and allometric growth patterns in hatchery-reared California halibut larvae. J. Fish Biol. 61:1217-1229.

Graham, W. M. 1989. The influence of hydrography on the larval dynamics and recruitment of five *Cancer* crab species in northern Monterey Bay. M. S. Thesis, University of California, Santa Cruz. 170 p.

Gregory, P. A. 2001. Silversides. Pp. 243–245 *In* W. S. Leet, C. M. Dewees, R. Klingbeil, and E. J. Larson (eds.), California's living marine resources: A status report. Calif. Dept. Fish and Game. 592 p.

Haaker, P. L. 1975. The biology of the California halibut, *Paralichthys californicus* (Ayres), in Anaheim Bay, California. Pp. 137–151 *in* E.D. Lane and C.W. Hill (eds.). The marine resources of Anaheim Bay. Calif. Dept. Fish and Game, Fish Bull. 165.

Hart, J. L. 1973. Pacific fishes of Canada. Fish. Res. Board Can., Bull. 180. 740 p.

Haugen, C. W. (ed.) 1990. The California Halibut, *Paralichthys californicus*, Resource and Fisheries. Calif. Dept. Fish Game, Fish Bull. 174. 475 pp.

Hendricks, T. J. 1980. Currents in the Los Angeles area. Pp. 243-256 *in* Coastal Water Research Project Biennial Report 1979-1980. So. Calif. Coast. Water Res. Proj., Long Beach, CA. 363 p.

Herbinson, K. T., M. J. Allen, and S. L. Moore. 2001. Historical trends in nearshore croaker (Family Sciaenidae) populations in southern California from 1977 through 1998. Pp. 253-264 *In* S. B. Weisberg and D. Hallock (eds.), Southern California Coastal Water Research Project Annual Report 1999-2000, Southern California Coastal Water Research Project, Westminster, CA.

Hickey, B. M. 1992. Circulation over the Santa Monica-San Pedro basin and shelf. Prog. Oceanog. 30:37-115.

Hickey, B. M., E. L. Dobbins, and S. E. Allen. 2003. Local and remote forcing of currents and temperature in the central Southern California bight. J. Geophysical Res. 108:C3 (26):1-26.

Hines, A. H. 1991. Fecundity and reproductive output in nine species of *Cancer* crabs (Crustacea, Brachyura, Cancridae). Canadian Journal of Fisheries and Aquatic Sciences. 48: 267–275.

Holbrook, S. J., R. J. Schmitt, and J. S. Stephens. 1997. Changes in an assemblage of temperate reef fishes associated with a climate shift. Ecological Applications 7: 1299–1310.

Horn, M. H. and L. G. Allen. 1978. A distributional analysis of California coastal marine fishes. J. Biogeogr. 5: 23-42.

Horn, M. H. and K. L. M. Martin. 2006. Rocky intertidal zone. Pp. 205–226 *in* The ecology of marine fishes: California and adjacent waters. L. G. Allen, D. J. Pondella II, and M. H. Horn, eds., Univ. Calif. Press., Los Angeles, CA. 660 p.

Hunter, J. R. and B. J. Macewicz. 1980. Sexual maturity, batch fecundity, spawning frequency, and temporal pattern of spawning for the northern anchovy, *Engraulis mordax*, during the 1979 spawning season. CalCOFI Rep. 21:139-149.

Hunter, J. R. and K. M. Coyne. 1982. The onset of schooling in northern anchovy larvae, *Engraulis mordax*. CalCOFI Rep. 23: 246–251.

Idica, E. Y. 2010. Contaminant transport in the Southern California Bight. Ph.D. dissertation in Civil Engineering, University of California Los Angeles. 137 p.

Intersea Research Corporation (IRC). 1981. Scattergood Generating Station Cooling Water Intake Study: 316(b) Demonstration Program. Prepared for the Los Angeles Dept. Water and Power, Los Angeles, CA. Nov. 1981.

Jaeger, E. C. and A. C. Smith. 1966. Introduction to the natural history of southern California. Univ. Calif. Press, Berkeley, CA. 104 p.

Jarvis, E. T., M. J. Allen, and R. W. Smith. 2004. Comparison of recreational fish catch trends to environment-species relationships and fishery-independent data in the southern California bight, 1980–2000. Calif. Coop. Oceanic Fish. Invest. Rep. 45:167–179.

Kramer, S. H. 1991. Growth, mortality, and movements of juvenile California halibut *Paralichthys californicus* in shallow coastal and bay habitats of San Diego County, California. Fish. Bull. 89(2):195–207.

Kramer, S. H. and J. S. Sunada. 2001. California halibut. Pp. 195–197 *in* W. S. Leet, C. M. Dewees, R. Klingbeil, and E. J. Larson (eds.). California's Living Marine Resources: A Status Report. Calif. Dept. Fish and Game. 592 pp.

Lamb, A. and P. Edgell. 1986. Coastal Fishes of the Pacific Northwest. Harbour Publishing Co. Ltd, Canada. 224 p.

Largier, J. 2003. Considerations in estimating larval dispersal distances from oceanographic data. Ecol. Appl. 13: S71–89.

Leet, W. S., C. M. Dewees, R. Klingbeil and E. J. Larson. 2001. California's living marine resources: A status report. California Department of Fish and Game. 592 p.

Leong, R. J. H. and C. P. O'Connell. 1969. A laboratory study of particulate and filter feeding of the northern anchovy (*Engraulis mordax*). Journal of the Fisheries Research Board of Canada, 26(3): 557–582.

Littell R. C., G. A. Milliken, W. W. Stroup and R. D. Wolfinger. 1996. SAS System for Mixed Models, Cary, NC: SAS Institute Inc.

Limbaugh, C. 1964. Notes on the life history of two Californian pomacentrids: garibaldi, *Hypsypops rubicundus* (Girard), and blacksmith, *Chromis punctipinnis* (Cooper). Pac. Sci. 28:41–50.

Love, M. S. 2011. Certainly more than you want to know about the fishes of the Pacific coast: A postmodern experience. Really Big Press, Santa Barbara, California. 650 p.

Love, M. S., C. W. Mecklenburg, T. A. Mecklenburg, and L. K. Thorsteinson. 2005. Resource Inventory of Marine and Estuarine Fishes of the West Coast and Alaska: A Checklist of North Pacific and Arctic Ocean Species from Baja California to the Alaska–Yukon Border. U. S. Department of the Interior, U. S. Geological Survey, Biological Resources Division, Seattle, Washington, 98104, OCS Study MMS 2005-030 and USGS/NBII 2005-001.

Love, M. S., G. E. McGowen, W. Westphal, R. J. Lavenberg, and L. Martin. 1984. Aspects of the life history and fishery of the white croaker, *Genyonemus lineatus* (Sciaenidae) off California. Fish. Bull., U.S. 82(1):179-198.

MacCall, A. D., K. R. Parker, R. Leithiser, and B. Jessee. 1983. Power plant impact assessment: a simple fishery production model approach. Fishery Bulletin 81: 613−619.

MBC and Tenera Environmental (Tenera). 2007. Redondo Beach Generating Station Clean Water Act Section 316(b) impingement mortality and entrainment characterization study. Prepared for AES Redondo Beach. December 2007. 392. p.

MBC, Tenera, and URS Corp. 2008. Scattergood Generating Station Clean Water Act Section 316(b) impingement mortality and entrainment characterization study. Prepared for City of Los Angeles Dept. of Water and Power, Los Angeles, CA.

MBC. 1987. Ecology of important fisheries species offshore California. OCS Study 86-0093. Prepared for Minerals Management Service, Pacific OCS Region. 251 p.

MBC. 1988. The State of Santa Monica Bay: Part I. Assessment of conditions and pollution impacts. Prepared for the So. Calif. Association of Governments. Oct. 1988.

MBC. 1993. Santa Monica Bay Characterization Study, 1993. Prepared for the Santa Monica Bay Restoration Project. April 1993.

MBC. 2006. National Pollutant Discharge Elimination System 2005 receiving water monitoring report, AES Redondo Beach L.L.C. Generating Station, Los Angeles County, California. Prepared for AES Redondo Beach L.L.C. 70 p.

McGowen, G. 1978. Effects of thermal effluent from Southern California Edison's Redondo Beach Steam Generating Plant on the warm temperate fish fauna of King Harbor Marina. Ichthyoplankton study report for Phase III. Occidental College, Los Angeles, CA. Submitted to So. Calif. Edison Co. 78-RD-47. Aug. 1978. 65 p.

Methot, R. D., Jr. 1989. Synthetic estimates of historical abundance and mortality for northern anchovy. American Fisheries Society Symposium 6: 66−82.

Methot, R. D., Jr. and D. Kramer. 1979. Growth of the northern anchovy, *Engraulis mordax*, larvae in the sea..Fish. Bull. 77:413–420.

Middaugh, D. P., M. J. Hemmer, J. M. Shenker, and T. Takita. 1990. Laboratory culture of jacksmelt, *Atherinopsis californiensis*, and topsmelt, *Atherinops affinis* (Pisces: Atherinidae), with a description of larvae. Calif. Fish Game 76(1):4–13.

Miller, D. J. and R. N. Lea. 1972. Guide to the coastal marine fishes of California. California Fish Bulletin No. 157. 249 p.

Miller, E. F., J. P. Williams, and D. J. Pondella, II. 2011. Queenfish (*Seriphus politus*) and white croaker (*Genyonemus lineatus*) larval growth parameters. CalCOFI Rep., Vol. 52: 75−79.

Moore, S. L. 2001. Age and growth of white croaker (*Genyonemus lineatus*) off Palos Verdes and Dana Point, California. Pp. 154-163 In: SCCWRP Annual Report 1999-2000. So. Calif. Coastal Water Res. Project, Westminster, CA. March 2001. 308 p.

Moore, S. L. and P. W. Wild. 2001. White croaker. Pp. 234-235 *In* W. S. Leet, C. M. Dewees, R. Klingbeil, and E. J. Larson (eds.), California's living marine resources: A status report. Calif. Dept. Fish and Game. 592 p.

Moser, H. G. (ed.). 1996. The Early Stages of Fishes in the California Current Region. California Cooperative Oceanic Fisheries Investigations, Atlas No. 33, National Marine Fisheries Service, La Jolla, California. 1505 p.

Moser, H. G. and P. E. Smith. 1993. Larval fish assemblages of the California current region and their horizontal and vertical distribution across a front. Bulletin of Marine Science 53: 645–691.

Moser, H. G. and T. Pommeranz. 1999. Vertical distribution of eggs and larvae of northern anchovy, Engraulis mordax, and of the larvae of associated fishes at two sites in the Southern California Bight. Fishery Bulletin 97: 920–943.

Moyle, P. B. and Cech. 1988. Fishes: An Introduction to Ichthyology. Department of Wildlife and Fisheries Biology, Univ. Calif. Davis. Prentice Hall, Englewood Cliffs, NJ.

Murdoch, W.W., R. C. Fay, and B. J. Mechalas. 1989. Final Report of the Marine Review Committee to the California Coastal Commission, MRC Doc. No. 89-02, 346 p.

National Data Buoy Center. 2011. http://www.ndbc.noaa.gov/station_page.php?station=icac1.

Nations, D. 1975. The genus *Cancer* (Crustacea: Brachyura): systematics, biogeography, and fossil record. Natural History Museum of Los Angeles County Science Bulletin, 23: 1-104.

Noble, M. A., K. J. Rosenberger, P. Hamilton, and J. P. Xu. 2009. Coastal ocean transport patterns in the central Southern California Bight. Pp. 193–226 *In* H. J. Lee and W. R. Normack (eds.). Earth Science in the Urban Ocean: The Southern California Continental Borderland. Geological Society of America, Special Paper 454.

Pacific Fisheries Information Network (PacFIN.). 2012. http://www.psmfc.org/pacfin/data.html.

Parker, K. R. and E. DeMartini. 1989. Chapter D: Adult-equivalent loss. *In* Technical Report to the California Coastal Commission. Marine Review Committee, Inc. p. 56.

Parrish, R. H., C. S. Nelson, and A. Bakun. 1986. Transport mechanisms and reproductive success of fishes in the California Current. Biological Oceanography 1(2):175-203.

Parrish, R. H., D. L. Mallicoate, and K. F. Mais. 1985. Regional variations in the growth and age composition of northern anchovy, *Engraulis mordax*. Fishery Bulletin 84: 483-496.

Pena, R. and S. Dumas. 2009. Development and allometric growth patterns during early larval stages of the spotted sand bass *Paralabrax maculatofasciatus* (Perdoidei: Serranidae). In C. Clemmenesen, A. M. Malzahn, M. A. Peck, and D. Schnack (eds.). Advances in early life history study of fish. Scientia Marina, Barcelona, Spain. p. 183-189.

Pondella, D. J. II, Stephens, J. S., Jr, and Craig, M. T. 2002. Fish production of a temperate artificial reef based on the density of embiotocids (Teleostei: Perciformes). – ICES Journal of Marine Science, 59: S88−S93.

Pondella, D. J., II, and J. S. Stephens, Jr. 1994. Factors affecting the abundance of juvenile fish species on a temperate artificial reef. Bulletin of Marine Science. 55(2):1216-1223.

Recreational Fisheries Information Network (RecFIN.). 2012. http://www.recfin.org/data.htm.

Ricker, W. E. 1975. Computation and interpretation of biological statistics of fish populations. J. Fish. Res. Board Canada 91: 382.

Robertson, D. R. and G. R. Allen. 2002. Shorefishes of the tropical eastern Pacific: an information system. Smithsonian Tropical Research Institution, Balboa, Panamá.

Roesijadi, G. 1976. Descriptions of the prezoeae of *Cancer magister* Dana and *Cancer productus* Randall and the larval stages of *Cancer antennarius* Stimpson (Decapoda, Brachyura). Crustaceana 31: 275–295.

Sakagawa, G. T. and M. Kimura. 1976. Growth of laboratory-reared northern anchovy, *Engraulis mordax*, from southern California. Fish. Bull. U.S. 74(2): 271–279.

Santa Monica Bay Restoration Commission (SMBRC). 2004. State of the Bay 2004: Progress and Challenges. 45 p.

SAS Institute. 2008. SAS/STAT user's guide, Version 9.2, SAS Institute Inc. Cary, NC.

Schlotterbeck, R. E. and D. W. Connally. 1982. Vertical stratification of three nearshore southern California larval fishes (*Engraulis mordax, Genyonemus lineatus*, and *Seriphus politus*). Fish. Bull. U.S. 80(4):895-902.

Schroeder, D. M. and M. S. Love. 2002. Recreational fishing and marine fish populations in California. CalCOFI Rep., 43: 185−190.

Schweitzer, C. E. and R. M. Feldmann. 2000. Re-evaluation of the Cancridae Latreille, 1802 (Decapoda: Brachyura) including three new genera and three new species. Contributions to Zoology, 69 (4). http://dpc.uba.uva.nl/ctz/vol69/nr04/art02.

Shanks, A. L. 1985. Behavioral basis of internal-wave-induced shoreward transport of megalopae of the crab *Pachygrapsus crassipes*. Marine Ecology Progress Series 24: 289−295.

Sikkel, P. C. 1989. Egg presence and developmental stage influence spawning-site choice by female garibaldi. Animal Behaviour 38(3): 447−456.

Sikkel, P. C. 1994a. Filial cannibalism in a paternal-caring marine fish: The influence of egg developmental stage and position in the nest. Animal Behaviour 47(5): 1149−1158.

Sikkel, P. C. 1994b. Why female garibaldi prefer males with young eggs: A test of the parental investment hypothesis. Ethology, Ecology & Evolution 6(2): 191−211.

Sikkel, P. C. 1995. Effects of nest quality on male courtship and female spawning-site choice in an algal-nesting damselfish. Bulletin of Marine Science 57(3): 682–689.

Smith, P. E. 1972. The increase in spawning biomass of northern anchovy, *Engraulis mordax*. Fish. Bull. U.S. 70: 849–874.

Smith, P. E. and S. L. Richardson. 1977. Standard techniques for pelagic fish egg and larva surveys. FAO Fisheries Technical Paper 175: 1–100.

Southern California Edison (SCE). 1983. Redondo Beach Generating Station 316(b) Demonstration. Prepared for the California Regional Water Quality Control Board, Los Angeles Region. January 1983.

Starr R. M., K. A. Johnson, E. A. Laman and G. M. Cailliet. 1998. Fishery resources of the Monterey Bay National Marine Sanctuary. California Sea Grant College System, University of California, La Jolla.

Steinbeck, J. R., J. Hedgepeth, P. Raimondi, G. Cailliet, and D. L. Mayer. 2007. Assessing power plant cooling water intake system entrainment impacts. Report to California Energy Commission. CEC-700-2007-010. 105 p.

Steinbeck, J. R. 2011. Desalination plant intake technology review. Prepared for California State Water Resources Control Board Expert Review Panel on Intake Impacts and Mitigation. http://www.swrcb.ca.gov/water_issues/programs/ocean/desalination/erp_intake052512.pdf, October 30, 2011.

Stephens, J. S. Jr., P. A. Morris, K. E. Zerba, and M. Love. 1984. Factors affecting fish diversity on a temperate reef II: the fish assemblage of Palos Verdes Point, 1974–1981. Environmental Biology of Fishes, 11:259-275.

Stephens, J. S., Jr., P. A. Morris, D. J. Pondella, T. A. Koonce, and G. A. Jordan. 1994. Overview of the dynamics of an urban artificial reef fish assemblage at King Harbor, California, USA, 1974-1991: A recruitment driven system. Bulletin of Marine Science. 55(2):1224-1239.

Stephens, John S. Jr., R. Larson, and D. Pondella. 2006. Rocky reefs and kelp beds. Pp. 227–252 *In* The Ecology of Marine Fishes: California and Adjacent Waters. L. G. Allen, D. J. Pondella, and M. H. Horn (eds.). University of California Press, Berkeley, 670 pp.

Stepien, C. A. 1986. Life history and larval development of the giant kelpfish, *Heterostichus rostratus* Girard, 1854. Fishery Bulletin. 84 (4): 809-826.

Swift, C. C. and G. D. Frantz. 1981. Estuarine fish communities of Ballona wetlands. *In* B. Schriber , ed. The biota of the Ballona region of Los Angeles County, F1-F30.

Swift, C. C., J. L. Nelson, C. Maslow, and T. Stein. 1989. Biology and distribution of the tidewater goby, *Eucyclogobius newberryi* (Pisces: Gobiidae) in California. Contributions in Science, 404: 1–19.

Tenera and MBC. 2008. El Segundo Generating Station Clean Water Act Section 316(b) impingement mortality and entrainment characterization study. Prepared for El Segundo Power LLC, El Segundo, CA.

Tenera. 2006. Compilation and analysis of CIAP nearshore survey data. Calif. Dept Fish Game. http://www.dfg.ca.gov/mrd/fir/sss.html#crane.

Tenera. 2011. Intake Screening Technology Support Studies: Morphology of Larval Fish Head Capsules. ESLO2011-005.1. Prepared for Pacific Gas and Electric. 26 p.

Terry, R. D., S. A. Keesling, and E. Uchupi. 1956. Submarine geology of Santa Monica Bay, California. Report to Hyperion Engineers, Inc., Geol. Dept., Univ. So. Calif., Los Angeles, CA. 177 p.

U. S. Environmental Protection Agency (USEPA). 1977. Guidance for evaluating the adverse impact of cooling water intake structures on the aquatic environment: Section 316(b) P.L. 92-500. 58 p.

Wang, J. C. S. 1986. Fishes of the Sacramento-San Joaquin Estuary and Adjacent Waters, California: a Guide to the Early Life Histories. Interagency Ecological Study Program for the Sacramento-San Joaquin Estuary, Technical Report 9.

Ware, R. R. 1979. The food habits of the white croaker *Genyonemus lineatus* and an infaunal analysis near areas of waste discharge in Outer Los Angeles Harbor. Masters Thesis, Calif. State Univ. Long Beach. August 1979. 163 p.

Watson, W. 1982. Development of eggs and larvae of the white croaker, *Genyonemus lineatus* Ayres (Pisces: Sciaenidae) off the southern California coast. Fish. Bull. U.S. 80(3): 403–417.

Watson, W. 1996. Clinidae: Clinid kelpfishes. Pp. 1164−1169 *in* The Early Stages of Fishes in the California Current System (H. G. Moser, Ed.), Vol. 33. Allen Press, Inc, La Jolla, California.

Wellington, G. M. and B. C. Victor. 1989. Planktonic duration of one hundred species of Pacific and Atlantic damselfishes (Pomacentridae). Mar. Biol. 101: 557−567.

Williams, G. C. 1954. Differential vertical distribution of the sexes in *Gibbonsia elegans* with remarks on two nominal subspecies of this fish. Copeia. 4: 267−273.

Winer B. J., D. R. Brown, and K. M. Michels. 1991. Statistical principles in experimental design. McGraw-Hill, New York, New York, USA.

Winn, R. N. 1985. Comparative ecology of three cancrid crab species (*Cancer anthonyi*, *C. antennarius*, *C. productus*) in marine subtidal habitats in southern California. Ph.D. Dissertation, Univ. So. Calif. 235 p.

Zalenke, B. 2012. West Basin Desalination Facility: Back-projection of trajectories from the intake structure off Redondo Beach for March 2011 through March 2012 source water samples. Report prepared for Tenera Environmental, July 2012. 18 p.