Duke Energy South Bay Power Plant

SBPP Cooling Water System Effects on San Diego Bay

VOLUME II: Compliance with Section 316(b) of the Clean Water Act for the South Bay Power Plant

August 12, 2004

Submitted to:

San Diego Regional Water Quality Control Board
9174 Sky Park Court, Suite 100
San Diego, CA 92123-4340

Submitted by:

Duke Energy South Bay, LLC
990 Bay Blvd.
Chula Vista, CA 91911

Duke Energy South Bay Power Plant

SBPP Cooling Water System Effects on San Diego Bay

VOLUME II: Compliance with Section 316(b) of the Clean Water Act for the South Bay Power Plant

Prepared by:

Tenera Environmental
225 Prado Rd. Suite D
San Luis Obispo, CA 93401

971 Dewing Ave. Suite 101
Lafayette, CA 94549

ESLO2003-037.6 August 12, 2004

Table of Contents

Appendices

List of Tables

2.0 Description of the South Bay Power Plant and Characteristics of the Source Water Body

3.0 Entrainment and Source Water Larval Study

4.0 Impingement Study Results

5.0 Cooling Water System Impact Summary Assessment

6.0 Evaluation of Alternative Intake Technologies

List of Figures

Executive Summary

The purpose of this report is to assess the effects of the cooling water intake system (CWIS) of the Duke Energy South Bay Power Plant (SBPP) in Chula Vista, California and to evaluate alternative intake technologies that may reduce potentially adverse environmental effects in compliance with Section 316(b) of the Clean Water Act. Section 316(b) requires that ". . . the location, design, construction, and capacity of cooling water intake structures reflect the best technology available for minimizing adverse environmental impact" (USEPA 1977).

This 316(b) study was mandated by the San Diego Region of the California Regional Water Quality Control Board (RWQCB) Tentative Order No. R9-2002-0022. In the order, RWQCB staff concluded that some of the results and conclusions from a previous 316(b) CWIS study were in need of revision because they did not reflect current plant operations or were not representative of existing conditions in south San Diego Bay. A letter dated May 24, 2002 from the Board's executive director to Duke Energy South Bay LLC contained several questions regarding the effects of the power plant's CWIS and described studies designed to address these questions and to collect additional information on present conditions in the power plant's cooling water source and discharge areas. The results of this study, and those from a companion volume (*Volume 1: 316(a) Demonstration for the South Bay Power Plant*) on the effects of the SBPP discharge on the biota of the receiving waters in south San Diego Bay, will be used in continuing the National Pollutant Discharge Elimination System (NPDES) permit renewal process for SBPP Permit Number CA0001368.

The San Diego Regional Water Quality Control Board (RWQCB) discussed the need for the additional information with a group of agency representatives and consultants who provided input on the design and implementation of the 316(a) and 316(b) studies at SBPP. The representatives included Duke Energy, Tenera Environmental, Merkel and Associates, San Diego Unified Port District, the RWQCB, California Department of Fish and Game, U. S. Fish and Wildlife Service, NOAA Fisheries, and U.S. EPA. The AWG members reviewed and commented on several drafts of the 316(b) Cooling Water Intake Effects Study Plan. The general approach to the studies was twofold: 1) to quantify the direct effects of the SBPP on fishes and invertebrates in San Diego Bay by measuring the annual abundance and biomass of these organisms impinged on the CWIS intake screens and, 2) to estimate the additional indirect effects on fish and invertebrate populations by measuring the quantity of their larval forms entrained along with the cooling water supply withdrawn from the bay and passing through the power plant.

The study plans for entrainment and impingement were designed to address the following specific questions:

- What are the species composition and abundance of the larval fishes, *Cancer* crabs, and spiny lobster entrained by SBPP and how do they compare to the source populations in south San Diego Bay?
- What are the potential impacts of entrainment losses on larval fish, *Cancer* crab, and spiny lobster populations due to operation of the CWIS?
- What are the species composition and abundance of the juvenile and adult fishes and macroinvertebrates impinged by SBPP?
- What are the potential impacts of impingement losses on populations of fishes and macroinvertebrates due to operation of the CWIS?

Entrainment and impingement studies were conducted to examine the effects of the CWIS using methods similar to an earlier 316(b) study conducted by SDG&E in 1980. Entrainment effects occur when small planktonic organisms are drawn through the power plant cooling water system and impingement effects occur when larger fishes and invertebrates are trapped against the intake screens (**Figure ES-1**). The current entrainment study estimated the number of the microscopic planktonic fish, *Cancer* crab, and spiny lobster larvae in front of the intakes and at other locations in south San Diego Bay. Entrainment and source water sampling was conducted at nine stations in south San Diego Bay (**Figure ES-2)** monthly from January 2001 through January 2002 and bi-monthly from December 2002 through October 2003. Plankton samples were collected by towing small-mesh nets using methods similar to those used in other long-term fishery investigations. Preserved samples were sorted in the laboratory and the fishes and target invertebrates were identified to the lowest practical taxon. Impingement was studied weekly over a 24-hr period from December 2002 through November 2003 by recording the numbers and weights of all fishes and selected macroinvertebrates that were rinsed from the screens of Units 1 & 2 and Units 3 & 4.

Entrainment effects were assessed using three independent models. Two of the models, Fecundity Hindcasting (*FH*) and Adult Equivalent Loss (*AEL*), used species life history information to estimate the potential numbers of adults represented by the entrainment losses. The third approach, Empirical Transport Modeling (*ETM*), compared entrainment larval densities to source water larval densities to calculate the effects of larval removal on the standing stock of larvae in south San Diego Bay. The source water volume used in the *ETM* calculations comprised the area of the bay south of the Coronado Narrows and encompassed the South and South-Central eco-regions of San Diego Bay. Tidal exchange ratios, source water volumes, cooling water volumes, larval concentrations and larval durations and were all variables used in the *ETM* calculations.

Conservative assumptions were used for developing the best estimates of losses due to power plant operation. For example, even though cooling water pumping volumes were 68–73% of maximum in the 2001–2003 period, maximum pump volumes were used in

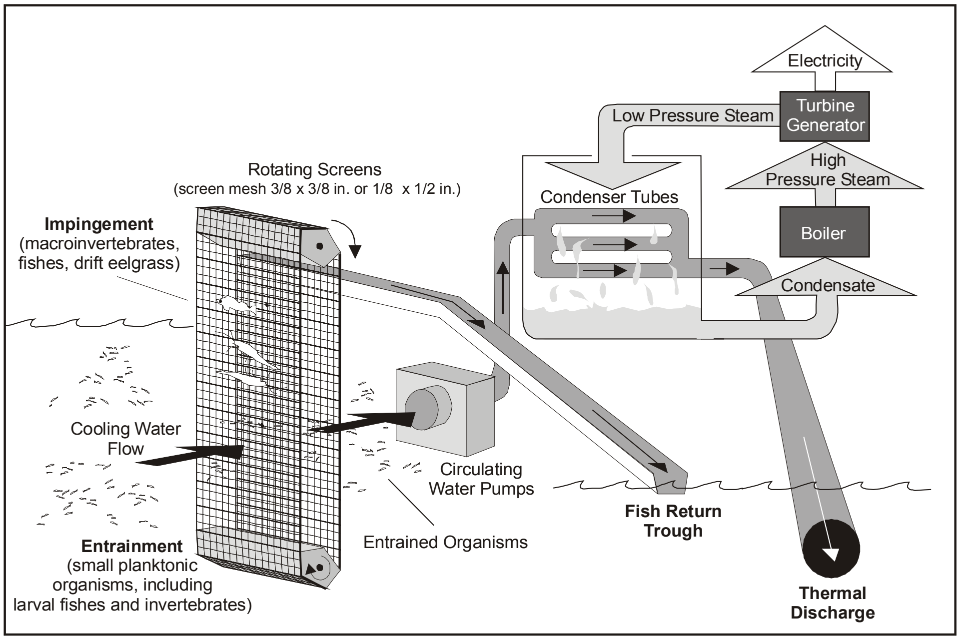


Figure ES-1. Conceptual diagram of impingement and entrainment processes and their relationship to the circulating water system at SBPP.

calculating potential entrainment and impingement losses. Further, although there is evidence that some organisms survive impingement and entrainment, the calculations assumed no survival.

The results of the 2001 and 2003 entrainment sampling study periods were as follows:

- Two taxa, CIQ gobies (comprised of arrow, cheekspot, and shadow gobies) and anchovies (comprised of bay and deepbody anchovies), comprised greater than 95 percent of the total estimated entrained larvae for both sampling periods. These are small forage fishes common in bays of southern California. Detailed assessments of entrainment effects were completed for the five taxa that comprised 99 percent of all of the entrained fish larvae (**Table ES-1**).
- California halibut, white seabass, and other commercial or recreational fishery species comprised less than 0.1 percent of the total estimated entrained larvae during both sampling periods. Because of their low abundances in entrainment samples, power plant effects on fishery species were not evaluated with the same modeling approaches used for the more abundant non-fishery taxa.
- During the first sampling period the greatest concentrations of larval fishes at the entrainment station occurred during June 2001, while during the second sampling period the greatest concentrations occurred during December 2002.

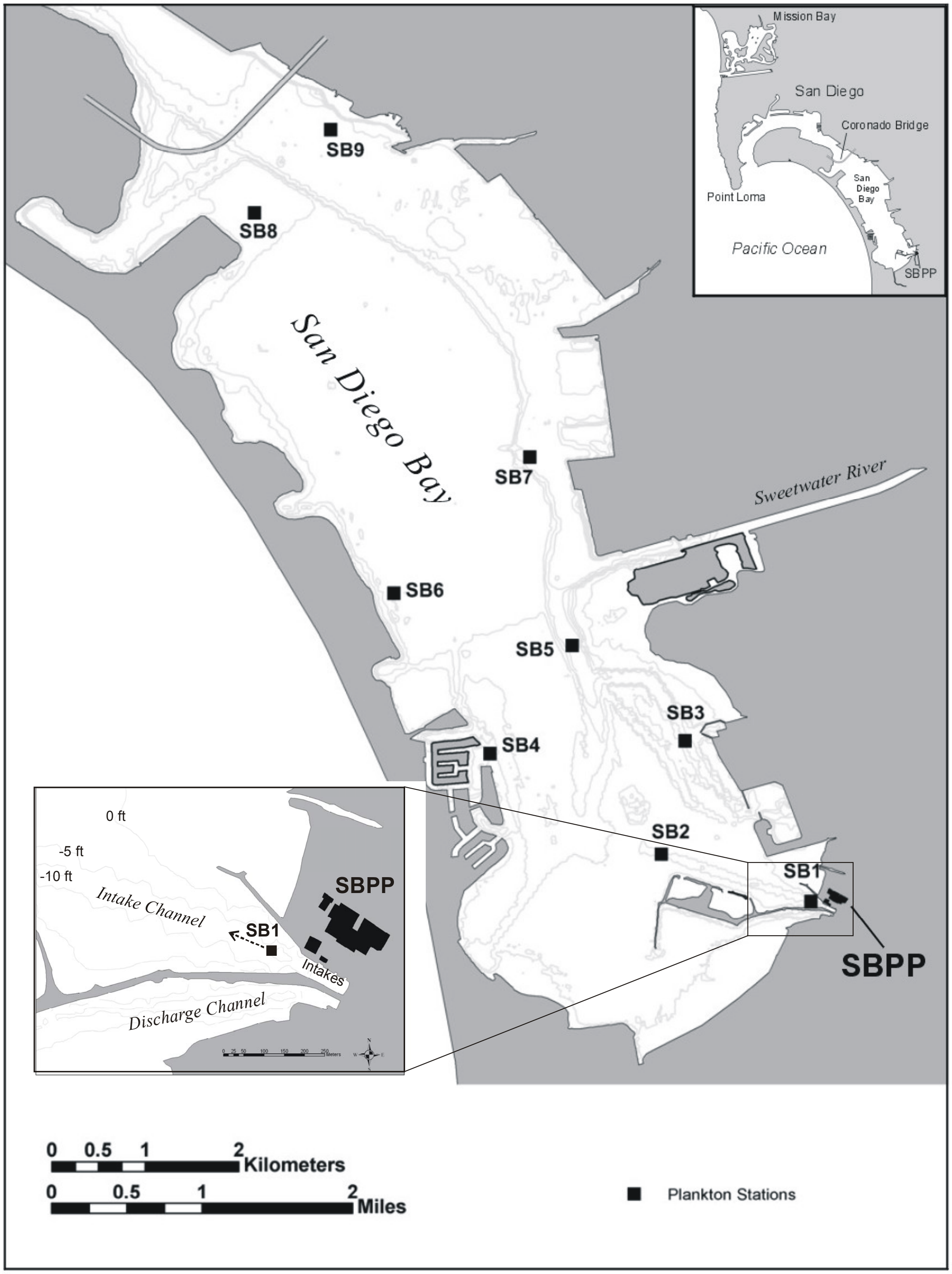


Figure ES-2. Location of SBPP entrainment (SB1) and source water plankton stations (SB2–SB9). Inset shows entrainment station in relation to SBPP with the direction and approximate length (100 m) of plankton tows. Impingement samples were collected directly from the intake screening system at SBPP.

- The larval fish community composition in south San Diego Bay changes along a gradient from north to south as a function of distance from the mouth of the bay. The abundances and numbers of species were lowest at the entrainment station and source water stations in the southernmost end of the bay.
- *ETM* estimates of entrainment mortality were 3–28 percent, although an estimate of 50 percent was calculated for longjaw mudsucker gobies during the 2003 sampling period. This estimate was affected by the reduced bi-monthly sampling effort during 2003. The *ETM* estimate from 2001, 17 percent, was considered to be more representative of entrainment effects on this species.
- The major results were very similar to the previous 316(b) study completed by SDG&E in 1980. Entrainment estimates for several of the species from this study were remarkably similar to estimates from the previous study. Our *ETM* estimates were similar to, or within the range of estimated effects on larval standing stock from the previous study.
- There was insufficient life history information and entrainment abundance to model adult equivalent losses for any of the fishery species, and silversides were the only taxa with assessment results that also had commercial landings data that could be used to value the losses. The *ETM* estimates of proportional larval mortality suggest losses of approximately 450,000 adult silversides. This extrapolation assumes a stable adult population and no compensation and would be very conservative for silversides due to the large variability in the adult population that far exceeds the 15 percent *ETM* estimate. The dollar value of entrainment losses of silversides was approximately $13,000.

Table ES-1. Summary of relative abundance of sampled fish larvae, estimated annual larval entrainment by SBPP, estimated annual source water population, estimated percent of source water larvae lost to entrainment by SBPP annually, and adult equivalent losses for the five most abundant larval fishes entrained. Estimates are based on 144 plankton samples from 2001, and 72 samples from 2003. Adult equivalent estimates are from the fecundity hindcasting method (*2FH*), and proportional losses are from *ETM* modeling.

	Percent Composition in Entrainment		Estimated Annual Larval Entrainment (in billions)		Estimated Annual Source Population (in billions)		Estimated Percent Larval Losses		Estimated Adult Equivalent Losses (in millions)	
Taxon	2001	2003	2001	2003	2001	2003	2001	2003	2001	2003
CIQ goby complex	75.6	89.0	1.83	1.39	8.51	5.21	21.5	26.7	2.17	1.65
Anchovies	21.3	6.8	0.52	0.11	4.95	1.39	10.5	7.9	0.21	0.05
Combtooth blennies	0.9	1.5	0.02	0.02	0.65	0.59	3.1	3.4	0.02	0.02
Longjaw mudsucker	0.9	1.6	0.02	0.02	0.12	0.04	17.1	50.2	<0.01	<0.01
Silversides	0.6	0.6	0.01	0.01	0.07	0.05	14.6	14.9	*	*

* Information unavailable to compute model estimate.

The results indicate low potential for entrainment effects on the five taxa analyzed. The increase in mortality due to entrainment, calculated for continuous full power operation, may be compensated for by increased survival of later larval and juvenile stages. The similarity in the estimates of entrainment losses between the 1979–1980 and 2001–2003 studies indicates that compensatory mechanisms are operating to maintain long-term stability in these populations. There is also evidence that some of these taxa may have behavioral adaptations to living in high current environments that would help reduce entrainment effects. The conclusion from this study that entrainment due to the SBPP cooling water system under full operation represents low potential risk to the target taxa populations is the same as the conclusion from the previous 316(b) Demonstration (SDG&E 1980).

Results from the 2002–2003 twelve-month impingement sampling program were as follows:

- A total of 50,970 individual fishes comprising approximately 50 taxa was collected from the 52 weekly impingement samples. The fishes weighed a total of 74 kg (163 lb).
- Total annual impingement of fishes under full operation flow rates was estimated to be 385,588 individuals weighing 556 kg (1,226 lb).
- The most abundant taxon both numerically and by weight impinged was anchovies (*Anchoa* spp.), comprising about 93 percent by number and 40 percent by weight of all of the fishes impinged. Most were juveniles.
- Crustaceans (shrimps, crabs, and lobster) and cephalopods (squid and octopus) were studied in more detail than other invertebrates because of their potential fishery value. A total of 1,106 crustaceans and cephalopods from 30 taxa was collected during the study. These individuals had a total wet weight of 3.1 kg (6.8 lb). In all, 80 invertebrate taxa were identified in the impingement samples.
- The estimated total annual impingement of target invertebrates under full operation was 9,019 individuals, with an estimated wet weight of 22.6 kg (49.8 lb).
- Most of the fishes impinged, over 96 percent of the total abundance and 87 percent of the biomass, were not commercially or recreationally fished species.
- There were several differences between the previous impingement study results (SDG&E 1980) and the current one. The estimated annual impingement in the prior study was 28,174 fishes weighing 4,459 kg (9,830 lb), while in the current study it was estimated at 385,588 fishes weighing 556.5 kg (1,226.4 lb).
- Anchovies (mainly juvenile slough anchovy) were more abundant in the recent study than the earlier study whereas round stingray, specklefin midshipman, diamond turbot, California halibut, and Pacific butterfish were less abundant.

- The estimated ex-vessel value for impingement losses under full power was less than $2,000 per year for the small numbers of fishes with commercial fishery landings.

The small magnitude of estimated impingement effects under full operation indicates that SBPP operation represents a low potential risk to target taxa populations. The previous 316(b) Demonstration also concluded that impingement effects were not significant (SDG&E 1980).

Alternative technologies, designs, and operational and maintenance features were evaluated to determine their potential to reduce biological losses and other environmental impacts to a greater extent than the existing facility. Evaluation of CWIS alternatives was conducted in accordance with EPA's Guidance for Evaluating the Adverse Impact of Cooling Water Intake Structures on the Aquatic Environment: Section 316(b) P.L. 92-500 (EPA Office of Water Enforcement, May 1, 1977). Section 316(b) of the Clean Water Act requires that "the location, design, construction, and capacity of cooling water intake structures reflect the best technology available for minimizing adverse environmental impact." The U.S. EPA interprets Best Technology Available (BTA) as "the best technology available commercially at an economically practicable cost." Determination of BTA includes the design, capacity, and location of the facility's cooling water intake, as well as cost considerations. The BTA determination is made on a case-by-case basis by assessing the relative biological value of reducing entrainment and impingement to the cost of the alternative.

Duke Energy leases both the power generating facilities located on the site and the site property from the Port of San Diego under an operating agreement that is due to expire in 2009. At the present time there are no plans to continue the operation of the existing facilities or replace them with newer equipment. Therefore several of the alternative intake technologies that might otherwise be considered feasible for SBPP are not feasible simply due to the fact that the time necessary to design, permit, and construct a number of the alternative technologies would extend beyond the life of the facility. However, Duke will remain in full compliance with both the spirit and intent of the 316(b) technology-based regulations regarding the location, design, and capacity of the facility's existing intake as the Company continues to operate to the end of its lease. Therefore this report includes a thorough assessment of alternative intake technologies though very few alternatives are feasible for the predicted future life of the facility.

A two-step system was used to evaluate alternatives to the existing CWIS. In the first step, potential alternatives were evaluated based on whether they met the BTA criteria for "proven and available." Alternatives that were not commercially available and had not been used successfully at a power plant similar in size and environmental setting to SBPP were eliminated from further consideration. Alternative intake technologies that met the site-specific proven and available criteria were further analyzed based on specific technical, economic, biological, and other environmental criteria. Thirty-two different alternatives were initially reviewed. Of these, 20 were determined not to be proven and available and thus were not considered further. These included four closed-cycle cooling systems, seven behavioral barriers, and nine physical barriers.

Five alternative cooling systems were evaluated:

- Closed-cycle cooling pond,
- Mechanical draft (wet) cooling tower,
- Natural draft cooling tower,
- Air-cooled condenser, and
- Wet/dry plume abatement cooling tower.

Wet/dry hybrid cooling towers using treated wastewater or desalinated water was the only viable closed-cycle cooling system for use at SBPP. This option was eliminated because of the short-term nature of Duke's SBPP lease, which expires in 2009. There would not be enough time to design, permit, and construct the cooling towers and the water treatment facilities. The costs of the two wet/dry alternatives relative to the life of the power plant were wholly disproportionate to the environmental benefits gained based on the entrainment and impingement data collected during the 2001 and 2003 studies. All the other closed-cycle cooling options were eliminated on the basis of physical limitations on the SBPP site, or unacceptable environmental impacts. For all the above reasons, the existing once-through cooling water system is preferred to any of the alternative cooling systems.

Safe and reliable reduction in cooling water pump operations coinciding with periods of reduced electrical generation or when units are out of service has also been identified as an effective method of reducing losses of organisms through entrainment and impingement. Seasonal curtailment of cooling system operations is generally practiced at power plants that entrain or impinge large numbers of commercially or recreationally important species. Entrainment and impingement studies showed that the effects of SBPP operation on local marine life were not detectable at the population levels of the species involved. More importantly, there is no certainty that reduced flows or seasonal curtailments to further reduce entrainment or impingement mortality would result in a detectable increase in population abundance for fish and invertebrate species inhabiting the San Diego Bay region and the adjacent coastal waters.

An offshore intake alternative design was also evaluated that includes the placement of underground intake pipes that would extend into the Pacific Ocean. Offshore intakes will always have higher intake velocities than typical shoreline intakes since the same amount of cooling water must be withdrawn through the smaller cross-sectional area of an offshore conduit compared to much larger cross-sectional areas of shoreline intakes. It was determined that the offshore alternative was not BTA when compared to the existing CWIS since it did not provide biological benefits. Furthermore, the cost of an offshore intake would be wholly disproportionate given that Duke's short-term lease with the Port of San Diego expires in 2009. At the present time there are no plans to continue the operation of the existing facilities or replace them with newer equipment after 2009.

Eight different behavioral technologies were evaluated. Of these only sound has been recently proven for a number of similar locations for impinged species. Though sound technology is still somewhat experimental in nature, it was evaluated at the second step. An ultrasound system is a relatively low cost technology to reduce impingement. An ultrasound system's lack of moving parts makes it generally maintenance free, except for inspections and cleaning of the sound transducer surfaces. System performance can be monitored automatically via underwater receivers and bioacoustical instrumentation. Ultrasound technology should be explored further as a technology to reduce the SBPP intake's impingement of fishes. Several studies and a growing body of research suggest that a properly designed ultrasound system could reduce SBPP's potential to impinge some pelagic fish species.

Thirteen different physical barrier screen technologies and two different fish diversion systems were evaluated for their potential to reduce entrainment and impingement. Of these, four of the screen technologies and the two fish diversion systems were determined to be proven and available. The costs for all of these technologies would be wholly disproportionate to any environmental benefit gained given that Duke's short-term lease with the Port of San Diego expires in 2009. In addition, these technologies trade decreases in impingement of larger organisms for increased environmental impacts on other life stages, sizes, or types of organisms and therefore do not represent BTA for the SBPP intake.

We recommend that the existing fish return system be upgraded to reduce bird predation and that the trough be extended so that it returns impinged organisms into deeper water.

The existing shoreline vertical traveling screen design represents the best technology available. This conclusion is based on the finding of relatively insignificant entrainment and impingement effects (including no population-level effects) and consideration of various demonstrated alternative technologies, including potential biological effectiveness for further reducing entrainment and impingement losses, engineering feasibility, and cost-effectiveness, as outlined in the guidance manual (USEPA 1977).

1.0 Introduction and Background

1.1 Introduction

The South Bay Power Plant (SBPP), operated by Duke Energy, is located on the southeastern shore of San Diego Bay in the city of Chula Vista, approximately 16 km (10 miles) north of the U.S.-Mexican border. The SBPP circulates water it withdraws from San Diego Bay once through the power plant's cooling water system to condense freshwater steam used in power production. After passing through the plant, the circulating water is returned to the bay through a discharge channel.

Since the quantity of cooling water withdrawn by the power plant exceeds 50 mgd, the intake location, design and capacity is subject to a cooling water intake technology regulation in the Clean Water Act described in Section 316(b) and commonly referred by its section name. Although Section 316(b) is described in a single short paragraph, its implementation has been expanded in guidance provided by EPA and various precedent rulings by the Administrator and from the court. Most recently a law suit by the Riverkeepers forced the court to order EPA to promulgate new rules. At the heart of the case was the contention that the plain language of 316(b) of the Clean Water Act required the permittee to minimize the environmental impacts of their cooling water withdrawals through the use of intake technology and did not allow for mitigation, such as restoring the production values of degraded marsh habitat, to offset impacts of the cooling water system intake. Promulgation of the rule proved difficult, causing EPA to seek the court's relief through a bifurcation of the new rule. The court agreed to separate the rules for proposed new power plants from rules for existing power plants.

Study plans for the present renewal of the SBPP NPDES permit were based on information requirements described in the draft rules for both new and existing power plants that were available in September 2002. The new rules called for detailed studies of impinged and entrained organisms, their source water populations, an assessment of the potential population-level impacts, value of any lost resources, an inventory of available and feasible intake technologies that would cost effectively reduce impacts from SBPP, and an analysis of the benefits of such technologies. All of these requirements of the proposed new rules, as well as additional specific information requested by the RWQCB, have been provided for and met in the present SBPP renewal studies, except one aspect of the new rule for existing facilities that finds the cost of EPA-standard intake technology at the facility would exceed its environmental benefits. In this case, the owner would propose to mitigate the facilities' intake effects with offsetting production from the restoration or preservation of habitat relevant to the entrained and impinged resources.

The present 316(b) assessment completes all of the new rule information and analysis requirements, finds that the cost of alternative intake technologies exceed their environmental benefit, but stops short of the new rule's required restoration alternative to offset intake effects. EPA issued its new rule for existing facilities in the midst of final preparations of this report. While all of the analyses to respond to benefits evaluation of restoration are at the ready, the Riverkeepers have sued the EPA again to prevent the use of habitat restoration to offset power plant intake effects described in the new rules for *new* facilities. It is therefore likely that Riverkeepers will also address the court on the same issue of restoration found in the new rule for *existing* facilities imminently due for publication in the Federal Register. However, if the new rules are published as anticipated, they allow several of years for compliance that might also be extended to accommodate the court's interest and any modification to the rules. It is reasonable to anticipate that the RWQCB will reflect both the new rule's uncertainties and allow by provision of the renewed NPDES permit an appropriate period for Duke Energy to comply.

Finally, as frequently noted in this report's cost-benefit analyses of alternative intake technologies for the SBPP, Duke Energy has only a few years remaining in their lease agreement with the San Diego Unified Port District to operate SBPP. At the present time, Duke Energy has no plans to continue operating the existing facility beyond November 2009. Duke Energy, in concert with many other representatives of the South Bay communities are actively investigating plans for a modernized power facility for the property. Duke Energy has clearly stated that any new facility that they might build to replace the existing facility would not have the need for a seawater intake. It should also be noted that the actual closure of SBPP is subject to approval by the State's Independent System Operator (ISO).

SBPP is seeking to renew its National Pollutant Discharge Elimination Permit (NPDES), issued by the California Regional Water Quality Control Board (RWQCB) – San Diego Region. In 2002, the RWQCB issued Tentative Order No. R9-2002-0022. In the order, RWQCB staff concluded that some of the previous studies of the power plant's intake and discharge effects on the water quality and biological resources of south San Diego Bay might be outdated and may not reflect current plant operations or be representative of existing conditions. A letter dated May 24, 2002 from the Board's executive director to Duke Energy South Bay LLC described several open issues regarding the effects of the power plant's intake and discharge systems. The studies described in the Board's directive were designed to address these open issues and to collect additional information on present conditions in the power plant's circulating water discharge and source water areas. The updated information forms the basis for continuing the NPDES permit renewal process for SBPP Permit Number CA0001368.

In addition to addressing the issues raised by the RWQCB, the studies were designed to fulfill requirements of the federal Clean Water Act (CWA) Section 316(a) for discharge effects and Section 316(b) for intake effects. The study design, sampling and laboratory processing methodologies, data, and assessment of impacts from these Section 316(a) and 316(b) studies are presented in two volumes. Volume I is an assessment of the effects of the circulating water discharge system, which fulfills CWA Section 316(a) requirements, and Volume II is an assessment of the effects of the circulating water intake system, which fulfills CWA Section 316(b).

This report, Volume II, addresses the questions related to potential impacts associated with the SBPP circulating water intake structure (CWIS). Compliance with CWA Section 316(b) requires that "the location, design, construction, and capacity of circulating water intake structures reflect the best technology available for minimizing adverse environmental impact" (EPA 1977). Because no single intake design can be considered to be the best technology available at all sites, compliance with the Act requires a site-specific analysis of intake-related organism losses and a site-specific determination of the best technology available for minimizing those losses. In this report, intake-related losses resulting from entrainment (the drawing of organisms into the CWIS) and impingement (the retention of organisms on the intake screens) are evaluated and discussed. Intake technologies are evaluated according to operating, engineering, economic, biological, and other environmental criteria to determine if the existing circulating water system represents the best technology available (BTA) for minimizing entrainment and impingement losses. The companion volume (*Volume I: 316(a) Demonstration for the South Bay Power Plant*) presents an updated analysis of the effects of the SBPP discharge on the biota of the receiving waters in south San Diego Bay.

1.2 Background

The SBPP is a gas and oil fueled generating plant located in south San Diego Bay, California near the U.S.-Mexico border. The plant has four major steam cycle units with a net generating capacity of 723 megawatts electric (MWe). Each unit can generate independently or in conjunction with any other unit. Generation typically cycles on a daily basis in response to demand for electricity. A complete description of the SBPP generating facility and characteristics of the surrounding bay environment are presented in Section 2.1 *Description of the South Bay Power Plant's Circulating Water System*, and Section 2.2 *San Diego Bay Environmental Setting*.

1.2.1 Regulatory Setting

Section 316(b) of the Clean Water Act regulates circulating water intake structures and requires that "the location, design, construction, and capacity of circulating water intake structures reflect the best technology available [BTA] for minimizing adverse environmental impact [AEI]. The Clean Water Act Statute does not specify required CWIS technologies or the methods by which EPA must make its determinations under Section 316(b). NPDES permit conditions imposed under 316(b) to satisfy the statute may be based either on applicable regulatory guidelines or, in their absence , on case-by-case best professional judgment determinations. To make Section 316(b) decisions, permit writers have relied on other cases and on EPA's (1977) informal draft guidelines "Guidance for Evaluating the Adverse Impact of Cooling Water Intake Structures on the Aquatic Environment: Section 316(b) P.L. 92-500." As explained in the introductory remarks of the present section, the new rules for existing facilities call for detailed studies of impinged and entrained organisms, their source water populations, an assessment of the potential population-level impacts, value of any lost resources, an inventory of available and feasible intake technologies that would cost effectively reduce impacts from SBPP, and an analysis of the benefits of such technologies.

In California, the State Water Resources Control Board (SWRCB) and the RWQCB are authorized to implement the Section 316(b) requirement. As is clear from the statute, the permit writer must consider two basic issues in making a finding that an intake technology meets BTA criteria for minimizing AEI:

1. Whether or not an AEI is caused by the intake and, if so,
2. What intake structure represents BTA to minimize that impact.

In response to the May 24, 2002 letter from the San Diego RWQCB, an alternative technology assessment is a part of this demonstration report (see Section 6 –

Technological, Design, And Operational Alternatives To Minimize Adverse Environmental Impacts).

1.2.1.1 Adverse Environmental Impact (AEI) Standard

Since there are no regulations defining AEI, permit decisions are made on a case by case basis. In several guidance documents issued since the 1970s, the EPA has indicated that assessment of AEI should be based on an evaluation of population level effects, not just losses of individual organisms. In its 1975 Draft BTA Guidelines, the EPA stated that "[a]dverse environmental impacts occur when the ecological function of the organism(s) of concern is impaired or reduced to a level which precludes maintenance of existing populations...". Additionally, in the 1976 Development Document, released in conjunction with the EPA's previous Section 316(b) rules, the EPA said that "[t]he major impacts related to cooling water use are those affecting the aquatic ecosystems. Serious concerns are with population effects that…may interfere with the maintenance or establishment of optimum yields to sport or commercial fish and shellfish, decrease populations of endangered organisms, and seriously disrupt sensitive ecosystems." A precedent-setting study of SBPP CWIS effects in 1979–1980 (SDG&E 1980) demonstrated no appreciable harm to fish or invertebrate populations in San Diego Bay based on extensive entrainment and impingement collections.

1.2.1.2 Best Technology Available (BTA) Standard

The second issue to be considered in making a Section 316(b) decision is whether the existing intake structure represents BTA to minimize adverse environmental impacts if they are occurring. Determination of BTA for any circulating water intake requires the following:

- consideration of the technical and engineering feasibility of alternative intake technologies,
- the potential for an intake technology to reduce or eliminate the "adverse environmental impact,"
- the potential for the technology to produce other environmental impacts reducing its net benefit, and
- the cost of the technology in relation to its potential environmental benefits.

Although no reference to cost is made in Section 316(b), legislative history suggests that Congress intended that costs be considered in 316(b) determinations. Specifically, a statement by the spokesman for the House Conferees indicates that Congress intended the "best technology available" to be interpreted to mean the "best technology available commercially at an economically practicable cost." Additionally, in responding to comments during the drafting of its 1977 Draft Guidance, the EPA said that BTA is the technology or group of technologies that minimize adverse impacts to the greatest possible degree at a cost that is not "wholly disproportionate" to the environmental benefits. This standard was also applied by the EPA Regional Administrator in the Pilgrim decision that states, "a decision regarding the required degree of minimization calls for a determination that the costs involved are not wholly out of proportion to the adverse environmental impact being avoided".

1.2.2 Effects of Impingement and Entrainment: Overview

The withdrawal of water by once-through circulating water systems has two major effects on the biological resources of the source water body: impingement and entrainment (**Figure 1.2-1**). Most circulating water systems employ some type of screening device to block large objects from entering the circulating water system. Fishes and other aquatic organisms large enough to be blocked by the screens may become impinged on the screens if the intake velocity exceeds their ability to move away or if they become entangled in debris that may be present in front of the CWIS. These organisms will remain impinged against the screens until the intake velocity is reduced so the organisms can move away or the screen is rotated and backwashed to remove them into a fish return trough. Some organisms are killed, injured, or weakened by impingement and others

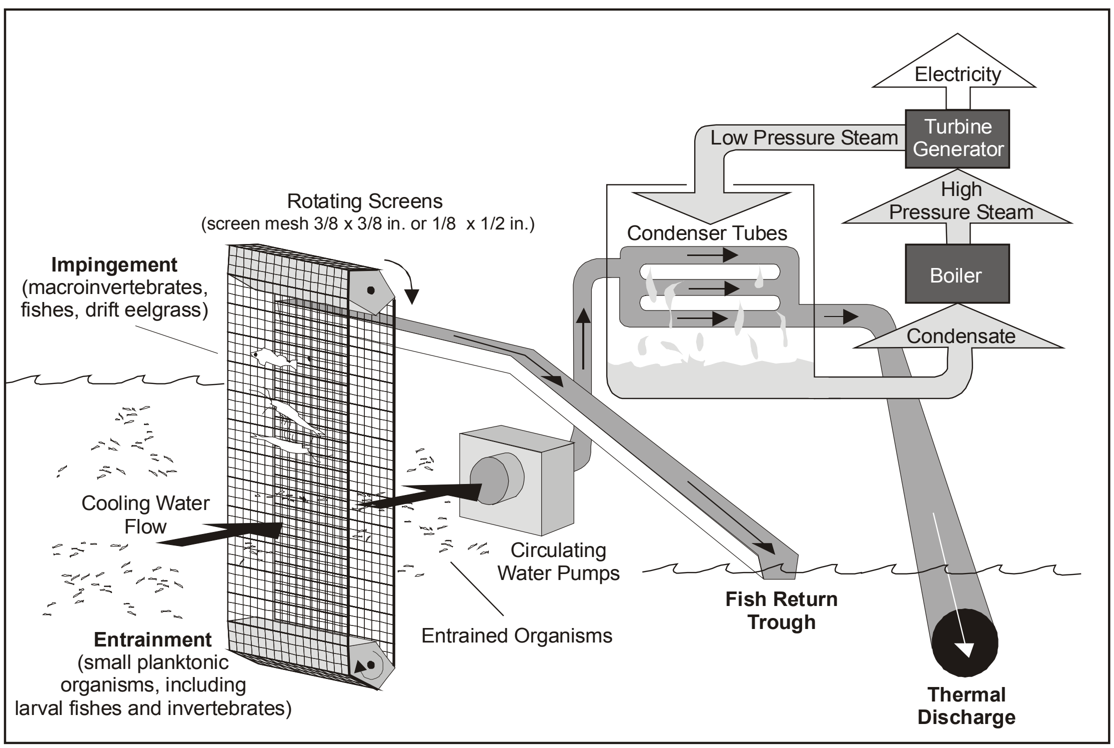


Figure 1.2-1. Conceptual diagram of impingement and entrainment processes and their relationship to the circulating water system at SBPP.

survive. Small planktonic organisms, including early life stages of larger organisms, pass through the mesh of the screen are entrained in the circulating water flow. These organisms are exposed to velocity and pressure changes due to the circulating water pumps, increased temperatures and, in some cases, chlorine exposure through the plant's condenser tubes. Although most individual organisms are killed by passage through the CWIS, the ultimate goal of the studies is to determine if effects are significant at the population level for the affected species. The additional mortality rates imposed by the CWIS on the high natural mortality rates of early life stages in most species typically cannot be measured in the natural population due to high natural variability in the ecosystem.

The effects of impingement and entrainment were studied at the SBPP in 1979–1980 (SDG&E 1980). The study also included sampling to characterize the biological resources in the source water of the region of the south bay near the plant. The primary focus of the study was to determine if the designation of the plant as a "high" impact intake system by the State was valid.

Impingement losses during the previous study were characterized for groups of critical taxa: slough anchovy *Anchoa delicatissima,* topsmelt *Atherinops affinis*, California halibut *Paralichthys californicus*, round stingray *Urolophus halleri*, specklefin midshipman *Porichthys myriaster*, and striped bass *Morone saxatilis*. Abundances of impinged invertebrates were not reported. A total of 13,335 fishes with an aggregate wet weight of 853 kg (1,881 lb) were sampled in 150 separate 24-hr impingement collections from February 1979 through January 1980. When impingement losses were compared with source water population abundances for the critical taxa, it was estimated that plant-induced losses comprised between 0.03 and 0.96 percent of source water (central and south San Diego Bay) populations during the 1979–1980 period. These losses were deemed insignificant in an ecological context. A very small fraction of the impinged fishes (0.3 percent) were commercially important species, and dollar losses to the fishery were not calculated. Impingement losses were also compared to natural mortality estimates for these species and were found to be insignificant.

It was determined that intake approach velocities at the screenwells were low enough to allow most fishes to avoid impingement by continuous or burst swimming. The report concluded that the biological impact of SBPP was insignificant in terms of impingement losses.

The original entrainment study was conducted using pump sampling for plankton at the intake structure and net sampling of plankton at three source water stations in central and southern San Diego Bay (SDG&E 1980). Entrainment effects on zoo- and ichthyoplankton were evaluated using comparisons of near-field and far-field densities, near-field and entrainment densities, and entrainment losses and source water resources. Statistically significant lower densities of zoo- and ichthyoplankton were detected in near-field samples when compared to far-field samples. This difference was attributed to either a localized effect of the plant or habitat differences between the areas. Entrainment losses for both zoo- and ichthyoplankton were low relative to the estimated source water standing stock, and were therefore not considered significant. The study concluded that the low level of impacts at all trophic levels did not support the State's designation of the plant's circulating water system as "high" impact, and that the intake system represented the best technology available for minimizing adverse environmental impacts.

1.3 Study Design

The study design for the SBPP circulating water intake technology evaluation required under Section 316(b) of the federal Clean Water Act was developed in cooperation with representatives of the San Diego RWQCB, California Department of Fish and Game, U.S. Fish and Wildlife, NOAA Fisheries and other interested parties. The study design was based on a survey and compilation of available background literature, results of recently completed SBPP intake studies, and circulating water system studies at other power plants.

Entrainment data presented in this report were collected from January 2001 through October 2003. From January 2001 through January 2002, entrainment and source water plankton net sampling was conducted monthly at both the intake station and at an array of source water stations to collect data for impact models that are used to update the previous 316(b) Demonstration study. An additional set of entrainment and source water samples were collected from the same stations every other month from December 2002 through October 2003. These entrainment and source water studies were designed to answer the following questions:

- What are the species composition and abundance of larval fishes, cancer crabs, and spiny lobsters entrained by the SBPP?
- What are the estimates of local species composition, abundance and distribution of source water stocks of entrainable larval fishes, cancer crabs, and spiny lobsters in southern and south-central San Diego Bay?

Field data on the composition and abundance of potentially entrained larval fishes, *Cancer* spp. megalopae, and larval spiny lobster *Panulirus interruptus* provides a basis to estimate the total number and types of these organisms passing through the power plant's CWIS. For the purposes of modeling and calculations, through-plant mortality was assumed to be 100 percent.

The purpose of this 316(b) impingement study is to characterize the juvenile and adult fishes and selected macroinvertebrates (e.g., shrimps, crabs, lobsters, squid, and octopus) impinged by the power plant's CWIS. The sampling program was designed to provide current estimates of the abundance, taxonomic composition, diel periodicity, and seasonality of organisms impinged at SBPP. In particular, the study focuses on the rates (i.e., number or biomass of organisms per cubic meter of water flowing per time into the plant) at which various species of fishes and macroinvertebrates are impinged. The impingement rate is subject to tidal and seasonal influences that vary on several temporal scales (e.g., hourly, daily, and monthly), while the rate of circulating water flow varies with power plant operations and can change at any time. A review of the previous impingement study at SBPP in 1979–1980 provides context for interpreting changes in the magnitude and characteristics of the present day impingement effects. Studies of the south bay fish assemblages independent of SBPP (e.g., Allen 1999) also provides information regarding the marine environment in southern and central San Diego Bay.

1.4 Report Organization

Section 2.0 provides a description of the SBPP and source water body characteristics. Section 3.0 describes the field collection, laboratory processing, and data analysis methods for the entrainment study, and presents results and impact assessments of target taxa due to entrainment. Section 4.0 describes the methods, results and impact assessment for target impingement taxa. Section 5.0 presents a synthesis of CWIS entrainment and impingement impact assessments, and Section 6.0 provides an evaluation of alternative intake technologies for SBPP.

Four appendices are also included with this report. The calculation of the source water volume for SBPP used in the ETM modeling procedure is explained in Appendix A. Details on the variance calculations for parameters used in the ETM model are explained in Appendix B. Complete summaries of the entrainment and source water sampling for fish larvae, California spiny lobster larvae, and *Cancer* crab megalopae are presented in Appendix C. Appendix D contains the weekly impingement results and annual impingement estimates for taxa enumerated in the sampling.

2.0 Description of South Bay Power Plant and Characteristics of the Source Water Body

2.1 Description of the South Bay Power Plant's Circulating Water System

2.1.1 Intake System

The South Bay Power Plant (SBPP) uses the waters of San Diego Bay for once-through cooling of its four electric generating units. Each unit is supplied by two circulating water pumps (CWP). Individual pump output varies between units, ranging from 148 m^3/min to 259 m^3/min (39,000 gallons per minute [gpm] to 68,400 gpm) based on the manufacturer's pump performance estimates. The quantity of circulating water circulated through the plant is dependent upon the number of pumps in operation (**Table 2.1-1**). With all pumps in operation, the circulating water flow through the plant is 1,580 m^3/min (417,400 gpm) or 2,275,000 m^3/day (601 million gallons per day [mgd]).

Table 2.1-1. Generating capacity and circulating water flow volumes of the South Bay Power Plant.

	Gross Generation	Flow from two CWP/unit	
Unit	(MWe)	(m^3/min)	(gpm)
1	152	295	78,000
2	156	295	78,000
3	183	472	124,600
4	232	518	136,800
Total	**723**	**1,580**	**417,400**

Circulating water is withdrawn from San Diego Bay via an intake channel that connects the SBPP with the southeast corner of the bay (**Figure 2.1-1**). The intake channel is about 180 m (600 ft) in length and has a bottom width of about 60 m (200 ft) at its widest point and then tapers to 15 m (50 ft) near the Unit 4 screenhouse. The maximum depth of the channel is approximately –5.4 m (–17.7 ft) mean lower low water (MLLW). The channel was constructed by dredging and diking operations during plant construction in the early 1960s. This dredged material was used to form part of the Chula Vista Wildlife Island that separates the intake and discharge channels. Variations in water level attributable to the tides, range from a low of –0.7 m (–2.3 ft) to a high of +2.5 m (+8.2 ft) MLLW.

The circulating water intakes utilized by the SBPP consist of three separate screenhouse structures for its four units. Units 1 and 2 share a single screenhouse structure while Units 3 and 4 have their own individual screenhouses. A floating boom has been deployed across the intake channel upstream of the screenhouses to stop floating debris and prevent it from entering the screenhouses. In the past, the plant has deployed a 1-inch mesh debris net across the channel during periods of high eelgrass and debris loading. The net was routinely deployed during the summer months from 1982 through 1986, but is now only used during periods of extraordinarily high debris influxes. As shown in **Figure 2.1-1**, water flow within the intake channel first approaches the screenhouse serving Units 1 and 2. The Unit 3 screenhouse is located an additional 40 m (131 ft) downstream, and the Unit 4 screenhouse another 28 m (92 ft) away, near the head of the channel.

Circulating water enters the screenhouses through stationary trash racks. The racks consist of vertical steel bars on 89-mm (3.5-in) centers with 76 mm (3.0 in) spacing between bars. The racks prevent larger organisms such as marine mammals and sea turtles from entering the system and screen out any large debris that could damage the traveling water screens and CWPs located behind the racks. Each screenhouse is equipped with one traveling water screen (TWS) for each CWP. These are vertical “thru flow” TWS. Water passes through vertical, ascending, rectangular trays or frames that support panels of stainless steel screen. Screen mesh size is either 9 mm (3/8 in) square or 3 mm by 13 mm (1/8 in by ½ in) rectangular depending on the TWS. Debris is impinged upon the screen mesh and carried upward, out of the water, with the ascending panels. As each panel reaches the top of its circuit through the TWS, debris is removed from the screen by high-pressure water spray. The panel then descends the backside of the TWS, completing its circuit. Debris washed from the screens by the water spray enters a trough that flows to the discharge basin near the point of discharge for Units 1 and 2. The screens are

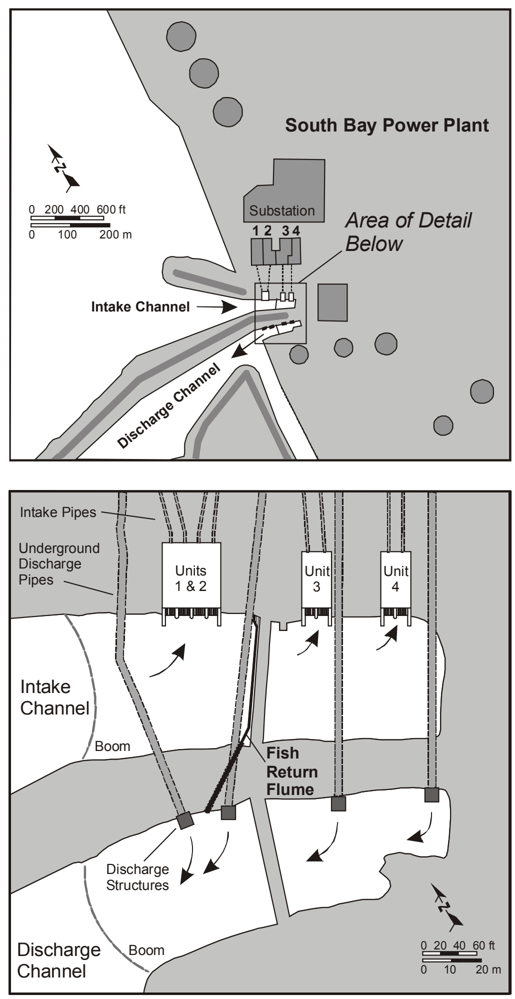


Figure 2.1-1. Diagram of SBPP circulating water intake and discharge structures.

automatically placed in operation when the build up of debris causes the pressure differential across the screen to reach a preset threshold. Water velocity through the TWS was calculated based on the cross sectional area of the submerged portion of the screens and the manufacturer's pump performance estimates (SDG&E 1980). Assuming clean traveling screens, the approach velocity through the Unit 1 and 2 TWS was estimated at 0.12 meters per second (mps) (0.4 feet per second (fps)) at high tide and 0.27 mps (0.9 fps) at low tide. Water velocity through the Unit 3 and 4 TWS was estimated at 0.21 mps (0.7 fps) at high tide, and 0.43 mps (1.4 fps) and 0.46 mps (1.5 fps) respectively, at low tide.

Directly behind the TWS are the circulating water pumps. Circulating water from the Unit 1 and 2 CWPs exits the screenhouse via four 122 cm (48 in) diameter conduits that carry the flow approximately 61 m (200 ft) to the units' condensers. Intake conduits for Units 3 and 4 (one for each CWP) are 152 cm (60 in) in diameter, and also 61 m long. At each of the condensers the circulating water is dispersed through several thousand thin walled condenser tubes. Units 1, 2, and 3 have dual pass condensers that direct the circulating water through the condenser twice. Unit 4's condenser is a single pass design. The Unit 1 condenser tubes are constructed of AL-6X, a stainless steel alloy, while the other condensers are copper-nickel tubes. Exhaust steam, exiting the plant's turbines, passes over the exterior of the tubes and is condensed by the circulating water flowing within the tubes. The resulting condensate is pumped back to the plant's boilers as part of the continuing steam cycle, and the circulating water exits the condenser as heated effluent. The change in circulating water temperature, or delta T, that occurs during passage through the condenser will vary with plant load and can also be affected, to a lesser degree, by condenser tube fouling, tube blockage (caused by debris), and fluctuations in circulating water flow caused by tidal shifts or degradation of CWP performance. Detailed information regarding the discharge system is found in *Volume 1: South Bay Power Plant 316(a) Thermal Discharge Assessment Report.*

2.1.1.1 Flow Volumes

The SBPP discharge (and intake) flow data were derived from the plant operator's daily logs. The logs specify which pumps were in operation for each hour of the day and usually, but not always, when a pump was started or stopped. For NPDES reporting purposes, and for the report, pump operation was rounded to the nearest hour (e.g., if a pump was shut off ten minutes into the hour, it was considered off for the entire hour; if it was shutoff 31 minutes after the hour, it was on for the entire hour). Pump output is based on manufacturer's pump curves. The volume of circulating water utilized by SBPP is dependent upon the number of CWPs that are in operation at any given time. Although the pumps are designed to operate at a constant motor speed and discharge volume, actual pump performance can be affected by changes in tide height, occlusion of the circulating water conduits by biofouling, and clogging of the condenser tubes by biofouling organisms or debris. Maximum volume with all eight pumps in continuous operation is

2,275,164 m^3/d (601.1 mgd). Daily average flow for the period from December 1, 1998 through September 30, 2003 ranged from 425,056 m^3/d (112.3 mgd), which represented the equivalent of both of the smaller Unit 1 or Unit 2 CWPs operating for 24 hours, to 2,275,164 m^3/d (601.1 mgd), which represented the continuous operation of all eight pumps (**Figure 2.1-2**). Maximum discharge volume occurred much more frequently between December 1998 and the end of 2000. Since that time, a decline in demand for

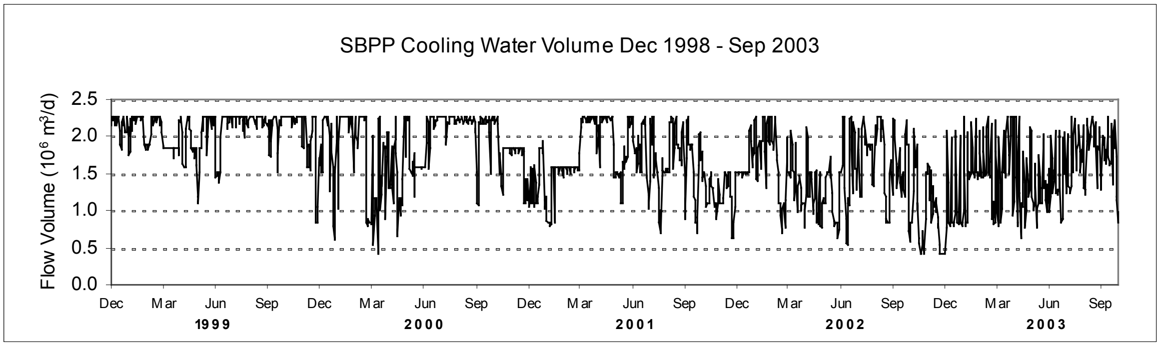


Figure 2.1.2. SBPP daily average circulating water flow from December 1, 1998 through September 30, 2003.

electricity from SBPP and the consequent reduction in generation have reduced the frequency of full flow operation periods. Unit 4 in particular saw limited use in 2002 and 2003. During 2003, SBPP operated all eight CWPs for a period of about 24 hours per week to accommodate the impingement sampling conducted as part of the 316(b) studies described in this report. As a result, the circulating water volumes for this period are more variable than those from 1999–2002.

2.1.1.2 Biofouling Control

Biofouling occurs when organisms such as microscopic plants (algae), some invertebrate species, and minute organisms colonize the circulating water system. Biofouling organisms prey on entrained species as they pass through the circulating water systems of power plants. Colonization of these organisms in the pipes and conduits of the power plant can cause loss of generation. Biofouling can be split into two general categories, microfouling and macrofouling.

SBPP uses chlorine injection to prevent or inhibit microfouling on the heat transfer surfaces of its condensers and ancillary heat exchangers. The current discharge limits for total residual chlorine at the point of discharge are dependent upon the number of power generating units that are in operation. The allowable discharge levels decrease as more cooling pumps are operated due to the increase in flow volume added to the discharge. If the cooling pumps for one unit are in operation, the allowable discharge is 144 parts per billion (ppb) of total residual chlorine (TRC). If 2, 3, or 4 units are in operation, the allowable discharge concentration is reduced to 111 ppb, 95 ppb, or 85 ppb TRC, respectively.

Chlorination of the cooling water system is intermittent rather than continuous. Treatments occur for a duration of 20 minutes, six times per day. Injection cycles are evenly spaced, occurring every four hours. During each cycle, half of the cooling water system of each generating unit is treated. This allows the SBPP to maintain chlorination levels that are adequate to effectively control microfouling organisms, but still remain within permissible discharge levels due to dilution and mixing with the water from the untreated half of each unit's cooling water system. The actual amount of chlorine added to each unit's cooling water system is unique because the systems have different flow capacities. Generating Units 1 and 2 have cooling water pumps that deliver 147,600 liters per minute (lpm) (39,000 gallons per minute (gpm)). Generating Unit 3 is cooled by pumps that supply 236,200 lpm (62,400 gpm), and generating Unit 4 has a cooling water flow of 258,900 lpm (68,400 gpm). Reported values are per pump, and each generating unit is cooled by two cooling water pumps. To remain within the limits of its NPDES permit, the SBPP currently uses 0.68 lpm (0.18 gpm) of chlorine for Units 1 & 2, 0.76 lpm (0.20 gpm) for Unit 3, and 1.02 lpm (0.27 gpm) for Unit 4. Treatment of cooling water systems only occurs during the scheduled intervals and is restricted to those generating units that are in operation. A description of TRC monitoring at SBPP is presented in *Volume 1: South Bay Power Plant 316(a) Thermal Discharge Assessment Report.*

SBPP uses mechanical cleaning as the principal means to control macrofouling within its circulating water systems. The intake conduits are cleaned approximately once a year by divers. The condenser waterboxes are cleaned of growth and debris as needed based on trends in cross-condenser differential pressure.

2.1.2 Discharge System

Upon exiting the condensers, circulating water from the four units is carried, via four individual pipes, to the discharge basin, located at the head of the discharge channel. The average travel time from the point of intake to the point of discharge is approximately two minutes. The discharge channel originates on the side of the jetty, opposite the head of the intake channel. A complete description of the discharge is included in *Volume 1: South Bay Power Plant 316(a) Thermal Discharge Assessment Report.*

2.2 San Diego Bay Environmental Setting

2.2.1 Physical Description

San Diego Bay is the largest estuary between San Francisco Bay and Baja California. The bay is relatively long and narrow, 25 km (15.5 mi) in length and 1–4 km (0.6–2.4 mi) wide, forming a crescent shape between the city of San Diego to the north and Coronado Island/Silver Strand to the south. The bay is separated into two distinct topographic regions, the outer bay, which is generally narrow and deep, and the inner bay, which is wide and shallow. Exchange with the ocean is limited to a single channel at the mouth. This north-south oriented channel is about 1.2 km (0.7 mi) wide, with depths between 5–15 m (16.4–49.2 ft) (SDUPD 1976).

San Diego Bay, like other tidally-influenced waters in California, has a mixed (diurnal plus semidiurnal) tide with the semidiurnal tide being the larger of the two. The diurnal tidal range is 5.5 ft (1.67 m) at Ballast Point near the north end of the bay and increases to 5.9 ft (1.8 m) near SBPP (**Appendix A**). The tidal prism of the bay (volume between MLLW and Mean Higher High Water [MHHW]) is approximately 7.6 x 10^7 m^3 (6.0 x 10^4 acre feet) (**Appendix A**).

Tidal currents can be reasonably strong near the entrance of the bay, up to 1.0 meters per second (mps) (3.3 feet per second [fps]), yielding an average tidal excursion (distance traveled by a parcel of water in one tide) of approximately 4.3 km (2.7 mi) (Chadwick et al. 1996a). The head of the bay (the South Bay region) is closed and without substantial tributaries. Thus, the horizontal motion of the tide near SBPP is small, with weak currents of approximately at 0.1–0.2 mps (0.3–0.6 fps). Because of the weak tidal currents near the head of the bay, the flushing and residence time of the bay are controlled by the two-layer estuarine circulation. An absence of freshwater inflow means that this estuarine circulation is also weak much of the year, with the residence time being on the order of one month for the innermost parts of the South Bay (**Appendix A**). A detailed analysis of current velocities in the vicinity of SBPP with consideration of the plant intake and discharge flows is presented in *Volume 1: South Bay Power Plant 316(a) Thermal Discharge Assessment Report* Section 2.5 *Receiving Water Currents and Bathymetry* of that report.

San Diego Bay is a low-inflow estuary. Rainfall averages approximately 26 cm (10.2 in) per year and significant freshwater inflow occurs for only a few months during the winter. Because of the low freshwater inflow, considerable solar heating and weak flushing of the head of the bay, bay waters become quite warm and slightly more saline (relative to adjacent coastal waters) in late summer through early winter. A detailed analysis of water temperatures in the vicinity of SBPP is presented in *Volume 1: South Bay Power Plant 316(a) Thermal Discharge Assessment Report* Section 2.3 *Receiving Water Temperature Monitoring*.

The head of the bay is quite shallow (approximately +0.5–2.0 m [+1.6–6.6 ft] deep MLLW), and winds play a role in driving currents that promote flushing. Wind waves up to approximately 0.5 m (1.6 ft) in height play a role in the vertical mixing of the SBPP thermal plume, but the weak horizontal density gradients of the system favor vertical uniformity of the ambient waters of the bay.

2.2.2 Biological Description

The SBPP is located along the southeastern shoreline of San Diego Bay, near the only remaining portions of the area's natural estuarine habitats. The shoreline and bathymetry of the bay have been altered through urbanization, waterfront development, and extensive dredging. The development of San Diego Bay and its use as a naval base and large commercial shipping hub has resulted in water quality changes, as well as the alteration of benthic substrates. Modifications, including fill projects, periodic dredging, and the construction of piers, wharves, and docks have significantly altered the shoreline, as well as intertidal and subtidal habitats.

In the past 70 years the shallow expanses of San Diego Bay and littoral habitats have been largely eliminated from the northern two-thirds of the bay and greatly reduced in the South Bay. Shallow submerged lands have been reduced to 65 percent of their original area in the South Bay (SDUPD 1990). Less than 40 percent of the area originally occupied by intertidal mud flats in the South Bay remains, and salt mashes have been reduced to a few remnant patches (SDUPD 1990). Between 1940 and 1960 chronic pollution of the bay from sewage and industrial discharges greatly reduced the abundance and diversity of species and blanketed large areas of the bottom with sludge (**Appendix A**). Regulation of discharges into the bay initiated during the 1970s has resulted in an improvement in water quality and a gradual recovery of the abundance and diversity of species.

South San Diego Bay is a relatively shallow basin (<3.7 m [<12 ft]), and is characterized by warm water temperatures and sluggish tidal currents. The Otay River flows into the South Bay approximately one mile south of the power plant and the Sweetwater River channel enters the bay about three miles north of SBPP. While San Diego Bay is still considered an estuarine system, freshwater inflow has been nearly eliminated by water diversion, utilization of groundwater, and infrequent runoff (Browning et al. 1973). Allen (1999) defined the South Bay ecoregion as areas of San Diego Bay that lie south of a line drawn in west-southwesterly direction from the Sweetwater River Channel to the Silver Strand State Beach on the San Diego Peninsula. The south central ecoregion extends from the Coronado Bridge south to the boundary of the south ecoregion at the Sweetwater River Channel.

The aquatic habitats in the vicinity of the SBPP are characteristic of a protected inshore marine environment. The flora and fauna of the region generally consists of communities living above, on, and within soft benthic substrates. Benthic substrates are composed mostly of alluvial sediments, including fine-grained sand, silt, and clay. Some expanses of bottom along the western shoreline of the bay are dominated by "cleaner", larger-grained sand (Browning et al. 1973). Because of the absence of freshwater inflow, the plant and animal communities are typical of marine and higher salinity estuarine environments. Aquatic habitats include submerged lands (or subtidal areas), eelgrass beds, mudflats, and salt marshes. Salt evaporation ponds located adjacent to the southernmost reach of the bay provide important habitat for shorebirds and migrating waterfowl.

The Chula Vista Wildlife Refuge adjacent to SBPP is an artificially constructed peninsula that separates the intake and discharge channels of the power plant. The island itself was largely constructed from dredge spoils, and portions of the access causeway are armored with rock rip-rap to prevent erosion. Tidal inlets within the reserve form wetland areas, and adjacent areas provide seasonal habitat for several species of nesting shorebirds, including endangered California least terns *Sterna antillarum browni* and western snowy plovers *Charadrius alexandrinus nivosus*.

2.2.2.1 Submerged Lands

Submerged lands encompass all subtidal and regularly submerged areas of the South Bay. Eelgrass beds are included within the 993 ha (2,454 ac) of area in the South Bay designated as submerged lands. With the exception of dredged channels, depths in the South Bay do not exceed 3.7 m (12 ft), and 57 percent of the acreage is less than 1.8 m (6 ft) (USFWS 1998). Twenty-six percent of the area is reported to be less than 0.9 m (3 ft) in depth (USFWS 1998).

The submerged land area in the South Bay supports over 450 invertebrate species (USDoN and SDUPD, 2000) and about 100 species of fishes, sharks, and rays (Allen 1999, USDoN and SDUPD, 2000). Common invertebrates include various types of worms, gastropod and bivalve mollusks, and crustaceans, but there is little information available pertaining to the current composition and abundance of individual species. Recreationally important fish species found in the South Bay include the barred sand bass *Paralabrax nebulifer*, spotted sand bass *Paralabrax maculatofasciatus*, diamond turbot *Hypsopsetta guttulata*, California halibut *Paralichthys californicus,* black croaker *Cheilotrema saturnum*, opaleye *Girella nigricans*, striped mullet *Mugil cephalus*, and bonefish *Albula vulpes*. Northern anchovy *Engraulis mordax*, two species of bay anchovy *Anchoa* spp., and Pacific sardine *Sardinops sagax* are occasionally abundant in the area. Round stingrays *Urolophus halleri* are known to be abundant in the South Bay (LES 1981) as are several other species of bottom-dwelling sharks and rays.

Allen (1999) conducted a study of the fisheries of San Diego Bay from July 1994 through April 1999. Different types of collection gear were used to allow for sampling in all of the available habitat types throughout the bay. The gear included large and small seines, square enclosures, purse seines, and beam and otter trawls. Allen divided the bay into four ecoregions with SBPP located in his "south region" (Allen 1999). In the south region the abundance of fishes was dominated by slough anchovy (66 percent), topsmelt (14 percent), arrow goby (3 percent), round stingray (3 percent), northern anchovy (2 percent), and shiner surfperch (2 percent). Based on the biomass of each fish taxa collected, the samples were dominated by round stingray (37 percent), followed by spotted sand bass (13 percent), bay ray (10 percent), barred sand bass (8 percent), slough anchovy (8 percent), and topsmelt (7 percent). Allen (1999) estimated that the total standing stock of the fishes in the bay's south ecoregion was about 79,000 kg (7.42 g/m^2). Allen (1999) also found that the fish composition and relative abundance in this area was similar to that reported by SDUPD (1990) from an earlier study conducted in 1988–1989.

A study of the fish community in the SBPP circulating water discharge channel was conducted quarterly from April 1997 through January 2000 (Merkel and Associates 2000a). Over 176,000 individuals representing 38 species were collected during this study. Numerically, the catch was dominated by slough anchovy (91.4 percent), followed by deepbody anchovy (1.4 percent), round stingray (1.1 percent), and topsmelt (1.0 percent). Density and biomass varied between the two discharge stations and also between seasons and years. Merkel and Associates (2000a) estimated that the mean biomass of the fishes, shark, and rays collected in the discharge channel for their entire study was 5.48 g/m^2.

Submerged lands in the South Bay are also an important resting and feeding area for waterfowl migrating along the Pacific Flyway. Surveys by USFWS during 1993–1994 found almost 200,000 birds at a time utilizing the habitat available in the South Bay (USFWS 1998). Common waterfowl include the surf scoter *Melanitta perspicillata*, scaup *Aythya* spp., black brant *Branta bernicla nigricans*, bufflehead *Bucephala albeola*, loons *Gavia* spp., and western grebe *Aechmophorus occidentalis*. Seabirds, such as gulls *Larus* spp. and cormorants *Phalacrocorax* spp., are also common in the area. Additionally, a number of listed (endangered and threatened) bird species, and species of special concern, have been observed in the South Bay. Bird species in the area that are protected under state or federal law include the California least tern, western snowy plover, brown pelican *Pelecanus occidentalis*, peregrine falcon *Falco peregrinus*, and bald eagle *Haliaeetus leucocephalus*.

Three species of marine mammals have been observed in the South Bay. During waterfowl surveys in the South Bay, the USFWS reported observing California sea lions

Zalophus californianus and Pacific bottle-nosed dolphins *Tursiops truncatus* within their study area (USFWS 1998). While most of the observations occurred in the northern half of the area, three bottle-nosed dolphins were observed in the southernmost regions of the bay. Harbor seals *Phoca vitulina* were reported near the discharge channel of the SBPP, possibly foraging for animals attracted to the heated effluent (ESA 1997). All marine mammals are federally protected.

Green sea turtles *Chelonia mydas* are attracted by the warm waters and flow of the discharge and occur in the discharge channel and vicinity of SBPP. Although green sea turtles migrate considerable distances, the SBPP discharge channel is the northernmost Pacific Coast location where they reside with any regularity (Eckert 1994). In 1978 all green sea turtles were afforded protection under the federal Endangered Species Act. Breeding populations of green sea turtles off Florida and Mexico were listed as endangered and all other populations were listed as threatened.

2.2.2.2 Eelgrass

Eelgrass was once abundant throughout much of San Diego Bay before shoreline development and dredging of navigational channels reduced its available habitat area. Now, over 90 percent of San Diego Bay's remaining eelgrass (*Zostera marina)* beds are located in the South Bay (USFWS 1998). The eelgrass beds in the South Bay cover a discontinuous area of approximately 279 ha (690 ac) and have been expanding since the discharge of raw sewage and industrial waste into the region was controlled (SDUPD 1990, USFWS 1998). Eelgrass beds are the most expansive in the vicinity of the Sweetwater Marsh National Wildlife Refuge (NWR), Crown Cove, and an area to the north of Emory Cove. Several dredged navigational channels break up the area of eelgrass coverage in the South Bay. A detailed description of changes in eelgrass cover in South Bay is presented in *Volume 1: South Bay Power Plant 316(a) Thermal Discharge Assessment Report.*

Eelgrass is a flowering marine plant that grows in the shallow, sunlit waters of protected bays and estuaries. Eelgrass forms extensive beds that root in soft benthic substrates. The beds provide cover and spawning substrate for many species of fishes and invertebrates, and are considered important nursery areas. Eelgrass is an important food source for green sea turtles and a variety of seabirds and migrating waterfowl. The black brant, a small migratory goose species, feeds heavily on eelgrass during its migrations along the coast.

Eelgrass is a wide-ranging plant species that occurs along the Pacific Coast of North America, from the Bering Strait south to lower Baja California and around to the Gulf of California. The species typically occurs in water temperatures from 5–27°C (41–80.5°F) (Phillips 1984). Eelgrass grows down to depths of 15.2 m (50 ft) if ample sunlight is present, but water turbidity in an area will limit its growth and the depth to which it occurs. Eelgrass is a marine species and is not tolerant of low salinities.

2.2.2.3 Mudflats

Mudflats are present along two-thirds of the shoreline of the South Bay and are absent only along the western shore in the vicinity of the Coronado Cays (USFWS 1998). Mudflat habitat occupies approximately 199 ha (492 ac) in the South Bay, and adjoins the salt evaporation ponds along its southern margins. The largest expanse of mudflat habitat in the region extends from the southern boundary of Emory Cove around the south end of the bay to the SBPP plant site. Another large expanse of mudflat habitat extends north along the eastern shoreline of the bay from the Chula Vista Boat Yards to the northern boundary of the Sweetwater Marsh NWR.

Mudflats are rich in organic matter and support a diverse assemblage of invertebrates. An extensive assortment of birds and fishes utilize this abundant invertebrate fauna as a primary food source. During low tide, shorebirds, such as the western snowy plover, Belding's savannah sparrow *Passerculus sandwichensis beldingi*, western sandpiper *Calidris mauri*, dunlin *Calidris alpina pacifica*, marbled godwit *Limosa fedoa*, willet *Catoptrophorus semipalmatus*, long-billed curlew *Numenius americanus*, northern phalarope *Phalaropus lobatus*, killdeer *Charadrius vociferus*, American avocet *Recurvirostra americana* and red knot *Calidris canutus*, forage the mudflats during their migrations along the Pacific Flyway. Over 26 species of shorebirds were identified as utilizing the South Bay habitats for a wintering ground (Browning et al. 1973). Sixty-seven species of birds were observed in mudflat and salt pond habitat during USFWS bird counts (USFWS 1998). When mudflats are submerged, a variety of terns (including the federally protected least tern), snowy plover (Species of Special Concern), grebes, and black skimmer *Rhynchops niger* use the habitat to forage for small fishes. These temperature and salinity-tolerant fishes include the California killifish *Fundulus parvipinnis* and two goby species (arrow goby *Clevelandia ios* and longjaw mudsucker *Gillichthys mirabilis*).

2.2.2.4 Saltmarsh

Salt marshes are the driest of the habitats in the South Bay that are influenced by the tides (USFWS 1998). The loss of salt marsh habitat has been particularly extensive due to shoreline development. The 23 ha (57 ac) of salt marsh that remain in the South Bay are distributed among six different locations. The largest patches of salt marsh habitat are located along the Sweetwater Marsh NWR northern boundary, adjacent to the J Street fill, and in the area south of Emory Cove. Small patches also occur in the Chula Vista Wildlife Reserve, the salt evaporation ponds, and along the Otay River channel. Salt marsh habitat not included in the 23 ha (57 ac) estimate, but considered critical, is also present in narrow strips along other tidally-influenced regions of the Otay River and along some portions of the salt pond dikes.

Salt marsh habitat is characterized by low-growing, salt-tolerant vegetation, and is typically dominated by pickleweed *Salicornia* spp. Salt marshes are used by a variety of shorebirds for nesting and feeding, and as escape areas during high tide. A great variety of shorebirds, herons, egrets, rails, and some waterfowl species may frequent small patches of salt marsh habitat. Fifty-seven species of birds were counted during bird surveys conducted by USFWS in a salt marsh along the Otay River (USFWS 1998). Until recently a 91 m (300 ft) stretch of salt marsh habitat along the Otay River supported nesting pairs of the light-footed clapper rail *Rallus longirostris levipes*, which is listed as a federally endangered species (USFWS 1998). Light-footed clapper rails are found exclusively in salt marshes and the extensive loss of this habitat type has coincided with the species' decline. The Belding's savannah sparrow, listed as threatened under state law, nests in salt marsh habitats within the South Bay. Over 100 nesting pairs were observed during USFWS bird surveys.

2.3 Source Water Volume

SBPP draws ocean water from San Diego Bay for once-through condenser cooling. As part of the modeling procedure for determining the impacts of larval losses from transit through the CWIS, it is necessary to specify the volume of the circulating water source. Detailed bathymetry of San Diego Bay was done by the U.S. Navy (1994) and refined for areas of the southern reaches of the bay for the present study (see companion *Volume 1: South Bay Power Plant 316(a) Thermal Discharge Assessment Report; Section 2.5: Receiving Water Currents and Bathymetry*).

Previous studies have defined ecoregions in San Diego Bay. Allen (1999) defined the south and south-central ecoregions as that portion from Coronado Bridge to the southern end of San Diego Bay, including the Sweetwater River channel to a point where it intersects the Interstate 5 interchange.

The calculation of a source water volume used in modeling power plant entrainment effects on fish larvae is presented in **Appendix A**. To summarize, the source volume for the SBPP was defined as the volume of water below Mean Water Level (MWL, the average of a large number of tidal observations) from the southern end of San Diego Bay northward to the Coronado Narrows (**Figure 2.3-1**). Computing the source volume required a compilation (using ArcView GIS software) of areas and volumes below fixed elevations (horizontal strata) and an analysis of currents and tidal dispersion for each source region. Variations in tidal range required that the South Bay be divided into four regions (**Figure 2.3-1**), with tidal datum levels determined for each, either directly from a tide gauge in the region or by interpolation from adjacent gauges. Tide gauges were available in Regions 2, 3 and 4, whereas datum levels in Region 1 had to be determined by interpolation.

Bathymetry for Regions 1 and 2 and the periphery of Regions 3 and 4 (west) were obtained from the U.S. Navy (U.S. Navy 1994). Bathymetry data collected by Merkel and Associates were used for most of Regions 3 and 4 (see companion *Volume 1: South Bay Power Plant 316(a) Thermal Discharge Assessment Report; Section 2.5: Receiving Water Currents and Bathymetry*). These data were collected using a Furuno FCV-600L single-beam fathometer operating at a frequency of 200 kHz. The echosounder was mounted on the port side of the vessel, with the 15º beamwidth transducer located approximately 0.15 m (0.5 ft) below the water surface. Tidal elevation corrections were made using a gauge located on the Broadway Pier, in the northern part of the bay. Tenera Environmental surveyed about 218 hectares adjacent to the discharge of the South Bay Power Plant. This bathymetric survey provided bottom depths of the discharge area with centimeter horizontal and vertical accuracy using a BioSonics 200 kHz digital echosounder (8º beamwidth transducer) with survey-quality base and roving GPS units.

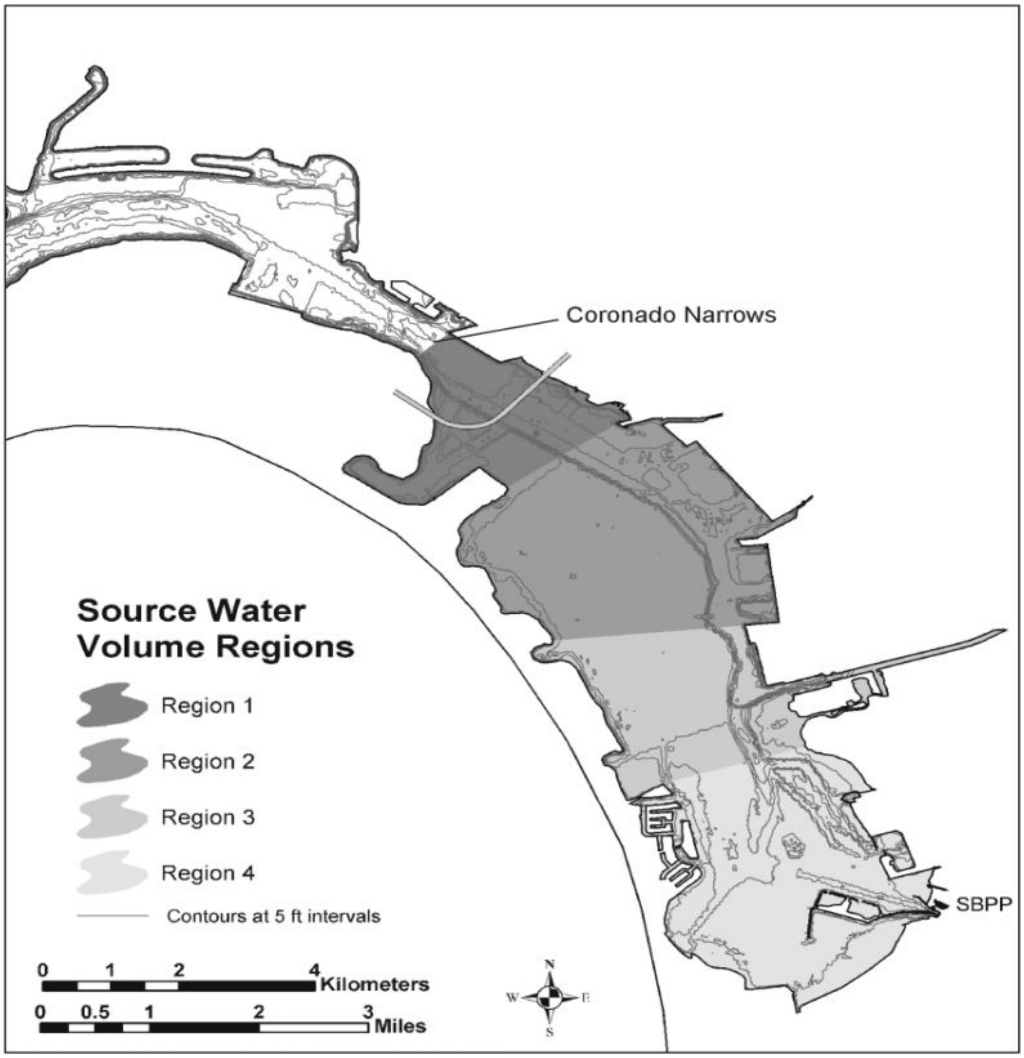


Figure 2.3-1. Source water volume regions of south San Diego Bay used to calculate total source water volumes for SBPP. From Jay and Largier (2003); *see Appendix A this report*.

The analyses in **Appendix A** of current patterns and tidal dispersion justify the definition of the South Bay (south of the Coronado Narrows) as an appropriate source volume for the purposes of modeling larval entrainment. Analyses of tidal currents measured at 18 locations throughout the interior of San Diego Bay showed that tidal currents exhibit a local maximum in the South Bay at the Coronado Narrows and increase toward the bay mouth. Tidal currents are weak in south bay, and mean flows are weak throughout the bay, except at isolated locations. Estimates of tidal dispersion were also formed using data from the same 18 current meters. While spatial patterns are generally similar to those from Largier (1995), there are differences in detail. The measurements presented in **Appendix A** provide superior temporal coverage to earlier studies, but some of the mechanisms (e.g., tidal pumping) found to be important at and seaward of the Narrows were not calculated. **Appendix A** shows that tidal dispersion has a local maximum at the Coronado Narrows, consistent with the idea that the Narrows acts as the "mouth" of

South Bay. The results suggest that larvae are likely removed from South Bay primarily, but not exclusively, by dispersion and that advection may be dominant during winter river-flow events. Such events have not to date been measured. These analyses confirm, in a quantitative manner, earlier definitions of eco-regions in San Diego Bay (Allen 1999, Merkel and Associates 2000b). The Narrows is, therefore, a logical seaward boundary for the SBPP source volume.

The resulting source water region areas and volumes are tabulated in **Table 2.3-1**. The sum of the areas is 30,326,646 m^2 (326,466,189 ft^2). The sum of the volumes used in the entrainment study calculations of mortality for the ETM (empirical transport model) is 149,612,092 m^3 (5,284,269,177 ft^2).

Table 2.3-1. Source water body surface area and water volume at mean water level (MWL) by region.

Region	Datum	Height (MLLW) ft	m	Area m^2	ft^2	Volume m^3	ft^3
1	MWL	2.93	0.90	4,241,241	45,656,798	33,754,018	1,192,185,160
2	MWL	2.94	0.90	10,173,006	109,512,412	70,387,388	2,486,068,457
3	MWL	2.99	0.91	6,355,524	68,417,214	25,060,179	885,120,494
4	MWL	3.05	0.93	9,556,875	102,879,765	20,410,508	720,895,066
			Total	**30,326,646**	**326,466,189**	**149,612,092**	**5,284,269,177**

3.0 Entrainment and Source Water Larval Study

3.1 Introduction

The purpose of the SBPP entrainment and source water studies was to evaluate the potential impacts of the circulating water intake system as required under Section 316(b) of the Federal Clean Water Act (CWA) (USEPA 1977). The San Diego Regional Water Quality Control Board (RWQCB) discussed the need for the additional information with a group of agency representatives and consultants who provided input on the design and implementation of the 316(a) and 316(b) studies at SBPP. The representatives included Duke Energy, Tenera Environmental, Merkel and Associates, San Diego Unified Port District, the RWQCB, California Department of Fish and Game, U. S. Fish and Wildlife Service, NOAA Fisheries, and U.S. EPA. The group members reviewed and commented on several drafts of the 316(b) Cooling Water Intake Effects Study Plan. The group agreed that the entrainment study should focus on early life stages of fishes, *Cancer* crabs, and California spiny lobster *Panulirus interruptus* that could pass through the 9 mm (3/8") mesh traveling screens and be entrained by the power plant's circulating water intake system (CWIS).

The entrainment study was designed to specifically address the following questions:

- What are the species composition and abundance of the larval fishes, *Cancer* crabs, and spiny lobster entrained by SBPP?
- What are the local species composition and abundance of the entrainable larval fishes, *Cancer* crabs, and spiny lobster in the South Bay region of San Diego Bay?
- What are the potential impacts of entrainment losses on larval fish, *Cancer* crab, and spiny lobster populations due to operation of the CWIS?

Plankton samples collected in the intake channel near the SBPP intake structures provided an estimate of the total number and types of these organisms passing through the power plant's CWIS. Data collected from source water surveys were used to estimate the abundance of the larval populations at risk of entrainment. The rationale used to calculate the volume of the source water is presented in **Appendix A**. The two estimates were used to provide an estimate of the fractional loss due to entrainment that can be translated into potential impacts on local fisheries or fish populations.

Many marine organisms have planktonic stages that can be entrained in circulating water intake systems. Particular taxa were selected in this study for further analyses based on their sampled abundance or economic or recreational value. Several approaches, where possible, were used in assessing the CWIS impacts on each taxon to yield more robust and comparable assessments. The three assessment modeling techniques used were Adult Equivalent Loss (*AEL*), Fecundity Hindcasting (*FH*), and Empirical Transport Modeling (*ETM*), which are described in Section 3.2.3 below. For the purposes of modeling and calculations, through-plant mortality was assumed to be 100 percent. Although many marine organisms have planktonic eggs that are also entrained by the power plant's CWIS these were not counted in our samples. Egg mortality was considered in the *FH* assessment model for fishes with planktonic eggs.

Typically, local population estimates for small, non-use (fishes without commercial or recreational fishery value) fishes are not available. The assessments in this study benefited from an extensive five-year study on the fishes of San Diego Bay completed by Allen (1999). This study provided population estimates for the south and south-central areas of San Diego Bay that corresponded reasonably well to the area identified as our source water. The population estimates from Allen's study were used to assess effects on local populations and compare the results among models.

3.1.1 Review of Previous Entrainment Study

Entrainment studies were previously conducted at the SBPP from February 1979 through February 1980 as part of the plant's initial Section 316(b) Demonstration requirement (SDG&E 1980). Pumps were used to sample plankton at the intake structure and nets were used at four source water stations in south-central and southern San Diego Bay. The assessment included zooplankton, as well as ichthyoplankton eggs and larvae.

The study was focused on "critical taxa" that included 14 groups of invertebrates and fishes. The fish larvae included Engraulidae (anchovies), Atherinopsidae (silversides), and Gobiidae (gobies) species complexes, black croaker *Cheilotrema saturnum*, California halibut *Paralichthys californicus*, and diamond turbot *Hypsopsetta guttulata*. Estimates of total annual entrainment were much greater for gobies than any of the other taxa (**Table 3.1-1**).

Table 3.1-1. Total annual entrainment estimates for "critical" fish larvae from 1979–1980 in the SBPP 316(b) Demonstration Study (SDG&E 1980).

Critical Taxa	Annual Entrainment Estimate
Goby species complex	2,200,000,000
Anchovy species complex	180,000,000
Silverside species complex	14,000,000
Diamond turbot	1,400,000
California halibut	420,000
Black croaker	41,000

Source water sampling was done at one station near the plant (near-field) and at three stations further north of the plant (far-field) (one located 4 km north and two located 8 km north). Statistical comparisons of abundances for the "critical taxa" showed no statistically significant differences in larval density between near-field and entrainment stations, but overall species composition and abundance at the near-field station was different from the far-field stations. Differences between near- and far-field stations were attributed to habitat differences as well as possible effects of power plant entrainment.

The study used a stock assessment model similar to the Empirical Transport Model (ETM) used in this study. This model and data from the previous SBPP 316(b) study are presented in MacCall et al. (1983). Comparisons of entrainment losses and source water abundances using this model showed that entrainment resulted in an estimated loss of less than 12 percent of the source water standing stock.

The study concluded that reductions in larval fish populations caused by entrainment through the SBPP CWIS had no significant ecological effects on populations of juveniles or adults in San Diego Bay. Based on the results of the 1979–1980 316(b) study the report concluded that the design and operation of the CWIS represented the best technology available for minimizing adverse environmental impacts.

3.2 Methods and Station Locations

The following sections provide information on the entrainment and source water sampling, laboratory processing, and methods used to assess entrainment impacts. Locations of entrainment and source water stations are also described.

3.2.1 Field Sampling

Entrainment and source water sampling was conducted monthly from January 2001 through January 2002 and bi-monthly from December 2002 through October 2003. This provided a complete year of data in 2001 (including January 2002) to describe seasonal differences in species abundances, and a comparison year in 2003 (including December 2002) to describe interannual variability. While the results from the second sampling were expected to confirm our initial entrainment assessment, it was recognized that the bi-monthly sampling would affect estimates for species with short larval durations that do not have extended spawning periods. The same set of entrainment and source water stations was sampled (**Figure 3.2-1; Table 3.2-1**) using the same methods during each study period. The first survey in January 2001 ended before all stations were sampled when the boat experienced mechanical problems. Data from this incomplete survey were not included in the analyses presented in this section.

3.2.1.1 Entrainment Sampling

Sample collection methods were similar to those developed and used by the California Cooperative Oceanic and Fisheries Investigation (CalCOFI) in their larval fish studies (Smith and Richardson 1977) but modified for sampling in the shallow areas of south San Diego Bay where depths can be less than 2 m (6.6 ft) during low tides. Entrainment samples were collected from a single station (SB1; **Figure 3.2-1**) located in the SBPP intake channel by towing a bongo frame with two 0.71 m (2.33 ft) diameter openings each equipped with 335-µm (0.013 in) mesh plankton nets and codends. Sampling vessels included a 24 ft. research vessel (*R/V Ecosystems*) with a side-mounted davit for towing the nets, and a 42 ft. trawler (*F/V J. B.*) with a stern-mounted A-frame and amidships winch. The start of each tow began approximately 125 m west of the Units 1&2 intake structure, proceeded in a northwesterly direction against the prevailing intake current, and ended approximately 225 m from the intake structure. A debris boom across the intake channel prevented a closer start point to the intake structure. Because the intake channel was bounded a separation dike to the south and a shallow mudflat to the north, and there was a constant current flow toward the intake structure, it was assumed that all of the water sampled at the entrainment station would have been drawn through the SBPP cooling water system

Entrainment samples were collected over a 24-hour period, with each period divided into six 4-hour sampling cycles. Two replicate tows were collected consecutively at the entrainment station during each cycle. Source water samples at Stations SB2-SB9 were collected from the same vessel during the remainder of each cycle. Concurrent surface water temperatures and salinities were measured at all stations with a digital probe (YSI Model 30).

At all stations, the bongo nets were lowered as close to the bottom as practical without contacting the substrate. Once the nets were near the bottom, the boat was moved forward and the nets retrieved at an oblique angle (winch cable at a 45° angle) to sample the widest strata of water depths possible at each station. The winch retrieval speed was maintained at approximately 0.3 m/sec (1 ft/sec). At the shallowest stations, the boat was moved forward before the nets were lowered into the water so that the codend did not contact the bottom prior to beginning the tow.

Total time of each tow was 1.5–2.0 minutes at a speed of approximately 1 kt during which a combined volume of at least 60m^3 (2,119 ft^3) of water was filtered through both nets. The sample volume was checked when the nets reached the surface, and if the target volume was not collected, the nets were returned to the water at the retrieval point and the tow extended to complete the sample. The water volume filtered was measured by calibrated flowmeters (General Oceanics Model 2030R) mounted in the openings of the nets. Flowmeters were lubricated and maintained before and after each survey, and checked periodically during a survey to ensure that the impeller assembly was spinning freely. Flowmeters were calibrated annually by averaging the readings from ten replicate trials over a measured distance of 10 meters, and applying conversion factors supplied by the manufacturer. Accuracy of individual instruments differed by less than 5% between calibrations.

Once the nets were retrieved from the water all of the collected material was rinsed into the codend. The contents of both nets were combined into one sample immediately after collection. Samples from the paired nets were not kept separate because they were not statistically independent samples and could not be used as replicates for analysis. The use of a bongo frame design minimizes disturbance from the tow bridle compared to a three-point attachment design and allows each net to collect an unobstructed sample. The sample was placed into a labeled jar and was preserved in 10 percent formalin. Each sample was given a serial number based on the location, date, time, and depth of collection. In addition, the information was recorded on a sequentially numbered data sheet. The serial number was used to track the sample through laboratory processing, data analyses, and reporting.

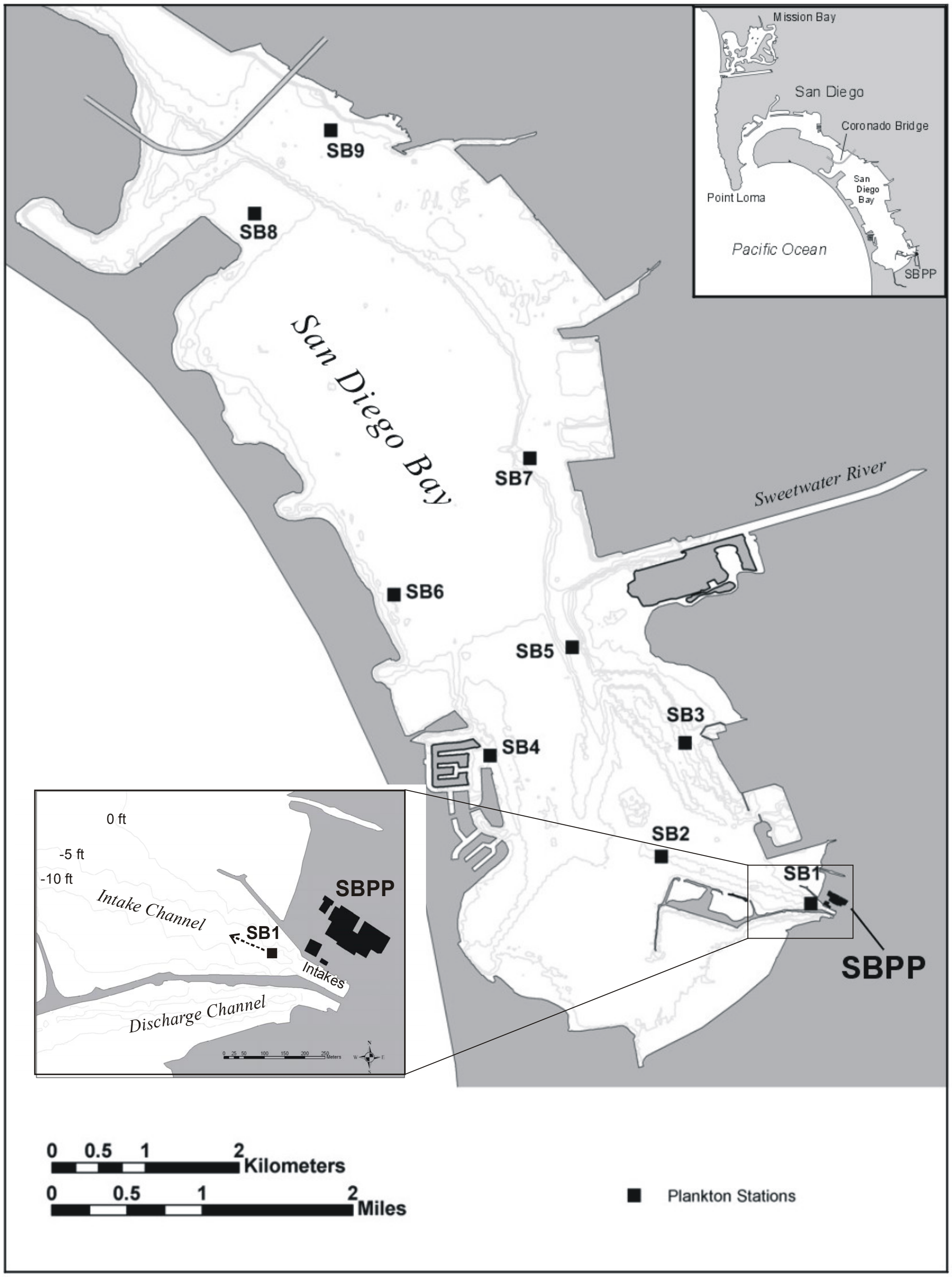


Figure 3.2-1. Location of 2001 and 2003 South Bay Power Plant entrainment (SB1) and source water plankton stations (SB2–SB9). Inset shows entrainment station in relation to SBPP circulating water intakes, and direction and approximate length (100 m) of plankton tows.

Table 3.2-1. Locations of entrainment (SB1) and source water (SB2−SB9) plankton stations. *Source water station also sampled in SDG&E (1980) study. Entrainment samples from this earlier study were collected with pumps at the SBPP intake structures.

Station	Distance from SBPP Intake in km (mi)	Latitude (N)	Longitude (W)	Depth below MLLW in meters (ft)
SB1	0.1 (0.08)	32° 36.869’	117° 05.942’	3.0 (10)
SB2	1.6 (1.0)	32° 37.140’	117° 06.805’	3.7 (12)
SB3	2.1 (1.3)	32° 37.795’	117° 06.668’	4.9 (16)
SB4	3.4 (2.1)	32° 37.723’	117° 07.794’	4.0 (13)
SB5*	3.6 (2.2)	32° 38.347’	117° 07.320’	6.7 (22)
SB6	5.1 (3.2)	32° 38.649’	117° 08.350’	3.7 (12)
SB7	5.5 (3.4)	32° 39.437’	117° 07.565’	11.0 (36)
SB8*	9.0 (5.6)	32° 40.846’	117° 09.153’	1.5 (5)
SB9*	9.5 (5.9)	32° 41.326’	117° 08.714’	11.0 (36)

3.2.1.2 Source Water Sampling

Samples were collected at eight source water stations in the south and south-central regions of San Diego Bay (**Figure 3.2-1**). The source water stations ranged in depth from approximately –2 m (–6.6 ft) MLLW at SB8 to –12 m (–39.4 ft) MLLW at SB9. The stations were stratified to include four channel locations on the east side of the bay and four shallower locations on the west side of the bay. This station array was chosen to include a range of depths and adjacent habitats in south San Diego Bay that would characterize the larval fish composition in the source waters. For example, stations on the east side of the bay were adjacent to saltmarsh habitat and would tend to have a greater proportion of larvae from species with demersal eggs that spawned in saltmarsh channels, while deeper channel stations in the northern end of the study area would tend to have more larvae of species that spawn in open water such as northern anchovy. The station locations also included the three plankton tow stations sampled during the previous 316(b) studies in 1979–1980 (SDG&E 1980) (SB5 off Sweetwater River Marsh; SB8 near the U.S. Navy amphibious base; and SB9 in the navigation channel south of Coronado Bridge).

Source water sampling was conducted using the same methods and during the same time period described above for entrainment sampling (Section 3.2.1.1) with target volumes for the oblique tows of approximately 60 m^3 (1.5–2.0 minute tow at approximately 1 knot). A single tow was completed at each of the source water stations during each of the six 4-hr cycles. Entrainment samples at Station SB1 were collected from the same vessel during the remainder of each cycle. Concurrent surface water temperatures and salinities were measured at all stations with a digital probe (YSI Model 30).

3.2.2 Laboratory Analysis

Laboratory processing consisted of sorting (removing), identifying, and enumerating all larval fishes, megalopal stages of *Cancer* spp. crabs, and spiny lobster larvae (puerulus and phyllosome stages), from the samples. Sorting and identification accuracy was verified and maintained by Tenera Environmental's quality control (QC) program which specified a minimum accuracy level of 95% (**Appendix B**). A total of 21 sorters and 4 taxonomists were involved in the processing of field samples. Each of the taxonomists had at least eight years of experience in the identification of larval fishes or invertebrates. Mr. William Watson of the Southwest Fisheries Science Center checked identifications of problematic specimens. The primary reference for identifications was Moser et al. (1996). During the study, a total of 160 quality control samples were processed for sorting accuracy with no failures, and 38 samples were processed for identification accuracy with one failure. All field and laboratory data were entered into a computer database, which was verified for accuracy against the original data sheets.

Myomere counts and pigmentation patterns were used to identify larval fishes to the lowest taxonomic classification possible, which was usually the species level, but sometimes the genus or family level for certain groups. For example, many species of the family Gobiidae share morphologic and meristic characters during early life stages (Moser et al. 1996) making accurate identifications to the species level questionable. These include early larvae of the arrow goby *Clevelandia ios*, cheekspot goby *Ilypnus gilberti*, and shadow goby *Quietula y-cauda*. These three species were combined into an unidentified goby category referred to as the 'CIQ goby complex'. Larval combtooth blennies *Hypsoblennius* spp. can be easily distinguished from other larval fishes (Moser et al. 1996). However, the three sympatric species that could occur in San Diego Bay cannot be distinguished from each other on the basis of morphometrics or meristics for some of the smaller sizes common in the samples. These combtooth blennies were grouped into an "unidentified combtooth blennies" category (i.e., *Hypsoblennius* spp.). Larvae from the three members of the silversides (family Atherinopsidae) that occur in San Diego Bay (California grunion *Leuresthes tenuis*, jacksmelt *Atherinopsis californiensis*, and topsmelt *Atherinops affinis*) also cannot be easily distinguished at a the smallest larval sizes and were therefore treated as a single group. Similarly, larvae for the deepbody anchovy *Anchoa compressa* and slough anchovy *Anchoa delicatissima* are also very difficult to distinguish and were therefore combined into one group *Anchoa* spp. Also combined into this *Anchoa* spp. group were all small (2–3 mm) Engraulidae (anchovy) individuals, as there were very few other species of this fish family identified from these samples.

Larvae were measured (notochord/standard lengths) to determine their length ranges in the entrainment samples. These estimates were used to calculate the period of time that the larvae were subject to entrainment. Approximately 100–200 larvae from each of the most abundant taxa, or species with recreational or commercial fishery importance, were measured using a video capture system and Optimus™ image analysis software. The number of larvae measured from the surveys in each period ('2001' or '2003' sample periods) was based on the percentage frequency of occurrence of a taxon in each survey. For example, if 20 percent of the California halibut in the first survey period were collected from the entrainment station during the June survey then approximately 20 fish were measured from that survey. The total number of fish measured for each taxon did not exactly equal 100 because at least one or two larvae were measured from surveys that had less than one or two percent of the total for that taxon.

3.2.3 Data Analysis

Sample concentrations of larval fishes, *Cancer* crab megalops, and spiny lobster larvae were computed by dividing the number of each taxon or species in each sample by the volume of water sampled. The mean entrainment concentrations for each survey for each taxon were calculated by averaging the two replicate samples at the entrainment station (SB1) during each cycle and then calculating an average concentration for the survey from the six cycles. The mean concentrations were used in calculating entrainment estimates for each survey that were used in the *ETM* modeling and combined to obtain annual entrainment estimates that were used in the *FH* and *AEL* models.

The mean survey concentrations for the source water were calculated by averaging the data from the six cycles at each station and then averaging the concentrations from the eight source water (SB2–SB9) and entrainment (SB1) stations. Although the mean and variance estimates for each survey were calculated by treating the stations as separate strata, the estimates only accounted for potential differences in sample size among strata and not the differences in volume for the areas that each station represented. We did not have the data necessary to accurately allocate portions of the bay into the strata represented by the stations. In the absence of this type of information the mean source water concentration for each survey weighted each station equally. The mean concentrations for each survey were used in calculating estimates of Proportional Entrainment (*PE*) used in the *ETM* modeling.

The mean survey concentrations were also used to calculate annual averages for the two sampling periods. The annual mean concentrations were used to describe the differences in composition among stations based on the rank order of abundance for the two sampling periods. The annual average concentrations were also used to compute the Bray-Curtis distance between each pair of stations (Digby and Kempton 1987). The scale of the Bray-Curtis distance measure is from 0 to 1.0, with a value of 0 indicating zero distance between samples or 100 percent similarity and a value of 1.0 describing a high degree of dissimilarity. Annual mean concentrations were log transformed prior to computing the Bray-Curtis distances to help account for the many order of magnitude differences in the abundances among taxa. The relationships among stations as depicted by the Bray-Curtis distances were analyzed using the ordination technique of non-metric multidimensional scaling (MDS) (Digby and Kempton 1987). This analysis depicts the dissimilarities among stations provided by the Bray-Curtis distance in a two-dimensional graphical display. Bray-Curtis distances and MDS analyses were calculated separately for the two study periods using PRIMER Version 5.0 (Clarke and Gorley 2001). The taxa responsible for the differences in the MDS were analyzed using the PRIMER SIMPER program (Clarke and Gorley 2001). The taxa were analyzed by grouping the stations into three areas: east shallow (SB1, SB2, and SB3), west shallow (SB4, SB6, and SB8), and channel (SB5, SB7, and SB9).

Data from the larval length measurements were used to estimate the period of time that the larvae are exposed to entrainment. Although there were differences in the distributions of the data for the two sampling periods the length data for the two sampling periods were combined for all of the target taxa except anchovies *Anchoa* spp. The measurements for the two periods combined provided a larger sample size and the best estimate of the size range of entrained larvae. The data for anchovies were not combined because of the statistically significant differences between periods and the large differences in the distributions for the two sampling periods. Potential outliers in the final data for all taxa were eliminated by calculating the duration using the range between lengths of the first and 99th percentiles. Although shrinkage of 6-18% is known to occur in larvae upon death and due to preservation (Porter et al. 2001, Fey 2002) this would not affect our calculation of larval duration since it is based on the differences between tow measurements which would both be subject to approximately equal levels of shrinkage. The larval duration was calculated by dividing the range by a larval growth rate for each taxon derived from the literature. The mean length at entrainment was also calculated from the data for use in the life history modeling described in Section 3.2.3.2 – *Demographic Modeling – Fecundity Hindcasting (FH) and Adult Equivalent Loss (AEL).*

3.2.3.1 Calculating Total Entrainment for the Study Periods

Data were summarized for the impact assessment models using the mean survey concentrations as described above. Entrainment estimates for each survey were computed by multiplying the mean concentration of larval fishes at Station SB1 times the maximum daily CWIS flow rate of 2,275,244 m^3 (601,056,000 gal). Entrainment for each survey period was estimated by multiplying the daily entrainment estimate by the number of days in each survey period (ca. 30 days for the '2001' study period and ca 60 days for the '2003' study period). These survey period entrainment estimates were summarized over each of the two study periods to determine the annual entrainment estimates used in the data summaries and demographic (*FH* and *AEL*) modeling approaches. The variances associated with these estimates may not be representative of the true variance because they do not incorporate the variation between replicate tows during each cycle and among days within survey periods.

3.2.3.2 Demographic Modeling – Fecundity Hindcasting (*FH*) and Adult Equivalent Loss (*AEL*)

The estimates of total entrainment for the two study periods were used in two demographic models that use information on the life history of the target organisms to calculate the numbers of female adult (through use of the *FH* model) or adult (through use of the *AEL* model) fishes represented by the entrainment losses. Both models translate larval entrainment mortality into adult fish losses, which are familiar units to resource managers. The models are conceptualized in **Figure 3.2-2** using life history information that is characteristic of a fish such as the sand bass that is found in San Diego Bay. Larval survival is usually very low in these species, especially at the earliest stages following spawning. In the example only 100 larvae out of 100,000 may survive to juvenile stages when their survival rates typically increase. Even though larval and juvenile survival in most fishes is very low, they can sustain their population by producing only two adults that survive to maturity over their reproductive lifespan. Larval survival estimates for the earliest larval stages from hatching to entrainment are used in the *FH* model to hindcast the adult female reproductive output eliminated by entrainment. Larval survival estimates for post-entrainment larval and juvenile stages are used in the *AEL* model to estimate the equivalent number of adults that the larvae lost due to entrainment would represent.

Both models require information on the life history of the target organisms and were only used when this information was available for the target organism or a closely related species. The sources for the life history parameters used in the models for each target organism are presented in the results. More detailed explanations of the two models and their assumptions are presented in the following sections.

Entrainment losses can be interpreted as population level impacts by comparing *FH* and *AEL* estimates to population estimates available from fishery or other data sources. These comparisons are of somewhat limited use because the *FH* and *AEL* estimates are for a single year and do not account for interannual variation in larval abundances. In addition such comparisons assume that the entrainment losses result in direct losses to the population that are not compensated for by reduced mortality at life stages that are not subject to entrainment. At SBPP *FH* and *AEL* estimates were compared with both fishery data and population estimates for south San Diego Bay from Allen (1999). The data from Allen provide the most useful comparison because his data were collected over a five-year period from 1995-1999 and therefore account for adult interannual variation over that period. The comparison assumes that the variation in the adult population during 2001 and 2003 would fall within the range of variation for the 1995-1999 period. The assumptions in these population-level comparisons and the assumption of 100% mortality for all entrained organisms result in conservative estimates that overestimate actual CWIS effects.

Fecundity Hindcasting (*FH*)

The *FH* approach combines larval entrainment losses with adult fecundity to estimate the adult female reproductive output eliminated by entrainment, assuming no compensatory reserve of the population. *FH* requires an estimate of survival for egg and early larval stages for the time period up to entrainment (**Figure 3.2-2**). The fact that *FH* only requires survivorship for the few days that the eggs or larvae are vulnerable to entrainment is an advantage of this approach over the *AEL* model that requires survival data from the average age at entrainment (a few days) through adult recruitment (up to a few years). Estimates of lifetime fecundity and early life-stage survival from sources other than entrainment are integrated into an estimate of loss by converting the estimated number of entrained larvae backwards to reproductively active females (i.e., hindcasting). In addition to life history information, the *FH* model requires estimates of total entrainment for the study period and the average age of the larvae at entrainment. The model is limited by the need to (1) obtain or model age-specific survival rates and total lifetime fecundities to predict the adult losses, and (2) secure information on the size of the adult population of interest to estimate population-level effects (i.e., relative losses). Therefore, the method was only used for those taxa where the minimal data of item (1), above, were available.

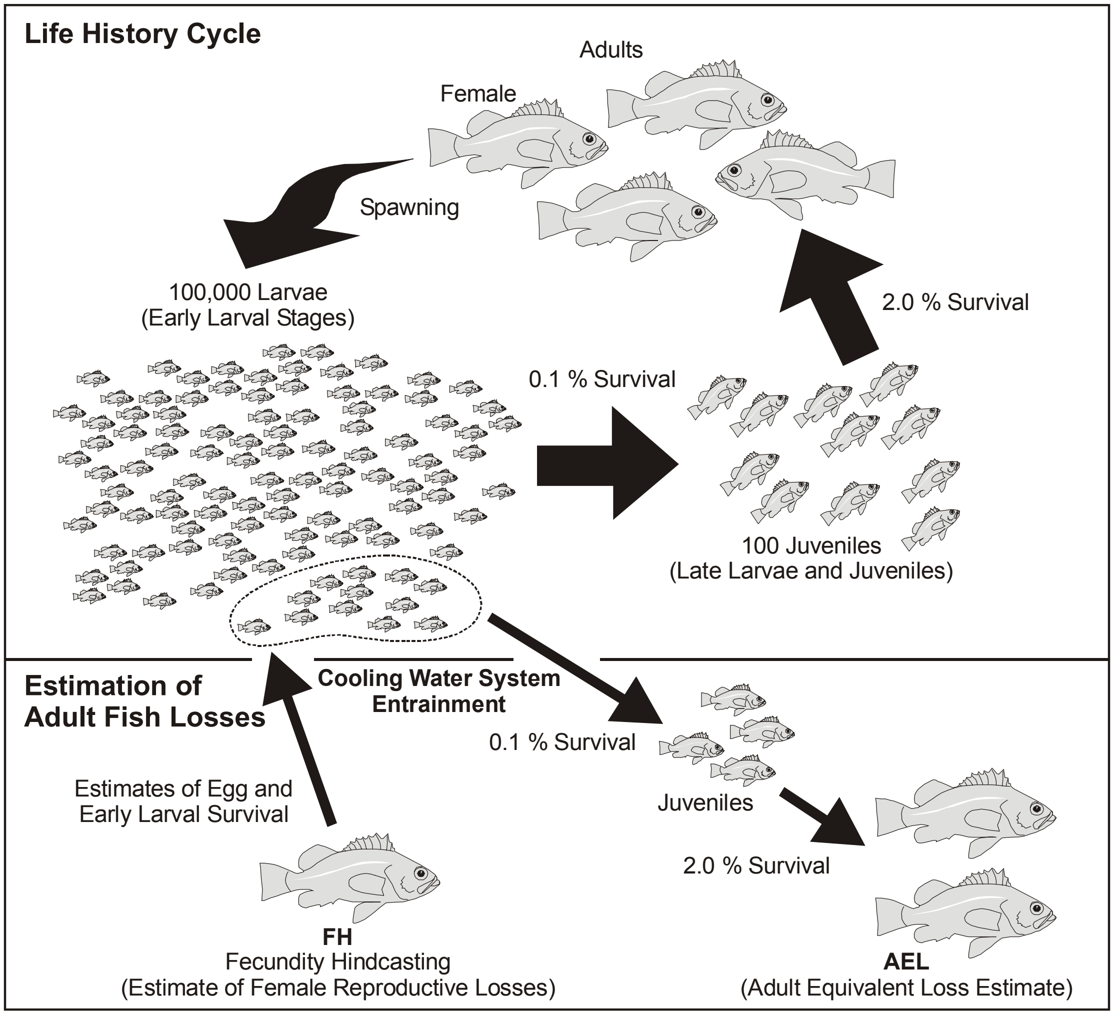


Figure 3.2-2. Life history diagram for fishes with planktonic larvae and examples of how survival estimates of life history stages are used in *FH* and *AEL* models.

The estimated total larval entrainment for a species ($\hat{E}_T$) was used to estimate the number of breeding females needed to produce the number of larvae entrained. The estimated number of breeding females ($\widehat{FH}$) whose fecundity was equal to the estimated total loss of entrained larvae is calculated as follows:

$$\widehat{FH} = \frac{1}{\bar{F}_T} \sum_{i=1}^{w} \frac{\hat{E}_{T_i}}{S_i} \tag{1}$$

where

$w =$ number of weeks the larvae are vulnerable to entrainment,

$\bar{E}_{T_i} =$ estimated total entrainment for the ith survey period ($i = 1,...,$ w),

$S_i =$ survival rate from eggs to larvae of the stage present in the ith survey period, and

$\bar{F}_T =$ average total life time fecundity for females, equivalent to the average number of eggs spawned per female over their reproductive years.

Equation 1 is based on the simplified case of a single synchronized spawning by a species. For species with overlapping or continuous spawning, larval abundance would have to be specified by time period and age class. However, we used the mean size of the larvae entrained to estimate the representative age of the entrained larvae, and then estimated a survival rate to this age. Two input parameters in Equation 1 that may not be available for many species, and thus may limit the method, are average fecundity ($\bar{F}_T$) and survival rates (S_i) from spawning to entrainment.

In practice, survival was estimated for either one or several age classes, depending on the data source, to the estimated age at entrainment. For example, if the mean age at entrainment was estimated at 12 days, and survival rates are available for the separate larval stages (e.g., egg, yolk-sac larvae, post-yolk-sac larvae) through entrainment, then these estimates will be combined to estimate survival over 12 days. For other taxa only a single survival estimate will be available for the period prior to entrainment. The expected total lifetime fecundity $\bar{F}_T$ was approximated by the expression

$$\begin{aligned}\bar{F}_T &= \text{Average eggs/year} \cdot \text{Average number of years of reproductive life} \\ &= \text{Average eggs/year} \cdot \left(\frac{\text{Longevity - Age at maturation}}{2} \right).\end{aligned} \qquad (2)$$

The expected length of reproductive life was approximated as the midpoint between the times of maturation and longevity. This approximation was based on the assumption of a linear survivorship curve between these events (i.e., uniform survival). For exploited species such as northern anchovy, the expected number of years of reproductive life may be much less than predicted using this assumption. Therefore, the estimated longevity was based on the oldest observed individual caught by the fishery, rather than by the oldest recorded fish.

The variance of $\hat{F}H$ was approximated by the Delta method (Seber 1982):

$$\hat{V}ar(\hat{F}H) = (\hat{F}H)^2 \left[CV^2(\hat{E}_T) + \sum_{j=1}^{n} CV^2(\hat{S}_j) + CV^2(\hat{\bar{F}}) + \left(\frac{\hat{V}ar(A_L) + \hat{V}ar(A_M)}{(A_L - A_m)^2} \right) \right]$$

where

$\mathrm{CV}(\hat{E}_T)$= CV of estimated entrainment,
$\mathrm{CV}(\hat{S}_j)$ = CV of estimated survival of eggs and larvae up to entrainment,
$\mathrm{CV}(\hat{\bar{F}})$ = CV of estimated average annual fecundity,
A_M = age at maturation, and
A_L = age at maturity.

The behavior of Equation 2 for *FH* appears log-linear, suggesting that an approximate confidence interval can be based on the assumptions that $\ln(\hat{FH})$ is normally distributed and uses the pivotal quantity

$$Z = \frac{\ln \hat{FH} - \ln FH}{\sqrt{\frac{\hat{Var}(\hat{FH})}{\hat{FH}^2}}}.$$

A 90% confidence interval for *FH* was estimated by solving for *FH* and setting Z equal to

+/-1.645, i.e.

$$\hat{FH} \cdot e^{-1.645\sqrt{\frac{\hat{Var}(\hat{FH})}{\hat{FH}^2}}} \quad \text{to} \quad \hat{FH} \cdot e^{+1.645\sqrt{\frac{\hat{Var}(\hat{FH})}{\hat{FH}^2}}}.$$

The *FH* model assumes the following:

- Values of parameters from the literature represent the population parameters for the time period and location of the SBPP study.
- No population reserve or compensation counters entrainment mortality.
- Estimates of annual egg production are representative of the average for the reproductive lifespan of each taxon.
- Reproductive life expectancy can be accurately calculated by assuming that time of death is uniformly distributed between age-at-maturation and age-of-longevity.
- Juvenile and egg survival rates are constant over time.
- The loss of the reproductive potential of one female is equivalent to the loss of an adult female.

Adult Equivalent Loss (*AEL*)

The *AEL* approach uses an estimate of the abundance of entrained or impinged organisms to forecast the loss of an equivalent number of adults. The approach requires survival estimates (had the larvae not been entrained) from entrainment to an age at recruitment to the adult population (**Figure 3.2-2**). In addition to life history information, the *AEL* model requires estimates of total entrainment for the study period and the average age of the larvae at entrainment. The model is limited by the need to (1) obtain or model age-specific survival rates to predict the adult losses, and (2) secure information on the size of the adult population of interest to estimate population-level effects (i.e., relative losses). Therefore, the method was only used for those taxa where the minimal data of item (1), above, were available.

Starting with the number of age class j larvae entrained ($\hat{E}_j$), it is conceptually easy to convert these numbers to an equivalent number of adults lost ($\widehat{AEL}$) at some specified age class from the formula:

$$\widehat{AEL} = \sum_{j=1}^{n} \hat{E}_j \hat{S}_j \tag{3}$$

where

$n =$ number of age classes,

$\hat{E}_j =$ estimated number of larvae lost in age class j, and

$\hat{S}_j =$ survival rate for the jth age class to adulthood (Goodyear 1978).

Age-specific survival rates from larval stage to recruitment into the fishery (through juvenile and early adult stages) must be included in this assessment method. For some commercial species, survival rates are known for adults in the fishery; but for most species, age-specific larval survivorship has not been well described.

When age-specific survival rates from larval stage to recruitment into the fishery were available, AEL was calculated using survival from a representative age of the entrained larvae at SBPP. This age was calculated by dividing the average larval length at entrainment (minus hatch length) by a literature-based growth rate. Age-specific survivorship for any interval of time (t) was then calculated following the formula (Ricker 1975)

$$\frac{N_t}{N_0} = e^{-Zt}$$

where

$N_t =$ number of animals in the population at time t,

$N_0 =$ number of animals in the population at time $t = 0$,

$\frac{N_t}{N_0} = S$ (finite survivorship to time t),

$e = 2.71828...$(base of the natural log), and

$Z =$ instantaneous mortality rate.

Survivorship to recruitment, to an adult age, was apportioned into several age stages, and *AEL* was calculated using the entrainment estimate as

$$\widehat{AEL} = \hat{E}_T \prod_{j=1}^{n} \hat{S}_j \tag{4}$$

where

$n =$ number of age classes from entrainment to recruitment and

$\hat{S}_j =$ survival rate from the beginning to end of the jth age class.

The variance of $\widehat{AEL}$ can be estimated using the Delta method of Seber (1982) as

$$\widehat{Var}(\widehat{AEL}) = \widehat{AEL}^2 \left(CV^2(\hat{E}_{Adj-T}) + \sum_{j=1}^{n} CV^2(\hat{S}_j) \right). \tag{5}$$

$\widehat{AEL}$ and $\widehat{FH}$ can be compared by assuming a stationary population where an adult female must produce two adults (i.e., one male and one female). Overall survival (S_T) can then be estimated from total lifetime fecundity ($\bar{F}_T$) by the quantity

$$\hat{S}_T = \frac{2}{\hat{\bar{F}}_T} = \hat{S}_{egg} \cdot \hat{S}_{larvae} \cdot \hat{S}_{adult},$$

which leads to

$$\hat{S}_{adult} = \frac{2}{\hat{\bar{F}}_T \cdot \hat{S}_{egg} \cdot \hat{S}_{larvae}}. \tag{6}$$

Substituting Equation 6 into the overall form of the *AEL* equation where

$$\widehat{AEL} = \hat{E}_T \cdot \hat{S}_{adult} \tag{7}$$

yields

$$\widehat{AEL} = \frac{2(\hat{E}_T)}{\hat{S}_{egg} \cdot \hat{S}_{larva} \cdot \hat{\bar{F}}_T}$$

where

$$\widehat{AEL} \equiv 2\widehat{FH}\ . \tag{8}$$

Without independent adult survival rates and assuming a 50:50 sex ratio, $\hat{AEL}$ and $\hat{FH}$ are deterministically related according to Equation 8, with an associated standard error of $\hat{SE}(\hat{AEL}) = 2\hat{SE}(\hat{FH})$. Equation 8 should be aligned so that the average female age is also the age of recruitment used in computing $\hat{AEL}$. This alignment is accomplished by solving the simple exponential survival equation (Ricker 1975)

$$N_t = N_0 \cdot e^{-Z(t-t_0)}$$

by substituting numbers of either equivalent adults or hindcast females, their associated ages, and mortality rates into the equation where,

N_t = number of adults at time t,
N_0 = number of adults at time t_0,
Z = instantaneous rate of natural mortality, and
t = age of hindcast animals (FH) or extrapolated age of animals (AEL).

This allows for the alignment of ages in either direction such that $2FH \equiv AEL$ since they are either hindcast or extrapolated to the same age.

The estimates of mortality calculated from the *AEL* and *FH* approaches can be compared for the same time periods for taxa where independent estimates are available for survival from entrainment to recruitment into the fishery and entrainment back to hatching. These comparisons serve as a method of cross-validation for the demographic approaches to impact assessment.

Calculations of *AEL* using data on survivorship from entrainment to recruitment into the fishery assume the following:

- Values in the literature on life history parameters represent the fish population in the time period and location of the SBPP study.
- If survivorship values from the literature are limited to single observations, values are assumed constant over time or representative of the mean.
- No population reserve or compensation counters entrainment mortality.
- Survival rates used in the calculations represent the life stage of the larvae or fish.

In some cases, survival rates estimated for a similar fish species were used. Should survivorship data from one species be substituted for another, there is the additional assumption that:

- Values of survivorship for the two species are the same.

For fish species where larval survival data are missing, expected survival could be estimated from fecundity and juvenile and adult survival data. However, in those cases where fecundity data were available, we did not have juvenile and adult survival estimates. To use fecundity data in calculating survival rates, there is the additional assumption that

- The fish population is stationary in size such that each adult female contributes two new offspring to the population of adults during its lifetime.

3.2.3.3 Empirical Transport Model

The empirical transport model (*ETM*) has been proposed by the U.S. Fish and Wildlife Service to estimate mortality rates resulting from circulating water withdrawals by power plants (Boreman et al. 1978, and subsequently in Boreman et al. 1981) as an alternative to the demographic models described above. The *ETM* model provides an estimate of incremental (a conditional estimate in absence of other mortality, Ricker 1975) mortality imposed by SBPP on local San Diego Bay larval populations by using empirical data (plankton samples) rather than relying solely on hydrodynamic and demographic calculations. Consequently, *ETM* requires an additional level of field sampling to characterize the abundance and composition of source water larval populations. The fractional loss to the source water population represented by entrainment is provided by estimates of proportional entrainment (*PE*) for each survey that can then be expanded to predict regional effects on appropriate adult 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 at a southern California power plant (Parker and DeMartini 1989). The *ETM* has also been used to assess impacts at the Salem Nuclear Generating Station in Delaware Bay, New Jersey (PSE&G 1993) as well as other power stations along the East Coast. Empirical transport modeling permits the estimation of conditional mortality due to entrainment while accounting for the spatial and temporal variability in distribution and vulnerability of each life stage to power plant withdrawals. The modeling approach described below uses a *PE* approach that is similar to the method described by MacCall et al. (1983) and used by Parker and DeMartini (1989) in their final report to the California Coastal Commission (Murdoch et al. 1989) for the San Onofre Nuclear Generating Station (SONGS). This estimate can then be summarized over appropriate blocks of time in a manner similar to that of the *ETM*.

The general equation to estimate *PE* for a day on which entrainment was sampled is:

$$\hat{P}E = \frac{\hat{N}_{Ei}}{\hat{N}_{Si}}$$

where

$\hat{N}_{Ei}$ = estimated number of larvae entrained during the day in survey i, calculated as (estimated density of larvae in the water entrained that day)$\times$(design specified daily cooling water intake volume),

$\hat{N}_{Si}$ = estimated number of larvae in the source water that day in survey i (estimated density of larvae in the souce water that day)$\times$(source water volume).

The *PE* value represents the effects of a number of processes operating over a day and is estimated for each survey for the two study periods.

If larval entrainment mortality is constant throughout the period and a larva is susceptible to entrainment over *d* days, then the proportion of larvae that escape entrainment in survey *i* is:

$$(1-\hat{P}E_i)^{\hat{d}}\ .$$

Larval duration from hatching to entrainment was calculated from growth rates using the length representing the upper 99th percentile of the length measurements. The value for *d* was computed by dividing an estimate of growth rate into the change in length based on this 99th percentile estimate. The minimum size used for computing the larval duration was determined after removing the smallest 1 percent of the values. This procedure eliminated outlier measurements in the data.

It is possible that aging was biased for the following reason, even though standard lengths of larval fishes (i.e., measurements of minimum, mean, and maximum), and larval growth rates were applied to estimate the ages of the entrained larvae. It was assumed that larvae shorter than the minimum length were collected very soon after hatching and were therefore, aged at zero days. Subsequent ages were estimated using this length as a basis. Other reported data for various species suggest that hatching lengths can be either smaller or larger than the size estimated from SBPP samples, and indicate that the smallest observed larvae represent either natural variation in hatch lengths within the population or shrinkage following preservation (Theilacker 1980). The possibility remains that all larvae from the observed minimum length to the greatest reported hatching length (or to some other size) could have just hatched, leading to overestimation of ages for all larvae. The same values were used for both the *FH* and *AEL* models to estimate age at entrainment and are subject to similar biases.

The surveys in each study period were used to estimate larval mortality (P_M) due to entrainment using the following equation

$$\hat{P}_M = 1 - \sum_{i=1}^{12} \hat{f}_i (1 - \hat{P}E_i)^{\hat{d}} \quad (9)$$

where

$\hat{P}E_i$ = estimate of proportional entrainment for the *i*th survey,

$\hat{f}_i$ = proportion of the total annual source water population present during the *i*th survey, and

$\hat{d}$ = the estimated number of days of larval life.

To establish independent survey estimates, it is assumed that during each survey a new and distinct cohort of larvae is subject to entrainment. Each of the surveys was weighted by $\hat{f}_i$ and estimated as the proportion of the total annual source water population present during each *i*th survey period. For each study period, the sum of the proportions equals one:

$$\hat{f}_i = \frac{\hat{N}_S}{\sum_{i=1}^{n} \hat{N}_{Si}} \text{ and } \sum_{i=1}^{n} \hat{f}_i = 1.$$

The variance of P_M was estimated using the Delta method (Seber 1982), and its formulation is presented in **Appendix C**.

The estimate of the population-wide probability of entrainment ($\hat{P}E_i$) is the central feature of the *ETM* approach (Boreman et al. 1981, MacCall et al. 1983). If a population is stable and stationary, then $\hat{P}_M$ also estimates the effects on the fully-recruited adult age classes when uncompensated natural mortality from larva to adult is assumed.

Assumptions associated with the estimation of P_M include the following:

- The samples at each survey period represent a new and independent cohort of larvae.
- The estimates of larval abundance for each survey represent a proportion of total annual larval production during that survey.
- The conditional probability of entrainment (PE_i) is constant within survey periods.
- Lengths and applied growth rates of larvae accurately estimate larval duration.

3.2.3.4 Target Taxa Selection

The sampling and processing was designed to quantify entrainment and source water populations of larval fishes, megalops stage *Cancer* crab, and spiny lobster larvae. Results from entrainment sampling were used to identify the taxa that would be evaluated for entrainment effects. The fishes comprising up to 99 percent of the total abundance in either sampling year were targeted for assessment. In addition, commercially or recreationally important fishes that were present in abundances that provided reasonable data for analysis were also selected. No invertebrates were included in the assessment because they were not present in high enough abundances to provided reasonable data for analysis. In fact, only a single *Cancer* crab megalops larva and no *Panulirus* larvae were collected from the entrainment samples.

3.3 Entrainment and Source Water Results

3.3.1 Community Overview

3.3.1.1 Entrainment Results

Totals of 23,039 and 7,589 larval fishes were collected from the SBPP entrainment station (SB1) during the 2001 and 2003 sampling periods, respectively (**Tables 3.3-1, 3.3-2,** and **Appendix D**). The count of fishes and invertebrates collected during Survey 1 (an incomplete survey), and the number of unidentified, damaged, or larval fish fragments are not presented in **Tables 3.3-1** or **3.3-2,** nor used in the analysis, but are presented in **Appendix D**. During the 2001 period the greatest concentrations of larval fishes at the entrainment station occurred during the June survey, while during the 2003 period the greatest concentrations occurred during the December 2002 survey (**Figures 3.3-1** and **3.3-2**); December 2002 was included in the 2003 study period. The maximum concentration of all larval fishes combined was much larger in the 2001 sampling period.

Only a single *Cancer* crab megalopae (collected during the 2001 period) and no spiny lobster larvae were collected from the entrainment station. Fish fragments and damaged fishes that could not be identified to species typically comprised less that 1% of the total catch and were not included in the summary analyses.

Total annual entrainment was estimated to be 2.42×10^9 and 1.57×10^9 for the 2001 and 2003 periods, respectively (**Tables 3.3-1** and **3.3-2**). Entrainment samples were dominated by gobies in the CIQ complex. They comprised the largest percentage of the total estimated entrainment for both sampling periods, 76 and 89 percent, respectively (**Figures 3.3-3a, b**). CIQ gobies and anchovies comprised greater than 95 percent of the total estimated entrainment for both sampling periods. The fewer number of taxa and the lower maximum concentrations in the 2003 study period probably reflect the reduced number of samples collected during 2003.

Based on the estimated entrainment from both sampling periods the following five taxa were evaluated for entrainment effects:

- CIQ goby complex (unidentified Gobiidae),
- longjaw mudsucker (*Gillichthys mirabilis*),
- anchovies (*Anchoa* spp.),
- silversides (Atherinopsidae), and
- combtooth blennies (*Hypsoblennius* spp.).

Table 3.3-1. Total annual entrainment estimates of fishes and target invertebrates based on monthly larval densities (sampled at Station SB1 from February 2001 through January 2002) and design maximum circulating water flows; *n*=144 tows at 1 station.

Taxon	Common Name	Total Larvae Sampled	Percent Comp.	Cum. Percent	Estimated Total Annual Entrainment	Standard Error
CIQ goby complex	gobies	17,878	75.64	75.64	1,830,898,760	21,724,769
Anchoa spp.	bay anchovies	4,390	21.27	96.91	514,808,619	5,071,239
Hypsoblennius spp.	combtooth blennies	226	0.92	97.83	22,334,999	258,893
Gillichthys mirabilis	longjaw mudsucker	249	0.91	98.74	21,953,225	405,184
Atherinopsidae	silversides	140	0.60	99.34	14,521,485	384,593
Syngnathus spp.	pipefishes	101	0.41	99.75	10,013,128	329,781
Acanthogobius flavimanus	yellowfin goby	19	0.09	99.85	2,260,696	89,422
Strongylura exilis	California needlefish	8	0.03	99.88	740,045	26,934
Sciaenidae	croakers	6	0.03	99.91	706,220	38,208
Hyporhamphus rosae	California halfbeak	3	0.01	99.92	346,465	34,112
Genyonemus lineatus	white croaker	3	0.01	99.93	340,216	22,636
Hypsopsetta guttulata	diamond turbot	3	0.01	99.95	277,819	15,795
Engraulis mordax	northern anchovy	3	0.01	99.96	269,386	30,975
Ruscarius creaseri	roughcheek sculpin	2	0.01	99.97	214,553	2,914
Odontopyxis trispinosa	pygmy poacher	2	0.01	99.97	214,553	2,914
Gobiesox spp.	clingfishes	2	0.01	99.98	179,103	22,315
Cheilotrema saturnum	black croaker	1	<0.01	99.99	137,775	24,745
Lepidogobius lepidus	bay goby	1	<0.01	99.99	113,911	20,459
Typhlogobius californiensis	blind goby	1	<0.01	100.00	107,251	20,268
Paralichthys californicus	California halibut	1	<0.01	100.00	89,571	17,914
	Total Fishes	**23,039**			**2,420,527,779**	
Cancer antennarius (megalopa)	brown rock crab	1			99,567	18,816

Table 3.3-2. Total annual entrainment estimates based on bi-monthly larval densities (sampled at Station SB1 from December 2002 through October 2003) and design maximum circulating water flows; *n*=72 tows at 1 station. No target invertebrate larvae were collected during this period.

Taxon	Common Name	Total Larvae Sampled	Percent Comp.	Cum. Percent	Estimated Total Annual Entrainment	Standard Error
CIQ goby complex	gobies	6,747	88.99	88.99	1,394,283,727	9,291,117
Anchoa spp.	bay anchovies	502	6.84	95.84	107,170,502	1,294,042
Gillichthys mirabilis	longjaw mudsucker	132	1.60	97.43	25,034,110	389,598
Hypsoblennius spp.	combtooth blennies	124	1.51	98.94	23,615,354	328,998
Atherinopsidae	silversides	50	0.63	99.57	9,792,101	293,011
Syngnathus spp.	pipefishes	23	0.30	99.86	4,691,648	126,804
Acanthogobius flavimanus	yellowfin goby	6	0.07	99.93	1,055,578	40,520
Sciaenidae	croakers	4	0.05	99.99	855,939	64,400
Engraulis mordax	northern anchovy	1	0.01	100.00	203,692	26,297
	Total Fishes	**7,589**			**1,566,702,650**	

These five taxa comprised over 99 percent of the total entrainment in both sampling periods (**Tables 3.3-1** and **3.3-2**). California halibut and other commercially or recreationally important fishes were collected in very low numbers at the entrainment station, and therefore were not analyzed further. Only a single California halibut larva was collected during the 2001 study period, and no halibut larvae were collected during 2003. The single larva from 2001 was used to estimate a total annual entrainment of 89,600 (**Table 3.3-1**). California halibut were identified as a 'critical taxa' in the previous 316(b) study (SDG&E 1980). Their estimate of annual entrainment losses of 420,000 larvae indicate that they collected only 4-5 larvae in their entrainment samples. Entrainment effects for California halibut were not assessed because of the low entrainment estimates from both studies. None of the target invertebrate taxa were evaluated for entrainment effects because only a single *Cancer* crab megalopae was identified from the entrainment samples.

3.3.1.2 Source Water Results

Totals of 123,304 and 41,195 larval fishes were collected from the SBPP source water stations (SB2–SB9) during the 2001 and 2003 sampling periods, respectively (**Tables 3.3-3** and **3.3-4**). The count of fishes and invertebrates collected during Survey 1 (an incomplete survey), and the number of unidentified, damaged, or larval fish fragments are not presented in **Tables 3.3-3** or **3.3-4,** nor used in the analysis, but are presented in **Appendix D**. Crab megalopae and spiny lobster larvae were more abundant at the source water stations than at the entrainment station. During the 2001 period the greatest mean concentrations of larval fishes at the source water stations occurred during the February and July surveys, while during the 2003 period the greatest mean concentrations occurred during the December 2002 survey (**Figures 3.3-4** and **3.3-5**); December 2002 was included in the 2003 study period. Mean concentrations were much larger in the 2001 sampling period when compared to the 2003 period.

Similar to entrainment samples, CIQ gobies and the anchovy complex were the most abundant larvae collected at source water stations during the 2001 and 2003 study periods, comprising 92 and 89 percent, respectively, of the total larvae collected (**Tables 3.3-3** and **3.3-4**). There were fewer taxa collected during the 2003 study period (25) when compared to the 2001 period (43), but the lists of the ten most abundant taxa were very similar between periods. Although the reduced sampling resulted in fewer taxa and overall lower numbers during the 2003 study period, California halibut were more abundant during 2003 and sand basses (*Paralabrax* spp.), another important fishery species, were collected in approximately equal numbers during both periods.

Table 3.3-3. Summary of larval fish and invertebrate abundances at source water stations from February 2001 through January 2002; *n*=288 tows at 8 stations.

Taxon	Common Name	Total Larvae Collected	Percent of Total	Cum. Percent
CIQ goby complex	CIQ gobies	94,641	76.75	76.75
Anchoa spp.	anchovies	18,646	15.12	91.88
Hypsoblennius spp.	Combtooth blennies	7,316	5.93	97.81
Atherinopsidae	silversides	796	0.65	98.46
Syngnathus spp.	pipefishes	326	0.26	98.72
Hypsopsetta guttulata	diamond turbot	251	0.20	98.92
Gillichthys mirabilis	longjaw mudsucker	212	0.17	99.09
Engraulis mordax	northern anchovy	205	0.17	99.26
Paralabrax spp.	sand basses	174	0.14	99.40
Labrisomidae	labrisomid kelpfishes	149	0.12	99.52
Acanthogobius flavimanus	yellowfin goby	142	0.12	99.64
Genyonemus lineatus	white croaker	128	0.10	99.74
Sciaenidae	croakers	87	0.07	99.81
Cheilotrema saturnum	black croaker	74	0.06	99.87
Paralichthys californicus	California halibut	32	0.03	99.90
Gibbonsia spp.	clinid kelpfishes	18	0.01	99.91
Trachurus symmetricus	jack mackerel	18	0.01	99.93
Serranidae	basses	13	0.01	99.94
Lepidogobius lepidus	bay goby	8	0.01	99.94
Citharichthys stigmaeus	speckled sanddab	7	0.01	99.95
Roncador stearnsi	spotfin croaker	7	0.01	99.96
Clupeiformes	herrings and anchovies	6	<0.01	99.96
Menticirrhus undulatus	California corbina	6	<0.01	99.97
Clinocottus analis	wooly sculpin	4	<0.01	99.97
Odontopyxis trispinosa	pygmy poacher	4	<0.01	99.97
Hippocampus ingens	Pacific seahorse	4	<0.01	99.98
Typhlogobius californiensis	blind goby	3	<0.01	99.98
Leptocottus armatus	Pacific staghorn sculpin	3	<0.01	99.98
Gobiesox spp.	clingfishes	3	<0.01	99.98
Artedius spp.	sculpins	2	<0.01	99.98
Atractoscion nobilis	white seabass	2	<0.01	99.99
Ruscarius creaseri	roughcheek sculpin	2	<0.01	99.99
Pleuronichthys ritteri	spotted turbot	2	<0.01	99.99
Cottidae	sculpins	2	<0.01	99.99
Oligocottus spp.	sculpins	2	<0.01	99.99
Paralichthyidae	lefteye flounders & sanddabs	2	<0.01	99.99
Citharichthys spp.	sanddabs	1	<0.01	100.00
Nannobrachium spp.	lanternfishes	1	<0.01	100.00
Sebastes spp.	rockfishes	1	<0.01	100.00
Hyporhamphus rosae	California halfbeak	1	<0.01	100.00
Porichthys myriaster	specklefin midshipman	1	<0.01	100.00
Gobiesox rhessodon	California clingfish	1	<0.01	100.00
Clupeidae	herrings	1	<0.01	100.00
	Fish Total	**123,304**		
Panulirus interruptus	California spiny lobster larvae	52		
Cancer anthonyi (megalops)	yellow crab	4		
Cancer antennarius (megalops)	brown rock crab	2		
	Target Invertebrate Total	**58**		

Table 3.3-4. Summary of larval fish abundances at source water stations from December 2002 through October 2003; *n*=288 tows at 8 stations. No target invertebrate larvae were collected during this period.

Taxon	Common Name	Total Larvae Collected	Percent of Total	Cum. Percent
CIQ goby complex	CIQ gobies	32,808	79.64	79.64
Anchoa spp.	anchovies	3,942	9.57	89.21
Hypsoblennius spp.	Combtooth blennies	3,292	7.99	97.20
Atherinopsidae	silversides	261	0.63	97.83
Syngnathus spp.	pipefishes	227	0.55	98.39
Paralabrax spp.	sand basses	158	0.38	98.77
Hypsopsetta guttulata	diamond turbot	137	0.33	99.10
Sciaenidae	croakers	83	0.20	99.30
Labrisomidae	labrisomid kelpfishes	69	0.17	99.47
Paralichthys californicus	California halibut	60	0.15	99.62
Engraulis mordax	northern anchovy	54	0.13	99.75
Gillichthys mirabilis	longjaw mudsucker	35	0.08	99.83
Cheilotrema saturnum	black croaker	20	0.05	99.88
Acanthogobius flavimanus	yellowfin goby	17	0.04	99.92
Atractoscion nobilis	white seabass	5	0.01	99.93
Hippocampus ingens	Pacific seahorse	5	0.01	99.95
Typhlogobius californiensis	blind goby	5	0.01	99.96
Genyonemus lineatus	white croaker	5	0.01	99.97
Paralichthyidae	lefteye flounders & sanddabs	3	0.01	99.98
Gibbonsia spp.	clinid kelpfishes	2	0.00	99.98
Citharichthys stigmaeus	speckled sanddab	2	0.00	99.99
Menticirrhus undulatus	California corbina	2	0.00	99.99
Labridae	wrasses	1	0.00	100.00
Gobiesox rhessodon	California clingfish	1	0.00	100.00
Lepidogobius lepidus	bay goby	1	0.00	100.00
	Total Fishes	**41,195**		

3.3.1.3 Station Comparisons

The entrainment and source water stations extend over a distance of over 9 km (5.6 mi) in south San Diego Bay and include both channel and shallow mudflat habitats. Despite the differences in location and habitat unidentified gobies in the CIQ complex were the most abundant fish larvae at all of the stations during both study periods (**Tables 3.3-5** and **3.3-6**). CIQ gobies were most abundant at Stations SB6 and SB7 during the first study period and at SB7 during the second study period. Overall, taxa richness generally increased from the entrainment station in the far south end of the bay to Station SB9 in the north closest to the Coronado Bridge during both study periods (**Figure 3.3-6a, b**).

Diversity (Shannon-Weiner H’) was generally low among all of the stations (**Figure 3.3-6a, b**) during both study periods. This was due to the dominance in the samples by a few taxa such as CIQ gobies, anchovies, and combtooth blennies.

Bray-Curtis distances computed on the average concentrations of fish larvae for the nine stations showed that the entrainment station, SB1, was most similar during both study periods to Station SB2, which is also located in the far south end of the bay (**Table 3.3-7a, b**). The entrainment station was least similar to Stations SB7 and SB9, which are located in channel areas to the north. These relationships were further analyzed using non-metric multidimensional scaling (MDS). The MDS analysis for the 2001 study indicates that the biological differences among stations almost match the geographic position of the stations (**Figure 3.3-7a**). Data collected from Stations SB1, SB2, and SB3, which are all located close together in the South Bay, all cluster together with Station SB3 being more similar to Station SB5 in the channel to the north, than it is to the three mudflat stations (SB4, SB6, and SB8). These three stations all cluster together and are most similar to Station SB2, which is located in the intake channel near mudflat habitat. Stations SB7 and SB9 are most similar to the other channel station, SB5. The MDS analysis for the 2003 study period showed similar results that were less clearly matched to the geographic positions of the stations (**Figure 3.3-7b**).

While the results from both years show that the larval fish community at the entrainment station is different from the stations to the north near the Coronado Bridge, the gradient of change and the differences among stations are relatively consistent. Although the distances among stations are similar in magnitude, the species causing those differences vary from the entrainment station (SB1) north to Station SB9. SIMPER analysis of the data shows that the fish species causing the differences between the east bay shallow (SB1, SB2, and SB3) and west bay shallow (SB4, SB6, and SB8) areas for both study periods is the longjaw mudsucker, which was collected in greatest abundances at the entrainment station SB1 (**Tables 3.3-8** and **3.3-9**). Diamond turbot also contributed to the differences during both periods because it was more abundant at the west bay shallow stations. The fish taxon responsible for most of the differences between the east bay shallow and deep channel stations during both study periods was the sand bass, which was collected in greatest abundances at the deep channel stations. Combtooth blennies, northern anchovy, and diamond turbot also contributed to the differences during both study periods and were also in greater abundances at the deep channel stations.

The results show that larval fish community composition changes along a gradient from north to south as a function of distance from the mouth of San Diego Bay and is also influenced by habitat differences. White croaker adults (Allen 1999) and larvae (this study) were found in higher abundance at northern stations in San Diego Bay. Larval California halibut were found in highest concentrations at the channel stations, the general area where Allen (1999) found the highest abundance of adults of this species. Larval combtooth blennies and longjaw mudsucker were also found in highest abundance in habitats where the adults reside. Longjaw mudsucker adults prefer the saltmarsh habitat in the eastern and southern portions of the bay, and adult combtooth blennies are in higher abundance along the east side of the bay (Allen 1999) where there are higher numbers of pier pilings.

Table 3.3-5. Mean larval fish concentrations (larvae per 1000 m^3) by station for monthly surveys from February 2001 through January 2002.

		Stations									
Taxon	**Common Name**	**SB1**	**SB2**	**SB3**	**SB4**	**SB5**	**SB6**	**SB7**	**SB8**	**SB9**	**Mean**
CIQ goby complex	gobies	2,095.9	1,549.6	2,391.7	2,914.0	3,003.0	4,109.9	3,995.8	2,743.1	2,400.4	**2,800.4**
Anchoa spp.	bay anchovies	556.5	476.4	231.4	159.6	938.9	1,327.7	1,042.7	520.4	73.3	**591.9**
Hypsoblennius spp.	combtooth blennies	27.2	45.7	140.8	81.6	210.8	84.6	575.7	94.4	453.6	**190.5**
Atherinopsidae	silversides	18.2	57.1	6.0	42.2	11.4	22.4	5.3	58.5	18.2	**26.6**
Syngnathus spp.	pipefishes	12.5	13.7	8.3	4.5	16.0	8.1	12.8	6.9	9.2	**10.2**
Gillichthys mirabilis	longjaw mudsucker	27.1	4.3	11.5	3.1	15.9	1.5	12.2	0.7	1.2	**8.6**
Engraulis mordax	northern anchovy	0.4	0.8	0.9	-	6.9	0.8	18.6	15.1	11.1	**6.1**
Hypsopsetta guttulata	diamond turbot	0.4	0.8	1.9	2.1	5.9	2.6	10.7	11.8	18.4	**6.1**
Acanthogobius flavimanus	yellowfin goby	2.4	3.5	0.6	12.0	2.9	15.1	1.0	1.9	2.0	**4.6**
Paralabrax spp.	sand basses	-	0.2	0.6	-	12.2	1.1	17.6	1.7	6.9	**4.5**
Labrisomidae	labrisomid kelpfishes	-	1.4	2.5	4.8	2.0	1.1	10.1	9.0	5.5	**4.0**
Genyonemus lineatus	white croaker	0.5	1.0	1.8	2.3	6.3	5.3	6.7	4.3	4.8	**3.7**
Sciaenidae	croakers	0.7	0.4	1.0	0.2	5.1	0.3	10.1	0.2	4.2	**2.5**
Cheilotrema saturnum	black croaker	0.2	0.3	0.5	0.8	4.1	3.0	3.9	0.8	3.8	**1.9**
Paralichthys californicus	California halibut	0.1	0.5	0.2	0.2	0.5	0.7	2.0	0.4	2.4	**0.8**
Gibbonsia spp.	clinid kelpfishes	-	-	0.2	1.8	0.8	0.5	-	0.7	0.8	**0.5**
Trachurus symmetricus	jack mackerel	-	-	-	-	-	-	-	-	3.5	**0.4**
Serranidae	sea basses	-	-	-	-	-	-	-	0.9	1.5	**0.3**
Lepidogobius lepidus	bay goby	0.1	-	0.3	0.4	0.2	-	0.5	0.2	0.4	**0.2**
Roncador stearnsi	spotfin croaker	-	-	0.4	-	0.6	-	0.4	0.4	0.2	**0.2**
Menticirrhus undulatus	California corbina	-	-	-	-	0.9	-	0.5	-	0.1	**0.2**
Citharichthys stigmaeus	speckled sanddab	-	-	-	0.4	-	-	-	0.2	1.0	**0.2**
Clupeiformes	herrings and anchovies	-	-	-	-	-	1.2	-	-	0.2	**0.2**
Odontopyxis trispinosa	pygmy poacher	0.3	-	-	0.6	-	0.3	-	-	0.2	**0.2**
Gobiesox spp.	clingfishes	0.2	-	-	0.3	-	-	-	0.6	-	**0.1**
Hippocampus ingens	Pacific seahorse	-	-	0.3	-	-	0.3	-	0.4	-	**0.1**
Clinocottus analis	wooly sculpin	-	-	-	-	-	-	0.7	-	0.2	**0.1**
Typhlogobius californiensis	blind goby	0.1	-	-	-	0.3	-	0.3	-	0.2	**0.1**
Strongylura exilis	California needlefish	0.9	-	-	-	-	-	-	-	-	**0.1**
Ruscarius creaseri	roughcheek sculpin	0.3	-	0.3	-	-	-	-	-	0.2	**0.1**
Leptocottus armatus	Pacific staghorn sculpin	-	-	-	0.2	-	-	0.3	0.3	-	**0.1**
Artedius spp.	sculpins	-	-	-	-	0.3	-	-	-	0.2	**0.1**
Hyporhamphus rosae	California halfbeak	0.4	0.2	-	-	-	-	-	-	-	**0.1**
Paralichthyidae	lefteye flounders & sanddabs	-	-	-	-	-	0.3	-	0.2	-	**0.1**
Cottidae	sculpins	-	-	-	-	0.2	-	-	0.2	-	**0.1**
Oligocottus spp.	sculpins	-	-	-	-	-	-	0.2	0.2	-	**0.1**
Pleuronichthys ritteri	spotted turbot	-	-	-	-	-	-	-	0.4	-	**0.1**
Atractoscion nobilis	white seabass	-	-	-	-	0.2	-	-	0.2	-	**<0.1**
Porichthys myriaster	specklefin midshipman	-	-	-	-	-	0.3	-	-	-	**<0.1**
Clupeidae	herrings	-	-	-	-	-	-	0.3	-	-	**<0.1**
Nannobrachium spp.	lanternfishes	-	-	-	-	-	-	0.2	-	-	**<0.1**
Gobiesox rhessodon	California clingfish	-	-	-	-	-	0.2	-	-	-	**<0.1**
Sebastes spp.	rockfishes	-	-	-	-	-	-	0.2	-	-	**<0.1**
Citharichthys spp.	sanddabs	-	-	-	-	-	-	-	-	0.2	**<0.1**
	Station Total	**2,744.3**	**2,155.7**	**2,801.3**	**3,231.0**	**4,245.4**	**5,587.0**	**5,728.8**	**3,474.2**	**3,024.3**	

Table 3.3-6. Mean larval fish concentrations (larvae per 1000 m^3) by station for bi-monthly surveys from December 2002 through October 2003.

Taxon	Common Name	Stations									Mean
		SB1	SB2	SB3	SB4	SB5	SB6	SB7	SB8	SB9	
CIQ goby complex	gobies	1,715.7	1,498.4	1,509.7	1,515.7	2,490.4	1,271.9	3,028.5	1,912.9	1,682.8	**1,847.3**
Anchoa spp.	anchovy	119.8	244.7	92.8	164.1	301.0	319.3	230.9	252.8	152.0	**208.6**
Hypsoblennius spp.	combtooth blennies	30.0	41.7	178.8	113.4	214.0	55.5	330.7	122.0	473.0	**173.2**
Atherinopsidae	silversides	12.5	7.4	3.4	71.7	3.5	28.8	2.7	10.3	3.2	**16.0**
Syngnathus spp.	pipefishes	5.6	7.1	12.4	6.2	23.2	13.3	19.6	7.6	8.8	**11.6**
Paralabrax spp.	sand basses	-	-	1.4	-	20.9	11.5	28.9	5.2	5.7	**8.2**
Hypsopsetta guttulata	diamond turbot	-	0.6	2.0	6.7	5.7	7.7	19.7	5.5	13.4	**6.8**
Gillichthys mirabilis	longjaw mudsucker	32.5	0.4	4.1	0.5	7.6	-	3.1	-	-	**5.4**
Sciaenidae	croakers	1.0	0.4	10.5	-	11.2	8.9	4.8	-	2.9	**4.4**
Labrisomidae	labrisomid kelpfishes	-	0.4	0.9	7.2	3.3	1.7	3.2	3.3	12.1	**3.6**
Paralichthys californicus	California halibut	-	-	1.0	0.5	4.2	3.3	10.4	0.7	4.9	**2.8**
Engraulis mordax	northern anchovy	0.3	0.4	-	-	11.4	0.4	7.9	1.4	3.1	**2.8**
Acanthogobius flavimanus	yellowfin goby	1.3	1.5	0.5	1.8	0.4	0.6	-	1.7	1.0	**1.0**
Cheilotrema saturnum	black croaker	-	-	-	-	1.3	2.0	2.1	2.7	0.5	**1.0**
Genyonemus lineatus	white croaker	-	-	-	-	1.4	-	0.9	-	-	**0.3**
Typhlogobius californiensis	blind goby	-	-	-	-	0.9	-	0.9	-	0.5	**0.2**
Hippocampus ingens	Pacific seahorse	-	-	0.5	0.9	-	-	0.4	0.4	-	**0.2**
Atractoscion nobilis	white seabass	-	-	-	-	-	-	0.8	-	1.2	**0.2**
Paralichthyidae	lefteye flounders & sanddabs	-	-	-	-	-	0.4	-	-	1.2	**0.2**
Menticirrhus undulatus	California corbina	-	-	-	-	-	-	-	-	1.1	**0.1**
Gibbonsia spp.	clinid kelpfishes	-	-	-	1.1	-	-	-	-	-	**0.1**
Citharichthys stigmaeus	speckled sanddab	-	-	-	-	-	0.4	-	-	0.6	**0.1**
Lepidogobius lepidus	bay goby	-	-	-	-	-	-	-	-	0.6	**0.1**
Labridae	wrasses	-	-	-	-	-	-	-	-	0.6	**0.1**
Gobiesox rhessodon	California clingfish	-	-	-	-	-	-	-	-	0.4	**0.1**
	Station Total	**1,918.7**	**1,802.9**	**1,817.8**	**1,890.0**	**3,100.4**	**1,725.6**	**3,695.2**	**2,326.6**	**2,369.3**	

Table 3.3-7. Bray-Curtis scores comparing larval fish composition by sampling stations in south San Diego Bay: a) 2001 period (February 2001–January 2002) and b) 2003 period (December 2002–October 2003).

a) 2001

Station	SB1	SB2	SB3	SB4	SB5	SB6	SB7	SB8	SB9
SB1	–	–	–	–	–	–	–	–	–
SB2	86.69	–	–	–	–	–	–	–	–
SB3	81.20	83.48	–	–	–	–	–	–	–
SB4	76.22	83.21	81.89	–	–	–	–	–	–
SB5	73.90	76.06	80.67	73.49	–	–	–	–	–
SB6	75.38	82.19	79.85	83.33	79.99	–	–	–	–
SB7	66.53	68.71	76.42	67.57	88.39	73.46	–	–	–
SB8	70.05	78.24	78.03	80.53	79.88	80.22	78.24	–	–
SB9	63.94	67.85	73.29	72.50	82.28	74.03	82.19	80.31	–

b) 2003

Station	SB1	SB2	SB3	SB4	SB5	SB6	SB7	SB8	SB9
SB1	–	–	–	–	–	–	–	–	–
SB2	87.13	–	–	–	–	–	–	–	–
SB3	78.13	79.90	–	–	–	–	–	–	–
SB4	75.74	82.09	77.85	–	–	–	–	–	–
SB5	67.12	68.70	81.42	69.64	–	–	–	–	–
SB6	70.20	77.03	80.15	77.61	82.96	–	–	–	–
SB7	62.30	65.83	77.18	68.46	91.50	79.77	–	–	–
SB8	73.77	82.67	79.15	85.26	80.07	85.15	78.85	–	–
SB9	63.29	69.07	74.89	73.18	79.75	79.50	82.34	79.99	–

Table 3.3-8. SIMPER scores for larval fishes (cumulative 75%) by station groups in south San Diego Bay from February 2001 through January 2002.

Taxon	Common Name	Contribution Percent	Cumulative Percent
Groups: East Bay Shallow & West Bay Shallow			
Gillichthys mirabilis	longjaw mudsucker	10.2	10.2
Acanthogobius flavimanus	yellowfin goby	8.1	18.2
Labrisomidae	kelpfishes	6.8	25.0
Hypsopsetta guttulata	diamond turbot	6.5	31.5
Atherinopsidae	silversides	6.2	37.8
Engraulis mordax	northern anchovy	6.1	43.9
Genyonemus lineatus	white croaker	5.7	49.6
Anchoa spp.	bay anchovies	5.4	55.1
Hypsoblennius spp.	combtooth blennies	5.0	60.0
Gibbonsia spp.	kelpfishes	4.0	64.0
Cheilotrema saturnum	black croaker	3.8	67.8
Syngnathus spp.	pipefishes	3.5	71.3
Paralabrax spp.	sand basses	3.3	74.6
CIQ goby complex.	CIQ gobies	3.2	77.8
Groups: East Bay Shallow & Deep Channel			
Paralabrax spp.	sand basses	10.4	10.4
Engraulis mordax	northern anchovy	9.0	19.4
Hypsoblennius spp.	combtooth blennies	8.5	27.9
Hypsopsetta guttulata	diamond turbot	8.1	36.0
Sciaenidae	croaker	6.4	42.4
Cheilotrema saturnum	black croaker	6.0	48.4
Genyonemus lineatus	white croaker	5.6	53.9
Anchoa spp.	anchovy	5.3	59.2
Labrisomidae	labrisomid kelpfishes	5.0	64.1
Gillichthys mirabilis	longjaw mudsucker	4.5	68.6
Atherinopsidae	silversides	4.1	72.7
Paralichthys californicus	California halibut	3.0	75.6

Table 3.3-9. SIMPER scores for larval fishes (cumulative 75%) by station groups in south San Diego Bay from December 2002 through October 2003.

Taxon	Common Name	Contribution Percent	Cumulative Percent
Groups: East Bay Shallow & West Bay Shallow			
Gillichthys mirabilis	longjaw mudsucker	13.0	13.0
Hypsopsetta guttulata	diamond turbot	11.4	24.4
Paralabrax spp.	sand basses	10.3	34.7
Atherinopsidae	silversides	10.2	44.9
Labrisomidae	kelpfishes	9.3	54.2
Sciaenidae	croakers	9.0	63.1
Hypsoblennius spp.	combtooth blennies	6.5	69.6
Cheilotrema saturnum	black croaker	6.1	75.7
Groups: East Bay Shallow & Deep Channel			
Paralabrax spp.	sand basses	12.5	12.5
Hypsopsetta guttulata	diamond turbot	9.9	22.3
Engraulis mordax	northern anchovy	9.2	31.5
Paralichthys californicus	California halibut	8.6	40.1
Hypsoblennius spp.	combtooth blennies	8.3	48.4
Labrisomidae	kelpfishes	7.5	56.0
Gillichthys mirabilis	longjaw mudsucker	7.0	63.0
Sciaenidae	croakers	5.6	68.5
Cheilotrema saturnum	black croaker	3.9	72.4
Syngnathus spp.	pipefishes	3.5	76.0

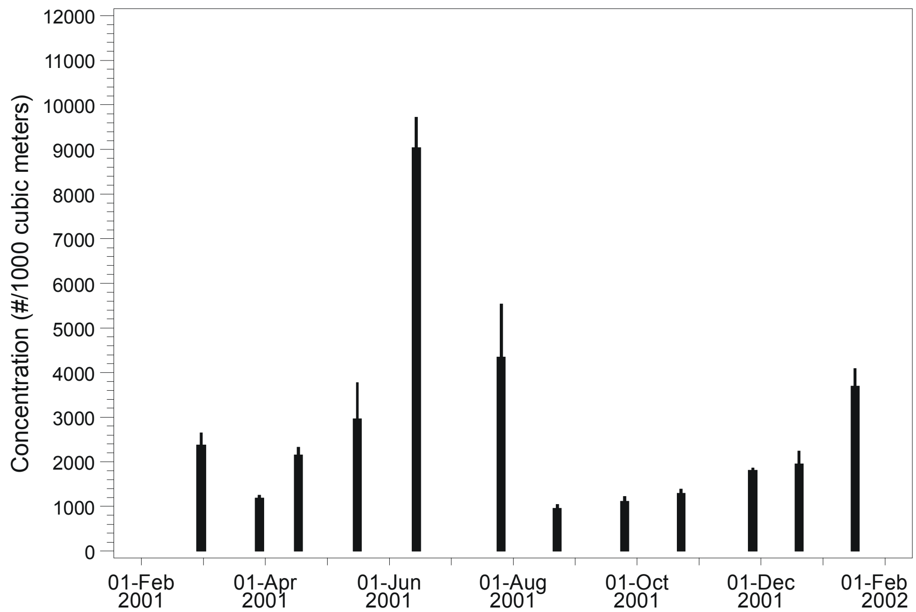


Figure 3.3-1. Mean concentration (# / 1,000 m^3) and standard error of all larval fishes collected at entrainment station SB1 during the 2001 period.

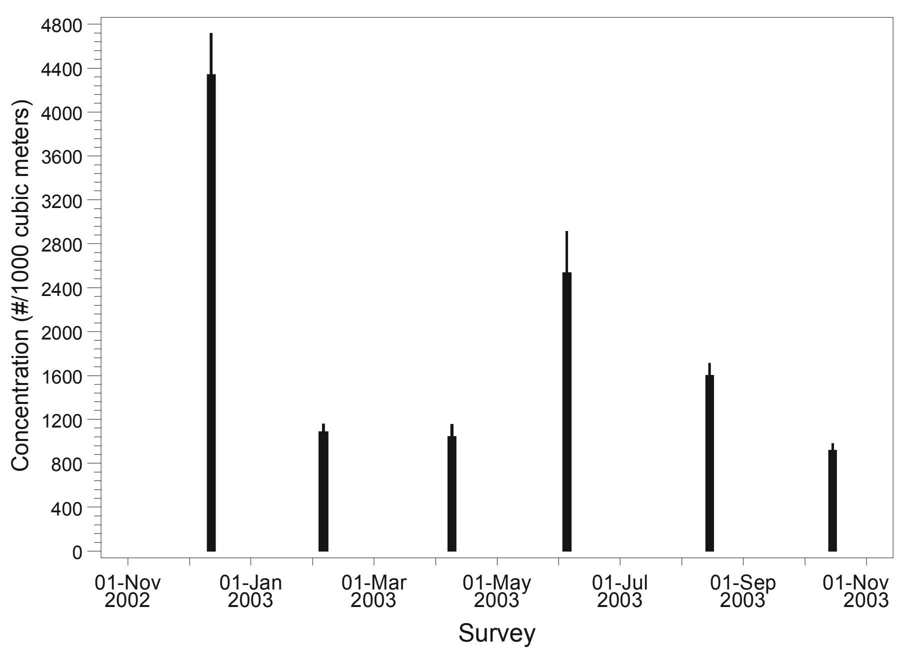


Figure 3.3-2. Mean concentration (# / 1,000 m^3) and standard error of all larval fishes collected at entrainment station SB1 during the 2003 period.

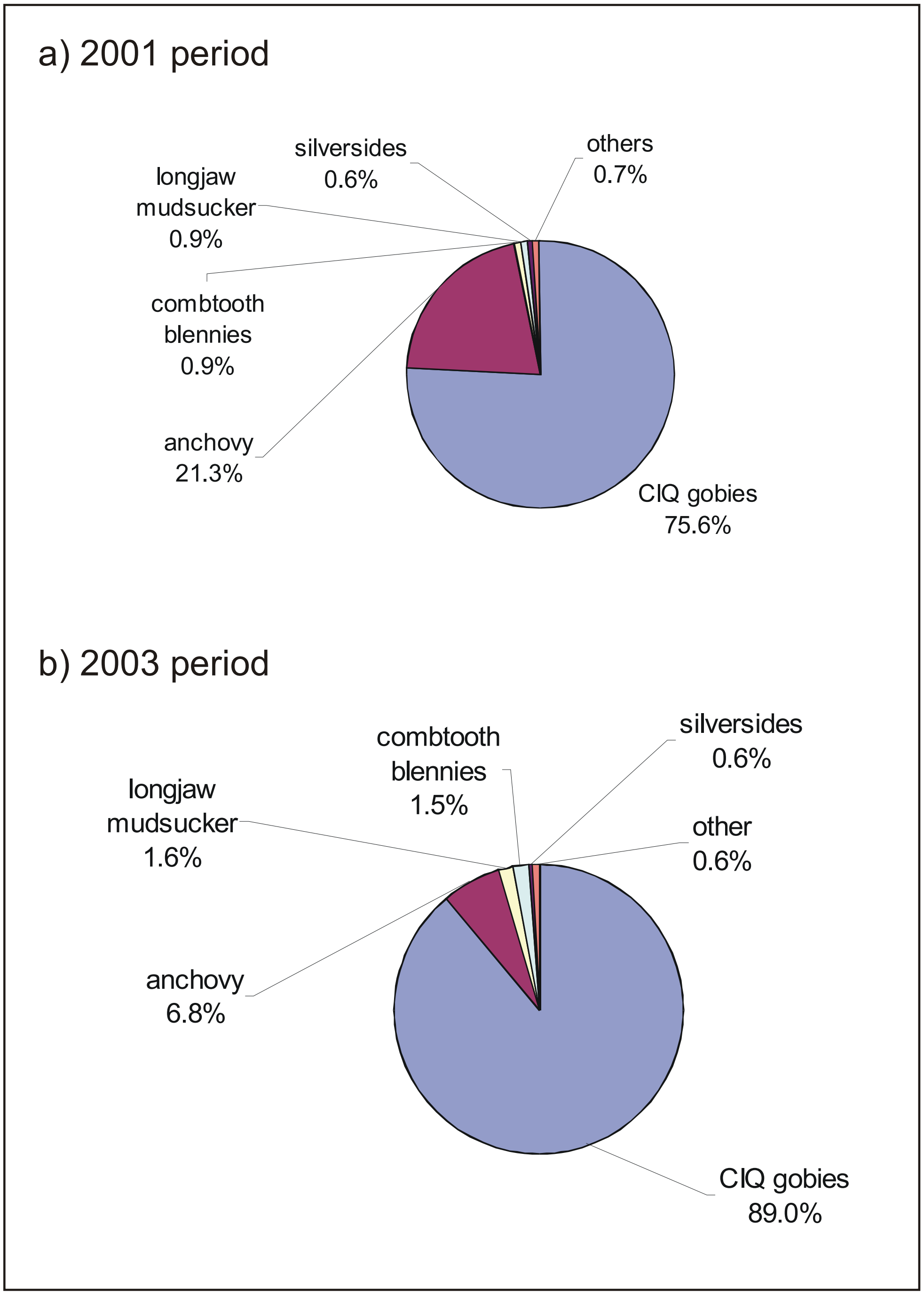


Figure 3.3-3. Percent composition of estimated total entrainment for a) 2001 period and b) 2003 period. The percentages for the taxa comprising the top 99 percent of the total estimated entrainment are listed while the remaining taxa are combined into 'others'.

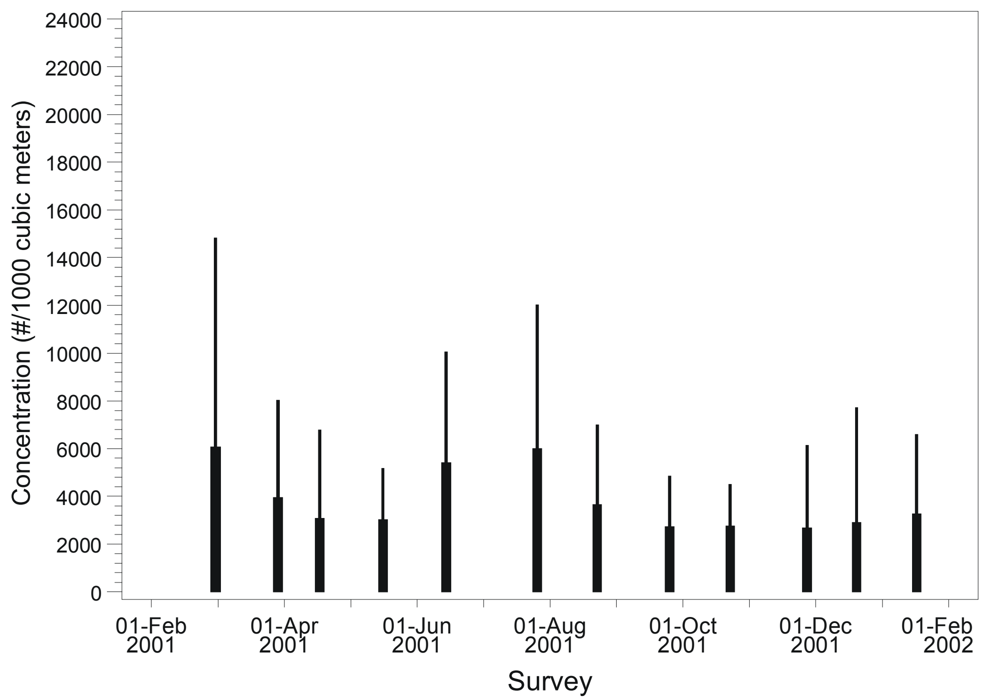


Figure 3.3-4. Mean concentration (# / 1,000 m^3) and standard error of all larval fishes collected at source water stations SB2–SB9 during the 2001 period.

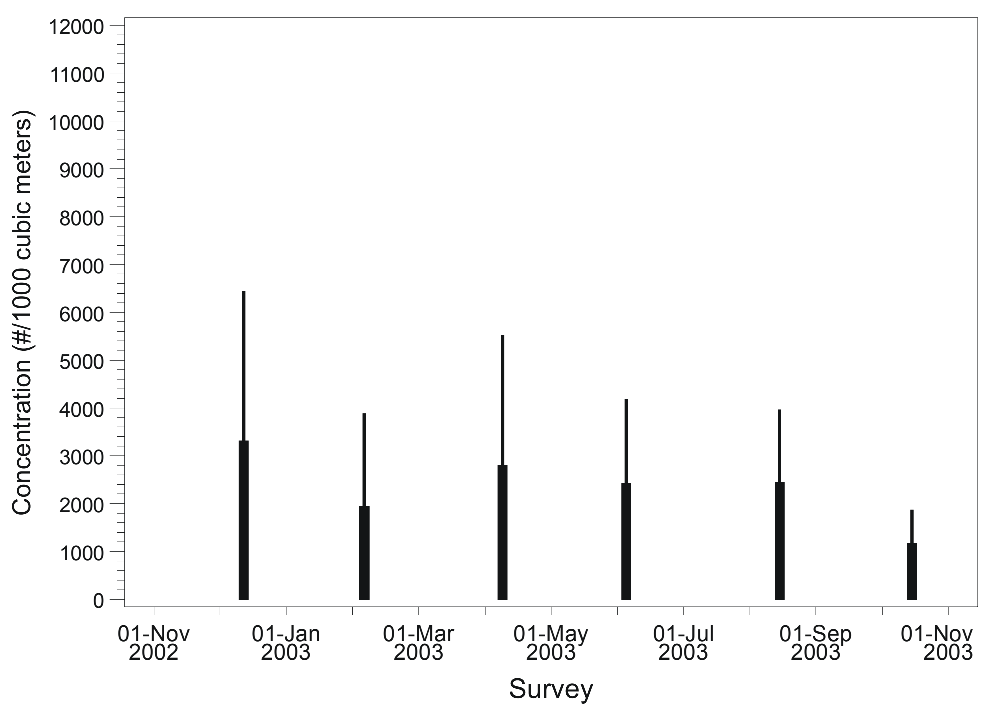
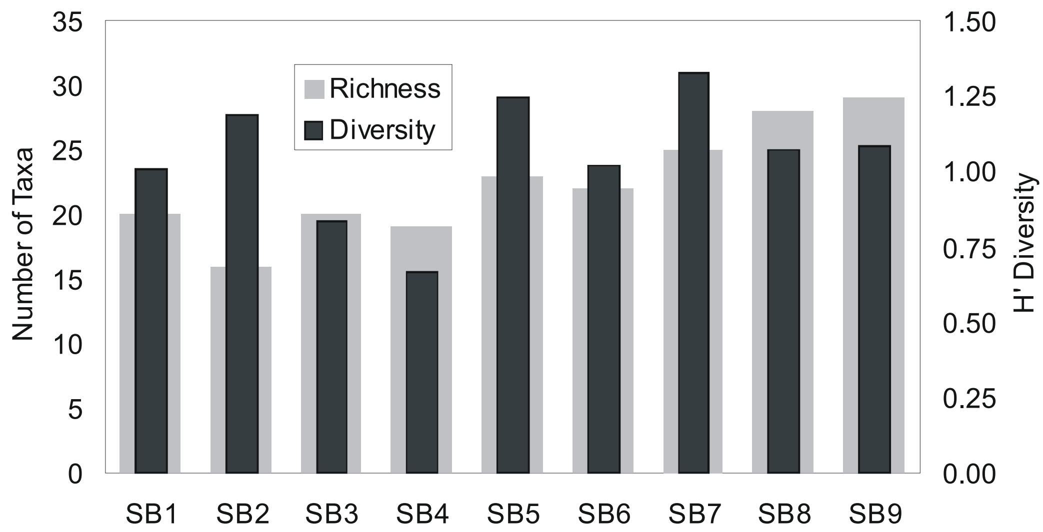


Figure 3.3-5. Mean concentration (# / 1000 m^3) and standard error of all larval fishes collected at source water stations SB2–SB9 during the 2003 period.

a) 2001



b) 2003

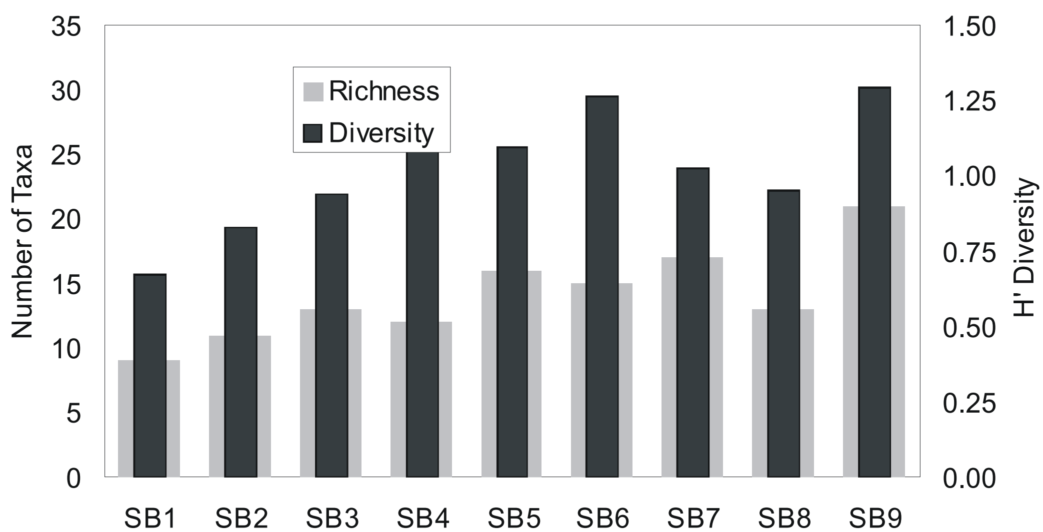
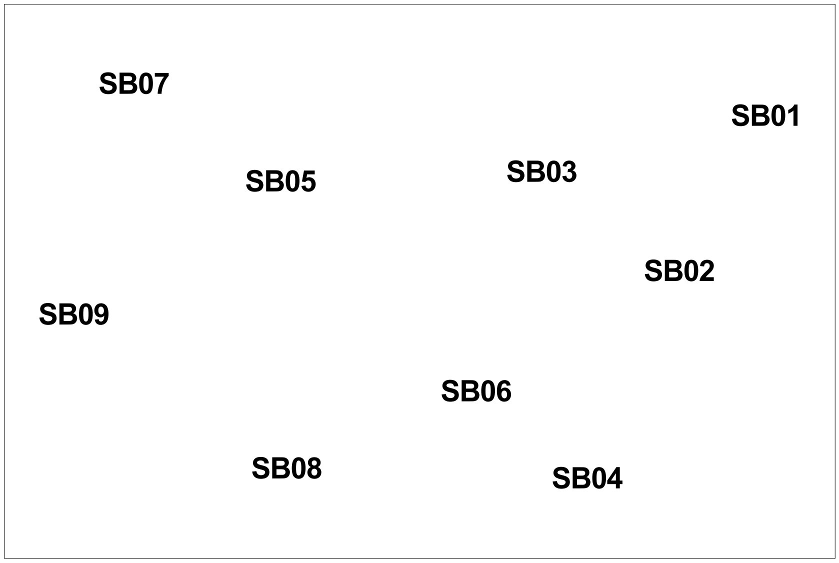


Figure 3.3-6. Taxa richness (number of taxa) and diversity (Shannon-Weiner H') computed from the annual mean concentrations of larval fishes at all stations (SB1–SB9) in south San Diego Bay: a) 2001 period and b) 2003 period.

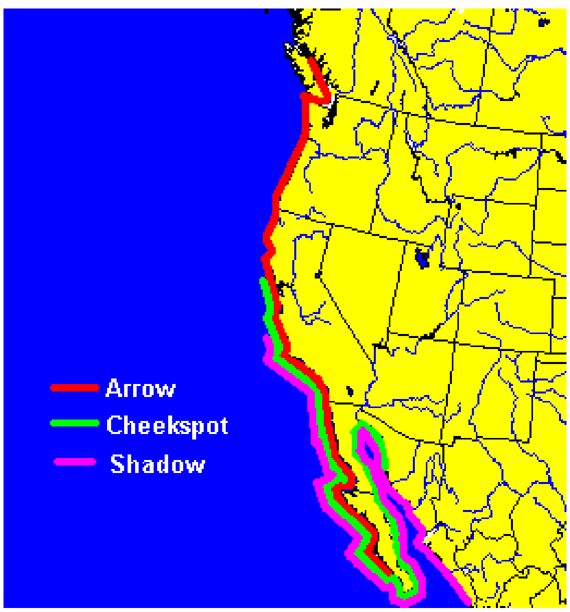
a) 2001



b) 2003

Figure 3.3-7. Non-metric multidimensional scaling analysis of Bray-Curtis dissimilarities of annual mean concentrations of larval fishes at all nine south San Diego Bay stations for the a) 2001 period and b) 2003 period. The relative positions of the stations are plotted on the graphs using the station labels. Refer to Figure 3.2-1 for station locations.

3.3.2 CIQ Goby complex (*Clevelandia ios, Ilypnus gilberti, Quietula y-cauda*)



Distribution map for CIQ gobies.

Range: Vancouver Island, British Columbia to Gulf of California;

Life History: Size up to 50 mm (2 in);
Age at maturity from 0.7–1.5 yr;
Life span ranges from <3 yr (arrow goby) to 5 yr (shadow goby);
Spawns year-round in bays and estuaries; demersal, adhesive eggs with fecundity from 225–1,400 eggs per female with multiple spawning 2–5 per yr;
Juveniles from 14.0–29.0 mm are less than 1 yr old;

Habitat: Mud and sand substrates of bays and estuaries; commensally in burrows of shrimps and other invertebrates.

Fishery: None.

3.3.2.1 Life History

Gobies belong to a speciose family (Gobiidae) of small, demersal fishes that are found worldwide in shallow tropical and subtropical environments. The family contains approximately 1,875 species in 212 genera (Nelson 1994, Moser et al. 1996). Twenty-one goby species from 16 genera occur from the northern California border to south of Baja California (Moser et al. 1996) and six species were found in San Diego Bay during a five-year study (Allen 1999).

Members of the goby family share a variety of distinguishing characteristics. Their body shape is elongate and can be either somewhat compressed or depressed (Moser et al. 1996). Most members of the family lack both a lateral line and swim bladder (Moyle and Cech 1988). Gobies generally have two dorsal fins, the first consisting of 2–8 flexible spines and the second containing a spine and several segmented rays. Their caudal fin is rounded and their pelvic fins are typically joined to form a cup-like disc (Moser et al. 1996). The eyes of most gobies are relatively large and are a dominant feature of their blunt heads. Goby species are extremely variable in coloration. They range from the drab, cryptically colored species that inhabit mudflats to the striking, brightly colored species of tropical and subtropical reefs (Moser et al. 1996).

One of the most important characteristics of the goby family is their small size. Due to their size and evolved tolerances for a variety of environmental conditions, gobies have been able to colonize habitats that are inaccessible to most other fishes. These include cracks and crevices in coral reefs, invertebrate burrows, mudflats, mangrove swamps, freshwater streams on oceanic islands, and inland seas and estuaries (Moyle and Cech 1988).

Gobies generally occur in shallow marine habitats, however many members of the family are euryhaline and are able to tolerate very low salinities and even freshwater. A number of goby species also have the ability to survive out of the water by “breathing” air. The longjaw mudsucker *Gillichthys mirabilis* can survive for days out of water if kept moist, and the mudskipper *Periopthalamus* spp. regularly leaves the water to forage for terrestrial insects among mangrove roots and exposed rocks (Moyle and Cech 1988). Gobies eat a variety of larval, juvenile, and adult crustaceans, mollusks, and insects. Many will also eat small fishes, fish eggs, and fish larvae.

Arrow goby *Clevelandia ios* occupy the most northerly range of the three species, occurring from Vancouver Island, British Columbia to Baja California (Eschmeyer et al. 1983). The reported northern range limits of both shadow goby *Quietula y-cauda* and cheekspot goby *Ilypnus gilberti* are in central California with sub-tropical southern ranges that extend well into the Gulf of California (Robertson and Allen 2002). Their physiological tolerances reflect their geographic distributions with arrow goby being less able to withstand warmer temperatures compared to cheekspot goby. When exposed to temperatures of 32.1°C for three days in a laboratory experiment, no arrow goby survived but 95 percent of cheekspot goby survived (Brothers 1975). Gobies exposed to warm temperatures on mudflats can seek refuge in their burrows where temperatures can be several degrees cooler than surface temperatures.

All three species have overlapping ranges in the San Diego region and occupy similar habitats. Arrow goby is generally the most abundant of the three species in San Diego Bay (juveniles and adults), followed by cheekspot and shadow gobies (Allen 1999). The life history of the arrow goby was reviewed by Emmett et al. (1991) and the comparative ecology and behavior of all three species were studied by Brothers (1975) in Mission Bay, approximately 6 km (3.7 mi) north of San Diego Bay. Arrow goby is the most abundant of the three species in bays and estuaries from Tomales Bay to San Diego Bay, including Elkhorn Slough (Calliet et al. 1977), Anaheim Bay (MacDonald 1975) and Newport Bay (Allen 1982). The species inhabits burrows of ghost shrimps *Neotrypnea* spp. and other burrowing invertebrates. In a 5-year study of fishes in San Diego Bay, approximately 75 percent of the estimated 4.5 million (standing stock) gobies were juveniles (Allen et al. 2002). Gobies were among the most abundant species sampled in the SBPP discharge canal from 1997 to 2000, but still only accounted for less than 1 percent of the total catch which was dominated by slough and deepbody anchovy (Merkel and Associates Inc. 2000a).

Myomere counts, gut proportions, and pigmentation characteristics can be used to identify most fish larvae to the species level. However, the arrow, cheekspot, and shadow gobies cannot be differentiated with complete confidence at most larval stages (Moser et al. 1996). Therefore, larval gobies collected during SBPP entrainment sampling that could not be identified to the species level were grouped into the 'CIQ' goby complex (for *Clevelandia*, *Ilypnus* and *Quietula*), or the family level 'Gobiidae' if specimens were damaged but could still be recognized as gobiids. Some larger larval specimens with well-preserved pigmentation patterns could be identified to the species level (W. Watson, Southwest Fisheries Science Center, pers. comm.) but those that were speciated in this study were subsequently combined into the CIQ complex for analysis.

The reproductive biology is similar among the three species in the CIQ complex. Arrow goby typically mature sooner than the other two species, attaining 50 percent maturity in the population after approximately 8 mo as compared to 16–18 mo for cheekspot and shadow gobies. Mature females for all three of these species are oviparous and produce demersal eggs that are elliptical in shape, typically adhesive, and attached to a nest substratum at one end (Matarese et al. 1989, Moser et al. 1996). Hatched larvae are planktonic and the duration of the planktonic stage was estimated at 60 d for populations in Mission Bay (Brothers 1975). Arrow goby mature more quickly and spawn a greater number of eggs at a younger age than either the cheekspot or shadow gobies. Fecundity is dependent on age and size of the female. For the Mission Bay populations of gobies Brothers (1975) measured fecundity ranged from 225–750 eggs per batch for arrow gobies (depending on adult size), 225–1,030 eggs for cheekspot, and 340–1,400 for shadow, for a mean value of 615 per batch for the CIQ complex. Mature females for the CIQ complex deposit 2–5 batches of eggs per year.

CIQ complex larvae hatch at a size of 2–3 mm (Moser et al. 1996). Data from Brothers (1975) were used to estimate an average growth rate of 0.16 mm/d for the approximately 60 days period from hatching to settlement. Brothers (1975) estimated a 60-day larval mortality of 98.3 percent for arrow goby larvae, 98.6 percent for cheekspot, and 99.2 percent for shadow. These values were used to estimate average daily survival at 0.93 for the three species. Once the larvae transform at a size of approximately 10–15 mm SL, depending on the species (Moser et al. 1996), the juveniles settle into the benthic environment. For the Mission Bay populations mortality following settlement was 99 percent per year for arrow goby, 66–74 percent for cheekspot goby, and 62–69 percent for shadow goby. Few arrow gobies in the Mission Bay study exceeded 3 yr of age based on otolith records, whereas cheekspot and shadow gobies commonly lived for 4 yr (Brothers 1975).

3.3.2.2 Sampling Results

CIQ complex goby larvae were the most abundant taxon collected during both sampling periods at both the entrainment and source water stations (**Tables 3.3-1**, **-2**, **-3**, and **-4**). Total annual entrainment estimates for the 2001 and 2003 sampling periods were 1.83 x 10^9 and 1.39 x 10^9 larvae, respectively (**Tables 3.3-1** and **3.3-2**). Entrainment estimates for each survey are presented in **Appendix D**. CIQ goby larvae were most abundant at the entrainment station during June and July of the 2001 sampling period, but tended to be more abundant at source water stations than the entrainment station in all but those months (**Figure 3.3-8a**). During the 2003 period peak abundances occurred in February at both entrainment and source water stations (**Figure 3.3-8b**). Variation in abundance probably reflects differences in the spawning periods for the three species comprising the complex. Brothers (1975) indicated that the peak spawning period for arrow goby occurs from November through April, while spawning in cheekspot and shadow goby is more variable and can occur throughout the year. A peak spawning period for shadow goby in June and July of Brothers' (1975) study corresponds to the increased larval abundances during those months in 2001 of this study. CIQ gobies were consistently more abundant in 2001 than 2003 among the source water stations. They were abundant at all stations during both periods and tended to occur in higher concentrations at Stations SB5 and SB7 adjacent to the Sweetwater River channel (**Figure 3.3-9**).

The length frequency distribution for a representative sample of CIQ goby larvae showed that the majority of the larvae were recently hatched based on the reported hatch size of 2–3 mm (Moser et al. 1996). The mean length for both sampling periods and the combined data was 3.1 mm (**Figures 3.3-10a, b**).

3.3.2.3 Circulating Water System Impact Assessment

The following sections present the results for demographic and empirical transport modeling of circulating water system effects. A comprehensive comparative study of the three goby species in the CIQ complex by Brothers (1975) provided the necessary life history information for both the *FH* and *AEL* demographic models.

Fecundity Hindcasting (*FH*)

The two annual entrainment estimates for CIQ gobies were used to estimate the number of breeding females needed to produce the number of larvae entrained (**Tables 3.3-10a,**

10b). No estimates of egg survival for gobies were available, but because egg masses in gobies are demersal (Wang 1986) and parental care, usually provided by the adult male, is common in the family (Moser et al. 1996), egg survival is probably high and was assumed to be 100 percent. Estimates of larval survival for the three species from Brothers (1975) were used to estimate an average daily survival of 0.93. A larval growth rate of 0.16 mm/d was estimated from Brothers (1975) using his reported transformation lengths for the three species and an estimated transformation age of 60 d. The mean length and the length of the first percentile (2.18 mm) were used with the growth rate to estimate that the mean age at entrainment was 5.8 d. Survival to the average age at entrainment was then estimated as $0.93^{5.8} = 0.66$. An average batch fecundity estimate of 615 eggs was based on calculations from Brothers (1975) on size-specific fecundities for the three species. Brothers (1975) found eggs with two to three different vittelogenic stages in the ovaries. Therefore, an estimate of 2.5 spawns per year was used in calculating *FH* (615 eggs/spawn × 2.5 spawns/year = 1,538 eggs/year). Average ages of maturity and longevity of 1.0 and 3.3 years, respectively, from Brothers (1975) for the three species were used in the model.

The estimated numbers of adult females whose lifetime reproductive output was entrained through the SBPP circulating water system for the two periods were 1,084,781 in 2001 and 826,093 in 2003 (**Table 3.3-10**).

Table 3.3-10. Results of *FH* modeling for CIQ goby complex larvae entrained during the a) 2001 and b) 2003 sampling periods. The upper and lower estimates are based on a 90% confidence interval of the mean.

Parameter	Mean	Std. Error	*FH* Lower Estimate	*FH* Upper Estimate	*FH* Range
a) 2001 Period					
FH Estimate	**1,084,781**	1,880,405	62,654	18,781,809	18,719,155
Total Entrainment	1,830,898,760	21,724,769	960,927	1,208,640	247,713
b) 2003 Period					
FH Estimate	**826,093**	1,431,472	47,761	14,288,342	14,240,581
Total Entrainment	1,394,283,727	9,291,117	755,864	896,322	140,458

Adult Equivalent Loss (*AEL*)

The parameters required for formulation of *AEL* estimates include larval survival from entrainment to settlement and survival from settlement to the average age of reproduction for a mature female. Larval survival from entrainment through settlement was estimated as $0.93^{60-5.8} = 0.02$ using the same daily survival rate used in formulating *FH*. Brothers (1975) estimated that mortality in the first year following settlement was 99 percent for arrow, 66–74 percent for cheekspot, and 62–69 percent for shadow goby. These estimates were used to calculate a daily survival of 0.995 that was used to estimate a finite survival of 0.21 for the first year following settlement. Daily survival through the average female age of 1.71 years from life table data for the three species was estimated as 0.994 and was used to calculate a finite survival of 0.195.

The estimated number of adult CIQ gobies equivalent to the number of larvae entrained through the SBPP circulating water system for the two periods was 1,579,926 in 2001 and 1,206,161 in 2003 (**Table 3.3-11**).

Table 3.3-11. Results of *AEL* modeling for CIQ goby complex larvae entrained during the a) 2001 period and b) 2003 period. The upper and lower estimates are based on a 90% confidence interval.

Parameter	Mean	Std. Error	*AEL* Lower Estimate	*AEL* Upper Estimate	*AEL* Range
a) 2001 Period					
AEL Estimate	**1,579,926**	2,738,709	91,252	27,354,701	27,263,449
Total Entrainment	1,830,898,760	21,724,769	1,399,539	1,760,313	360,774
b) 2003 Period					
AEL Estimate	**1,203,161**	2,084,863	69,562	20,810,207	20,740,645
Total Entrainment	1,394,283,727	9,291,117	1,100,876	1,305,446	204,570

Empirical Transport Model (*ETM*)

The larval duration used to calculate the *ETM* estimates for CIQ gobies for both the 2001 and 2003 sampling periods was based on the lengths of entrained larvae combined for the two sampling periods. The difference between the lengths of the 1^{st} (2.2 mm) and 99^{th} (5.8 mm) percentiles was used with a growth rate of 0.16 mm/d to estimate that CIQ goby larvae were vulnerable to entrainment for a period of 22.9 days.

ETM estimates of P_m for CIQ gobies were similar for the two sampling periods (**Table 3.3-12**). *PE* estimates ranged from 0.004 to 0.02 for the two periods. The small range in both the *PE* estimates and the values of f_i within both of the sampling periods indicate that goby larvae are present in the source water throughout the year and are removed through entrainment at a relatively constant rate of 0.4 to 2.0 percent per day. This is also represented in the *PE* estimates that are similar in value to the ratio of the circulating water system to source water volumes (0.015). The largest fractions of the source water population during the 2001 sampling period occurred in the February (f_i = 0.2165) and July (f_i = 0.1064) surveys. The June and July surveys had the highest entrainment station concentrations during the 2001 sampling period (**Figure 3.3-8**), resulting in higher *PE* estimates for those surveys. Similarly, the highest *PE* estimates during the 2003 sampling period were associated with the surveys with the highest entrainment station concentrations (**Figure 3.3-9**).

Table 3.3-12. *ETM* parameters for CIQ goby complex larvae. *ETM* calculations based on South Bay source water volume of = 149,612,092 m^3, and daily circulating water volume = 2,275,244 m^3.

Survey Date	PE_i Estimate	PE_i Estimate Std. Error	f_i	f_i Std. Error
a) 2001 Period				
28-Feb-01	0.0057	0.0014	0.2165	0.0387
29-Mar-01	0.0045	0.0008	0.0977	0.0151
17-Apr-01	0.0109	0.0023	0.0491	0.0096
16-May-01	0.0175	0.0060	0.0475	0.0052
14-Jun-01	0.0247	0.0060	0.0620	0.0099
26-Jul-01	0.0225	0.0077	0.1064	0.0179
23-Aug-01	0.0038	0.0006	0.0676	0.0101
25-Sep-01	0.0070	0.0010	0.0704	0.0074
23-Oct-01	0.0075	0.0008	0.0661	0.0063
27-Nov-01	0.0105	0.0020	0.0774	0.0145
20-Dec-01	0.0103	0.0030	0.0584	0.0135
17-Jan-02	0.0173	0.0032	0.0811	0.0120
	P_m = 0.2147 *S.E.* = 0.4294			
b) 2003 Period				
12-Dec-02	0.0200	0.0032	0.2650	0.0285
6-Feb-03	0.0081	0.0011	0.1474	0.0171
9-Apr-03	0.0063	0.0011	0.2191	0.0242
5-Jun-03	0.0196	0.0036	0.1610	0.0170
15-Aug-03	0.0133	0.0020	0.1196	0.0140
15-Oct-03	0.0141	0.0013	0.0879	0.0070
	P_m = 0.2671 *S.E.* = 0.4739			

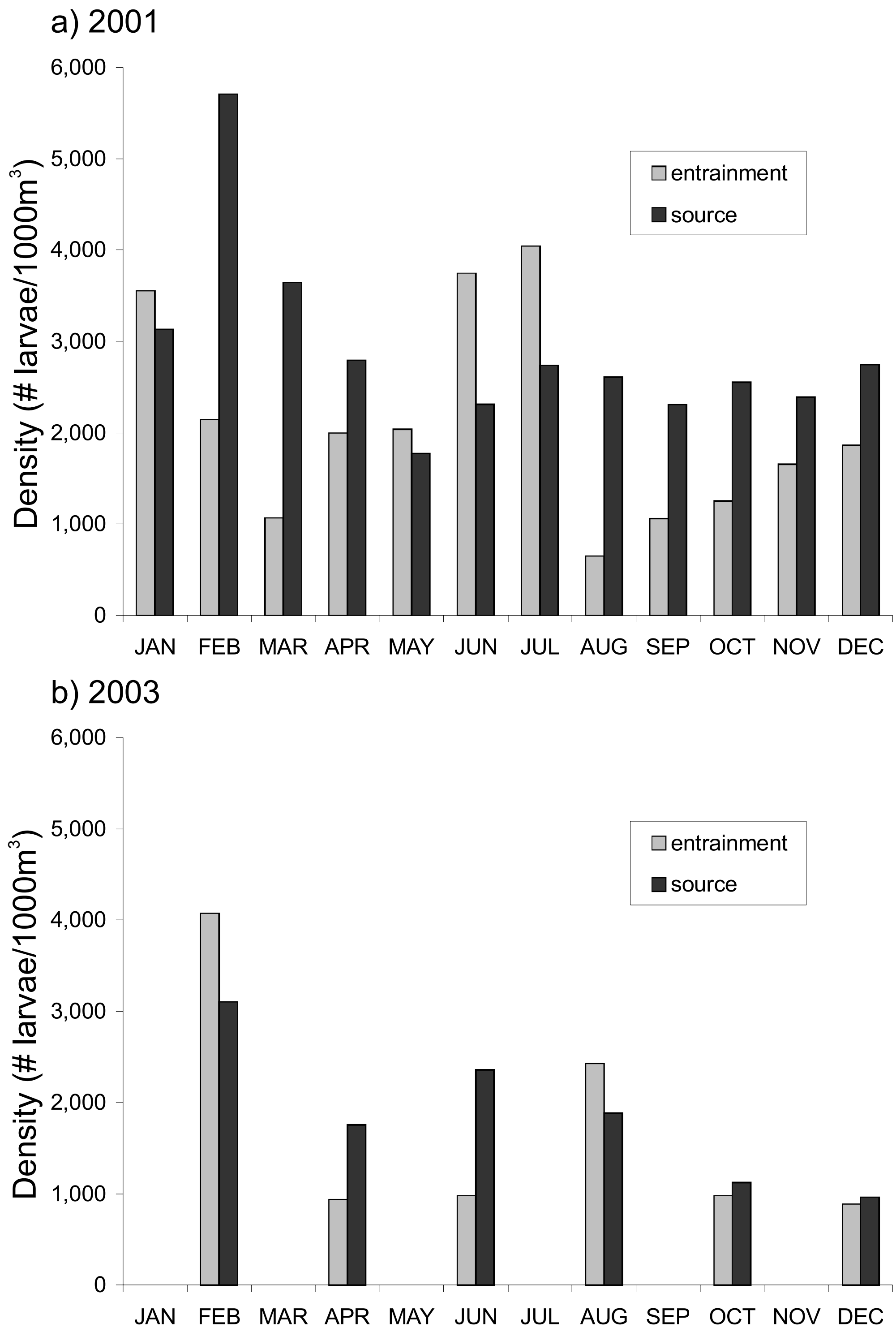


Figure 3.3-8. Comparison of mean density (#/1000 m^3) of CIQ goby complex larvae between entrainment station (SB1) and source water stations (SB2-SB9): a) 2001 period (Note: January values from 2002); b) 2003 period (Note: December values from 2002).

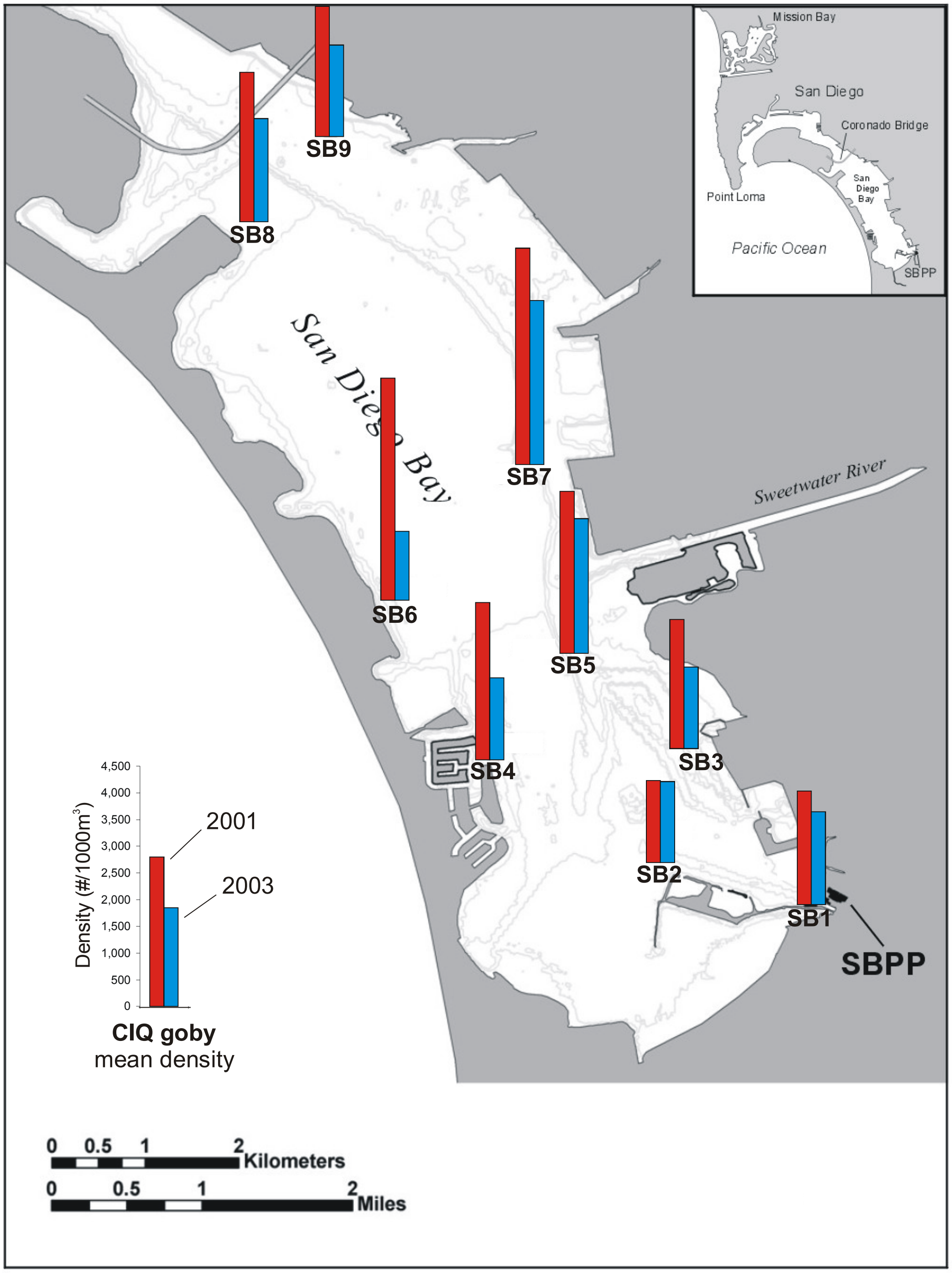


Figure 3.3-9. Annual mean density (#/1000 m^3) of CIQ goby complex larvae at entrainment station (SB1) and source water stations (SB2-SB9) in south San Diego Bay during the 2001 and 2003 sampling periods. Mean density for all stations combined is shown with scale.

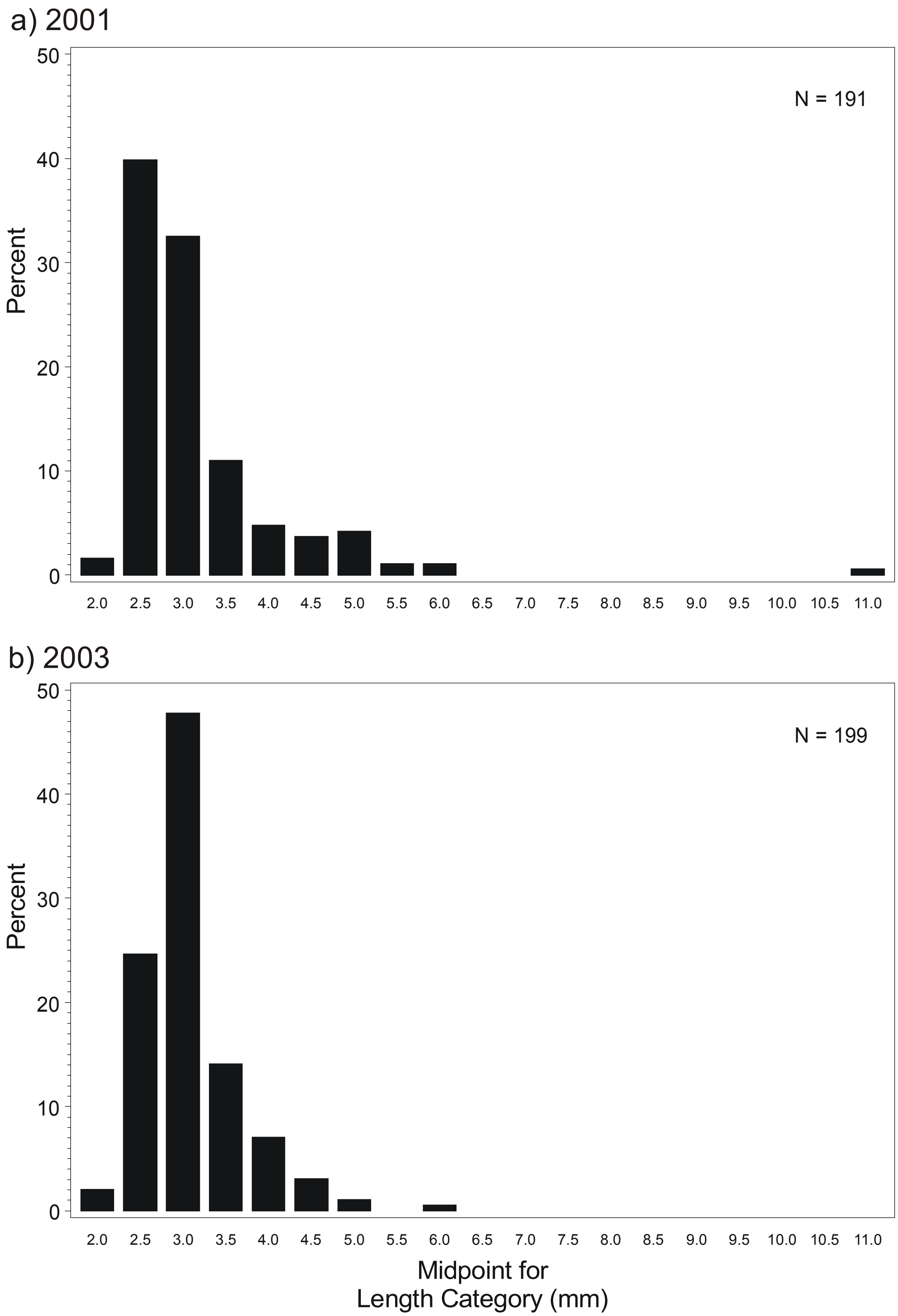
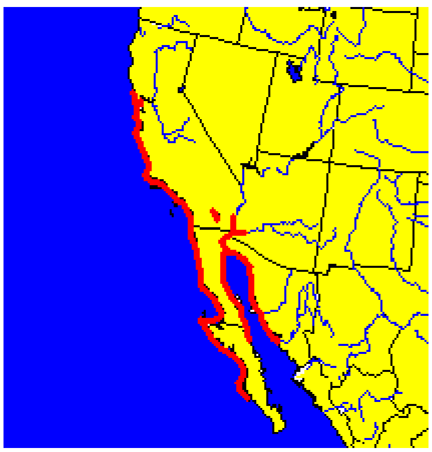


Figure 3.3-10. Length frequency distribution of CIQ goby complex larvae collected at entrainment station SB1: a) 2001 period, b) 2003 period.

3.3.3 Longjaw mudsucker (*Gillichthys mirabilis*)

Photo: Drew Talley



Distribution map for longjaw mudsucker.

Range: Tomales Bay, California to Bahia Magdalena, Baja California; in Arizona: the Salt River and the lower Colorado River; introduced into the Salton Sea, California;

Life History: Size: up to 210 mm (8 in);
Age at maturity: <1 yr;
Fecundity: spawns 2–3 times per season, batch size from 4,000–27,000 eggs;
Life span: 2 yr.

Habitat: Tidal flats, shallow brackish water, upper salt marsh channels;

Fishery: Small commercial and recreational bait fishery.

3.3.3.1 Life History

The longjaw mudsucker *Gillichthys mirabilis* (mudsucker) is a medium to large species of goby that commonly inhabits bays, estuaries, tidal sloughs, and salt ponds along the Pacific coast of North America. They are readily distinguished from other similar-looking gobies by their disproportionately long maxillary, which extends to near the margin of the gill opening. The native distribution of mudsuckers is in bay habitats from Tomales Bay, California to Magdelena Bay along the Pacific Coast, with an isolated population in the northern reaches of the Gulf of California (Wang 1986). An introduced population in the Salton Sea is descended from 500 individuals that were released by CDFG in November 1930 to establish a bait species for sportfishes (Barlow and De Vlaming 1972). Naturalized populations of introduced mudsuckers also occur in Arizona (Roosevelt Lake on the Salt River) and the lower Colorado River where they are commonly used as fishing bait.

Mudsuckers are able to tolerate a wide range of environmental conditions. The species can be abundant on tidal flats and in shallow muddy backwaters (Love 1996). They were the dominant fish in the upper reaches of a reconstructed tidal marsh in San Diego Bay, and are particularly well adapted to narrow channels with high salinities, low dissolved oxygen, and steep clay banks (Williams and Zedler 1999). They are able to live in water with salinities ranging from 80 ppt (about 2.5 times that of seawater) to nearly freshwater, and are able to withstand water temperatures as high as 35°C (95°F) for short durations although their preferred temperature range is between 9–23°C (48–73°F) (De Vlaming 1971, Love 1996). In addition to extracting oxygen from the water with their gills, mudsuckers can survive extended periods out of water by absorbing oxygen from air taken in (gulped) and held in their large and highly vascularized buccal cavity (Moyle and Cech 1988), and by limited cutaneous respiration through their fins (Barlow 1961). Mudsuckers often retreat into shrimp or crab burrows when tidal flats are exposed during low tides (Love 1996) but can also move short distances across the flats using pectoral locomotion (Todd 1968).

Mudsuckers were one of the least abundant species of goby collected in fish studies done by Allen (1999) in San Diego Bay, and they were only found in the southernmost region of the bay. They were rare (1 small individual collected) in samples from the discharge channel of SBPP during a three-year fish study from 1996–1999 (Merkel and Associates, Inc. 2000a).

Mudsuckers reach a maximum size of about 210 mm (8.3 in) and may live to about 2 years of age (Walker et al. 1961, Love 1996). They become sexually mature and are capable of spawning in their first year at a size of approximately 25–51 mm (1–2 in). After reaching maturity mudsuckers can spawn 2–3 times a year (Barlow and De Vlaming 1972). Spawning activity begins as early as November in San Francisco Bay and Tomales Bay (Wang 1986), peaks in spring, and may extend through July (Barlow and De Vlaming 1972). The timing of spawning is controlled by environmental cues such as seasonal changes in day length and water temperature (Moyle and Cech 1988). Females are oviparous and lay from 4,000 to 9,000 adhesive, club-shaped eggs, which are attached to the sides of the burrow with central stalks (Weisel 1947). The eggs are guarded by the male and require a 10–12 d incubation period at 18°C (64°F) (Weisel 1947, Wang 1986). Barlow (1961) reported an annual reproductive output of between 8,000 and 27,000 eggs per female.

After hatching at a size of 3–4 mm TL the larvae are pelagic and occur at all depths within the shallow water column. The larvae are easily recognized and distinct from other sympatric gobiids because of their distinct pigmentation pattern (Moser et al. 1996). Transformation from the planktonic to the benthic environment occurs at 8–12 mm TL (0.31–0.47 in) and is accompanied by an increase in dorsal pigmentation (Barlow 1963). Young mudsuckers grow rapidly, with 25–40 mm juveniles averaging a daily length increase of 0.54 mm per day, based on data from Weisel (1947). This estimate is compatible with data from Walker et al. (1961) who reported that young-of-the-year mudsuckers reached a size of 60–80 mm (2.4–3.1 in) by August of the year. If most of these juveniles grew from larvae that hatched shortly after the peak spawning period in late March and early April, their size would be the result of a daily growth rate of 0.55 mm per day ([70–3.5 mm]/122 days). The average growth rate for these early juvenile stages was used to estimate a larval duration of approximately 12 days based on an average length of 10 mm at settlement and hatch size of 3.5 mm. Growth rates slow by December with the modal size of yearling goby ranging from 80–115 mm SL (3.2–4.5 in) (Walker et al. 1961). Males were observed to grow slightly faster than females. Brothers (1975) reported a total larval mortality of 99 percent over a two-month period for the 'CIQ' goby complex. These values were used to estimate a daily larval survival of 0.927 that was used in the entrainment modeling.

Mudsuckers are carnivorous and juveniles feed on a variety of invertebrates and occasionally on small fishes. Their diet includes harpacticoids and other copepods, nematodes, and fly larvae of the family Heliidae (Walker et al. 1961, Wang 1986). As adults they feed on crustaceans such as crabs and ghost shrimp (Love 1996). In the Salton Sea, the most important food of adult mudsucker is the pile worm *Neanthes* spp., although they also consume barnacles, a variety of insect larvae, and occasionally Desert pupfish (Walker et al. 1961). Mudsuckers are preyed upon by many species of birds and fishes.

Mudsuckers are used as live bait in a variety of recreational fisheries, and especially for corvina *Cynoscion* spp. in the Salton Sea (Walker et al. 1961). Most mudsuckers used for bait are captured in cylindrical minnow traps. From 1987 through 1996 annual reported landings of mudsuckers in California ranged from 10 lb in 1994 to 557 lb in 1987. No landings were reported in 1990 or 1991.

3.3.3.2 Sampling Results

Longjaw mudsucker larvae were the fourth most abundant taxa of fish larvae collected in entrainment samples during the 2001 sampling period with the total annual entrainment estimate for that period being 2.20×10^{7} larvae (**Table 3.3-1**). The total annual entrainment estimate increased to 2.50×10^{7} larvae during the 2003 sampling period (**Table 3.3-2**). However, longjaw mudsucker larvae were less abundant in the source water samples than in entrainment samples during both sampling periods (**Tables 3.3-3, 3.3-4** and **Figure 3.3-11**). Entrainment estimates for each survey are presented in **Appendix D.** Longjaw mudsucker larvae were generally collected in highest abundance, in both the entrainment and source water samples, during the winter and early spring with no larvae being found during the summer. Wang (1986) indicated that spawning activity begins as early as November in San Francisco Bay and Tomales Bay. Barlow and De Vlaming (1972) state that spawning activity for the longjaw mudsucker peaks in spring and may extend through July. The skewed distribution of longjaw mudsucker larvae at the southeastern stations (**Figure 3.3-12**) corresponds with the distribution of adult habitat in the saltmarshes along the southeastern bay margins. Highest larval concentrations at the entrainment station SB1 reflected its proximity to the saltmarsh habitats located between SBPP and the Chula Vista Marina, and along the northern margin of the Chula Vista Wildlife Island.

The length frequency distribution of longjaw mudsucker larvae showed that a majority of the larvae were recently hatched based on the average of the reported hatch length of 3.5 mm (Moser et al. 1996). Mean lengths during the 2001 and 2003 sampling periods were 3.5 and 3.7 mm, respectively, with a mean length of 3.6 mm for the combined data (**Figure 3.3-17a, b**).

3.3.3.3 Circulating Water System Impact Assessment

The following sections present the results for demographic and empirical transport modeling of circulating water system effects. There was very little species-specific life history information available for longjaw mudsucker. Larval survival was estimated using data on other species of gobies from Brothers (1975) and there was enough information on longjaw mudsucker reproduction to parameterize the *FH* demographic model. Larval growth was estimated from information in Weisel (1947) and Walker et al. (1961). Not enough information was available to parameterize the *AEL* model, which was not calculated for this species.

Fecundity Hindcasting (*FH*)

The two annual entrainment estimates for longjaw mudsucker were used to estimate the number of breeding females needed to produce the number of larvae entrained (**Table 3.3-13**). No estimates of egg survival for longjaw mudsuckers were available, but because egg masses in gobies are demersal (Wang 1986) and parental care, usually provided by the adult male, is common in the family (Moser et al. 1996), egg survival is probably high and was assumed to be 100 percent. The mean length for larval longjaw mudsuckers in entrainment samples was 3.6 mm. A larval growth rate of 0.54 mm/d was derived from growth rates derived from data in Weisel (1947) and Walker (1961). The mean length and the length at the 1st percentile (3.0 mm) were used with the growth rate to estimate that the mean age at entrainment was 1.2 d. Brothers (1975) reports that a total mortality of 99 percent for a 2-month larval period is reasonable for other, similar, species of gobies. A daily survival rate of 0.93 was used to calculate survival to the average age at entrainment as $0.93^{1.2} = 0.91$. An average batch fecundity estimate of 6,500 eggs was based on data from Weisel (1947) and Barlow (1861). An estimate of 2.5 spawns per year was used based on information in Walker et al. (1961) and Barlow and De Vlaming (1972). Average ages of maturity and longevity of 1.0 and 2.0 yr (Walker et al. 1961), respectively, were used in the model.

The estimated number of adult females whose lifetime reproductive output was entrained through the SBPP circulating water system for the two periods were 1,478 in 2001 and 1,686 in 2003 (**Table 3.3-13**).

Table 3.3-13. Results of *FH* modeling for longjaw mudsucker larvae entrained during the a) 2001 period and b) 2003 period. The upper and lower estimates are based on a 90% confidence interval of the mean.

Parameter	Mean	Std. Error	*FH* Lower Estimate	*FH* Upper Estimate	*FH* Range
a) 2001 Period					
FH Estimate	**1,478**	2,564	85	25,656	25,571
Total Entrainment	21,953,225	405,184	1,235	1,722	487
b) 2003 Period					
FH Estimate	**1,686**	2,926	97	29,302	29,205
Total Entrainment	25,034,110	389,598	1,365	2,006	641

Empirical Transport Model (*ETM*)

The larval duration used to calculate the *ETM* estimates for longjaw mudsucker for both the 2001 and 2003 sampling periods was based on the lengths of entrained larvae combined for the two sampling periods. The difference between the lengths of the 1st (3.0 mm) and 99th (4.7 mm) percentiles was used with a growth rate of 0.54 mm/d to estimate that mudsucker larvae were vulnerable to entrainment for a period of 3.2 d.

The *ETM* estimate of P_m for longjaw mudsucker for the 2003 sampling period was much larger than the estimate for the 2001 sampling period (**Table 3.3-14**). *PE* estimates ranged from 0.00 to 0.92 for the two sampling periods. The *PE* estimates are generally larger than the ratio of the circulating water system to source water volumes (0.015). This is due to the greater abundances of mudsucker larvae at the entrainment station relative to the source water stations (**Tables 3.3-5** and **3.3-6**). During the 2003 sampling period mudsucker larvae weren't collected at three of the source water stations. This resulted in the high *PE* estimates and the large estimate of P_m for 2003.

Table 3.3-14. *ETM* parameters for longjaw mudsucker larvae. *ETM* calculations based on South Bay source water volume of = 149,612,092 m^3, and daily cooling water volume = 2,275,244 m^3.

Survey Date	PE_i Estimate	PE_i Estimate Std. Error	f_i	f_i Std. Error
a) 2001 Period				
28-Feb-01	0.0444	0.0142	0.3458	0.0514
29-Mar-01	0.0277	0.0175	0.0982	0.0464
17-Apr-01	0.4358	0.2728	0.0170	0.0092
16-May-01	0.9200	0.9614	0.0044	0.0044
14-Jun-01	—	—	—	—
26-Jul-01	—	—	—	—
23-Aug-01	—	—	—	—
25-Sep-01	0.0409	0.0520	0.0184	0.0144
23-Oct-01	0.0089	0.0101	0.0247	0.0132
27-Nov-01	0.1324	0.0482	0.1568	0.0448
20-Dec-01	0.0222	0.0081	0.1419	0.0276
17-Jan-02	0.0490	0.0157	0.1928	0.0426
	P_m = 0.1711 *S.E.* = 0.3925			
b) 2003 Period				
12-Dec-02	0.5946	0.2161	0.2750	0.0754
6-Feb-03	0.1299	0.0290	0.6634	0.0809
9-Apr-03	0.0336	0.0412	0.0616	0.0422
5-Jun-03	—	—	—	—
15-Aug-03	—	—	—	—
15-Oct-03	—	—	—	—
	P_m = 0. 5018 *S.E.* = 0. 5368			

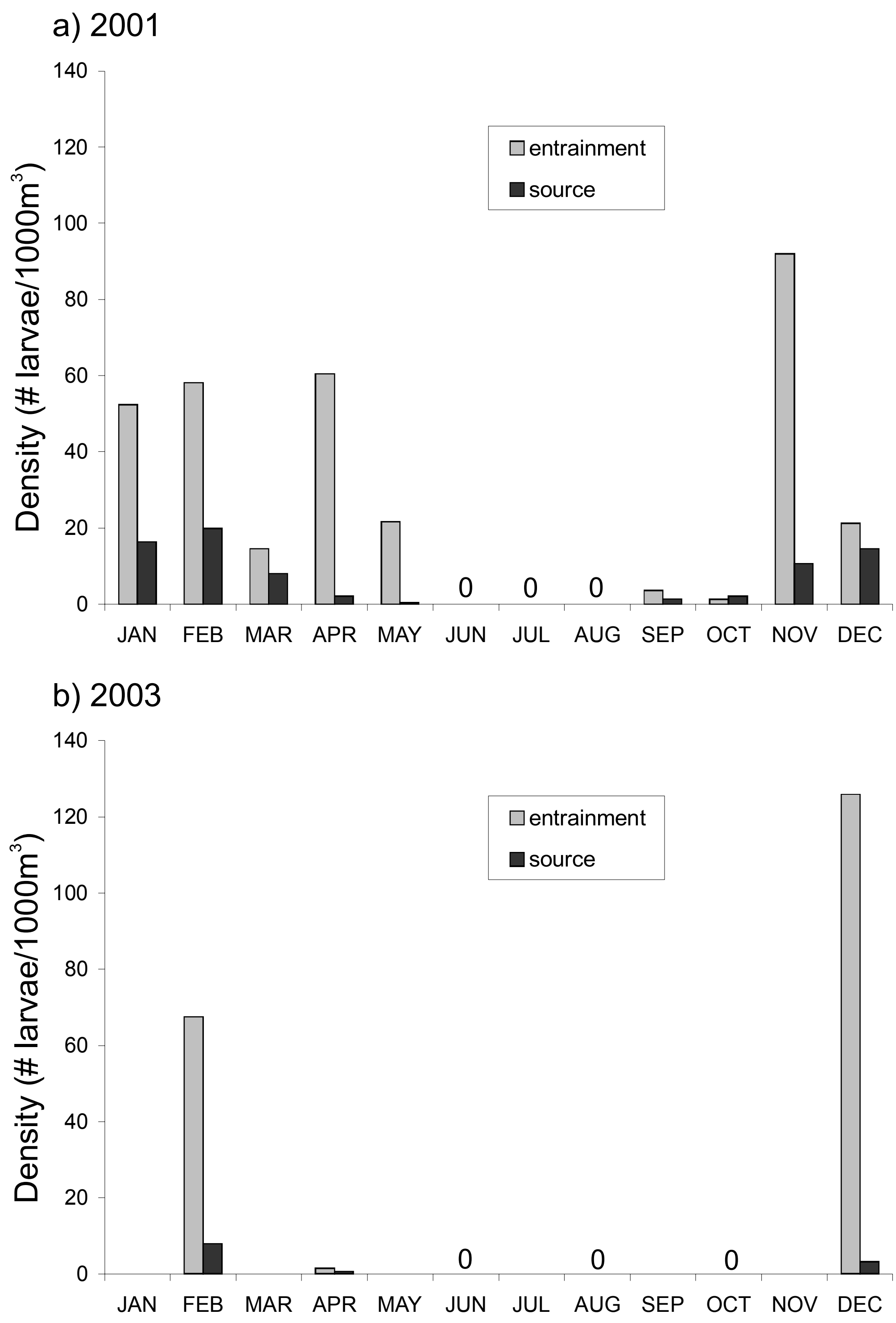


Figure 3.3-11. Comparison of mean density (#/1000 m^3) of longjaw mudsucker larvae between entrainment station (SB1) and source water stations (SB2−SB9): a) 2001 period (Note: January values from 2002); b) 2003 period (Note: December values from 2002).

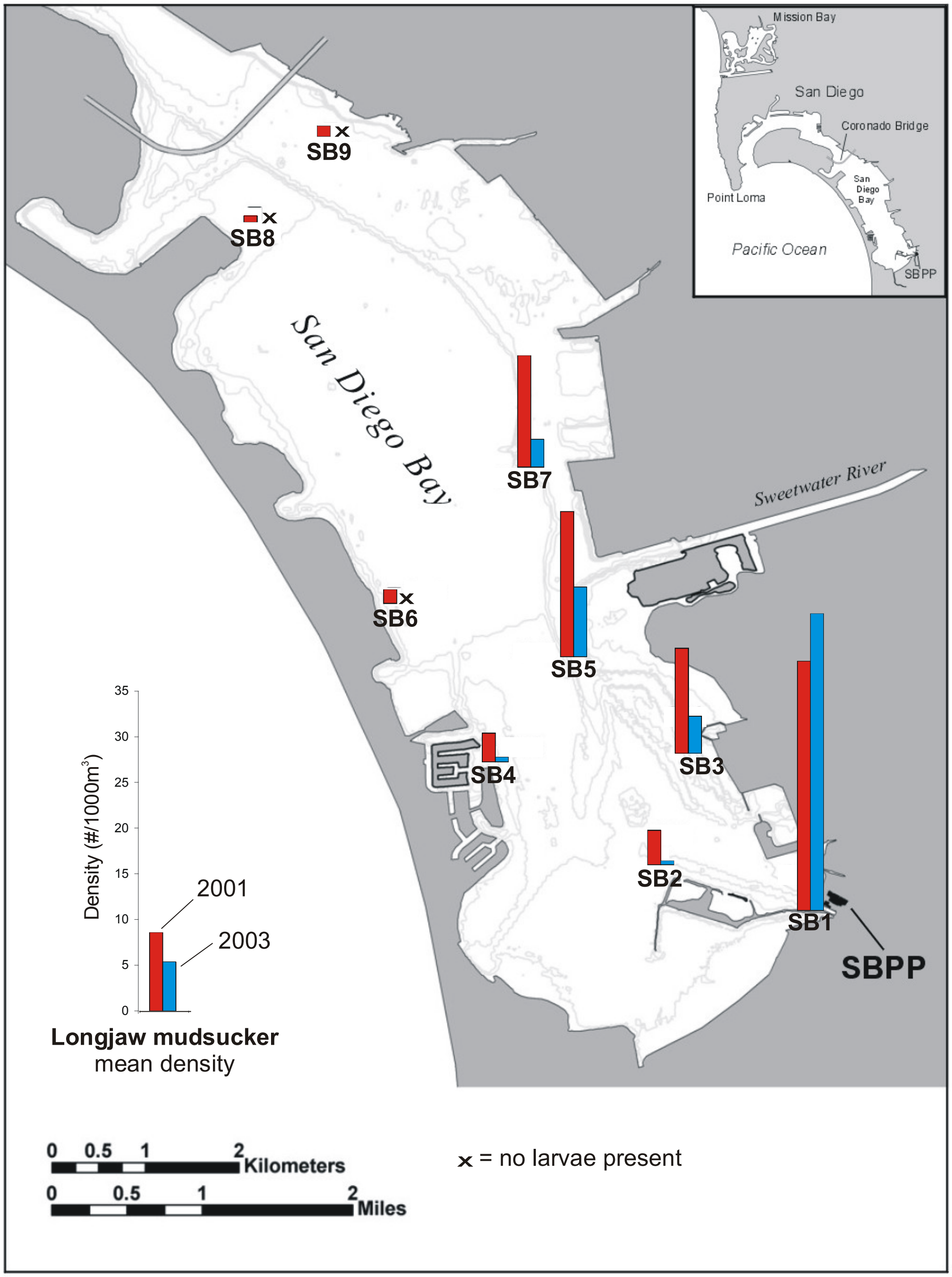


Figure 3.3-12. Annual mean density (#/1000 m^3) of longjaw mudsucker larvae at entrainment station (SB1) and source water stations (SB2-SB9) in south San Diego Bay during the 2001 and 2003 sampling periods. Mean density for all stations combined is shown with scale.

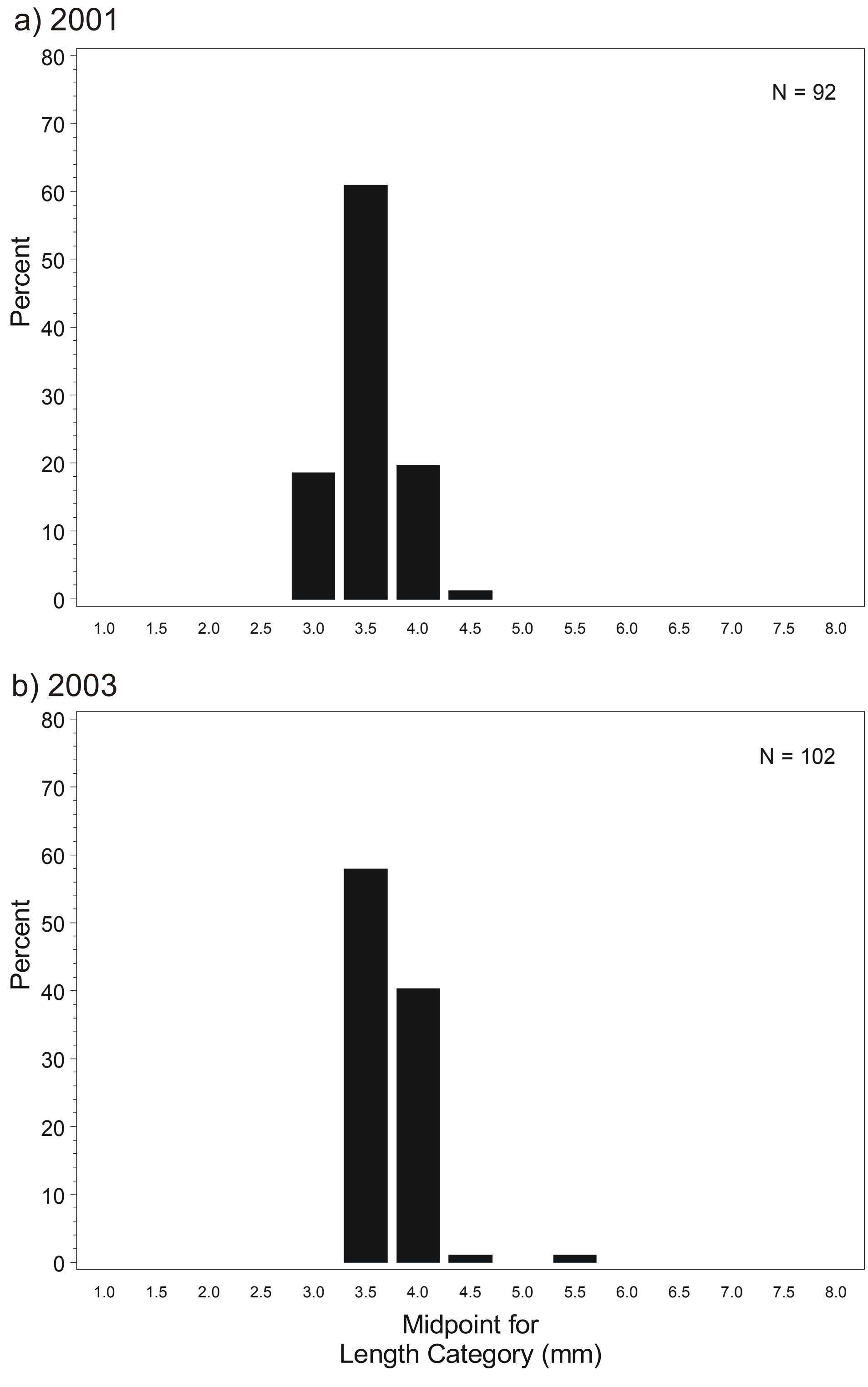


Figure 3.3-13. Length frequency distribution of longjaw mudsucker larvae collected at entrainment station SB1: a) 2001 period, b) 2003 period.

3.3.4 Anchovies (*Anchoa delicatissima and A. compressa*)

Distribution map for anchovies.

Range: Slough anchovy—Long Beach Harbor to Magdalena Bay, Baja California,
Deepbody anchovy—Morro Bay, California to Todos Santos Bay, Baja California.

Life History: Size: Slough anchovy to 94 mm (3.7 in), deepbody anchovy to 175 mm (7 in);
Size at maturity: Slough anchovy 50 mm (2 in), deepbody anchovy 70 mm (2.8 in);
Fecundity: females spawn multiple batches in summer months with annual reproductive output of approximately 7,000 eggs (slough anchovy) and 15,000–28,000 eggs (deepbody);
Life span: 3–6 yr.

Habitat: Primarily in estuaries and bays;

Fishery: No commercial fishery but occasionally used as bait for recreational fisheries.

3.3.4.1 Life History

The slough anchovy *Anchoa delicatissima* and the deepbody anchovy *Anchoa compressa* are two of the eleven species of Engraulidae (anchovies) larvae that have been identified in the California Cooperative Oceanic Fisheries Investigations (CalCOFI) study area (Moser et al. 1996). The CalCOFI study area covers more than one million square kilometers between the Oregon-California border and the tip of Baja California and extends from 3 to 400 nautical miles offshore. The two species of *Anchoa* fluctuate in relative abundance from year to year in southern California estuaries, but the causes of these changes are not known (CMI 2003, Emmett et al. 1991). The recreational take of *Anchoa* species for bait is not accurately known, but the RecFIN database (PSMFC 2003) reports that approximately 9,000 deepbody anchovy were landed between 1995 and 2002.

Both species of *Anchoa* typically occur in shallow bay and estuarine environments (Horn and Allen 1976). The slough anchovy is found from Long Beach Harbor to Magdalena

Bay, Baja California (Miller and Lea 1972). Maximum age is approximately 3 years (Heath 1980), with a maximum length of 94 mm (3.7 in) (Miller and Lea 1972). Females tend to grow larger than males. The deepbody anchovy is found from Morro Bay to Todos Santos Bay, Baja California (Miller and Lea 1972). It utilizes estuaries during all life stages, moving from the lower portions to the upper reaches during spawning season (CMI 2003, Emmett et al. 1991). Deepbody anchovy reach a maximum length of approximately 175 mm (7 in) and can live up to 6 years (Love 1996).

In a five-year study of fishes in San Diego Bay, Allen (1999) reported that slough anchovy comprised almost 20 percent of all individuals sampled and about 7 percent of the total biomass. Slough anchovy varied greatly in abundance over each year of the study (July 1994 through April 1999) but were found in virtually every month of the study in the southern regions of the bay (69 percent frequency). Peak abundance occurred in July of most years due to heavy recruitment of young-of-the-year (YOY). Deepbody anchovy were present in nearly 25 percent of all samples but accounted for less than 0.1 percent of total abundance and biomass.

Slough anchovy were the dominant anchovy species in the SBPP discharge channel and one of the most numerous finfishes in south San Diego Bay, accounting for 91.4 percent of all individuals captured over a three-year sampling period (Merkel & Associates, Inc. 2000a). They were found in high numbers in summer months and most individuals were small post-larvae or juveniles in the 30–40 mm SL range. Deepbody anchovy comprised 1.4 percent of individuals collected, and was most abundant in winter and spring surveys.

Both slough and deepbody anchovies mature at approximately 1 yr of age at sizes of 50 mm (2 in) and 70 mm (2.8 in), respectively (Heath 1980). They are broadcast spawners with external fertilization. Spawning occurs from May to September in Newport Bay (White 1977), with most of the spawning in south San Diego Bay occurring from April through June (McGowen 1981). Spawning takes place mainly at night (Heath 1980, Edmands 1983). This species appears to spawn primarily in the lower reaches of bays and estuaries, whereas the deepbody anchovy utilizes the upper reaches of bays for spawning (Edmands 1983). McGowen (1981) found a much higher proportion of deepbody eggs in back bay samples (58 percent) than slough anchovy eggs (2 percent). Bay anchovy *Anchoa mitchilli*, a congener, also spawn at night and about every 1–4 days in estuaries along the east coast (Farooqi et al. 2003) from May to September (Rilling and Houde 1999a). This species, however, matures at two to three months, with batch fecundity of 429–1,186 (Farooqi et al. 2003). Jung (2002) estimated female bay anchovy spawn 42 times each season producing 1,500–1,800 eggs per batch.

Mean annual fecundity of slough anchovy is approximately 7,000 eggs per female or 1,418 eggs/g female weight (Heath 1980). Deepbody anchovy annual fecundity ranges from 15,000–28,000 eggs per female, which equates to an average of 1,268 eggs/g female weight (Heath 1980). No information on spawning frequency for these species was found. Instead, an annual fecundity value of 45,110 eggs for bay anchovy *Anchoa mitchilli* reported by Luo and Musick (1991, in Jung 2002) was used in demographic modeling of entrainment effects at SBPP because larval survival from this species was also used in the modeling. Bay anchovy populate bays and estuaries and spawn at night (Rilling and Houde 1999a). Data from bay anchovy indicate high egg mortality with a daily survival rate of 0.50 (Houde 1987). Larvae of slough and deepbody anchovies are approximately 2.0–2.5 mm long at hatching (Moser et al. 1996, Farooqi et al. 2003). Bay anchovy grow at a rate of approximately 0.6–0.7 mm/d (Rilling and Houde 1999b) to about 20–25 mm before taking on juvenile characteristics within approximately 30 days (Moser et al. 1996, Farooqi et al. 2003). Data from bay anchovy were used to estimate daily survival at 0.87 for the larval stages through transformation (Rilling and Houde 1999b).

3.3.4.2 Sampling Results

Anchovy larvae were the second most abundant taxon collected in both the entrainment and source water samples during both sampling periods (**Tables 3.3-1**, **-2**, **-3**, and -**4**). Total annual entrainment estimates for the 2001 and 2003 sampling periods were 5.15×10^8 and 1.07×10^8 larvae, respectively (**Tables 3.3-1** and **3.3-2**). Entrainment estimates for each survey are presented in **Appendix D**. Anchovy larvae were mainly found from May through August with highest abundances during the June 2001 survey (**Figure 3.3-14a**). Mean densities were lower at the entrainment station than the combined source water stations in all surveys except June 2001. Although *Anchoa* spp. are reported to spawn multiple times during a year, these results from south San Diego Bay indicate a relatively short seasonal spawning period.

Highest concentrations of anchovy larvae were generally found in the central and western portions of the bay (**Figure 3.3-15**). The lowest concentrations occurred at Station SB9, a deep water channel station characterized by cooler water temperatures and swifter currents compared to the other stations.

The length frequency distribution for a representative sample of anchovy larvae showed that the majority of the larvae were recently hatched based on the reported hatch size of 2.0–2.5 mm (Moser et al. 1996, Farooqi et al. 2003). The mean length was 3.0 mm during 2001 and 4.4 mm during 2003 (**Figures 3.3-16 a, b**). The results, especially for the 2001 sampling period, show that a large number of the larvae were smaller than the reported hatch length. This may be due to shrinkage of the samples during preservation (Theilacker 1980) or geographic variation that results in hatch lengths for south San Diego Bay that are different from reported data. There was a significant difference between sampling periods in the mean size ($p < 0.01$) and distribution (KS test $p < 0.01$). The difference in the size distributions between years may have resulted from the decreased sampling frequency during the 2003 period. The bi-monthly frequency would have a greater likelihood of missing the period immediately after spawning when smaller larvae would be more abundant. The data from the 2003 sampling period do not appear to be representative of the size range of larvae that may be entrained based on the 2001 sampling period, therefore only the data from the 2001 sampling period were used in calculating the lengths of various life stages used in the impact assessment models.

3.3.4.3 Circulating Water System Assessment

The following sections present the results for demographic and empirical transport modeling of circulating water system effects. There was very little species-specific life history information available for slough or deepbody anchovies. Although larval growth was estimated from information on slough anchovy (Emmett et al. 1991) other life history information from an Atlantic coast species, bay anchovy *Anchoa mitchelli*, had to be used to parameterize the *FH* demographic model.

Fecundity Hindcasting *(FH)*

The two annual entrainment estimates for anchovy were used to estimate the number of breeding females needed to produce the number of larvae entrained (**Table 3.3-15**). Egg survival over the estimated 3-day planktonic period (Emmett et al. 1991) was estimated as 0.13 from mortality data on bay anchovy eggs (Houde 1987). Larval survival was estimated using daily mortality estimates for bay anchovy from Rilling and Houde (1999). Information in Emmett et al. (1991) indicates that slough anchovy larvae mature at 20–25 mm in 30 days. This information and an estimated hatch length of 2.3 mm were used to estimate a larval growth rate of 0.67 mm/d. The mean length (3.0 mm) and the length of the first percentile (1.3 mm) were used with this growth rate to estimate a mean age at entrainment of 2.5 d. Survival to the average age at entrainment was then estimated as $0.79^{2.5} = 0.56$. The estimate for total annual fecundity of 45,110 was based on data from bay anchovy (Luo and Musick 1991 in Jung 2002). Average ages of maturity and longevity of 1.0 and 3.0 years, respectively, from Emmett et al. (1991) for slough anchovy were used in the model.

The estimated numbers of adult females whose lifetime reproductive output was entrained through the SBPP circulating water system for the two periods were 106,876 and 22,249 for the 2001 and 2003 sampling periods, respectively (**Table 3.3-15**).

Table 3.3-15. Results of *FH* modeling for anchovy complex larvae entrained during the a) 2001 period and b) 2003 period. The upper and lower estimates are based on a 90% confidence interval.

Parameter	Estimate	Std. Error	*FH* Lower Estimate	*FH* Upper Estimate	*FH* Range
a) 2001 Period					
FH Estimate	**106,876**	185,216	6,177	1,849,086	1,842,909
Total Entrainment	514,808,619	5,071,239	96,821	116,931	20,110
b) 2003 Period					
FH Estimate	**22,249**	38,597	1,282	386,078	384,796
Total Entrainment	107,170,502	1,294,042	18,677	25,821	7,144

Empirical Transport Model (*ETM*)

The larval duration used to calculate the *ETM* estimates for the anchovy complex for both the 2001 and 2003 sampling periods was based on the larval lengths from the 2001 sampling period. The difference between the lengths of the 1^{st} (1.3 mm) and 99^{th} (7.8 mm) percentiles was used with a growth rate of 0.67 m/d to estimate that the larvae were vulnerable to entrainment for a period of 9.6 days.

ETM estimates of P_m for the anchovy complex were similar for the two sampling periods even though the *PE* estimates showed considerable variation among surveys (**Table 3.3-16**). *PE* estimates ranged from 0.00 to 0.28 for the two sampling periods. Even though anchovy abundances were highly variable in entrainment samples throughout the year (**Figure 3.3-18** and **-19**), the *PE* estimates for the surveys with the largest fractions of the source water population were close in value to the ratio of the circulating water and source water volumes (0.015). Results for both sampling periods show that anchovy larvae are most abundant in the source water during the summer surveys with the largest fractions of the source water population occurring during the July survey (f_i = 0.49) surveys during the 2001 sampling period, and during the August survey (f_i = 0.70) during the 2003 sampling period. The highest *PE* estimate occurred during the March 2001 survey when abundances for both the entrainment and source water stations were low.

Table 3.3-16. *ETM* parameters for anchovy larvae. *ETM* calculations based on South Bay source water volume of = 149,612,092 m^3, and daily circulating water volume = 2,275,244 m^3.

Survey Date	PE_i Estimate	PE_i Estimate Std. Error	f_i	f_i Std. Error
a) 2001 Period				
28-Feb-01	—	—	0.0001	0.0001
29-Mar-01	0.2784	0.1151	0.0003	0.0001
17-Apr-01	—	—	0.0001	0.0001
16-May-01	0.0146	0.0036	0.1070	0.0161
14-Jun-01	0.0298	0.0051	0.3187	0.0460
26-Jul-01	0.0012	0.0003	0.4949	0.0538
23-Aug-01	0.0058	0.0016	0.0743	0.0153
25-Sep-01	0.0029	0.0024	0.0036	0.0012
23-Oct-01	—	—	0.0007	0.0005
27-Nov-01	—	—	—	—
20-Dec-01	—	—	0.0002	0.0001
17-Jan-02	—	—	0.0001	0.0001
	P_m = 0.1048 *S.E.* = 0.3132			
b) 2003 Period				
12-Dec-02	0.0170	0.0196	0.0050	0.0040
6-Feb-03	0.0154	0.0173	0.0008	0.0004
9-Apr-03	0.0001	0.0001	0.1017	0.0168
5-Jun-03	0.0052	0.0024	0.1862	0.0442
15-Aug-03	0.0105	0.0016	0.7000	0.0445
15-Oct-03	0.0144	0.0141	0.0064	0.0033
	P_m = 0.0787 *S.E.* = 0.2814			

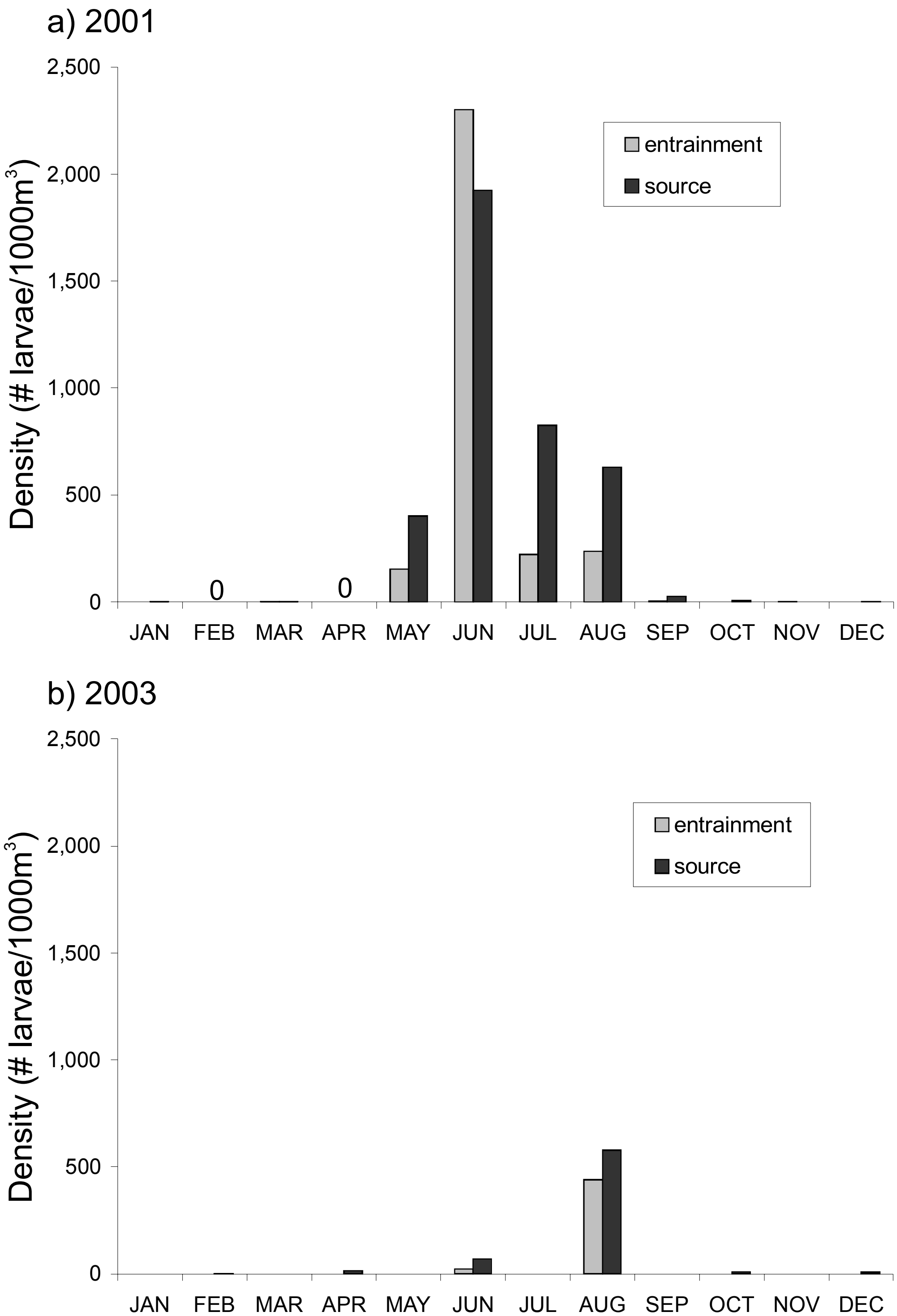


Figure 3.3-14. Comparison of mean density (#/1000 m^3) of anchovy larvae between entrainment station (SB1) and source water stations (SB2-SB9): a) 2001 period (Note: January values from 2002); b) 2003 period (Note: December values from 2002).

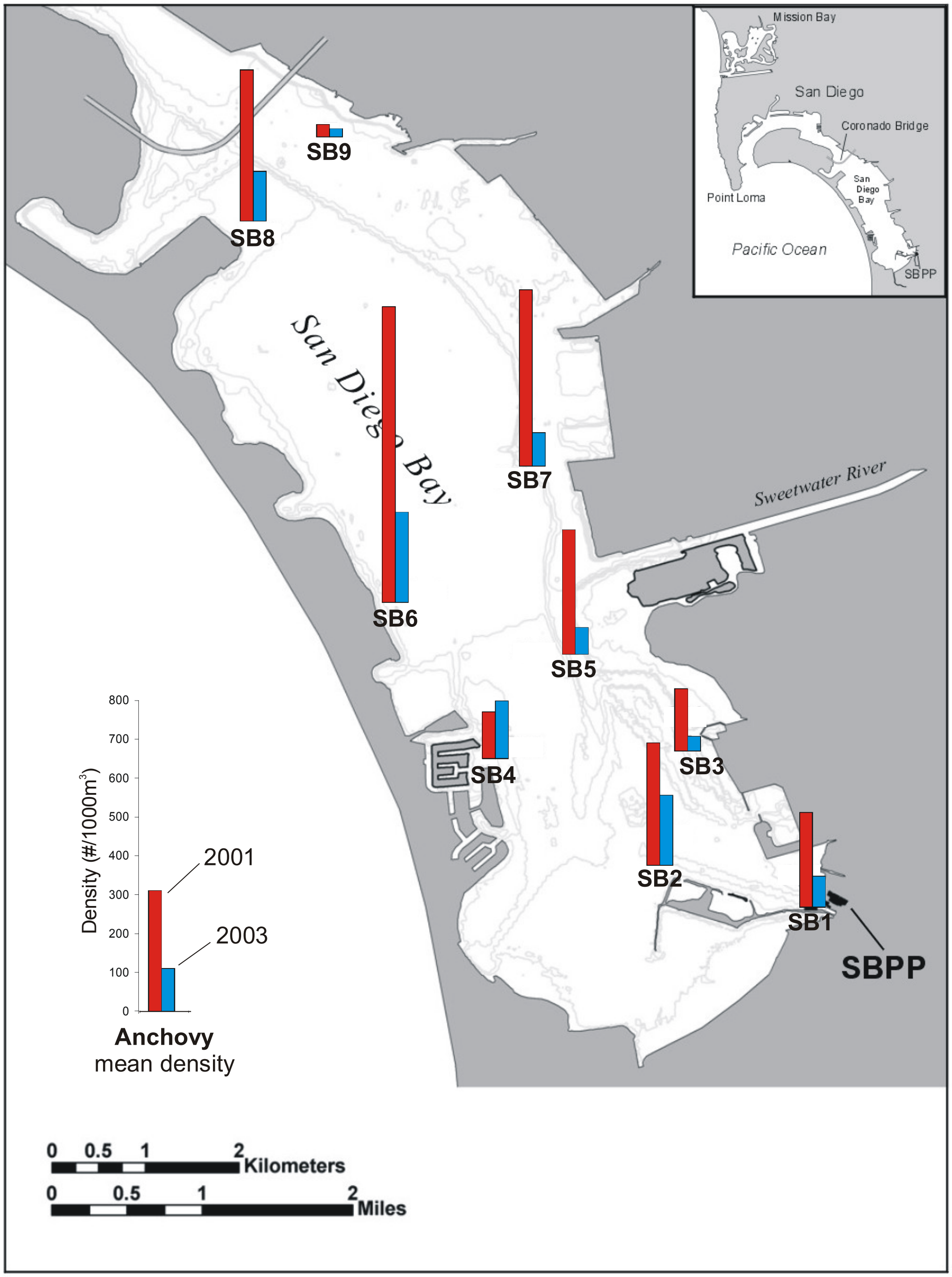


Figure 3.3-15. Annual mean density (#/1000 m^3) of anchovy larvae at entrainment station (SB1) and source water stations (SB2-SB9) in south San Diego Bay during the 2001 and 2003 sampling periods. Mean density for all stations combined is shown with scale.

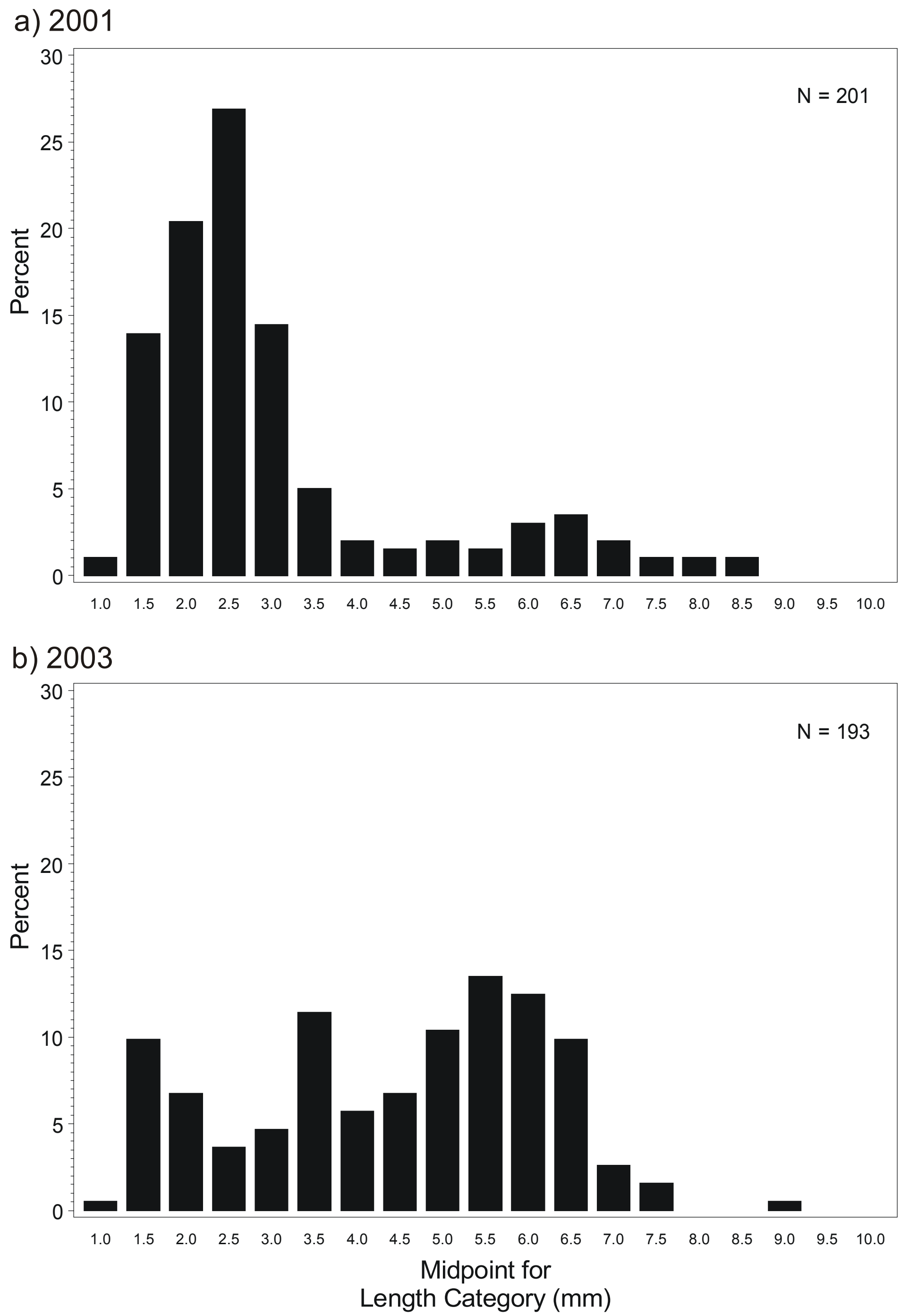
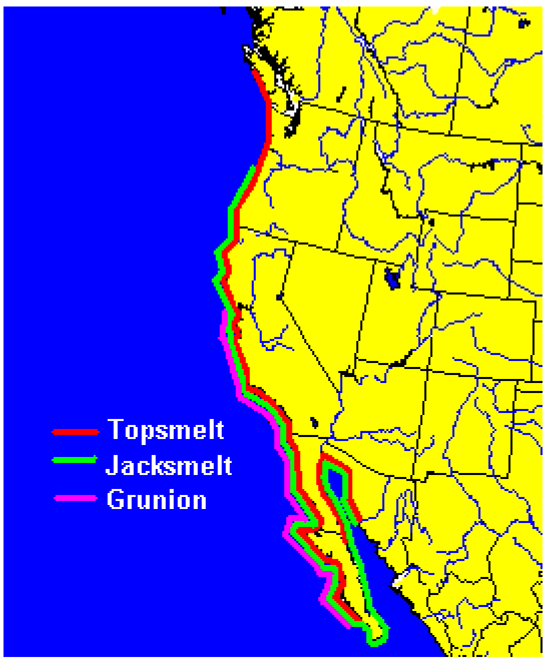


Figure 3.3-16. Length frequency distribution of anchovy larvae collected at entrainment station SB1: a) 2001 period, b) 2003 period.

3.3.5 Silversides (Atherinops affinis, Atherinopsis californiensis, and Leuresthes tenuis)



Distribution map for Atherinopsidae complex.

Range: Topsmelt—Vancouver Island, British Columbia, to southern Baja California and the upper Gulf of California;
Jacksmelt—Yaquina Bay, Oregon through Gulf of California;
Grunion—San Francisco to southern Baja California.

Life History: Size: topsmelt to 37 cm (14.5 in), jacksmelt to 44 cm (17 in), grunion to 19 cm (7.5 in);
Age at maturity: all species 2–3 yr;
Life span: topsmelt 8 yr, jacksmelt 9–10 yr, grunion 4 yr;
Annual fecundity: topsmelt 1,000 eggs, jacksmelt >2,000 eggs, grunion 1,000–3,000 eggs.

Habitat: Bays, estuaries, nearshore surface waters to depths of 29 m (95 ft).

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

3.3.5.1 Life History

Three species of silversides (family Atherinopsidae) occur in California ocean waters: topsmelt *Atherinops affinis*, jacksmelt *Atherinopsis californiensis*, and the California grunion *Leuresthes tenuis*. There is a limited fishery for silversides that are marketed fresh for human consumption or for bait (Leet et al. 2001). The commercial fishery for silversides has been conducted with a variety of gear. Historically, set-lines have been used in San Francisco Bay for jacksmelt, and during the 1920s beach nets, pulled ashore by horses, were used at Newport Beach (Leet et al. 2001). Commercial catches of jacksmelt have varied sharply over the past 80 years fluctuating from more than two million pounds in 1945 to 2,530 pounds in 1998 and 1999 (Leet et al. 2001). This is an incidental fishery and the large fluctuations in the catch records reflect demand, not actual abundances (Leet et al. 2001).

Topsmelt are found from Vancouver Island British Columbia, to the Gulf of California, (Miller and Lea 1972), with a disjunct distribution in the northern gulf (Robertson and

Allen 2002). These schooling fishes are very common in estuaries, kelp beds, and along sandy beaches. Topsmelt have a wide salinity tolerance and can survive in a range conditions from 0–90 ppt (Love 1996). Adults mature within 2–3 years to an approximate length of 10–15 cm (4–6 in), can reach a length of 37 cm (14.5 in), and have a life expectancy of up to eight years (Love 1996). Both topsmelt and jacksmelt are caught by sportfishers from piers and along shores.

In a five-year study of fishes in San Diego Bay, topsmelt ranked second in abundance and fifth in biomass, comprising about 23 percent of the individuals and 9 percent of the total weight (Allen 1999). Topsmelt were captured in all samples with peak abundances generally occurring in April due to heavy recruitment of young-of-the-year (YOY). Topsmelt occurred in a wide size range over the study and were represented by four age classes. Typically, YOY and juvenile topsmelt primarily occupied the intertidal zone while adult fish also occupied nearshore and midwater channel sub-habitats. Merkel and Associates, Inc. (2000a) also found topsmelt within the SBPP discharge channel. Topsmelt were collected nearly every month and represented about 1 percent of the total catch.

The spawning activity of topsmelt corresponds to changes in water temperature (Middaugh et al. 1990). In Newport Bay, topsmelt spawn from February to June peaking in May and June (Love 1996). Females deposit the eggs on marine plants and other floating objects where fertilization occurs (Love 1996). Fecundity is a function of female body size with individuals in the 110–120 mm range spawning approximately 200 eggs per season, and fish 160 mm or greater spawning 1,000 eggs per season (Fronk 1969). Topsmelt eggs maintained in the laboratory hatched 10–14 d after fertilization (Middaugh et al. 1990). Moser et al. (1996) reported that topsmelt hatch at lengths of 4.3–5.4 mm and transformation to the juvenile form occurs at 14–21 mm. Middaugh et al. (1990) reported a hatch length of 5.4 mm and growth to an average length of 9.7 mm after 8 days. These values were used to estimate a larval growth rate of 0.53 mm/d.

Jacksmelt is a pelagic species found in estuaries and coastal marine environments from Yaquina Bay, Oregon to the Gulf of California (Eschmeyer et al. 1983, Robertson and Allen 2002). Jacksmelt is the largest member of the three species of the silverside 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 18–20 cm (7.0–7.8 in) SL, and may live to a maximum age of nine or ten years (Clark 1929).

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 1996). Jacksmelt larvae hatch at an average length of 8.3 mm and reach a length of 11 mm after 8 d (Middaugh et al. 1990). These laboratory data were used to estimate a larval growth rate of 0.34 mm/d. The greatest concentrations of Atherinopsid larvae in south San Diego Bay were found from April through June (McGowen 1981).

California grunion are found from San Francisco to Magdalena Bay, Baja California (Miller and Lea 1972) but are most abundant from Point Conception southward (Love 1996). These are pelagic, schooling fish, usually seen from just behind the surf line to depths of about 18 m (60 ft). Grunion reach 19 cm (7.5 in) in length, with a life span of up to four years. They mature at one year old at a length of approximately 12–13 cm (5 in).

The commercial use of grunion is limited as this species forms a minor portion of the commercial "smelt" catch (Leet et al. 2001). Grunion are taken incidentally in bait nets and other round haul nets, and limited quantities are used as live bait, though no commercial landings have been reported (Leet et al. 2001). In the 1920s, the recreational fishery was showing signs of depletion, and a regulation was passed in 1927 establishing a closed season of three months, April through June. The fishery improved, and in 1947, the closure was shortened to April through May. Grunion may be taken by sport fishermen, using their hands only, and no holes may be dug in the beach to entrap them (Leet et al. 2001)

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 (peaking late March to early June) (Love 1996). The female swims onto the beach and digs into the wet sand, burying herself up to her pectoral fins or above. The male or males curve around her with vents touching her body, and when the female lays her eggs beneath the sand, males emit sperm, which flows down her body and fertilizes the eggs (Love 1996). Females spawn four to eight times per season at about 15-day intervals, producing 1,000–3,000 eggs. Eggs hatch at temperatures between 13.9–28.3°C (57–83°F). The eggs remain in the sand until they are liberated by the next tide high enough to reach them (approximately 10 days). Larvae hatch at approximately 6.5–7.0 mm transforming into juveniles at 15–20 mm (Moser et al. 1996).

3.3.5.2 Sampling Results

Silverside larvae were the fifth most abundant taxa of fish larvae collected in the entrainment samples during both the 2001 and 2003 sampling periods with a total annual entrainment estimates of 1.45×10^7 and 9.79×10^6 larvae, respectively (**Tables 3.3-1** and **3.3-2**). They were the fourth most abundant larvae collected in the source water samples during both the 2001 and 2003 sampling periods (**Tables 3.3-3** and **-4**). Entrainment estimates for each survey are in **Appendix D**. Silverside larvae were collected in highest abundance in both entrainment and source water samples generally from December through April during the 2001 sampling period (**Figures 3.3-17a**), but had the greatest density in the 2003 period during June (**Figures 3.3-17b**). Silverside larvae were most abundant at shallow stations in the western portion of the bay and least abundant at the channel stations on the eastern side of the bay (**Figure 3.3-18**).

The length frequency distribution of silverside larvae showed that a majority of the larvae were recently hatched based on the average of the reported hatch lengths for topsmelt and jacksmelt (average = 6.9 mm) from Middaugh et al. (1990) (**Figure 3.3-19a, b**). The mean lengths from the 2001 and 2003 sampling periods were 7.4 and 7.1 mm, respectively , with a mean length of 7.3 mm for the combined data.

3.3.5.3 Circulating Water System Impact Assessment

The following sections present the results for demographic and empirical transport modeling of circulating water system effects. Although there was information on the early life history for California grunion, there was very little species-specific information available for the other two species, topsmelt and jacksmelt, that were collected in greater abundances during the study. Therefore, circulating water system effects were estimated using only the *ETM* and neither of the demographic models. Larval growth for topsmelt and jacksmelt was estimated from laboratory studies (Middaugh et al. 1990).

Empirical Transport Model (*ETM*)

The larval duration used to calculate the *ETM* estimates for silversides for both the 2001 and 2003 sampling periods was based on the lengths of entrained larvae combined for the two sampling periods. The difference between the lengths of the 1^{st} (4.4 mm) and 99^{th} (11.9 mm) percentiles was used with a growth rate of 0.44 mm/d to estimate that silverside larvae were vulnerable to entrainment for a period of 17.3 d.

The *ETM* estimates of P_m for silversides for the two sampling periods were very similar in value (**Table 3.3-17**). *PE* estimates ranged from 0.00 to 0.08 for the two sampling periods. The *PE* estimates vary considerably from the ratio of the circulating water system to source water volumes (0.015). This is consistent with the entrainment and source water abundances that showed considerable variation throughout both sampling periods and among stations. The largest *PE* estimates did not occur when the largest fractions of the source water populations were present resulting in P_m estimates that were small relative to the long period that the larvae are potentially vulnerable to entrainment. The results are also consistent with results showing that mean larval silverside concentrations were generally higher at the source water stations.

Table 3.3-17. *ETM* parameters for silversides. *ETM* calculations based on South Bay source water volume of = 149,612,092 m^3, and daily circulating water volume = 2,275,244 m^3.

Survey Date	PE_i Estimate	PE_i Estimate Std. Error	f_i	f_i Std. Error
a) **2001 Period**				
28-Feb-01	0.0085	0.0034	0.6405	0.0841
29-Mar-01	0.0004	0.0004	0.1625	0.0596
17-Apr-01	0.0116	0.0072	0.0683	0.0349
16-May-01	0.0603	0.0328	0.0048	0.0022
14-Jun-01	0.0499	0.0706	0.0010	0.0010
26-Jul-01	—	—	—	—
23-Aug-01	—	—	—	—
25-Sep-01	0.0823	0.1165	0.0007	0.0007
23-Oct-01	—	—	0.0064	0.0033
27-Nov-01	0.0186	0.0158	0.0100	0.0050
20-Dec-01	0.0422	0.0163	0.0340	0.0098
17-Jan-02	0.0186	0.0072	0.0718	0.0268
	P_m = 0.1460 *S.E.* = 0.3734			
b) **2003 Period**				
12-Dec-02	0.0589	0.0200	0.1148	0.0628
6-Feb-03	0.0036	0.0029	0.2252	0.1242
9-Apr-03	0.0294	0.0227	0.1209	0.0625
5-Jun-03	0.0009	0.0012	0.5268	0.2209
15-Aug-03	—	—	—	—
15-Oct-03	0.0204	0.0238	0.0122	0.0093
	P_m = 0.1487 *S.E.* = 0.4121			

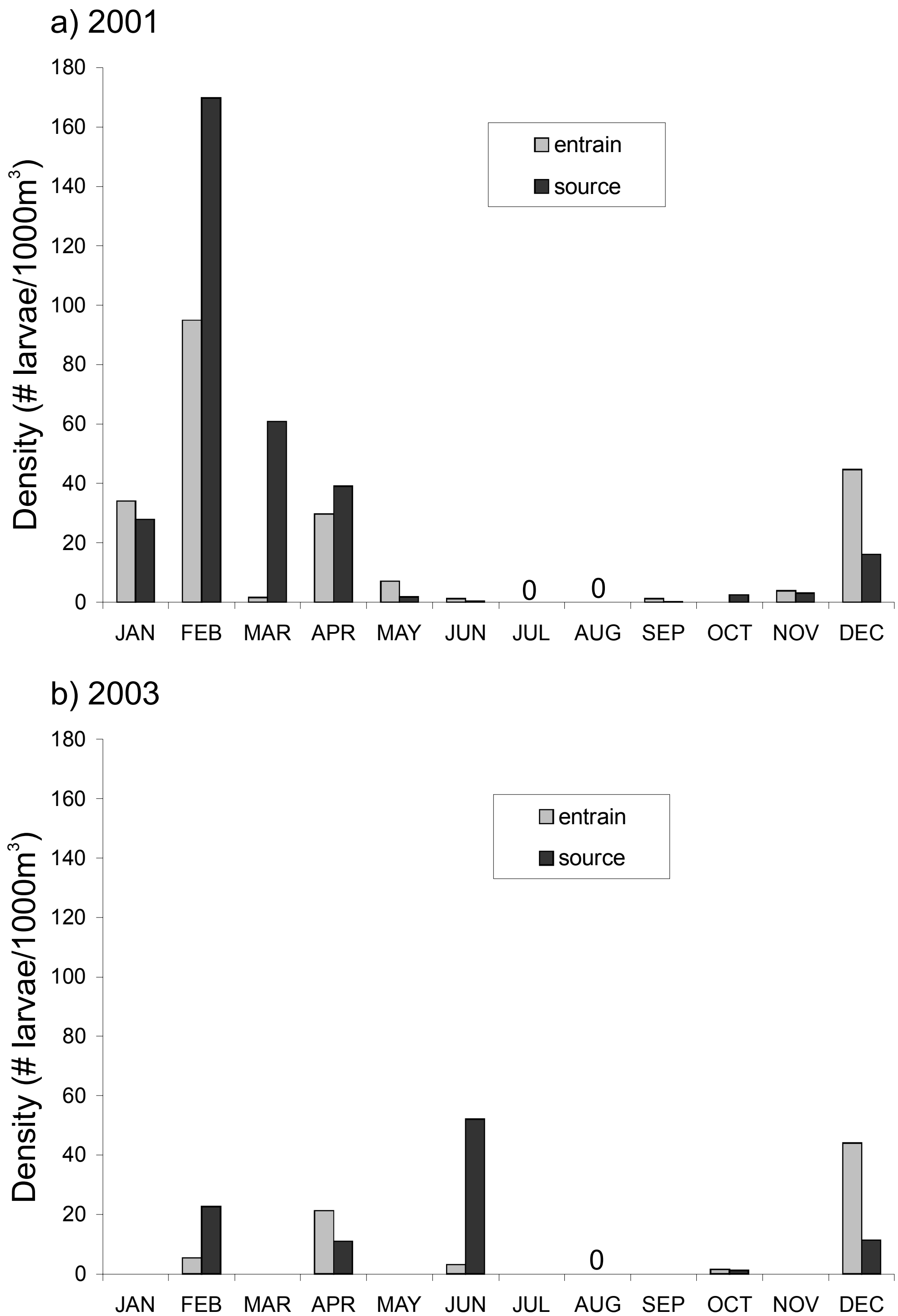


Figure 3.3-17. Comparison of mean density (#/1000 m^3) of silverside larvae between entrainment station (SB1) and source water stations (SB2-SB9): a) 2001 period (Note: January values from 2002); b) 2003 period (Note: December values from 2002).

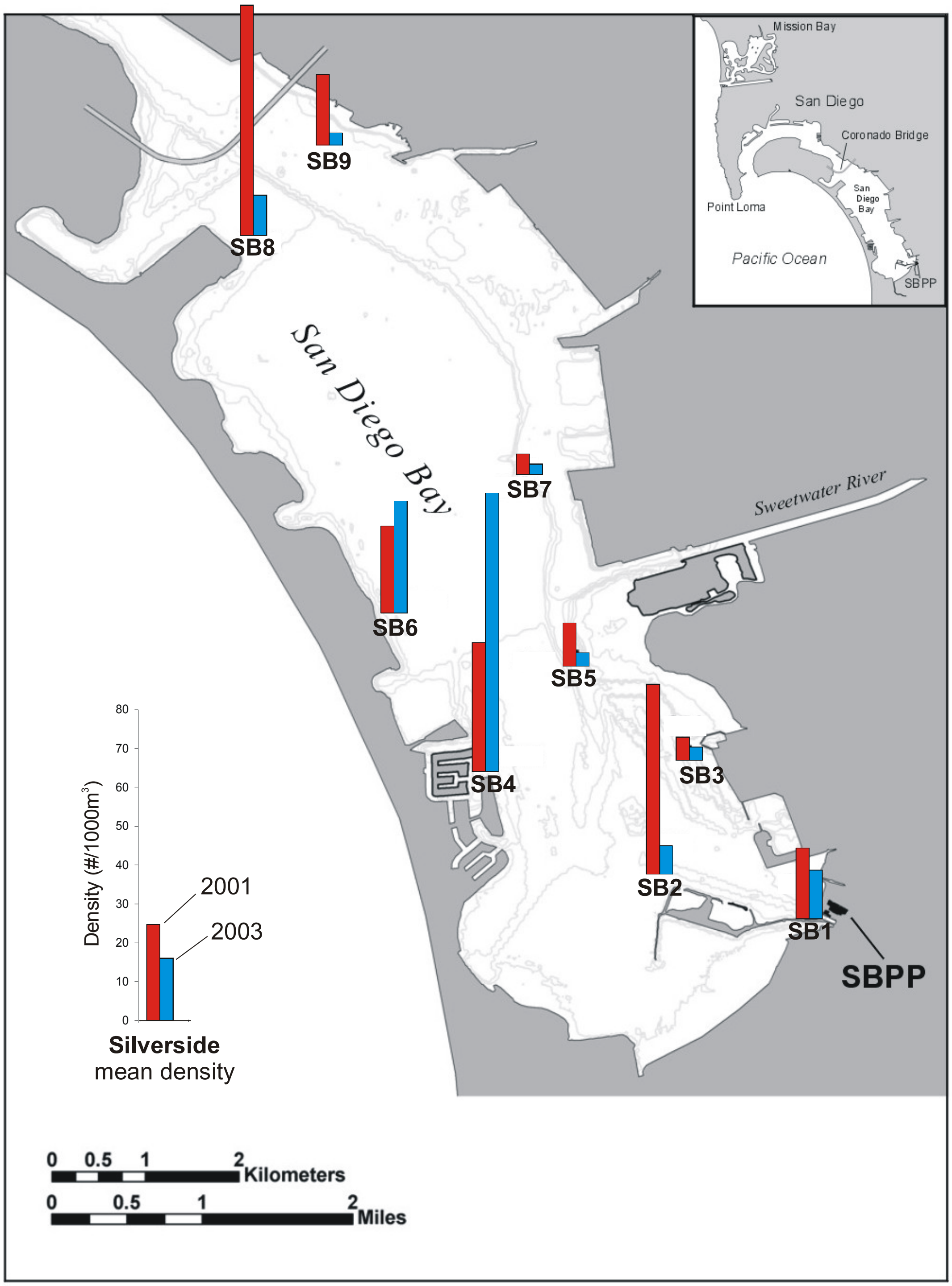


Figure 3.3-18. Annual mean density (#/1000 m^3) of silverside larvae at entrainment station (SB1) and source water stations (SB2-SB9) in south San Diego Bay during the 2001 and 2003 sampling periods. Mean density for all stations combined is shown with scale.

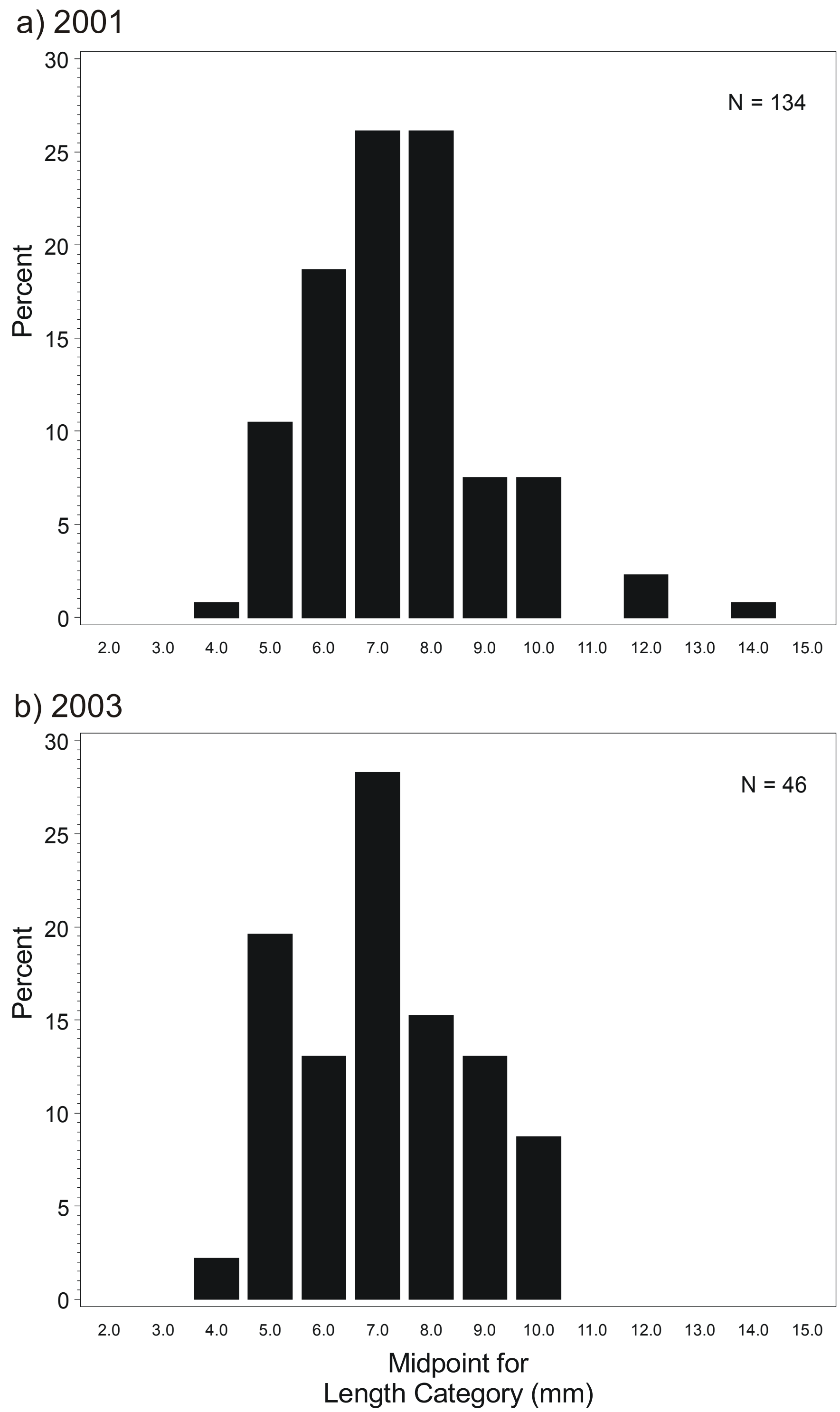
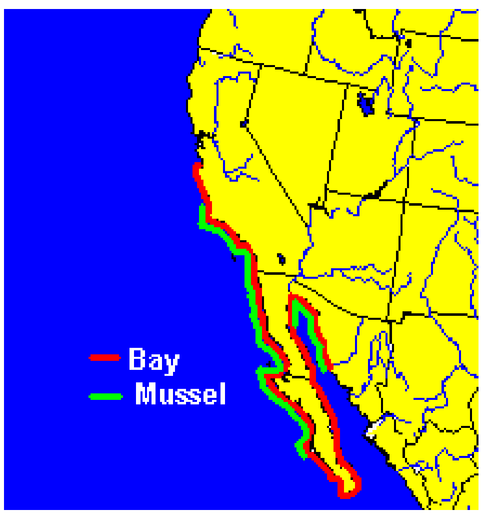


Figure 3.3-19. Length frequency distribution of silverside larvae collected at entrainment station SB1: a) 2001 period and b) 2003 period.

3.3.6 Combtooth Blennies (*Hypsoblennius* spp.)



Distribution map for combtooth blennies.

Range: Bay blenny—Monterey Bay to Gulf of California.
Mussel blenny—Morro Bay to Magdalena Bay, Baja California and the northern Gulf of California.

Life History: Size: bay blenny to 15 cm (5.9 in), mussel blenny to 13 cm (5.1 in);
Age at maturity: both species 0.5 yr;
Life span: bay blenny 6 yr., mussel blenny 5 yr.
Fecundity: bay blenny 300–3,000 eggs, mussel blenny 200–2,000 eggs;

Habitat: Bay blenny—soft bottom in bays and estuaries, associated with submerged aquatic vegetation and mussels on mooring buoys; to a depth of 9 m (30 ft),
Mussel blenny—empty worm tubes and barnacle tests on pilings, mussel beds, crevices in shallow rock reefs; to 12 m (40 ft);

Fishery: No commercial or recreational fishery.

3.3.6.1 Life History

Combtooth blennies are a prominent group among the subtropical and tropical fish fauna that inhabit inshore rocky habitats throughout much of the world. They are members of the family Blenniidae within the order Blennioidei. The family Blenniidae, the combtooth blennies, contains about 345 species in 53 genera (Nelson 1994, Moser 1996). They derive their common name from the arrangement of closely spaced teeth in their jaws.

Combtooth blennies are all relatively small fishes that typically grow to a total length of less than 200 mm (7.9 in) (Moser et al. 1996). Most have blunt heads that are topped with some arrangement of cirri (Moyle and Cech 1988, Moser 1996). Their bodies are generally elongate and without scales. Dorsal fins are often continuous and contain more soft rays than spines (Moyle and Cech 1988). Coloration in the group is quite variable, even among individuals of the same species (Stephens et al. 1970).

Blennies inhabit a variety of hard substrates in the intertidal and shallow subtidal zones of tropical and subtropical marine habitats throughout the world. They may occur to depths of 24 m (80 ft) but are more frequently found in water depths of less than 5 m (15 ft) (Love 1996). Combtooth blennies are common in rocky tidepools, reefs, breakwaters, and on pier pilings. They are also frequently observed on encrusted buoys and boat hulls. Combtooth blennies are omnivores and eat both algae and a variety of invertebrates, including limpets, urchins, and bryozoa (Stephens 1969, Love 1996).

Combtooth blennies are represented along the California coast by three members of the genus *Hypsoblennius*: bay blenny *H. gentilis*, rockpool blenny *H. gilberti*, and mussel blenny *H. jenkinsi*. These species co-occur throughout much of their range although they occupy different habitats. The bay blenny is found along both coasts of Baja California and up the California coast to as far north as Monterey Bay, (Miller and Lea 1972, Robertson and Allen 2002). The rockpool blenny occurs from Magdalena Bay, Baja California to Point Conception, California (Miller and Lea 1972, Stephens et al. 1970). The range of the mussel blenny extends from Morro Bay to Magdalena Bay, Baja California and in the northern Gulf of California (Tenera 2001, Robertson and Allen 2002). In San Diego Bay the only *Hypsoblennius* species recorded in a five year study of the bay's fishes was the bay blenny, although the sampling methods did not include inspections of piling habitats where mussel blennies are known to be common. Mussel blenny larvae have been positively identified from plankton samples collected in the present study. No rockpool blenny habitat occurs in the bay and no specimens have been collected in the various studies, so the *Hypsoblennius* spp. complex designation in the present study was used to describe only bay and mussel blenny species.

The two species have different habitat preferences. The mussel blenny is only found subtidally and inhabits mussel beds, the empty drill cavities of boring clams, barnacle tests, or in crevices among the vermiform snail tubes *Serpulorbis* spp. (Stephens 1969, Stephens et al. 1970). They generally remain within one meter of their chosen refuge (Stephens et al. 1970). The bay blenny is usually found subtidally but appear to have general habitat requirements and may inhabit a variety of intertidal and subtidal areas (Stephens et al. 1970). They are commonly found in mussel beds and on encrusted floats, buoys, docks, and even fouled boat hulls (Stephens 1969, Stephens et al. 1970). Bay blenny are often found in bays and are tolerant of nearly estuarine conditions (Stephens et al. 1970). They are among the first fish species to colonize new or disturbed marine habitats such as new breakwaters or mooring floats after recolonization by attached invertebrates (Stephens et al. 1970, Moyle and Cech 1988).

Bay blenny grow to a slightly larger size and live longer than mussel blenny, reaching a size of 15 cm (5.9 in) and living for 6–7 years (Stephens 1969, Stephens et al. 1970, Miller and Lea 1972). Mussel blennies grow to 13 cm (5.1 in) and have a life span 3–6 years (Stephens et al. 1970, Miller and Lea 1972). Male and female growth rates are similar. Female blennies mature quickly and reproduce within the first year, reaching peak reproductive potential in the third year (Stephens 1969). The spawning season typically begins in the spring and may extend into September (Stephens et al. 1970). Blennies are oviparous and lay demersal eggs that are attached to the nest substrate by adhesive pads or filaments (Moser et al. 1996). Males are responsible for tending the nest and developing eggs. Females spawn 3–4 times over a period of several weeks (Stephens et al. 1970). Males guard the nest aggressively and will often chase the female away, however, several females may occasionally spawn with a single male. The number of eggs a female produces varies proportionately with size (Stephens et al. 1970). The mussel blenny spawns approximately 500 eggs in the first reproductive year and up to 1,500 eggs by the third year (Stephens et al. 1970). Total lifetime fecundity may be up to 7,700 eggs (Stephens 1969).

Larvae are pelagic and average 2.7 mm (0.11 in) in length two days after hatching at a size of 2.3–2.6 mm (0.9–1.0 in) (Stephens et al. 1970, Moser 1996). The planktonic phase for *Hypsoblennius* spp. larvae may last for 3 months (Stephens et al. 1970, Love 1996). *Hypsoblennius* larvae are visual swimmers (Ninos 1984). Captured larvae released by divers have been observed to use surface water movement and near-surface currents to aid swimming. After release the swimming larvae orient to floating algae, bubbles on the surface, or the bottoms of boats or buoys. The overall swimming speed was measured at 17 cm/s (0.33 kt) for rockpool blenny and 14.8 cm/s (0.28 kt) for mussel blenny (Ninos 1984). Size at settlement ranges from 12–14 mm (0.5–0.6 in). After the first year mussel and bay blenny averaged 40 and 45 mm (1.6 and 1.8 in) total length, respectively (Stephens et al. 1970).

The three species of *Hypsoblennius* found in California waters are morphologically similar as early larvae (Moser 1996, Ninos 1984). For this reason most *Hypsoblennius* identified in SBPP plankton tows collections were identified as *Hypsoblennius* spp. Certain morphological features (e.g., preopercular spines) develop at larger sizes and allow taxonomists to identify some larvae to the species level. All larger, more developed *Hypsoblennius* collected in this study were identified as mussel blenny. The only individual collected in SBPP impingement samples was also a mussel blenny. Since the majority of the smaller *Hypsoblennius* specimens were most likely mussel blenny, life history information for this species was used to model entrainment impacts on this group.

Stephens (1969) estimated the survival for mussel blenny from egg to settling (50 d) at 0.31. This is a rough estimate based on assumptions of population size and fecundity. To check the estimate we recalculated mortality for half year intervals using the predicted age group abundance based on 1,284 fish in Stephens (1969) study. Blenny daily larval survival was estimated as 0.8875 using a computed adult daily survival from Stephens (1969) of 0.9983. Adult survival was computed by fitting an exponential mortality function to observed field abundances of ages 0.52, 1, 2, 3, 4, 5, and 6 yr fish. The larval survival was then estimated using N_0 equal to the estimated lifetime fecundity of 3,281. A larval growth rate of 0.198 mm/d for mussel blenny was determined by regression of the growth equation; $Y_t = 1.2845\ e^{3.92*(1-e^{\wedge}\ -0.0177t)}$ (Stevens and Moser 1982) determined for 300+ days growth to 60–65 mm SL.

3.3.6.2 Sampling Results

Combtooth blenny larvae were the third or fourth most abundant taxon collected in both the entrainment and source water samples during both sampling periods (**Tables 3.3-1**, **-2**, **-3**, and **-4**). Total annual entrainment estimates for the 2001 and 2003 sampling periods were 2.23 x 10^7 and 2.36 x 10^7 larvae, respectively (**Tables 3.3-1** and **3.3-2**). Entrainment estimates for each survey are presented in **Appendix D**. Combtooth blenny larvae were substantially more abundant at the source water stations than the entrainment station and occurred throughout the year during both sampling periods (**Figures 3.3-20a**, **20b**). Maximum densities were recorded at the entrainment station in winter, whereas source water concentrations generally peaked in summer and early-fall. In contrast to anchovies and silversides, larval blennies were most abundant at deeper channel stations on the eastern margin of the bay (**Figure 3.3-21**). Adult mussel blennies occur mainly in association with rock revetments, pilings and other hard substrates which were most abundant on the central and eastern sides of the bay, and their larval distribution seemed to reflect this association.

The length frequency distribution for a representative sample of combtooth blenny larvae showed that the majority of the larvae were recently hatched based on the reported hatch size of 2.5 mm (Moser et al. 1996). The mean lengths for the 2001 and 2003 sampling periods were 2.6 and 2.7 mm, respectively, with a mean length of 2.7 mm for the combined data(**Figures 3.3-22a, b**).

3.3.6.3 Circulating Water System Impact Assessment

The following sections present the results for demographic and empirical transport modeling of circulating water system effects. There was very little species-specific life history information available for combtooth blennies. Larval survival was estimated using data Stephens (1969) and (Stevens and Moser 1982) and there was enough other information on reproduction to parameterize the *FH* demographic model. Larval growth was estimated from information from (Stevens and Moser 1982).

Fecundity Hindcasting (*FH*)

The two annual entrainment estimates for combtooth blenny were used to estimate the number of breeding females needed to produce the number of larvae entrained (**Table 3.3-18**). No estimates of egg survival for combtooth blenny were available, but because egg masses are attached and guarded by the male (Stephens et al. 1970), egg survival is probably high and was assumed to be 100 percent. The mean length for larval combtooth blenny larvae in entrainment samples was 2.7 mm. A larval growth rate of 0.20 mm/d was derived from growth rates derived from data in Stevens and Moser (1982). The mean length and the length at the 1^{st} percentile (1.9 mm) were used with the growth rate to estimate that the mean age at entrainment was 3.8 d. A daily survival rate of 0.89 computed from Stephens (1969) was used to calculate survival to the average age at entrainment as $0.89^{3.8} = 0.63$. An average batch fecundity estimate of 550 eggs was based on data from Stephens (1969), and an estimate of 2.3 spawns per year based on information from Stevens and Moser (1982) where used to calculate an annual fecundity of 1,281 eggs. An average longevity for mussel blenny of 3–6 yr from Stephens (1969) and an age of maturation of 0.4 yr from Stevens and Moser (1982) were used in the model.

The estimated numbers of adult females whose lifetime reproductive output was entrained through the SBPP circulating water system for the two periods were very similar between sampling periods: 10,757 in 2001 and 11,374 in 2003 (**Table 3.3-18**).

Table 3.3-18. Results of *FH* modeling for combtooth blenny larvae entrained during the a) 2001 period and b) 2003 period. The upper and lower estimates are based on a 90% confidence interval of the mean.

Parameter	Mean	Std. Error	*FH* Lower Estimate	*FH* Upper Estimate	*FH* Range
a) 2001 Period					
FH Estimate	**10,757**	18,646	621	186,217	185,596
Total Entrainment	22,334,999	258,893	9,577	11,937	2,360
b) 2003 Period					
FH Estimate	**11,374**	19,737	655	197,518	196,863
Total Entrainment	23,615,354	328,998	9,398	13,349	3,951

Empirical Transport Model (*ETM*)

The larval duration used to calculate the *ETM* estimates for combtooth blenny for both the 2001 and 2003 sampling periods was based on the lengths of entrained larvae combined for the two sampling periods. The difference between the lengths of the 1^{st} (1.9 mm) and 99^{th} (5.1 mm) percentiles was used with a growth rate of 0.20 mm/d to estimate that combtooth blenny larvae were vulnerable to entrainment for a period of 16.1 d.

The *ETM* estimates of P_m for both sampling periods were very similar in value (**Table 3.3-19**). *PE* estimates ranged from 0.0001 to 0.0126 for the two sampling periods. The

PE estimates are all less than the ratio of the circulating water system to source water volumes (0.015). This is due to the greater abundances of combtooth blenny larvae at the source water stations relative to the entrainment station (**Tables 3.3-5** and **3.3-6**). This resulted in the low *PE* and P_m estimates.

Table 3.3-19. *ETM* parameters for combtooth blenny. *ETM* calculations based on South Bay source water volume of = 149,612,092 m^3, and daily circulating water volume = 2,275,244 m^3.

Survey Date	PE_i Estimate	PE_i Estimate Std. Error	f_i	f_i Std. Error
a) **2001 Period**				
28-Feb-01	0.0126	0.0034	0.0282	0.0036
29-Mar-01	0.0028	0.0005	0.0651	0.0085
17-Apr-01	0.0012	0.0002	0.0511	0.0052
16-May-01	0.0001	0.0001	0.1132	0.0110
14-Jun-01	0.0008	0.0002	0.1250	0.0118
26-Jul-01	0.0011	0.0004	0.1419	0.0262
23-Aug-01	0.0020	0.0003	0.1095	0.0103
25-Sep-01	0.0012	0.0002	0.1578	0.0161
23-Oct-01	0.0028	0.0006	0.0629	0.0066
27-Nov-01	0.0030	0.0006	0.0966	0.0084
20-Dec-01	0.0030	0.0006	0.0305	0.0043
17-Jan-02	0.0089	0.0028	0.0183	0.0025
	P_m = 0.0310 *S.E.* = 0.1774			
b) **2003 Period**				
12-Dec-02	0.0084	0.0018	0.1180	0.0147
6-Feb-03	0.0091	0.0018	0.0812	0.0107
9-Apr-03	0.0014	0.0005	0.2122	0.0197
5-Jun-03	0.0002	0.0000	0.1342	0.0208
15-Aug-03	0.0002	0.0001	0.3223	0.0307
15-Oct-03	0.0007	0.0003	0.1320	0.0146
	P_m = 0.0337 *S.E.* = 0.1849			

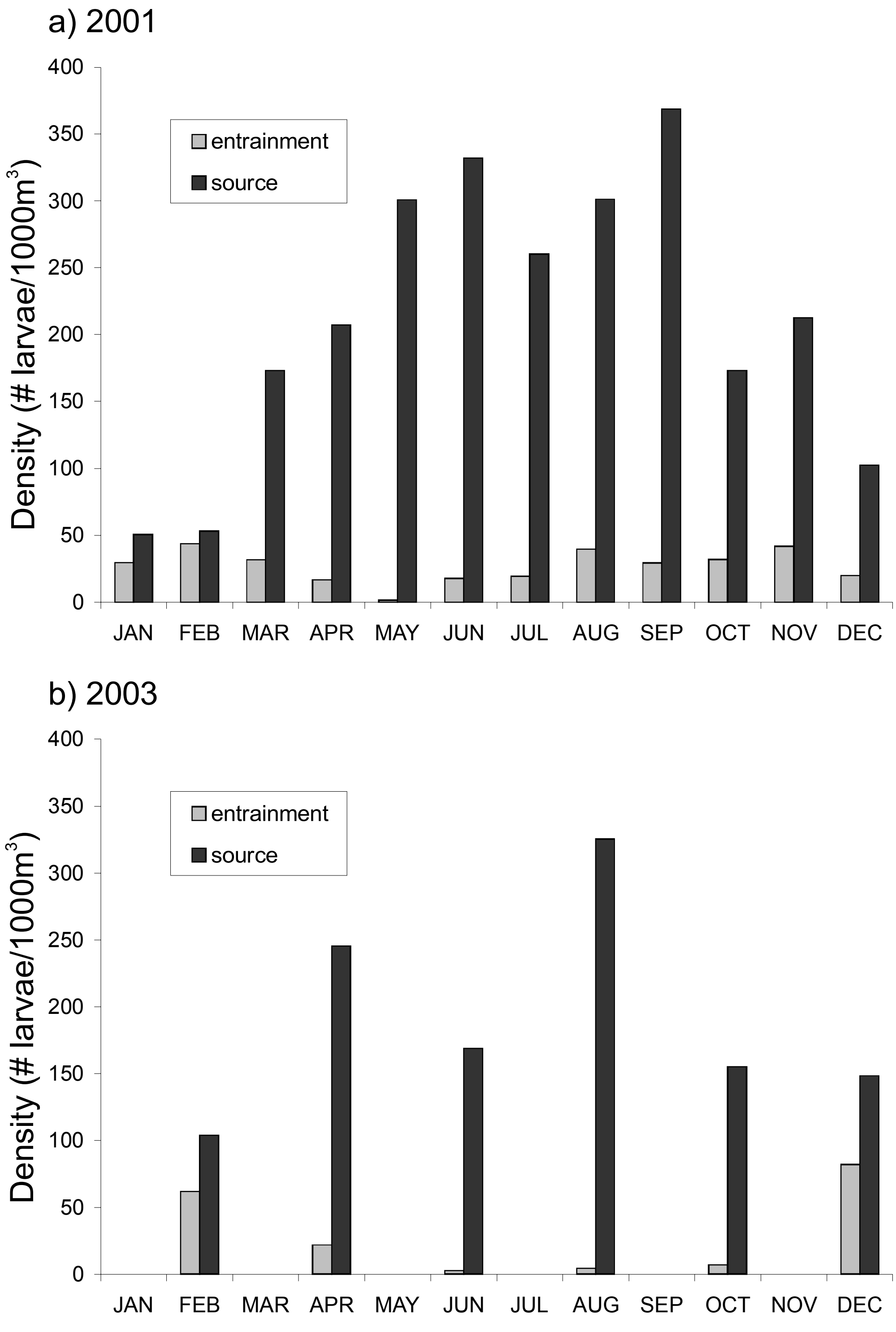


Figure 3.3-20. Comparison of mean density (#/1000 m^3) of combtooth blenny larvae between entrainment station (SB1) and source water stations (SB2-SB9): a) 2001 period (Note: January values from 2002); b) 2003 period (Note: December values from 2002).

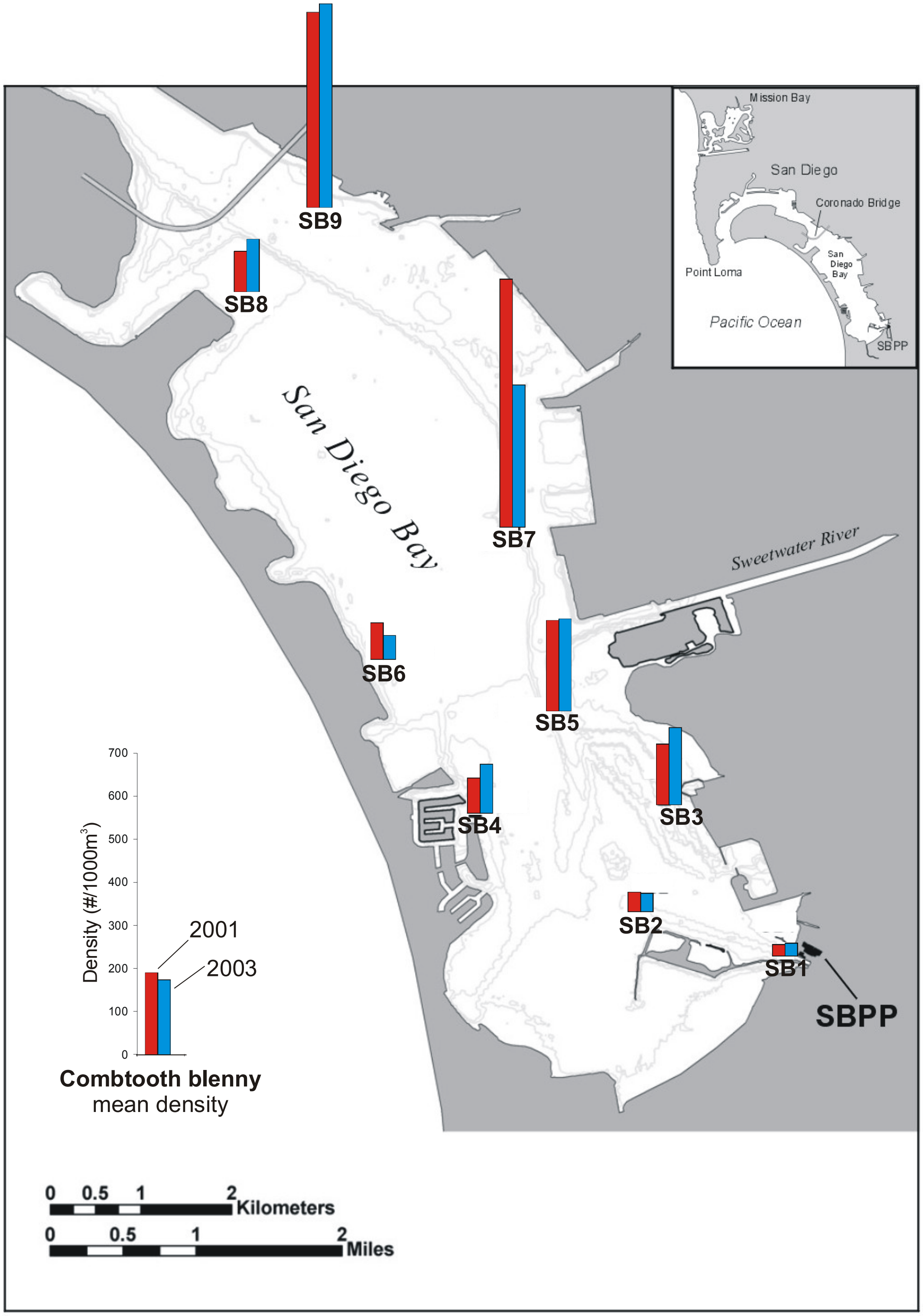


Figure 3.3-21. Annual mean density (#/1000 m^3) of combtooth blenny larvae at entrainment station (SB1) and source water stations (SB2-SB9) in south San Diego Bay during the 2001 and 2003 sampling periods. Mean density for all stations combined is shown with scale.

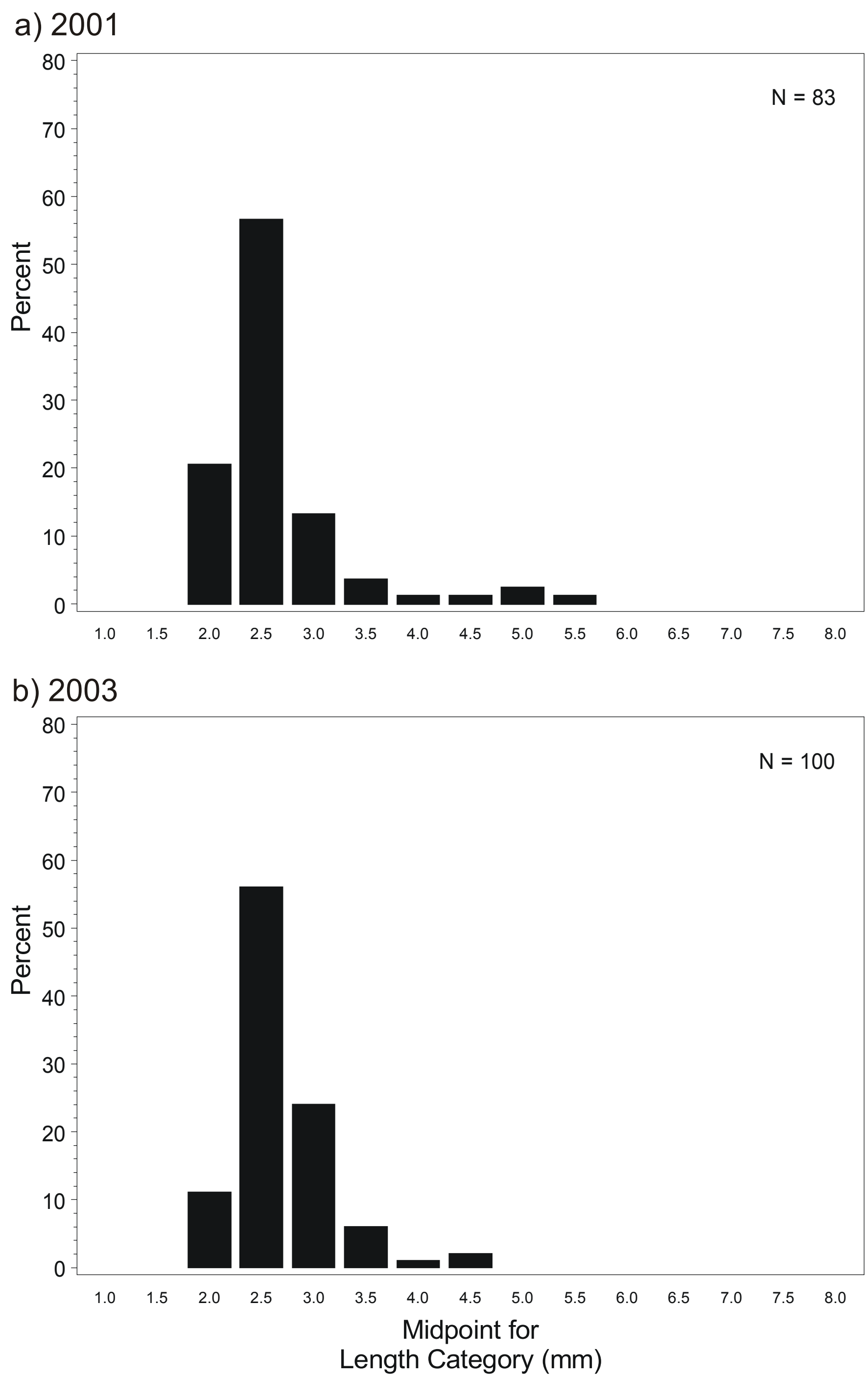


Figure 3.3-22. Length frequency distribution of blenny larvae collected at entrainment station SB1: a) 2001 period and b) 2003 period.

4.0 Impingement Study

4.1 Introduction

The purpose of the SBPP impingement study was to evaluate the potential impacts of the operation of the circulating water intake system as required under Section 316(b) of the Federal Clean Water Act (CWA) (USEPA 1977). The San Diego Regional Water Quality Control Board (RWQCB) reviewed the need for and design of the studies with representatives of Duke Energy, Tenera Environmental, Merkel and Associates, USFWS, and other agencies. The group reviewed and approved the final 316(b) Cooling Water Intake Effects Study Plan.

The impingement study was designed to specifically address the following questions:

- What are the species composition and abundance of the juvenile and adult fishes and macroinvertebrates impinged by SBPP?
- What are the potential impacts of impingement losses on populations of fishes and macroinvertebrates due to operation of the CWIS?

4.1.1 Review of Previous Impingement Study

A previous impingement study was conducted at SBPP from February 1979 through January 1980 as part of the plant's initial Section 316(b) Demonstration requirement (SDG&E 1980). A total of 150 separate 24-hour collections from the circulating water intake screening system were completed during this earlier study. A total of 13,335 fishes, sharks, and rays of 63 taxa with an aggregate wet weight of 853 kg (1,881 lb) were collected. The analysis focused on six “critical” fish taxa: slough anchovy *Anchoa delicatissima*, topsmelt *Atherinops affinis*, California halibut *Paralichthys californicus*, round stingray *Urolophus halleri*, speckled midshipman *Porichthys myriaster*, and striped bass *Morone saxatilis*. No information on invertebrate impingement was reported in this earlier study.

The results of the 1979–1980 impingement study can be summarized as follows:

- Annual impingement by SBPP based on the actual number of fishes collected was estimated to be 28,174 fishes weighing 4,459 kg (9,830 lb).
- The numerically most abundant impinged species in order of decreasing abundance were round stingray, topsmelt, deepbody anchovy, specklefin midshipman, slough anchovy, and Pacific butterfish. The six species with the

heaviest wet weight biomass were specklefin midshipman, round stingray, deepbody anchovy, Pacific butterfish, topsmelt, and diamond turbot.

- Maximum impingement occurred during May. Impingement was generally higher from November through June than from July through October.
- Slough anchovy was impinged throughout the year with greatest impingement during May and June.
- Impingement of topsmelt was greatest during November and December and lowest during June through October.
- The total number of fishes impinged at night was greater than during the day, especially for topsmelt, specklefin midshipman, round stingray, and deepbody anchovy.
- Impingement abundance appeared not to be correlated with changes in ambient water temperature, tidal height, or weather conditions.
- Impingement abundance generally increased as a function of the plant's circulating water flow.

The study concluded that impingement losses caused by the operation of the SBPP CWIS had no significant effects on the adult fish populations in San Diego Bay and that the design and operation of the CWIS represented the best technology available for minimizing adverse environmental impacts.

A description of the fish communities in south San Diego Bay is presented in Section 2.2–*San Diego Bay Environmental Setting* of this report and Section 3.4–*Fish Communities* of Volume 1: South Bay Power Plant 316(a) Thermal Discharge Assessment.

4.2 Methods

The following sections provide information on impingement sample collection and field processing, and also on methods used to assess impingement impacts. The impingement sampling program was designed to provide current estimates of the abundance, taxonomic composition, diel periodicity, and seasonality of organisms impinged at SBPP. This was accomplished by calculating the rates (i.e., number or biomass of organisms per m^3 water flowing per time into the plant) at which various species of fishes and macroinvertebrates were impinged. Impingement rates are subject to tidal and seasonal influences that vary on several temporal scales (e.g., hourly, daily, and monthly) while the rate of circulating water flow varies with power plant operations.

4.2.1 Sampling

The SBPP has three separate shoreline intake structures, one for Units 1 & 2 and one each for Units 3 & 4, which withdraw water from south San Diego Bay for cooling purposes. These structures house the bar racks, the vertical traveling water screens, and the circulating water pumps. Seawater entering the intake structures first passes through the bar racks, followed by the traveling screens and then the pumps. All material that passes through the bar racks but was larger than the traveling screen mesh (either 9 mm (3/8 in) square or 3 mm by 13 mm (1/8 in by ½ in) rectangular openings) was impinged and was subsequently rinsed from the screens by a high-pressure wash system when the screens were rotated for cleaning. The material rinsed from the screens normally is returned to the discharge area (see **Figure 2.1-1**) via a trough, but was collected during impingement sampling. A more complete description of the entire circulating water intake system including the operation of the traveling screens is presented in Section 2.1–*Description of the South Bay Power Plant's Circulating Water System* of this report.

Impingement sampling at SBPP was conducted during a 24-hr period one day each week from December 5, 2002 through November 26, 2003. Each sampling period was divided into six approximately 4-hr cycles. In almost every survey throughout this study all eight circulating water pumps (two per unit) were operated during the entire 24-hr sampling period. Before each weekly sampling effort, all of the screens were rotated and rinsed clean of all impinged material. A trap door in the screen wash trough was then opened so that all impinged material would fall into a collection basket. The collection baskets used during this study were the same ones used in the earlier impingement study and were constructed from stainless steel and had ¼ inch diameter holes. During each cycle the traveling screens remained stationary for a period of approximately 3.5 hr, and were then rotated and rinsed for 30 min. This rinse period allowed the entire traveling screen to be rinsed of all material that had been impinged since the last screen wash cycle. In a few instances during impingement collections, the screen wash system started automatically due to a high differential pressure prior to the end of the cycle. The material that was rinsed from the screens during the automatic screen washes was combined with the material collected at the end of that cycle. All debris and organisms rinsed from the Units 1 & 2 traveling screens were kept separate from the material from the Units 3 & 4 traveling screens.

All fishes and selected macroinvertebrates collected at the end of each 4-hr cycle were removed from the debris and then identified and counted. Individual weights and lengths of bony fishes and sharks and rays were recorded (standard length [SL] for the bony fishes and total length [TL] for the sharks and rays). Any mutilated fishes were identified, if possible, and the total weight recorded by taxa. No length measurements were recorded for mutilated fishes. Carapace width was measured for crabs and total length was measured for shrimps and cephalopod mollusks. Weight was also recorded for these invertebrates. Other invertebrates, including hydroids, anemones, sea jellies, barnacles, worms, brittlestars, bryozoans, tunicates, gastropods, and bivalves, were not enumerated or weighed but were only recorded as present when found in the impinged material.

During periods when many fishes or invertebrates were impinged during a single cycle, a maximum of 50 individuals of any one taxa were measured and weighed. All lengths were recorded to the nearest 0.1 mm and all weights to the nearest 0.1 g. The condition (alive, dead, or mutilated) of the organism was also recorded as was the amount and type of impinged debris. In addition, the operating status of the circulating water pumps and traveling screens was also recorded. All data were recorded on sequentially numbered data sheets, verified, and subsequently entered into a computer database.

A quality control (QC) program was implemented to ensure the correct identification, enumeration, length and weight measurements of the organisms were recorded on the data sheet. Random cycles were chosen for QC re-sort to verify that all the collected organisms were removed from the impinged material.

A log containing hourly observations of the operating status (on or off) of the eight circulating water pumps for the entire study period was obtained from the power plant's operation staff. This provided a record of the volume of circulating water pumped through the plant, which was used to calculate impingement rates.

4.2.2 Data Analysis

To estimate taxa-specific impingement rates we first calculated the circulating water flow during each of the six cycles of the 24-hr survey. The total time for each cycle (generally

4 hr) was multiplied by the manufacturer's rated flow of each of the pumps that had operated during the cycle. Each unit has two pumps with the following flow rates: Units 1 & 2 pumps–148 m^3/min /pump (39,000 gpm), Unit 3 pumps–236 m^3/min/pump (62,300 gpm), and Unit 4 pumps–259 m^3/min/pump (68,400 gpm). In the few instances when the traveling screen was not operational during sampling, the water flow for that pump was not added into the total for that cycle, as impinged organisms were not collected from that screen**.** The circulating water flow rate for each cycle (obtained from the plant's operator pump logs showing which pumps were operating and manufacturer's rated flow for each operating pump) was then used to calculate an average daily impingement rate and associated standard error per volume of circulating water for each taxa for the two unit pairs (Units 1 & 2 or Units 3 & 4). Although many of the impinged fishes were juveniles, for analysis purposes it was assumed that they were all adults and that none of the impinged organisms survived.

An adjustment was made to the total weight of each taxa to compensate for any mutilated fishes that were collected and not weighed. The average weight of non-mutilated individuals of a given taxa collected in each cycle was assigned to any mutilated individuals in that cycle. This adjusted weight was then used in all biomass calculations.

The estimated daily impingement rate was then used to calculate estimated weekly, monthly, and annual impingement. The days between the impingement collections were assigned to a weekly survey period by setting the collection day as the median day within the period and assigning the days on either side of the collection date to the closest adjacent sampling day to create a weekly survey period. In most cases, the weekly survey periods were 7 d, but in a few instances the survey period varied from 5–9 d in length. The total calculated flow for each weekly survey period was multiplied by the taxon-specific impingement rate calculated from the daily sampling to obtain estimates of the weekly impingement rates of both counts and biomass for each taxon. Finally, the estimated abundance and biomass impingement rate for each survey period was summed to determine monthly and annual estimates of impingement for each taxon for the yearlong study period.

Historically, SBPP operates its circulating water pumps only when its units are in operation. **Figure 2.1-2** presents the pump flow volume during the yearlong impingement study. All of the pumps were generally only operated during the impingement collection days yielding an annual pumping capacity of approximately 68% of the design maximum. To determine the maximum or "worst-case" possible impingement impacts, the following analysis was based on a hypothetical year when all of the pumps were operated 24 hr every day.

During impingement sampling all fishes and invertebrates that were retained on the traveling screens were rinsed from the screens, flowed along the trough and was deposited into the collection baskets. Data for all impinged taxa are presented in this report, but a subset of the taxa was selected for more detailed analysis. This included fishes that comprised the top 95 percent of the total abundance and biomass plus any taxon that was commercially or recreationally important and in the top 95 percent of the total abundance or biomass. The impinged commercially or recreationally important invertebrates that were in the top 90 percent of the total abundance or biomass are also discussed in more detail in the following sections.

4.3 Fish Impingement Results

4.3.1 Fish Community Overview

A total of 50,970 fishes weighing a total of 74 kg (163 lb) and comprising approximately 50 taxa was impinged during this 12-month study (**Table 4.3-1**). The vast majority of the collected fishes (over 93 percent) were anchovies (*Anchoa* spp.). The next most common fishes were silversides Atherniopsidae (mainly topsmelt *Atherinops affinis*), pipefishes *Syngnathus* spp., California halfbeak *Hyporhampus rosae*, specklefin midshipman *Porichthys myriaster*, gobies Gobiidae, and round stingray *Urolophus halleri*. The wet weight biomass was also dominated by anchovies (39.9 percent) followed by round stingray, specklefin midshipman, bat ray *Myliobatis californica*, and silversides.

The estimated total annual abundance of impinged fishes at SBPP was 385,588 based on continuous flow of all eight circulating water pumps for an entire year (**Table 4.3-1**). The estimated annual biomass of impinged fishes was 556.2 kg (1,226.4 lb).

Figure 4.3-1 presents the numbers and biomass of fishes collected during each of the 52 impingement surveys for all four units combined. High abundances of impinged fishes were recorded during February and November. Peaks in biomass of impinged fishes occurred during February, April, and May. Generally there was a greater abundance and biomass of fishes impinged from Units 1-4 during nighttime hours than during the daylight hours (**Figures 4.3-2** and **4.3-3**).

The intake screen wash system for Units 1 & 2 at SBPP is separate from that of Units 3 & 4, and impingement data were recorded separately for each of the two unit groups **(Appendix D).** About 80 percent of the total abundance and 86 percent of the total biomass of fishes was impinged at Units 3 & 4 (**Table 4.3-1** and **Appendix E**).

Four fish taxa were evaluated in detail for potential power plant effects on their local populations: anchovies, silversides, sand basses *Paralabrax* spp., and shortfin corvina *Cynoscion parvipinnis*. These species were chosen for detailed evaluation because they comprised a significant fraction of the impinged fishes by number or biomass, and have commercial or recreational fishery value.

Table 4.3-1. Summary of SBPP Units 1-4 fish impingement from December 2002 through November 2003 (52 surveys).

		Sampled Abundance			Estimated Annual Impingement			
Taxon	Common Name	(#)	(kg)	(lb)	(#)	Std. Err.	(kg)	Std. Err.
Anchoa spp.	anchovies	47,746	29.53	65.12	359,420	105,476.2	222.01	58.62
Atherinopsidae	silversides	1,293	2.72	6.00	11,664	8,106.0	25.69	36.39
Syngnathus spp.	pipefishes	433	0.32	0.71	3,218	642.2	2.35	0.51
Hyporhamphus rosae	California halfbeak	361	0.43	0.95	2,765	722.6	3.21	1.05
Porichthys myriaster	specklefin midshipman	253	9.23	20.35	1,850	665.3	66.16	48.90
Gobiidae	gobies	241	0.17	0.37	1,791	420.2	1.25	0.42
Urolophus halleri	round stingray	203	16.46	36.29	1,532	490.6	124.57	45.56
Cymatogaster aggregata	shiner surfperch	77	0.28	0.62	549	216.2	2.24	1.71
Strongylura exilis	California needlefish	63	1.37	3.03	510	330.1	11.73	13.76
Cynoscion parvipinnis	shortfin corvina	60	1.57	3.46	428	178.3	10.93	9.85
Hypsopsetta guttulata	diamond turbot	54	0.63	1.38	382	197.8	4.52	4.20
Fundulus parvipinnis	California killifish	26	0.03	0.06	191	107.8	0.20	0.14
Myliobatis californica	bat ray	24	4.40	9.70	172	102.2	30.97	19.91
Hippocampus ingens	Pacific seahorse	23	0.19	0.42	165	83.1	1.27	1.41
Heterostichus rostratus	giant kelpfish	22	0.20	0.45	163	87.4	1.43	1.08
Seriphus politus	queenfish	21	0.52	1.15	152	91.0	3.66	6.10
Acanthogobius flavimanus	yellowfin goby	10	<0.01	0.01	73	73.2	0.03	0.03
Gymnura marmorata	California butterfly ray	8	1.46	3.21	56	48.8	10.37	11.07
Leptocottus armatus	Pacific staghorn sculpin	6	0.03	0.07	58	59.8	0.27	0.39
unidentified fish	unidentified fish	6	0.01	0.03	42	48.5	0.09	0.13
Cololabis saira	Pacific saury	6	0.01	0.02	42	39.1	0.05	0.06
Pleuronectidae	flounders	6	0.01	0.01	56	81.7	0.05	0.07
Sardinops sagax	Pacific sardine	4	0.15	0.33	28	42.0	1.05	1.86
Porichthys notatus	plainfin midshipman	4	0.15	0.32	28	34.3	1.03	2.50
Paralabrax spp.	sand basses	3	1.60	3.52	24	34.7	13.06	19.29
Gillichthys mirabilis	longjaw mudsucker	3	0.04	0.08	24	35.0	0.27	0.39
Pleuronichthys spp.	turbots	3	<0.01	0.01	21	51.7	0.02	0.05
Sciaenidae	croakers	3	<0.01	<0.01	26	35.7	0.01	0.01
Mugil cephalus	striped mullet	2	1.50	3.31	14	24.2	10.50	25.72
Cynoscion nobilis	white seabass	2	0.09	0.21	14	24.2	0.65	1.56
Engraulis mordax	northern anchovy	2	0.01	0.01	16	28.3	0.05	0.10
Lepidogobius lepidus	bay goby	2	<0.01	<0.01	14	24.2	0.01	0.01
Cheilotrema saturnum	black croaker	1	0.46	1.01	7	17.1	3.22	7.89
Dasyatis brevis	diamond stingray	1	0.42	0.93	7	17.1	2.94	7.21
Paralichthys californicus	California halibut	1	0.01	0.03	7	17.1	0.09	0.21
Blenniidae	combtooth blennies	1	0.01	0.03	5	11.3	0.06	0.14
Tridentiger trionocephalus	chameleon goby	1	0.01	0.02	9	21.7	0.09	0.23
Gibbonsia spp.	clinid kelpfishes	1	<0.01	0.01	9	20.0	0.03	0.07
Hypsoblennius spp.	combtooth blennies	1	<0.01	0.01	7	17.1	0.02	0.05
Pleuronichthys ritteri	spotted turbot	1	<0.01	<0.01	7	17.1	0.01	0.03
Porichthys spp.	midshipmans	1	<0.01	<0.01	7	17.1	0.01	0.02
Albula vulpes	bonefish	1	<0.01	<0.01	7	17.1	0.01	0.02
Lepidopsetta bilineata	rock sole	1	<0.01	<0.01	7	17.0	0.01	0.02
Stichaeidae	pricklebacks	1	<0.01	<0.01	7	17.1	<0.01	0.01
larval/post-larval fish	larval/post-larval fish	1	<0.01	<0.01	7	17.3	<0.01	0.01
Pleuronectiformes	flatfishes	1	<0.01	<0.01	6	14.6	<0.01	<0.01
	TOTAL	**50,984**	**74.03**	**163.23**	**385,588**		**556.18**	
	% impingement total from Units 1-2	20.3	14.4					
	% impingement total from Units 3-4	79.7	85.6					

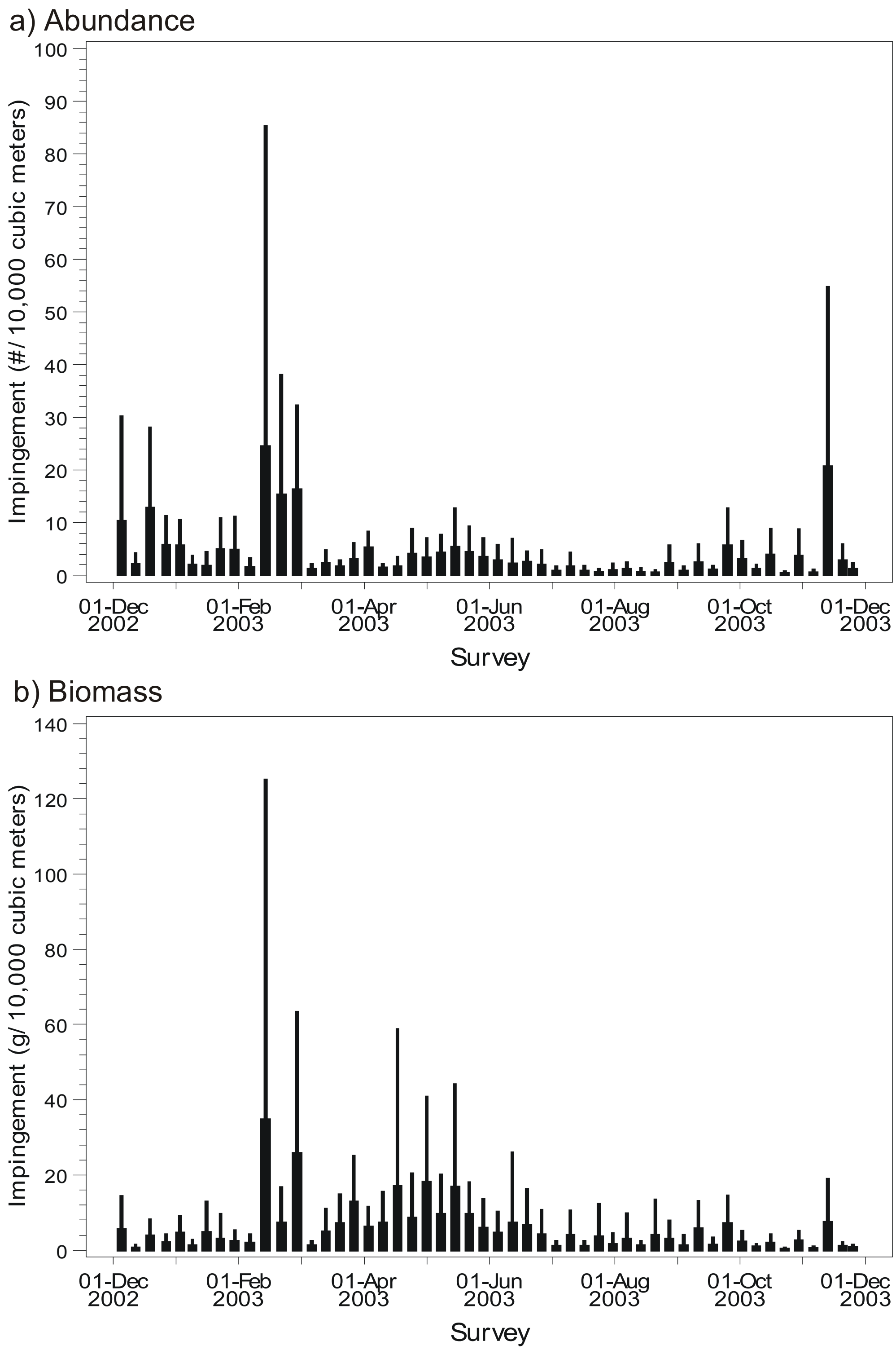


Figure 4.3-1. Mean concentration and standard error of fishes impinged at SBPP Units 1-4 December 2002 through November 2003 (*n*=52 surveys): a) abundance, and b) biomass.

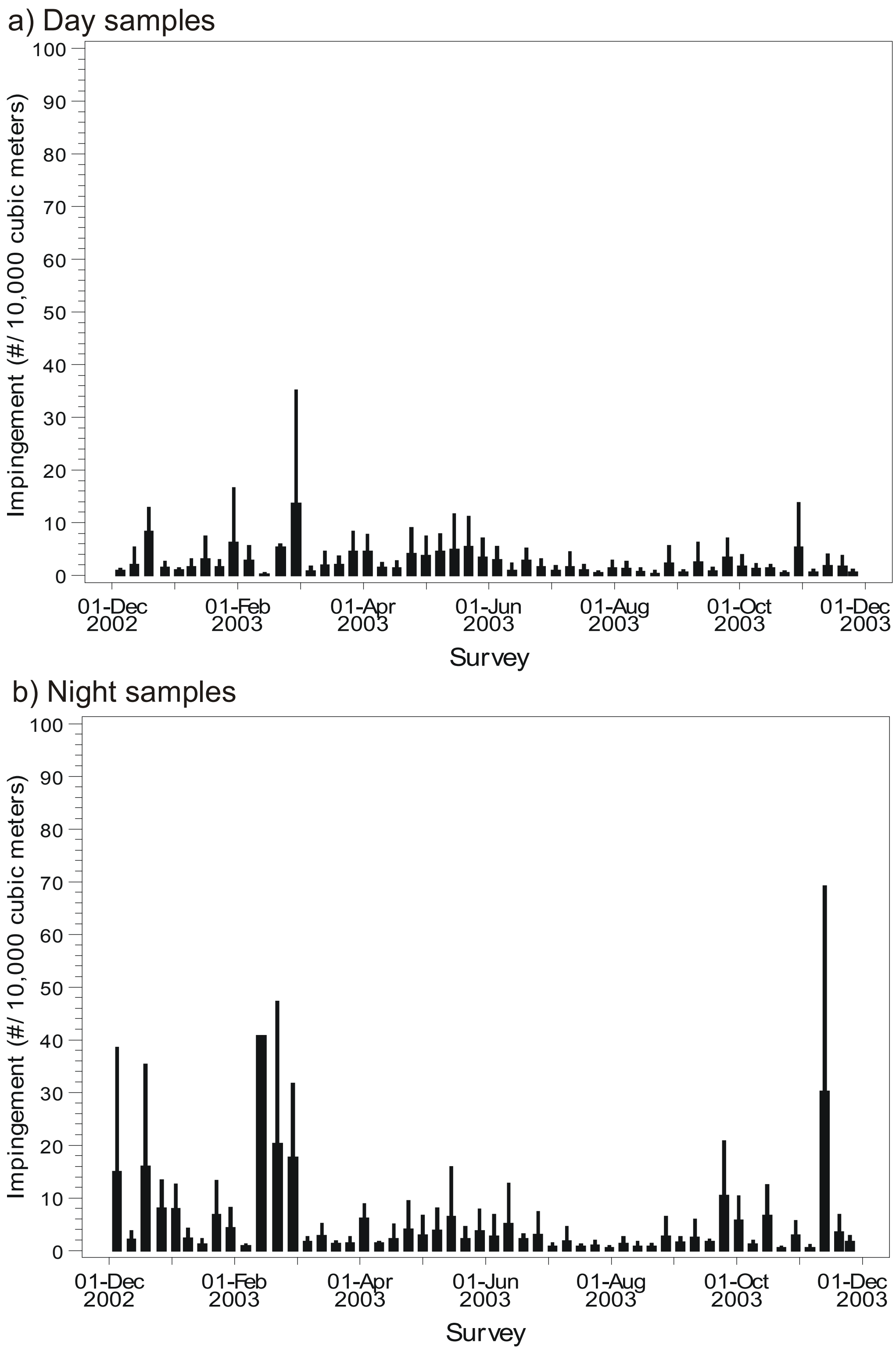


Figure 4.3-2. Mean abundance and standard error of fishes impinged at SBPP Units 1-4 December 2002 through November 2003 (n=52 surveys): a) day samples, and b) night samples.

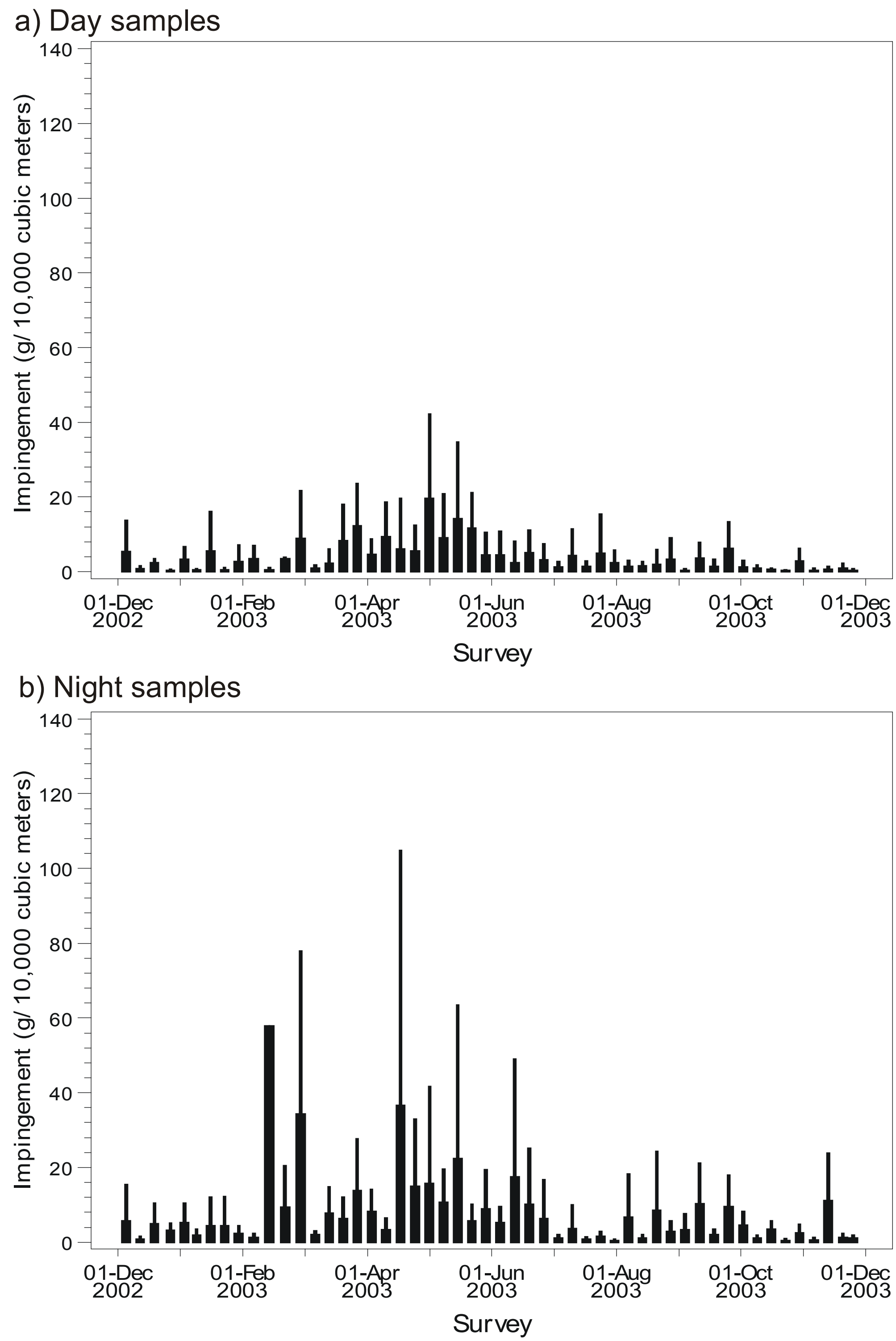


Figure 4.3-3. Mean biomass and standard error of fishes impinged at SBPP Units 1-4 December 2002 through November 2003 (*n*=52 surveys): a) day samples, and b) night samples.

4.3.2 Anchovies (*Anchoa* spp.)

The slough anchovy *Anchoa delicatissima* and the deepbody anchovy *Anchoa compressa* are both common to abundant in the southern portions of San Diego Bay (Allen (1999), Merkel & Associates, Inc. 2000a). The two species of *Anchoa* are known to fluctuate in relative abundance from year to year in southern California estuaries, but the causes of these changes are not known (CMI 2003, Emmett et al. 1991). Anchovy larvae were abundant in plankton samples collected as part of the entrainment impact portion of the present study, and their life history is presented in Section 3.3.4 of this report.

4.3.2.1 Results

Most of the individuals from this group were identified as either slough anchovy or, for the small juveniles that were difficult to separate into species, *Anchoa* spp. (**Appendix D**). Only three adult deepbody anchovy were identified during the entire impingement study. It is likely, therefore, that a vast majority of the small individuals recorded as *Anchoa* spp. were actually slough anchovy. The abundance and biomass of all three taxa were combined for analysis.

Anchovies were the most abundant group of impinged fishes (47,732 individuals) and also had the greatest biomass (29.5 kg [65.1 lb]) (**Table 4.3-1**). They were impinged throughout the year and were most abundant during winter and least abundant during late summer (**Figure 4.3-4**). There was a wide range of lengths impinged (from 12–125 mm [0.5–4.9 in]) with the mean length of 41 mm (1.6 in) (**Figure 4.3-5)**. The smallest monthly mean length was recorded during December 2002 (30.2 mm [1.2 in]) and the monthly mean lengths gradually increased (to 51.3 mm [2.0 in]) in July 2003 as the cohort matured (**Appendix D**). Smaller individuals started to appear in August 2003, with a subsequent decrease in monthly mean length through November 2003 (mean 34.5 mm [1.4 in]).

4.3.2.2 Impact Assessment

Based upon the impinged abundance and biomass of anchovies presented above and an assumed continual operation of all eight SBPP circulating water pumps, the estimated annual impingement abundance of anchovies was 359,420 (±105,476 std. error), with a biomass of 222.0 kg (489.5 lb) (±58.6 kg std. error) (**Table 4.3-1**). The vast majority of the anchovies in this group were the slough anchovy, a species that has a maximum length of 94 mm (3.7 in). Although many of the anchovies impinged were small (mean length 41 mm [1.6 in]), for purposes of this assessment they were all assumed to be reproductively mature adults.

Slough anchovy were the dominant anchovy species in the SBPP discharge channel (Merkel & Associates, Inc. 2000a) and one of the most numerous finfishes in south San Diego Bay (Allen 1999). They accounted for 91.4 percent of all individuals captured over a three-year sampling period (April 1997 through January 2000) conducted in the discharge channel (Merkel & Associates, Inc. 2000a). In most instances, both juveniles and adults were captured during sampling (Merkel & Associates, Inc. 2000a). In a five-year study (July 1994 through April 1999) of fishes in San Diego Bay, Allen (1999) reported that slough anchovy comprised almost 20 percent of all individuals sampled and about 7 percent of the total biomass from all regions of San Diego Bay. Slough anchovy varied greatly in abundance over each year of the study (July 1994 through April 1999) but were found in virtually every month of the study in the southern regions of the bay. Peak abundance occurred in July of most years due to heavy recruitment of young-of-the-year (YOY). Allen (1999) estimated the standing stock of *Anchoa delicatissima* in the south and south central ecoregions at about 10,170 and 22,580 kg, respectively. Based on this estimate, SBPP impinges about 0.7 percent of the estimated *A. delicatissima* in the southern portion of San Diego Bay.

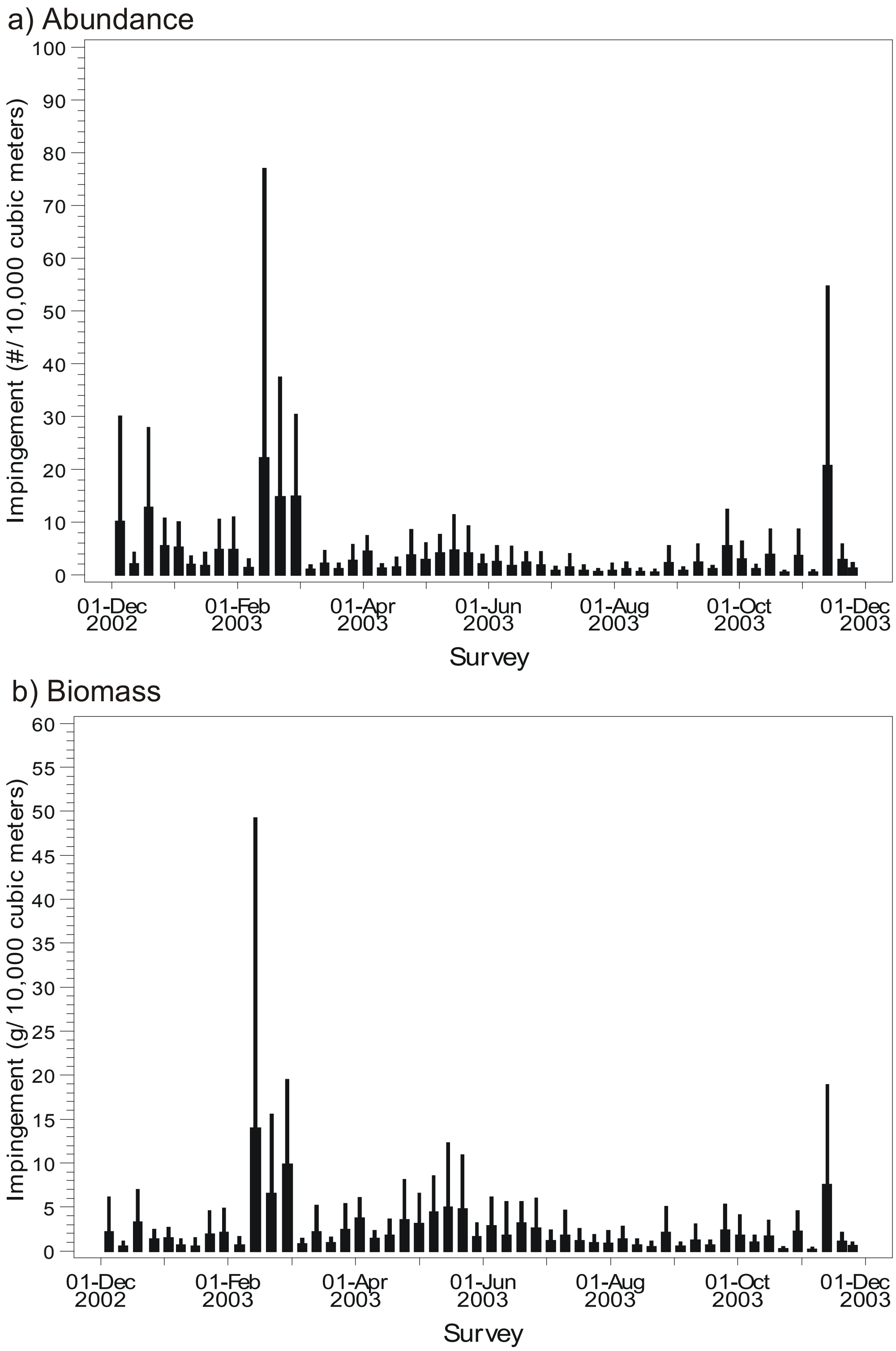


Figure 4.3-4. Mean concentration and standard error of anchovies *Anchoa* spp. impinged at SBPP Units 1-4 December 2002 through November 2003 (n=52 surveys): a) abundance, and b) biomass.

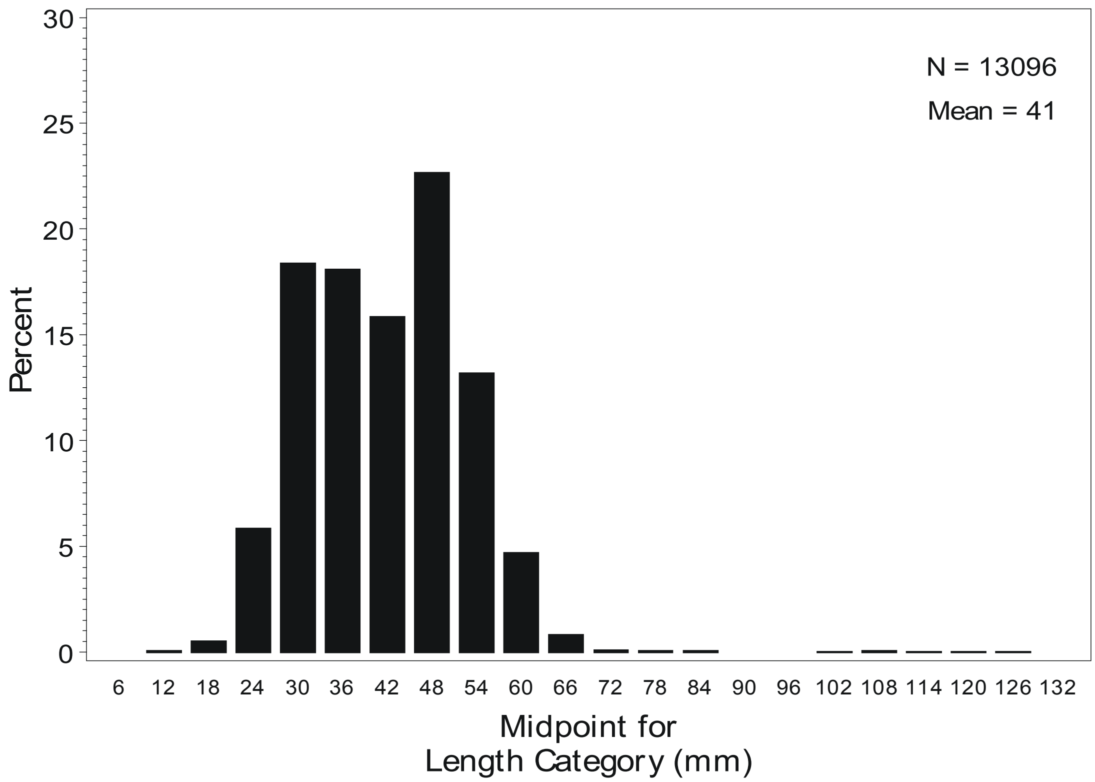


Figure 4.3-5. Size frequency distribution of anchovies *Anchoa* spp. from SBPP Units 1-4 impingement samples.

4.3.3 Silversides (Atherinopsidae)

Three species of silversides (family Atherinopsidae) occur in San Diego Bay: topsmelt *Atherinops affinis*, jacksmelt *Atherinopsis californiensis*, and the California grunion *Leuresthes tenuis*. Silverside larvae were abundant in plankton samples collected as part of the entrainment impact portion of this study, and their life history is presented in Section 3.3.5 of this report.

4.3.3.1 Results

Two silverside species, topsmelt and jacksmelt, were impinged during the study. Of the 1,293 silversides there were 1,273 topsmelt, 4 jacksmelt, and 16 others that could not be identified to the species level and were recorded as Atherinopsidae. The impinged silversides had a combined total weight of 2.72 kg (6.0 lb) (**Table 4.3-1**) in the 52 weekly surveys. It was the second most abundant fish impinged and had the fifth highest biomass.

Impingement of silversides was seasonal, occurring mainly from February through May (**Figure 4.3-6**). The majority of the biomass was recorded during one survey in February 2003. Lengths ranged from 14–228 mm [0.6–9.0 in] SL, with a mean length of 41.6 mm (1.6 in) (**Figure 4.3-7**).

4.3.3.2 Impact Assessment

Based on the impinged abundance and biomass of silversides presented above and the continual operation of all eight circulating water pumps, the estimated annual impingement abundance of this group was 11,664 individuals (± 8,106 std. error), weighing 25.7 kg (56.7 lb) (±36.4 kg std. error). The majority of the silversides impinged were topsmelt, which can grow to a maximum length of 370 mm (14.6 in). Although many of the silversides impinged were small (mean length ca. 42 mm [1.7 in]), for purposes of this assessment they were all assumed to be adults.

In a five-year study of fishes in San Diego Bay, topsmelt ranked second in abundance and fifth in biomass, comprising about 23 percent of the individuals and 9 percent of the total weight (Allen 1999). Topsmelt were captured in all samples with peak abundances generally occurring in April due to heavy recruitment of young-of-the-year (YOY). Typically, YOY and juvenile topsmelt primarily occurred in the intertidal zone while adult fish occupied nearshore and midwater channel sub-habitats. Merkel and Associates, Inc. (2000a) collected topsmelt during almost every month, representing about 1 percent of the total catch by number. No jacksmelt or grunion were collected in the discharge channel.

Allen (1999) estimated the standing stock of topsmelt in the south and south central ecoregions was about 5,384 and 5,492 kg, respectively. Based on the total of these two estimates, the biomass of silversides impinged by SBPP represents about 0.2 percent of the estimated standing stock for this area of San Diego Bay.

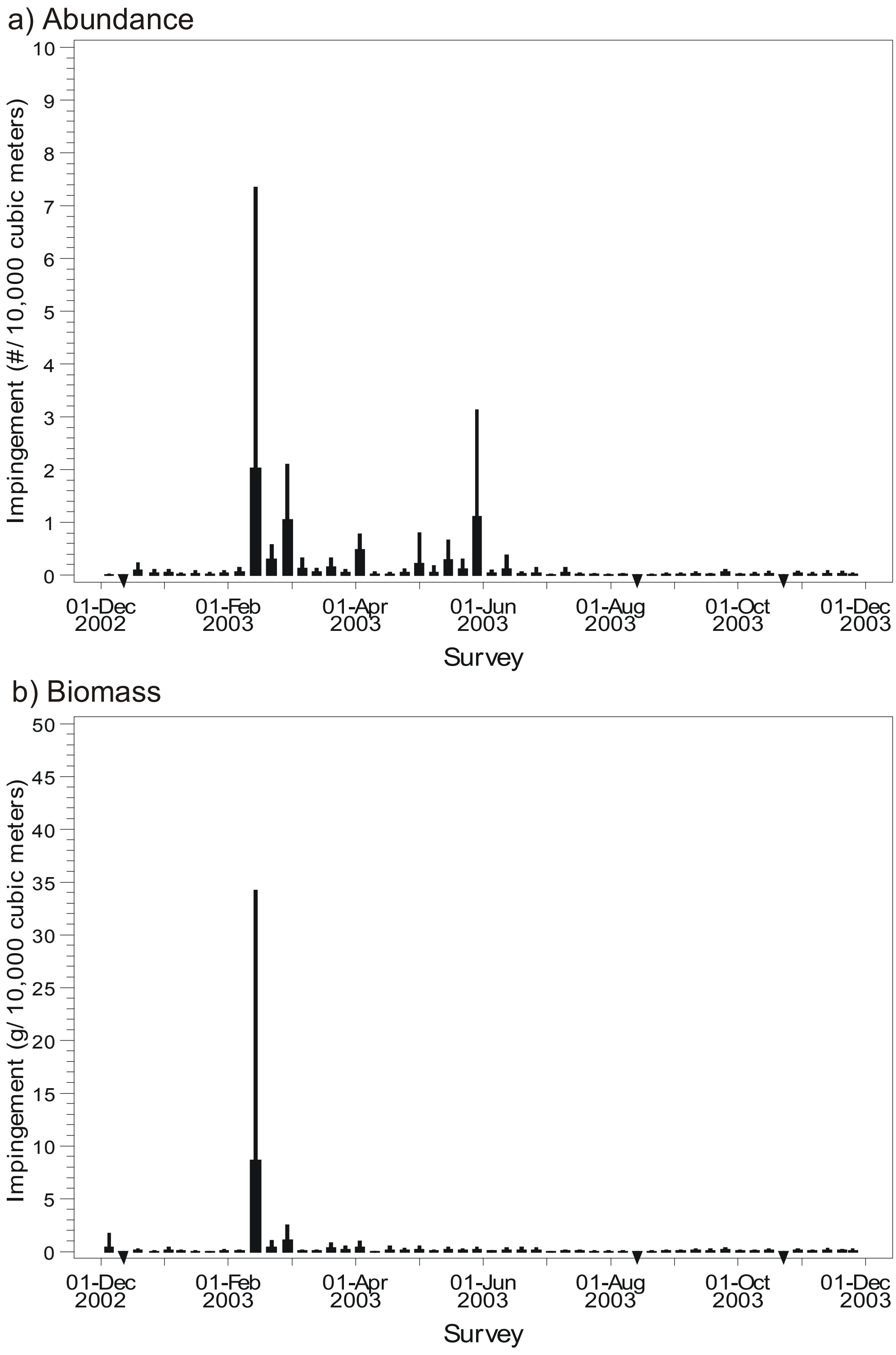


Figure 4.3-6. Mean concentration and standard error of silversides Atherinopsidae impinged at SBPP Units 1-4 December 2002 through November 2003 (n=52 surveys): a) abundance, and b) biomass. Triangles indicate surveys where none was present.

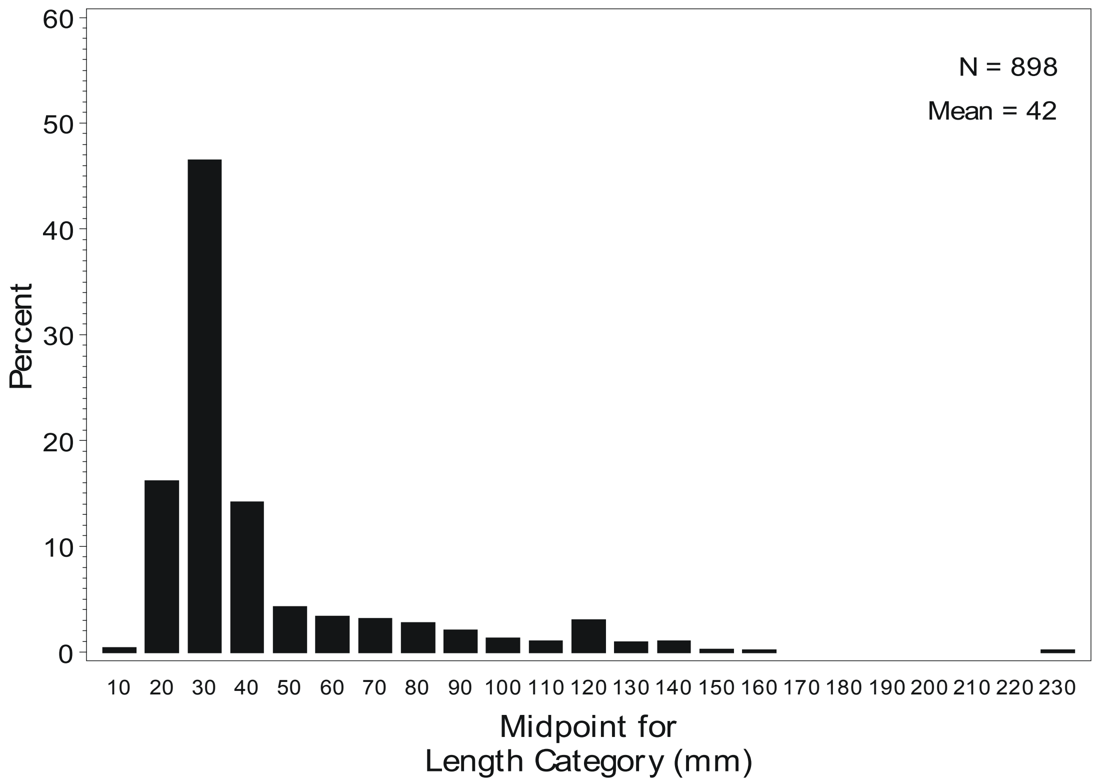
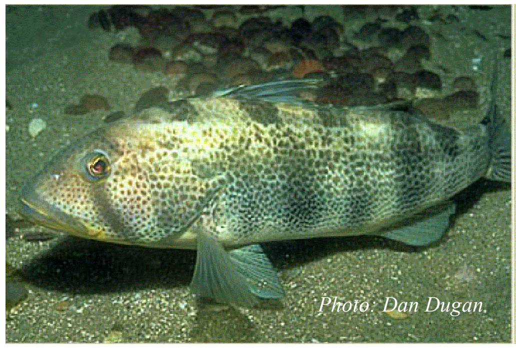
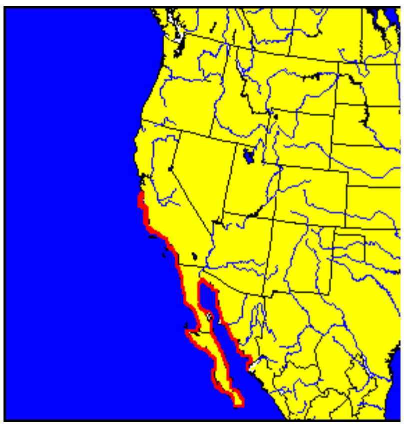


Figure 4.3-7. Size frequency distribution of silversides Atherinopsidae from SBPP Units 1-4 impingement samples.

4.3.4 Sand basses (*Paralabrax* spp.)





Distribution map for *Paralabrax* spp.

Range: Monterey, California to Mazatlan, Mexico, including the Gulf of California;

Life History: Size: to 65 cm (25.6 in);
Size at maturity: to 26.7 cm (10.5 in);
Fecundity: up to 185,00 eggs; Life span: to 24 yr;

Habitat: shallow water rock-sand ecotone; nearshore sand flats, near kelp beds, rocky areas, and bays.

Fishery: Sport fishery only; no commercial fishery allowed.

4.3.4.1 Life History and Fishery

Three species of basses, family Serranidae, genus *Paralabrax*, occur in the San Diego region: spotted sand bass, *P. maculatofasciatus,* barred sand bass *P. nebulifer*, and kelp bass *P. clathratus*. In San Diego Bay, the spotted and barred sand basses comprise the majority of serranids and are the focus of this section.

Spotted sand bass is found from Monterey, California to Mazatlan, Mexico, including the Gulf of California (Miller and Lea 1972). However Love (1996) reports that they not common north of Newport Bay in southern California. Although they have been taken in water as deep as 61 m (200 ft) they are usually found shallower than 6.1 m (20 ft) (Love 1996).

In a study of fish populations in San Diego Bay, Allen et al. (2002) reported that spotted sand bass were captured throughout the bay during their July 1994 through April 1999 study period and were considered one of the most important resident fish species. Abundance of spotted sand bass was variable over time with peaks in abundance occurring in April and July in the northern portion of San Diego Bay and October and January in the southern portion. Young-of-the-year (YOY) recruitment occurred in October, primarily in the north central and south central portions of the bay (Allen 1999)

Love (1996) summarized the life history of the spotted sand bass. Adults can reach 56 cm (22 in) in length and live to at least 14 yr. Females mature within the first year and half are mature when they are approximately 15 cm (6 in) long. Males reach maturity at approximately 3 yr with half of the individuals mature at 18 cm (7 in). Some individuals in the populations are protogynous, changing sex from female to male as they grow. Spawning in California populations occurs from June through August.

The barred sand bass, *Paralabrax nebulifer*, can be found to depths of 183 m (600 ft). Adults and sub-adults are most abundant from 5 to 26 m (16 to 85 ft) over nearshore sandy flats, near kelp beds, rocky areas, and in bays and estuaries whereas juveniles are often found in eelgrass beds during fall and winter. Barred sand bass was the most common trawl-caught fish in Mission Bay, and is also common in San Diego Bay and in lower Newport Bay, California. Love (1996) reports that the greatest abundance of adult barred sand bass appears to be near "edge" habitats where rocky and sandy substrates meet. Eggs and larvae are pelagic, while juveniles and adults are benthopelagic. Adults usually remain within a few meters over the substrate.

Abundance of *P. nebulifer* was variable over time with peaks in abundance occurring in April, July, and October mainly in the northern portion of San Diego Bay (Allen 1999). Barred sand bass captured in San Diego Bay were mainly juveniles representing a relatively narrow range of sizes indicating that the bay is an important nursery area for this primarily coastal species. YOY recruitment occurred in October, primarily in the north central and south central portions of the bay.

Sport fishery catch estimates of spotted sand bass in the southern California region from 1998 to 2002 ranged from 17,000–74,000 annually with a mean of 45,600 fish (RecFIN 2003). Sport catches of barred sand bass during the same time period were much larger, ranging from 410,000 to 1,130,000 and averaging 773,400 fish.

4.3.4.2 Results

Three *Paralabrax* spp. with a total weight of 1.6 kg (3.5 lb) were impinged at SBPP during this study (**Table 4.3-1**). These individuals were collected in December and May (**Figure 4.3-8**). They measured 229, 284, and 308 mm SL (9.0, 11.2, and 12.1 in) (**Appendix D**).

4.3.4.3 Impact Assessment

Based on the impinged abundance and biomass of sand bass presented above and the continual operation of all eight circulating water pumps, the estimated annual impingement abundance of this group was 24 individuals (±34.7 std. error), weighing 13.1 kg (28.8 lb) (±19.3 kg std. error).

In a five-year study of fishes in San Diego Bay, barred sand bass ranked fourteenth in abundance and second in biomass, comprising about 0.3 percent of the individuals and 14 percent of the total weight (Allen 1999). Barred sand bass were collected at almost all stations during almost every month of the year. Peaks in abundance of this species occurred in October and January and all size classes, from juveniles to adults, were typically represented.

Allen (1999) estimated the stock of barred sand bass in south and south central ecoregions (the area considered the source water for the entrainment portion of this report) at 19,442 and 10,536 kg, respectively. Based on a combination of these two estimates, the biomass of sand bass impinged by SBPP represents about 0.003 percent of the estimated stock for this portion of San Diego Bay.

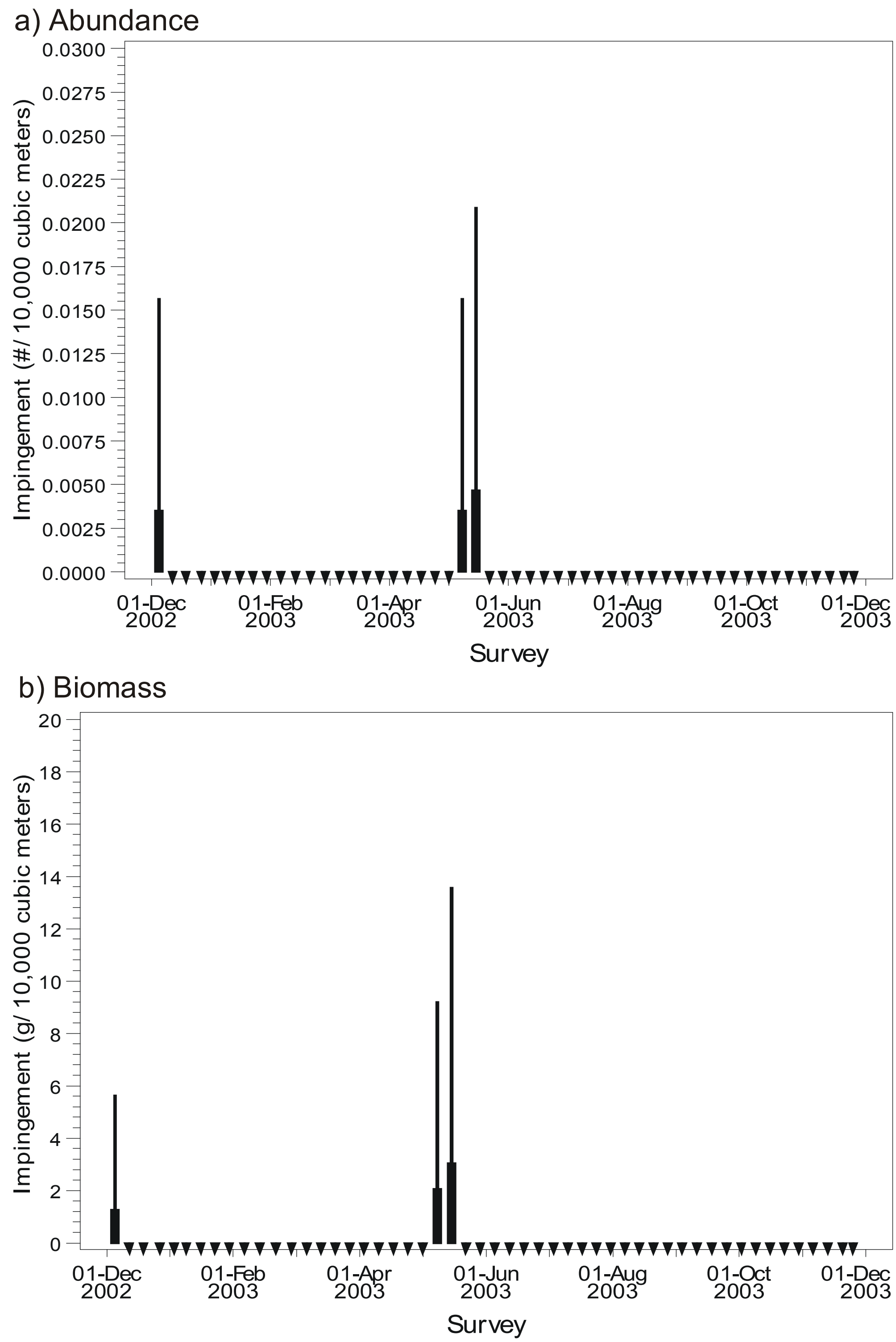


Figure 4.3-8. Mean concentration and standard error of sand basses *Paralabrax* spp. impinged at SBPP Units 1-4 December 2002 through November 2003 (n=52 surveys): a) abundance, and b) biomass. Triangles indicate surveys where none was present.

4.3.5 Shortfin corvina (*Cynoscion parvipinnis*)

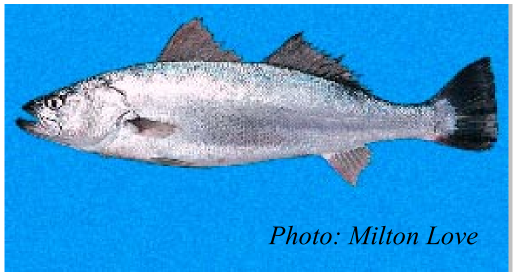
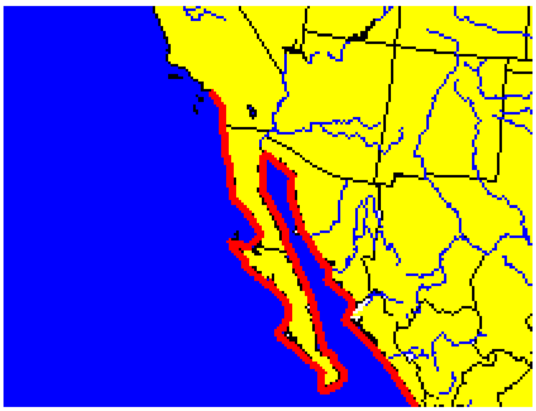


Photo: Milton Love



Distribution map for shortfin corvina.

Range: Huntington Beach, California to Mazatlan, Mexico, including the Gulf of California; uncommon north of Baja California, Mexico;

Life History: Size: to 60.0 cm (23.6 in) TL;
Fecundity: no specific information available;
Life span: no estimate available;

Habitat: Inshore sandy or muddy areas to depths of 50 m (164 ft);

Fishery: Commercially in Mexico; incidentally taken in California.

4.3.5.1 Life History and Fishery

The shortfin corvina *Cynoscion parvipinnis* is a member of the croaker family (Sciaenidae) that occurs in shallow sandy or muddy areas from Huntington Beach, California to Mazatlan, Mexico, including the Gulf of California (Eschmeyer et al. 1983). Allen (1999) refers to shortfin corvina as being a southern or “Panamic Province” species, generally being found further south in the eastern subtropical and tropical Pacific. He reasoned that the warmer waters in San Diego Bay provide a warm water refuge for this and other southern species.

Shortfin corvina has been reported to feed on octopus, squid, and small fishes and can grow to a maximum size of 60 cm (23.6 in) (Robertson and Allen 2002). As with other related members of the croaker family, eggs are pelagic and larvae develop planktonically. There is no specific information available on the age, growth, or reproductive capacity of shortfin corvina.

Shortfin corvina are present throughout San Diego Bay but are most abundant in the south ecoregion (Allen 1999). They are probably caught incidentally in the sport fishery but are too uncommon to yield significant landings. The species is sometimes mistaken for the orangemouth corvina *Cynoscion xanthulus* because they both have an orange mouth. However, the orangemouth corvina grows to a much larger size, and is distributed mainly in the Gulf of California and the Salton Sea where it is targeted as both a sport and commercial fishery species (Riedel et al. 2001).

4.3.5.2 Results

A total of 60 shortfin corvina, weighing 1.6 kg (3.5 lb), were impinged by the SBPP circulating water system during this study (**Table 4.3-1**). This species was the tenth most abundant fish impinged and ranked seventh in biomass. Impingement of this species was seasonal, with the highest abundance occurring from August to through October when the greatest numbers of juveniles were impinged (**Figure 4.3-9a**). Although the peak in abundance was recorded during fall, the greatest biomass was impinged during March and April (**Figure 4.3-9b**). The length range of the impinged corvina was 32–370 (1.3-14.6 in), with a mean length of 96 mm (3.8 in) (**Figure 4.3-10**).

4.3.5.3 Impact Assessment

Based on the impinged abundance and biomass of shortfin corvina presented above and the continual operation of all eight circulating water pumps, the estimated annual impingement of shortfin corvina was 428 individuals (±178.3 std. error), weighing 10.9 kg (24.1 lb) (±9.85 kg std. error).

Allen (1999) estimated the stock of shortfin corvina in the south and south central ecoregions (the area considered the source water for the entrainment portion of this report) at 201 and 508 kg, respectively. Based on a combination of these two estimates, the biomass of shortfin corvina impinged by SBPP represents about 1.5 percent of the estimated stock for this portion of San Diego Bay.

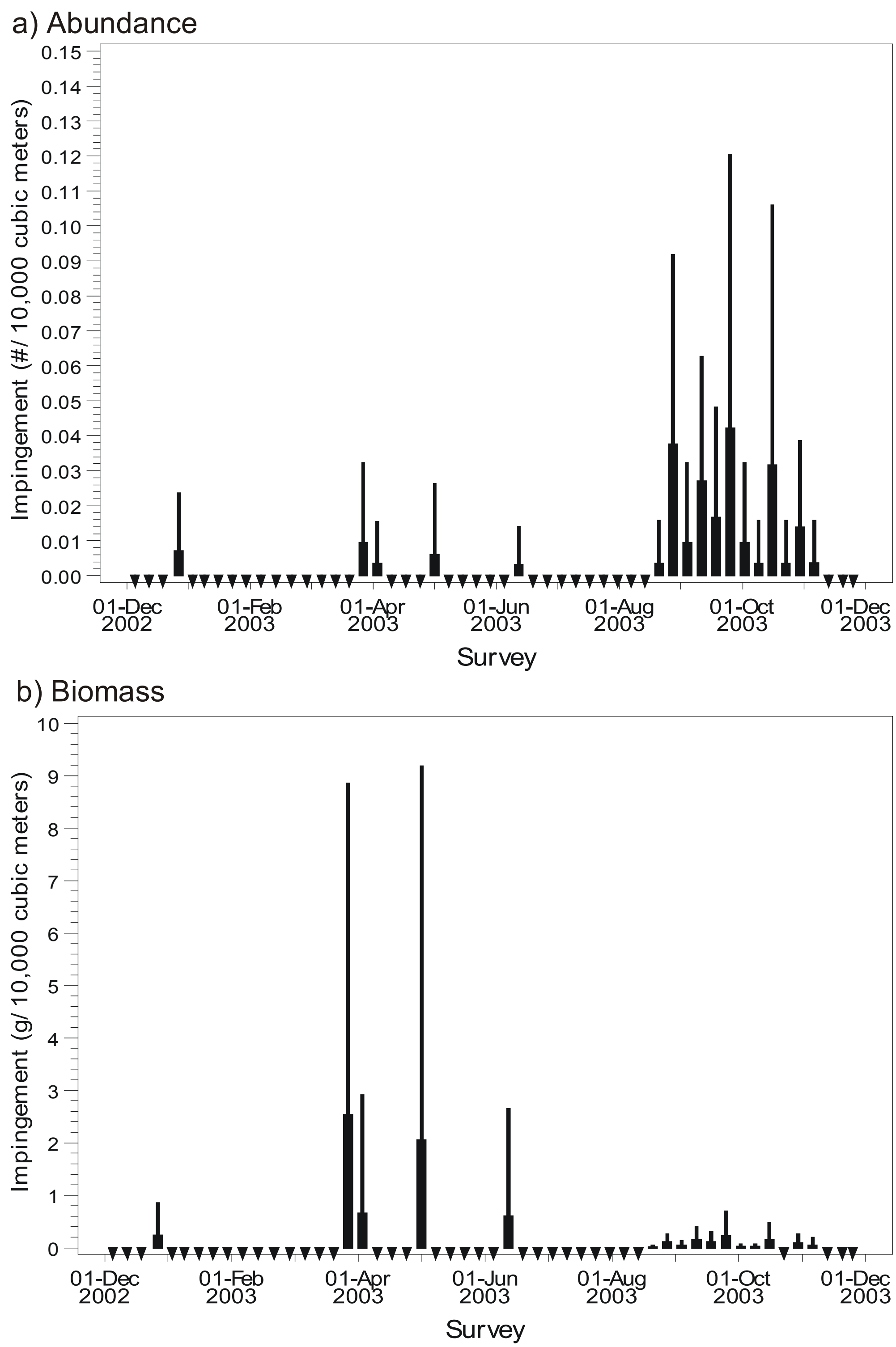


Figure 4.3-9. Mean concentration and standard error of shortfin corvina *Cynoscion parvipinnis* impinged at SBPP Units 1-4 December 2002 through November 2003 (n=52 surveys): a) abundance, and b) biomass. Triangles indicate surveys where none was present.

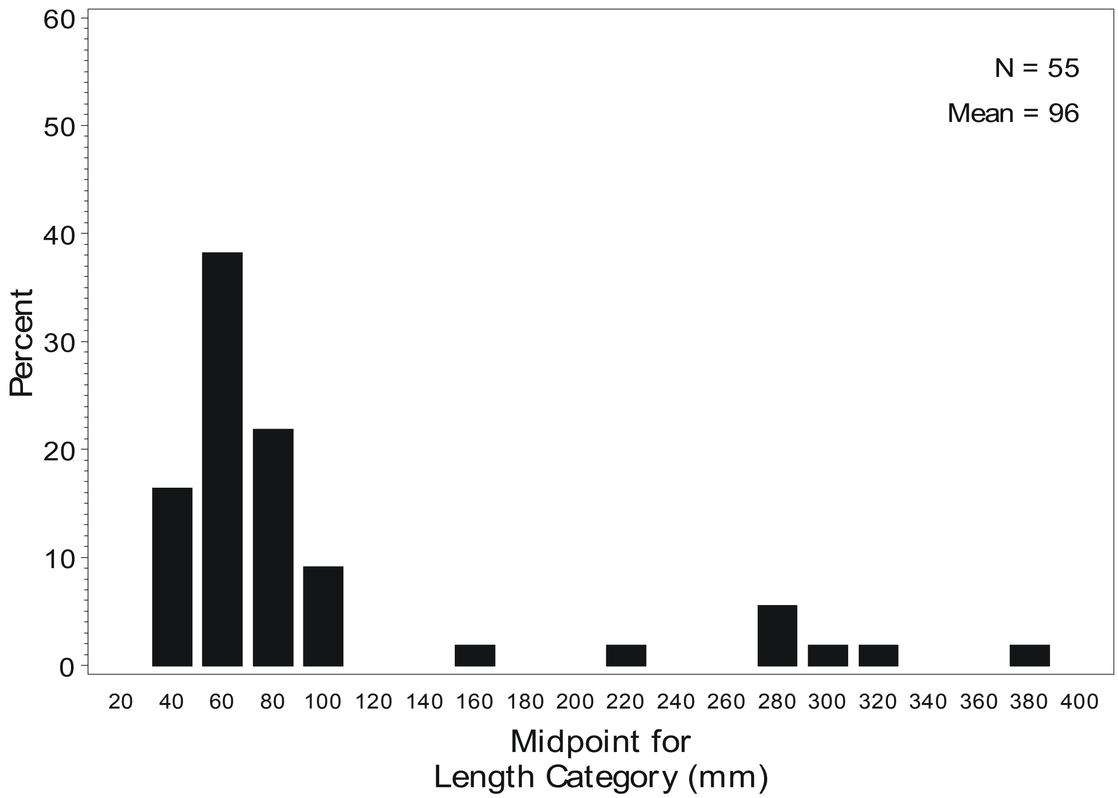


Figure 4.3-10. Size frequency distribution of shortfin corvina *Cynoscion parvipinnis* from SBPP Units 1-4 impingement samples.

4.4 Macroinvertebrate Impingement Results

4.4.1 Macroinvertebrate Community Overview

A wide variety of invertebrates were impinged on the traveling screens at SBPP. All invertebrates were removed from the impinged eelgrass and other debris, but only a subset of the taxa were enumerated. All of the others, including hydroids, sponges, jellyfish, bryozoans, mussels, snails, sea stars, worms, and tunicates, many of which were colonial life forms or non-swimming species that likely became dislodged from the intake structure itself, were only recorded as present. A total of 1,106 invertebrates with a total wet weight biomass of 3.08 kg (6.76 lb) and comprising 30 taxa were enumerated during this one-year study; an additional 50 taxa were noted as present but were not counted or weighed (**Table 4.4-1**). The top 90 percent most abundant taxa were tidepool shrimps *Heptacarpus* spp., pistol shrimps *Alpheus* spp., black-clawed crabs *Lophopanopeus* spp., Xantus' swimming crab *Portunus xantusii*, littoral pistol shrimp *Synalpheus lockingtoni*, brown shrimp *Penaeus californiensis*, market squid *Loligo opalescens*, yellow shore crab *Hemigrapsus oregonensis*, and striped shore crab *Pachygrapsus crassipes*. The invertebrates comprising the top 90 percent of the total impinged biomass were California spiny lobster *Panulirus interruptus*, brown shrimp, octopus *Octopus* spp., pistol shrimp, tidepool shrimp, Xantus' swimming crab, and littoral pistol shrimp. Although the abundance of impinged invertebrates was almost identical between Units 1 & 2 (49.5 percent) and Units 3 & 4 (50.5 percent) almost 78 percent of the invertebrate biomass was impinged on the Units 3 & 4 traveling screens (**Table 4.4-1** and **Appendix E**). This weight difference between the unit pairs was mainly due to a few larger spiny lobsters, brown shrimp, and octopus that were impinged on the Units 3 & 4 screens.

The estimated total annual abundance of all counted impinged invertebrates at SBPP was 9,019 individuals based on continuous flow of all eight circulating water pumps for an entire year (**Table 4.4-1**). The estimated annual impingement biomass of all invertebrates was 22.6 kg (49.84 lb).

Four invertebrate taxa were evaluated in detail for potential power plant effects on their local populations: brown shrimp, California spiny lobster, market squid, and octopus. As explained in Section 4.2.2 *Data Analysis*, these species were chosen because they comprised a significant fraction of the impinged macroinvertebrates by numbers or biomass, and have commercial or recreational fishery value.

Table 4.4-1. Summary of SBPP Units 1-4 macroinvertebrate impingement from December 2002 through November 2003 (52 surveys); '+' taxa present but not counted or weighed.

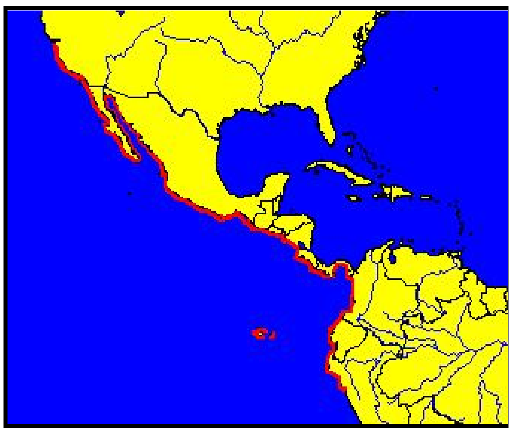
		Sampled Abundance			Estimated Annual Impingement			
Taxon	Common Name	(#)	(kg)	(lb)	(#)	Std. Err.	(kg)	Std. Err.
CRUSTACEA AND CEPHALOPODA (TARGET TAXA)								
Heptacarpus spp.	tidepool shrimps	408	0.32	0.72	3,205	582.8	2.46	0.55
Alpheus spp.	pistol shrimps	249	0.40	0.88	2,230	2,704.6	3.50	4.10
Lophopanopeus spp.	black-clawed crabs	134	0.10	0.23	1,238	448.3	0.75	0.58
Portunus xantusii	Xantus' swimming crab	77	0.16	0.36	562	173.3	1.16	0.54
Synalpheus lockingtoni	littoral pistol shrimp	42	0.11	0.23	317	247.8	0.83	0.72
Penaeus californiensis	brown shrimp	32	0.66	1.46	225	107.2	4.85	3.10
Loligo opalescens	market squid	26	0.01	0.02	208	131.8	0.07	0.07
Hemigrapsus oregonensis	yellow shore crab	17	0.03	0.06	121	124.8	0.20	0.32
Pachygrapsus crassipes	striped shore crab	14	0.01	0.03	109	74.1	0.09	0.08
Stomatopoda	mantis shrimps	13	0.01	0.03	93	62.9	0.09	0.08
Crangon spp.	bay shrimps	13	0.01	0.02	100	78.9	0.06	0.06
Pyromaia tuberculata	tuberculate pea crab	12	0.02	0.04	84	67.7	0.13	0.15
Crangon nigromaculata	spotted bay shrimp	11	0.01	0.01	80	67.2	0.04	0.04
Octopus spp.	octopuses	9	0.50	1.09	74	61.2	3.71	3.85
Hippolytidae	hippolytid shrimps	9	<0.01	0.01	67	63.8	0.03	0.05
Caridea	unidentified shrimps	9	<0.01	0.01	69	70.6	0.03	0.04
Palaemon macrodactylus	oriental shrimp	7	0.01	0.01	48	51.5	0.04	0.05
Loxorhynchus spp.	spider crabs	5	0.01	0.01	35	42.3	0.04	0.05
Majidae	spider crabs	4	<0.01	<0.01	38	43.5	0.02	0.03
Panulirus interruptus	California spiny lobster	2	0.70	1.54	13	22.6	4.41	8.00
Synalpheus spp.	pistol shrimps	2	<0.01	0.01	14	24.2	0.03	0.05
Hemigrapsus spp.	shore crabs	2	<0.01	<0.01	16	25.7	0.01	0.02
Pugettia spp.	kelp crabs	2	<0.01	<0.01	16	26.5	0.01	0.01
Neotrypaea spp.	ghost shrimps	1	<0.01	0.01	7	17.1	0.02	0.04
Uca crenulata	Mexican fiddler crab	1	<0.01	<0.01	7	17.1	0.01	0.04
Herbstia parvifrons	crevice spider crab	1	<0.01	<0.01	7	17.4	0.01	0.02
Brachyura	unidentified crabs	1	<0.01	<0.01	7	17.7	0.01	0.02
Xanthidae	unidentified mud crabs	1	<0.01	<0.01	8	18.8	0.01	0.01
Alpheidae	unidentified shrimps	1	<0.01	<0.01	13	33.0	<0.01	<0.01
Erileptus spinosus	spider crab	1	<0.01	<0.01	8	19.6	<0.01	<0.01
OTHER INVERTEBRATES								
Acmaeidae	limpets	+	-	-	-	-	-	-
Aglaophenia spp.	ostrich-plume hydroid	+	-	-	-	-	-	-
Amphipoda	amphipods	+	-	-	-	-	-	-
Anthozoa	anemones	+	-	-	-	-	-	-
Aurelia aurita	moon jelly	+	-	-	-	-	-	-
Balanus spp.	barnacles	+	-	-	-	-	-	-
Barnacles	unidentified barnacles	+	-	-	-	-	-	-
Bivalva	unidentified clams	+	-	-	-	-	-	-
Caprellidea	caprellid shrimps	+	-	-	-	-	-	-
Cerithidea californica	California hornsnail	+	-	-	-	-	-	-
Chione spp.	bivalves	+	-	-	-	-	-	-
Ciona intestinalis	simple tunicate	+	-	-	-	-	-	-
Crepidula spp.	slipper snails	+	-	-	-	-	-	-
Crucibulum spinosum	cup-and-saucer limpet	+	-	-	-	-	-	-
Encrusting bryozoa	encrusting bryozoa	+	-	-	-	-	-	-
Epiactis prolifera	brooding anemone	+	-	-	-	-	-	-
Foliose bryozoa	foliose bryozoa	+	-	-	-	-	-	-

(table continued)

Table 4.4-1 *(continued)*. Summary of SBPP invertebrate impingement from December 2002 through November 2003 (52 surveys); '+' taxa present but not counted or weighed.

		Sampled Abundance			Estimated Annual Impingement			
Taxon	Common Name	(#)	(kg)	(lb)	(#)	Std. Err.	(kg)	Std. Err.
Gammaridea	amphipods	+	-	-	-	-	-	-
Hermissenda crassicornis	horned nudibranch	+	-	-	-	-	-	-
Hydrozoa	unidentified hydroid	+	-	-	-	-	-	-
Isopoda	isopods	+	-	-	-	-	-	-
Macoma spp.	clams	+	-	-	-	-	-	-
Musculista senhousia	green mussel	+	-	-	-	-	-	-
Mysidacea	mysids	+	-	-	-	-	-	-
Mytilus galloprovincialis	Mediterranean mussel	+	-	-	-	-	-	-
Mytilus spp.	mussels	+	-	-	-	-	-	-
Nudibranchia	nudibranch	+	-	-	-	-	-	-
Obelia spp.	hydroid	+	-	-	-	-	-	-
Ophiothrix spp.	brittle stars	+	-	-	-	-	-	-
Ophiuroidea	brittle stars	+	-	-	-	-	-	-
Ostrea spp.	oyster	+	-	-	-	-	-	-
Pectinidae	scallops	+	-	-	-	-	-	-
Pelecypoda	unidentified bivalve	+	-	-	-	-	-	-
Platyhelminthes	flatworm	+	-	-	-	-	-	-
Polychaeta	segmented worms	+	-	-	-	-	-	-
Polynoidae	scale worms	+	-	-	-	-	-	-
Encrusting porifera	sponge	+	-	-	-	-	-	-
Pycnogonida	sea spiders	+	-	-	-	-	-	-
Pycnopodia helianthoides	juvenile sunflower star	+	-	-	-	-	-	-
Scyphozoa	sea jelly	+	-	-	-	-	-	-
Slipper snail	slipper snail	+	-	-	-	-	-	-
Solen rostraformis	rosy jackknife clam	+	-	-	-	-	-	-
Spirorbidae	polychaete worm	+	-	-	-	-	-	-
Styela montereyensis	stalked tunicate	+	-	-	-	-	-	-
Styela spp.	simple tunicates	+	-	-	-	-	-	-
Tetraclita rubescens	barnacle	+	-	-	-	-	-	-
Tubularia spp.	hydroid	+	-	-	-	-	-	-
Tunicata (simple)	simple tunicate	+	-	-	-	-	-	-
Tunicata (colonial/social)	colonial/social tunicates	+	-	-	-	-	-	-
Uca crenulata	Mexican fiddlercrab	+	-	-	-	-	-	-
	TOTAL	**1,106**	**3.08**	**6.79**	**9,019**		**22.60**	
% impingement total from Units 1-2	49.5	22.2						
% impingement total from Units 3-4	50.5	77.8						

4.4.2 Brown Shrimp (*Penaeus californiensis*)



Distribution map for brown shrimp.

Range: From San Francisco Bay, California to Sachura Bay, Peru and the Galapagos Islands;

Life History: Size: to 250 mm (9.8 in) total length;
Fecundity: up to 30,000 per spawn;
Life span: no estimate available;

Habitat: Over mud or sand bottoms in depths from 3–220 m (10–720 ft);

Fishery: Commercially in Mexico; incidentally taken in California.

4.4.2.1 Life History and Fishery

The brown shrimp *Penaeus californiensis* (also referred to by the common names of golden prawn, yellow-leg shrimp, or two-spot prawn) is found over mud or sand bottoms in depths of 3–220 m (10–720 ft) from San Francisco Bay, California to Sachura Bay, Peru and the Galapagos Islands (Jensen 1995, Rodriguez de la Cruz 1976 *in* de la Rosa-Vélez et al. 2000). Although its peak abundances are at approximately 55 m (180 ft), it is also associated with shallow mangrove estuarine habitats of coastal embayments.

The brown shrimp completes its entire life cycle in the marine environment. Spawning may occur throughout the year. Under laboratory conditions, females produce a wide range of clutch sizes (1,500–230,000 eggs) with a wide range of viability (3–78 percent) (Moore et al. 1974). Eggs are demersal and hatch within 12–15 hr after deposition (Rodriguez de la Cruz and Rosales 1970 *in* de la Rosa-Vélez et al. 2000). The planktonic larvae metamorphose through a series of 11 larval stages before reaching the semi-benthic postlarval stage. Once the rostrum is fully developed, 12–17 days after hatching, the animal is considered a juvenile.

Postlarvae prefer low salinity water and tend to immigrate from the sea to nursery grounds in estuaries, bays, and coastal lagoons. Older and larger postlarvae prefer lower salinities and can adapt to lower salinities more quickly than younger individuals (Mair 1980). Laboratory experiments indicate that higher temperatures (28–32°C) and salinities (>36 ppt) increase the shrimp's susceptibility to disease (Vargas-Albores et al. 1998). Adults are probably omnivorous and feed at night, as do other penaeid shrimps, and they are probably preyed upon in turn by large fishes (Divita et al. 1983). Brown shrimp became more abundant in the San Diego area after the 1997 El Nino Southern Oscillation (ENSO) event (SCAMIT 1998).

This species is important in the commercial "jumbo" shrimp fishery in Mexico and it is also taken in California fisheries but typically as incidental catch and not as a targeted species. There were no landings records specifically listed for *P. californiensis* in the PacFIN database from 1981–2002 (PacFIN 2003) or the CDFG landings data for 2002 (CDFG 2003). In the Mexican fishery, the species is taken in bottom trawling nets throughout the Gulf of California along with a similar species of shrimp, *P. stylirostris*. In the northern Gulf of California, shrimp landings averaged approximately 500 MT from 1980–1990, and then declined to approximately 200 MT annually through 1997 (Galindo-Bect et al. 2000).

4.4.2.2 Results

A total of 32 brown shrimp weighting 0.66 kg (1.5 lb) were collected from the impingement samples (**Table 4.3-1**). They were the sixth most abundant invertebrate taxon impinged and had the second highest biomass. Brown shrimp were impinged at low levels from December through mid-July. None were impinged from mid-July through November (**Figure 4.4-1**). Lengths of brown shrimp ranged from 11–210 mm (0.4–8.3 in) TL, with a mean length of 88.3 mm (3.5 in) TL. The individual weights ranged from 0.2–83.1 g (<0.01–0.2 lb) with a mean weight of 20.7 g (0.05 lb).

4.4.2.3 Impact assessment

Based on the impinged abundance and biomass of brown shrimp presented above and the continual operation of all eight circulating water pumps, the estimated annual impingement abundance of this group was 225 individuals ($\pm$107.2 std. error) weighing 4.9 kg (10.7 lb) ($\pm$3.1 kg std. error) (**Table 4.4-1**). No information is available as to the quantity of this species caught in the fishery or its abundance in San Diego Bay.

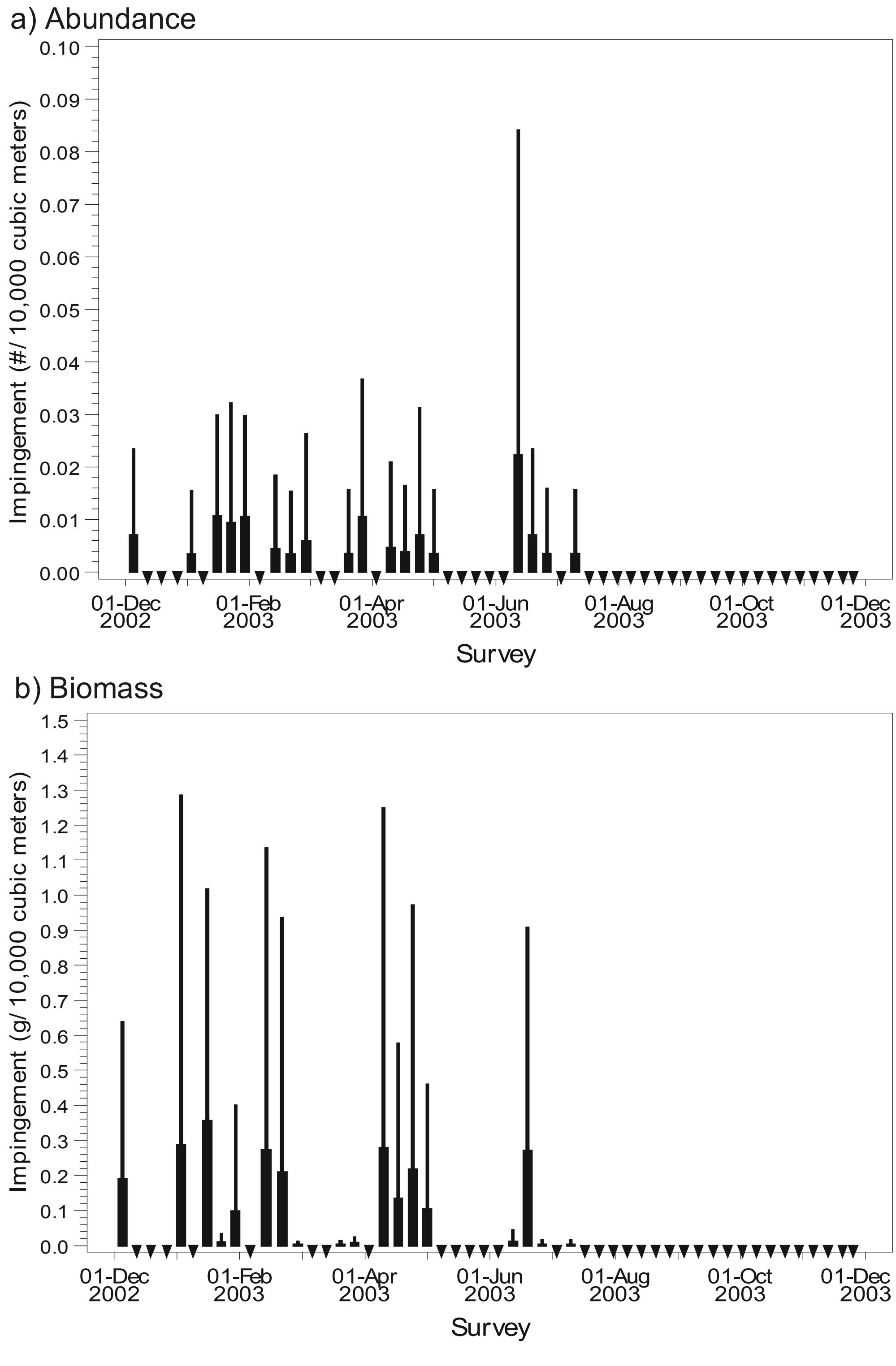
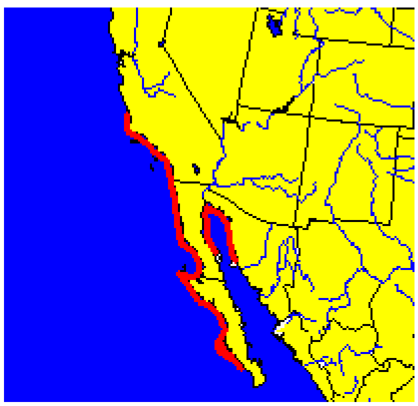


Figure 4.4-1. Mean concentration and standard error of brown shrimp *Penaeus californicus* impinged at SBPP Units 1-4 December 2002 through November 2003 (n=52 surveys): a) abundance, and b) biomass. Triangles indicate surveys where none was present.

4.4.3 California Spiny Lobster (*Panulirus interruptus*)



Distribution map of California spiny lobster.

Range: From Monterey Bay, California to southern Baja California and northern Gulf of California, Mexico;

Life History: Size: to 75 cm (2.5 ft) total length;
Fecundity: 50,000–800,000 eggs;
Life span: 20–30 years;

Habitat: Nearshore surfgrass beds and rocky habitat in depths from intertidal to 75 m (0–245 ft);

Fishery: Commercial and recreational fishery throughout range.

4.4.3.1 Life History and Fishery

The California spiny lobster *Panulirus interruptus* inhabits coastal waters of the Pacific Southwest from Monterey Bay, California, to southern Baja California, Mexico. The majority of the population is centered between Point Conception and central Baja California (Lindberg 1955, Johnson 1960). There is an isolated population in the northern waters of the Gulf of California (Duffy 1973). Adult lobsters usually inhabit rocky areas from the intertidal zone to depths of 75 m (245 ft). Lobsters make an annual offshore-onshore migration stimulated by water temperature and an increase in wave action. In winter months, male and female lobsters are found in depths of 15 m (50 ft) or greater. Mating occurs in December through March while the lobsters are offshore. Starting in late March through May they move onshore into depths of less than 9 m (30 ft). They generally migrate in small groups after dark.

Spawning occurs from March through August with primary activity during May, June, and July (Allen 1916). Females move inshore and release 50,000–800,000 eggs (Shaw 1986). The extruded eggs are fertilized by sperm released from a tar-like spermatophore deposited by the male on the under side of the female's sternum. The fertilized eggs then attach to the lobster's pleopods. There the eggs develop for 9–10 wk before hatching.

The larval development of spiny lobster is protracted and complex compared to other crustaceans. The larvae undergo 11 pelagic stages (Johnson 1956). The first larval stages or phyllosomes have transparent, dorsoventrally flattened bodies and long spider-like legs. The average body length is 1.4 mm (0.06 in) for stage I phyllosomes and 29 mm (1.1 in) for stage IV phyllosomes. Only 3 percent of larvae survive to reach stage IV. During the larval period, the phyllosomes drift with the prevailing currents feeding on other planktonic organisms. After 5–9 months, the phyllosome larvae metamorphose into stage XI, the puerulus stage. Here the animal resembles the adult form, although the body is still transparent and the second antennae are three times the length of the body (Johnson 1956). The puerulus actively swims inshore where it settles to the bottom if the habitat is suitable. The larvae are commonly found in surf grass, *Phyllospadix torreyi.* The puerulus stage lasts approximately 60–90 d. Ten days after settling, the puerulus become fully pigmented and begins life as a benthic juvenile. Most juvenile lobsters spend their first two years in nearshore surf grass beds, mussel beds, or shallow rocky crevices.

Approximately 90 percent of females are sexually mature at 69 mm (2.7 in) carapace length (CL) (Shaw 1986). Males mature at 3–6 yr and females mature at 5–9 yr. Growth rates are highly variable depending on food resources, water temperature, size, and sex of the animal. Males tend to grow faster and live longer than females. Males reach the minimum legal harvest CL of 83 mm (3.3 in) in 7–10 yr and females after 12 yr. Lobsters shelter in crevices or holes during daylight hours to avoid a variety of predators including sheephead, cabezon, kelp bass, octopus, California moray eel, giant sea bass, rockfishes, leopard shark, and horn shark. At night lobsters leave the safety of the den to search for food. Being omnivores, they consume algae and a wide variety of invertebrates such as snails, mussels, sea urchins, clams, and fishes, as well as injured or newly molted lobsters.

There has been a commercial fishery in southern California for spiny lobster since the 1800s. Fishermen use weighted wire mesh boxes or "traps" baited with fish or crushed mussels to attract the lobsters. The traps are usually clustered around rocky outcrops or along depth contours of less than 30 m (100 ft). Seasonal landings have varied over the years from a peak in 1949–1950 of 476,000 kg (1.05 million lb) to a low in 1974–1975 of 69,000 kg (152,000 lb) (Shaw 1986). From 1979–1998 landings have ranged from 181,000–431,000 kg (400,000–950,000 lb). San Diego County is located in the central portion of the spiny lobster range where up to 60 percent of California landings occur.

The average landings for San Diego County in 1999–2002 were 95,868 kg (211,353 lb). Annual revenue generated by lobster landings in San Diego County from 1981–2003 averaged $1,226,000 (PacFIN 2003). There is also a substantial sport fishery. Lobsters are taken by skin divers and scuba divers, as well as with hoop nets. Although there are little data, it is estimated that annual sport take is equal to half of the commercial catch (Frey 1971). Fluctuations in landings can be due to factors other than population such as weather events like El Nino or La Nina. California Department of Fish and Game tracks the number of sub-legal lobsters per number of traps fished. Based on the proportion of short and legal lobsters taken, it is believed that the lobster population in California is well managed and in a healthy status.

4.4.3.2 Results

A total of only 2 spiny lobsters, with a combined weight of 0.7 kg (1.5 lb), were impinged during the entire one-year study (**Table 4.4-1, Figure 4.4-2**). Their lengths were 81 and 178 mm (3.2 and 7.0 in) TL **(Appendix D)**. Because of their large size relative to other impinged invertebrates, this species had the highest impingement biomass.

4.4.3.3 Impact Assessment

Based on the impinged abundance and biomass of California spiny lobster and the continual operation of all eight circulating water pumps, the estimated annual impingement abundance of this group was 13 individuals (±22.6 std. error) weighing 4.41 kg (±8.0 kg std. error) (9.71 lb) (**Table 4.4-1**).

In 2002 there were 191,390 lb of spiny lobster landed by the commercial fishery in the San Diego area (CDFG 2003). The value of these lobster was reported as $1,240,940, which equals about $6.50/lb. Based on an average size spiny lobster in the commercial fishery of 3 lb, the estimated adult equivalent biomass value of impinged lobsters is approximately $250 annually.

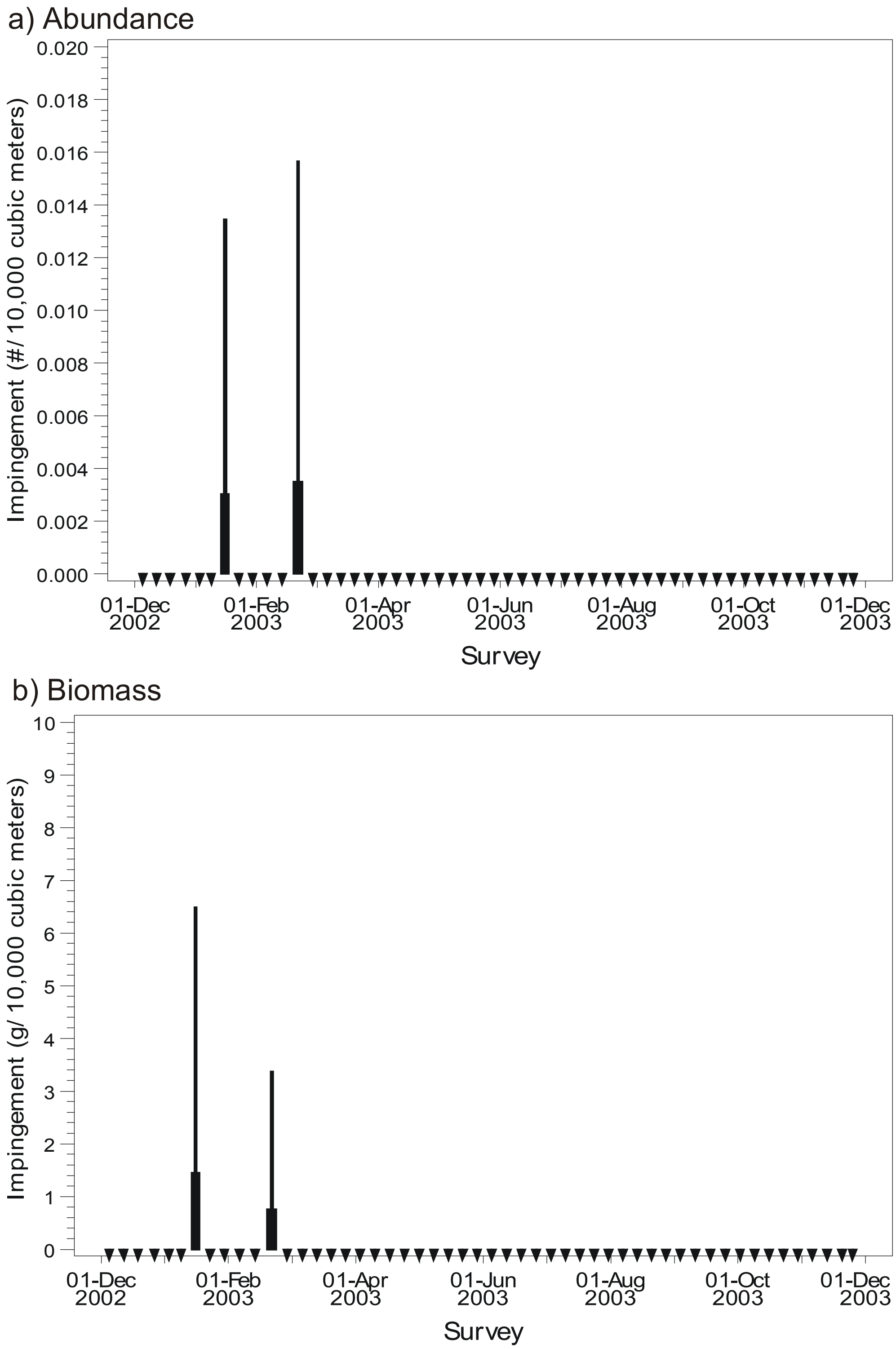
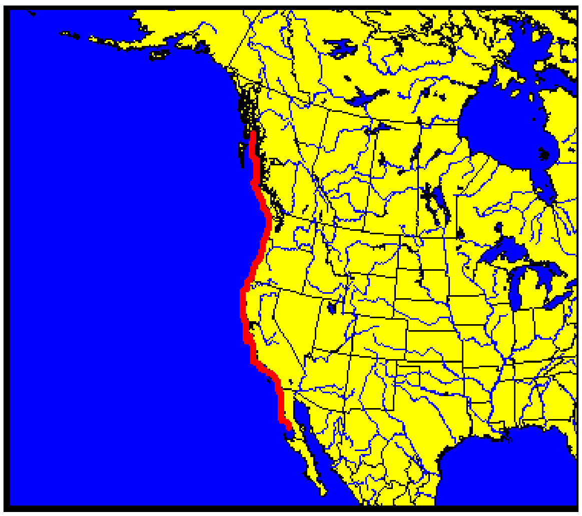


Figure 4.4-2. Mean concentration and standard error of California spiny lobster *Panulirus interruptus* impinged at SBPP Units 1-4 December 2002 through November 2003 (n=52 surveys): a) abundance, and b) biomass. Triangles indicate surveys where none was present.

4.4.4 Market Squid (*Loligo opalescens*)



Distribution map for market squid.

Range: From southern Alaska to Isla Guadalupe, Mexico;

Life History: Size: Males to 275 mm (11 in) (not including tentacles) and females to approximately 200 mm (8 in);
Size at maturity: dorsal mantle lengths as small as 70–80 mm (2.8 to 3.1 in);
Fecundity: 180–300 eggs encased in a capsule, may extrude 20–30 capsules;
Life span: <1 yr;

Habitat: Pelagic, living in coastal waters but returning to shallow inshore waters to spawn;

Fishery: Commercial, marketed for human consumption or sold as bait.

4.4.4.1 Life History and Fishery

The market squid is a member of the family Lolinginidae in the order Decapoda that also contains octopus. Market squid range from southern Alaska to Isla Guadalupe, Mexico, and Bahía Asuncíon, Baja California (Morris et al. 1980), but are most common from British Columbia southward (Morris et al. 1980). They are pelagic, living in coastal waters and moving to semi-sheltered bays and other locations with suitable substrata (sand or mud bottoms) to spawn in depths ranging from just below the intertidal down to 180 m (540 ft) (Fields 1965, Kato and Hardwick 1975).

Male market squid reach 275 mm (11 in) dorsal mantle length (DML), and females attain 200 mm (8 in) DML (UCLA 1999). Water temperature was found to be positively correlated with female growth between northern and southern portions of the southern California bight but males were unaffected by temperature differences (McDaniel et al. 2003). Male and female market squid reach maturity at around 70–80 mm (ca. 3 in) DML in as little as six months (Butler et al. 1999, FWIE 1999). At 15 mm (0.6 in) DML, squid are reported to be approximately 50 days old. Recent age estimates indicate that the market squid may complete their life cycle in less than one year (Butler et al. 1999).

Market squid spawn year-round from San Francisco to Baja California, but exhibit two spawning peaks annually (Starr et al. 1998). Spawning activity begins in the southern California population in December and continues through March. In Monterey Bay, they begin spawning in April and continue through November (McInnis and Broenkow 1978, Morris et al. 1980). Both male and female squid are terminal spawners and die after spawning.

The female produces from 180–300 eggs encased in a cylindrical capsule and may extrude 20–30 capsules during a spawning event (Starr et al. 1998, FWIE 1999). Recent research on market squid reproduction corroborates reports by Starr et al. (1998) and FWIE (1999) that estimated around 5,500 eggs per spawning female (Macewicz et al. 2000). Egg cases are attached with thin stalks to the bottom substratum (Fields 1965). Subsequent layers are then deposited until large clusters are formed (Starr et al. 1998). Egg cases have been observed in depths ranging from 3–180 m (10–590 ft) (FWIE 1999) and the eggs hatch in 15–90 d, depending on water temperature (Fields 1965, Yang et al. 1986).

Approximately 90 percent of the seasonal harvest of market squid in California occurs in the southern California bight. Large fluctuations in annual landings are thought to be correlated with changes in ocean climate that affect market squid reproduction and survival.

The commercial landings of market squid, in the San Diego Area, have fluctuated greatly from 1981 through 2001, with an average reported landing of 4.1 metric tons (MT) (PacFIN 2003). The largest reported landing of 18.8 MT occurred in 1988, while the minimum reported landing of 0.6 MT occurred in 1994. No commercial landings of market squid were reported for 2002 (CDFG 2003, PacFIN 2003).

4.4.4.2 Results

A total of 26 market squid weighting 0.01 kg (0.02 lb) were collected from the impingement samples (**Table 4.4-1**). They were ranked as the seventh most abundant invertebrate impinged. They were only impinged from December through early March (**Figure 4.4-3**). All individuals were very small ranging in length from 12–24 mm (0.5–0.9 in) in overall length (**Appendix D**).

4.4.4.3 Impact Assessment

Based on the impinged abundance and biomass of market squid and the continual operation of all eight circulating water pumps, the estimated annual impingement abundance of this group was 208 individuals (±132.0 std. error) weighing 0.07 kg (0.15 lb) (±0.07 kg std. error) (**Table 4.4-1**).

In 2002 there were 1,585 pounds of 'jumbo squid' landed by the commercial fishery in the Mission Bay area of San Diego (PacFIN 2002). The value of these squid was reported as $882, which equals about $0.56/lb. Based on weight of 0.5 lb for an average size squid, the value of the adult equivalent squid biomass impinged at SBPP is less than $60 annually.

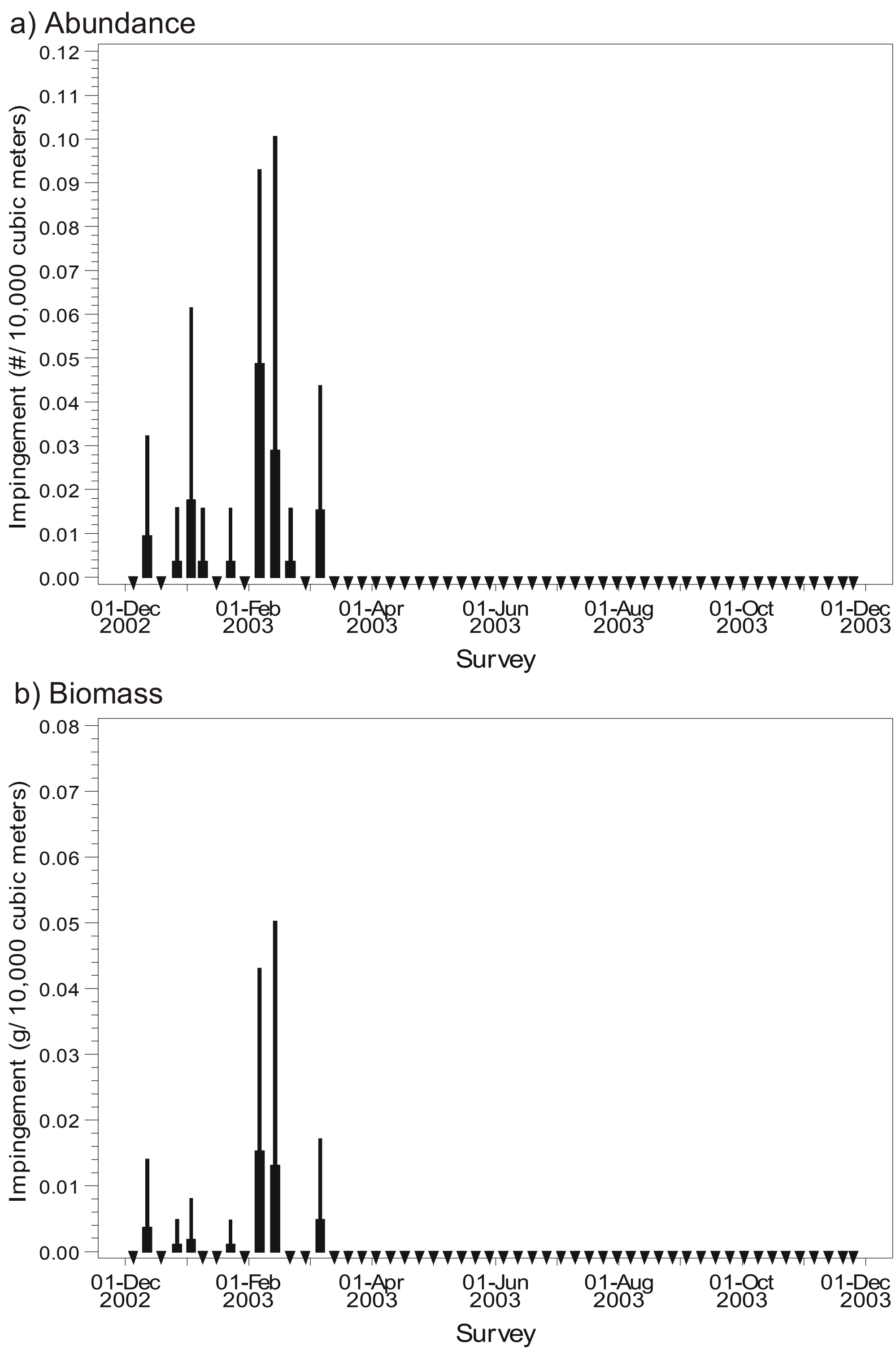
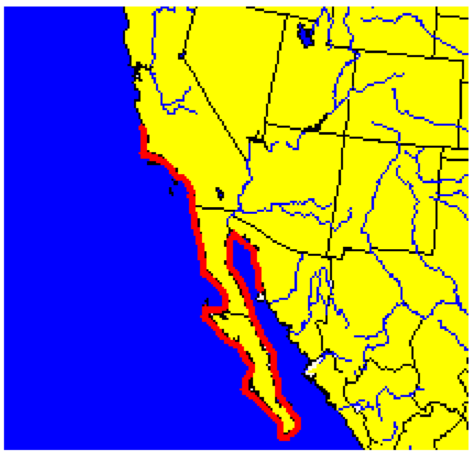


Figure 4.4-3. Mean concentration and standard error of market squid *Loligo opalescens* impinged at SBPP Units 1-4 December 2002 through November 2003 (n=52 surveys): a) abundance, and b) biomass. Triangles indicate surveys where none was present.

4.4.5 Two-spotted Octopus (*Octopus* spp.)



Distribution map for *Octopus* spp.

Range: *O. bimaculoides*: San Simeon (San Luis Obispo Co.) to Ensenada, Baja California; *O. bimaculatus*: Santa Barbara to Gulf of California;

Life History. Size: Dorsal mantle length from 5–20 cm (2.0– 7.9 in) at maturity;
Fecundity: varies with species and size;
Life span: varies with species; approximately 0.5–3 years;

Habitat: *O. bimaculoides* found from the middle and low intertidal zones and mud flats to the subtidal, on rocks or in kelp beds, to depths of 20 m; *O. bimaculatus* from the lower intertidal zone to 50 m.

Fishery: Commercial and recreational.

4.4.5.1 Life History and Fishery

The two-spotted octopus group consists of two similar species: *Octopus bimaculoides* and *O. bimaculatus*. *Octopus bimaculoides* occurs from San Simeon (San Luis Obispo Co.) to Ensenada, Baja California, and *O. bimaculatus* has a more southerly distribution extending into the Gulf of California (Morris et al. 1980). They occur from the middle intertidal zone to depths of 20–50 m (66–164 ft) in kelp beds, rock, or mud substrates. They can also shelter in large gastropod shells or discarded bottles and cans.

Morris et al. (1980) summarized the life history of *O. bimaculoides*. Two-spotted octopuses begin laying eggs primarily from January through May. Females lay their eggs under rocks from late winter to early summer, and brood them continuously from 2–4 mo until hatching. MacGinitie and MacGinitie (1968) report that female *O. bimaculoides* weighing approximately 0.5 lb will lay approximately 600 eggs. The eggs are attached by slender stalks, are about 0.5 in. long and 1/6 inch in diameter. The young remain on the bottom after hatching, often moving into the intertidal.

Adults feed on a variety of fishes, mollusks, and crustaceans (MacGinitie and MacGinitie 1968). In the rocky intertidal zone *O. bimaculoides* drills and feeds principally on limpets (Morris et al. 1980).

O. bimaculatus occupies holes and crevices in a wide range of hard substrate habitats (Ambrose 1988). Females lay their eggs under rocks from late winter to early summer, and brood them continuously 2–4 mo. At Santa Catalina Island, with an average octopus of 260 g (0.6 lb) (71 mm [2.8 in] mantle length [ML]), the average clutch size is approximately 20,000 eggs (Ambrose 1981). The young remain on the bottom after hatching, often moving into the intertidal zone (Morris et al. 1980).

4.4.5.2 Results

A total of 9 octopuses weighing 0.5 kg (1.1 lb) was collected from impingement samples (**Table 4.4-1**). They were the 14^{th} most abundant invertebrate impinged and third in biomass. They were impinged from February through August (**Figure 4.4-4**).

4.4.5.3 Impact assessment

Based on the impinged abundance and biomass of octopus and the continual operation of all eight circulating water pumps, the estimated annual impingement abundance of this group was 74 individuals (±61.2 std. error) weighing 3.71 kg (9.18 lb) (±3.85 kg std. error) (**Table 4.4-1**).

In 2002 there were 174 pounds of 'unspecified octopus' landed by the commercial fishery in the Mission Bay area of San Diego (PacFIN 2002). The value of these octopus was reported as $96, which equals about $0.55/lb. Based on an average weight of 2.0 lb, the value of octopus impinged annually at SBPP is less than $100.

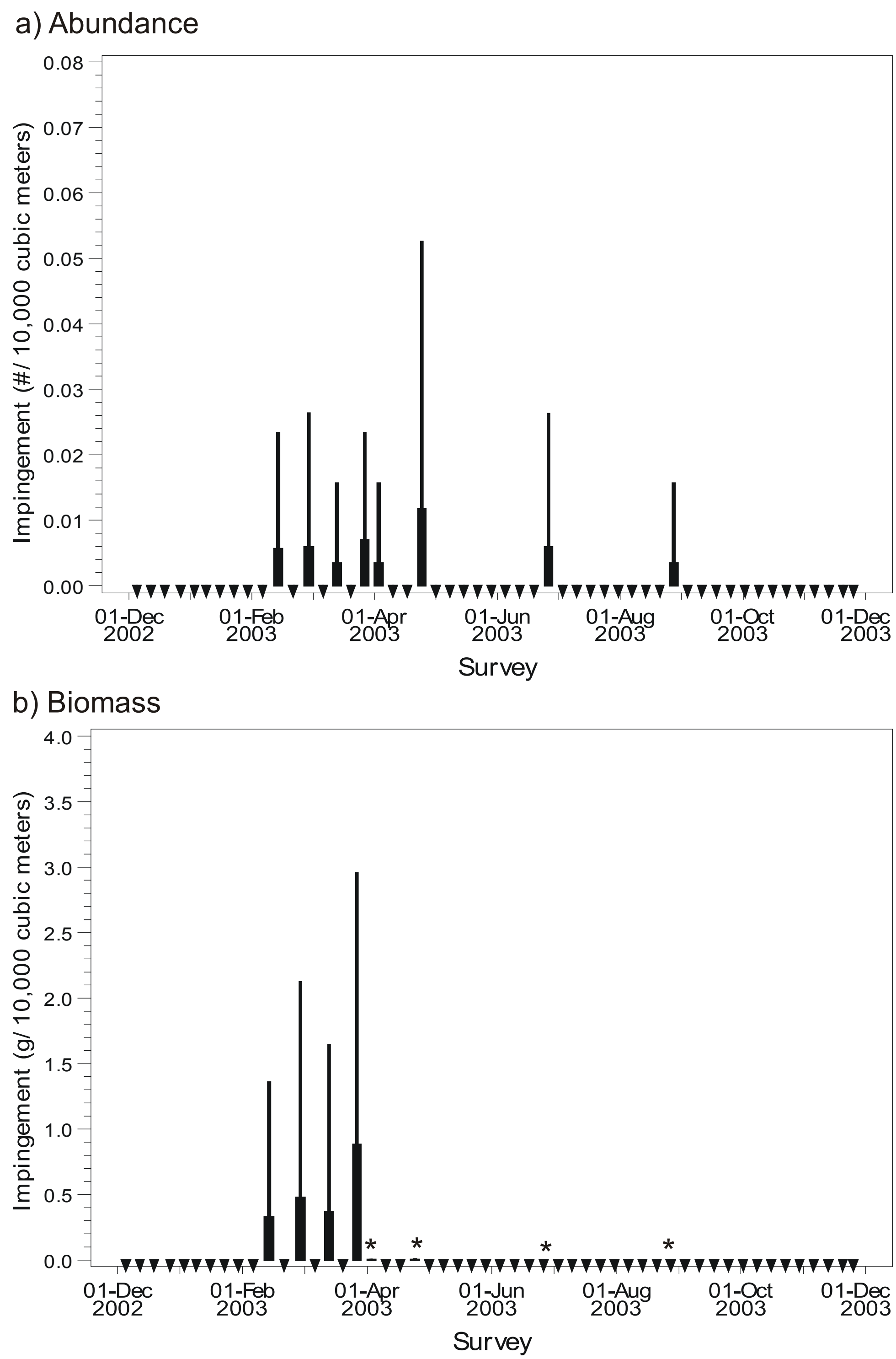


Figure 4.4-4. Mean concentration and standard error of two-spotted octopus *Octopus* spp. impinged at SBPP Units 1-4 December 2002 through November 2003 (n=52 surveys): a) abundance, and b) biomass. Triangles indicate surveys where none was present. * Biomass not measured during survey.

5.0 Cooling Water Assessment Summary

5.1 Overview of Assessment

The purpose of the SBPP entrainment and impingement studies was to evaluate the potential effects of the cooling water intake system as required under Section 316(b) of the Federal Clean Water Act (CWA) (USEPA 1977). As part of this evaluation, an earlier 316(b) study conducted in 1979 (SDG&E 1980) was updated and information from the 2001 and 2003 entrainment and impingement studies will be used by the San Diego Regional Water Quality Control Board in support of the NPDES permitting process for SBPP. Data on larval fishes, megalopal crabs, and larval spiny lobster collected near the SBPP intakes were used to estimate entrainment losses, while impingement losses were based on direct measurements of the abundance and biomass of fishes and selected macroinvertebrates retained on the SBPP intake screens.

Our ability to evaluate CWIS effects was limited to the fishes and invertebrates that were in high abundances in entrainment or impingement samples. The abundances of the majority of the entrained and impinged species were low and would not result in any risk of population-level effects. In addition, there is a great deal of uncertainty that the assessments for species that are in low abundance and collected infrequently are representative of actual CWIS effects. However, by focusing on the most abundant species, we were able to estimate the magnitude of effects on the component species in the biological community. After evaluating the results of entrainment and impingement only two groups of fishes—anchovies and silversides—were found to be abundant enough to be affected by both entrainment and impingement. Based on the data collected in our studies, it was determined that the collective entrainment and impingement losses would have some small but undetectable effect on biological community functioning.

The life history of component species in the community must be considered when discussing potential effect to the populations. Although the study focused on species potentially affected by entrainment and impingement processes, it is important to note that several fish species in south San Diego Bay have early life stages that are not susceptible to these processes. Live-bearers, such as surfperches, some sharks, and some rays, produce young that are fully developed and too large to be affected by entrainment. Live-bearers together comprise nearly 40 percent of the fish biomass in the bay (Allen 1999). Another common species in south San Diego Bay, striped mullet, also is not susceptible to entrainment because it spawns offshore and only the juveniles and adults subsequently utilize the bay habitat. From the standpoint of impingement effects, one of the most abundant groups of species in the bay, gobiid fishes, are generally not susceptible to impingement after transformation to the juvenile life stage because they are bottom-dwelling species that typically do not move up into the water column. Even fish species that swim in the water column are generally not susceptible to impingement effects as they mature because they are able to swim against the slow approach velocity of the cooling water inflow. For example, at the SBPP intakes it was not uncommon to see small schools of adult striped mullet swimming directly in front of the intakes and not being impinged during times when circulating water pumps were operating.

Overall, the 316(b) assessment relied on a synthesis of results from modeling the effects of larvae removed from the system through entrainment and juveniles and adults removed from the system through impingement. In both cases, estimated losses were calculated using the following set of conservative assumptions that would result in the greatest projected effects on a target species:

- all entrainment and impingement loss estimates were calculated based on maximum design cooling water flows, although actual cooling water withdrawals were only a small fraction of the maximum due to variable demand for power generation throughout the year;
- entrainment modeling assumed no survival of larvae through the cooling water system;
- no density-dependent compensatory effects were included in the models that would result in increased survivorship for later life-stages not subject to CWIS effects; and
- estimated economic losses of impingement fishery species were scaled up to assume that all impinged individuals represented fishes of adult size potentially lost to the fishery, without applying projected mortality rates to the impinged juveniles.

Overall, our conclusions are consistent with those from the earlier 316(b) study done in 1979–1980 (SDG&E 1980) that the operation of SBPP does not substantially affect populations of the most abundant or economically important fishes and invertebrates in San Diego Bay. Studies by Allen (1999) found that slough anchovy comprised over half of the fishes by number in the south-central and south ecoregions of San Diego Bay. Results from the present study show that SBPP may account for a loss of approximately 8–10 percent of the larval population annually and represent an equivalent loss of approximately 1–2 percent of the adult standing stock. Another major group of fishes in the bay affected by entrainment was the CIQ goby complex, with larval losses estimated at 21–27 percent of the source water population. Under the most conservative assumptions, the SBPP CWIS may account for losses from 1.2 to 2.2 million adult CIQ gobies per year out of an estimated standing stock of over 10 million. For the invertebrate species investigated, there were no substantial direct effects of the CWIS on their populations. Particularly for species with commercial fishery importance, such as lobsters, crabs, and squid, the results indicate that SBPP would not affect the adult populations of these species.

5.2 Summary of Entrainment Results

Results from entrainment sampling were used to identify the most abundant taxa that were subsequently evaluated for entrainment effects. Three independent models were used in assessing entrainment losses. Two of the models, Fecundity Hindcasting (*FH*) and Adult Equivalent Loss (*AEL*), used species life history information to estimate the potential numbers of adults represented by the entrainment losses. The third approach, the Empirical Transport Model (*ETM*), compared entrainment larval densities to source water larval densities to calculate the effects of larval removal on the standing stock of larvae in south San Diego Bay. Results from the three models are summarized in **Table 5.2-1** and discussed in the following sections.

Table 5.2-1. Summary of estimated SBPP entrainment effects on target taxa for 2001 and 2003 periods based on *FH*, *AEL*, and *ETM* (P_m) models. The *FH* estimate is multiplied by 2 to test the relationship that 2•*FH = AEL*.

	Estimated Annual Entrainment		2·*FH*		*AEL*		P_m	
Taxon	**2001**	**2003**	**2001**	**2003**	**2001**	**2003**	**2001**	**2003**
CIQ goby complex	1.8×10^9	1.4×10^9	2,169,562	1,652,186	1,579,926	1,203,161	0.215	0.267
longjaw mudsucker	2.2×10^7	2.5×10^7	2,956	3,372	*	*	0.171	0.502
anchovies	5.1×10^8	1.1×10^8	213,752	44,498	*	*	0.105	0.079
silversides	1.5×10^7	9.8×10^6	*	*	*	*	0.146	0.149
combtooth blennies	2.2×10^7	2.4×10^7	21,514	22,748	*	*	0.031	0.034

* Information unavailable to compute model estimate.

The list of target taxa included fishes comprising the top 99 percent of the total abundance in either sampling year, and commercially or recreationally important fishery species. Totals of 23,039 and 7,589 larval fishes were collected from the SBPP entrainment station (SB1) during the 2001 and 2003 sampling periods, respectively (**Tables 3.3-1** and **3.3-2**). Total annual entrainment was estimated to be 2.42×10^9 and 1.57×10^9 for the 2001 and 2003 periods, respectively (**Tables 3.3-1** and **3.3-2**). The CIQ goby complex comprised the largest percentage of the total estimated entrainment for both sampling periods, 76 and 89 percent, respectively (**Figures 3.3-1** and **3.3-2**). Together, the CIQ goby and the *Anchoa* spp. complexes comprised greater than 95 percent of the total estimated entrainment for both sampling periods. Other abundant taxa included combtooth blennies, longjaw mudsucker, and silversides. California halibut and other commercial or recreational fishery species comprised less than 0.1 percent of the total estimated entrainment during both sampling periods (**Table 5.2-2**).

Table 5.2-2. Summary of entrainment estimates comparing commercially or recreationally important fishery taxa with non-use taxa collected at entrainment station SB1 for the 2001 and 2003 periods.

Taxon	Common Name	Total Larvae Collected	Percent Composition	Estimated Annual Larval Entrainment
a) 2001 Period				
Fishery Taxa				
Sciaenidae	croakers	6	0.03	706,220
Genyonemus lineatus	white croaker	3	0.01	340,216
Hypsopsetta guttulata	diamond turbot	3	0.01	277,819
Engraulis mordax	northern anchovy	3	0.01	269,386
Cheilotrema saturnum	black croaker	1	0.01	137,775
Paralichthys californicus	California halibut	1	0.01	89,571
Subtotal Fishery Taxa		17	0.08	1,820,988
Subtotal Non-Use Taxa		23,022	99.92	2,418,706,791
	2001 Totals	**23,039**	**100.00**	**2,420,527,779**
b) 2003 Period				
Fishery Taxa				
Sciaenidae	croakers	4	0.05	855,939
Engraulis mordax	northern anchovy	1	0.01	203,692
Subtotal Fishery Taxa		5	0.06	1,059,631
Subtotal Non-Use Taxa		7,584	99.94	1,565,643,019
	2003 Totals	**7,589**	**100.00**	**1,566,702,650**

No commercially or recreationally important fishes or target invertebrate taxa were evaluated in detail for entrainment effects because of their low abundances in entrainment samples. No endangered or threatened fish or invertebrate species were collected at entrainment or source water stations during either study period.

The available life history information for the smaller forage species that were the focus of our assessment was generally very limited. An exception was the comprehensive comparative study by Brothers (1975) of the three goby species in the CIQ complex from Mission Bay just to the north of San Diego Bay. The life history information in Brothers (1975) allowed us to calculate estimates for CIQ gobies using all three assessment models. *FH* and *ETM* estimates were calculated for all of the other taxa except silversides, which was assessed using only the *ETM* approach (**Table 5.2-1**).

The previous 316(b) demonstration study (SDG&E 1980) used an assessment method that was similar to the *ETM* that compared entrainment losses with estimates of source water population abundances. The source water volume used in their calculations was the entire volume of San Diego Bay at mean tide, which was estimated to be 2.3 x 10^8 m^3. The source water volume used in our *ETM* calculations was less, 1.5 x 10^8 m^3, because it only included the southern half of the bay. This is a more conservative estimate of the source water volume for SBPP and the percentage losses to standing stock from the previous 316(b) study would need to be increased by approximately 35 percent to make them comparable with the current study. Our *ETM* estimates were calculated assuming maximum design flows for the entire year; therefore actual losses are significantly less.

Allen (1999) provides area and habitat-specific density estimates that could be used with our estimates of the source water volume to calculate source water populations for the south San Diego Bay area. Allen (1999) separated the density estimates by area into intertidal, nearshore, and channel habitats. We calculated estimates of the areas for these three habitats using data from our source water volume calculations as follows:

- intertidal – area below +1.0 ft MLLW but less than –2.0 ft,
- nearshore – area deeper than –2.0 ft but shallower than –10 ft;
- channel – bottom area in the south bay deeper than –10 ft MLLW.

The calculated areas for the three habitats were:

- intertidal – 318 ha, 11 percent of source water area
- nearshore – 783 ha, 28 percent of source water area
- channel – 1,737 ha, 61 percent of source water area

These estimates are slightly different than the ones provided by Allen (1999), because they were based on more recent detailed bathymetry that was used in the current source water calculations.

5.2.1 CIQ Goby Complex

Goby larvae were the most abundant taxon collected during the study (**Tables 3.3-1** and **3.3-2**). Entrainment data were used in the *FH* and *AEL* models to estimate that 1.2–2.2 million equivalent adults are removed from the south San Diego Bay population each year if all of the SBPP circulating water pumps were operated continuously for the entire year (**Table 5.2-1**). The estimates are close to the relationship of $2 \cdot FH = AEL$ and provide some assurance that the life history parameters from Brothers (1975) used in the calculations were reasonably accurate.

Allen's (1999) study of the fishes of San Diego Bay provides information that was used to estimate that the average standing stock in the south bay source water for the 1995 through 1999 period was 10.6 million CIQ gobies. The large estimate of 10.6 million for the source water population does not seem unreasonable given the large number of goby larvae collected at the source water stations (**Tables 3.3-5** and **3.3-6**). Despite the large number entrained, goby larvae were actually collected in greater numbers at several of the source water stations and were highest at Stations SB6 and SB7 (**Tables 3.3-5** and **3-3-6**; see **Figure 3.2-1** for station locations). Gobies are most abundant in mud bottom habitats such as the long shallow channel leading to the plant's intake. Allen (1999) estimated average densities of almost 5.0 CIQ gobies per m^2 at his intertidal stations in the south ecoregion. The small size of the larvae collected near the intakes (**Figures 3.3-12**) indicate that most of the entrained larvae were recently hatched and may have even originated come from goby egg masses hatched in the intake channel area.

The *FH* and *AEL* estimates were used to estimate that under maximum design flows the SBPP CWS results in losses of 11–21 percent of the average standing stock of CIQ gobies in the south bay source water estimated for the 1995 through 1999 period. Usually this type of comparison is very problematic because the larval and adult populations used in the comparison experience large interannual variation and there is no way of knowing how representative the estimates are for the time period being compared. In this study there were five years of data on the adult population from Allen (1999) and two years of entrainment data from our study, which helped account for some of the interannual variation. This may help explain why the *FH* and *AEL* estimates of percentage losses to adult standing stock are reasonably close to the *ETM* estimates from the current study (21 and 27 percent; **Table 5.2-1**) and the percentage losses estimated from the previous study (SDG&E 1980). When adjusted for the differences in source water volume used in the two studies, the estimates of the average annual losses of 12 percent and losses during peak periods of abundance of 28 percent from the previous study are very close in value to our estimates using *FH* and *AEL*. The current study included are more thorough characterization of the source water population than the previous study that, in combination with natural variation, probably explains the differences between the *ETM* estimates from this study and the estimates of percentage losses from the previous study. The comparisons of the estimates of entrainment mortality from both studies, and from using multiple assessment models in this study, all help provide assurance that our assessment of entrainment effects is reasonably accurate.

The previous 316(b) demonstration concluded that these levels of entrainment losses should have no measurable effect on the overall goby population in south San Diego Bay. This is especially true for gobies that appear to be well adapted to habitats with strong tidal currents. Brothers (1975) calculated that tidal exchange alone in Mission Bay would result in a larval survival rate of only 0.02 percent over a 15-day period, but his data showed survival rates of 0.8 to 1.7 percent over a two-month period. He concluded that the larvae are probably capable of some oriented behavior to areas with reduced tidal exchange and thus enhance their survival through behavioral mechanisms.

In addition to the similarity in the estimates of entrainment effects between this study and the earlier study in 1979–1980 (SDG&E 1980) the estimated number of larvae entrained, 2.2 billion (**Table 3.1-1**), is very close to our recent estimates of 1.4 to 1.8 billion larvae (**Tables 3.3-1** and **3.3-2**). While this type of comparison cannot be used to conclude that there are no CWS effects, since there are no data from before plant operation for comparison, it may indicate that there has not been a long-term downward trend in goby larval production during plant operation. The absence of a long-term downward trend in goby populations that may be attributed to plant operation is supported by Allen's (1999) abundance data on arrow and shadow gobies that showed increases through time during his 5-year study. Any initial effects of the power plant that may have occurred and the additional larval mortality due to continued operation appear to have been compensated for since the larval production and adult populations appear to be relatively stable based on these comparisons. The absence of any long term downward trend in goby abundance may be partially due to potential behavioral mechanisms that help them survive in estuarine conditions with high tidal currents and that also help reduce entrainment effects.

5.2.2 Longjaw Mudsucker

The longjaw mudsucker is a species of goby with a distinct larval form that could be separated in plankton samples from other co-occurring goby species. Longjaw mudsucker was the fourth most abundant taxon collected in entrainment samples in 2001, and the third most abundant in 2003 (**Tables 3.3-1** and **3.3-2**). The 2*FH* estimates of approximately 3,000 adults lost annually were very similar for the two study periods (**Table 5.2-1**). *AEL* estimates were not calculated because no species-specific survival estimates were available for the later stage larvae and juveniles.

The limited fishery information on mudsuckers, which are occasionally taken for bait, was insufficient to develop any population estimates. However, Allen's (1999) study of the fishes of San Diego Bay and a study by Williams and Zedler (1999) of fish densities in salt marsh habitat near SBPP provided data to estimate standing stock of mudsuckers. Mudsuckers are primarily found in intertidal channels, so a density estimate of $0.86/m^2$ from Williams and Zedler's 8-year study was applied to the intertidal area of the source water body and reduced by a factor of 100 to account for the specific intertidal channel marsh habitat sampled in their study. This resulted in an estimate of 27,427 mudsuckers in the source water. Allen's (1999) sampling likely underestimated mudsucker abundance in the bay because upper marsh areas were not sampled, but the average densities weighted by gear type resulted in an estimate for the source water area of 1,978 mudsuckers. An average standing stock abundance of the two studies yields an estimate of approximately 15,000 mudsuckers. Based on this source population estimate, *FH* estimates of entrainment effects (average approximately 3,000 adults annually) under maximum design flows over the entire year are equivalent to losses of approximately 20 percent of the adult standing stock.

Adult longjaw mudsucker were collected in very low abundances (19 fishes over five years) from only the southernmost stations in Allen's study (1999). These low abundances do not allow us to draw any conclusions regarding possible trends in the adult populations of this species over time. In the 2003 study, mudsucker larvae were found in greatest abundance at the entrainment station, especially during 2003 when they were not present at several of the source water stations (**Tables 3.3-5** and **3.3-6**). This pattern of abundance resulted in the relatively high *ETM* estimates, especially for the 2003 study period (**Table 5.2-1**).

The estimate for the 2003 study period was most likely affected by the reduced sampling effort. Entrainment and source water abundances peaked twice during 2001 indicating that spawning activity can occur from October through May (**Figures 3.3-13** and **3.3-15**). During 2003, mudsucker larvae were only collected from the first three surveys in both entrainment and source water samples and were in very low abundances in source water samples (**Figures 3.3-14** and **3.3-16**). The bi-monthly sampling schedule clearly affected the source water and entrainment estimates during the 2003 sampling period. The results from 2001 indicate that the best sampling schedule for longjaw mudsucker would start in early fall and continue through the following spring. Therefore, the *ETM* estimate for the 2001 study period is believed to represent the best estimate of entrainment effects.

The entrained larvae were all less than 4 days old based on their length frequency distribution (**Figure 3.3-17**) and a larval growth rate of ca. 0.5 mm/d derived from data in Weisel (1947) and Walker et al. (1961). The patterns of abundance and the size of the entrained larvae indicate a main source population that is near the power plant intakes, probably inhabiting the salt marsh channel areas directly north of SBPP at Telegraph Creek and west of the plant within the Chula Vista Wildlife Reserve. Similar to other species of gobies, longjaw mudsucker larvae actively avoid strong tidal currents that can transport them offshore and away from their bay habitat (Brothers 1975). Barlow (1963) observed postlarval mudsuckers swimming into strong tidal currents and then descending to the bottom where they were able to maintain their position.

Longjaw mudsuckers are highly prized as bait fish because they are extremely hardy and can survive wide ranges in salinity, oxygen and periods when they are stranded out of water (Love 1991). Although there are no similar studies on the larvae, high entrainment survival has been shown to be very high in other species of gobies (EPRI 2000). Therefore, some level of entrainment survival could be expected despite the assumption of 100% mortality used in the assessment models. The models also assume that the plant CWIS is continuously operating. These conservative assumptions, in combination with behavioral mechanisms that help reduce entrainment effects, the potential for entrainment survival, and the limited time period that the larvae are exposed to entrainment all help support the conclusion that there is little potential for any long-term effects on longjaw mudsuckers in south San Diego Bay.

5.2.3 Anchovies

Anchovies (*Anchoa* spp.) were the second most abundant taxon in entrainment samples during both study periods (**Tables 3.3-1** and **3.3-2**). Entrainment estimates based on maximum design flow for the entire year were used to calculate 2*FH* estimates of 44,000–214,000 equivalent adults (**Table 5.2-1**). *AEL* estimates were not calculated for anchovies because no species-specific survival estimates were available for the later stage larvae and juveniles. There were only very limited fishery data for slough and deepbody anchovies, so Allen's (1999) study of the fishes of San Diego Bay provided the only information to obtain an estimate of 9.4 million anchovies (slough and deepbody combined) in the south bay source water. Based on this source population estimate, 2*FH* estimates of entrainment effects under maximum design flows over the entire year are equivalent to losses of 0.5 to 2.3 percent of the adult standing stock.

The entrainment and source water sampling results indicate that abundances of anchovy larvae are highly variable both among stations (**Tables 3.3-5** and **3.3-6**) and seasonally (**Figures 3.3-18**, **-19**, **-20**, and **-21**). The five-year study by Allen (1999) also showed considerable variation among years. Although entrainment abundances would also be expected to show similar interannual variation, the estimated total annual entrainment from 2001, and the earlier 1979–1980 study periods were very similar, and the 2003 estimate was higher but still less than an order of magnitude different (**Tables 3.1-1** and **5.2-1**). The *ETM* estimates for the 2001 and 2003 study periods of approximately 8−10 percent (**Tables 5.2-1**) are also similar to estimated losses to larval standing stock from the earlier 316(b) Demonstration (annual average estimate 5.2 percent and peak abundance estimate 8.1 percent) (SDG&E 1980) when those estimates are increased to account for the differences in source water volume assumptions between studies.

The consistency of the estimates of total annual entrainment and losses to standing stock (*ETM*) between the two studies may indicate that, while highly variable over short time scales, the anchovy population in the south bay is fairly stable over longer time periods. This short-term variability is related to the reproductive biology of anchovies. Anchovies have high fecundity and may spawn every few days during the reproductive season (Jung 2002). The population can produce abundant larvae but initial mortality rates are very high (Rilling and Houde 1999). This contributes to the high spatial and temporal variability seen in the data, but this life history strategy also helps to ensure a relatively stable population over longer periods of time in variable environments. The incremental mortality caused by entrainment (~10 percent) is proportionally small relative to variation in natural mortality and probably does not represent a significant risk to the local population. This conclusion is supported by the consistency in the estimates between the previous and current studies, which cannot be used to conclude that there are no CWS effects, since there are no data from before plant operation for comparison, but may indicate that there has not been a long-term downward trend in goby larval production during plant operation.

5.2.4 Silversides

Silversides were the fifth most abundant taxon in entrainment samples during both study periods (**Tables 3.3-1** and **3.3-2**). *AEL* and *FH* estimates were not calculated from the entrainment estimates because there were no species-specific survival estimates for jacksmelt or topsmelt larval and juvenile stages. Therefore, no accurate estimates of equivalent adult losses due to larval silverside entrainment could be developed. Allen's (1999) study of the fishes of San Diego Bay was used to provide information on adult populations of silversides in San Diego Bay. Densities of jacksmelt and topsmelt from his study were used to calculate an estimate of 3.1 million adult silversides (jacksmelt and topsmelt combined) in the south bay source water.

The entrainment and source water sampling results indicated that silverside larval abundances were highly variable among stations (**Tables 3.3-5** and **3.3-6**). Topsmelt was the second most abundant adult fish collected by Allen (1999). His results showed that abundances of both adult topsmelt and jacksmelt were highly variable among years. Although entrainment abundances might be expected to show similar interannual variation, the estimates from the 2001 and the earlier 1979–1980 study periods are almost equal, 1.5×10^7 and 1.4×10^7, respectively (**Tables 3.1-1** and **5.2-1**). While this type of comparison cannot be used to conclude that there are no CWS effects, since there are no data from before plant operation for comparison, it may indicate that there has not been a long-term downward trend in silverside larval production during plant operation. Any initial effects of the power plant that may have occurred and the additional larval mortality due to continued operation appear to have been compensated for since the larval production and adult populations appear to be relatively stable based on this comparison. This conclusion is supported by abundance data from Allen (1999) that shows no long-term declines for adult jacksmelt and topsmelt and the *ETM* estimates for the 2001 and 2003 study periods (ca. 15 percent, **Table 5.2-1**) that are bounded by the estimates from the earlier study of average annual larval losses of silversides of 1.1 percent and losses during peak periods of 23 percent (SDG&E 1980).

Based on the estimate of the adult population in the San Diego Bay source water derived from Allen (1999) the *ETM* estimates of proportional larval mortality indicate losses of approximately 450,000 adult silversides annually. This approach assumes a stable adult population and no compensation, but can be used to place the *ETM* estimates into some context in the absence of other assessment models. This approach would be very conservative for silversides due to the large variability in the adult population that far exceeds the 15 % *ETM* estimate. Commercial landings for jacksmelt fluctuate greatly from year to year. Total statewide landings for silversides in 2002 were reported to be 37 mt (81,449 lb) with a value of $25,275 (CDFG California Commercial Landings for 2002). Based on these values the average value per pound was calculated to be approximately $0.31. The weight at maturity for silversides was estimated from a length to weight relationship for topsmelt of $\text{Weight}=0.0000992 \cdot \text{Length}^{2.59}$ from Calliet et al. (2000) to calculate weights of 52 g (0.12 lbs) for jacksmelt and 14 g (0.03 lbs) for topsmelt. The proportions of jacksmelt and topsmelt in entrainment were estimated as 70 and 30 percent, respectively, from the silverside larvae that could be identified to species. These values were used to estimate a total ex-vessel dollar value of approximately $13,000 for silverside entrainment losses.

5.2.5 Combtooth Blennies

Combtooth blennies were the third most abundant taxon collected in entrainment samples in 2001, and the fourth most abundant in 2003 (**Tables 3.3-1** and **3.3-2**). *AEL* estimates were not calculated because no species-specific survival estimates were available for the later larval stages and juvenile stages. The 2*FH* estimates were very similar for the two study periods and yielded calculated losses of approximately 22,000 adults annually as a result of entrained larvae (**Table 5.2-1**). The abundance of adult blennies in the source water population could not be accurately estimated because the only data available were for bay blenny *H. gentilis* from Allen's (1999) study using trawl and seine methods, and most of the entrained blenny larvae in the present study were probably mussel blenny *H. jenkinsi* based on the identification of advanced stage larvae. The entrainment and source water larval sampling from this study showed highest abundances at the two northernmost source water stations on the east side of the bay (SB7 and SB9) (**Tables 3.3-5** and **3.3-6**). Mussel blennies are probably much more abundant in San Diego Bay than bay blennies but live in specific habitats (on pier pilings, boat moorings and rocks within empty mussel shells, worm tubes, barnacle tests and boring clam holes) that cannot be sampled except by direct diver observation. Ninos (1984) measured average densities of mussel blennies at Catalina Island of 17.6/m^2 and Stephens et al. (1970) noted them as abundant on pilings and floats in Newport Bay. Given the extensive bayfront development along the eastern margin of San Diego Bay and the large area of potential habitat for mussel blennies, it is reasonable to estimate that the adult population could be on the order of several hundred thousand individuals. Entrainment effects based on 2*FH* would be equivalent to losses of less than 10 percent of the adult standing stock.

Abundances of adult combtooth blennies were highly variable among years and did not indicate any trend in the adult population (Allen 1999). Some of this variability is probably due to the sampling methods, which did not target preferred blenny habitats. Although the adult population may be highly variable, the entrainment and *ETM* estimates were almost equal for the 2001 and 2003 sampling periods (**Table 5.2-1**). This may indicate a more stable adult population that is not declining over time due to increased larval mortality from the plant. Our estimates of entrainment effects include assumptions that the plant CWIS is continuously operating and 100 percent mortality of all entrained organisms. These conservative assumptions and the low *ETM* estimates of ca. 3% for both years indicate that entrainment would not significantly affect combtooth blennies in south San Diego Bay.

5.2.6 Entrainment Effects Conclusion

The results indicate low potential for entrainment effects on bay-wide populations of the five target taxa. The results for some of the taxa were very similar to results for the 1979–1980 316(b) Demonstration (SDG&E 1980), suggesting that the populations for these taxa have not been experiencing any long-term declines in abundance over the 20+ year period between studies that might be caused by increased mortality on the larvae due to entrainment. This absence on any evidence for long-term declines is also supported by data from Allen's (1999) five-year study on the adult fishes of San Diego Bay. Although adult abundances of some species sampled in that study varied considerably among years, there was no indication of any declining population trends that may have been due to increased mortality from entrainment. While these types of comparison can be used to argue that there are no declining population trends they cannot be used to conclude that there were no CWS effects, since there are no data from before plant operation for comparison.

The *ETM* estimates of losses to larval source populations due to entrainment were all low relative to the natural variation observed in the adult populations. It is not known to what extent different fish populations may compensate for the increased larval mortality rates caused by power plant operation, but the magnitude of losses at the adult population level are below those that can be directly measured by sampling because of naturally high interannual variation in abundances. Similarly, any potential effects on trophic level functioning in south San Diego Bay as a result of entrainment losses could not be measured directly. Small incremental increases in mortality due to entrainment, even under maximum design flows, may be compensated for by increased survival of later larval and juvenile stages. The similarity in the estimates of entrainment losses between the 1979–1980 and 2001–2003 studies may indicate that compensatory mechanisms are operating to maintain long-term stability in some populations. There is also evidence that some of these taxa, such as gobies, have behavioral adaptations to living in high current environments that may also help to reduce entrainment effects.

The approximate direct annual value of fishery species lost as a result of larval entrainment by the SBPP cooling water system could not be calculated for most fishes and invertebrates because there was insufficient life history information and entrainment abundance to model adult equivalent losses using the AEL approach described in Section 3.2.3–*Data Analysis.* Silversides was the only taxon with assessment results that also had commercial landings data that could be used to assign monetary value to the losses. The *ETM* estimates of proportional larval mortality indicate losses of approximately 450,000 adult silversides. This extrapolation assumes a stable adult population and no compensation, but can be used to place the *ETM* estimates into some context in the absence of data from other assessment models. This approach would be very conservative for silversides due to the large variability in the adult population that far exceeds the 15 % *ETM* estimate. The approximate ex-vessel dollar value of entrainment losses to silversides was $13,000 (**Table 5.2-3**).

Table 5.2-3. Estimates of annual dollar value for fishery taxa entrained (2001 period) and impinged (2003 period) at SBPP. Weights are for sizes of mature individuals typical for the fishery. Ex-vessel values based on CDF&G landings data in San Diego, Los Angeles or statewide during 2002.

Taxon	ETM Estimated Entrainment Effects	Estimated Annual Impingement	Typical Fishery Weight (g)	Typical Fishery Weight (lb)	Price/lb[5]	Entrainment Species Valuation ($)	Impingement Species Valuation ($)	Total Value ($)
shiner surfperch	0	549	45	0.1	$1.02		$56	$56
northern anchovy[1]	-	16	19	<0.1	$0.27		<$1	<$1
Pacific sardine[2]	-	28	125	0.3	$0.30		$2	$2
topsmelt	135,087	11,617	14	<0.1	$0.31	$1,256	$108	$1,364
jacksmelt	315,204	47	52	0.1	$0.31	$11,725	$2	$11,727
croakers[3]	-	33	70	0.2	$1.42		$8	$8
Shortfin corvina[4]	-	428	350	0.8	$2.11[6]		$697	$697.
white seabass	-	14	3,773	8	$2.11		$245.	$245.
queenfish	-	152	25	0.1	$0.79		$7	$7
sand bass	-	24	781	1.7	$2.11[6]		$87	$87
California halibut	-	7	1,916	4.2	$3.17		$93	$93
diamond turbot	-	382	190	0.4	$0.40		$64	$64
striped mullet	0	14	378	0.8	$2.11[6]		$24	$24
market squid	0	208	226	0.5	$0.56		$58	$58
octopus	0	74	907	2.0	$0.55		$81	$81
Cal. spiny lobster	0	13	1,360	2.5	$6.50		$211	$211
						$12,981	$1,747	$14,728

[1] Weight from Clark and Phillips 1952
[2] Weight from Hill et al. (1999)
[3] Life history data from white croaker; entrainment AEL estimate includes all croaker species
[4] Life history data from California corbina
[5] From California Department of Fish and Game Poundage and Value of Landings by Area for 2002 http://www.dfg.ca.gov/mrd/landings02/table15.pdf
[6] Value not available for species; price/lb value for white seabass used instead

5.3 Summary of Impingement Results

The purpose of the SBPP impingement studies was to evaluate the potential effects of the cooling water intake system as required under Section 316(b) of the Federal Clean Water Act (CWA) (USEPA 1977). Data on fishes and selected invertebrates impinged by the operation of the SBPP intake cooling water system were used to complete this portion of the 316(b) Demonstration. Results are summarized in **Table 5.2-3** and discussed in the following sections.

Results from the 12-month long sampling effort were used to identify the most abundant fish and invertebrate taxa that were evaluated for impingement effects. A total of 50,970 individual fishes comprising approximately 50 taxa, and over 1,000 invertebrates comprising 80 taxa, were collected from the impingement samples (**Table 4.3-1**). The fishes had a total wet weight biomass of 74 kg (163 lb). Total annual impingement of fishes under maximum design flows was estimated to be 385,588 individuals fishes weighing 556 kg (1,226 lb). The most abundant impinged taxon both numerically and by weight was anchovies (*Anchoa* spp.) (**Table 4.3-1**), comprising about 94 percent by number and 40 percent by biomass of all of the fishes impinged. Other fish taxa important to the commercial or recreational fishery that were analyzed as part of this assessment were silversides, sand basses, and shortfin corvina.

The only invertebrates counted, measured, and weighed during impingement studies were crustaceans (shrimps, crabs, and lobster) and cephalopods (squid and octopus). The rest of the invertebrates, including hydroids, sponges, jellyfish, bryozoans, worms, mussels, snails, sea stars, and tunicates, were only recorded as present. A total of 1,106 crustaceans and cephalopods comprising 30 taxa were enumerated during this study (**Table 4.4-1**). These individuals had a total wet weight biomass of 3.1 kg (6.8 lb). The estimated total annual abundance of impinged invertebrates under maximum design flows was 9,019 individuals, with an estimated annual biomass of 22.6 kg (49.8 lb) (**Table 4.4-1**). The most abundant target invertebrates were small shrimps and crabs. Brown shrimp, California spiny lobster, market squid, and octopus were analyzed as part of this assessment because their counts or biomass were within the top 90 percent of all targeted invertebrates and they were important as fishery species.

There were several differences between the previous impingement study results (SDG&E 1980) and the 2003 study. The estimated annual fish impingement in the prior study was 28,174 individuals weighing 4,459 kg (9,830 lb), while in the 2003 study it was estimated at 385,588 fish weighing 556 kg (1,226 lb). The abundance of total fishes increased while the biomass decreased, with both changes being on the order of one magnitude. Other differences are as follows:

- There was a larger number of very small anchovies (mainly slough anchovy) that were impinged during the 2003 study than in the previous one. This increase was the main reason for the large difference in the total number of fishes impinged.
- There were more impinged individual pipefishes, California halfbeaks, and gobies during the 2003 study.
- There was a lower number and biomass of round stingray, specklefin midshipman, deepbody anchovies, diamond turbot, California halibut and Pacific butterfish during the 2003 study. These weight differences accounted for the majority of the overall biomass differences between the two studies.
- There were fewer impinged silversides (mainly topsmelt) and shiner surfperch during the 2003 study.
- The average individual fish weight of slough anchovy, topsmelt, specklefin midshipman, shiner surfperch, and diamond turbot was lower during the 2003 study, while the average weight of round stingrays was slightly higher in 2003 than in the 1979–1980 study.

During Allen's (1999) five-year study, there was no consistent trend in changes in abundance over time for any of the above listed taxa in south San Diego Bay. Impingement of invertebrates was not assessed during the 1979–1980 study (SDG&E 1980) and therefore no comparisons can be made.

5.3.1 Fishes

5.3.1.1 Anchovies

This complex was mainly composed of individuals that were identified in the field samples as slough anchovy (*Anchoa delicatissima*) or unidentified anchovy (*Anchoa* spp.); only three deepbody anchovy (*A. compressa*) were identified during this study. The *Anchoa* spp. individuals were generally small and not possible to identify to the species level in the field, but were probably slough anchovy. The estimated annual impingement of anchovies under maximum design flows was 359,420 individuals with a biomass of 222.0 kg (489.5 lb). The estimated annual impingement of slough and deepbody anchovies in the 1979–1980 study was 3,132 and 3,689, respectively (no weights were presented).

This group is not commercially or recreationally important, and there are no fishery landing data to compare to impingement estimates. Allen (1999) estimated the stock of slough anchovies in his south and south central ecoregions of San Diego Bay (the area considered the source water for the entrainment portion of this report) at 10,170 and 22,580 kg, respectively (total=32,750 kg). Based on these estimates, SBPP impinges about 0.7 percent of the estimated biomass of slough anchovies in the southern portion of San Diego Bay.

5.3.1.2 Silversides

Topsmelt composed the majority of the impinged silversides at SBPP. The estimated annual impingement abundance of this group under maximum design flows was 11,664 individuals, weighing 25.7 kg (56.7 lb). Even though many of the silversides impinged were small they were assumed to be adults for this assessment. The estimated annual impingement of topsmelt in the 1979–1980 study was 5,147 (no weight was presented).

The estimated stock of topsmelt in south and south central area of San Diego Bay was 5,384 and 5,492 kg, respectively (total=10,876 kg) (Allen 1999). A comparison of this estimated stock of topsmelt to the annual biomass impinged shows that SBPP impinges about 0.2 percent of the estimated stock. Using a value of $0.35/lb and $0.78/lb for topsmelt and jacksmelt, respectively, from commercial landing data (CDF&G 2003) the total estimated value of impinged silversides under maximum design flows was $129.

5.3.1.3 Sand basses

Two sand basses, spotted and barred, are common in the San Diego area. Only three sand basses were impinged at SBPP during this study (**Table 4.3-1**). The estimated annual impingement abundance of sand bass under maximum design flows was 24 individuals, weighing 13.1 kg (28.8 lb) (**Table 4.3-1**). No estimates of impingement of sand bass were presented in the earlier impingement study (SDG&E 1980).

Allen (1999) estimated the stock of barred sand bass in the south and south central portions of San Diego Bay at 19,442 and 10,536 kg, respectively. Using these values, it was estimated that SBPP impinges about 0.04 percent of the estimated stock for this portion of San Diego Bay. Sport catches of barred sand bass from 1998–2002 ranged from 410,000 to 1,130,000 and averaged 773,400 fish in the southern California region (RecFIN 2003).

5.3.1.4 Shortfin corvina

The estimated annual impingement of shortfin corvina under maximum design flows was 428 individuals, weighing 10.9 kg (24.1 lb) (**Table 4.3-1**). Allen (1999) states that San Diego Bay is near the northern limits of the range for this species and that the standing stock of shortfin corvina in the south and south-central portions of San Diego Bay is approximately 700 kg. SBPP therefore impinges about 1.5 percent of the estimated stock of shortfin corvina. No estimates of annual impingement of shortfin corvina were presented in the earlier impingement study (SDG&E 1980). This species is not abundant in California and is caught only incidentally in the sport fishery.

5.3.2 Macroinvertebrates

5.3.2.1 Brown shrimp

The estimated annual impingement abundance of brown shrimp was 225 individuals weighing 4.9 kg (10.7 lb) (**Table 4.4-1**). This species is not a targeted fishery species in California as San Diego is near its northern range limit and generally not in high abundance in the fishery catch. It is anticipated that effect caused by the removal of this small quantity of shrimp from the local population would be minimal.

5.3.2.2 California spiny lobster

The estimated annual impingement abundance of spiny lobster was 13 individuals weighing 4.41 kg (9.71 lb) (**Table 4.4-1**). The 2002 price to the fishery for spiny lobster was about $6.50/lb (CDF&G 2003). Based on an average size spiny lobster in the commercial fishery of 3 lbs, the estimated adult equivalent biomass value of impinged lobsters was approximately $250 annually (**Table 5.2-3**).

5.3.2.3 Market squid

The estimated annual impingement abundance of market squid was 208 individuals weighing 0.07 kg (0.15 lb) (**Table 4.4-1**). During 2002, the value of jumbo squid to the fishery landed in San Diego equals about $0.56/lb (CDF&G 2003). Based on an average size squid weight of 0.5 lb, the value of the adult equivalent squid biomass impinged at SBPP is less than $60 annually (**Table 5.2-3**).

5.3.2.4 Two-spotted octopus

The estimated annual impingement abundance of two-spotted octopus was 74 individuals weighing 3.71 kg (9.18 lb) (**Table 4.4-1**). The 2002 value of octopus landed by the commercial fishery in the San Diego area was about $0.55/lb (CDF&G 2003). Assuming an average weight of 2.0 lb for an adult octopus, the fishery value of the number of adult equivalent octopuses would be approximately $80 annually (**Table 5.2-3**).

5.3.3 Impingement Effects Conclusion

The majority of the fishes impinged, over 96 percent of the abundance and 87 percent of the biomass, is not targeted by commercial or recreational fisheries. The estimated dollar value for impingement losses under maximum design flows was estimated as less than $1,500 for the fishes with commercial fishery landings (**Table 5.2-3).** The fishes analyzed as part of this assessment were also compared to stock estimates from Allen's (1999) five-year study of the fishes of San Diego Bay. SBPP impinges from 0.04 to 1.5 percent of the standing stock biomass of these taxa in the southern portions of San Diego Bay. The small magnitude of the estimated impingement effects under maximum design flows indicate that

SBPP operation represents a low potential risk to the target taxa populations. The previous 316(b) Demonstration also concluded that impingement effects were not significant (SDG&E 1980).

The majority of the target invertebrates impinged, 94 percent of the abundance and 42 percent of the biomass, are not targeted by commercial or recreational fisheries. The estimated dollar value for impingement losses to target invertebrates under maximum design flows was estimated as approximately $400 annually for the small numbers with commercial fishery landings (**Table 5.2-3**). This estimate does not include brown shrimp that are not part of a local fishery. Estimated impingement totals were very low indicating very little potential risk to the populations of targeted invertebrates.

SBPP currently has a functional system to return impinged material to San Diego Bay. All impinged material, both organisms and debris, is rinsed from the traveling screens into a trough. This material and bay water then flows along the trough and is deposited back into the bay near the Units 1 and 2 points of discharge in the discharge channel. At times birds rest on the edge of the trough and appear to feed on the organisms as they pass by in the trough. This system would be more efficient in returning organisms back to the bay if it had some type of low cover to minimize bird feeding as is recommended in Section 6.0 –*Technological, Design, and Operational Alternatives to Minimize Adverse Environmental Impacts.*

5.4 Summary Conclusion on Cooling Water System Effects

The combined effects of entrainment on larvae and impingement on juveniles and adults are evaluated for two of the target taxa: anchovies and silversides (**Table 5.3-1**). These two taxa were chosen because of their susceptibility to both entrainment and impingement and their numerical dominance as larvae and adults in entrainment and impingement samples, respectively. The estimates of the adult stocks for these two taxa are based on data from Allen (1999) by multiplying the densities measured for the various habitats in his south and south-central ecoregions by our estimates of the surface areas for the three habitats from our source water volume calculations. Allen's stock estimates are based on sampling densities that include fishes in various age classes and therefore are not necessarily estimates of adult standing stock. Similarly our impingement estimates include fishes of various sizes. The estimate of entrainment mortality assumes a linear relationship between larval losses and resulting losses at any later stages and therefore could also be applied to Allen's stock estimate. Estimates of combined cooling water system effects for anchovies and silversides yielded similar results (**Table 5.4-1**).

The estimated annual value of the combined entrainment and impingement losses was less than $15,000 (**Table 5.2-3**). This estimate includes the value of impingement losses for several fishes and invertebrates, but only the value of silverside losses was estimated for entrainment. Entrainment effects were only estimated for the most abundant fishes that, with the exception of silversides, had no commercial landings that could be used to value the losses. Entrainment of other commercially or recreationally important fishes and invertebrates was very low (**Table 5.2-2**). The estimated annual value is conservative because of the low entrainment of other commercially or recreationally important fishes and the use of the *ETM* estimate to value silverside losses. Therefore, the actual ex-vessel dollar value of fishery losses would be expected to be considerably less than the $15,000 estimate.

Table 5.4-1. Summary of SBPP cooling water intake system effects for anchovies and silversides.

Taxon	Stock Estimate[1]	Average Entrainment % Mortality[2]	Average Impingement % Mortality[3]	Total % Mortality
anchovies	9,400,621	9.2	3.8	13.0
silversides	3,052,820	14.7	0.4	15.1

[1] Estimate combined from south and south-central ecoregions from Allen (1999) using source water volume calculated for south San Diego Bay (see Section 2.3–*Source Water Volume*).

[2] Average ETM estimate from 2001 and 2003 periods.

[3] Estimate calculated from annual impingement and Allen's (1999) stock estimate.

The recent Nearshore Fisheries Management Plan (NFMP) (CDF&G 2002) provides guidelines for setting the Total Allowable Catch (TAC) for 19 species of nearshore finfish species and has applicability to the analysis of power plant related losses. The new guidelines for developing regulations and catch limits were implemented to prevent overfishing, rebuild depressed stocks, ensure conservation, and promote habitat protection and restoration. In fisheries with a moderate level of data the default fishing rate for these 19 species would be the rate that reduces the average recruits per spawner to 50 percent of the unfished level. These guidelines for TAC were established for fishes that are generally longer-lived with populations with multiple age-classes that may not have the high levels of surplus production characteristic of fishes that have shorter lifespans, higher fecundity, and more variable populations. The TACs from the NFMP would therefore be very conservative for anchovies and silversides.

Our mortality estimates are also conservative because they do not assume any migration into the south San Diego Bay population from the rest of San Diego Bay and offshore for silversides. The estimates are also based on the plant operating at maximum design flows for the entire year and 100 percent mortality of all entrained organisms. The plant has not operated at maximum design flows the past few years, and some level of entrainment survival has been shown to occur for even fragile larvae such as anchovies (EPRI 2000). Finally, the *ETM* estimates assume a linear relationship between larval losses and resulting losses at later stages and include no adjustment for compensatory mechanisms that may result in reduced mortality in the larvae and juveniles that survive entrainment. Losses due to the power plant cooling water system are well below the guidelines for TAC in the NMFP and indicate little potential for impacting these taxa.

Stock estimates for these two taxa were variable over the 1994–1999 period but did not indicate any declining trends in abundance (Allen 1999). This and the similarity in the entrainment estimates between the 1979–1980 316(b) Demonstration (SDG&E 1980) and this study indicates that these two taxa are not experiencing any long-term population declines that might be due to the CWIS. These comparisons and the other results indicate that SBPP does not adversely affect fish and invertebrate populations in south San Diego Bay. The report from the previous 316(b) Demonstration reached the same conclusion using a fisheries model for establishing levels of maximum sustainable yield. The consistency of the results and findings from the two studies support the conclusion that the SBPP cooling water intake system as presently designed and operated has minimal adverse environmental effects in south San Diego Bay.

6.0 Technological, Design, and Operational Alternatives to Minimize Adverse Environmental Impacts

6.1 Background

Section 316(b) of the Clean Water Act requires that "the location, design, construction, and capacity of cooling water intake structures reflect the best technology available for minimizing adverse environmental impact." This requirement applies both to new sources that are seeking National Pollutant Discharge Elimination System (NPDES) permits for the first time and to existing sources, such as the SBPP, that seek to renew existing permits.

In compliance with Section 316(b) objectives, the design of the SBPP Units 1–4 intake employs design features to minimize entrainment and impingement. The design and operation of the CWIS are described in Section 2.0, along with a discussion of the physical and biological characteristics of the source water. Entrainment, impingement, and effects of the CWIS are described in Sections 3.0 through 5.0.

The U.S. Environmental Protection Agency interprets Best Technology Available (BTA) as "the best technology available commercially at an economically practicable cost." Determination of BTA includes the design, capacity, and location of the facility's cooling water intake, as well as cost considerations. It should be noted that the current SBPP NPDES permit describes the existing cooling water system as BTA.

Duke Energy leases both the site property and the power generating facilities located on the site from the Port of San Diego under a lease agreement that expires in November 2009. Thereafter, a 3-month period has been designated during which time the "must run" status of the plant will be evaluated. If at the end of that period the plant is considered by the California Independent System Operator (ISO) to be a "must run" facility, the lease will continue in effect until that status is terminated. If the ISO determines that the facility is no longer a "must run" plant, Duke is obligated to demolish the plant unless the Port waives this requirement. At this time, Duke Energy believes it is unlikely that it will continue to operate the existing plant after November 2009, and any operating scenarios after that date are highly speculative, both in terms of the identity of the operator and the rate at which the plant would operate, if at all. Since Duke Energy's operation and control of the SBPP is currently expected to cease in February 2010, the evaluation of CWIS alternatives for purposes of this study assumes a four-year horizon to allow for a minimum of one year for planning, permitting, designing, and financing an alternative prior to amortizing its construction and O&M costs. In the evaluation of CWIS alternatives, project costs are amortized from the point of project financing approval. Should the plant continue to operate after that time, the NPDES permit will need to be renewed again and the CWIS alternatives can be re-evaluated at that time.

Sections 3 through 5 of this study conclude that entrainment and impingement effects of the SBPP do not pose a significant risk to the fish or invertebrate populations found in San Diego Bay and thus do not rise to the level of an “adverse environmental impact” as that term is used in Section 316(b) of the Clean Water Act. Consequently, the cost of most intake alternatives evaluated in this study were found to be wholly disproportionate to the environmental benefit gained.

In addition, in light of the likely shutdown of the existing plant in November 2009, alternative CWIS technologies that might otherwise be considered feasible for SBPP become infeasible because the time necessary to design, permit, and construct them exceeds the remaining anticipated operating life of the plant. As such, these alternatives are considered not “available” and therefore are not BTA.

Notwithstanding our conclusion that the existing intake represents BTA, this study includes a review and evaluation of alternative cooling water system technologies, including those that are considered infeasible due to the few remaining years of plant operation. This analysis may assist in identifying cost-effective alternatives that would reduce entrainment or impingement effects during the remaining years SBPP will be operated by Duke Energy even though such reductions are not required by the Clean Water Act. Our analysis of alternatives focuses on alternatives that would reduce the risk of population-level impacts as well as reduce the number of individuals lost to entrainment or impingement.

In determining BTA, alternative technologies, designs, and operational and maintenance features were evaluated to determine their potential to reduce biological losses and other environmental impacts to a greater extent than the existing facility. This section summarizes site-specific analyses of these alternatives on the basis of their availability, feasibility, cost-effectiveness, and the degree to which they minimize biological impacts. Sections 3.0 through 5.0 provide the site-specific framework used to evaluate the relative biological effectiveness and engineering feasibility of the alternatives considered.

Evaluation of CWIS alternatives was conducted in accordance with EPA’s Guidance for Evaluating the Adverse Impact of Cooling Water Intake Structures on the Aquatic Environment: Section 316(b) P.L. 92-500 (EPA Office of Water Enforcement, May 1, 1977). EPA recently adopted rules for 316(b) determinations for new facilities and is in the process of finalizing rules for existing facilities that are expected to be promulgated in February 2004. At the present time, BTA determinations for the SBPP are governed by the 1977 guidance, as refined through subsequent case law and EPA administrative decisions.

The range of alternatives available for upgrading an existing power plant is typically narrower than the range of alternatives for a new power plant. The cost of retrofitting an existing power plant to use an entirely different cooling technology can be so prohibitive or so disproportionate to the environmental benefits gained that the retrofitting is not considered BTA. Existing sites also may lack the space to accommodate large retrofits or provide choices for relocating CWIS. Economic considerations play an important role in arriving at a BTA determination. The Clean Water Act does not require traditional cost/benefit analyses of different alternatives or elimination of all adverse impacts associated with the operation of CWIS. Rather, it requires BTA for the CWIS that minimizes entrainment and impingement effects. The key question is the magnitude of any adverse environmental impact associated with a given technology. This determination is made on a case-by-case basis by assessing the biological value of reducing entrainment and impingement relative to the cost of the alternative.

The alternatives considered for the SBPP are divided into three groups:

- **Capacity or Flow Reduction Options.** The term "capacity" refers to the volume or flow of cooling water drawn through the intake. Levels of entrainment and impingement damage are in many cases correlated with the amount of water withdrawn. Reducing volume can minimize entrainment damage and may reduce impingement.

- **Location Options.** These options refer to the position or site occupied by the CWIS. Location has been referred to as the most important factor in minimizing adverse impacts from a CWIS because many adverse impacts can be avoided simply by not siting the intake in areas of sensitive or important natural resources. Changing the location of a CWIS to minimize adverse environmental impacts is clearly easier for a new facility than an existing facility. Nevertheless, it may be possible in some cases to reduce impacts by replacing an existing CWIS with a new one at a new location.

- **Design Options.** EPA has interpreted the design component of BTA to refer to various elements that make up the CWIS itself. These include screening systems intended to keep fish adults, juveniles, larvae, or eggs from being drawn into the plant, and fish bypass and return systems intended to minimize the adverse impacts of impingement. The design component of BTA also includes various types of pumps and intake technologies, such as velocity caps, that influence the volume and velocity of water drawn into the plant.

A two-step system was used to evaluate alternatives to the existing CWIS. In the first step, potential alternatives were evaluated based on whether they are commercially available and have been used successfully at a power plant similar in size and environmental setting to SBPP, i.e., whether they met the BTA criteria for "proven and available." Alternatives that did not meet these criteria were eliminated from further consideration.

In the second-step, alternative intake technologies that met the site-specific proven and available criteria were further analyzed based on the following specific technical, economic, biological, and other environmental criteria:

- **Technical Criteria.** Technical criteria addressed the compatibility of each alternative intake technology with the existing facility design and site layout, including space availability on land or in the nearby shoreline area. Each alternative was evaluated based on site-specific considerations, including engineering feasibility, operations, and reliability.

- **Economic Criteria.** BTA determinations for intake technologies under the 316(b) guidelines provide that the cost of alternative technologies not be disproportionate to environmental benefits that would be gained by application of the technology. Each alternative intake technology was evaluated by estimating capital costs, annual operating and maintenance (O&M) costs, and other costs, such as a reduction of generating capacity. The present value (PV) of these costs was calculated so the alternative technologies could be compared on a constant dollar basis. A discount rate of 7 percent and an analysis period of four years were assumed in these calculations.

- **Potential Biological Benefits.** Each technology and operational alternative that would reduce the loss of aquatic organisms and satisfy the proven and available criteria was investigated to determine whether it would reduce SBPP CWIS effects.

- **Other Environmental Impacts.** A key objective is to minimize overall environmental impacts. Elements critical to meeting this objective include technologies and configurations that minimize impacts on the environment and community. Consistent with these objectives, alternative intake technologies were evaluated with respect to noise impact, visual impact, land use requirements, construction impacts, offsite impacts, safety, and waste disposal.

Twenty-nine different alternatives were initially reviewed. **Table 6.1-1** lists alternative CWIS technologies that are evaluated in this section. The table identifies the technologies and options that were determined to be proven and available and that were evaluated in more detail.

Table 6.1-1. Alternatives to minimize adverse environmental impacts.

	Demonstrated Proven, Available, and Technically Feasible for SBPP	Sufficient Time to Design, Permit, and Construct [1]	Other Potential Environmental Impacts	Entrainment Effects	Impingement Effects
Flow Reduction Options					
1. Closed-cycle Cooling Pond	NO. The required 900 acres for pond is not feasible for site.	NO	Increased PM_{10} emissions, brine discharge effects, salt drift on vegetation, and reduced bay view.	Reduced by corresponding cooling water flow reductions up to 90%.	Reduced, but by an unpredictable amount.
2. Mechanical Draft (Wet/Dry) Cooling Towers — utilizing four different water sources					
2a. City Water	NO. Alternative not is considered feasible since it is very unlikely it would meet "best and highest use" test for city water.	NO	Increased PM_{10}, emissions, blowdown discharge effects, noise levels, and reduced bay view.	Reduced by corresponding cooling water flow reductions of 100%.	Reduced by corresponding cooling water flow reductions of 100%.
2b. Seawater	NO. Insurmountable air quality impacts.	NO	Increased PM_{10}, emissions, blowdown discharge effects, noise levels, visible plumes, and reduced bay view.	Reduced by corresponding cooling water flow reductions up to 90%.	Reduced, but by an unpredictable amount.
2c. Reclaimed Wastewater	YES	NO	Increased PM_{10}, emissions, noise levels, visible plumes, and reduced bay view.	Reduced by corresponding cooling water flow reductions of 100%.	Reduced by corresponding cooling water intake reductions of 100%.

Table 6.1-1 (continued). Alternatives to minimize adverse environmental impacts.

	Demonstrated Proven, Available, and Technically Feasible for SBPP	Sufficient Time to Design, Permit, and Construct [(1)]	Other Potential Environmental Impacts	Entrainment Effects	Impingement Effects
2d. Desalinated Water	YES	NO	Increased noise levels, effects of RO brine discharge, visible plumes and reduced bay view.	Reduced by corresponding cooling water flow reductions of 90%.	Reduced, but by an unpredictable amount.
3. Natural Draft Cooling Tower	YES	NO	Extreme visual impact, increased PM_{10}, emissions, discharge effects, noise levels, and visible plumes.	Reduced by corresponding cooling water flow reductions of up to 90%.	Reduced, but by an unpredictable amount.
4. Air-Cooled Condenser	YES	NO	Large visual impact and potential increased offsite noise levels.	Reduced by corresponding cooling water flow reductions of 100%.	
5. Cooling Water Pump Flow Reduction	NO. Inconsistent with current NPDES permit discharge temperature limit.	NO		Reduced in proportion to reduction of cooling water flow.	Reduced, but by an unpredictable amount.

Table 6.1-1 (continued). Alternatives to minimize adverse environmental impacts.

	Demonstrated Proven, Available, and Technically Feasible for SBPP	Sufficient Time to Design, Permit, and Construct (1)	Other Potential Environmental Impacts	Entrainment Effects	Impingement Effects
Location Options					
6. Offshore Intake	YES	NO	Increased disturbance to the ocean and bay bottom community, and navigational hazard.	Possibly lower numbers of entrained organisms but greater numbers of commercial and recreational species.	Significant increases in entrapment and impingement of juvenile and adult fishes.
Design Options—Behavioral Barriers					
7. Chemicals	NO. No known seawater application.	N/A		No change, possible toxicity effects on entrained larval.	
8. Magnetic Field	NO. No known seawater application.	N/A		No change.	
9. Electrical Barrier	NO. No known seawater application.	N/A		No change.	
10. Chains and Cables	NO. No known seawater application.	N/A	Severe biofouling potential.	No change.	
11. Strobe Light	YES	NO	Disturbance to light-sensitive non-target species.	No change.	Reduced, but by an unpredictable amount.
12. Air Bubble Curtain	YES	NO	Disturbance to other non-target species.	No change, possible mechanical damage from bubbles on entrained larval.	Reduced, but by an unpredictable amount.
13. Velocity Gradient	YES	NO	Disturbance to other non-target species.	No change.	Reduced, but by an unpredictable amount.

Table 6.1-1 (continued). Alternatives to minimize adverse environmental impacts.

	Demonstrated Proven, Available, and Technically Feasible for SBPP	Sufficient Time to Design, Permit, and Construct (1)	Other Potential Environmental Impacts	Entrainment Effects	Impingement Effects
14. Sound	YES	YES	Disturbance to sound-sensitive non-target species.	No change.	Reduced, but by an unpredictable amount.
Design Options—Physical Barriers					
15. Media Filter	NO. No known seawater application for SBPP-sized cooling water intake flow.	N/A		Would be reduced up to 100%.	May be reduced 100%.
16. Porous Dike	NO. No known seawater application for SBPP-sized cooling water intake flow.	N/A	Disturbance to the shoreline and ocean bottom, and navigational hazard.	Would be reduced up to 100%.	Impingement of juvenile and adult fishes may be reduced up to 100%.
17. Sand Filter	NO. No known seawater application for SBPP-sized cooling water intake flow.	N/A	Disturbance to the shoreline and ocean bottom.	Entrainment would be reduced 100%.	Impingement of juvenile and adult fishes would be reduced up to 100%.
18. Radial Well	NO. No known seawater application for SBPP-sized cooling water intake flow.	N/A	Disturbance to the shoreline and ocean bottom.	Entrainment would be reduced 100%.	Impingement of larval, juvenile and adult fishes would be reduced 100%.
19. Cylindrical Wedge-Wire Screen	NO	NO	Disturbance to the ocean bottom community.	Entrainment would be reduced by an amount equal to the wedge-wire mesh size.	Impingement of juvenile and adult fishes would be reduced up to 100%.

Table 6.1-1 (continued). Alternatives to minimize adverse environmental impacts.

	Demonstrated Proven, Available, and Technically Feasible for SBPP	Sufficient Time to Design, Permit, and Construct (1)	Other Potential Environmental Impacts	Entrainment Effects	Impingement Effects
20. Stationary Screen	NO	NO	Disturbance to the ocean bottom community.	No change.	
21. Horizontal Traveling Screen	YES	NO	Disturbance to the shoreline and ocean bottom communities.	No change.	Impingement of juvenile and adult fishes might be reduced.
22. Rotary Drum Screens	YES	NO	Disturbance to the ocean bottom community, and navigational hazard.	No change.	Impingement of juvenile and adult fishes might be reduced.
23. Fine-mesh Vertical Traveling Screen	YES	YES		Fewer organisms would be entrained.	Higher numbers of organism would be impinged.
24. Center-flow/Dual-Flow Screen	YES	NO	Disturbance to the ocean bottom community.	No change.	Impingement of juvenile and adult fishes might be reduced.
25. Barrier Net	NO. No known seawater application for SBPP-sized cooling water intake flow.	NO	Disturbance to the ocean bottom community and navigational hazard.	No change.	Impingement of adult fishes would be reduced by an unpredictable amount.
26. Aquatic Filter Barrier	NO. No known seawater application for SBPP-sized cooling water intake flow.	NO	Disturbance to the ocean bottom community and navigational hazard.	Up to 50% fewer organisms would be entrained.	Higher numbers of organism would be impinged.

Table 6.1-1 (continued). Alternatives to minimize adverse environmental impacts.

	Demonstrated Proven, Available, and Technically Feasible for SBPP	Sufficient Time to Design, Permit, and Construct (1)	Other Potential Environmental Impacts	Entrainment Effects	Impingement Effects
Fish Diversion, Collection, and Conveyance Systems					
27. Louver Diversion System	YES	NO	Potential disturbance to the ocean bottom community.	No change.	Impingement of juvenile and adult fish might be reduced.
28. Angled Screens	YES	NO	Potential disturbance to the ocean bottom community.	No change.	Entrapment and impingement of juvenile and adult fish might be reduced.
Miscellaneous					
29. Biofouling Control	YES	N/A		Fewer entrained organisms would be cropped by biofouling organisms.	No change.

(1) Duke Energy leases both the power generating facilities located on the site and the site property from the Port of San Diego under an operating agreement that is due to expire in 2009. Currently Duke has no plans to operate the facility beyond the period of the lease.

6.2 Capacity (Flow) Reduction Options for Minimizing Environmental Impacts

6.2.1 Cooling Systems

In the existing once-through cooling system, circulating seawater absorbs heat from the steam exiting the steam turbine generators for each unit, thereby condensing it. The heated circulating water from the condensers and closed-cooling exchangers is discharged through a channel back to San Diego Bay. In contrast, closed-cycle cooling water systems recirculate cooling water. Heat is rejected to the atmosphere by closed-cycle cooling water systems largely by evaporation, and cooling water that is lost through evaporation must be replenished. Another cooling system, the air-cooled condenser, eliminates the need for closed-cycle cooling water by directly condensing the turbine exhaust steam in a large array of fan-cooled convective heat exchangers.

Four alternative cooling systems were evaluated:

- Closed-cycle cooling pond,
- Mechanical draft (wet) cooling towers, including wet/dry plume abatement towers
- Natural draft cooling tower, and
- Air-cooled condenser.

It is important to note that EPA is not authorized to directly order the installation of cooling towers because, although closely related to the CWIS, cooling towers are not considered part of the CWIS itself. At the same time, however, EPA has also consistently concluded that CWA § 316(b) does authorize EPA to impose a capacity (or flow) limit based on the permittee's ability to meet that limit using technologies, such as cooling towers, that have been determined to be appropriate at the particular plant. Such a limit imposes a performance standard for CWIS capacity, which the permittee may meet in any manner it chooses.

6.2.1.1 Closed-Cycle Cooling Pond

This cooling alternative is a recirculating system based on the use of a man-made evaporative cooling pond (i.e., a shallow reservoir with a large surface area to effectively dissipate heat from the water). Warm water from the steam turbine condensers is discharged to a cooling pond. The water cools as a result of convection and radiation to the surrounding air and is then pumped back to the condenser to repeat the cycle.

Although cooling ponds require the least amount of make-up water of all alternatives, they require a large amount of land. It was previously estimated that a cooling water pond large enough dissipate the rejected heat from SBPP would be approximately 900 acres (SDG&E 1973). The use of this system is normally limited to plant sites with significant amounts of excess acreage. The SBPP site is approximately 150 acres in total area, and the land for a cooling pond is unavailable. This option therefore was eliminated from further consideration.

6.2.1.2 Mechanical Draft (Wet) Cooling Tower

This closed-cycle cooling water alternative is a recirculating cooling water system with mechanical draft or "wet" cooling towers. This design involves pumping recirculating water through the condenser to remove heat from the turbine generator exhaust steam, thereby condensing it. The hot water leaves the condenser and flows to a system of distribution headers located above the heat transfer surfaces (i.e., "fill" sections) in the cooling tower. The water droplets fall through the fill section while fans draw (induce) air upward, causing some of the water to evaporate. Cooling occurs primarily by evaporation and contact cooling of the water by the cooler-induced airflow. The cooled water is collected in a basin, then pumped back through the condenser to repeat the cycle.

This alternative involves consumptive water use. Although it is called a closed system, "make-up" water is needed to replace water that is lost due to evaporation, drift (non-evaporated water droplets entrained in the airflow that exit the tower), and blowdown (cooling water that is bled off to control concentrations of dissolved solids in the water). Four sources of make-up water were evaluated: potable city water, seawater, reclaimed wastewater, and desalinated water. This alternative would reduce the plant's use of San Diego Bay water, thereby reducing both entrainment and impingement effects. Transitory thermal loading to the bay would also be reduced with the use of a cooling tower.

This alternative would add visible structures and would generate additional noise at SBPP. The structure would be approximately 45 to 60 ft tall and would increase the visual mass of the power plant to viewers from all directions, and particularly from the east. A vapor plume would be visible when ambient air conditions promote condensation of moisture in the saturated exhaust air. Therefore, a wet/dry plume abated cooling tower alternative was evaluated.

This system incorporates use of a mechanical draft or wet cooling tower with a dry section at the top of the tower. In the dry section, a portion of the hot recirculating water is routed internally to a series of heat exchangers where the outside surface is exposed to the moisture-laden air from the wet section. These exchangers heat the moisture-laden air (without adding additional moisture) to keep the air/water mixture from becoming supersaturated, thus reducing or eliminating the presence of a visible plume.

The four potential sources of make-up water are evaluated for this alternative in the following paragraphs:

City Water

Units 1-4 would require a wet cooling tower evaporation rate of about 5,000 gpm for cooling purposes at maximum load. The cooling tower would require at least this amount of make-up water regardless of the cycles of concentration in the tower.

The State Water Resources Control Board (SWRCB) and the California Energy Commission (CEC) both endorse policies that discourage the use of local fresh water resources for power plant cooling use. A longstanding SWRCB resolution[1] and CEC licensing procedures[2] state that fresh water should be used for power plant cooling only as a last resort if use of other water sources (such as reclaimed water, ocean water, brackish water, or irrigation runoff) would be "environmentally undesirable or economically unsound." As described below, there are probably other water sources available.

Demands on local water resources by other users and developers in the area effectively preclude supplying the required amount of fresh water cooling tower makeup, up to at least 7 million gallons per day (MGD), in addition to the significant amount of fresh water already supplied to the plant for boiler feed water makeup and other uses. For this reason and due to the state regulatory policy described above, fresh water is not considered further in this analysis.

Seawater

Cooling towers designed for seawater would operate at up to 2 cycles of concentration and would require approximately 10,000 to 15,000 gpm of make-up water. The seawater make-up would be provided by modifying a portion the existing once through cooling system. Cooling tower blowdown would be returned to San Diego Bay through the existing once-through cooling water discharge system. The PM_{10} emissions from seawater cooling towers would be significant, far greater than for potable city water, reclaimed wastewater, or desalinated water. Drift would also lead to increased fine particulate salt emissions from the facility in the form of dissolved solids emitted with the drift droplets. For the seawater towers considered, the estimated additional particulate emissions to the atmosphere associated with drift would be about 1,300 lb/day at maximum operation.[3] This quantity of additional (low elevation) PM_{10} emissions at the facility would likely cause significant, adverse air quality impacts and would be very difficult or impossible to permit by the San Diego Air Pollution Control District. For this reason, the use of seawater for make-up water is eliminated from further consideration.

[1] SWRCB, *Water Quality Control Policy on the Use and Disposal of Inland Waters Used for Power Plant Cooling*, Resolution No. 75-58, June 19, 1975.

[2] California Energy Commission, *Energy Facility Licensing Process, Developers Guide of Practices and Procedures*, Staff Report/Draft, November 2000.

[3] Assuming drift is 0.0005 percent of recirculating water.

Reclaimed Wastewater

Assuming about 4 cycles of concentration for a cooling tower using reclaimed water, about 6,000–7,000 gpm (10 MGD) of makeup water would be required at maximum loads. This make-up water could come from the City of San Diego South Bay wastewater treatment plant located about six miles south of the SBPP, near the Mexican border. This scheme would require a water conveyance pipeline approximately 16–20 inches in diameter and the associated pumps and controls to transport the reclaimed water to SBPP. A second 8–12-inch diameter pipeline might also be installed to convey the cooling tower blowdown (1,500–2,000 gpm) back to the wastewater treatment plant for disposal. The alignment of the pipelines would be underground in public rights-of-way, mainly under public streets and possibly through unavoidable environmentally sensitive areas such as a national wildlife preserve.

Based on a representative, but preliminary, reclaimed water quality analysis, it appears that no supplemental water treatment system would be required at the SBPP site to use reclaimed water for cooling tower makeup. However, the expected composition of the reclaimed water would require the diligent implementation of a carefully designed cooling water chemical conditioning program, particularly for corrosion and biological control.

The technical, economic criteria, and the environmental impacts and benefits of wet/dry closed-cycle cooling alternative using reclaimed wastewater are discussed below.

Technical Criteria
The mechanical draft cooling system could consist of two cooling towers, each comprised of about 14 cells and each tower measuring approximately 600 ft long by 45 ft wide by 60 ft tall.

It is generally desirable to locate a cooling tower as close to the condensers as practical in order to minimize the length of the very large diameter, underground cooling water supply and return lines between the tower and the condensers. These large towers should be oriented with the long axis perpendicular to the prevailing wind direction for good operating characteristics. In addition, locations that are upwind of sensitive equipment or sensitive off-site features, such as the station switchyard or major highways, should be avoided to minimize problems associated with drift droplet deposition and potential fogging. Considering that the prevailing wind direction for the site is from the WNW and the fact that the wet/dry tower design should significantly reduce fogging potential along Interstate 5 just downwind, the preferred location for the two cooling towers will likely be in the former wastewater pond area just east of the powerhouse.

Economic Criteria
Based on an economic study at a similar facility, the estimated total installed capital cost for the cooling towers only, <u>excluding</u> the significant additional costs for the required six-mile, off-site reclaimed water supply and return pipelines, would be approximately \$20–\$30 million. The equivalent amortized cost over the analysis period (four years) would be about \$6–\$9 million per

year. Inclusion of the other required costs (not estimated at this time due to lack of definition) would add significantly to this cost, perhaps doubling it or more.

In addition, the capital cost estimates for this alternative do not include the costs associated with acquisition of additional PM_{10} offsets for cooling tower drift emissions.

Potential Biological Benefits
A 100 percent reduction in entrainment and impingement would occur if the once-through cooling option were replaced by a wet/dry plume abated tower using city-supplied reclaimed wastewater.

Other Environmental Impacts
Based on an assessment of the physical and operating characteristics of a plume abatement wet/dry cooling tower, this cooling alternative may have other environmental impacts compared to a once-through cooling system. Operation of the cooling tower will generate drift, including dissolved salts. Any additional PM_{10} emissions would be mitigated. The cooling tower would also add a noise source to SBPP, but it could be designed to meet applicable noise standards.

Evaluation Summary
Even though a wet-dry mechanical draft closed-cycle cooling system may be feasible assuming the availability of reclaimed wastewater, it is clear that under the scenario of Duke's lease with the Port of San Diego the cost would be wholly disproportionate to any environmental benefit. However, it is very unlikely that evaporating large quantities of potable water to cool a power plant in an area with critical water supplies would be ever be deemed the best and highest use for desalinated water. There is no practical basis for considering this alternative, as discussed in Section 6.1.

Desalinated Water

Make-up water could come from a desalination plant. Assuming 10 cycles of concentration for a cooling tower using 100 percent desalinated water, about 5,600 gpm (8 MGD) of makeup water would be required.

Technical Criteria
For the purpose of this conceptual analysis, it is assumed that a desalination plant would be located within the SBPP site. It would probably consist of seawater filtration, reverse osmosis desalination membranes, filter backwash systems, and ancillary equipment such as pumps, chemical storage tanks, etc. The new plant would require at least one or two acres of space and should be located as near as practical to the existing seawater cooling water intake and discharge system to minimize piping and pumping costs. This facility would be designed to desalinate 8-10 MGD of seawater to acceptable water standards (and would be potentially capable of achieving drinking water standards for other markets if advantageous to Duke or another owner). Existing fuel tanks, which are no longer required for fuel storage, would be converted to storage tanks for the desalinated water.

The existing cooling water intake system could be readily modified to provide seawater feed for a new desalination facility. A seawater feed rate of 16–20 MGD (11,000–14,000 gpm) would be required to produce 8–10 MGD (5,600–7,000 gpm) of desalinated water which could used for cooling tower makeup, boiler feed water makeup, and other fresh water uses both inside and outside the power plant facility. The process would also produce about 8–10 MGD of waste brine, containing about twice the salinity of the seawater feed, which would be disposed through the existing once-through cooling water discharge system.

Economic Criteria
Based on a published study for a 9 MGD desalination plant proposed for a central California location,[4] the estimated total installed capital cost for this alternative is approximately $74 million. The equivalent amortized cost of this capital expenditure only, excluding the very significant cost of power for the facility and other operating and maintenance costs, would be approximately $22 million per year over the analysis period (four years).

Potential Biological Benefits
A 100 percent reduction in entrainment and impingement would occur if the once-through cooling option were replaced by a wet/dry plume abatement tower using desalinated water.

Other Environmental Impacts
Based on an assessment of the physical and operating characteristics of a plume abatement wet/dry cooling tower, this cooling alternative may have other environmental impacts compared to a once-through cooling system. The cooling tower may have an adverse, but not significant, air quality and visual impacts on the surrounding community. The cooling tower would also add a noise source to SBPP, although it could be designed to meet applicable noise standards. Finally, operation of the desalination plant would result in the generation of a waste brine, containing about twice the salinity of seawater, which would probably need to be disposed in San Diego Bay.

Evaluation Summary
Even though a wet-dry mechanical draft closed-cycle cooling system may be feasible assuming the use of desalinated water, it is clear that under the scenario of Duke's short-term lease with the Port of San Diego, the cost would be wholly disproportionate to any environmental benefit gained given. It is also unlikely that evaporating large quantities of potable water to cool a power plant in an area with critical water supplies would ever be deemed the best and highest use for desalinated water. As discussed in the Section 6.1, there is no practical basis for considering this alternative.

6.2.1.3 Natural Draft Cooling Tower

A natural draft cooling tower system is similar in principle to the mechanical draft system described above. The primary difference is that this alternative replaces mechanical fans to

[4] California PUC, *Carmel River Dam Contingency Plan, Plan B,* 2002 (costs for off-site conveyance and storage omitted).

move the cooling air with what is essentially an enormous chimney. Air is drawn in at the base of the tower as warmer, more buoyant air exits the top of the tower. This circulating air contacts the returned cooling water inside the tower and cools it, mainly by evaporation. The cooling water recirculation, blowdown, makeup rates, and water quality issues would be similar to the mechanical draft system described above.

Most of the potential negative impacts described for the mechanical draft cooling towers would also be associated with a natural draft cooling tower for the SBPP. The blowdown discharge to San Diego Bay would be the same. Drift and the resulting particulate PM_{10} emissions would also occur, although at somewhat reduced rates. Visible condensate plumes would also periodically occur at the top of the tower. However, the most significant environmental impact on the surrounding area would be the visual impact of the enormous tower. It was previously estimated that a natural draft tower designed for SBPP would be 450 ft high and 395 ft in diameter at the base (SDG&E 1973). This technology is proven and available for some facilities, especially those remote from residential areas, but it is very undesirable at the SBPP facility due to visual impacts; it was therefore eliminated from further consideration.

6.2.1.4 Air-Cooled Condenser

The air-cooled condenser (ACC) is an array of fan-cooled heat exchangers that use ambient air to remove heat from the turbine generator exhaust steam. The air-cooled heat exchangers remove heat and thus condense steam by drawing ambient air upwards from below the condenser across exterior finned cooling tubes. An ACC with 50 cells and overall dimensions of 460 ft by 220 ft (101,000 square ft) by at least 100 ft high was evaluated for this analysis.

Because the ACC has such a large footprint, there are limited scenarios for its location. Due to the very large diameter ducts required to transfer the steam from each turbine exhaust to the ACC, it is desirable to locate the ACC as close as practical to the steam turbines. However the necessary space on the north side of the powerhouse building adjacent to the steam turbine generators is not available. The empty area on the north side between Units 2 and 3, about 60 ft by 100 ft, is much too small. The remaining area on that side of the building is occupied by the station switchyard. Conceivably, it may be possible to locate an ACC in the former wastewater pond area to the east of the powerhouse, but the very long steam ducts for this option would significantly add to the cost of the system. In addition, because of the size of the ACC, it would have significant and unacceptable visual impacts.

From a technical standpoint, use of an ACC provides the worst performance (lowest MW output, highest turbine back-pressure, and highest heat rate) of all cooling system alternatives considered. This degraded efficiency is made even worse by the additional steam turbine back-pressure that would be caused by the extra-long steam ducts due to the relatively remote location of the ACC. It has the highest capital cost, and, except for the cooling pond, and would require the most space on the site. While this technology is proven and available for some facilities, it is technically impractical at SBPP, and would have significant environmental impacts; therefore, it was eliminated from further consideration.

6.2.2 Circulating Water Pump Flow Reduction

Once-through cooling systems typically rely on two main circulating water pumps to deliver cooling water to a single steam turbine condenser. During normal plant operation both circulating water pumps will be in service. The pumps are sized to deliver sufficient cooling water flow at maximum turbine load so that the condenser will operate at its design performance level and the cooling water discharge will not exceed a maximum permitted temperature. When cooling water flow is reduced or restricted without also reducing the unit load (i.e., quantity of turbine exhaust steam), the discharge temperature will rise to discharge temperature limits requiring unit load reduction.

Seasonal curtailment of energy production is strongly influenced by uncertainties associated with generation requirements. SBPP is under the control of the California Independent System Operator (ISO), and all SBPP units are currently designated Reliability Must Run (RMR) units. This means that the ISO determines which units operate at what loads. Current SBPP practice is to shut down unneeded circulating water pumps within the constraints of ISO dispatch instructions. Seasonal curtailment of power generation as a method of reducing entrainment and impingement losses for SBPP is also not considered a feasible alternative from the ISO perspective.

6.3 Intake Location Options for Minimizing Adverse Environmental Impacts

6.3.1 Intake Location

The SBPP currently withdraws cooling water from an intake channel area that extends into the southeastern margin of San Diego Bay. The intake channel has a bottom width of 61 m (200 ft) at its widest point and tapers to 15 m (50 ft) near the Unit 4 screenhouse. The bottom depth of the channel is approximately –5.4 m (–17.7 ft) MLLW.

6.3.1.1 Offshore Intake Location

An offshore intake alternative design was evaluated that includes the placement of two underground intake pipes that would extend from the existing CWIS under San Diego Bay and Silver Strand Beach and into the Pacific Ocean, a total distance of about 4,500 ft. Two intake pipes would extend from the beach approximately 450 ft into the Pacific Ocean, with each terminus fitted with a submerged velocity cap sized for the an appropriate horizontal intake velocity that would be protective of aquatic organisms. The depth of the ocean water at the velocity cap location would be approximately 30 ft. The submerged velocity caps would be designed to minimize entrapment of organisms by creating horizontal flow and limiting the intake velocity to a suitably low value. The top of each velocity cap would be designed to be at least 20 ft below the ocean's lowest low water level. The velocity cap opening for the cooling water intake would be located about 10 ft above the ocean floor to avoid drawing mud and debris into the system.

Two cooling water intake pipes of approximately 10–12 ft in diameter would be required to provide the necessary cooling water intake water flow and to enable periodic heat treatment to remove accumulated biologic growth from the inside of the pipes.

Technical Criteria

The offshore intake pipe design supported on the ocean bottom would probably utilize large reinforced concrete support pads with pipe support saddles on the ocean floor. The purpose of each pad is to restrain, support, and distribute the load of the intake pipes on the bottom of the ocean without substantially disturbing the ocean sediment. Construction of the offshore, underground lines in this relatively shallow portion of the bay would probably be accomplished by driving sheet piles and dewatering, excavating, and backfilling sequentially along multiple segments of the route.

Economic Criteria

Due to the very speculative nature of this alternative, a specific estimated capital cost for the offshore intake alternative design has not been performed. Given the exceedingly high cost of in-water construction, the extensive environmental mitigation measures likely to be required, and the enormous task of obtaining preconstruction approvals and permits, the total capital cost is surely in the many tens of millions of dollars, and may be more than $100 million.

Potential Biological Benefits

Offshore intakes will always have higher intake velocities than typical shoreline intakes since the same amount of cooling water must be withdrawn through the smaller cross-sectional area of an offshore conduit compared to much larger cross-sectional areas of shoreline intakes. The higher intake velocities associated with offshore intakes lead to higher impingement rates. Existing offshore intakes located along California's ocean coastline typically terminate as a vertical riser in 30–50 ft of water. Since the same volume of SBPP intake water would have to pass through a significantly smaller opening compared to the existing shoreline intake structure, intake velocities would be several times higher than the velocity of the existing CWIS.

Commercially or recreationally important species comprised less than one percent of the total number of larval fishes collected at the shoreline entrainment station in 2001 and 2003. It is possible that risk of entraining larvae of commercially and recreationally important species such as sea bass would increase through use of an offshore intake.

Many of the dominant groups of fishes and invertebrates (gobies, flounder, crabs, and shrimps) would typically be found in or near the bottom habitat in the vicinity of the offshore site. Pelagic fish species, such as anchovy, are commonly found in large schools moving through the water column. Submerged offshore intakes have higher approach velocities than shoreline systems and use conduits within which fishes can become entrapped, resulting in an increase in the number of organisms impinged.

Further, there is a distinct possibility that the physical presence and nature of an offshore intake would attract many of the fishes and invertebrates inhabiting this part of the Pacific Ocean, and would cause higher entrapment and impingement rates than those at the proposed onshore intake.

Other Environmental Impacts

The physical configuration and operating characteristics of the offshore intake pipes would have the following additional adverse impacts compared to the existing shoreline intake structure:

- Installation of the new cooling water intake lines would require substantial submarine excavation, of tens of thousands of cubic yards of the ocean and bay bottom.

- Significant disruption of recreational and commercial use of the bay during the construction period.

- Offshore intake pipes would become fouled with mussels, barnacles, and marine growth and could require the addition of a chemical feed system and/or warm water recirculation system as well as periodic mechanical cleaning to control such marine organisms.

Evaluation Summary

The offshore alternative evaluated is not BTA when compared to the existing CWIS. It does not provide biological benefits and has very substantial increased costs. The installation of intake pipes is not considered feasible due to the extensive bayfloor and seafloor disruption that would be required. The offshore alternatives would not constitute an improvement to the existing shoreline CWIS. There is no evidence that an offshore intake would reduce entrainment of fishes and invertebrates compared to the existing shoreline intake; there is clear evidence the numbers of entrapped and impinged fishes and invertebrates would be significantly greater by orders of magnitude than at the existing intake.

Section 6.1 discusses the impracticality of this alternative and its low cost-benefit potential.

6.4 Design Options for Minimizing Adverse Environmental Impacts

6.4.1 Behavioral Barriers

Behavioral technologies are still considered experimental by many regulatory and resource management agencies despite numerous studies involving existing devices and new technologies (EPRI 1999). Devices such as chemical injectors, magnetic fields, electric barriers, and hanging chains and cables have received national attention as fish protection measures at various intake facilities, but mostly for freshwater applications. These alternatives are sensitive to corrosion and biofouling in the marine environment and would therefore not be suitable for San Diego Bay. Strobe lights have been used effectively to repel several fish species under laboratory conditions and at water intake facilities; however, they have not been proven effective for reducing impingement of marine fishes (Brown 1999). Air bubble curtains generally have been ineffective in diverting or blocking fishes in a variety of field conditions (EPRI 1999). Limited practical applications of the use of velocity gradients and sound have been developed and research is continuing. Eight different behavioral technologies were evaluated. Of these only sound has been recently proven for a number of similar locations for impinged species. Though sound technology is somewhat still experimental in nature, it was evaluated at the second step.

Behavioral technologies have received considerable attention, particularly over the past ten years. Behavioral barrier technology is limited to reducing the entrapment and impingement of juvenile and adult fishes, but it is not generally suitable for macroinvertebrates. Because organisms susceptible to entrainment through the 3/8-inch mesh of the existing intake would be small in size and would possess limited swimming ability, behavioral barriers would not reduce entrainment of any life stages of any organisms.

6.4.1.1 Sound

Results of recent studies have suggested the potential for infrasound sources to repel fishes effectively. A response in fishes close to a sound source is probably related more to particle motion than acoustic pressure. Particle motion is very pronounced in the near field of a sound source and is most likely the major component of what fishes sense from infrasound (frequencies less than 50 Hz). In the first practical application of infrasound for repelling fishes, Knudsen et al. (1992, 1994) found a piston-type particle motion generator operating at 10 Hz to be effective in repelling Atlantic salmon smolts in a tank and in a small diversion channel. Following the success of Knudsen et al., there was a general belief in the scientific community that infrasound could be an effective fish repellent since there was a physiological basis for understanding the response of fishes to particle motion.

During the last decade, behavioral guidance systems have come to the forefront of fish passage research (Popper and Carlson 1998). Stimuli such as light, sound, and electric shock have been used to elicit avoidance responses from fishes in an attempt to repel them from structures such as the cooling water intake pipes at power plants or the turbine intakes at hydroelectric generating stations. Of these technologies, high-frequency sound (or ultrasound) has been demonstrated to elicit avoidance responses in alewife *Alosa pseudoharengus* (Dunning et al. 1992), blueback herring *A. aestivalis* (Nestler et al. 1992), and American shad *A. sapidissima* (Mann et al. 1997) in tanks and enclosures. The results of these experiments have led to the development of fish deterrent and guidance systems that have subsequently been tested at several electric generating stations with varying degrees of success (Nestler et al. 1992, Ross et al. 1993, 1996, Popper 1998).

The application of sound to control fish behavior was reviewed by Popper and Carlson (1998). They found that ultrasound is more effective in guiding pelagic fishes (such as anchovy) than demersal fishes (such as gobies or California halibut). There is a good potential that the use of ultrasound would reduce the impingement of pelagic anchovy by deterring their presence in the area of an intake, but would have little if any effect on the impingement of demersal fishes.

The focus of recent fish protection studies involving underwater sound technologies has been on the use of new types of low- and high-frequency acoustic systems that were not previously available for commercial use. High-frequency sound (120 kHz or more) effectively and repeatedly repels members of the genus *Alosa* (American shad, alewife, and blueback herring) at sites throughout the U.S. (Ploskey et al. 1995, Dunning 1995, Con Ed 1994). Only one thermal power plant, the James A. Fitzpatrick Nuclear Power Plant, which is in a freshwater environment, has installed a sound system intended to reduce impingement, specifically impingement of alewife (EPRI 1999). Other studies have not shown sound to be consistently effective in repelling species such as largemouth bass, smallmouth bass, yellow perch, walleye, rainbow trout (EPRI 1998), gizzard shad, Atlantic herring, and bay anchovy (Con Ed 1994). Given the species-specific responses to different frequencies and the variable results that often have been produced, additional site-specific, species-specific, and species lifestage-specific research would be required to evaluate the potential usefulness of the technology at a specific intake system.

Technical Criteria

The potential for sound technology to reduce impingement at the SBPP intakes could be determined from results of hydroacoustic studies conducted in the intake area. The hydroacoustic surveys could be designed to also provide site-specific information from the intake area on the potential effectiveness of sound.

Economic Criteria

An sound system is a relatively low cost technology to reduce impingement. An sound system's lack of moving parts makes it generally maintenance free, except for inspections and cleaning of the sound transducer surfaces. System performance can be monitored automatically via underwater receivers and bioacoustical instrumentation.

Potential Biological Benefits

Sound technology has the potential to reduce impingement at the SBPP intakes.

Other Environmental Impacts

None are known at the present time. The sound frequencies would be carefully tuned to avoid any disturbance to sensitive receptors in the immediate area, such as marine mammals.

Evaluation Summary

Sound technology should be explored further as a technology to reduce the SBPP intake's impingement of fishes. Several studies and a growing body of research suggest that a properly designed sound system could reduce SBPP's potential to impinge some pelagic fish species.

6.4.2 Physical Barriers

Several physical barrier technologies are available but are still developmental. Media filters, porous dikes, sand filters, and radial well intakes have never been used to provide power plant cooling water from a marine source. Debris accumulation, biofouling, and sedimentation are major constraints in using media filters in the marine environment. Porous dikes may be effective in preventing the passage of juvenile and adult fishes based on the results of small-scale pilot studies and laboratory testing. However, entrainable organisms will generally be trapped in the porous medium or entrained into the pump flow. No recent research has been performed with porous dikes, sand filters, or other forms of media filter intakes. No practical way to apply media filters to CWISs has been identified, and the status of these technologies is unlikely to change in the future (EPRI 1999). In the absence of demonstrated performance capabilities and operational reliability, porous dikes, sand filters, and other media filters are not considered proven or available technology for the SBPP.

Adjustable vertical barriers are used to redirect intake flows to reduce entrainment rates by selecting a level of the water column for withdrawal that has relatively lower concentrations of larvae or other organisms. There is no clear evidence that an adjustable vertical barrier could reduce entrainment rates because the concentrations of larvae in the shallow area of the intake are relatively uniform throughout the water column. Moreover, this design has not been demonstrated successfully in sites comparable to San Diego Bay. Therefore, the adjustable vertical barrier alternative was eliminated from additional consideration of intake alternatives.

Cylindrical wedge-wire screens with 0.5- to 2.0-mm bar spacing to protect early life stages (eggs and early larvae) of fishes from entrainment have not been evaluated at a large-scale CWIS sited in a marine environment where biofouling organisms exist (EPRI 1999). Therefore, cylindrical wedge-wire screens are not proven and effective in the marine environment of SBPP's CWIS.

Stationary screens have had little application. No information was found on recent advances or installations of flat-panel screens for use as fish barriers. Flat panel screens also require a much larger surface area than do conventional traveling screens for the passage of the same volume of water. Use of these screens for cooling water intakes is precluded except for small-volume intakes where the space is available and the screens can be maintained in a clean condition to minimize head loss.

The horizontal traveling screen combines elements of both diversion and collection devices. Years of design, research, and development efforts at two sites have demonstrated its lack of operational reliability (EPRI 1999).

Rotary drum screens are often considered as technologies for protecting fishes in freshwater environments, but they have never been used in a marine environment. A constant water elevation is required for effective drum screen operation. This intake technology would be infeasible in the tidally-influenced San Diego Bay.

Thirteen different physical barrier screen technologies to reduce entrainment and impingement were evaluated, and of these, four of the technologies were determined to be proven and available. The screening technologies that are evaluated include vertical traveling screens, center- and dual-flow screens, barrier nets, and the new fine-mesh aquatic filter barrier.

6.4.2.1 Fine-mesh Vertical Traveling Screens with Improved Fish Return System

The fine-mesh vertical traveling screen is a physical barrier that employs a screen to prohibit passage of all but the smallest organisms into the CWIS. As the size of the openings in the screen mesh is reduced, entrainment of organisms is also reduced, but some impingement does occur. The screen is configured as a series of vertically rotating panels that are periodically cleaned to remove impinged organisms and debris. The cooling water approach velocity has an effect on impingement and entrainment. Approach velocity can be significantly reduced by increasing the number of screens and increasing the intake opening size. Many California power plants currently have intakes equipped with vertical traveling screens. The California Energy Commission and the RWQCB recently approved and deemed BTA a new intake design at the Moss Landing Power Plant that is equipped with vertical traveling screens.

Technical Criteria

This alternative considers replacing the existing vertical traveling screens with new fine mesh (5/32-in) traveling screens, adding 15–30 percent additional cross–sectional screen area[5] to Units 1 & 2, and Units 3 & 4 screenhouse structures, and improving the existing fish return system. The total number of screens must increase in order to ensure that the through-screen velocity through the proposed smaller-sized mesh remains the same as currently experienced with the larger mesh screens. Currently there are a total of eight traveling screens for Units 1-4 (two screens per unit). A vertical fine-mesh traveling screen of the type shown on **Figure 6.4-1** could be installed at SBPP.

The screen design would include a primary low-pressure seawater spray system designed to gently dislodge impinged organisms and entangling debris before the secondary high-pressure spray system removes the remaining impinged debris. The system would be designed to operate continuously to return impinged organisms to the bay quickly and in good condition. The improved spray wash system will increase the efficiency of the overall removal of debris from the screen surface and intake well, reducing the potential for organism entanglement and maintaining low screen approach velocities. Reductions in the amount of debris immediately in front of the intake will lower the potential for entanglement and impingement of organisms such as fishes, crabs, and shrimps. Lower screen approach velocities are also expected to reduce the potential for impingement of weak or entangled organisms by reducing the amount of energy needed by these organisms to move away from the intake facility.

The existing fish return system could be improved by enclosing the return trough to prevent bird predation and extending the terminus of the trough into deeper water.

Economic Criteria

Based on similar modifications installed at another power plant CWIS, the cost of replacing the existing traveling screens with new fine-mesh traveling screens, purchasing additional screens, constructing the structures necessary to house the screens and improving the fish return system is estimated to be in the range of $6–8 million, or about $1.8–2.4 million per year amortized capital cost.

Potential Biological Benefits

Entrainment of larger fishes would be reduced by replacing the existing traveling screens with screens equipped with fine mesh. Impingement of larger larvae on the fine-mesh is expected to increase by an undetermined amount.

[5] Typical range in open area loss in changing from 3/8” mesh to 5/32” mesh screen material, as estimated by major traveling screen supplier.

Other Environmental Impacts

Additional fine mesh traveling screens will be required to maintain the existing velocities. These new screens require larger structures than the existing intake structures. This expansion would increase impacts associated with the necessary construction activities and increase the amount of fill of the bay. Disturbance of the benthic community would occur as a result of intake construction activities.

Evaluation Summary

There is no practical basis for implementing this alternative and it would most likely fail any reasonable cost-benefit analysis (see Section 6.1).

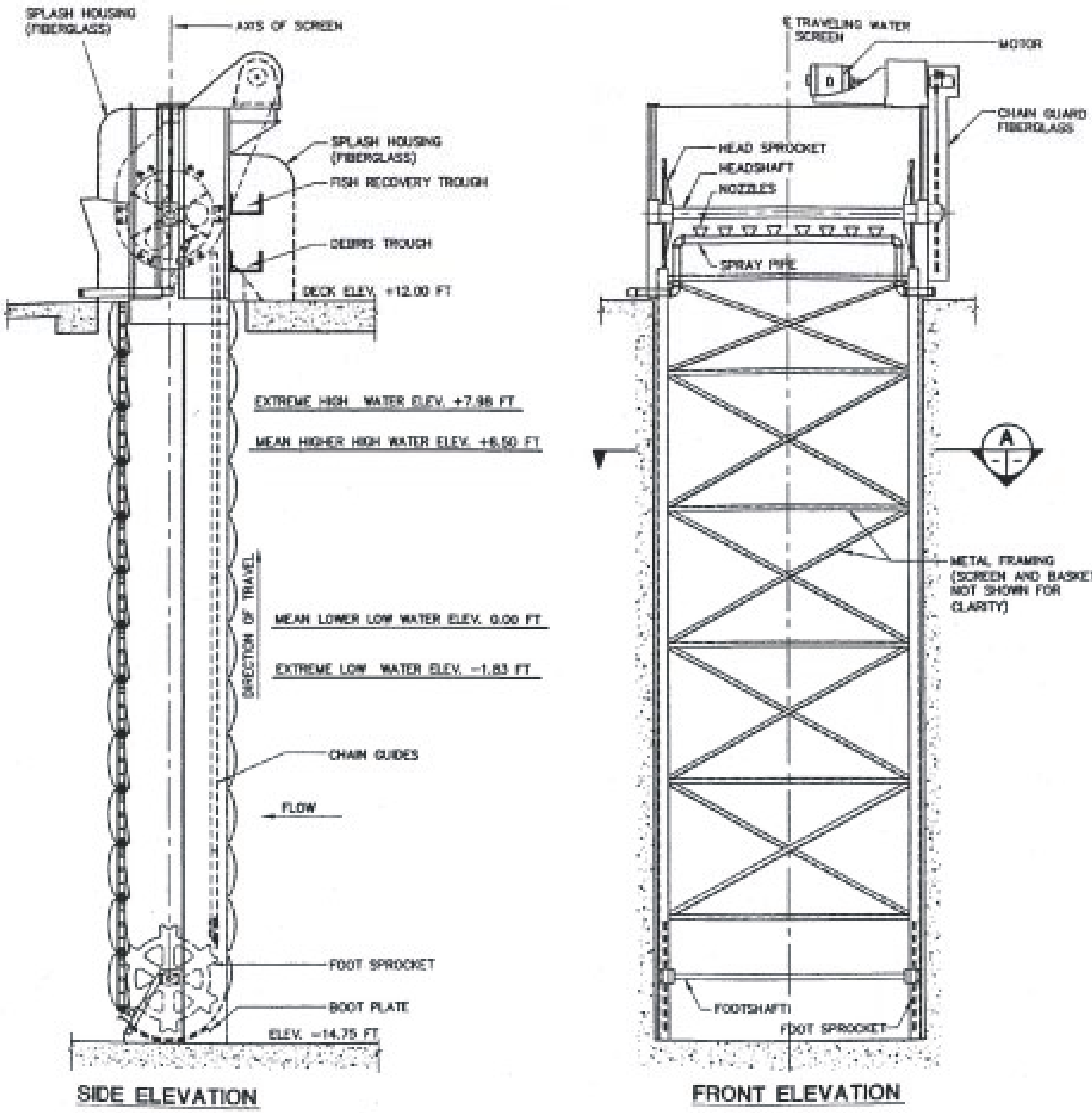


Figure 6.4-1. Example of fine-mesh vertical traveling screen system.

6.4.2.2 Center-flow and Dual-flow Screens

Center-flow and dual-flow screens use fine-mesh screens to reduce impingement. The center flow screen is designed to pass the water through the center passage (between the screens) with the water exiting on both sides of the screen conveyor (**Figure 6.4-2**). Because the screens are equipped with fine mesh, the screen surface dimensions are increased to control water velocity through the screen.

The dual-flow screen design concept is the same as a center-flow design, except that water enters from both screens into the center passage. These two designs allow the use of a finer mesh material without increasing through-screen velocity. Both concepts are used in conjunction with fish return conveyance systems. The screen is positioned so the fishes, macroinvertebrates, and debris are trapped in the direction of the flow. Wall-mounted structural components guide the screen trays and baskets. Low-pressure spray nozzles are used to dislodge fishes, macroinvertebrates, and debris into a trough or holding tank for return to the source water body.

Technical Criteria

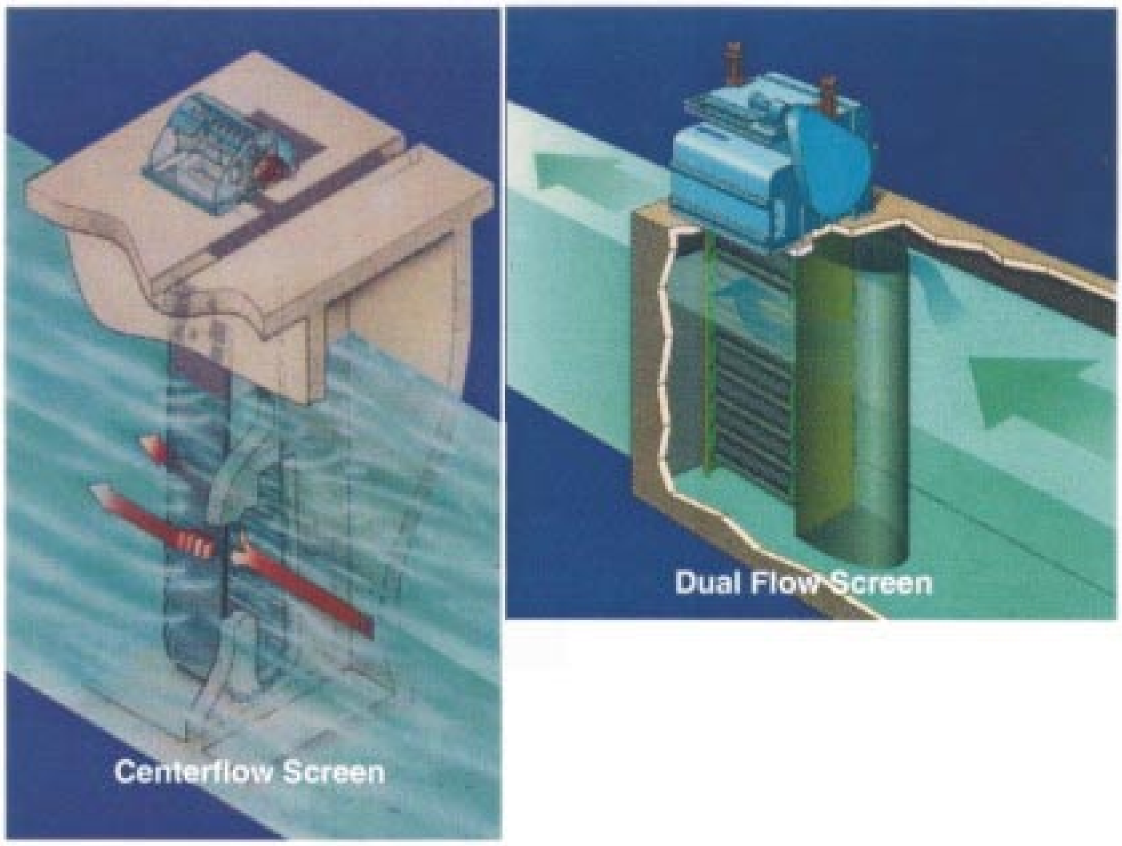
Because of their orientation to the current in an intake structure, center- or dual-flow screens require a larger intake structure that would project farther out into the bay than the existing shoreline configuration with vertical traveling screens.

Economic Criteria

Detailed economic criteria for this alternative were not evaluated because it would not further reduce impingement effects of the proposed CWIS, as discussed below.

Potential Biological Benefits

The biological effectiveness of center- and dual-flow screen systems was evaluated in experiments at the Barney M. Davis Power Station at Laguna Madre, Texas (Murray and Jinnette 1978), and a dual-flow system was evaluated at the Roseton Generating Station on the Hudson River in New York. The overall survival rate for all impinged organisms was 86 percent at the Davis Station using center-flow fine-mesh screens. The impingement survival rate was higher than that found with conventional screens at the Roseton Station. Flow velocities through the fine mesh at the Davis Station ranged from 1.7–3.1 fps; the flow velocity at Roseton was 0.75 fps. At the Davis Station, impingement survival was approximately 95 percent for menhaden, which accounted for 33 percent of the number of fish impinged. Increased impingement mortality was observed during times of increased debris loading.



Source:
US Filter

Figure 6.4-2. Center-flow/dual-flow screens.

Other Environmental Impacts

Center- and dual-flow screens require a larger intake structure than the existing intake structure that contains the vertical traveling screens. Since additional space required for a larger structure is constrained to the landward side of the existing site, the enlarged structure would need to expand into the San Diego Bay. This expansion would increase impacts associated with the necessary construction activities and would increase the amount of fill of the bay.

Evaluation Summary

In the absence of a demonstrated potential for long-term survival for impinged ichthyoplankton, center- and dual-flow screens do not offer alternative intake technology to reduce the combined entrainment and impingement losses. Insufficient data preclude a detailed comparison of the potential survival of early life stages of fishes impinged on center- and dual-flow screens

As discussed in the Section 6.1, there is no practical basis for implementing this alternative nor would the alternative be expected to pass a cost to environmental benefit analysis.

6.4.2.3 Barrier Net

A barrier net is a large fish net equipped with float lines and anchor lines strategically located to deflect aquatic organisms from an intake structure, thereby reducing entrainment and impingement. The design and location of barrier nets are site-specific and take into consideration the characteristics of local fish populations and concentrations of debris. The net mesh size is selected to prevent fishes from passing into the intake without entrapping them. Barrier nets have been effective in reducing impingement rates at several power plants that have long intake canals leading to the cooling water intake pumps.

Technical Criteria

Barrier nets are a viable option for protecting some fish species from entrapment and impingement where water velocity is relatively low (generally less than 1 fps), debris loading is light, and where the organisms are of a relatively uniform size. The size of net mesh is specific to certain sizes and species of fishes.

The tidal current speeds, moderate debris loads, and wide variety in the size of organisms found at the SBPP site do not provide ideal conditions for a barrier net application. Moreover, studies report significant problems with biofouling buildup. Maintaining clean surfaces has been so difficult in freshwater that one of the common problems with barrier nets is that they sink because of biofouling or have to be removed for cleaning. Biofouling is even more difficult to control in the marine environment, and barrier nets could conceivably fail within a matter of a few months.

Economic Criteria

Based on recent studies a similar plant, the capital cost of a barrier net was estimated to be in the range of $2 million (equivalent to amortized capital of $0.6 million per year).

Potential Biological Benefits

Given the proper hydraulic conditions (primarily low velocity) and located in areas without heavy debris loading, barrier nets have been effective in preventing fishes from entering water intake canals. Several barrier nets located in the Midwestern U.S. have been studied (Michaud and Taft 1999). At the Ludington Pumped Storage Plant on Lake Michigan, a 2.5-mile-long barrier net set around the intake jetties successfully reduced the impingement of all the fish species found in the vicinity of the intake (Reider et al. 1997). The net was first deployed in 1989, and the original design was modified to be 96 percent effective for four species (yellow perch, rainbow smelt, alewife, and chub.)

The Chalk Point Station on the Patuxent River used a two-barrier net system located at the mouth of the intake canal (Loos 1986). The outermost net (1.25-inch stretch mesh) trapped most of the debris and jellyfish, while a finer mesh (0.75-inch stretch mesh) inner net prevented impingement of smaller marine organisms (**Figure 6.4-3**). Modifications of the original system increased its effectiveness and achieved an 84 percent reduction in impingement of crabs.

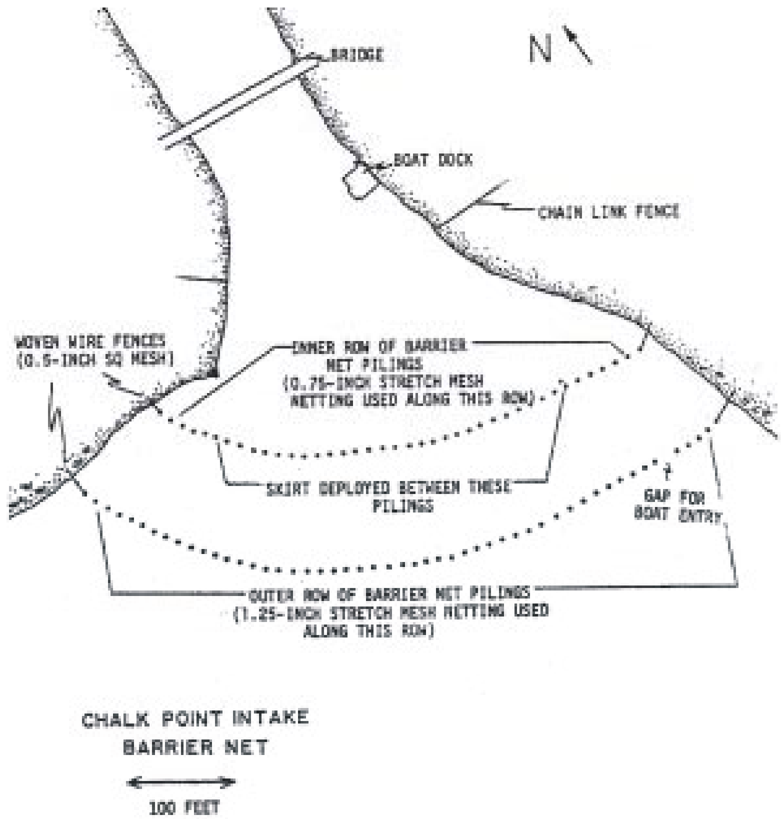
Other Environmental Impacts

A barrier net at the SBPP CWIS could entangle and possibly kill sea turtles, marine birds, and marine mammals.

Evaluation Summary

Barrier nets are most effective when used to block fishes from entering intake canals. It is possible that a barrier net might be effective at reducing impingement; however, it would also increase the risk of injury or death to large numbers of marine birds, sea turtles, and some marine mammals that are at risk of entanglement and death. Barrier nets do not represent BTA for the SBPP intake and were eliminated from further consideration.

This alternative cannot be practically implemented, as discussed in Section 6.1, in the time frame of the remaining term of Duke's lease.



Source:
EPRI, 1999

Figure 6.4-3. Chalk Point barrier net configuration.

6.4.2.4 Aquatic Filter Barrier

An aquatic filter barrier (AFB) is a type of physical barrier to block and divert organisms that might be entrained or impinged in a CWIS. The mesh size of a filter barrier is small enough to screen out small planktonic eggs and larval organisms that are susceptible to entrainment. The AFB eliminates impingement losses of juvenile and adult fishes and can dramatically reduce entrainment losses. The net is deployed in front of the intake and is designed to have a large screening surface area so that through-velocities are extremely low. The net requires some sweeping flow along its surface and an air burst system to keep it clean. A very fine-mesh filter net of polyester fiber strands pressed into a water-permeable fabric mat net is currently being manufactured and tested by Gunderboom, Inc.

Technical Criteria

A successful AFB design must integrate the biological characteristics of the target organisms with engineering parameters, site constraints, and plant operating characteristics. Functional design considerations for SBPP include the exclusion of larvae in the size range of 1–3 mm from maximum intake flows of 417,400 gallons per minute (gpm). The semi-porous barrier material is manufactured with appropriate diameter perforations to meet particle size filter specification and in lengths and widths of sufficient surface area to allow its use at the plant intake. The screens are typically designed to allow filter flows of 10 gpm/square ft. An AFB designed specifically for SBPP would be more than 40,000 square feet in cross-sectional area. Bottom depths of the installation area generally determine the AFB's dimensions. The SBPP AFB would have a length of more than 4,000 ft assuming an average depth of 10 ft in an installation area across the front of the power plant site. Configuration of an AFB of sufficient length 4,000 ft or more) is highly constrained by the length of the power plant site waterfront on the intake side of the jetty separating the intake and discharge channels. This distance is about 1,000 ft. Given this limitation, the AFB would need to extend 500 to 1,000 ft or more offshore and would enclose a relatively large area that would be restricted from marine uses and present a barrier to vessel movement.

Economic Criteria

The cost of an AFB for the SBPP would depend upon a number of environmental design factors, including the filter pore size chosen to most effectively protect the area's marine life. Recent installations at other sites have required capital investments in the range of $4–$6 million. Annual Operations and Maintenance (O&M) costs would depend to a large extent on site conditions, and no reliable estimates have been made.

Potential Biological Benefits

The biological effectiveness of the barrier net to reduce entrainment is a direct function of the barrier's pore diameters. If the net is functioning, there is no doubt of its effectiveness. However, maintaining an AFB in good operating condition has proven to be a very difficult challenge. In 1993, Orange and Rockland Utilities, Inc. became one of the first utilities to experiment with filter barriers by installing a 3.0-mm, mesh net at its Bowline Point Generating Station on the Hudson River (LMS 1996a). Fine suspended silt caused the net to clog and sink in 1993, and efforts in 1994 to spray clean the nets could not control fouling by the alga *Ectocarpus* spp. In both years, source water abundance of the target ichthyoplankton species, bay anchovy, was too low to determine the biological effectiveness of the net, and fouling finally caused the physical failure of the net's piling supports. However, the researchers concluded that with further development, a fine mesh barrier might yet prove effective in preventing entrainment at Bowline Point.

Beginning in 1995, Orange and Rockland Utilities, Inc. sponsored an evaluation of a new generation of filter barrier net manufactured by Gunderboom, Inc. The filter barrier net, consisting of the polyester fiber strands described above, was tested to determine its ability to minimize ichthyoplankton entrainment at the Lovett Generating Station on the Hudson River (LMS 1996b, 1997, and 1998; ASA 1999). Difficulties in keeping the boom deployed and a lack of adequate cleaning techniques were reported in 1995–1997 studies. The filter barrier was redesigned to address these problems. A computer-controlled air sparging system was added to continuously remove the buildup of silt in the fabric's mesh; this may have resolved cleaning and reliability problems at this site. The 1998 study results showed a large reduction in entrainment and demonstrated reliable operation following barrier modification. At this time, the Gunderboom systems are being evaluated for a number of power plant cooling systems, including Lovett.

Other Environmental Impacts

The installation of an AFB at the SBPP could potentially interfere with boat traffic.

Evaluation Summary

The installation of an AFB for the SBPP CWIS would reduce the entrainment and impingement effects. However, SBPP CWIS site lacks the sweeping flows necessary for this alternative to be effective. Furthermore, given the recent biofouling problems and the large enclosed surface area required to configure a suitable AFB, installation of an AFB would be operationally unreliable. The use of AFB technology at the SBPP would not be BTA and was not included any further consideration of alternatives.

Furthermore, the cost an AFB would be wholly disproportionate to any environmental benefit gained given that Duke's short-term lease with the Port of San Diego expires in 2009. Section 6.1 explains why in general terms this alternative cannot be implemented.

6.4.3 Fish Diversion, Collection, and Conveyance System Alternatives

A fish collection system is not an alternative to the previously evaluated physical barrier technologies. Rather, its purpose is to supplement and enhance the performance of screen barriers.

6.4.3.1 Louver Diversion System

A louver diversion system consists of an array of evenly spaced, vertical panels in the area of the intake angled to create flows that lead juvenile and adult fishes away from the intake; louvers are not effective in reducing entrainment. The design of the system considers both the approach velocity of flow to the louvers and the swimming speed of the fishes. The purpose of the louvers is to create directional flow that will stimulate fishes to avoid the intake. A louver system's effectiveness is highly dependent on the characteristics and lifestages of fish species and, to a lesser degree, on site specifics.

Most environmental regulatory agencies find louvers generally less acceptable than a number of more effective fish protection systems. The louver system has been applied in rivers to effectively divert migrating fishes. Studies of these louver applications have been reported to be 80–95 percent effective over a wide variety of species and range of conditions (EPRI 1986, 1994). Louver arrays require sweeping flows. Louver systems have also proven effective in collecting fishes inside an onshore intake screen that were entrapped by an offshore submerged intake.

Technical Criteria

Most existing louver applications have been located in rivers to protect migratory fish species such as salmon. Since louver systems do not provide a barrier to debris that could block the power plant's condenser tube system, a screen is also required for power plant CWIS operation.

Economic Criteria

Conceptual designs of louver diversion systems for the SBPP intake were not available and consequently costs could not be estimated. The use of louver technology is not sufficiently developed at this time to venture a conceptual design.

Potential Biological Benefits

Northeast Utilities Service Company studied the use of louvers to divert juvenile and adult clupeids and Atlantic salmon smolts on the Connecticut River. They found that 76 percent of marked and 86 percent of unmarked shad and other clupeids were guided to bypass facilities (Harza and RMC 1992, 1993; Stira and Robinson 1997). The results of a separate experiment using Atlantic salmon smolts indicated that 85 to 90 percent of the individuals were guided to a bypass (Harza and RMC 1992). The configuration of this system is shown on **Figure 6.4-4**.

Other Environmental Impacts

Louvers constructed into the area in front of the existing intake would temporarily disturb the bay's benthic community. Any such disturbance would be short-term, and its own recruitment and recovery processes would rapidly restore the benthic community.

Evaluation Summary

The alternative would not be effective for SBPP's CWIS since it only addresses impingement impacts, which, based on data presented in Sections 4.0 and 5.0, are not a significant issue. Louvers are not considered a proven alternative intake technology for SBPP. The cost-benefit of the modification was not able to be determined. Therefore, this alternative was eliminated from further consideration and evaluation.

See Section 6.1 for a discussion of why this alternative can be ruled out.

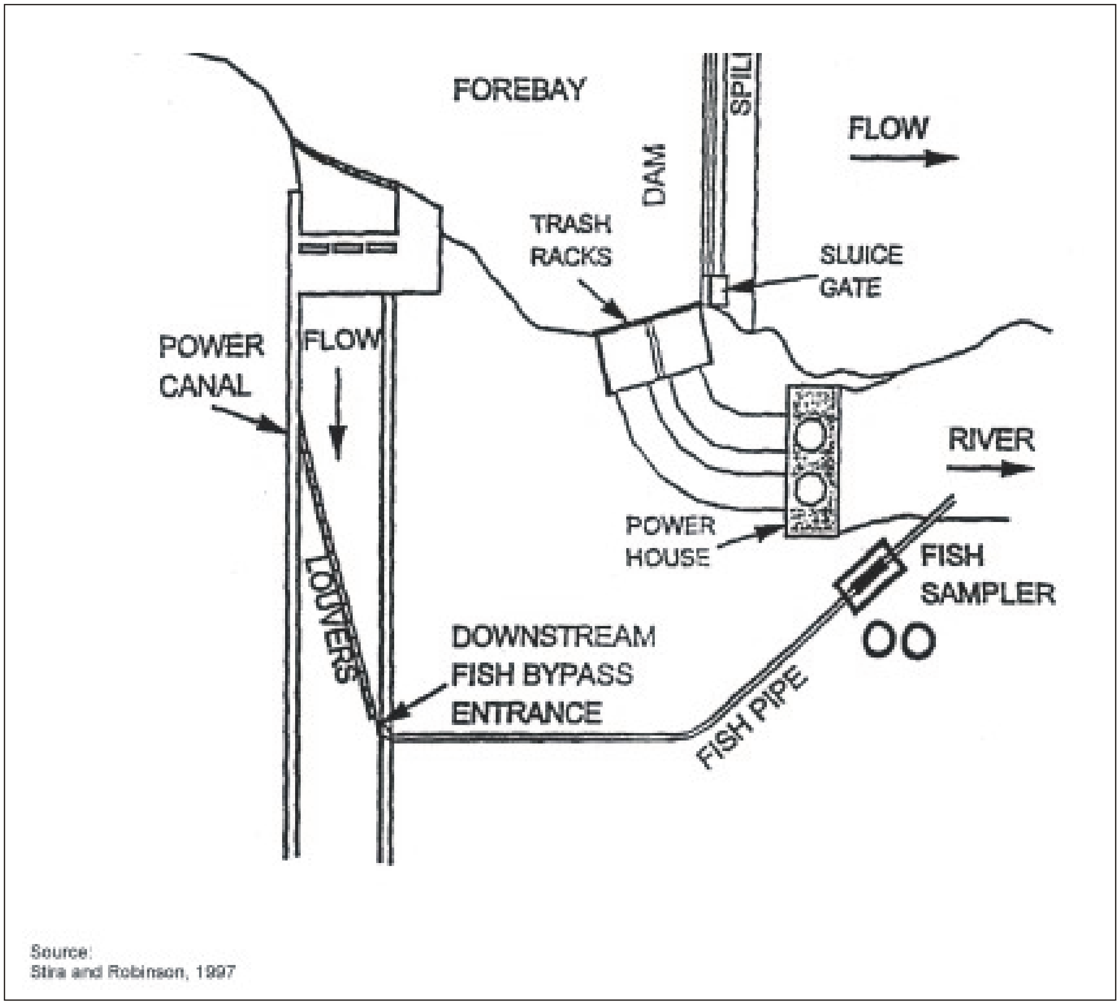


Figure 6.4-4. Holyoke fish bypass louver system.

6.4.3.2 Angled Screens

The angled screen design is composed of a series of vertical traveling screens angled strategically to maximize diversion of aquatic species. The organisms are diverted to a primary bypass line. The organisms captured in the primary bypass line will typically be led to a secondary bypass line, holding tank, or released back to the natural habitat. Angled screens have been studied for possible use at CWIS to protect a variety of fishes in freshwater, riverine, estuarine, and marine environments (EPRI 1999). Angled screens have been used at both irrigation and hydroelectric intake facilities. Given the proper physical and hydraulic conditions, the angled screen system can be very effective in diverting fishes to the bypass line.

Technical Criteria

Angled screen diversion systems are used mainly when impingement rates are very high, typically at intakes that are equipped with long offshore intake conduits and other configurations where organisms are entrapped and cannot escape contacting the intake. Since angled screens must be turned to the current, the overall intake structure required to house them must project out in front of the pumps. At SBPP such an installation would have to be built out into the San Diego Bay in the area in front of the existing intake structures.

Economic Criteria

This alternative would not be cost effective for the existing CWIS since it only addresses impingement impacts, which, based on data presented in Sections 4.0 and 5.0, are not a significant issue.

Potential Biological Benefits

Installations of angled screens in combination with diversion and fish return systems are effective at removing entrapped and/or impinged organisms with varying degrees of return survival. There have been various studies on angled screen application to different plants/facilities around the U.S. In Brayton Point Station Unit 4 at Mt. Hope Bay, Massachusetts, an 18-month biological effectiveness evaluation was conducted. The study determined the species, number, and initial/extended survival life of fishes diverted in the bypass line (Davis et al. 1988) and found that the survival rates at Brayton varied from 25 percent for fragile species to 65 percent for hardy species. The overall diversion efficiency of all species was 76.3 percent (Davis et al. 1988). The system was not very effective for young bay anchovy but was sufficient to protect the other species.

The angled screen system at the Danskammer Point Generating Station on the Hudson River was tested in 1981 (EPRI 1999). The diversion effectiveness study was conducted over a three-year period and examined young/older fishes and ichthyoplankton (EPRI 1999). The diversion efficiency range was from 95.4–100 percent, with a mean of 99.4 percent. The study determined that the overall efficiency (diversion efficiency × initial survival × latent [96-hour] survival) ranged from 67.9 percent for alewife to 98.7 percent for spottail shiner, with a mean percentage of 84.4 (EPRI 1999). The angled screen system has proven that it can protect the young-of-the-year and older fishes, and it is an effective device for preventing impingement.

Other Environmental Impacts

As noted above, an angled screen system requires an area leading up to the pumps in which the screens are installed at an angle to the flow. This would take up additional area from the bay or from land inshore of the existing intake structure building. Building into the bay would result in effects related to the additional construction activities and loss of additional bay habitat from the larger structure.

Evaluation Summary

The low potential biological benefits relative to the costs for installation and operation at SBPP preclude angled screens from being considered a viable alternative. Angled screens have been primarily used at locations where impingement rates have been high. Impingement rates for the existing SBPP have been measured and found to be low. Angled screens were found not to be BTA.

This alternative cannot realistically be implemented in the time remaining in Duke's lease agreement (see Section 6.1 for a further discussion of the reasons for this).

6.4.4 Alternate Biofouling Control

The biofouling control procedure currently used at SBPP consists of intermittent chlorination for slime control and biofouling control.

Alternative biofouling control schemes that can be considered for application at SBPP include the following:

- Increased chlorine dosage,
- Increased frequency of chlorination from intermittent dosage to continuous application,
- Use of alternative chemical toxins, including bromine, chlorine dioxide, chlorine bromide, and ozone,
- Application of toxic coatings on cooling system conduit walls, and
- Oxygen depletion (stagnation).

6.4.4.1 Technical Criteria

Chlorination is currently used at SBPP in an effort to control slime accumulation on condenser surfaces and colonization of the cooling water systems by macroinvertebrates such as barnacles, mussels, and hydroids. The chemicals that are used to clean the intake and discharge structures are sodium hypochlorite. The use of bromine compounds is also permitted in the plant's NPDES permit, although this form of treatment is not currently used. The cooling water stream is initially dosed with sodium hypochlorite, which converts to chlorine to treat microfoulings. The level of chlorine at the point of discharge is limited to 0.2 mg/l (200 ppb) total residual chlorine (TRC). Chlorination is limited by the NPDES permit to two hours per day per generating unit, with a maximum usage of 334 pounds of chlorine per day (Addendum 1 to SBPP's NPDES permit).

Although entrainment impacts were assessed with an assumption that 100 percent of entrained organisms would be cropped during transit by biofouling organisms, this conservative assumption probably overestimates actual losses that could be minimized by rigorous control of biofouling growth in the SBPP's CWIS.

6.4.4.2 Economic Criteria

Since biofouling treatments will continue following standard procedures, no additional costs are anticipated.

6.4.4.3 Potential Biological Benefits

All of these alternatives, with the exception of increasing chlorination frequency to continuous application, are expected to have the potential to reduce entrainment cropping by controlling the colonization of CWS conduits by marine fouling organisms. Because the chlorine is also toxic to entrained fish eggs, larvae, and juveniles and invertebrates, continuous chlorination would potentially result in 100 percent entrainment mortality.

6.4.4.4 Other Environmental Impacts

Since biofouling treatments are conducted following standard procedures within NPDES permit limits, no environmental effects are anticipated.

6.4.4.5 Evaluation Summary

Biofouling treatment periods and frequencies for the existing units are necessary and beneficial, and it is recommended that they continue.

6.5 Comparison of Alternatives

An examination was made of the relative effect of operating the plant's CWIS on fish and macroinvertebrate populations. Entrainment and impingement impact analyses (Sections 3.0–5.0) showed that operations would not cause significant adverse impacts on the populations of fishes, crabs, and lobster inhabiting San Diego Bay. There is no empirical evidence that San Diego Bay populations of gobies and other bay/estuarine species are limited by their habitat carrying capacity. Most of the organisms entrained are species that are distributed widely by ocean currents along the Pacific coast and by the tidal exchange with San Diego Bay. The broad extent and movement of these species along the coast would act to reduce the risk of localized population effects. In addition, species of entrained larvae have very high natural mortality rates that are substantially greater than entrainment effects. None of the entrained species are state or federally protected.

For these reasons, it was concluded that the impact of SBPP operation on populations of local marine life would continue to be undetectable at the population levels of the species involved. More importantly, there is no certainty that implementation of alternative intake technologies designed to further reduce entrainment or impingement mortality would result in a detectable increase in population abundance for fish and invertebrate species inhabiting the San Diego Bay region and the adjacent coastal waters.

Furthermore, the costs of the alternatives evaluated would be wholly disproportionate to any environmental benefit gained given that Duke's short-term lease with the Port of San Diego expires in 2009. As previously stated there are no plans at the present time to continue the operation of the existing facilities after 2009. There is simply not enough time to design, permit, construct, and install these alternatives. The length of the analysis period (four years) would also make the amortized costs of these alternatives wholly disproportionate to any environmental benefits received.

6.6 Conclusion

The wet/dry plume abated cooling tower using reclaimed wastewater or desalinated water was the only technically viable closed-cycle cooling system for use at SBPP. This option was eliminated because of the limited remaining duration of Duke's SBPP lease, which expires in 2009. There would not be enough time to design, permit, and construct the cooling towers and the water conveyance pipeline. The costs of the two wet/dry alternatives over the analysis period were wholly disproportionate to the environmental benefit gained based on the entrainment and impingement data collected during the 2001 and 2003 studies. All the other closed-cycle cooling options were eliminated on the basis of physical limitations on the SBPP site or unacceptable environmental impacts. All closed-cycle alternatives significantly reduce plant output, due primarily to reduced steam turbine generator efficiency and secondarily to increased auxiliary plant loads. Likewise, all options would result in varying levels of visual impacts and require significant land areas that may not be available. A number of the options would also result in increased air emissions and increased noise levels. For all the above reasons, the existing once-through cooling water system is preferred to a closed-cycled cooling system.

Safe and reliable reduction in cooling water pump operations, within the thermal discharge constraints of the current NPDES permit, coinciding with periods when a unit is out of service is an effective method of reducing the losses of organisms through entrainment and impingement, and is the current practice at the SBPP.

The existing shoreline vertical traveling screen design represents the best technology available. This conclusion is based on the finding of relatively insignificant entrainment and impingement effects (including no population level effects) and consideration of various demonstrated alternative technologies, including potential biological effectiveness for further reducing entrainment and impingement losses, engineering feasibility, and cost-effectiveness, as outlined in the EPA guidance manual.

7.0 Literature Cited

Allen, B. M. 1916. Notes on the spiny lobster (*Panulirus interruptus*) of the California coast. Univ. Calif. Publ. Zool. 19(12): 139-152.

Allen, L. G. 1982. Seasonal abundance, composition, and productivity of the littoral fish assemblage in upper Newport Bay, California. Fish Bull., U. S. 80 (4): 769-790.

Allen, L. G. 1999. Fisheries Inventory and Utilization of San Diego Bay, San Diego, California. Final Report: Sampling Period July 1994 to April 1999. Prepared for U.S. Navy and the San Diego Unified Port District.

Allen, L. G., A. M. Findlay, and C. M. Phalen. 2002. Structure and standing stock of the fish assemblages of San Diego Bay, California from 1994 to 1999. Bull. So. Cal. Acad. Sci. 101(2): 49-85.

Ambrose, R. F. 1981. Observations on the embryonic development and early post-embryonic behavior of *Octopus bimaculatus*. Veliger, 24: 139-146 pp.

Ambrose, R. F. 1988. Population dynamics of *Octopus bimaculatus*: Influence of life history patterns, synchronous reproduction and recruitment. Malacologia, 29(1): 23-29 pp.

Applied Science Associates (ASA). 1999. Ichthyoplankton Monitoring Study Deployment of a Gunderboom System at Lovett Generating Station Unit 3, 1998. Orange and Rockland Utilities, Inc.

Barlow G. W. 1961. Intra- and interspecific differences in rate of oxygen consumption in gobiid fishes of the genus *Gillichthys*. Biol. Bull. Mar. Biol. Lab., Woods Hole. 121: 220-229.

Barlow, G. W. and V. D. De Vlaming. 1972. Ovarian cycling in longjaw gobies, *Gillichthys mirabilis*, from the Salton Sea. California Department of Fish and Game, Fish Bull. 58(1): 50-57.

Barlow, G. W. 1963. Species structure of the gobiid fish *Gillichthys mirabilis* from coastal sloughs of the eastern Pacific. Pacific Science 17(1): 47-72.

Boreman, J., C. P. Goodyear, and S. W. Christensen. 1978. An empirical transport model for evaluating entrainment of aquatic organism by power plants. United States Fish and Wildlife Service. 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 plants sited on estuaries. Transactions of the American Fishery Society 110: 253-260.

Brothers, E. B. 1975. The comparative ecology and behavior of three sympatric California gobies. Ph.D. Thesis, University of California at San Diego.

Brown, R. E. 1999. The potential of strobe lighting as a cost-effective means for reducing impingement and entrainment. Proceeding of the EPRI/DOE Power Generation Impacts on Aquatic Resources Conference, Atlanta, GA (April 1999).

Browning, B. M., J. W. Speth, and W. Gayman. 1973. The natural resources of San Diego Bay: Their status and future. CA Dept. Fish Game. Coastal Wetlands Series #5. 105 pp.

Butler, J., D. Fuller, and M. Yaremko. 1999. Age and growth of market squid (*Loligo opalescens*) off California during 1998. CalCOFI Rept. 40:191-195.

Calliet, G. M., B. Antrim, D. Ambrose, S. Pace, and M. Stevenson. 1977. Species composition, abundance and ecological studies of fishes, larval fishes, and zooplankton in Elkhorn Slough. pp. 216-386 *in* J. Nybakken, G. Calliet, and W. Broenkow (eds.). Ecological and hydrographic studies of Elkhorn Slough, Moss Landing Harbor and nearshore coastal waters, July 1974 to June 1976. Moss Landing Marine Lab., Moss Landing, CA.

Chadwick, D. B., J. L. Largier, and R. T. Cheng, 1996. The role of thermal stratification in tidal exchange at the mouth of San Diego Bay. Proceedings of the 7th International Conference on the Physics of Estuaries and Coastal Seas, American Geophysical Union.

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

Clarke, K. R. and R. N. Gorley. 2001 PRIMER Version 5.0: User Manual/Tutorial. PRIMER-E, Plymouth, England.

Conservation Management Institute (CMI). 2003. *http://fwie.fw.vt.edu/WWW/macsis/lists/M010301.htm*

Consolidated Edison Company of New York, Inc (Con Ed). 1994. Evaluation of Underwater Sound to Reduce Impingement at the Arthur Kill Station.

Davis, R. W., J. A. Matousek, M. J. Skelly, and M. R. Anderson. 1988. Biological evaluation of Brayton Point Station Unit 4 Angled Screen Intake. In Proceedings: Fish Protection at Steam and Hydroelectric Power Plants, San Francisco, CA, October 28–31, 1987. Sponsored by Electric Power Research Institute (EPRI). CS/EA/AP-5663-SR.

de la Rosa-Vélez, J., R. Escobar-Fernández, F. Correa, M. Maqueda-Cornejo, and J. de la Torre-Cueto. 2000. Genetic structure of two commercial penaeids (*Penaeus californiensis* and *P. stylorostris*) from the Gulf of California, as revealed by allozyme variation. Fish. Bull., U. S. 98: 647–683.

de Vlaming, V. L. 1971. Thermal selection behavior in the estuarine goby, *Gillichthys mirabilis* Cooper. J. Fish Biol. 3: 277-286.

Digby, P. G. N. and R. A. Kempton. 1987. Multivariate Analysis of Ecological Communities. Chapman and Hall. New York. 206 p. New York.

Divita, R., M. Creel and P. F. Sheridan. 1983. Foods of coastal fishes during brown shrimp, *Peneus aztecus*, migration from Texas estuaries. Fish. Bull., U. S. 81: 396-404.

Duffy, J. M. 1973. The status of the California spiny lobster resource. Calif. Dept. Fish and Game Mar. Res. Tech. Rep. 10. 15 p.

Dunning, D. 1995. Ultrasound Deterrence: Alewife at a Nuclear Generating Station in New York. *In*: T. J. Carlson and A. N. Popper (eds.): Using Sound to Modify Fish Behavior at Power-Production and Water-Control Facilities: A Workshop. Prepared for U.S. Department of Energy and Bonneville Power Administration, DOE/BP-62611-11.

Dunning, D. J., Q. E. Ross, P. Geoghegan, J. J. Reichle, J. K. Manezes, and J. K. Watson. 1992. Alewifes avoid high-frequency sound. North American Journal of Fisheries Management 12: 407-416.

Eckert, S. A. 1994. Results of trawling for sea turtles in the South San Diego power plant intake channel. Unpublished report prepared for SDG&E.

Edmands, F. A. II. 1983. The diel distribution and transport of ichthyoplankton collected by stationary nets in Newport Bay, California, July 1979. M.A. Thesis, Calif. State Univ., Fullerton, CA, 112p.

Emmett, R. L., S. A. Hinton, S. L. Stone, and M. E. Monaco. 1991. Distribution and abundances 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, 329pp.

Environmental Science Associates (ESA). 1997. Initial Study for San Diego Gas & Electric Company's Application No. 97-12-039.

Electric Power Research Institute (EPRI). 1986. Assessment of Downstream Migrant Fish Protection Technologies for Hydroelectric Application. EPRI Report No. AP-4711.

EPRI. 1994. Research Update on Fish Protection Technologies for Water Intakes. EPRI Report No. TR-104122.

EPRI. 1998. Evaluation of Fish Behavioral Barriers. EPRI Report No. TR-109483.

EPRI. 1999. Fish Protection at Cooling Water Intakes. EPRI Report No. TR-114013.

Eschmeyer, W.N. and E.S. Herald. 1983. A Field Guide to the Pacific Coast Fishes of North America. Peterson Field Guide. Houghton Mifflin Company. 336 pp.

Farooqi, T., R.F. Shaw and J.G. Ditty. 2003. Engraulidae. *In*: Richards, W.J. (ed) Preliminary Guide to the Identification of the Early Life History Stages of

Ichthyoplankton of the Western Central Atlantic. Southeast Fisheries Science Center, Miami, Florida (cited December 3, 2003 at http://www4.cookman.edu/noaa/Ichthyoplankton/ENGRAULIDAE.PDF)

Fey, D. P. 2002. Length correction of larval and early-juvenile herring (*Clupea harengus* L.) and smelt (*Osmerus eperlanus* L.) after preservation in formalin and alcohol. Bull. Sea Fish. Inst. Poland 155:47-51.

Fields, W. G. 1965. The structure, development, food, relations, reproduction, and life history of the squid *Loligo opalescens* Berry. Calif. Dept. Fish and Game, Fish Bull. No. 131:108 pp.

Fish and Wildlife Information Exchange. (FWIE). 1999. Department of Fisheries and Wildlife Sciences. Virginia Tech. 1999. http://fwie.fw.vt.edu/WWW/macsis/lists/M070013.htm.

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

Fronk, R. H. 1969. Biology of *Atherinops affinis littoralis* Hubbs in Newport Bay. MS Thesis, Univ. Calif. Irvine. 106 pp.

Galindo-Bect, S. M., E. P. Glenn, H. M. Page, K. Fitzsimmons, L. A. Galindo-Bect, J. M. Hernandez-Ayon, R. L. Petty, J. Garcia-Hernandez, and D. Moore. 2000. Penaeid shrimp landings in the upper Gulf of California in relation to Colorado River freshwater discharge. Fish. Bull., U. S. 98: 222–225.

Harza Engineering Company Inc. and RMC Environmental Services. 1992. Response of Atlantic Salmon Smolts to Louvers in the Holyoke Canal, Spring 1992. New England Power Company.

Harza Engineering Company Inc. and RMC Environmental Services. 1993. Response of Juvenile Clupeids to Louvers in the Holyoke Canal, Fall 1992. New England Power Company.

Heath, K. L. 1980. Comparative life histories of two species of anchovies, *Anchoa delicatissima* and *A. compressa* (Family Engraulidae) from Newport Bay, California. M.A. Thesis CSU Fullerton, 71 p. (cited in Emmett et al 1991)

Horn, M. H. and L. G. Allen. 1976. Numbers of species and faunal resemblance of marine fishes in California bays and estuaries. Bull. So. Cal. Acad. Sci. 75(2): 159-170.

Houde, E. D. 1987. Fish early life dynamics and recruitment variability. Am. Fish Soc. Symp. 2: 17-29.

Jensen, G. C. 1995. Pacific coast crabs and shrimps. Sea Challengers. Monterey, California. 87 pp.

Johnson, M. W. 1956. The larval development of the California spiny lobster, *Panulirus interruptus*, with notes on *Panulirus gracilis*. Proc. Calif. Acad. Sci. 29: 1-19.

Johnson, M. W. 1960. Production and distribution of larvae of the spiny lobster *Panulirus interruptus* with records on *Panulirus gracilis*. Bull. Scripps Inst. Oceanogr. 6: 413-462.

Jung, S. 2002. Fish community structure and the spatial and temporal variability in recruitment and biomass production in Chesapeake Bay. PhD. Thesis, University of Maryland at College Park. 349 p.

Knudsen, F. R., P. S. Enger, and O. Sand. 1992. Awareness reactions and avoidance responses to sound in juvenile Atlantic salmon, *Salmo salar* L. Journal of Fish Biology. 40:523-534.

Knudsen, F. R., P. S. Enger, and O. Sand. 1994. Avoidance responses to low frequency sound in downstream migrating Atlantic salmon, *Salmo salar* L. Journal of Fish Biology. 45:227-233.

Largier, J. L. 1995. San Diego Bay circulation: a study of the circulation of water in San Diego Bay for the purpose of assessing, monitoring and managing the transport and potential accumulation of pollutants and sediment in San Diego Bay. Scripps Institution of Oceanography Tech. Rep. # 1-188-190-0. Final Report prepared for Calif. State Water Res. Control Board and the Calif. Reg. Water Quality Control Board, San Diego Region.

Lindberg, R. G. 1955. Growth, population dynamics and field behavior in the spiny lobster, *Panulirus interruptus* (Randall). Univ. Calif. Publ. Zool. 59(6): 157-248.

Lawler, Matusky & Skelly Engineers (LMS). 1996a. Effectiveness Evaluation of a Fine Mesh Barrier Net Located at the Cooling Water Intake of the Bowline Point Generating Station. Prepared for Orange and Rockland Utilities, Inc.

LMS. 1996b. Lovett Generating Station Gunderboom Evaluation Program 1995. Orange and Rockland Utilities, Inc. LMSE-96/0291&169/330.

LMS. 1997. Lovett Generating Station Gunderboom Evaluation Program 1996. Orange and Rockland Utilities, Inc. LMSE-97/0191&169/321.

LMS. 1998. Lovett Generating Station Gunderboom Evaluation Program 1998. Orange and Rockland Utilities, Inc. 169-335.

Lockheed Environmental Sciences (LES). 1981. South Bay Power Plant Receiving Water Monitoring Program - A Four-Year Cumulative Analysis Report (1977-1980). Carlsbad. CA.

Loos, J. L. 1986. Evaluation of Benefits to PEPCO of Improvements in the Barrier Net and Intake Screens at Chalk Point Station Between 1984 and 1985. Prepared for Environmental Affairs Group Water and Land Use Department Potomac Electric Power Company Washington, D. C.

Love, M. 1996. Probably more than you want to know about fishes of the Pacific Coast. Really Big Press, Santa Barbara, CA. 381 pp.

Luo, J. and J. Musick. 1991. Reproductive biology of the bay anchovy in Chesapeake Bay. Trans. Am. Fish Soc 120: 701-710.

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(3): 613-619.

MacDonald, C. K. 1975. Notes on the family Gobiidae from Anaheim Bay. pp. 117-121 *in* E. D. Lane and C. W. Hill (eds.). The marine resources of Anaheim Bay. Calif. Dept. Fish and Game, Fish Bull. 165.

Macewicz, B. J., N. C. H. Lo, and J. R. Hunter. 2000. Lifetime fecundity of the market squid, *Loligo opalescens*. SWFSC, La Jolla, CA. Presented at CalCOFI Conference 2000, Nov. 1 – 3, 2000. Lake Arrowhead Conference Center. University of California, Los Angeles. Lake Arrowhead, California.

MacGinitie, G. E. and N. MacGinite. 1968. Natural History of Marine Animals. McGraw-Hill, New York, pp. 390-399.

Mair, J. McD. 1980. Salinity and water-type preferences of four species of postlarval shrimp (Penaeus) from West Mexico. J. Exp. Mar. Biol. Ecol. 45: 69-82.

Mann, D. A., Z. Lu, and A. N. Popper. 1997. A clupeid fish can detect ultrasound. Nature 389: 341.

Matarese, A. C., A. W. Kendall Jr., D. M. Blood, and B. M. Vintner. 1989. Laboratory guide to early life history stages of northeast Pacific fishes. NOAA Tech. Rept. NMFS 80, 652 pp.

McGowen, G.E. 1981. Composition, distribution, and seasonality of ichthyoplankton populations near an electricity generating station in South San Diego Bay, California. Rapp. P.-v. Réun. Cons. Int. Explor. Mer. 178: 112-114.

Merkel & Associates, Inc. 2000a. South Bay Power Plant cooling water discharge channel fish community characterization study, April 1997 through January 2000, Final Report. Prepared for Duke Energy North America and San Diego Regional Water Quality Control Board. 56 pp. + App.

Merkel & Associates, Inc. 2000b. Environmental controls on the distribution of eelgrass (*Zostera marina* L.) in south San Diego Bay: An assessment of the relative roles of light, temperature, and turbidity in dictating the development and persistence of seagrass in a shallow back-bay environment. Prepared for Duke Energy North America. 81 pp. + App.

Michaud, D. T. and E. P. Taft. 1999. Recent Evaluation of Physical and Behavioral Barriers for Reducing Fish Entrainment at Hydroelectric Projects in the Upper

Midwest. Proceeding of the EPRI/DOE Power Generation Impacts on Aquatic Resources Conference, Atlanta, GA (April 1999).

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. California Department of Fish and Game 76(1): 4-13.

Miller, D. J. and R. N. Lea. 1972. Guide to the coastal marine fishes of California. Calif. Dept. Fish and Game, Fish Bull. 157. 249 pp.

Moore, D. W., R. W. Sherry and F. Montanez. 1974. Maturation of *Penaeus californiensis* in captivity. Proc. 5th Ann. Meet. World Maric. Soc. 5: 445-447.

Morris, R. F., D. P. Abbott, and E. C. Haderlie. 1980. Intertidal invertebrates of California. Stanford University Press. Palo Alto, CA. 690 pp.

Moser, H. G., W. Watson, S. R. Charter, D. A. Ambrose and E. M. Sandknop. 1996. The Early Stages of Fishes in the California Current Region. California Cooperative Oceanic Fisheries Investigations, Atlas No. 33. Allen Press Inc., Lawrence, Kansas.

Moyle, P. B. and Cech. 1988. Fishes: An Introduction to Ichthyology. Department of Wildlife and Fisheries Biology, U.C. 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, Marine Review Committee Doc. No. 89-02, 346 pp.

Murray, L. S., and T. S. Jinnette. 1978. Survival of Dominant Estuarine Organisms Impinged on Fine Mesh Traveling Screens at the Barney M. Davis Power Station. Central Power and Light Company.

National Marine Fisheries Service (NMFS). 2003. Personal communication from the Fisheries Statistics and Economics Division. MRFSS Queries. *http://www.st.nmfs.gov/st1/recreational/queries/index.html* [cited October 28, 2003].

Nelson, J. S. 1994. Fishes of the world, 3rd Ed. John Wiley and Sons, Inc., New York. 600 pp.

Nestler, J. M., G. R. Ploskey, and J. Pickens. 1992. Responses of blueback herring to high frequency sound and implications for reducing entrainment at hydropower dams. North American Journal of Fisheries Management 12(4): 667-683.

Ninos, M. 1984. Settlement and metamorphosis in *Hypsoblennius* (Pisces, Blenniidae). Ph.D Thesis University of Southern California. 86 pp.

Pacific Coast Fisheries Information Network (PacFIN). 2003. *http://www.psmfc.org/pacfin/*

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

Phillips, R C. 1984. The ecology of eelgrass meadows in the Pacific northwest: a community profile. U.S. Fish Wildl. Serv. OBS 84/24, 85pp

Ploskey, G. R., J. M. Nestler, G. N. Weeks, and C. Schilt. 1995. Evaluation of an Integrated Fish-Protection System. *In:* Waterpower 95, Proceedings of the International Conference on Waterpower. American Society of Civil Engineers, New York, NY.

Popper, A. N. and T. J. Carlson. 1998. Application of sound and other stimuli to control fish behavior. Transactions of the American Fisheries Society 127(5): 673-707.

Porter, S. M., A. L. Brown, and K. M. Bailey. 2001. Estimating live standard length of net-caught walleye pollock (*Theregra chalcogramma*) larvae using measurements in addition to standard length. Fish. Bull. 99:691-696.

Public Service Electric and Gas Company (PSE&G). 1993. Appendix I—Modeling. Permit No. NJ0005622. Prepared by Lawler, Matusky, and Skelly Engineers, Pearl River, NY. Comments on NJPDES Draft. 82 pp.

Recreational Fisheries Information Network (RecFIN). 2003. *http://www.psmfc.org/recfin/*

Reider, R. H., D. D. Johnson, P. B. Latvaitis, J. A. Gulvas, and E. R. Guilfoos. 1997. Operation and Maintenance of the Ludington Pumped Storage Project Barrier Net. *In:* Fish Passage Workshop, Milwaukee, WI, May 6-8, 1997. Sponsored by Alden Research Laboratory, Conte Anadromous Fish Research Laboratory, Electric Power Research Institute, and Wisconsin Electric Power Company.

Ricker, W. E. 1975. Computation and interpretation of biological statistics of fish populations. Fishery Research Board of Canada Bulletin 91. 382 pp.

Riedel R., L. Helvenston, and B. A. Costa-Pierce. 2001. Fish biology and fisheries ecology of the Salton Sea. U.S. Bureau of Reclamation Administrative Report, http://www.usbr.gov/lc/region/saltnsea/pdf_files/ssfishdft-fin_rpt.pdf. 58 p.

Rilling, G. C. and E. D. Houde. 1999a. Regional and temporal variability in distribution and abundance of bay anchovy, *Anchoa mitchilli*, eggs, larvae, and adult biomass in Chesapeake Bay. Estuaries 22(4): 1096-1109.

Rilling, G. C. and E. D. Houde. 1999b. Regional and temporal variability in growth and mortality of bay anchovy, *Anchoa mitchilli*, larvae in Chesapeake Bay. Fish Bull. 97: 555-569.

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

Rodriguez de la Cruz, M. C. 1976. Sinopsis biológica de las especies del género *Penaeus* del Pacifico Mexicano. Mem. Simp. Biol. Y Din. Pobl. Cam. Guymas, Son. I:282–316.

Rodriguez de la Cruz, M. C. and F. J. Rosales. 1970. Sinopsis de *Penaeus californiensis* Holmes. FAO, Rome. 35 p.

Ross, Q. E., D. J. Dunning, J. K. Menezes, M. J. Kenna, Jr., and G. Tiller. 1996. Reducing impingement of alewives with high frequency sound at a power plant on Lake Ontario. North American Journal of Fisheries Management 16:548–559.

Ross, Q. E., D. J. Dunning, R. Thorne, J. K. Menezes, G. W. Tiller, and J. K. Watson. 1993. Response of alewives to high-frequency sound at a power plant intake on Lake Ontario. North American Journal of Fisheries Management 13: 291-303.

San Diego Gas & Electric Co. (SDG&E). 1980. South Bay Power Plant Cooling Water Intake System Demonstration Summary. Prepared for: California Regional Water Quality Control Board San Diego Region, San Diego CA.

San Diego Unified Port District (SDUPD) and California State Coastal Conservancy. 1990. South San Diego Bay Enhancement Plan. 4 volumes. Prepared by Michael Brandman Associates, Inc.

San Diego Unified Port District (SDUPD). 1976. Master Plan (1972) amendment submitted to State lands Commission as required by Public Resources Code 63.

Southern California Association of Marine Invertebrate Taxonomists (SCAMIT). 1998. SCAMIT Newsletter 16:10, p. 7.

San Diego Gas and Electric Company (SDG&E). 1973. Thermal distribution and biological studies for the South Bay Power Plant. Final Report – Volume 1 Summary and conclusions, May 1973. Prepared by Pioneer Service and Engineering Company.

Seber, G. A. F. 1982. The Estimation of Animal Abundance and Related Parameters. McMillan. London. 654 p.

Shaw, W. N. 1986. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Pacific Southwest) spiny lobster. U.S. Fish Wildl. Serv. Biol. Rep. 82(11.47). U.S. Army Corps of Engineers, TR EL-82-4. 10pp.

Smith, E. L. 1972. Temperature fluctuations in San Diego Bay. San Diego State University.

Smith, P. E. and S. L. Richardson. 1977. Standard Techniques for Pelagic Fish Egg and Larva Surveys. FAO Fisheries Technical Paper 175:1-100.

Starr, R. M., K. A. Johnson, E. A. Laman, and G. M. Cailliet. 1998. Fishery resources of the Monterey Bay National Marine Sanctuary. Publ. No. T-042. California Sea Grant College System, University of California, La Jolla, CA. 102 pp.

Stephens, J. S. Jr. 1969. Growth, longevity, and the effect of size on the biology of certain Blennioid fishes. Final Report. National Science Foundation GB 5940. 83 pp.

Stephens, J. S. Jr., R. K. Johnson, G. S. Key and J. E. McCosker. 1970. The comparative ecology of three sympatric species of California blennies of the genus *Hypsoblennius* Gill (Teleostomi, Blenniidae). Ecol Monogr. 40(2):213-233.

Stevens, E. G. and H. G. Moser 1982. Observations on the early life history of the mussel blenny, *Hypsoblennius jenkinsi*, and the bay blenny, *Hypsoblennius gentilis*, from specimens reared in the laboratory. CalCOFI Rep. Vol 23:269-275.

Stira, R. J., and D. A. Robinson. 1997. Effectiveness of a Louver Bypass System for Downstream Passage of Atlantic Salmon Smolts and Juvenile Clupeids in the Holyoke Canal, Connecticut River, Holyoke, Massachusetts. *In:* Fish Passage Workshop, Milwaukee, WI, May 6-8, 1997. Sponsored by Alden Research Laboratory, Conte Anadromous Fish Research Laboratory, Electric Power Research Institute, and Wisconsin Electric Power Company.

Tenera Environmental Services. 2001. Morro Bay Power Plant Modernization Project: 316(b) Resource Assessment. Report 2000-107.8 prepared for Duke Energy Morro Bay LLC, Oakland, CA.

Theilacker, G. H. 1980. Changes in body measurements of larval northern anchovy, *Engraulis mordax*, and other fishes due to handling and preservation. Fishery Bulletin 78:685-692.

Todd, E. S. 1968. Terrestrial sojourns of the longjaw mudsucker, *Gillichthys mirabilis*. Copeia 1: 192-194.

U.S. Department of the Navy, Southwest Division (USDoN, SWDIV) and San Diego Unified Port District (SDUPD). 2000. San Diego Integrated Natural Resources Management Plan, September 2000. San Diego, CA. Prepared by Tierra Data Systems, Escondido, CA.

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 pp.

U. S. Fish and Wildlife Service. 1998. Proposed South San Diego Bay Unit, National Wildlife Refuge. Draft Environmental and Land Protection Plan. U.S. Fish Wild. Serv. 89 pp.

U.S. Navy, Southwest Div. Nat. Resource Branch 1994. San Diego Bay Eelgrass Survey. Jan. 1 1994.

University of California Los Angeles. (UCLA) 1999. http://www.lifesci.ucla.edu/odc/html/body_marketsquid.html.

Vargas-Albores, F., Hinojosa-Baltazar, P., Portillo-Clark, G. and Magallon-Barajas, F. 1998. Influence of temperature and salinity on the yellow-leg shrimp, *Penaeus californiensis* Holmes, prophenoloxidase system. Aquacul. Res. 29:549-555.

Walker, B. W., R. R. Whitney, and G. W Barlow. 1961. Fishes of the Salton Sea: The ecology of the Salton Sea, California, in relation to the sport fishery of California. California Department of Fish and Game Fish Bulletin. 113: 88-91. Sacramento, CA.

Wang, J. C. S. 1986. Fishes of the Sacramento- San Joaquin Estuary and adjacent waters, California: A guide to the early life stages. Tech. Rept. 9, January 1986.

Weisel, G. F. 1947. Breeding behavior and early development of the mudsucker, a gobiid fish of California. Copeia 2: 77-85.

White, W.S. 1977. Taxonomic composition, abundance, distribution and seasonality of fish eggs and larvae in Newport Bay, California. M.A. Thesis CSU Fullerton, 107 p.

Williams, G. D. and J. B. Zedler. 1999. Fish assemblage composition in constructed and natural tidal marshes of San Diego Bay: relative influence of channel morphology and restoration history. Estuaries 22: 702-716.