Appendix E- Guidance Documents for Assessing Entrainment Including Additional Information on the Following Loss Rate Models: Fecundity Hindcasting (FH), Adult Equivalent Loss (AEL) and Area Production Forgone using an Empirical Transport Model (ETM/APF)

Associated with the Draft Final Staff Report Including the Draft Final Substitute Environmental Documentation for the Draft Final~~Proposed~~ Desalination Amendment

Documents included:

Steinbeck, J.R., J. Hedgepeth, P. Raimondi, G. Cailliet and D.L. Mayer. 2007. Assessing Power Plant Cooling Water Intake System Entrainment Impacts.

Raimondi, P. 2011. Variation in Entrainment Impact Based on Different Measures of Acceptable Uncertainty. Prepared for California Energy Commission, Public Interest Energy Research Program. http://www.energy.ca.gov/2011publications/CEC-500-2011-020/CEC-500-2011-020.pdf

ASSESSING POWER PLANT COOLING WATER INTAKE SYSTEM ENTRAINMENT IMPACTS

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EXECUTIVE SUMMARY

Steam electric power plants and other industries that withdraw cooling water from surface water bodies are regulated in the U.S. under Section 316(b) of the Clean Water Act of 1972. Of the industries regulated under section 316(b), steam electric power plants have the largest cooling water volumes with some large plants exceeding two billion gallons per day. Environmental effects of cooling water withdrawal result from impingement of larger organisms on screens that block material from entering the cooling water system and the entrainment of smaller organisms into and through the system.

Concerns regarding the environmental effects of entrainment result from the large volume of cooling water potentially used by coastal power plants. In California, the 21 coastal power plants potentially withdraw up to 64 billion liters (17 billion gallons) of seawater per day. This process results in the loss of billions of aquatic organisms, including fishes, fish larvae and eggs, crustaceans, shellfish and many other forms of aquatic life from California's coastal ecosystem each year. There has been increased focus on the effects of power plant cooling water intake systems because the biological resources of the world's oceans, and California's coast in particular, are in serious decline. Long-term declines, which started in the early 1970s, have occurred in 60 percent of the fishes for which landings are reported. Despite the potential contribution of cooling water withdrawal to these declines, recent studies have only been completed at a few of the California power plants (California Energy Commission 2005). Regulations for Section 316(b) of the Clean Water Act published in July 2004 (USEPA 2004) will result in new studies on the environmental effects of cooling water systems at many of the existing power plants in California and throughout the country. The results of these studies will help determine the environmental effects of cooling water withdrawal on biological communities.

While the assessment of impingement effects is relatively straightforward, the assessment of entrainment effects require thoughtful consideration of all aspects of the study design. The difficulties in entrainment assessments arise from several factors. The organisms entrained include planktonic larvae of fishes and invertebrates that are difficult to sample and identify. The entrained larvae are also part of larger source water populations that may extend over large areas or be confined to limited habitats making it difficult to determine the effects of entrainment losses. The early life histories of most fishes on the Pacific coast are also poorly described limiting the usefulness of demographic models for assessing entrainment effects. All of these factors make the assessment of

cooling water system entrainment difficult. The purpose of this report is to present, by example, some of the considerations for the proper design and analysis of entrainment studies.

This report describes three studies for assessing entrainment at coastal power plants in California. They represent a range of marine and estuarine habitats: the South Bay Power Plant in south San Diego Bay, and the Morro Bay and Diablo Canyon Power Plants in central California. These studies utilized a multiple modeling approach for assessing entrainment effects. When appropriate life history information was available for a species, demographic modeling techniques were used to calculate the numbers of adults represented by the losses of fish eggs and larvae due to entrainment. The primary approach for assessment at these plants was the Empirical Transport Model (*ETM*), originally developed for use with power plants entraining water from rivers, and then adapted for use on the open coast and in estuaries in southern California. The *ETM* utilizes the same principles used in fishery management to estimate effects of fishing mortality on the sustainability of a stock. Just as fishery managers use catch and population size to estimate fishery mortality, the *ETM* requires estimates of both entrainment and source water larval populations. The source water population is the abundance of organisms at risk of entrainment as determined by biological and hydrodynamic/oceanographic data. The process of defining the source water and obtaining an estimate of its population varied among the three plants and also among species within studies. The purpose of this paper is to present the multiple modeling approaches used for power plant entrainment assessments, with the main focus being a comparison of the processes used to define the source water populations used in the *ETM* modeling from the three power plants.

The results showed that standard demographic models were generally not usable with species found along the California coast due to the absence of life history information for most of them. The results for the *ETM* ranged from very small levels (<1.0%) of proportional mortality due to entrainment for wide ranging pelagic species such as northern anchovy to levels as high as 50% for fishes with more limited habitat that were spawned near power plant intake structures. The results of the *ETM* were generally consistent with the biology and habitat distributions of the fishes analyzed.

Based on our experiences with these and other studies we believe that a prescriptive approach to the design of entrainment assessments is not possible, and therefore, we provide some general considerations that might be helpful in the design, sampling, and analysis of entrainment impact assessments. These include ensuring that organisms that could be affected by entrainment are effectively sampled and that the sampling will account for any endangered, threatened, or other listed species that could be affected by entrainment. In addition to identifying species potentially affected, it is critical to determine the source water areas potentially affected including the distribution of habitats that might be differentially affected by CWIS entrainment. The sampling plan also needs to account for the design, location, and hydrodynamics of the power plant intake structure. The sampling frequency should accommodate important species that might have short spawning seasons. This may require that the sampling frequency be seasonally adjusted based on presence of certain species. The relative effects of entrainment estimated by the *ETM* model should be much less subject to interannual variation than absolute estimates using Fecundity Hindcasting (*FH*), Adult Equivalent Loss (*AEL*) or other demographic models. Therefore, if source water sampling is done in conjunction with entrainment sampling then one year is a reasonable period of sampling for these studies. The size of the source water sampling area should be based on the hydrodynamics of the system. In a closed system this may be the entire source water. In an open system, ocean or tidal currents and dispersion should be used to determine the appropriate sampling area for estimating daily entrainment mortality (*PE*) for the larger source water population.

Some practical considerations for sample collection and processing include adjusting the sample volume for the larval concentrations in the source waters. This is best done using preliminary sampling with the gear proposed for the study. Age of larvae are best determined using analysis of otoliths, but if this is not possible be sure that length frequencies measured from the entrainment samples are realistic based on available life history and account for egg stages that would be subject to entrainment if fish eggs are not sorted and identified from the samples. This is easily accommodated in the *ETM* approach by adding the duration of the planktonic egg stage to the larval duration calculated from the otolith or length data.

Although we believe that the *ETM* is best approach for assessment, results from multiple models provide additional information for verifying results and for determining effects at the adult population level. One approach for assessment at the adult population level is through converting *ETM* results into an estimate of the habitat necessary to replace the production lost due to entrainment (Area of Production Foregone [APF]). The APF is calculated by multiplying the area of habitat present within the estimated source water by the proportional entrainment mortality estimated from *ETM*. This approach may be useful for scaling restoration projects to help offset losses due to entrainment.

The *ETM* can also be used to estimate the number of equivalent adults lost by entrainment by applying the mortality estimate to a survey of the standing stock. This can be compared with estimates from *FH* and *AEL*. When making these types of comparisons it is important to hindcast or extrapolate the *FH* and *AEL* model estimates to the same age. This may not necessarily result in the same estimates from both models unless the data used in the two models are derived from a life table assuming a stable age distribution. The USEPA (2002) used *AEL* and another demographic modeling approach, production foregone, to estimate the number of age-1 individuals lost due to power plant impingement and entrainment. The accuracy of estimates from any of these demographic models is subject to the underlying uncertainty in aging, survival, and fecundity estimates and population regulatory, behavioral, or environmental factors that may be operating on the subject populations at the time the life history data were collected.

Uncertainty associated with the *ETM* is primarily derived from sampling error that can be controlled by careful design using some of the guidelines provided in this report. With a good sampling design, the *ETM* provides a site-specific, empirically based approach to entrainment assessment that is a major improvement over demographic modeling approaches. In addition, the results can be used to estimate entrainment effects on other planktonic organisms, in estimating cumulative effects of multiple power plants and other sources of mortality, and in scaling restoration efforts to offset losses due to entrainment. We hope that the information in this report will assist others in the design and analysis of CWIS assessments that will be required as a result of the recent publication of new rules for Section 316(b) of the Clean Water Act (USEPA 2004).

1.0 INTRODUCTION

Steam electric power plants and other industries (e.g., pulp and paper, iron and steel, chemical, manufacturing, petroleum refineries, and oil and gas production) use water from coastal areas for cooling resulting in impacts to the marine organisms occupying the affected water bodies. Industries that withdraw cooling water from surface water bodies are regulated in the U.S. under Section 316(b) of the Clean Water Act of 1972 [33 U.S. Code Section 1326(b)]. 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 impacts." Of the industries regulated under section 316(b), steam electric power plants have the largest cooling water volumes ranging from tens of thousands to millions of $m^3 d^{-1}$ (Veil et al. 2003). A survey in 1996 reported that 44% of the power plants in the U.S. utilized a steam electric process involving once-through cooling (Veil 2000). Electricity is generated at these plants by heating purified water to create high-pressure steam, which is expanded in turbines that drive generators and produce electricity (Figure 1-1). After leaving the turbines, steam passes through a condenser where high volume cooling water flow cools and condenses the steam, which is then re-circulated back through the system.

Regulatory guidance for complying with section 316(b), that was first proposed by the U.S. Environmental Protection Agency (EPA) in 1976, was successfully challenged in the courts by a group of 58 utility companies in 1977 and never implemented (Bulleit 2000). As a result, section 316(b) was implemented by the states using a broad range of approaches; some states developed fairly comprehensive programs while others never adopted any formal regulations (Veil et al. 2003). The EPA has recently published new regulations for 316(b) compliance (USEPA 2004) as part of the settlement of a lawsuit against the EPA by environmental groups headed by the Hudson Riverkeeper (Nagle and Morgan 2000). As a result of these new regulations power plants throughout the U.S. are now required to reduce the environmental effects of their cooling water intake systems (CWIS).

The withdrawal of water by once-through cooling water systems has two major impacts on the biological organisms in the source water body: impingement and entrainment (Figure 1-1). Almost all power plants with once-through cooling employ some type of screening device to block large objects from entering the cooling water system (impingement). Fishes and other aquatic organisms large enough to be blocked by the screens may become impinged if the intake velocity exceeds their ability to move away. These organisms will remain impinged against the screens until intake velocity is reduced such that organisms can move away or the screen is backwashed to remove them. Some organisms are killed, injured, or weakened by impingement. Small planktonic organisms or early life stages of larger organisms that pass through the screen mesh are entrained in the cooling water flow. These organisms are exposed to high velocity and pressure due to the cooling water pumps, increased temperatures and, in some cases, chemical treatments added to the cooling water flow to reduce biofouling.

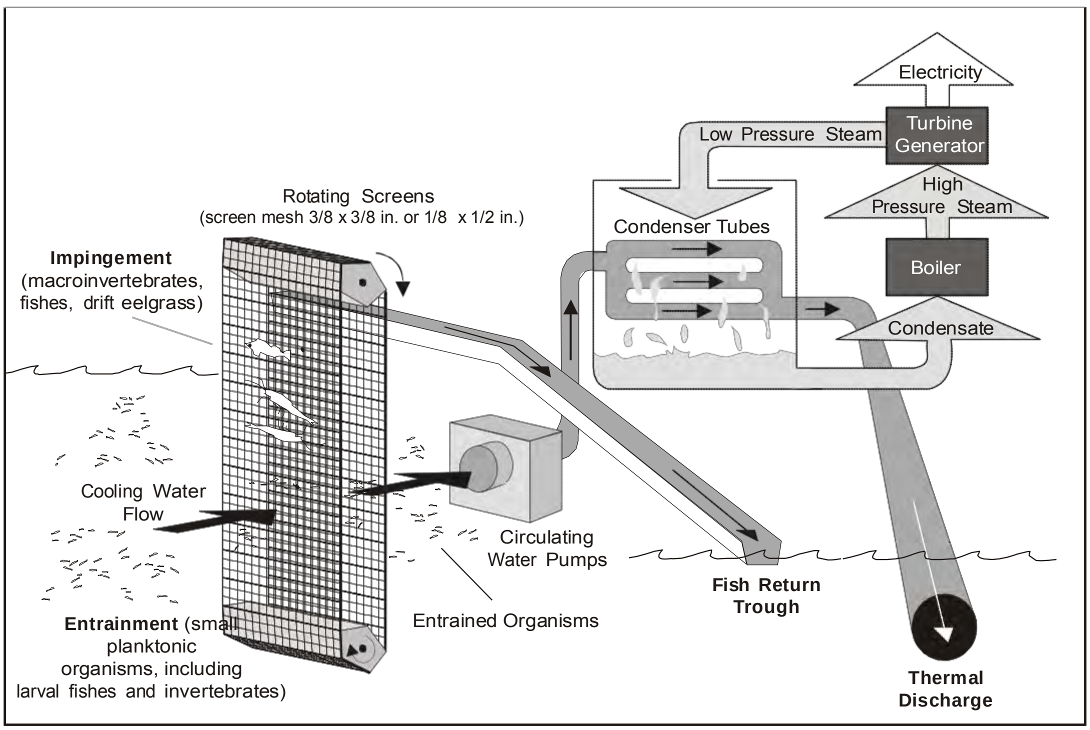


Figure 1-1. Conceptual diagram of power plant cooling water systems at South Bay, Morro Bay, and Diablo Canyon Power Plants, and relationship of impingement and entrainment processes to circulating water system. A fish return trough is present only at the South Bay Power Plant.

Most impingement and entrainment [316(b)] studies on CWIS effects at power plants were completed in the late 1970s and early 1980s using draft guidance issued by the EPA (USEPA 1977). More recently, many power plants throughout the country began to upgrade and expand their generating capacities due to increased demands for power. The California Energy Commission (CEC), which had regulatory authority for these projects in California, required utility companies to determine the impacts of these CWIS changes. Although existing

CWIS are regulated in California through National Pollution Discharge Eliminations System (NPDES) permits issued by the nine Regional Water Quality Control Boards (RWQCB) in the state, the projects done under the regulatory authority of the CEC also required coastal zone permits under the California Coastal Act and therefore were conducted in compliance with the California Environmental Quality Act (CEQA). The CEC and the RWQCBs required new studies in anticipation of the publication of new EPA regulations, but also because data on CWIS impacts were not available for some of the plants and studies at other plants were usually over 20 years old. As a result, we had the opportunity in California to develop approaches to assessing CWIS impacts that might prove useful to researchers at power plants throughout the U.S. These studies involved regulatory agency staff, scientists, consultants, and industry representatives, usually meeting and working under the heading of Technical Workgroups. This collaborative process was first used for studies at the Pacific Gas & Electric Company Diablo Canyon Power Plant and was initiated and directed by Mr. Michael Thomas at the Central Coast Regional Water Quality Control Board (CCRWQCB) (Ehrler et al. 2003). This process was also used on studies for plant re-powering projects under CEC and RWQCB review at the Moss Landing, Morro Bay, Potrero and Huntington Beach Power Plants.

This paper focuses on methods for assessing only entrainment effects (not impingement), and specifically, entrainment effects on ichthyoplankton. Entrainment affects all types of planktonic organisms, but most studies do not assess holoplankton (phytoplankton and zooplankton that are planktonic for their entire life) because their broad geographic distributions and short generation times reduce the effects of entrainment on their populations. In contrast, the potential for localized effects on certain fish populations is much greater, especially for power plants located in riverine or estuarine areas where a large percentage of the local population may be at risk of entrainment (Barnthouse et al. 1988, Barnthouse 2000). Although the potential for similar effects exists for certain invertebrate meroplankton (e.g. crab and clam larvae), taxonomy of early larval stages of many invertebrates is not sufficiently advanced to allow for assessments at the species-level. The different larval stages of many invertebrates may also require different mesh sizes and sampling techniques that increase the costs and complexity of a study. In contrast, as a result of programs such as the California Coastal Oceanographic Fisheries Investigations (CalCOFI) program, operating since 1950, ichthyoplankton of the west coast have been well described and long-term data sets exist on the abundances of many larval fishes (Moser 1996).

The best-documented and most extensive 316(b) studies from the period of the late 1970s and early 1980s were from the Hudson River power plants (Barnthouse et al. 1988, Barnthouse 2000). Impacts of cooling water withdrawals from three plants were extensively studied using long-term, river-wide sampling and analyzed using mathematical models designed to predict the effects on striped bass and other fish populations. After many years of debate surrounding a lawsuit, the case was settled out of court. Two of the most important factors in laying the groundwork for the settlement were the converging estimates of the effects from different researchers and the development of models that estimated conditional mortality from empirical data that reflected the “complex interactions of a host of factors” and helped identify the “relative importance of each component of the analysis” (Englert and Boreman 1988).

Numerous demographic modeling approaches have been proposed and used for projecting losses from CWIS impacts (Dey 2003). Equivalent adult (Horst 1975, Goodyear 1978), production foregone (Rago 1984), and variations of these approaches and models (Dey 2003) translate entrainment losses of egg and larval stages into equivalent units (adult fishes, biomass, etc.) that otherwise would not have been lost to the population. Although these models are the most commonly used methods for CWIS assessment and were used by the EPA to support the new 316(b) regulations (USEPA 2004), there can be problems with their application and interpretation. The models require life history parameters (larval duration, survival, fecundity, etc.) that are available for only a limited number of species, generally those managed for commercial or recreational fishing. Our experience has shown that on the California coast, taxa (the term ‘taxa’ [‘taxon’ singular] is used to refer to individual species or broader taxonomic categories that cannot be identified to species) that are usually entrained in highest numbers are small, forage fishes that have very limited life history information available.

However, these models are attractive because their interpretation appears to be straightforward since they convert larval forms into “equivalent units” that are more easily understood by the public, regulators, and managers. The estimates of numbers or biomass of fish from the models can also be added to losses from impingement and compared with commercial or recreational fishery data to provide cost estimates of the losses. Unfortunately, these interpretations are available for only a few taxa, there is usually no scale for determining the significance of the losses to the source water populations, and the studies are only done for a 1-2 yr period, not accounting for inter-annual variation in larval abundances.

Our assessments included a modified version of the Empirical Transport Model (*ETM*) (Boreman et al. 1978, 1981) which circumvented the problems with existing demographic modeling. This model was first developed for use with power plants entraining water from rivers, but MacCall et al. (1983) used the same general approach for entrainment assessments at power plants on the open coast and in estuaries in southern California. In contrast to demographic models, it does not require detailed life history information. The *ETM* provides an estimate of the mortality caused by entrainment to a source water population independent of any other sources of mortality, i.e., conditional mortality (Ricker 1975). Inherent in this approach is the requirement for an estimate of the source water population of larvae affected by entrainment. The source water population is the abundance of organisms at risk of entrainment as determined by biological and hydrodynamic/oceanographic data. The *ETM* is based on the same principles used in fishery management to estimate effects of fishing mortality on a source water population or stock (Boreman et al. 1981, MacCall et al. 1983). Although not specifically required for calculating estimated losses, an estimate of the source water population is also required to provide a context for the losses estimated by demographic models.

The process of defining the source water and obtaining an estimate of its population varies among studies and also among taxa within studies. The purpose of this paper is to present the multiple modeling approaches used for power plant entrainment assessments, with the main focus being a comparison of the processes used to define the source water populations used in the *ETM* modeling from three power plants in California, South Bay Power Plant (SBPP), Morro Bay Power Plant (MBPP), and Diablo Canyon Power Plant (DCPP), which represent a range of marine and estuarine habitats (Figure 1-2). This comparison allows us to compare the approaches and assess the influence of the source water on the proportional mortality of affected fish and invertebrate larval taxa.

The source water population definitions for the three studies were based on the hydrodynamic and biological characteristics of the water bodies where the facilities were located. This is necessary to characterize the sources of the water that is drawn into a power plant. This is fairly simple if the source of cooling water is a lake that is so well mixed that the larval concentrations are uniform. In this case the only necessary information to estimate the mortality on the larvae is the volume of the lake and the plant cooling water volume. In this simple example the mortality is the ratio of the cooling water volume to the source water volume since the concentration of larvae entrained will be equal to the concentration in the source water. In the case of SBPP, samples were collected throughout the entire source water since the larval composition in the habitats within the south part of San Diego Bay were potentially different even though the source water volume for SBPP was treated as a closed system similar to the lake in the above example. The source water for MBPP included both bay and ocean components requiring biological sampling in both locations and calculations to include the effects of tides on the source water. The effects of ocean currents affected the source water potentially entrained for DCPP and the ocean component of the MBPP source water. As a result the source water potentially affected by entrainment was much larger than the areas sampled for these two studies requiring additional measurements and modifications to the model. The many factors that need to be considered in the design of these kinds of studies can be examined by comparing the different approaches taken at the three facilities.

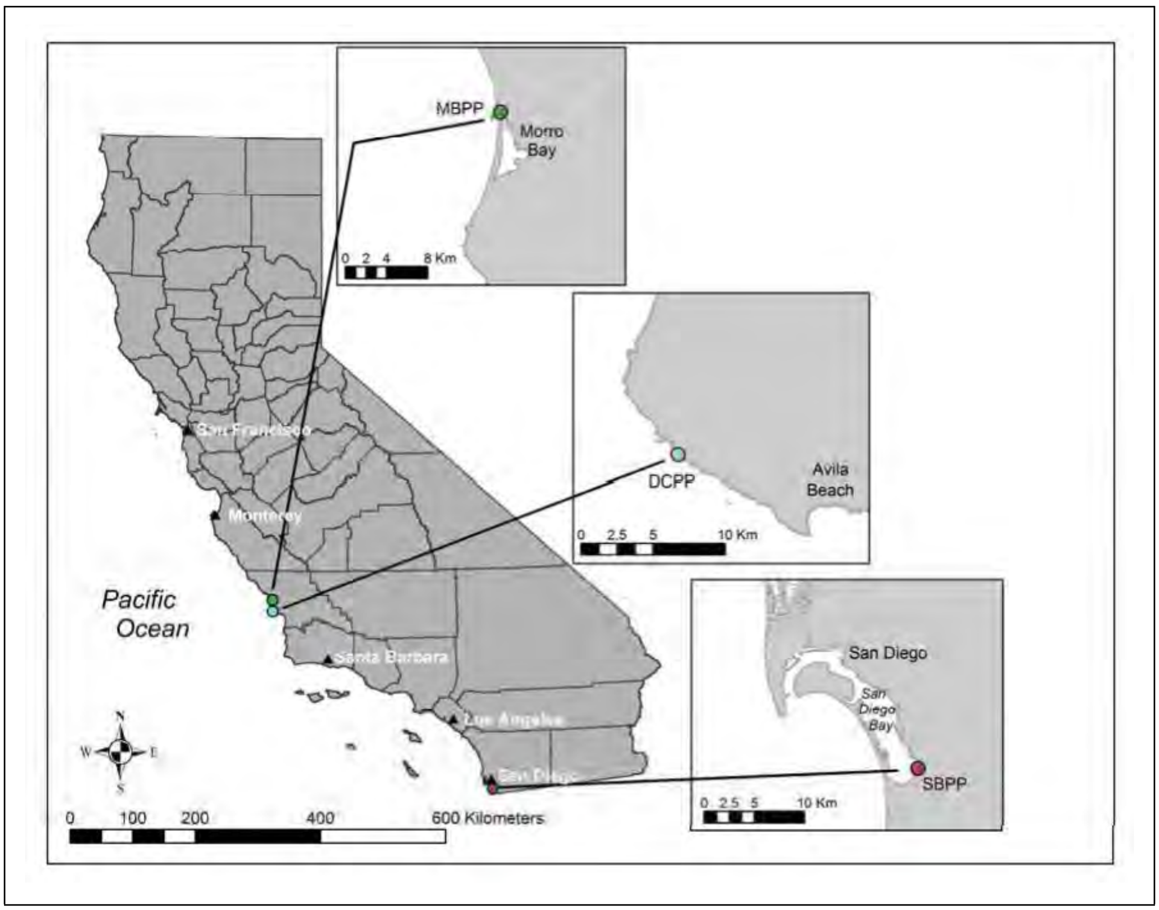


Figure 1-2. Locations of Morro Bay (MBPP), Diablo Canyon (DCPP), and South Bay Power Plants (SBPP).

During the course of these studies we have modified the assessment approaches and this process has continued as we have participated in additional, more recent studies. Therefore one of the additional purposes of this paper is to present these more recent changes in our assessment methods even though they may differ from methods presented in the three example studies.

Our experiences resulting from these studies are especially pertinent with the recent publication of new rules for Section 316(b) of the Clean Water Act (USEPA 2004), and CEC and California Coastal Commission (CCC) requirements for modernizing power plants in California. The new 316(b) rules require that information on the source water body be submitted as part of 316(b) compliance [40 CFR 125.95(b)(2)]. Although not stated in the new rules, it seems appropriate that CWIS impacts would be evaluated based on the source water body information. The CEC and CCC have required this in recent studies and most likely will continue this practice. Hopefully the information in this paper will assist others in the design and evaluation of CWIS assessments that will be required under the new rules.

2.0 METHODS

2.1 POWER PLANT DESCRIPTIONS

The studies we will be presenting as examples were conducted at three power plants: SBPP, MBPP, and DCPP (Figure 1-2). The CWIS for all three plants share several features: shoreline intake structures with stationary trash racks that consist of vertical steel bars to prevent larger objects and organisms from entering the system and traveling water screens (TWS) located behind the bar racks that screen out smaller organisms and debris from the system (Figure 1-1).

Entrainment occurs to organisms that pass through the smaller mesh of the TWS. These organisms are exposed to increased temperatures and pressures as they pass through CWS. The surfaces of the piping in the CWS can be covered with biofouling organisms that feed on organisms that pass through the system. Although studies have shown that there may be some survival after CWS passage (Mayhew et al. 2000), most of these studies were conducted at power plants in rivers and estuaries on the east coast or in the Gulf of Mexico where biofouling was not recognized as a large problem compared with coastal environments. In addition, these studies only examined survival after passage through the system, and did not include comparisons of intake and discharge concentrations where losses due to cropping should be factored into CWS survival. For example, during testing used to determine the appropriate entrainment sampling location losses between the intake and discharge at the Moss Landing Power Plant sometimes exceeded 95 percent and were always greater than 50 percent (Pacific Gas and Electric Co. 1983). For these reasons, our assessments of CWS effects have assumed that entrained organisms experience 100% mortality.

The SBPP, operated by Duke Energy, is located on the southeastern shore of San Diego Bay in the city of Chula Vista, California, approximately 16 km north of the U. S. – Mexican border (Figure 2-1). The plant draws water from San Diego Bay for once-through cooling of its four electric generating units, which can produce a maximum of 723 MWe (Table 2-1). With all pumps in operation, maximum water flow through the plant is 1,580 m^3min^{-1} (2.3 million m^3d^{-1}).

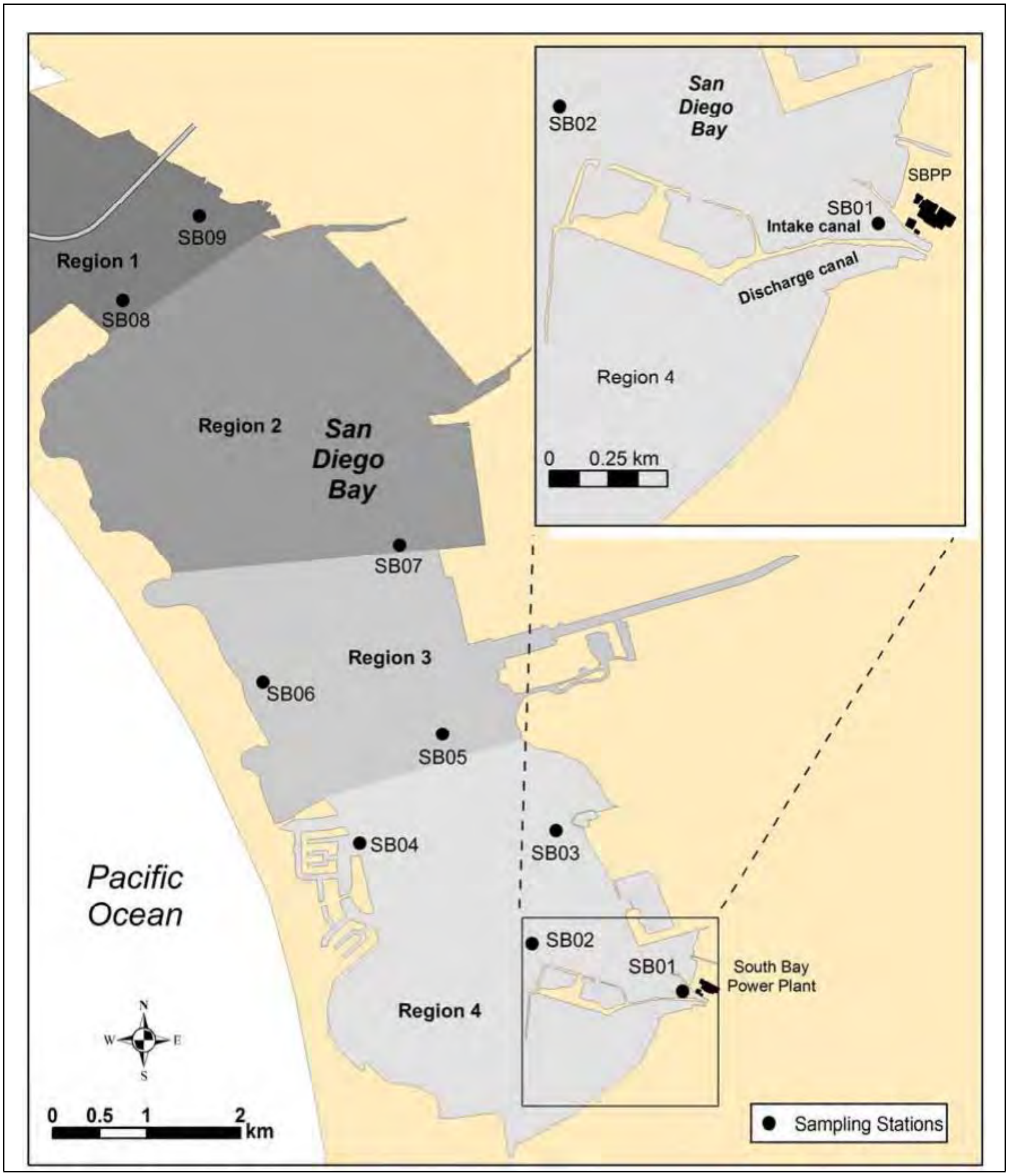


Figure 2-1. Location of South Bay Power Plant entrainment (SB01) and source water stations and detail of power plant intake area. Shaded areas represent regions of the bay used in calculating bay volumes.

The MBPP, operated by Duke Energy, is located on the northeastern shoreline of Morro Bay, which is approximately midway between San Francisco and Los Angeles, California (Figure 2-2). The plant draws water from Morro Bay for once-through cooling of its four electric generating units, which can produce a total of 1,002 MWe (Table 2-1). With all pumps in operation, water flow through the plant is 1,756 m^3min^{-1} (2.53 million m^3d^{-1}). Morro Bay studies were done as part of the permitting requirements for an upgrade to the plant that result in a decrease in flow to 1,086 m^3min^{-1} (1.56 million m^3d^{-1}). Therefore, all of the entrainment estimates and modeling were calculated using this flow rate.

Table 2-1. Characteristics of the South Bay (SBPP), Morro Bay (MBPP) and Diablo Canyon (DCPP) Power Plants.

Power Plant	Number of Power Generating Units	Total Maximum Megawatt Electric (Mwe) Output	Number of Circulating Water Pumps	Total Maximum Daily Flow (m^3)
SBPP	4	723	8 (2/unit)	$2.3x10^6$
MBPP	4	1,002	8 (2/unit)	$2.5x10^6$
DCPP	2	2,200	4 (2/unit)	$9.7x10^6$

The DCPP, operated by Pacific Gas and Electric Company, is located on the open coast midway between the communities of Morro Bay and Avila Beach on the central California coast in San Luis Obispo County (Figure 2-3). The intake structure for the plant is located behind two breakwaters that protect it from waves and surge. The plant has two nuclear-fueled generating units that can produce a total of 2,200 MWe (Table 2-1). With the main pumps and smaller auxiliary seawater system pumps in operation, total water flow through the plant is 6,731 m^3min^{-1} or (9.7 million m^3d^{-1}).

2.2 SOURCE WATER AND SOURCE POPULATION DEFINITIONS

The concept of defining the source water potentially affected by CWS operation is inherent in the assessment process, but was not defined as a necessary component of a 316(b) assessment until the recent publication of the new 316(b) rules. The new rules require all existing power plants with CWS capacities greater than 189,000 m^3d^{-1} to complete a Comprehensive Demonstration Study that includes a qualitative description of the source water. A more detailed quantitative definition of source water is not necessary for demographic modeling approaches, but is required to place calculated losses into context. The Empirical Transport Model (*ETM*) requires a more specific definition since the model calculates the conditional mortality due to entrainment on an estimate of the population of organisms in the source water that are potentially subject to entrainment.

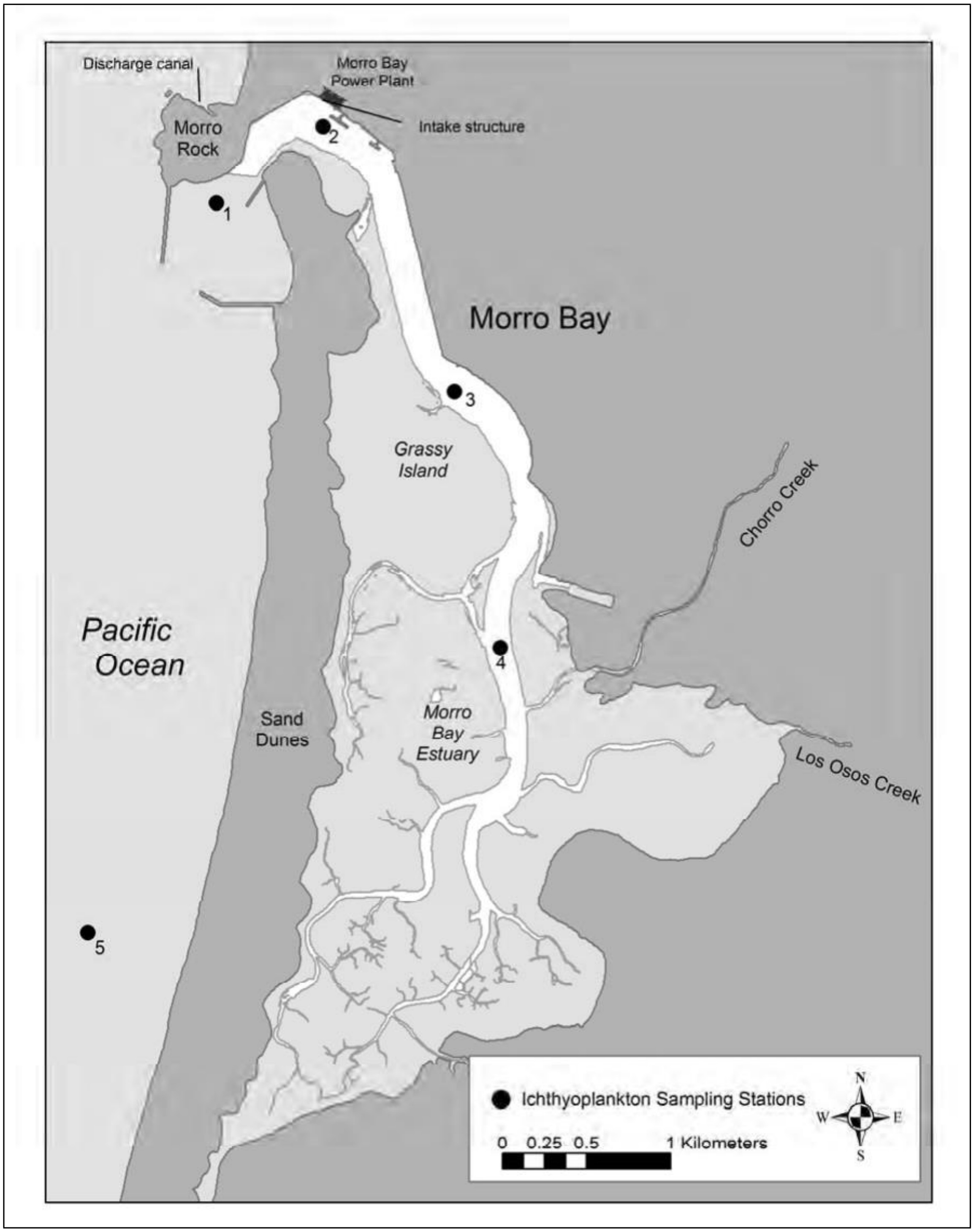


Figure 2-2. Locations of Morro Bay Power Plant entrainment (Station 2) and source water stations. White area depicts the main tidal channels in the bay, light gray areas are submerged at high tide, and dark gray areas are above the mean high tide line.

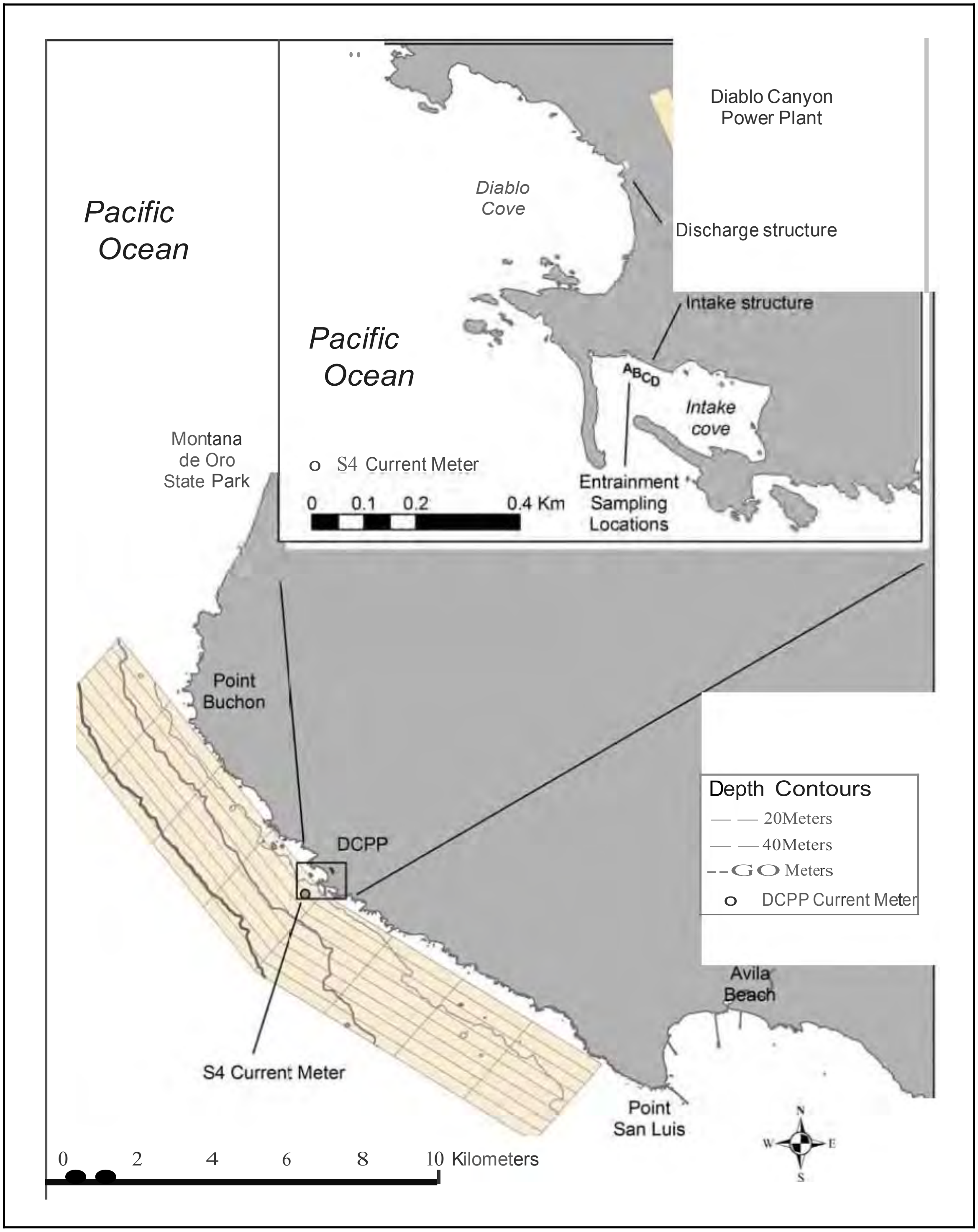


Figure 2-3. Locations of Diablo Canyon Power Plant (DCPP) entrainment stations (A, B, C, D, in insert) and source water sampling grid.

Critical to properly defining the source water for these studies was physical data that was either collected during the studies or from other sources to estimate the volume of the areas sampled and the total size of the source water. At SBPP and MBPP, hydrographic data collected for the study from several sources was used to estimate volume of the two water bodies. That volume was used as the total source water volume for SBPP. In addition to the volume of Morro Bay, current data from offshore and information on tides was used to estimate the total source water volume which included both bay and ocean components. Data from the same current meter used in the DCPP study were used in the MBPP study to calculate an average current speed over the period of January 1, 1996 – May 31, 1999. Current direction was ignored in calculating the average speed. The current speed was used to estimate unidirectional displacement over the period of time that the larvae in the sampling area offshore from Morro Bay were exposed to entrainment (described below). At DCPP, hydrographic data from National Oceanic and Atmospheric Administration was used to estimate the volumes of each of the 64-nearshore sampling stations (described below). In addition, data on alongshore and onshore current velocities were measured using an InterOceans S4 current meter positioned approximately 1 km west of the DCPP intake at a depth of approximately 6 m (Figure 2-3). The direction in degrees true from north and speed in cm/s were estimated for each hour of the nearshore study grid survey periods. These data were used to estimate the size of the area that could have acted as a source for larvae in the nearshore sampling area (described below).

South Bay Power Plant

The SBPP draws ocean water from the southernmost end of San Diego Bay (Figure 2-1). Allen (1999) divided San Diego Bay into four eco-regions and defined the south and south-central eco-regions as the area from the Coronado Bridge to the southern end of San Diego Bay. Analyses of current patterns and tidal dispersion were used to justify the use of the south and south-central eco-regions (south of the Coronado Narrows) as an appropriate source volume for the purposes of modeling the effects of entrainment by SBPP. These analyses were done by Dr. John Largier, formerly at Scripps Institute of Oceanography, and now at Bodega Marine Laboratory of the University of California at Davis, and Dr. David Jay, Oregon Health and Science University (Tenera Environmental 2004). The analysis of tidal currents measured at 18 locations throughout the interior of San Diego Bay showed that tidal currents exhibited a local maximum in the south bay at the Coronado Narrows and increased toward the bay mouth. Estimates of tidal dispersion were formed using data from the same 18 current meters, which showed spatial patterns generally similar to those from Largier (1995).

The results of Largier (1995) showed that tidal dispersion had a local maximum at the Coronado Narrows, consistent with the idea that the Narrows acts as the “mouth” of south bay. South of the Narrows currents and tidal dispersion are much reduced. Mixing throughout the south bay was estimated to take from one week to a month, typical of the period of time that the larvae were estimated to be exposed to entrainment. The results suggested that larvae are likely removed from the south bay primarily, but not exclusively, by dispersion and that advection may only be dominant during winter river-flow events. The analyses confirmed, in a quantitative manner, Allen’s (1999) definitions of eco-regions in San Diego Bay and helped verify the use of the Coronado Narrows as a logical seaward boundary for the SBPP source volume.

Since retention times in the south bay exceeded the average larval durations for most of the taxa examined, the source water was treated as a static volume. Volume was calculated 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 2-1). Computing the source volume required compiling the areas and volumes below fixed elevations (horizontal strata). Variations in tidal range required that the South Bay be divided into four regions, 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 were obtained from the U.S. Navy and supplemented with data collected for this study. Estimates of the average concentrations of the organisms inside the bay were multiplied by the sum of the estimated volumes from the four areas (Table 2-2) to obtain estimates of the bay source water populations that were used in the calculations of mortality for the *ETM*.

Table 2-2. Source water body surface area and water volume at mean water level (MWL) by region for south San Diego Bay.

Region	Datum	Height (m)	Area (m^2)	Volume (m^3)
1	MWL	0.90	4,241,241	33,754,018
2	MWL	0.90	10,173,006	70,387,388
3	MWL	0.91	6,355,524	25,060,179
4	MWL	0.93	9,556,875	20,410,508
			30,326,646	149,612,092

Morro Bay Power Plant

The MBPP source water was divided into two sub-areas, bay water and nearshore coastal water, because the location of the intake structure near the harbor entrance entrained both bay and nearshore taxa (Figure 2-2). The source water for MBPP could not be treated as a static volume, such as the source water for SBPP, because of the location of the power plant intake near the harbor entrance, which made it subject to tidal flows on a daily basis, and the smaller volume of the bay relative to an area such as San Diego Bay. To compensate for daily tidal movement past MBPP, the volume of the Morro Bay source water component was calculated as the sum of the bay's twice daily exchange of its 15.5 million m^3 tidal prism, adjusted for tidal exchange, (Mean High Water to Mean Low Water) and the bay's non-tidal volume of 5.4 million m^3. The volume of the tidal prism was adjusted to account for the portion of the Morro Bay outflow that returned with the incoming tide. Since volume was used to estimate the total supply of entrained larvae, inclusion of the re-circulated tidal prism volume would double count a portion of the larval supply and underestimate potential entrainment effects. This was accounted for using a tidal exchange ratio (TER), calculated for Morro Bay. The TER is the fraction of the total tidal exchange that consists of "new" water coming into the estuary, i.e., water that did not leave the estuary on the previous tidal cycle (Largier et al. 1996). In Morro Bay, the "total tidal exchange" is synonymous with the tidal prism, except for the amount estimated by TER.

The TER is difficult to estimate from measurements because the currents that prevail outside of any estuary mouth are complex and variable, and it is quite sensitive to processes inside and outside the estuary, especially complex currents, river inflow and density stratification (Largier et al. 1996). However, a method was developed (Largier et al. 1996) that measures the TER from the change in salinity of water flowing in and out of the entrance of an estuary. Applying this method, the Morro Bay TER was calculated to be between 70 and 80% of the average daily tidal prism by Dr. David Jay (Tenera Environmental 2001). A TER of 75% was used in calculating the bay source water volume, which was equal to the twice-daily tidal exchange of the average tidal prism, adjusted for the TER, added to the bay's non-tidal volume. Estimates of the average concentrations of organisms from the stations inside the bay (Stations 1−4) were multiplied by this volume to obtain estimates of the bay source water populations (Table 2-3). Since tidal exchange was used in calculating the source volume for Morro Bay, the plant's intake flow volume was calculated over a complete daily tidal cycle of two highs and two lows which was 24 hours and 50 minutes.

Table 2-3. Volumes for Morro Bay and Estero Bay source water sub-areas.

Area	Volume (m^3)
Morro Bay	15,686,663
Estero Bay Sampling Area	20,915,551

The area sampled outside Morro Bay in Estero Bay was treated as a static volume (Table 2-3) that was equal to the volume of Morro Bay uncorrected for tidal exchange. This volume for Estero Bay was used because it represented the volume of water exchanged with the bay that could be subject to entrainment. Estimates of the average concentrations of the organisms from the station just inside the bay (Station 1) and the station down-coast (Station 5) were multiplied by this volume to obtain estimates of the Estero Bay populations in the area sampled. The total size of the source water beyond the area sampled was estimated using ocean current data. Morro Bay and Estero Bay larval estimates were calculated separately so that the large source volume in Estero Bay did not inflate the source water estimates for bay taxa that were in much lower abundances outside the bay.

Diablo Canyon Power Plant

The DCPP nearshore sampling was designed to only provide information on abundance and distribution in the vicinity of DCPP of larval fishes and the invertebrates selected for detailed assessment, since it was recognized that the actual source water would be much larger for some taxa and also vary by taxa and seasonally due to changing oceanographic conditions. In establishing the nearshore sampling area, we considered that ocean currents in the area generally move both up and down the coast past DCPP. The currents also showed inshore/offshore oscillations, but these occurred less frequently and generally at a lower magnitude. The nearshore sampling area contained 64 stations or ‘cells’ (Figure 2-3) that was centered on the Intake Cove at DCPP. The northern extent of the sampling area was near Point Buchon and the southern half, a mirror image of the northern portion, extended to near Point San Luis. The shape of the sampling area reflected a slight bend (approximately 20º) in the coast at DCPP. The sampling area extended a distance of 8.7 km to both the north and south and an average distance of 3 km offshore. Regions inshore of the sampling area were in shallow water with partially submerged rocks, making the areas unsafe for boat operations and sampling. Volumes in each of the 64 cells were estimated using the surface area of the cell and the average depth based on available bathymetry data. The number of larvae in each cell was estimated by multiplying the average concentration during each survey by the volume of water sampled.

2.3 SAMPLING

Sampling at all three of the facilities was designed to provide estimates of both entrainment and source water concentrations that accounted for the differences in the cooling water volumes at the three plants and were representative of the range of habitats and organisms potentially affected by entrainment in each area. As a result of the differences among the three plants and funding available, the combined entrainment and source water sampling efforts ranged from five stations for the MBPP study to 68 stations for the DCPP study.

Sample collection methods were similar to those developed and used by CalCOFI in their larval fish studies (Smith and Richardson 1977). Sampling at all three plants was conducted using a bongo frame with two 71-cm diameter rings with plankton nets constructed of 333-*u*m mesh. Each net was fitted with a Dacron sleeve and a cod-end container to retain the organisms. Each net was equipped with a calibrated General Oceanics flowmeter, which allowed the calculation of the amount of water filtered. Net lengths varied according to the depth of the water sampled. Shorter nets, 1.8 m in length, were used for entrainment sampling in the shallower intake cove at DCPP. Longer nets, 3.3 m in length were used for all other sampling. All of the nets were lowered as close to the bottom as possible and retrieved using oblique or vertical tows to sample the entire water column. Once the nets were retrieved from the water all of the collected material was rinsed into the codend. The target volume of each tow at both the entrainment and source water stations was 40-60 m^3 for both nets combined. The sample volume was checked when the nets reached the surface and the tow continued or started over if the target volume was not collected. The contents of both nets were either combined into one sample immediately after collection, or treated as a single sample for analysis.

Entrainment sampling at all three plants was done in the waters outside of the plant CWIS as close as possible to the intake structure bar racks. This sampling design assumed that the concentrations from the waters in front of the CWIS are the same as the concentrations in the cooling water flow. Sampling was done outside of the CWIS because of the numerous problems involved in sampling inside the plant or at the discharge. Sampling inside the plant usually involves sampling with a pump that generally obtains a small volume relative to plankton nets in a given period of time. Although samples inside the CWIS may be well mixed, the cooling water flow inside the system is exposed to biofouling organisms that can significantly reduce the concentration of larval fish and other organisms. Sampling outside the plant also allowed entrainment samples to be used in characterizing source water populations. This was critical to the *ETM* calculations and allowed source water estimates to be calculated for taxa that may have only been collected from entrainment samples.

South Bay Power Plant

Entrainment and source water sampling was conducted monthly from January 2001 through January 2002 (Tenera Environmental 2004). Entrainment samples were collected from Station SB1 located in the SBPP intake channel (Figure 2-1). Each tow proceeded out the intake channel against the prevailing intake current. The intake channel was bounded by a separation dike to the south and a shallow mudflat to the north, and there was a constant current flow toward the intake structure. Therefore it was assumed that all of the water sampled at the entrainment station would be 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 (Figure 2-1). A single tow was completed at each of the source water stations during each of the six 4-hr cycles.

The stations for the SBPP study (Figure 2-1) 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. The source water stations ranged in depth from approximately –2 m Mean Lower Low Water (MLLW) at SB8 to –12 m MLLW at SB9. 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 water. For example, stations on the east side of the bay were adjacent to salt marsh habitat and would tend to have a greater proportion of larvae from fishes with demersal eggs that spawned in salt marsh channels, such as gobies, 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 (*Engraulis mordax*).

Morro Bay Power Plant

Entrainment and source water sampling was conducted from December 1999 through December 2000 (Tenera Environmental 2001). Entrainment samples were collected weekly from in front of the MBPP intake structures (Station 2; Figure 2-2). Samples were collected over a continuous 24-hour period with each period divided into six, 4-hour sampling cycles. Two tows were conducted during each cycle. During the same period, monthly source water samples were collected at four stations in addition to the entrainment station (Figure 2-2). Initially, source water surveys were collected twice per day during daylight hours on high and low tides, but after two months of sampling in February 2000, sample collection for source water surveys was expanded to cover the entire 24-hour period and was no longer linked to tidal cycle.

Fewer stations were sampled in the MBPP study relative to the SBPP study due to the smaller size of the estuary. Station 1 was located just inside the entrance to Morro Bay and was intended to characterize water from outside the bay that was subject to entrainment during incoming tides. Only two other source water stations (stations 3 and 4) were located in Morro Bay because the areas that could be sampled in the south part of the bay were limited to narrow navigation channels. This was not considered to be a problem because of the large tidal prism relative to the size of the bay resulted in shallower portions of the bay draining through the deeper navigation channels where the sampling occurred. Station 5 was located outside of the bay approximately 4.7 km down coast (i.e., south of the harbor mouth) and was intended to characterize open coastal taxa potentially subject to entrainment.

Diablo Canyon Power Plant

Collection of the DCPP entrainment samples occurred from October 1996 through June 1999 (Tenera Environmental 2000). This was the longest period of sampling among the three studies. The sampling was continued longer than one-year because of El Niño conditions during the first year, which were agreed by the Technical Workgroup as not representative of normal conditions. Entrainment samples were collected once per week from four permanently moored sampling stations located directly in front of the intake structure that were sampled in a random order during eight 3-hour cycles (Figure 2-3). Two samples were collected at each station during each cycle. The first 9 surveys were collected with 505 *u*m mesh nets, but due to extrusion of larval fishes through the net mesh observed during these first few surveys, subsequent surveys were collected with 335 *u*m mesh.

The boundaries and shape of the nearshore sampling area were chosen to ensure that the area would be large enough to characterize the larvae from the fishes potentially influenced by the large volume of the DCPP CWIS, and would be representative of the variety of nearshore habitats found in the area. These were the same reasons used to justify the large sampling effort (64 stations) relative to the SBPP and MBPP studies. Sampling of the nearshore study area occurred monthly from July 1997 through June 1999. Two randomly positioned stations within each of the 64 cells of the grid were sampled once each survey. The study grid was sampled continuously over 72 hours using a “ping-pong” transect to limit temporal and spatial biases in the sampling pattern and to optimize shipboard time. The starting cell (constrained to the 28 cells on the perimeter of the grid) and the initial direction of the transect (constrained to the two cells diagonally, adjacent to the starting cell) were selected at random. When the adjacent diagonal cell had previously been sampled, one of the two adjacent cells in the direction of travel was randomly selected to be sampled next. To minimize temporal variation between entrainment and study grid sampling, source water surveys were scheduled to bracket the 24-hour entrainment survey, overlapping by one day before and after the collection of entrainment samples.

Entrainment and nearshore sampling efforts did not start at the same times and therefore the entire sampling period was divided into five analysis periods. All of the weekly entrainment samples from October 1996 through November 1998 were processed so this period was divided into two yearlong analysis periods. Results for these periods are not presented because they were only used to generate estimates directly from entrainment data. The nearshore sampling period was also divided into two yearlong analysis periods. Only the entrainment samples collected during the sampling of the nearshore area were processed from December 1998 through June 1999 so entrainment data from July 1998 through June 1999 were used to generate model estimates for a fifth analysis period that could be directly compared with model estimates that incorporated data from the nearshore sampling area.

2.4 SELECTION OF TAXA FOR DETAILED ASSESSMENT

Although almost all planktonic forms (phyto-, zoo-, and ichthyoplankton) are affected by entrainment, these three studies and most other 316(b) studies have focused on a few organism groups, typically ichthyoplankton and zooplankton. The effects on phytoplankton and invertebrate holoplankton are typically not studied because their large abundances, wide distributions, and short generation times should make them less susceptible to CWIS impacts. The groups of organisms selected for assessment in these studies included larval fishes and larvae from commercially or recreationally important invertebrates such as *Cancer* spp. crabs and California spiny lobster (*Panulirus interruptus*).

The workgroup also looked at including kelp spores, fish eggs, squid paralarvae, and abalone and bivalve larvae in the assessment. The risk of a significant impact on adult kelp populations by entrainment of kelp spores was determined to be negligible due to the large number of spores produced along the coast. Additionally, it is not possible to identify the species of kelp based on gametes or spores. Fish eggs were not included because they are difficult to identify to species and the most abundant fishes in these studies had egg stages that were not likely to be entrained; they either have demersal/adhesive eggs or are internally fertilized and extrude free-swimming larvae. Squid paralarvae are also unlikely to be entrained because they are competent swimmers immediately after hatching. Abalone larvae were not included because they are at low risk of entrainment and cannot be effectively sampled or identified during early life stages when they would be susceptible to entrainment (Tenera Environmental 1997). In addition, algal spores, fish eggs, and abalone and bivalve larvae would all require smaller mesh than the mesh used for ichthyoplankton and separate sampling efforts.

The final list of fish and invertebrates analyzed in each of the studies (Table 2-4) was determined by technical workgroups after all of the samples had been processed and data from the entrainment samples summarized. The assessments included taxa from the organism groups that were in highest abundance in the entrainment samples (generally those comprising up to 90% of the total abundance) and commercially or recreationally important fishes and invertebrates that were in high enough abundances to allow for their assessment. It was also realized that organisms having local adult and larval populations (i.e., source not sink species) were more important than species such as the northern lampfish (*Stenobrachius leucopsarus*), which is an offshore, deep-water species whose occurrence in entrainment was likely due to onshore currents that transported the larvae into coastal waters from their primary habitat. These 'sink species' were not included in the assessments.

Table 2-4. Taxa used in assessments at South Bay (SBPP), Morro Bay (MBPP) and Diablo Canyon (DCPP) power plants.

Scientific Name	Common Name
SBPP – taxa comprising 99 percent of total entrainment abundance	
Clevlandia ios, Ilypnus gilberti, Quietula y-cauda	CIQ goby complex
Gillichthys mirabilis	longjaw mudsucker
Anchoa spp.	anchovies
Atherinopsidae	silversides
Hypsoblennius spp.	combtooth blennies
MBPP – taxa comprising 90 percent of total entrainment abundance plus commercial taxa	
unidentified Gobiidae	gobies
Leptocottus armatus	Pacific staghorn sculpin
Stenobrachius leucopsarus	northern lampfish
Quietula y-cauda	shadow goby
Hypsoblennius spp.	combtooth blennies
Sebastes spp. V_De	KGB rockfishes
Atherinopsis californiensis	jacksmelt
Clupea pallasii	Pacific herring
Genyonemus lineatus	white croaker
Scorpaenichthys marmoratus	cabezon
Cancer antennarius	brown rock crab
Cancer jordani	hairy rock crab
Cancer anthonyi	yellow crab
Cancer gracilis	slender crab
Cancer productus	red rock crab
Cancer magister	Dungeness crab
DCPP – ten most abundant taxa plus commercial taxa	
Sardinops sagax	Pacific sardine
Engraulis mordax	northern anchovy
Sebastes spp. V / *S. mystinus*	blue rockfish complex
Sebastes spp. V_De/V_D_	KGB rockfish complex
Oxylebius pictus	painted greenling
Artedius lateralis	smoothhead sculpin
Orthonopias triacis	snubnose sculpin
Scorpaenichthys marmoratus	cabezon
Genyonemus lineatus	white croaker
Cebidichthys violaceus	monkeyface prickleback
Gibbonsia spp.	Clinid kelpfishes
Rhinogobiops nicholsii	blackeye goby
Citharichthys spp.	sanddabs
Paralichthys californicus	California halibut
Cancer antennarius	brown rock crab
Cancer gracilis	slender crab

The list of taxa reveals one of the problems with these studies. In some cases larvae cannot be identified to the species level and can only be identified into broader taxonomic groupings. Myomere and pigmentation patterns were used to identify many species, however this can be problematic for some species. For example, sympatric members of the family Gobiidae share morphologic and meristic characters during early life stages (Moser 1996) making identification to the species level difficult. In the MBPP study we grouped those gobiids we were unable to identify to species into an “unidentified gobiid” category (i.e., unidentified Gobiidae). In the SBPP study we were able to determine that the unidentified gobies were comprised of three species (Table 2-4). Larval combtooth blennies (*Hypsoblennius* spp.) can be easily distinguished from other larval fishes (Moser 1996). However, the three sympatric species along the central California coast cannot be distinguished from each other on the basis of morphometrics or meristics. These combtooth blennies were grouped into the “unidentified combtooth blennies” category (i.e., *Hypsoblennius* spp.). Many rockfish species (*Sebastes* spp.) are closely related, and the larvae share many morphological and meristic characteristics, making it difficult to visually identify them to species (Moser et al. 1977, Moser and Ahlstrom 1978, Baruskov 1981, Kendall and Lenarz 1987, Moreno 1993, Nishimoto in prep.). Identification of larval rockfish to the species level relies heavily on pigment patterns that change as the larvae develop (Moser 1996). Of the 59 rockfishes known from California marine waters (Lea et al. 1999), at least five can be reliably identified to the species level as larvae (Laidig et al. 1995, Yoklavich et al. 1996): blue rockfish (*Sebastes mystinus*), shortbelly rockfish (*S. jordani*), cowcod (*S. levis*), bocaccio (*S. paucispinis*), and stripetail rockfish (*S. saxicola*). The *Sebastes* larvae we collected could only be identified into broad sub-generic groupings based on pigment patterns; these larvae were grouped using information provided by Nishimoto (in prep.; Table 2-5). The use of these broad taxonomic categories presents problems in determining the most appropriate life history parameters to use in the demographic models. This involved calculating an average value or determining the most appropriate value from different sources and species.

Table 2-5. Pigment groups of some preflexion rockfish larvae from Nishimoto (in-prep).

The code for each group is based on the following letter designations:

V_ = long series of ventral pigmentation (starts directly at anus)	De = elongating series of dorsal pigmentation (scattered melanophores after continuous ones)
V = short series of ventral pigmentation (starts 3-6 myomeres after anus)	d = develops dorsal pigmentation (1-2 or scattered melanophores)
D_ = long series of dorsal pigmentation (4 or more in a continuous line) extending to above anus	P = pectoral blade pigmentation
D = short series of dorsal pigmentation (4 or more in a continuous line) not extending to anus	p = develops pectoral pigmentation (1-2 or scattered melanophores)

CODE	SPECIES	COMMON NAME
V D	Long ventral series, short dorsal series, no pectoral pigment	
	S. atrovirens	kelp
	S. chrysomelas	black and yellow
	S. maliger	quillback
	S. nebulosus	China
	S. semicinctus	halfbanded
V De	Long ventral series, elongating dorsal series, pectoral pigment	
Or	S. auriculatus	brown
V DeP	*S. carnatus*	gopher
Or	*S. caurinus*	copper
V dep	*S. dalli*	calico
	S. rastrelliger	grass
V	Short ventral series, no dorsal series, no pectoral	
	S. aleutianus	rougheye
	S. alutus	Pacific Ocean perch
	S. brevispinis	silvergrey
	S. crameri	darkblotched
	S. diploproa	splitnose
	S. elongatus	greenstriped
	S. macdonaldi	Mexican
	S. miniatus	vermilion
	S. nigrocinctus	tiger
	S. proriger	redstripe
	S. rosaceus	rosy
	S. ruberrimus	yelloweye
	S. serriceps	treefish
	S. umbrosus	honeycomb
	S. wilsoni	pygmy
	S. zacentrus	sharpchin

2.5 OTHER BIOLOGICAL DATA

All of the assessment models required some life history information from a species to enable the calculation of entrainment effects. Age-specific survival and fecundity rates are required for the fecundity hindcasting (*FH*) and adult equivalent loss (*AEL*) demographic models. Calculation of *FH* requires egg and larval survivorship up to the age of entrainment plus estimates of lifetime fecundity, while *AEL* requires survivorship estimates from the age at entrainment to adult recruitment. Species-specific survivorship information (e.g., age-specific mortality) from egg or larvae to adulthood was not available for many of the taxa considered in the assessments at the three plants. Life history information was gathered from the scientific literature and other sources. Uncertainty surrounding published life history parameters is seldom known and rarely reported, but the likelihood that it is very large needs to be considered when interpreting results from the demographic approaches for estimating entrainment effects. Accuracy of the estimated entrainment effects from demographic models such as *FH* and *AEL* depend on the accuracy of age-specific mortality and fecundity estimates. In addition, these data are unavailable for many species limiting the application of these models to large numbers of species.

All three modeling approaches (*FH*, *AEL*, and *ETM*) required an age estimate of the entrained larvae. The larval ages were estimated using the length of the entrained larvae and an estimate of the larval growth rate for each species obtained from the scientific literature and other sources. The size range from the minimum to the average size of the larvae was used to calculate the average age of the entrained larvae that was used in the *FH* and *AEL* models, while the size range from the minimum to the maximum size of the larvae was used to calculate the maximum age of the entrained larvae and the period of time that the larvae were subject to entrainment for the *ETM* model. Minimum and maximum lengths used in these calculations were adjusted to account for potential outliers in the measurements by using the 1st and 99th percentile values in the calculations. These values were chosen based on our examination of the distributions of the length measurements and other values may be more appropriate for other studies or species depending upon the data. The size range was estimated for each taxon from a representative sample of larvae from the SBPP and MBPP studies, while all of the entrained larvae of the taxa selected for detailed assessment were measured from the DCPP study. All of the measurements were made using a video capture system attached to a microscope and Optimas^{TM} image analysis software.

2.6 DATA REDUCTION

Entrainment Estimates

Estimates of daily larval entrainment for all ichthyoplankton and selected invertebrate larvae for all of the plants were calculated from data collected at the entrainment stations located directly in front of the power plant intake structures. Daily entrainment estimates were used to calculate daily incremental entrainment mortality estimates used in the *ETM*. Estimates of entrainment over annual study periods were used in the *FH* and *AEL* demographic modeling.

Daily entrainment estimates and their variances were derived from the mean concentration of larvae (number of larvae per cubic meter of water filtered) calculated from the samples collected during each 24-hr entrainment survey. These estimates were multiplied by the daily intake flow volume for each plant (MBPP and SBPP studies used engineering estimates of cooling water flow and DCPP used actual daily flow) to obtain the number of larvae entrained per day for each taxon as follows:

$$E_i = v_i \cdot \overline{\rho}_i , \qquad (1)$$

where v_i = total intake volume for the survey day of the i^{th} survey period, and $\overline{\rho}_i$ = average concentration for the survey day of the i^{th} survey period.

Entrainment was estimated for the days within each weekly (MBPP and DCPP) or monthly survey period (SBPP). The number of days in each period was determined by setting the sampling date at the mid-point between sample collections. Daily cooling water intake volumes were then used to calculate entrainment for the study period by summing the product of the entrainment estimates and the daily intake volumes for each survey period. These estimates and their associated variances were then added to obtain annual estimates of total entrainment and variance for each taxon as follows:

$$E_T = \sum_{i=1}^{n} \left(\frac{V_i}{v_i} \right) E_i , \qquad (2)$$

where

v_i = intake volume on the survey day of the ith survey period (i=1,...,n);
V_i = total intake volume for the ith survey period (i=1,...,n); and
E_i = the estimate of daily entrainment during the entrainment survey of the ith survey period.

with an associated variance of

$$Var(E_T) = \sum_{i=1}^{n} \left(\frac{V_i}{v_i}\right)^2 Var(E_i), \tag{3}$$

using the sampling variances of entrainment on the survey day of the i^{th} period, $Var(E_i)$. The daily sampling variance for SBPP and MBPP was calculated using the average concentrations from samples collected during each cycle, while the daily sampling variance for DCPP was calculated by treating each sampling cycle as a separate strata using data from the four entrainment stations. Both methods underestimated the true variance because they did not incorporate the variance associated with the within-survey period variation and daily variations in intake flow due to waves, tide, and other factors not measured by the power plant. One hundred percent mortality was assumed for all entrained organisms.

For the study at DCPP estimates of annual entrainment were scaled to better represent long-term trends for a taxa by using ichthyoplankton data collected inside the Intake Cove at DCPP (Figure 2-3). These data were used to calculate an index of annual trends in larval abundance for the period of 1990 through 1998. This multi-year annualized index consisted of five months (February–June) of larval fish concentrations from 1990, six months (January–June) from 1991, and seven months (December–June) from all subsequent years. The estimated annual entrainment (E_T) was adjusted to the long-term average using the following equation:

$$E_{Adj-T} = \left(\frac{\bar{I}}{I_i}\right) \cdot E_T, \tag{4}$$

where

E_{Adj-T} = adjusted estimate of total annual entrainment to a long-term average, 1990 – 1998;

I_i = index value from DCPP Intake Cove surface plankton tows for each ith year; and

$\bar{I}$ = average index value from DCPP Intake Cove surface plankton tows, 1990 – 1998.

The abundances used in calculating the index were not expected to be representative of the abundances calculated from the DCPP entrainment data since they were only collected during 5 to 7 months of the year in contrast to the entrainment sampling that occurred continuously from October 1996 through June 1999. The use of the index assumes that the difference in abundance is approximately equal over time, although the validity of this assumption probably varied among taxa. Variance for adjusted annual entrainment can then be expressed as follows:

$$Var(E_{Adj-T}) = \left(\frac{\bar{I}}{I_i}\right)^2 \cdot Var(E_T), \tag{5}$$

assuming the indices are measured without error. Ignoring the sampling error of the indices will underestimate the true variance, but will qualitatively account for the change in scale associated with multiplying the annual entrainment estimate by a scalar. The variance of E_{Adj-T}, however, does not take into account the between-day, within-station variance, interannual variation, nor the variance associated with the indices used in the adjustment. Hence, the actual variance of the E_{Adj-T} estimate is likely to be greater than the value expressed above.

The Intake Cove surface tow index was assumed to have the following relationship:

$$E(I_i) = C \cdot E_i\,, \tag{6}$$

where

$E(I_i)$ = expected value of the index for the ith year;
E_i = entrainment for the ith year; and
C = proportionality coefficient.

If this relationship holds true and the differences over time are constant, then the inter-annual variance in the index has the following relationship:

$$Var(I_i) = C^2 Var(E_i). \tag{7}$$

Therefore, the coefficients of variation (CV) for I and E across n years have the following relationship:

$$CV(\bar{I}) = \frac{\sqrt{\frac{Var(I)}{n}}}{\bar{I}} \quad \frac{\sqrt{\frac{C^2 Var(E)}{n}}}{C\bar{E}} = CV(\bar{E})\,. \tag{8}$$

Hence, the CV for the Intake Cove surface tow index should be a measure of the CV for entrainment across years. In the case of E and I, variances include sampling errors that may not be equal. Therefore, the CV of I was used to estimate variation in entrainment across years.

The use of adjusted entrainment in *FH* and *AEL* models at DCPP provided results that better represented average long-term effects. Adjusted entrainment values were not used in calculating *ETM* results because the computation of *ETM* relies on a proportional entrainment (*PE*) ratio using estimates from paired entrainment and nearshore larval sampling. Moreover, if the assumptions of the *ETM* model are valid, then the estimate already represents average long-term entrainment effects because the *PE* ratio should largely be a function of the ratio of the cooling water to source water volumes, which is constant if the plant is operating at full power compared to ichthyoplankton abundances that vary over time. This would especially be true if the *PE* were averaged over several taxa, assuming that the effects of larval behavior cancel across all the species. As a result the use of adjusted entrainment in *FH* and *AEL* models also provided a better basis to compare results from all three models when they were converted into a common currency through the use of population or fishery stock assessments. This advantage of the ETM could be affected if actual cooling water flows varied considerably seasonally and among years.

2.7 SOURCE WATER ESTIMATES

Average concentrations calculated from source water stations were used to estimate source water populations of species or taxa groups using the same method used for calculating entrainment estimates for each i^{th} survey period. At SBPP a single source water estimate was calculated, while at MBPP, separate estimates were calculated for Morro Bay and Estero Bay source water components.

At DCPP separate estimates were calculated for each of the 64 grid stations based on the depth and surface area of each station. In addition, an adjustment was made to the estimated number of larvae in the row 1 cells of the study grid to help compensate for the inability to safely collect samples inshore of the grid (Figure 2-3). The estimated volume of water directly inshore of the study grid was multiplied by the concentration of larvae collected in the row 1 cells, except for cells directly offshore from the power plant and the cell furthest upcoast which is more offshore than the rest of the cells in row 1 due to the bend in the coastline at Point Buchon. The adjustment was not done for the volume of water inshore of that cell because it would have added a substantial volume to that cell and the composition and abundance would not have been representative of the other inshore areas. The average concentration from the entrainment stations was used for the areas inshore from the two cells directly offshore from the Intake Cove where entrainment samples were collected. The estimated number of larvae in each grid station and from the areas inshore of the grid was added to obtain an estimate of the sampled source water populations.

2.8 IMPACT ASSESSMENT MODELS

Demographic Approaches

Adult equivalent loss models (Goodyear 1978) evolved from impact assessments that compared power plant losses to estimates of adult populations or commercial fisheries harvests. In the case of adult fishes impinged by intake screens, the comparison was relatively straightforward. To compare numbers of impinged sub-adults and juveniles and entrained larval fishes to adults, it was necessary to convert these losses to adult equivalents using demographic factors such as survival rates. Horst (1975) provided an early example of the equivalent adult model (*EAM*) to convert numbers of entrained early life stages of fishes to their hypothetical adult equivalency. Goodyear (1978) extended the method to include survival for several age classes of larvae.

Demographic approaches, exemplified by *EAM*, produce an absolute measure of loss beginning with simple numerical inventories of entrained or impinged individuals and increasing in complexity when the inventory results are extrapolated to estimate numbers of adult fishes or biomass. We used two different but related demographic approaches in assessing entrainment impacts at all three facilities: *AEL* (Goodyear 1978), which uses the larval losses to estimate the equivalent number of adult fishes that would not have been lost to the population and *FH* (Horst 1975, Goodyear 1978, MacCall, pers. comm.), which estimates the number of adult females at the age of maturity whose reproductive output has been lost due to entrainment. The method is similar to the Egg Production Method described by Parker (1980, 1985) and implemented in Parker and DeMartini (1989) at San Onofre Nuclear Generating Station except they used only eggs to hindcast adult equivalents.

Both *AEL* and *FH* approaches require an estimate of the age at entrainment for each taxon that was estimated by dividing the difference between the smallest (represented by the 1st percentile value) and the average lengths of a representative sample of larvae measured from the entrainment samples by a larval growth rate obtained from the literature. This assumes that the period of vulnerability to entrainment starts when the larvae are either hatched or released and that the smallest larvae in our samples represent newly hatched or released larvae. This minimum value was checked against reported hatch and release sizes for the taxa analyzed in these studies and in most cases was less than these reported values.

Additionally, age-specific survival and fecundity rates are required for calculating *FH* and *AEL*. *FH* requires egg and larval survivorship up to the age of entrainment plus estimates of fecundity, age at maturity and longevity, while *AEL* requires survivorship estimates from the age at entrainment to adult recruitment. Furthermore, to make estimation practical, the affected population is assumed to be stable and stationary, and age-specific survival and fecundity rates are assumed to be constant over time. In addition, the *FH* method assumes that all of the females instantaneously reach 100% maturity at the age of maturity.

Species-specific survivorship information from egg or larvae to adulthood was limited for many of the taxa considered in these studies. These rates when available were inferred from the literature along with estimates of uncertainty. Uncertainty surrounding published demographic parameters is seldom known and rarely reported, but the likelihood that it is very large needs to be considered when interpreting results from the demographic approaches for estimating entrainment effects. The ratio of the standard deviation to the mean (*CV*) was assumed to be 30% for all life history parameters used in the models for the SBPP and MBPP studies and 100% for the DCPP study. The larger *CV* was used at DCPP because it was the first study we conducted and we wanted to use a large *CV* to ensure that the confidence intervals adequately reflected the large degree of uncertainty associated with the estimates. The smaller *CV* used for SBPP and MBPP does not reflect increased confidence in the life history data, but our realization that the larger *CV* used at DCPP resulted in confidence intervals for the estimates that spanned several orders of magnitude minimizing their usefulness in the assessment.

Fecundity Hindcasting

The *FH* approach couples larval entrainment losses to adult fecundity using survivorship between stages to estimate the numbers of adult females at the age of maturity whose reproductive output has been lost due to entrainment, i.e., hindcasting the numbers of adult females at the age of maturity effectively removed from the reproductively active population. Accuracy of the estimate of impacts using this model is dependent upon an accurate estimate of survival from parturition through the estimated average age at entrainment and total lifetime female fecundity. If it can be assumed that the adult population has been stable at some current level of exploitation and that the male:female ratio is constant at 50:50, then fecundity and mortality are integrated into an estimate of adult loss at the age of female maturity by converting entrained larvae back into adult females and multiplying by two to approximate the total number of equivalent adults at the age of female maturity.

A potential advantage of *FH* is that survivorship need only be estimated for a relatively short period of the larval stage (e.g., egg to larval entrainment). The method requires age-specific mortality rates and fecundities to estimate equivalent adult losses. Furthermore, this method, as applied assumes a 50:50 male:female ratio, hence the loss of a single female's reproductive potential was equivalent to the loss of two adult fish. Other assumptions included the following:

- Life history parameter values from the literature are representative of the population for the years and location of the study.
- Size of the stock does not affect survivorship or the rate of entrainment mortality (no density dependence).
- Reported values of egg mass were lifetime averages in order to calculate an unbiased estimate of lifetime fecundity.
- Total lifetime fecundity was accurately estimated by assuming that the mortality rate was uniform between age-at-maturity and longevity.
- 'Knife-edge' recruitment into the adult population at the age of maturity.
- Loss of the reproductive potential of one female was equivalent to the loss of an adult female at the age of maturity.

The estimated number of females at the age of maturity whose lifetime reproductive potential was lost due to entrainment was calculated for each taxon as follows:

$$FH = \frac{E_T}{TLF \cdot \prod_{j=1}^{n} S_j}, \qquad (9)$$

where

E_T = total entrainment estimate;

S_j = survival rate from parturition to the average age of the entrained larvae at the end of the j^{th} stage; and

TLF = average total lifetime fecundity (*TLF*) for females, equivalent to the average number of eggs spawned per female over their reproductive years.

While E_T was used in the modeling at SBPP and MBPP, $E_{Adj\text{-}T}$ was used at DCPP. In practice, survival was estimated by either one or several age classes, depending on the data source, to the estimated age at entrainment. The expected *TLF* was approximated by the following expression:

$$\begin{aligned} TLF &= \text{Average eggs/year} \cdot \text{Average number of years of reproductive life} \\ &= \text{Average eggs/year} \cdot \left(\frac{\text{Longevity - Age at maturation}}{2} \right). \end{aligned} \quad (10)$$

The number of years of reproductive potential was approximated as the midpoint between the ages of maturity and longevity. This approximation was based on the assumption of a linear uniform survivorship curve between these events (i.e., a uniform survival rate). Total lifetime fecundity for the studies at SBPP was calculated by adding 1 to the difference between longevity and age-at-maturity. This was done to account for spawning during the two ages used in the calculation. For heavily exploited species such as northern anchovy and sardine (*Sardinops sagax*), the expected number of years of reproductive potential may be much less than predicted using this assumption. Therefore, for the DCPP study the estimated longevity for heavily exploited fishes was based on the oldest observed individual caught by the fishery, rather than by the oldest recorded fish. If life table data are available for a taxon, then the lifetime fecundity should be estimated directly rather than using the approximation presented in Equation10. The variance of *FH* was approximated by the Delta method (Seber 1982) and is presented in Appendix A.

Adult Equivalent Loss

The *AEL* approach uses abundance estimates of entrained or impinged organisms to project the loss of equivalent numbers of adults based on stage-specific survival and age-at-recruitment (Goodyear 1978). The primary advantage of this approach, and of *FH*, is that it translates power plant-induced early life-stage mortality into numbers of adult fishes, which are familiar units to resource managers. Adult equivalent loss does not require source water estimates of larval abundance in assessing effects. This latter advantage may be offset by the need to gather age-specific mortality rates to predict adult losses and the need for information on the adult population of interest for estimating population-level effects (i.e., fractional losses). Other assumptions of *AEL* using data on survivorship from entrainment to recruitment into the fishery assume the following:

- Published values of life history parameters are representative of the fish population in the years and location for the specific study.
- If survivorship values from the literature are limited to single observations, values are assumed constant over time or representative of the mean survivorship.
- Survival rates used in the calculations are representative and constant for the life stage of the larvae or fish in the calculations.
- Size of the stock does not affect survivorship or the rate of entrainment mortality (no density dependence).

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

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

For fish species where larval survival data are missing, expected survival could be estimated using fecundity combined with juvenile and adult survival data. This approach requires the following additional assumption:

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

Starting with the number of age class j larvae entrained, it is conceptually easy to convert the numbers to an equivalent number of adults lost at some specified age class using the following formula:

$$AEL = \sum_{j=1}^{n} E_j S_j \,, \tag{11}$$

where,

n = number of age classes;

E_j = estimated number of larvae lost per year in age class j; and

S_j = survival rate for the j^{th} age class of the 1..n classes between entrainment and adulthood.

In practice, survival was estimated by either one or several age classes, depending on the data source, from the estimated age at entrainment to recruitment into the fishery. Survivorship to recruitment, at an adult age, was apportioned into several age stages, and *AEL* was calculated as follows:

$$AEL = E_T \prod_{j=1}^{n} S_j \ , \tag{12}$$

where,

S_j = survival rate over the j[th] age class.

The variance of *AEL* was approximated by the Delta method (Seber 1982) and is presented in Appendix A.

Alignment of FH and AEL Estimates

AEL and *FH* can be compared by assuming a stationary population where an adult female must produce two adults (i.e., one male and one female). These two adults are products of survival and total lifetime fecundity (*TLF*) modeled by the following expression:

$$2 = S_{egg} \cdot S_{larvae} \cdot S_{adult} \cdot TLF, \tag{12}$$

which leads to the following:

$$S_{adult} = \frac{2}{TLF \cdot S_{egg} \cdot S_{larvae}}. \tag{13}$$

Substituting into the overall form of the following *AEL* equation:

$$AEL = E_T \cdot S_{adult} \ , \tag{14}$$

yields the following:

$$AEL = \frac{2(E_T)}{S_{egg} \cdot S_{larva} \cdot TLF}. \tag{15}$$

Assuming a 50:50 sex ratio, without independent survival rates, *AEL* and *FH* are deterministically related as $AEL \equiv 2FH$. The two estimates can be aligned so that female age at maturity is also the age of recruitment used in computing *AEL*. Otherwise, an alignment age can be accomplished by solving the simple exponential survival growth equation (Ricker 1975, Wilson and Bossert 1971):

$$N_t = N_0 \cdot e^{-Z(t-t_0)} \ , \tag{16}$$

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 for a population under equilibrium in either direction so they are either hindcast or extrapolated to the same age such that *AEL*≡2*FH*. Estimates of entrainment mortality calculated from *AEL* and *FH* approaches can be compared for similar time periods in taxa for which independent estimates are available for (1) survival from entrainment to the age at maturity, and (2) entrainment back to the number of eggs produced. This comparison serves as a method of cross-validating the two demographic models. Substantial differences between the model estimates may indicate that the population growth rate implied by the model parameters is unrealistically high or low.

FH estimates the number of females at the age of maturity whose reproductive output is lost. The total number of females N_F of all ages in the population can be estimated by the average fecundity as

$$N_F = \frac{E_T}{\bar{F} \cdot \prod_{j=1}^{n} S_j}. \tag{17}$$

AEL can be extrapolated to all mature female ages and summed to make a comparison to 2•N_F using the preceding assumptions. The number of females whose reproductive output is lost in the population, N_F, will be greater than the females estimated by *FH*. The analogue, sum of extrapolated *AEL* over adult ages, will be greater than *AEL* and represents the number of adult males and females lost.

Empirical Transport Model

The *ETM* estimates conditional probability of mortality (P_M) associated with entrainment and requires an estimate of proportional entrainment (*PE*) as an input. Proportional entrainment is an estimate of the daily entrainment mortality on larval populations in the source water, independent of other sources of mortality. Following Ricker (1975), *PE* is an estimate of the conditional mortality rate. Proportional entrainment was calculated using the ratio of intake and source water abundances. In previous entrainment studies using the *ETM* method, intake concentrations were assumed from weighted population concentrations (Boreman et al. 1981). As proposed by the U.S. Fish and Wildlife Service (Boreman et al. 1978, 1981), *ETM* has been used to assess entrainment effects at the Salem Nuclear Generating Station in Delaware Bay, New Jersey and at other power stations along the east coast of the United States (Boreman et al. 1978, 1981; PSE&G 1993). Variations of this model have been discussed in MacCall et al. (1983) and used to assess impacts at the San Onofre Nuclear Generating Station (SONGS; Parker and DeMartini 1989).

The *ETM* estimates conditional mortality due to entrainment, while accounting for spatial and temporal variability in distribution and vulnerability of each life stage to cooling water withdrawals. The original form of the *ETM* incorporated many time-, space-, and age-specific estimates of mortality as well as information regarding spawning periodicity and larval duration (Boreman et al. 1978, 1981). Most of this information is limited or unknown for the taxa that were investigated for our studies. Thus, the applicability of this form of the *ETM* will be limited by the absence of empirically derived or reported demographic parameters needed as input to the model. The approach used in these studies only requires an estimate of the time the larvae are susceptible to entrainment. By compounding the *PE* estimate over time, the *ETM* can be used to estimate entrainment over a time period using assumptions about species-specific larval life histories, specifically the length of time in days that the larvae are in the water column and exposed to entrainment.

On any one sampling day *i*, the conditional entrainment mortality can be expressed as follows:

$$PE_i = \frac{E_i}{N_i}, \tag{18}$$

where

E_i = total numbers of larvae entrained during a day during the i^{th} survey; and

N_i = numbers of larvae at risk of entrainment, i.e., abundance of larvae in the sampled source water during a day during the i^{th} survey.

Survival over one day = 1-PE_i, and survival over the number of days (d) that the larvae are vulnerable to entrainment = $(1\text{-}PE_i)^d$, where d is estimated from the lengths of a representative sample of larvae collected over the entire study period. Values used in calculating PE are population estimates based on respective larval concentrations and volumes of the cooling water system flow and source water areas. The estimate of daily entrainment (E_i) was calculated using the methods described previously. The abundance of larvae at risk in the source water during the i^{th} survey can be directly expressed as follows:

$$N_i = V_S \cdot \overline{\rho}_{N_i}, \tag{19}$$

where

V_S = the static volume of the source water (N); and

$\overline{\rho}_{N_i}$ = the average larval concentration in the source water during the i^{th} survey.

We note that the daily estimate of survival used by MacCall et al. (1983) and Boreman et al. (1981) is $S=e^{-PE}$, which assumes the Baranov catch equation, $E=FN$, where F corresponds to PE and N is the average population size (Ricker 1975). Our estimate of daily survival assumes that N is the population size prior to entrainment. In our studies the outcome is approximately the same regardless of the type of survival estimates because PE values were weighted by large populations. When entrainment becomes relatively large it is recommended to use the Baranov-based estimate as in MacCall et al. (1983) because mortality estimates are reflective of average population size and also are larger.

At SBPP, and for taxa that were determined to primarily inhabit Morro Bay in the MBPP study, the estimated volumes of source water bodies previously described were used to estimate the abundance using an average concentration based on all of the samples from the source water for a given survey on a single day. At DCPP the equation to estimate PE for a day on which entrainment was sampled was:

$$PE = \frac{N_E}{N_G}, \tag{20}$$

where

N_E = estimated number of larvae entrained during the day, calculated as (estimated concentration of larvae in the water entrained that day) x (design specified daily cooling water intake volume); and

N_G = estimate of larvae in nearshore sampling area that day, calculated as $\sum_{i=1}^{64}$[(average concentration per cell) · (cell volume)] for i = 1, ..., 64 grid cells.

where the estimated cell concentrations were obtained from the 72-hour source water survey that contained the 24-hour entrainment sampling period. In addition, an adjustment was made to the estimated number of larvae in the row 1 cells of the study grid to help compensate for the inability to safely collect samples inshore of the grid (Figure 2-3). The estimated volume of the water directly inshore of the study grid was multiplied by the concentration of larvae collected in the row 1 cells, except for cells A1, D1, and E1, as previously described.

Regardless of whether the species has a single spawning period per year or multiple overlapping spawnings the estimate of total larval entrainment mortality can be expressed as the following:

$$P_M = 1 - \sum_{i=1}^{n} f_i (1 - P_S PE_i)^d , \qquad (21)$$

where

PE_i = estimate of proportional entrainment for the ith survey (i = 1,...,n);

P_S = proportion of sampled source water to total estimated source water;

f_i = annual proportion of total larvae hatched during the ith survey; and

d = estimated number of days that the larvae are exposed to entrainment.

To establish independent survey estimates, it was assumed that each new survey represented a new, distinct cohort of larvae that was subject to entrainment. Each of the surveys was weighted using the proportion of the total population at risk during the i^{th} survey (f_i). In the original study plan and analyses for MBPP and DCPP studies we proposed to use the proportion of larvae entrained during each survey period as the weights for the *ETM* model. Weights were proposed to be calculated as follows:

$$f_i = \frac{E_i}{E_T} , \qquad (22)$$

where E_i is estimated entrainment during the i^{th} survey, and E_T is estimated entrainment for the entire study period. This formulation conflicts with the formula for PE that uses the population in the source water during each survey to define the population at risk. If the weights are meant to represent the proportion of the population at risk during each survey then the weights should be calculated as follows:

$$f_i = \frac{N_i}{N_T}, \tag{23}$$

where N_i is the source population spawned during the i^{th} survey, and N_T is the sum of the N_is for the entire study period. Weights calculated using the entrainment estimates redefined the population at risk as the population entrained and represented a logical inconsistency in the model. Weights calculated using the source water estimates were used at SBPP and were used in final analyses of the data from the MBPP and DCPP studies in this paper.

The number of days that the larvae of a specific taxon were exposed to the mortality estimated by *PE*, was estimated using length data from a representative number of larvae from the entrainment samples. At SBPP, a single estimate of larval exposure was used in the calculations. The number of days (*d*) from hatching to entrainment was estimated by calculating the difference between the values of the 1st and upper 99th percentiles of the length measurements for each entrained larval taxon and dividing this range by an estimate of the larval growth rate for that taxon that was obtained from the scientific literature. The 1st and upper 99th percentiles were used to eliminate potential outlier measurements in the length data. In earlier studies at MBPP and DCPP, two estimates of *d* were calculated for each taxon and these were used to calculate two *ETM* estimates. The first estimate used an estimate of *d* calculated using the difference in length between the 1st and upper 99th percentiles and was used to represent the maximum number of days that the larvae were exposed to entrainment. The second estimate used an estimate of *d* calculated using the difference in length between the 1st percentile and the average length and was used to represent the average number of days that the larvae were exposed to entrainment.

The estimate of P_S in the *ETM* model is defined by the ratio of the area or volume of sampled source water to a larger area or volume containing the population of inference (Parker and DeMartini 1989). If an estimate of the larval (or adult) population in the larger area is available, the value of P_S can be computed directly using the estimate of the larval or adult population in the sampling area, defined by Ricker (1975) as the proportion of the parental stock. If the distribution in the larger area is assumed to be uniform, then the value of P_S for the proportion of the population will be the same as the proportion computed using area or volume.

For the SBPP study the entire source water was sampled (P_S = 1.0) and P_S was not incorporated in the *ETM*. At the MBPP, P_S was not incorporated in the *ETM* for fishes that were primarily associated with the estuarine habitats in Morro Bay. The P_S was included for fish and crab taxa whose adult distributions extended out into the nearshore waters. Estimates of the population of inference for these taxa were unavailable, therefore, P_S was estimated using the distance the larvae could have traveled based on the duration of exposure to entrainment and current speed as follows:

$$P_S = \frac{L_G}{L_P}, \qquad (24)$$

where

L_G = length of sampling area; and
L_P = length of alongshore current displacement based on the period (d) of larval vulnerability for a taxon.

The length of alongshore displacement was calculated using average current speed for the period of January 1, 1996 – May 31, 1999 from an InterOceans S4 current meter deployed at a depth of -6 m MLLW in approximately 30 m of water about 1 km west of the DCPP Intake Cove, south of Morro Bay. The current direction was ignored in the calculations, but was predominantly alongshore. The current speed was used to estimate unidirectional displacement over the period of time that the larvae were exposed to entrainment. The value of alongshore displacement (L_P) was compared with the alongshore length of the sampled waterbody (L_G). The distance between the west Morro Bay breakwater and Station 5 is 4.8 km; a value of 9.6 km (twice the distance) was used for L_G. This value was used because it places Station 5 in the center of the sampled waterbody.

For the MBPP study we only presented a single estimate of P_M for the taxa that used an adjustment for P_S in the *ETM*, because any changes due to the increased duration were inversely proportional to the changes in P_S, and resulted in nearly equal estimates of P_M. (The exponential model [MacCall et al. 1983], $1-e^{P_s PEt}$, gives equal estimates for P_S inversely proportional to t). The estimate of the standard error is increased due to the extended period of entrainment risk, so two estimates of the standard error were presented for these taxa.

The sampling for the DCPP study was also extrapolated to provide an estimate of entrainment effects outside the nearshore sampling area. Boreman et al. (1981) point out that if any members of the population are located outside the sampled area, then the *ETM* will overestimate the conditional entrainment mortality for the entire population. In their study of entrainment at SONGS, Parker and DeMartini (1989) incorporated the inference population (which was an extrapolation to the entire Southern California Bight from the coast to a depth of 75 m, an area extending about 500 km) directly into their estimate of *PE*. In the DCPP *ETM* analyses, *PE* was multiplied by the estimated fraction of the population in the nearshore sampling area (P_s). The size of the population affected by entrainment varied from relatively small (e.g., the size of the sampling area) to very large (e.g., fishery management units, zoogeographic range). For some species an area approximately the size of the study grid represented the population of inference, and in these cases, $P_S \approx 1$. For other species, the population of inference was larger than the study grid. The population of inference depended not only on the species, but also what appealed usefully to intuition, as a number of methods could be used for extrapolation. Therefore, the *ETM* was calculated over a range of values of P_S for each of the taxa selected for detailed assessment. The resulting curves were used to determine the *ETM* at any value of P_S. The curves were interpreted as a continuous probability function representing the risk of entrainment to the larvae at different values of P_S. Point estimates of P_M (and their ranges) were also calculated for each taxon.

The relationship between P_M and P_S was represented by the sets of curves for each of the taxa analyzed for DCPP. Two point estimates of P_S were also computed to account for the variation in the distribution of adult fishes included in the assessment. For offshore and subtidal taxa whose larval distribution extends to the offshore edge of the study grid, P_S was calculated as follows:

$$P_S = \frac{N_G}{N_P}, \tag{25}$$

where N_G is the number of larvae in the study grid, and N_P is the number of larvae in the population of inference. The numerator N_G, presented earlier in the calculation of *PE*, was calculated as follows:

$$N_{G_i} = \sum_{k=1}^{64} A_{G_k} \cdot \bar{D}_k \cdot \rho_{i,k} \;, \tag{26}$$

where

A_{G_k} = area of grid cell k;

$\bar{D}_k$ = average depth of the *k*th grid cell; and

ρ_{ik} = concentration (per m^3) of larvae in *k*th grid cell during survey *i*.

N_P was estimated by an offshore and alongshore extrapolation of the study grid concentrations, using water current measurements. The following conceptual model was formulated to extrapolate larval concentrations (per m^3) offshore of the grid:

$$P_S = \frac{N_G}{N_P} = \frac{\sum_{i=1}^{K_G} L_{G_j} \cdot W_j \cdot \bar{D}_j \cdot \rho_j}{\sum_{i=1}^{K_P} L_{P_j} \cdot W_j \cdot \bar{D}_j \cdot \rho_j} \;, \tag{27}$$

where

L_{G_j} = alongshore length of grid in the *j*th stratum;

W_j = width of *j*th stratum;

L_{P_j} = alongshore length of population in *j*th stratum based on current data;

$\bar{D}_j$ = average depth of *j*th stratum; and

ρ_j = average density of larvae in *j*th stratum.

For this model, the grid was subdivided into K_G alongshore strata (i.e., K_G=8 rows in the grid) and the population into K_G>K_G alongshore strata. This approach described discrete values in intervals of a continuous function. Therefore, to ease implementation, an essentially equivalent formula used grid cell concentrations during the i^{th} sampling period, $\rho_{i,k}$ for a linear extrapolation of density (# per m^2 calculated by multiplying $\rho_{i,k}$ by the cell depth) as a function of offshore distance, *w*:

$$P_{S_i} = \frac{N_{G_i}}{N_{P_i}} = \frac{N_{G_i}}{N_{G_i}\left(\frac{L_{P_i}}{L_G}\right) + L_{P_i}\int_{W_0}^{W_{Max}} \rho(w)dw} \;, \tag{28}$$

where L_P = alongshore length of population in the i^{th} study period based on current displacement. The limits of integration are from the offshore margin of the study grid, W_O, to a point estimated by the onshore movement of currents or where the density is zero or biologically limited, W_{max}. Note that this point will usually occur outside the study grid area and that the population number, N_P, is composed of two components that represent the alongshore extrapolation of the grid population and the offshore extrapolation of the alongshore grid population (Figure 2-3).

Alongshore and onshore current velocities used in the calculations were measured at a current meter positioned approximately 1 km west of the DCPP intake at a depth of approximately 6 m (Figure 2-3). The direction in degrees true from north and speed in cm/s were estimated for each hour of the nearshore study grid survey periods. Figure 2-4 shows the results of current meter analysis in which hourly current vectors were first rotated orthogonal to the coast by 49 degrees west of north. The movement of water was then tracked during the period from April 1997 through June 1999. A total alongshore length can be calculated from these data using the maximum up-coast and down-coast current movement over the larval duration period prior to each survey period. The maximum upcoast and downcoast current vectors measured during each survey period were added together to obtain an estimate of total alongshore displacement. This contrasts with the approach for the MBPP where average current speed was used in calculating alongshore movement. Transport of larvae into the nearshore via onshore currents was also accounted for and used to set the limits of the offshore density extrapolation. Within this scenario, there were two subclasses:

1. For species in which the regression of density versus offshore distance had a negative slope, the offshore distance predicted where density was zero (i.e., integral of zero) was calculated. The alongshore distance was calculated from the water current data.
2. For species in which the regression of density versus offshore distance had a slope of $\geq$0, either the offshore distance from the water current data or an average distance based on the depth distribution of the adults offshore was used. Literature values (e.g., CalCOFI) were used to place a limit on both the distance and density values used in the offshore extrapolation.

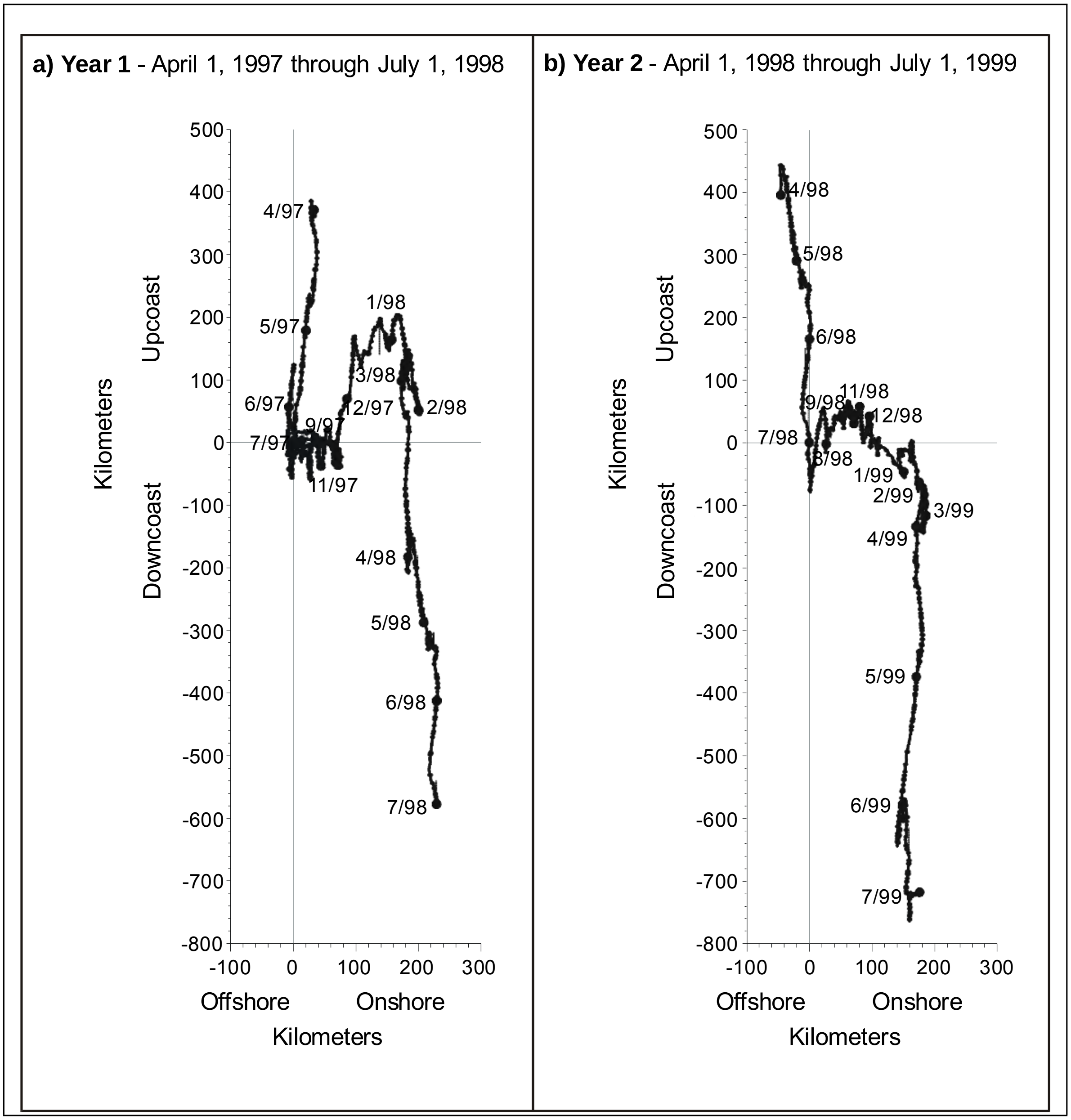


Figure 2-4. Relative cumulative upcoast/downcast and onshore/offshore current vectors from current meter located approximately 1 km west of the Diablo Canyon Power Plant intake at a depth of 6 m. Dates on current vectors are the dates of each survey.

Parameter values needed in performing the extrapolation were obtained by using analysis of covariance based on all of the data from the surveys for the study period from July 1997 through June 1999. The following quadratic model was tested in the analysis:

$$\rho_{ij} = \alpha_i + \beta w_{ij} + \gamma w^2_{if} + \varepsilon_{ij} , \qquad (29)$$

where

ε_i = normally distributed error term with mean of zero;

w_{ij} = distance for the ith observation in the jth survey;

ρ_{ij} = larval density per m^2 for the ith observation in the jth survey; and

α, β, γ = regression coefficients.

The following linear model produced a better fit in all cases:

$$\rho_{ij} = \alpha_i + \beta w_{ij} + \varepsilon_{ij}. \qquad (30)$$

A common slope, β, for all surveys and unique intercepts, α_i, for each survey were derived from the model. It is reasonable to assume a common slope, but differences in abundance between surveys required fitting different intercepts.

Similar to the demographic models there are also assumptions associated with the *ETM* approach. Although there are fewer life history parameters necessary for the *ETM*, it shares with the demographic models the assumption that the life history data used to calculate the period of time the larvae are exposed to entrainment are representative of the population in the years and location for the specific study and accurately estimates the period of larval exposure. Since the *ETM* is only estimating the entrainment mortality on the population of larvae, assumptions regarding compensation would only be important in interpreting the effects on adult populations. An assumption inherent to all the models is that the sampling resulted in representative estimates of entrainment for the period surveyed. Additional assumptions of the *ETM* include the following:

- The sampling resulted in representative estimates of the source water populations of larvae susceptible to entrainment and that the PE estimated from the entrainment and source water population samples is representative of entrainment mortality during the survey period.
- The estimates of the source water population represent the proportion for the survey period (fi) of total larval production.
- The samples during each survey period represent a new and independent cohort of larvae.

Although it would seem that there are also assumptions associated with the definition of the source water population relative to the population of inference, these assumptions become less critical if the *ETM* results are converted, for example, to Area of Production Foregone (APF). The APF is a useful method for converting the results of *ETM* into a context for resource managers and is presented in Section 4.0.

Variance calculations for *PE* are presented in Appendix A. Variance calculations for the estimate of P_M are not presented because of the different approaches and parameters that will be used in the *ETM* calculations for each study.

3.0 RESULTS

Detailed results for an example taxon from each plant are presented to compare the modeling approaches for different source water body types. Results at SBPP are presented for the arrow, cheekspot, and shadow (*Clevlandia ios, Ilypnus gilberti,* and *Quietula y-cauda* [CIQ]) goby complex, which was the most abundant fish larvae collected during the study. At Morro Bay and Diablo Canyon, the kelp, gopher, and black-and-yellow (*S. atrovirens, S. carnatus,* and *S. chrysomelas* [KGB]) rockfish complex results provided illustrative data. These results provide example calculations for the *FH* and *AEL* models as well as for the *ETM* so that all three modeling approaches can be compared between sites.

The example taxa are indicative of the source water at the three study sites. Since SBPP used a fixed source water body volume the *ETM* model for all of the taxa analyzed, including CIQ gobies, was calculated similarly. At MBPP, the *ETM* model for the taxa that were designated as primarily inhabitants of Morro Bay was calculated using a fixed source water volume using calculations identical to those for CIQ gobies for the SBPP study. Therefore, we decided to present the *ETM* results for the KGB rockfish at MBPP since the source water for this taxon included both the bay and a nearshore area, the size of which was estimated using current meter data. A similar approach was taken for the DCPP study and, therefore, the results for the KGB rockfish complex are also presented for that study to provide a comparison with the results for MBPP.

3.1 SOUTH BAY POWER PLANT

A total of 23,039 larval fishes in 20 taxonomic categories ranging from ordinal to specific classifications was collected from 144 samples at the SBPP entrainment station (SB1) during monthly sampling from February 2001 through January 2002 (Table 3-1). These samples were used to estimate that total annual entrainment of fish larvae was 2.42×10^9. Entrainment samples were dominated by gobies in the CIQ complex, which comprised about 76% of the total estimated entrainment. Five taxa evaluated for entrainment effects (Table 2-4) comprised greater than 99% of the total number of fish larvae entrained. No invertebrates were evaluated because only a single *Cancer* crab megalopae was collected.

The entrainment and source water stations extend over a distance of greater than 9 km in south San Diego Bay and include both channel and shallow mudflat habitats. Despite the differences in location and habitat, CIQ complex gobies were the most abundant fish larvae at all of the stations (Appendix B). Other fishes showed considerable variation in abundance among stations. For example, combtooth blennies (*Hypsoblennius* spp.) were much more abundant along the eastern shore north of SBPP where there are more piers and other structures, whereas longjaw mudsuckers (*Gillichthys mirabilis*) were in highest abundance near the power plant. Overall, taxa richness generally increased from the entrainment station in the far south end of the bay to Station SB9 in the north.

Table 3-1. Total annual entrainment estimates of larval fishes at South Bay Power Plant based on monthly larval densities (sampled at Station SB1 from February 2001 through January 2002) and the plant's designed maximum circulating water flows; *n*=144 tows at one station. Data and estimates for taxa comprising <0.01 percent of the composition not presented individually but lumped under other taxa.

Taxa	Common Name	Total Larvae Collected	Est. Total Annual Entrainment	Entrain. Percent Comp.	Entrain. Cum. Percent
CIQ goby complex	gobies	17,878	1,830,899,000	75.64	75.64
Anchoa spp.	bay anchovies	4,390	514,809,000	21.27	96.91
Hypsoblennius spp.	combtooth blennies	226	22,335,000	0.92	97.83
Gillichthys mirabilis	longjaw mudsucker	249	21,953,000	0.91	98.74
Atherinopsidae	silversides	140	14,521,000	0.60	99.34
Syngnathus spp.	pipefishes	101	10,013,000	0.41	99.75
Acanthogobius flavimanus	yellowfin goby	19	2,261,000	0.09	99.85
Strongylura exilis	Calif. needlefish	8	740,000	0.03	99.88
Sciaenidae	croakers	6	706,000	0.03	99.91
	Other 11 taxa	22	2,291,000	0.09	100.00
	Total	23,039	2,420,528,000		

SBPP Results for CIQ Gobies

The following sections present results for demographic and empirical transport modeling of SBPP entrainment effects. All three modeling approaches are presented for the CIQ goby complex. CIQ goby larvae were most abundant at the entrainment station during June and July (Figure 3-1). Brothers (1975) indicated that the peak spawning period for arrow goby occurred from November through April, while spawning in cheekspot and shadow goby was 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 this study.

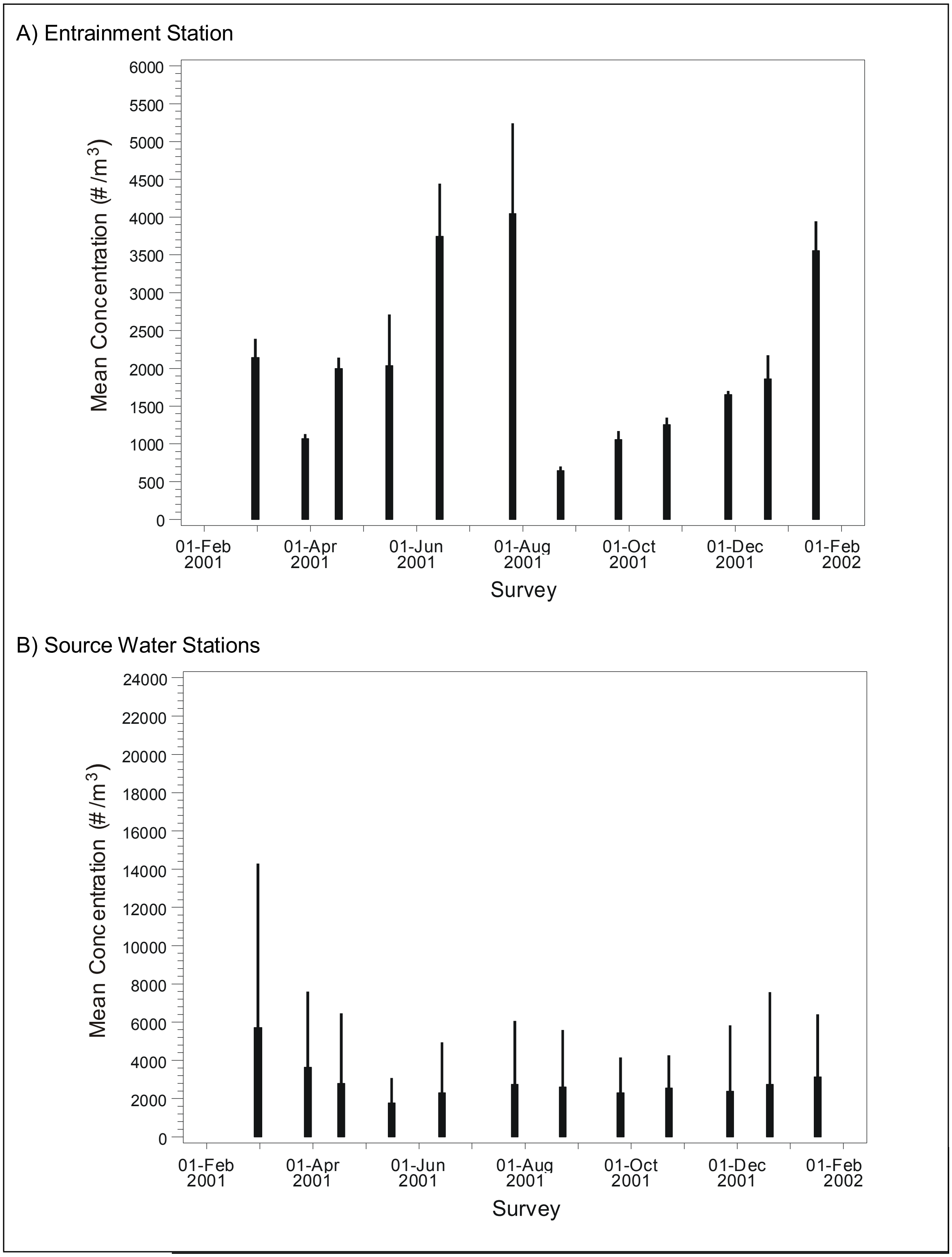


Figure 3-1. Monthly mean larval concentration (standard error shown at top of dark bars) of the *Clevlandia ios, Ilypnus gilberti*, and *Quietula y-cauda* (CIQ) goby complex larvae at SBPP; A) intake entrainment station and B) source water stations.

The *ETM* required an estimate of the length of time the larvae are susceptible to entrainment. The length frequency distribution for a representative sample of CIQ goby larvae showed that the majority of larvae were recently hatched based on the reported hatch size of 2–3 mm (Moser 1996) (Figure 3-2). The mean length of the collected CIQ goby larvae was 3.1 mm and the difference between the lengths of the 1st (2.2 mm) and 99th (5.8 mm) percentile values were used with a growth rate of 0.16 mm^{-d} estimated from Brothers (1975) to determine that CIQ goby larvae were vulnerable to entrainment for a period of 22.9 days. The growth rate of 0.16 mm^{-d} was determined using Brothers (1975) reported transformation lengths for the three species and an estimated transformation age of 60 d.

The comprehensive comparative study of the three goby species in the CIQ complex by Brothers (1975) also provided the necessary life history information for both *FH* and *AEL* demographic models and shows how life history data from the scientific literature are used in the modeling.

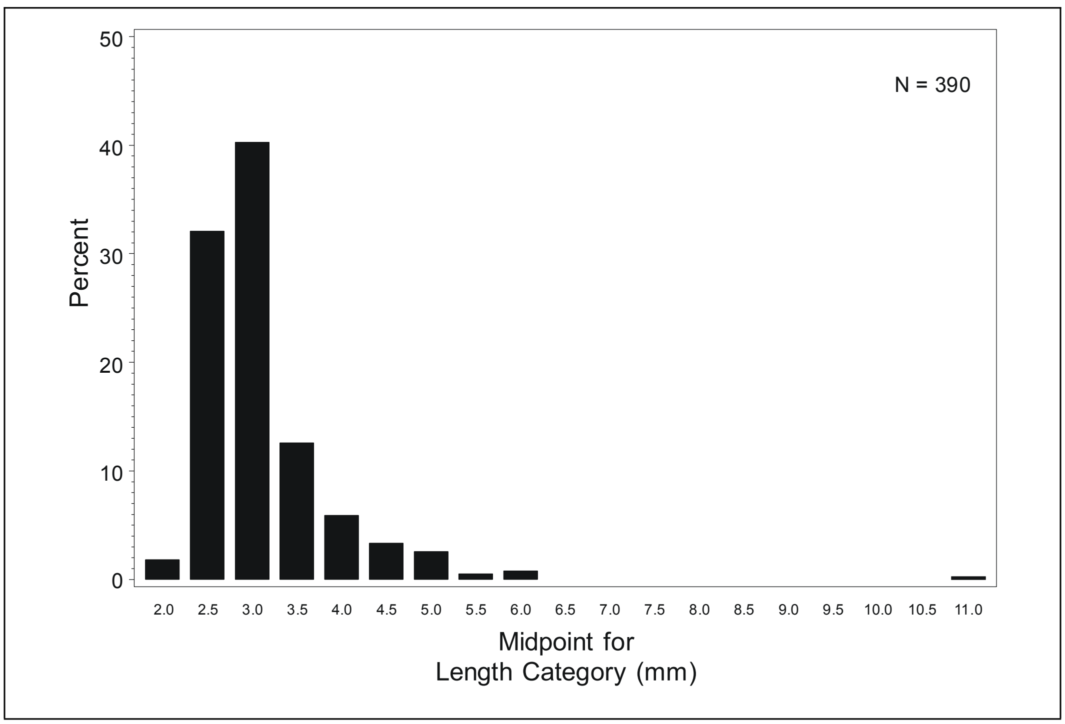


Figure 3-2. Length frequency distribution for *Clevlandia ios, Ilypnus gilberti*, and *Quietula y-cauda* (CIQ) goby complex larvae from the South Bay Power Plant entrainment station.

Fecundity Hindcasting

The annual entrainment estimate for CIQ gobies was used to estimate the number of adult females at the age of maturity whose reproductive output was lost due to entrainment (Table 3-2). No estimates of egg survival for gobies were available, but because goby egg masses are demersal (Wang 1986) and parental care, usually provided by the adult male, is common in the family (Moser 1996), egg survival is probably high and was assumed to be 100 percent. Average larval mortality of 99% over the two months between hatching and transformation for the three species of CIQ gobies from Brothers (1975) was used to estimate a daily survival rate of 0.931 as follows: $0.931 = (1\text{-}0.99)^{(6/365.25)}$. Mean length and length of the first percentile (2.2 mm) were used with the growth rate of 0.16 mm^{-d} to estimate a mean age at entrainment of 5.8 d. Survival to average age at entrainment was then estimated as $0.931^{5.8} = 0.659$. 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 at two to three different stages of development in the ovaries; therefore, an estimate of 2.5 spawns per year was used in calculating *FH* (615 eggs/spawn x 2.5 spawns/year = 1,538 eggs/year). The TLF for the studies at SBPP was calculated by adding 1 to the difference between the average ages of maturity (1.0) and longevity (3.3) from Brothers (1975) to account for spawning of a portion of the population during the first year. The *FH* model was used to estimate that the number of adult females at the age of maturity whose lifetime reproductive output was entrained through the SBPP circulating water system was 1,085,000 (Table 3-2). The standard error for the entrainment estimate was used to estimate a confidence interval based on just the sampling variance that was considerably less than a confidence interval for the estimate calculated using an assumed CV of 30% for all of the life history parameters.

Table 3-2. Results of fecundity hindcasting (*FH*) modeling for CIQ goby complex larvae entrained at South Bay Power Plant. The upper and lower estimates are based on a 90% confidence interval of the mean. *FH* was recalculated using the upper and lower confidence interval estimates for total entrainment.

	Estimate	**Estimate Std. Error**	***FH* Lower Estimate**	***FH* Upper Estimate**	***FH* Range**
FH Estimate	1,085,000	1,880,000	63,000	18,782,000	18,719,000
Total Entrainment	1.83×10^9	21,725,000	961,000	1,209,000	248,000

Adult Equivalent Loss

Three survival components were used to estimate *AEL*. These were 1) larval survival from the age of entrainment to the age of settlement, 2) survival from settlement to age 1, and 3) from age 1 to the average female age. Larval survival from average age at entrainment through settlement at 60 days was estimated as $0.931^{60-5.8} = 0.021$ using the same daily survival rate used in formulating *FH*. Brothers (1975) estimated that mortality in the first year following settlement was 91% for arrow, 66–74% for cheekspot, and 62–69% for shadow goby. These estimates were used to calculate a daily survival rate of 0.995 as follows:

$$0.995 = \frac{(1-0.91)^{1/(365.25-60)} + (1-0.70)^{1/(365.25-60)} + (1-0.65)^{1/(365.25-60)}}{3}$$

This value was used to calculate a finite survival of 0.211 for the first year following settlement as follows: $0.211 = 0.995^{(365.25-60)}$. Adult daily survival from one year through the average female age of 1.71 years from life table data for the three species provided by Brothers (1975) was estimated as 0.99. This value was used to calculate a finite survival of 0.195 as follows: $0.195 = (0.99)^{((1.71*365.25)-365.25)}$. The product of the three survival estimates and the entrainment estimate were used to estimate that the number of larvae entrained through the SBPP circulating water system number were equivalent to the loss of 1,580,000 adult CIQ gobies (Table 3-3). The standard error for the entrainment estimate was used to estimate a confidence interval based on just the sampling variance that was considerably less than a confidence interval for the estimate calculated using an assumed CV of 30% for all of the life history parameters.

Table 3-3. Results of adult equivalent loss (*AEL*) modeling for CIQ goby complex larvae entrained at South Bay Power Plant. The upper and lower estimates are based on a 90% confidence interval of the mean. *AEL* was recalculated using the upper and lower confidence interval estimates for total entrainment.

	Estimate	**Estimate Std. Error**	***AEL* Lower Estimate**	***AEL* Upper Estimate**	***AEL* Range**
AEL Estimate	1,580,000	2,739,000	91,300	2.74×10^7	2.73×10^7
Total Entrainment	1.83×10^9	2.17×10^7	1,399,000	1,760,000	361,000

Empirical Transport Model

The *ETM* estimates for CIQ gobies were calculated using the data in Appendix C and a larval duration of 22.9 days. Average larval concentrations from the entrainment and source water sampling were multiplied by the cooling water and source water volumes, respectively, to obtain the estimates that were used in calculating *PE* estimate for each survey. Weights were calculated by multiplying the source water estimate for each survey by the number of days in the survey period. Estimates for the surveys were summed and the proportion (f_i) for each survey calculated.

Daily mortality (PE_i) estimates ranged from 0.004 to 0.025 for the twelve surveys with an average value of 0.012 (Table 3-4). This average *PE* was similar to the volumetric ratio of the cooling water system to source water volumes (0.015), which was bounded by the range of PE_i estimates. PE_i estimates equal to the volumetric ratio would indicate that the CIQ goby larva were uniformly distributed throughout the source water and were withdrawn by the power plant at a rate approximately equal to that ratio. The small range in both the PE_i estimates and the values of f_i indicate that goby larvae were present in the source water throughout the year. The largest fractions of the source water population occurred in the February (f_i = 0.2165) and July (f_i = 0.1064) surveys which was consistent with the spawning periods for arrow and shadow gobies, respectively. June and July surveys also had the highest entrainment station concentrations resulting in higher PE_i estimates for those surveys (Figure 3-1).

Results for Other Taxa

The modeling results for other taxa selected for detailed assessment showed that both demographic modeling approaches could only be calculated for the CIQ goby complex (Table 3-5) due mainly to a lack of larval survival estimates for the life stages between larvae and adult. The alignment of the 2**FH* and *AEL* estimates would have been improved by extrapolating *AEL* to the age of maturity rather that the average female age of 1.7 years. Differences in the *FH* model results among taxa were generally proportional to entrainment estimates as shown by decreasing 2**FH* estimates for the top four taxa. As the results for the *ETM* model show, proportional effects of entrainment on the source populations vary considerably for the five taxa and do not reflect differences in entrainment estimates, but the combination of larval concentrations at entrainment and source water stations. The *ETM* estimates of P_M ranged from 0.031 (3.1%) to 0.215 (21.5%) with the estimated effects being lowest for combtooth blennies and highest for CIQ gobies and longjaw mudsuckers.

Table 3-4. Estimates of proportional entrainment (*PE*) and proportion of source water population present for CIQ goby larvae at South Bay Power Plant entrainment and source water stations from monthly surveys conducted from February 2001 through January 2002.

Survey Date	*PE* Estimate	Proportion of Source Population for Period (*f*)
28-Feb-01	0.0057	0.2165
29-Mar-01	0.0045	0.0977
17-Apr-01	0.0109	0.0491
16-May-01	0.0175	0.0475
14-Jun-01	0.0247	0.0620
26-Jul-01	0.0225	0.1064
23-Aug-01	0.0038	0.0675
25-Sep-01	0.0070	0.0704
23-Oct-01	0.0075	0.0661
27-Nov-01	0.0105	0.0773
20-Dec-01	0.0103	0.0584
17-Jan-02	0.0173	0.0811
Average =	0.0118	

Table 3-5. Summary of estimated South Bay Power Plant entrainment effects based on fecundity hindcasting (*FH*), adult equivalent loss (*AEL*), and empirical transport (*ETM*) estimates of proportional mortality (P_m) models. The *FH* estimate is multiplied by 2 to test the relationship that $2 \cdot FH \equiv AEL$.

Taxa	Entrainment Estimate	% Source Numbers	*2*FH*	*AEL*	P_M
CIQ goby complex	1.83×10^9	76.75	2,170,000	1,580,000	0.215
anchovies	5.15×10^8	15.12	214,000	*	0.105
combtooth blennies	2.23×10^7	5.93	21,500	*	0.031
longjaw mudsucker	2.19×10^7	0.17	2,960	*	0.171
silversides	1.45×10^7	0.65	*	*	0.146

* Information unavailable to compute model estimate.

3.2 MORRO BAY POWER PLANT

A total of 30,270 larval fishes in 87 taxonomic categories ranging from ordinal to specific classifications was collected from 609 samples at the MBPP entrainment station during weekly sampling from January 2000 through December 2000 (Table 3-6). These data were used to estimate total annual entrainment of fish larvae at 5.08×10^8. Entrainment samples were dominated by unidentified gobies, which comprised 77% of the total estimated entrainment of fish larvae. The top seven taxa comprising greater than 90% of the total and three other commercially or recreationally important fishes in the top 95% (white croaker *Genyonemus lineatus*, Pacific herring *Clupea pallasii*, and cabezon *Scorpaenichthys marmoratus*) were evaluated for entrainment effects along with six species of *Cancer* crab megalopae (Table 2-4) (results for *Cancer* crab not presented).

Table 3-6. Total annual entrainment estimates of fishes and invertebrates at Morro Bay Power Plant based on weekly larval densities sampled at Station 2 (n=609 tows) from January to - December 2000 and the plant's maximum circulating water flows. Data and estimates for taxa comprising <0.01 percent of the composition are not presented individually but lumped as other taxa.

Taxon	Common Name	Total Collected	Estimated Annual # of Entrained Larvae	Percent of Total	Cumulative Percent
Gobiidae unid.	gobies	22,964	393,261,000	77.37	77.37
Leptocottus armatus	Pacific staghorn sculpin	1,129	17,321,000	3.41	80.78
Stenobrachius leucopsarus	northern lampfish	1,018	14,549,000	2.86	83.64
Quietula y-cauda	shadow goby	845	13,504,000	2.66	86.30
Hypsoblennius spp.	combtooth blennies	572	10,042,000	1.98	88.27
Sebastes spp. V_De	KGB rockfishes	360	6,407,000	1.26	89.53
Atherinopsis californiensis	jacksmelt	384	6,266,000	1.23	90.76
Rhinogobiops nicholsi	blackeye goby	226	3,778,000	0.74	91.51
Gillichthys mirabilis	longjaw mudsucker	186	3,286,000	0.65	92.15
Lepidogobius lepidus	bay goby	181	3,233,000	0.64	92.79
Clupea pallasii	Pacific herring	242	3,030,000	0.60	93.39
Scorpaenichthys marmoratus	cabezon	171	2,888,000	0.57	94.54
Atherinopsidae unid.	silversides	163	2,720,000	0.54	95.08
Atherinops affinis	topsmelt	153	2,575,000	0.51	95.58
Sebastes spp. V	rockfishes	150	2,453,000	0.48	96.07
Tarletonbeania crenularis	blue lanternfish	142	2,213,000	0.44	96.50
Engraulis mordax	northern anchovy	155	2,136,000	0.42	96.92
larval fish - damaged	larval fish - damaged	74	1,283,000	0.25	97.18
Gibbonsia spp.	clinid kelpfish	98	1,141,000	0.22	97.40
Bathymasteridae unid.	ronquils	67	1,119,000	0.22	97.62
Cottidae unid.	sculpins	59	1,009,000	0.20	97.82
Artedius lateralis	smoothhead sculpin	46	739,000	0.15	97.96
Oligocottus spp.	sculpin	40	620,000	0.12	98.09
Stichaeidae unid.	pricklebacks	41	616,000	0.12	98.21
Chaenopsidae unid.	tube blennies	31	551,000	0.11	98.32
Cebidichthys violaceus	monkeyface eel	28	505,000	0.10	98.41
Bathylagus ochotensis	popeye blacksmelt	28	495,000	0.10	98.51
	59 other taxa	483	7,564,000	2.93	100.00
	Total Larvae	30,270	508,296,000		

Species composition for entrainment at MBPP was much more diverse than the results from SBPP. This may have resulted from the more frequent weekly sampling at MBPP and the location of the power plant near the entrance to the bay relative to the back bay location of SBPP. Entrainment was dominated by fishes that primarily occur as adults in the bay, such as gobies, but also included numerous fishes that are more typically associated with nearshore coastal habitats, such as rockfish and cabezon.

MBPP Results for the KGB Rockfish Complex

Detailed results and details on the data used in the three modeling approaches at MBPP are presented for the KGB larval rockfish complex. KGB rockfish had the sixth highest estimated entrainment (6,407,000) or 1.3% of the total larval fishes (Table 3-6). Consistent with the annual spawning period for most rockfishes (Parrish et al. 1989), larvae occurred in entrainment samples from January through June with the highest abundances in April (Figure 3-3). Results from source water surveys showed the same abundance peaks seen in samples collected at the MBPP intake station (Figure 3-4). Although not collected every month, KGB rockfish larvae were collected from all of the stations inside Morro Bay during the April survey. They reached their greatest concentration at the Estero Bay Station 5 during the May survey when they were less common at the stations inside Morro Bay.

The length frequency distribution for a representative sample of KGB rockfish larvae showed a relatively narrow size range of 3.4 to 5.4 mm (1^{st} and 99^{th} percentile values = 3.5 and 5.1) with an average size of 4.3 mm (Figure 3-5). These results indicate that most of the larvae are less than the maximum reported size at extrusion of 4.0–5.5 mm (Moser 1996) and are therefore subject to entrainment for a relatively short period of time. There are no studies on the larval growth rates for the species in the KGB rockfish complex so a larval growth rate of 0.14 mm^{-d} from brown rockfish (Love and Johnson 1999, Yoklavich et al. 1996) was used in estimating that the average age at entrainment was 5.5 d and the maximum age at entrainment, based on the 99^{th} percentile values was 11.3 d.

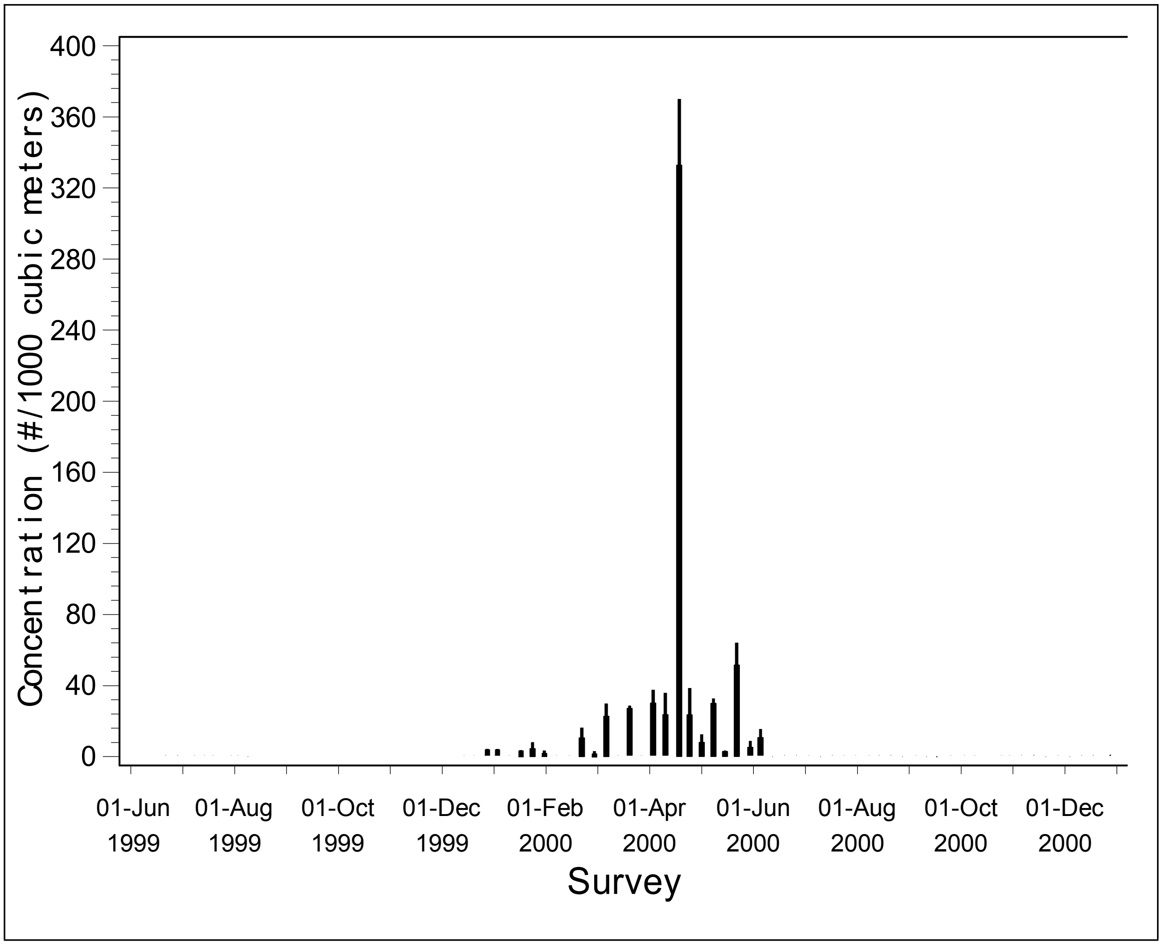


Figure 3-3. Weekly mean larval concentration of kelp, gopher, and black-and-yellow (KGB) rockfish complex larvae at the Morro Bay Power Plant intake entrainment station.

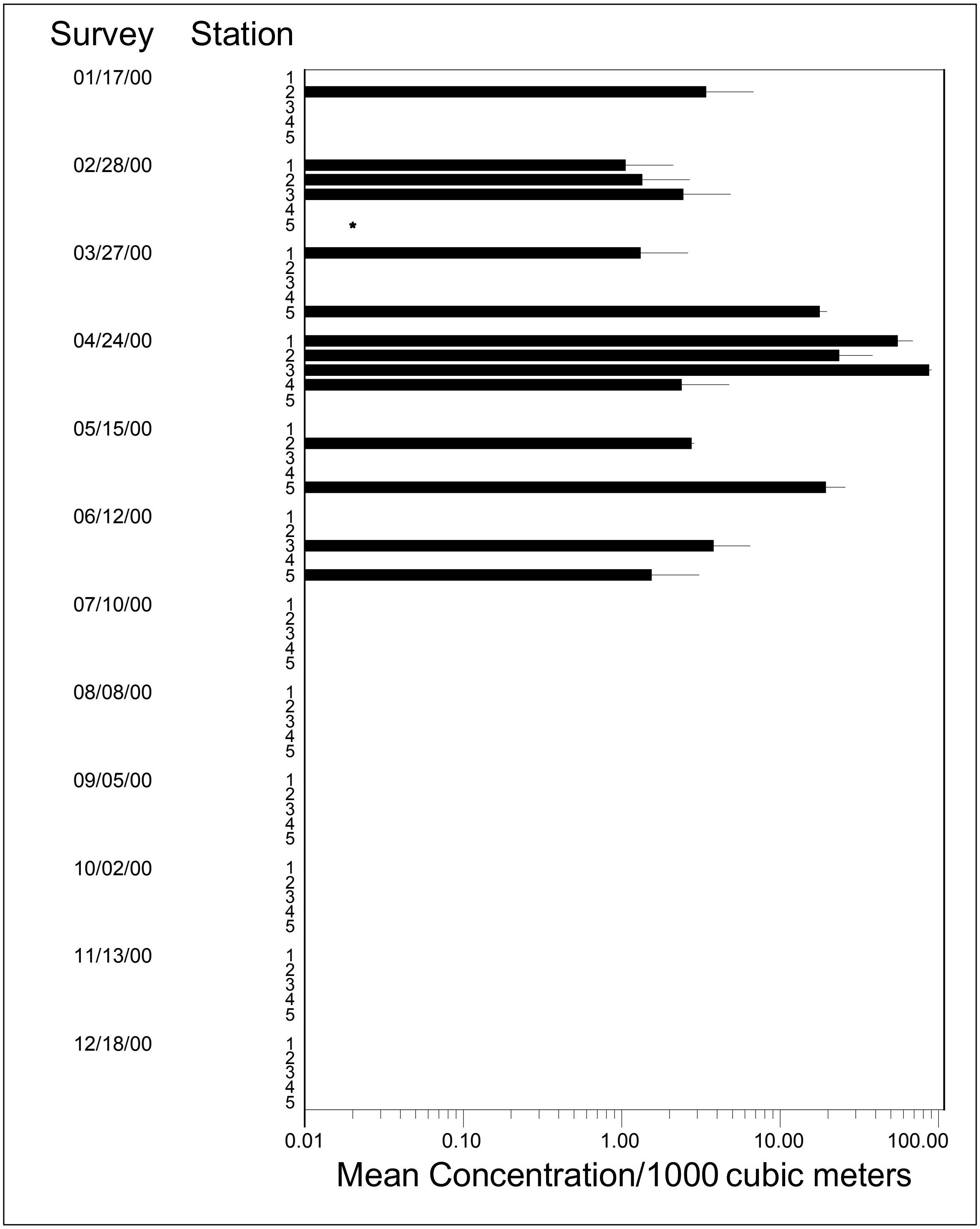


Figure 3-4. Comparison of average concentrations of kelp, gopher, and black-and-yellow (KGB) rockfish complex larvae at the Morro Bay Power Plant intake (Station 2), Morro Bay source water (Stations 1, 3, and 4), and Estero Bay (Station 5) from January 2000 through December 2000 with standard error indicated (+1 SE). Entrainment data only plotted for paired surveys. *No samples were collected during February 2000 at Station 5. Note that data are plotted on a $\log_{10}$ scale.

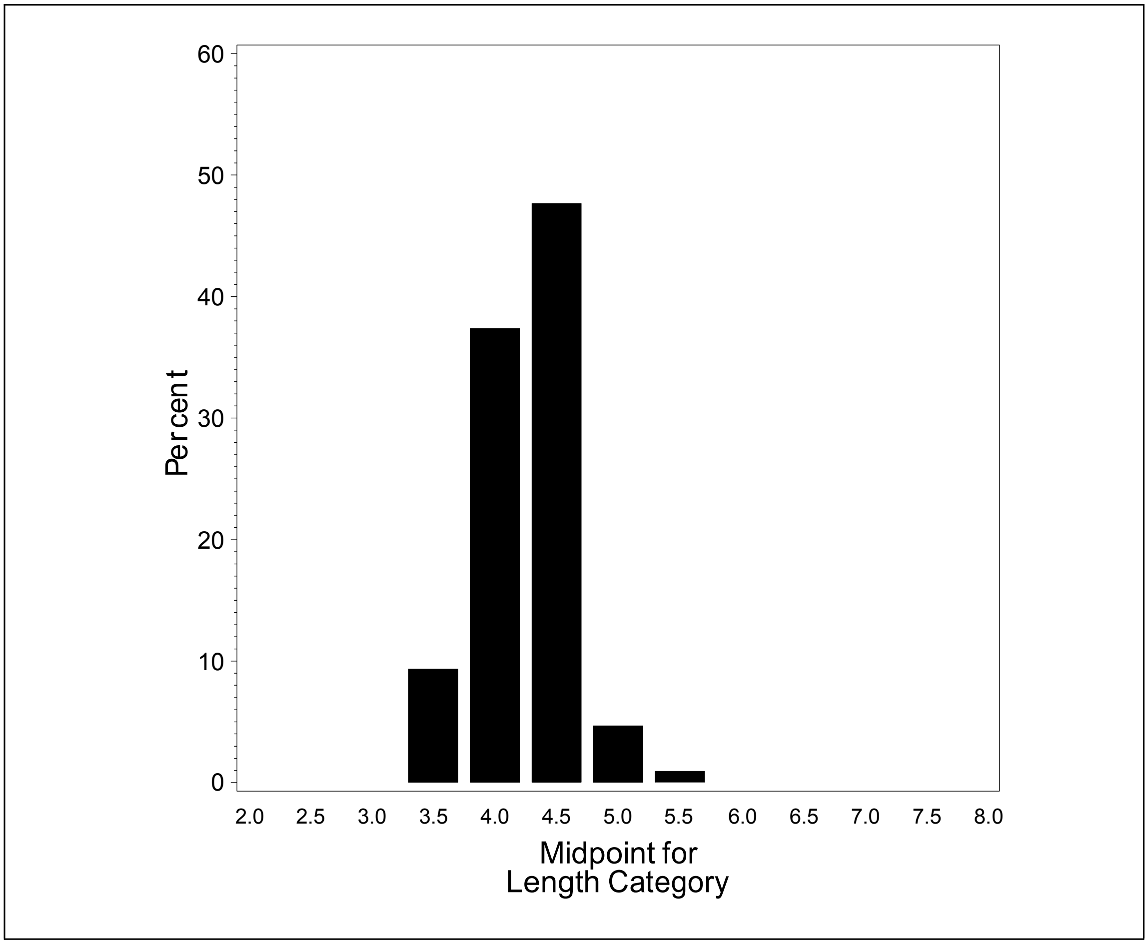


Figure 3-5. Length frequency distribution for kelp, gopher, and black-and-yellow (KGB) rockfish complex larvae from the Morro Bay Power Plant entrainment station.

Fecundity Hindcast Model

Total annual larval entrainment for KGB rockfish was used to estimate the number of adult females at the age of maturity whose reproductive output was lost due to entrainment (Table 3-7). The parameters required for formulation of *FH* estimates for KGB rockfishes were compiled from references on different rockfish species. Rockfishes are viviparous and release larvae once per year. A finite survival rate of 0.463 for the larvae from time of release to the average age at entrainment was estimated using an instantaneous mortality rate of 0.14/day from blue rockfish (Mary Yoklavich, NOAA/NMFS/PFEG, Pacific Grove, CA, pers. comm. 1999) over 5.5 days ($0.463 = e^{(-0.14*5.5)}$). An average annual fecundity estimate of 213,000 eggs per female was used in calculating *FH* (DeLacy et al. 1964: 52,000-339,000; MacGregor 1970: 44,118-104,101 and

143,156-182,890; Love and Johnson 1999: 80,000-760,000). Estimates of five years as the age at maturity and 15 years for longevity were used in calculating *FH* (Burge and Schultz 1973, Wyllie Echeverria 1987, Lea et al. 1999). The model estimated that the reproductive output of 13 adult females at the age on maturity was entrained by the MBPP (Table 3-7). Variation due to sampling error had only a small effect on the range of estimates.

Adult Equivalent Loss

Total annual MBPP entrainment of KGB rockfish was used to estimate the number of equivalent adults theoretically lost to the population. The parameters required for formulation of *AEL* estimates for KGB rockfish were derived from data on larval blue rockfish survival. Survivorship of KGB rockfishes from parturition to an estimated recruitment age of three years was partitioned into six stages (Table 3-8). The estimate of *AEL* was calculated assuming the entrainment of a single age class having the average age of recruitment. The estimated number of equivalent adults corresponding to the number of larvae that would have been entrained by the proposed MBPP combined-cycle intake was 23 (Table 3-9). The uncertainty of the *AEL* estimate due to sampling error was very small.

Although the *FH* and *AEL* estimates were very close to the theoretical relationship of $2FH \equiv AEL$, the *AEL* was only extrapolated to age three. The estimate would decrease by extrapolating to five years, the age of maturity used in the *FH* calculations.

Table 3-7. Annual estimates of adult female kelp, gopher, and black-and-yellow (KGB) rockfish losses at Morro Bay Power Plant based on larval entrainment estimates using the fecundity hindcasting (*FH*) model for the January – December 2000 data. Upper and lower estimates represent the changes in the model estimates that result from varying the value of the corresponding parameter in the model.

	Estimate	**Estimate Std. Error**	**Upper *FH* Estimate of**	**Lower *FH* Estimate**	***FH* Range**
FH Estimate	13	8	37	5	32
Entrainment	6,407,000	189,000	14	12	2

Table 3-8. Survival of the kelp, gopher, and black-and-yellow (KGB) rockfish complex larvae to an age of three years, based on blue rockfish (*Sebastes mystinus*) data.

Lifestage	Day (Start)	Day (End)	Instantaneous Natural Daily Mortality (Z)	Lifestage Survival (S)
Early larval 1	0	5.5	0.14	0.463
Early larval 2	5.5	20	0.14	0.131
Late larval	20	60	0.08	0.041
Early juvenile	60	180	0.04	0.008
Late juvenile	180	365	0.0112	0.126
Pre-recruit	365	1,095	0.0006	0.645

Note: Survival was estimated from release as $S = e^{(-Z)(\mathrm{Day(end)}-\mathrm{Day(Start)})}$. Daily instantaneous mortality rates (Z) for blue rockfish larvae were used to calculate KGB larval survivorship and were provided by Mary Yoklavich (NOAA/NMFS/PFEG, Pacific Grove, CA, pers. comm. 1999). Annual instantaneous mortality was assumed as 0.2/year after two year average age of entrainment was estimated as 5.5 days based on average size at entrainment and a growth rate of 0.14 mm/day (0.006 in./day) (Yoklavich et al. 1996).

Empirical Transport Model

The estimated P_M value for the KGB rockfish complex was 0.027 (2.7%) for the period of entrainment risk applied in the model (11.3 days) (Table 3-10) (All of the data used in the *ETM* calculations are in Appendix D). The model included an adjustment for P_S (0.088) because this taxon occupies nearshore habitats that extend well beyond the sampling areas. The value of P_S was computed by using alongshore distance of the sampled source water area (9.6 km) and dividing it by the alongshore distance the larvae could have traveled during the 11.3 day larval duration at an average current speed of 11.3 cm/s. The *PE* estimates ranged from 0 to 0.3097 (Table 3-10). Although the largest *PE* estimate occurred for the January survey, the largest fraction of the population was collected during the April survey (f_i = 0.7218) when the *PE* estimate was an order of magnitude lower.

Table 3-9. Annual estimates of adult kelp, gopher, and black-and-yellow (KGB) rockfish losses at Morro Bay Power Plant due to entrainment using the adult equivalent loss (*AEL*) model for the January – December 2000 data. Upper and lower estimates represent the changes in the model estimates that result from varying the value of the corresponding parameter in the model.

	Estimate	Estimate Std. Error	Upper *AEL* Estimate	Lower *AEL* Estimate	*AEL* Range
AEL Estimate	23	15	69	8	61
Total Entrainment	6,407,000	189,000	24	22	2

Table 3-10. Estimates of KGB rockfish larvae at MBPP entrainment and source water stations from monthly surveys conducted from January 2000 through December 2000 used in calculating empirical transport model (*ETM*) estimates of proportional entrainment (*PE*) and annual estimate of proportional mortality (P_M). The daily cooling water intake volume used in calculating the entrainment estimates was 1,619,190 m^3, and the volume of the source water used in calculating the source water population estimates was 15,686,663 m^3. Bay volume = 20,915,551 m^3. The larval duration used in the calculations was 11.28 days. More detailed data used in the calculations are presented in Appendix E.

Survey Date	Bay *PE*	Offshore *PE*	Total *PE*	Proportion of Source Population for Period (f)
17-Jan-00	0.3097	0	0.3097	0.0099
28-Feb-00	0.1052	0.0988	0.0509	0.0239
27-Mar-00	0	0	0	0.1076
24-Apr-00	0.0533	0.0661	0.0295	0.7218
15-May-00	0.3785	0.0220	0.0208	0.1197
12-Jun-00	0	0	0	0.0169
10-Jul-00	0	0	0	0
8-Aug-00	0	0	0	0
5-Sep-00	0	0	0	0
2-Oct-00	0	0	0	0
27-Nov-00	0	0	0	0
18-Dec-00	0	0	0	0
	$\bar{x}$ = 0.0705	$\bar{x}$ = 0.0156	$\bar{x}$ = 0.0342	

Results for Other Taxa

The modeling results for other taxa selected for detailed assessment showed that both demographic models could only be used with about half of the fishes analyzed (Table 3-11). Differences in the demographic model results among taxa were generally proportional to the differences in entrainment estimates as shown by the decreasing 2*FH estimates for the six fishes analyzed. An exception was KGB rockfishes that had lower model estimates in proportion to their entrainment due to the longer lifespan and later age of maturity of this taxa group relative to the other fishes analyzed. The *ETM* estimates of P_M for the analyzed fishes ranged from 0.025 (2.5%) to 0.497 (49.7%) with the estimated effects being lowest for fishes with source populations that extended outside the bay into nearshore areas. The highest estimated effects occurred for combtooth blennies that are commonly found as adults among the fouling communities on the piers and structures that are located along the waterfront near the MBPP intake.

Table 3-11. Summary of estimated Morro Bay Power Plant entrainment effects based on fecundity hindcasting (*FH*), adult equivalent loss (*AEL*), and empirical transport (*ETM*) estimates of proportional mortality (P_M) models. The *FH* estimate is multiplied by 2 to test the relationship that $2 \cdot FH = AEL$. *ETM* model (P_M) calculated using nearshore extrapolation of source water population.

Taxon	Common Name	Total Entrainment	2*FH	AEL	P_M
Gobiidae	unidentified gobies	3.9×10^8	796,000	268,000	0.116
Leptocottus armatus	Pacific staghorn sculpin	1.7×10^7	*	*	0.051
Stenobrachius leucopsarus	northern lampfish	1.5×10^7	*	*	0.025
Quietula y-cauda	shadow goby	1.3×10^7	12,700	7,440	0.028
Hypsoblennius spp.	combtooth blennies	1.0×10^7	8,720	8,080	0.497
Sebastes spp. V_De	KGB rockfishes	6.4×10^6	26	*	0.027
Atherinopsis californiensis	jacksmelt	6.3×10^6	*	*	0.217
Genyonemus lineatus	white croaker	3.0×10^6	106	*	0.043
Clupea pallasii	Pacific herring	3.0×10^6	86	532	0.164
Scorpaenichthys marmoratus	cabezon	2.9×10^6	*	*	0.025

* - Information unavailable to compute model estimate.

3.3 DIABLO CANYON POWER PLANT

There were 97,746 larval fishes identified and enumerated from the 4,693 entrainment samples processed for the DCPP study (Table 3-12). These were placed into 178 different taxonomic categories ranging from ordinal to specific classifications. This list of taxa was much more diverse than the studies at SBPP and MBPP due to length of the sampling effort, number of samples collected, and greater variety of habitats found in the area around the DCPP. The taxa in highest abundance were those whose adults were generally found close to shore, in shallow water. One exception was the thirteenth most abundant taxon, the northern lampfish, whose adults are found midwater and to depths of 3,000 m (Miller and Lea 1972). Fourteen fish taxa (Table 2-4) were selected for detailed assessment using the *FH*, *AEL*, and *ETM* approaches based on their numerical abundance in the samples and their importance in commercial or recreational fisheries.

There were 43,785 larval fishes identified and enumerated from the 3,163 samples processed from the nearshore sampling area. These comprised 175 different taxa ranging from ordinal to specific levels of classification. Adults of these taxa live in a variety of habitats, from intertidal and shallow subtidal to deep-water and pelagic habitats. The taxa in highest abundance in the nearshore sampling area were those whose adults were typically pelagic or subtidal; the more intertidally or nearshore distributed species were found in lower abundance in the sampling area.

<u>DCPP Results for the KGB Rockfish Complex</u>

Larval rockfishes in the KGB complex showed distinct seasonal peaks of abundance at the DCPP intake structure, with their greatest abundance tending to occur between March and July (Figure 3-6). An El Niño began developing during the spring of 1997 (NOAA 1999) and was detected along the coast of California in fall of that year (Lynn et al. 1998). This may have slightly affected the density in 1998 compared with the previous year. The El Niño event did not affect seasonal peaks in abundance between years; during both periods KGB rockfish larvae first starting appearing in February, reached peak abundances in April-May, and were not present following late-July.

Table 3-12. Fishes collected during Diablo Canyon Power Plant entrainment sampling. Fishes comprising less than 0.4% of total not shown individually but lumped under “other taxa”.

Taxon	Common Name	Count	Percent of Total	Cumulative Percent
Sebastes spp. V_De (KGB rockfish complex)	rockfishes	17,576	18.0	18.0
Gibbonsia spp.	clinid kelpfishes	9,361	9.6	27.6
Rhinogobiops nicholsi	blackeye goby	7,658	7.8	35.4
Cebidichthys violaceus	monkeyface eel	7,090	7.3	42.6
Artedius lateralis	smoothhead sculpin	5,598	5.7	48.4
Orthonopias triacis	snubnose sculpin	4,533	4.6	53.0
Genyonemus lineatus	white croaker	4,300	4.4	57.4
Cottidae unid.	sculpins	3,626	3.7	61.1
Gobiidae unid.	gobies	3,529	3.6	64.7
Engraulis mordax	northern anchovy	3,445	3.5	68.3
Stichaeidae unid.	pricklebacks	2,774	2.8	71.1
Sebastes spp. V (blue rockfish complex)	rockfishes	2,731	2.8	73.9
Stenobrachius leucopsarus	northern lampfish	2,326	2.4	76.3
Sardinops sagax	Pacific sardine	2,191	2.2	78.5
Scorpaenichthys marmoratus	cabezon	1,938	2.0	80.5
Oligocottus spp.	sculpins	1,708	1.7	82.2
Bathymasteridae unid.	ronquils	1,336	1.4	83.6
Oxylebius pictus	painted greenling	1,133	1.2	84.8
Oligocottus maculosus	tidepool sculpin	1,035	1.1	85.8
Liparis spp.	snailfishes	900	0.9	86.7
Chaenopsidae unid.	tube blennies	817	0.8	87.6
Pleuronectidae unid.	righteye flounders	698	0.7	88.3
Clinocottus analis	wooly sculpin	683	0.7	89.0
Sebastes spp. V_D	rockfishes	656	0.7	89.7
Ruscarius creaseri	roughcheek sculpin	633	0.6	90.3
Artedius spp.	sculpins	623	0.6	90.9
Lepidogobius lepidus	bay goby	541	0.6	91.5
Bathylagus ochotensis	popeye blacksmelt	497	0.5	92.0
Paralichthys californicus	California halibut	378	0.4	92.4
Parophrys vetulus	English sole	361	0.4	92.8
Sebastes spp.	rockfishes	357	0.4	93.1
Osmeridae unid.	smelts	356	0.4	93.5
Neoclinus spp.	fringeheads	352	0.4	93.9
	144 other taxa	6,006	6.1	100.0
	Total Larvae	97,746		

There were 17,863 larval KGB rockfishes identified from 774 of samples collected at the DCPP intake structure between October 1996 and June 1999 representing 20% of the entrainment samples collected and processed during that period. Annual estimated numbers of KGB rockfish larvae entrained at DCPP varied relatively little between the 1996–97 Analysis Period 1 (268,000,000) and the 1997–98 Analysis Period 2 (199,000,000) (Table 3-13). An approximation of 95% confidence intervals (± 2 std. errors) for the two estimates overlap indicating that the differences between them were probably not statistically significant and that entrainment of KGB rockfish larvae was relatively constant between years.

Estimates of annually entrained KGB rockfish larvae were adjusted (Table 3-13) to the long-term average DCPP Intake Cove surface plankton tow index, calculated as the ratio between the 9 yr average of DCPP Intake Cove sampling (Figure 3-7) and the average annual index estimated from these same tows during the year being adjusted. Average indices for the years 1997 and 1998 were 0.070 and 0.065 larvae/m^3, respectively, and the long-term average index for 1990–98 was 0.072 larvae/m^3. Thus, the ratios used to adjust the 1997 and 1998 estimates of larvae entrained were 1.03 and 1.13, respectively, indicating that larval density was slightly lower than the long-term average during those years. Adjustments resulted in an estimate of 275,000,000 entrained KGB rockfish larvae for 1996–97 Analysis Period 1 and 222,000,000 for 1997–98 Analysis Period 2 (Table 3-13). The same trends in overall abundance as noted for unadjusted entrainment values were apparent in the adjusted values; namely, larval KGB rockfish abundance changed little between analysis periods. Annual estimates of abundance during the study period were low relative to the long-term average index of larval abundance from the Intake Cove plankton tows as indicated by the index ratios greater than one.

Larval KGB rockfishes generally occurred in the nearshore sampling area with similar seasonality to that observed at the DCPP intake structure with peak abundance occurring in May of both 1998 and 1999 (Figure 3-6). There were 5,377 KGB rockfish larvae identified from 701 samples representing 23% of the nearshore sampling area samples collected and processed from July 1997–June 1999. The mean concentrations in May of each sampling year were very similar (1998: 0.29/m^3; 1999: 0.28/m^3), indicating little change in abundance between the El Niño and subsequent La Niña years. The pattern of abundances in the nearshore sampling area differed between years with larger abundances of larvae in the sampling cells closest to shore during 1999 (Figure 3-8b). Regression analyses of the data for the two sampling periods showed declining abundances with increasing distance offshore (negative slope) for the 1999 period and almost no change with increasing distance offshore for the 1998 period (Appendix F).

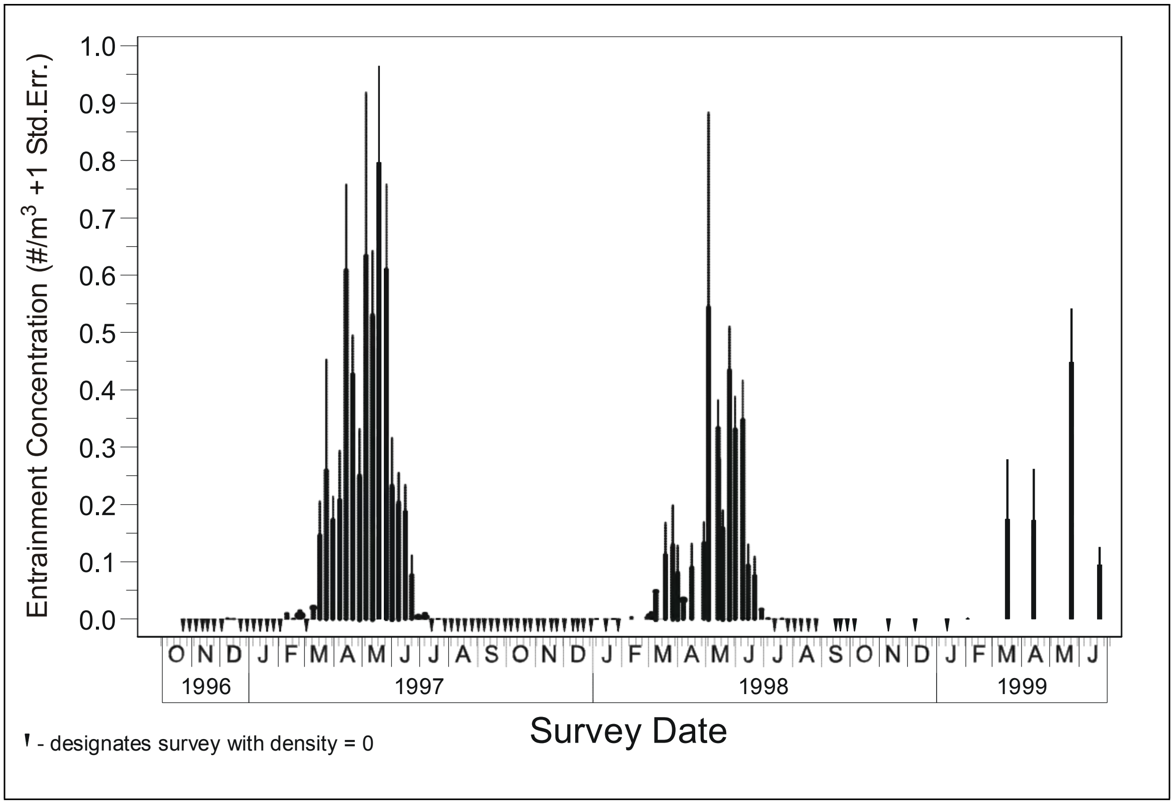


Figure 3-6. Weekly mean larval concentrations of kelp, gopher, and black-and-yellow (KGB) rockfish complex larvae at the Diablo Canyon Power Plant intake entrainment stations. Dark bars represent mean concentration and thinner bars represent one standard error.

Table 3-13. Diablo Canyon Power Plant entrainment estimates (E_T) and standard errors for kelp, gopher, and black-and-yellow (KGB) rockfish complex. E_{Adj-T} refers to the number entrained after adjustment to a long tern mean density. Note: The results for analysis periods 2 and 3 are the same because the overlap between the periods occurred during the peak larval abundances of KGB rockfish larvae.

Analysis Period	**E_T**	**$SE(E_T)$**	**E_{Adj-T}**	**$SE(E_{Adj-T})$**
1) Oct 1996 – Sept 1997	268,000,000	24,000,000	275,000,000	24,700,000
2) Oct 1997 – Sept 1998	199,000,000	25,900,000	222,000,000	28,900,000

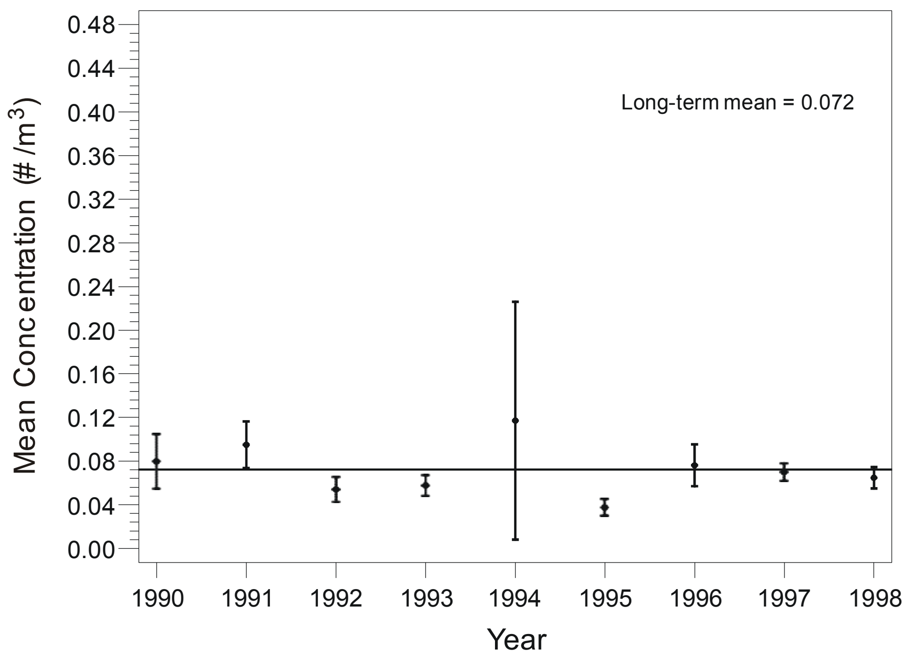


Figure 3-7. Annual mean concentration (+/- 2 standard errors) for kelp, gopher, and black-and-yellow (KGB) rockfish complex larvae collected from surface plankton tows in DCPP Intake Cove. Data were collected from December through June for every year except 1990 when only data from February through June were collected. The horizontal line is the long-term mean for all years combined.

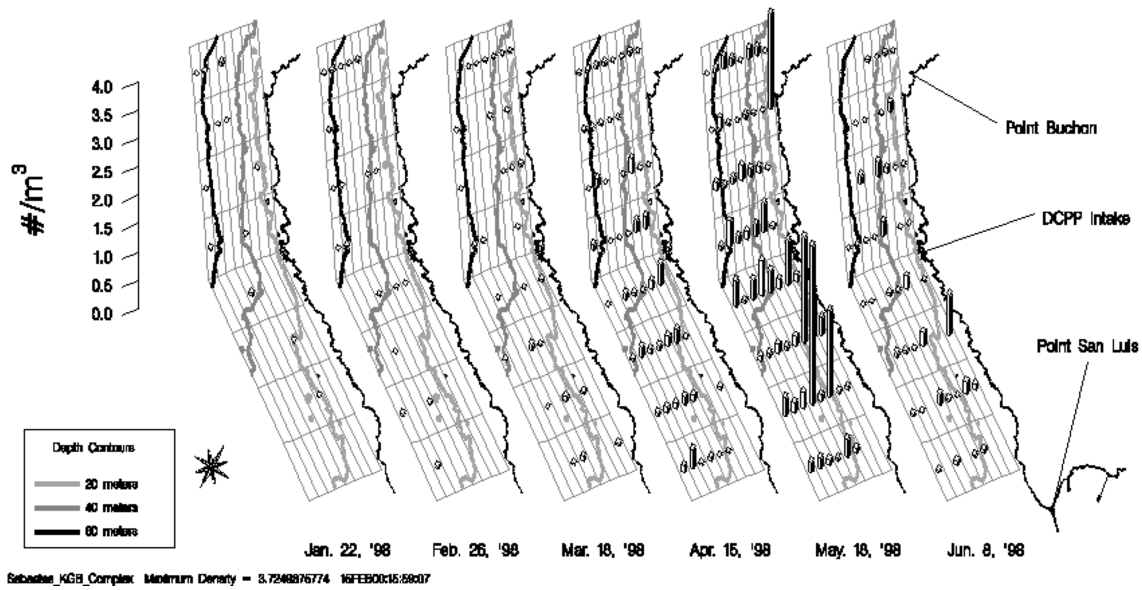
Standard lengths of all measured KGB rockfish larvae collected at the DCPP intake structure between October 1996 and June 1999 (9,926 larvae) ranged from 2.4 to 8.0 mm (mean = 4.2 mm) (Figure 3-9). The lengths of entrained KGB larvae, excluding the largest 1% and smallest 1% of all measurements, ranged from 3.3 to 5.6 mm. Similar to the KGB assessment at Morro Bay, a growth rate of 0.14 mm/d (Mary Yoklavich, NOAA / NMFS / PFEG, Santa Cruz, CA, pers. comm. 1999) was used to estimate the age of entrained larvae. Assuming that the size of the smallest 1% represents post-extrusion larvae that are aged zero days, then the estimated ages of entrained larvae ranged from zero up to ca. 16.4 d post-extrusion for the size of the largest 1% of the larvae. The estimated average age of KGB larvae entrained at DCPP was 6.4 d post-extrusion. The reported extrusion size for species in this complex ranges from 4.0–5.5 mm (Moser 1996).

Fecundity Hindcasting

The same life history parameter values used for the MBPP study (Table 3-8) were also used to calculate *FH* estimates for the KGB rockfish complex for the DCPP study. Average age at entrainment was estimated as 6.2 d. This was calculated by subtracting the value of the 1st percentile value of the lengths (3.3 mm) from the mean length at entrainment (4.2 mm) and dividing by the larval growth rate for brown rockfish of 0.14 mm/d (Love and Johnson 1999; Yoklavich et al. 1996) that was also used in the MBPP study. The survival rate of the KGB larvae from size at entrainment to size at recruitment into the fishery was partitioned into six stages from parturition to recruitment using the same approach presented for the MBPP study (Table 3-14). The survival rate from extrusion to the average age at entrainment using data from blue rockfish was estimated as 0.419 ($0.419 = e^{(-0.14)(6.2)}$).

The estimated number of adult KGB rockfish females at the age of maturity whose reproductive output was been lost due to entrainment was 617 for the 1996–97 period and 497 for the 1997–98 period (Table 3-14). The similarity between the estimates was a direct result of the similarity between adjusted entrainment estimates for the two periods. Low FH estimates resulted from the relatively high fecundity of adults and young average entrainment age estimated for larvae in this complex and not including other sources of mortality such as losses due to fishing in the model. The variation in the entrainment estimate had very little effect on the model estimates relative to the variation resulting from the life history parameters.

A) January 1998 – June 1998 surveys



B) January 1999 – June 1999 surveys

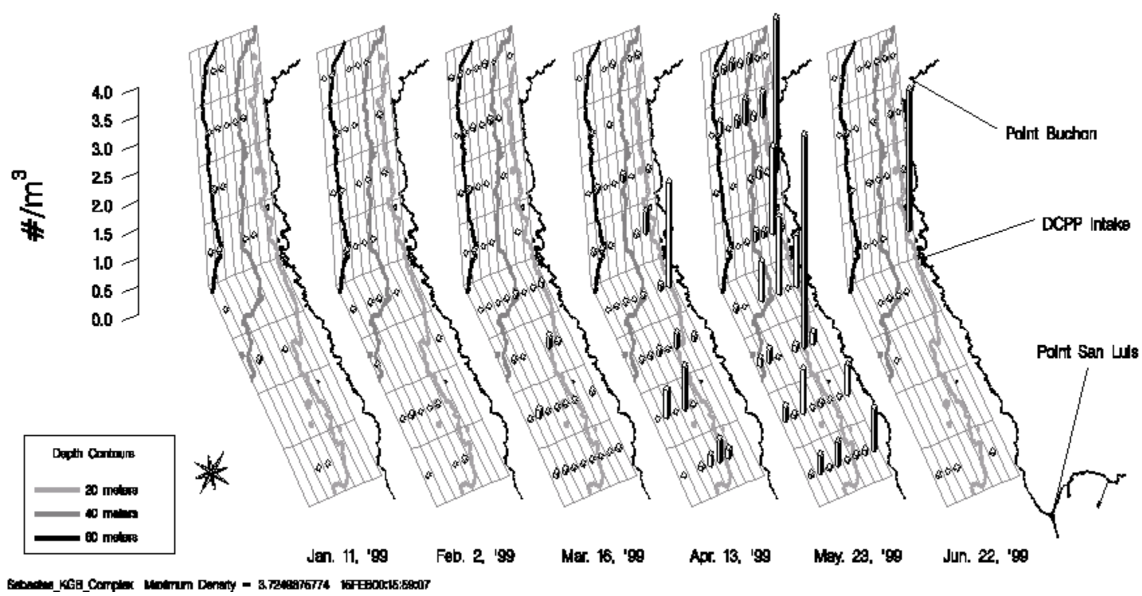


Figure 3-8. Average concentration for kelp, gopher, and black-and-yellow (KGB) rockfish complex larvae in each of the 64 nearshore stations for surveys done from A) January 1998 through June 1998, and B) January 1999 through June 1999 for Diablo Canyon Power Plant. Surveys done in other months are not shown because there were few or no KGB rockfish complex larvae collected.

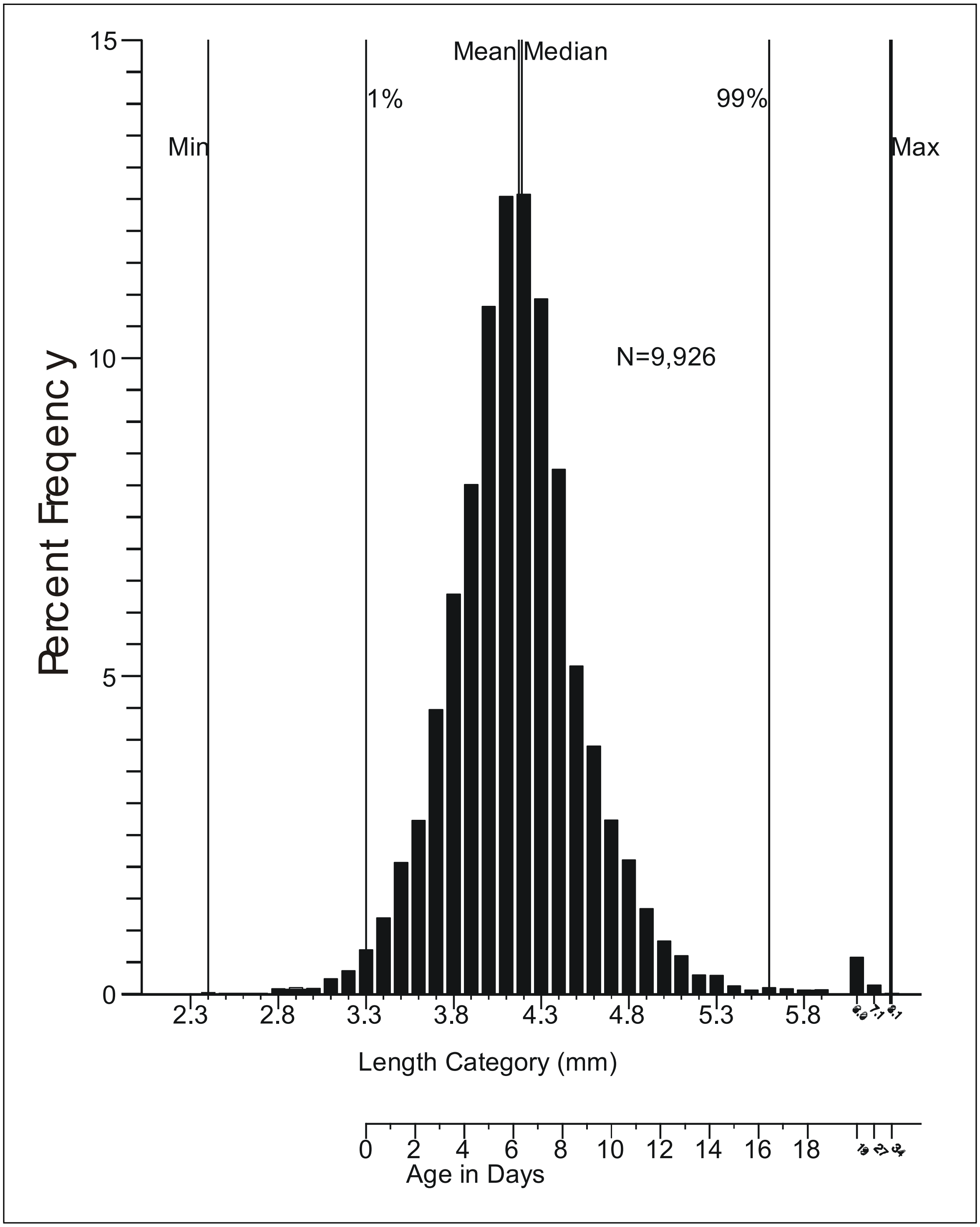


Figure 3-9. Length frequency distribution for kelp, gopher, and black-and-yellow (KGB) rockfish complex larvae measured from entrainment stations at Diablo Canyon Power Plant intake from October 1996 to June 1999. The x-scale is not continuous at larger lengths. Alternate x-scale shows age in days estimated using growth rate of 0.14 mm^{-d}.

Table 3-14. Diablo Canyon Power Plant fecundity hindcasting (*FH*) estimates for kelp, gopher, and black-and-yellow (KGB) rockfish complex for two year-long analysis periods. Upper and lower estimates represent the changes in the model estimates that result from varying the value of the corresponding parameter in the model.

Analysis Period	Adjusted Entrainment Estimate	Estimate Std. Error	Upper *FH* Estimate	Lower *FH* Estimate	*FH* Range
1) Oct 1996–Sept 1997					
FH Estimate	617	1,470	31,500	12	31,488
Adjusted Entrainment	275,000,000	24,700,000	708	526	182
2) Oct 1997–Sept 1998					
FH Estimate	497	1,190	25,400	10	25,390
Adjusted Entrainment	222,000,000	28,900,000	603	391	212

Adult Equivalent Loss

Similar to the *FH* calculations the same life history parameter values from blue rockfish used for the MBPP study (Table 3-8) were also used to calculate *AEL* estimates for KGB rockfish at DCPP. The *AEL* estimates were extrapolated forward from the average age at entrainment of 6.2 d, the same value used in the *FH* hindcasting. Survivorship, to an assumed recruitment age of 3 yr, was apportioned into these life stages, and *AEL* was calculated assuming the entrainment of a single age class having the average age of recruitment. Survival from the average age at entrainment (6.2 d) to the age at transformation (20 d) was estimated as 0.145 ($0.145 = e^{(-0.14)(20-6.2)}$). The other stages used the survival estimates from Table 3-14.

Paralleling the *FH* results, estimates of adult equivalents lost due to larval entrainment were fairly similar among survey periods (Table 3-15). The *AEL* estimate of 1,120 adults predicted from E_{T-Adj} at DCPP during 1996–97 reflects the slightly higher abundance of KGB rockfish larvae present during this year when compared to the 1997–1998 period (*AEL*= 905). The relatively constant larval abundance and subsequent estimates of effects varied little among survey periods, indicating that recruitment for the species in this complex remained relatively constant over the two years.

Similar to the results for MBPP, the *FH* and *AEL* estimates for DCPP were very close to the theoretical relationship of $2FH \equiv AEL$, the *AEL* was only extrapolated to age three. The estimate would decrease by extrapolating to five years, the age of maturity used in the *FH* calculations.

Table 3-15. Diablo Canyon Power Plant adult equivalent loss (*AEL*) estimates for kelp, gopher, and black-and-yellow (KGB) rockfish complex. Upper and lower estimates represent the changes in the model estimates that result from varying the value of the corresponding parameter in the model.

Analysis Period	Adjusted Entrainment Estimate	Estimate Std. Error	Upper *AEL* Estimate	Lower *AEL* Estimate	*AEL* Range
1) Oct 1996–Sept 1997					
AEL Estimate	1,120	3,410	166,000	8	165,992
Annual Entrainment	275,000,000	24,700,000	1,290	958	332
2) Oct 1997–Sept 1998					
AEL Estimate	905	2,750	134,000	6	133,994
Annual Entrainment	222,000,000	28,900,000	1,100	712	388

Empirical Transport Model

The data used in computing the *ETM* estimates of P_M for KGB rockfish for the two study periods are presented in Tables 3-16 and 3-17 and in more detail in Appendices E and F. Average *PE* estimates for the two periods were similar in value and the values of f_i showed that the largest weights were applied to the *PE* values for the April and May surveys in both periods (Table 3-16). The estimate of larval duration of 16.4 days was used in the *ETM* calculations for both study periods.

The *ETM* model used for DCPP included adjustments for P_S similar to the model used at MBPP. Unlike the MBPP study, P_S was calculated using two approaches. The first approach was similar to the MBPP study, but instead of using average current speed, alongshore current displacement was used to estimate the alongshore distance that could have been traveled by KGB rockfish larvae during the day of the survey and during the 16.4 day period prior to the survey that they were susceptible to entrainment (Table 3-17). The ratio of the alongshore length of the nearshore sampling area to the alongshore current displacement was used to calculate an estimate of P_S for each survey. The second approach used the alongshore current displacement to determine the alongshore length of the source water population, but also used onshore current movement over the same period to determine the offshore distance of the source water population. During the 1997-1998 period when the pattern of abundances within the nearshore sampling area was slightly increasing with distance offshore (positive slope) the offshore extent of the extrapolated source water population was set using the onshore current displacement (Table 3-17A and Appendix F). When the pattern of abundances showed a decline with distance offshore during

1998-1999 the estimated offshore extent was the distance offshore that the extrapolated density was equal to zero (x-intercept), or the offshore extent of the sampling area (3,008 m) if the x-intercept was inside of the sampling area (Table 3-17B and Appendix F). This was typically less than the measured onshore displacement during the surveys. The P_S was calculated as the ratio of the estimated number of KGB rockfish larvae in the nearshore sampling area to the estimated number in the source water area. The average values of P_S were used in the *ETM* calculations.

The *ETM* estimates for KGB rockfish are presented with the results of the other taxa included in the assessment for the DCPP (Table 3-18). *ETM* estimates of proportional mortality (P_M) were calculated using two methods to estimate the proportion of source water sampled (P_S). One method assumed that the source water only extended alongshore and did not extend outside of the nearshore sampling area. Only this first estimate was calculated for three fishes that occur primarily as adults in the shallow nearshore. The other method assumed that the source water extended alongshore and could extend some distance outside of the nearshore sampling area. Only this estimate was calculated for two fishes that occur as adults over large oceanic areas. Both estimates were calculated for the other nine fishes. No estimate was calculated for Pacific sardine in the Analysis Period 4 because of very low abundances that year.

Estimates of P_M were relatively similar in value between periods for the estimates calculated using the alongshore displacement estimate of P_S. There was a much greater difference between periods for the estimates calculated using the P_S based on extrapolating the source water population extending both alongshore and offshore. This was a result of the difference in the pattern of abundances in the nearshore sampling area between sampling periods (Figure 3-8). The source population was extrapolated further offshore during the 1997-1998 period resulting in a larger source water population estimate, which resulted in a smaller estimate of P_S and a smaller estimate of P_M.

Results for Other Taxa

Modeling results for the other taxa selected for detailed assessment showed that, similar to the results for MBPP, demographic models could only be used for half of the fishes analyzed (Table 3-18). There was a large variation in the demographic model results among taxa that was not necessarily reflective of the differences in entrainment estimates. This was the result of the large variation in life history among the fishes analyzed. For example, although the entrainment estimates for Pacific sardine and blue rockfish were similar the demographic model results were different by greater than two orders of magnitude.

Table 3-16. Estimates used in calculating empirical transport model (*ETM*) estimates of proportional entrainment (*PE*) for kelp, gopher, and black-and-yellow (KGB) rockfish complex for Diablo Canyon Power Plant from monthly surveys conducted for two periods A) July 1997 through June 1998, and B) July 1998 through June 1999. The larval duration used in the calculations was 16.4 days. More detailed data used in the calculations are presented in Appendices E and F.

A) July 1997 – June 1998

Survey Date	PE_i	PE_i Std. Error	f_i	f_i Std. Error
21-Jul-97	0.0107	0.0151	0.0004	0.0004
25-Aug-97	0	0	0	0
29-Sep-97	0	0	0	0
20-Oct-97	0	0	0	0
17-Nov-97	0	0	0	0
10-Dec-97	0	0	0.0003	0.0003
22-Jan-98	0.0008	0.0009	0.0121	0.0053
26-Feb-98	0.0021	0.0013	0.0180	0.0038
18-Mar-98	0.0587	0.0297	0.0279	0.0050
15-Apr-98	0.0076	0.0035	0.1732	0.0214
18-May-98	0.0036	0.0008	0.6384	0.0334
8-Jun-98	0.0353	0.0084	0.1297	0.0165
	0.0167	Sum =	1.00000	

B) July 1998 – June 1999

Survey Date	PE_i	PE_i Std. Error	f_i	f_i Std. Error
21-Jul-98	0.0033	0.0035	0.0035	0.0011
26-Aug-98	0	0	0	0
16-Sep-98	0	0	0	0
6-Oct-98	0	0	0	0
11-Nov-98	0	0	0	0
9-Dec-98	0	0	0	0
12-Jan-99	0	0	0.0240	0.0053
3-Feb-99	0.0005	0.0005	0.0243	0.0045
17-Mar-99	0.0327	0.0198	0.0809	0.0108
14-Apr-99	0.0137	0.0075	0.1906	0.0328
24-May-99	0.0115	0.0026	0.5926	0.0456
23-Jun-99	0.0170	0.0125	0.0841	0.0509
	0.0131	Sum =	1.00000	

Table 3-17. Onshore and alongshore current meter displacement used in estimating proportion of source water sampled (P_S) from monthly surveys conducted for two periods A) July 1997 through June 1998, and B) July 1998 through June 1999 for kelp, gopher, and black-and-yellow (KGB) rockfish complex at the Diablo Canyon Power Plant. More detailed data included in Appendices E and F.

A) July 1997 – June 1998

Survey Date	Cumulative Alongshore Displacement (m)	Onshore Current Displacement (m)	Estimated Offshore Extent of Source Water (m)	Offshore P_S	Alongshore P_S
21-Jul-97	31,300	4,820	4,820	0.0153	0.5545
25-Aug-97	–	–	–	–	–
29-Sep-97	–	–	–	–	–
20-Oct-97	–	–	–	–	–
17-Nov-97	–	–	–	–	–
10-Dec-97	146,000	31,600	31,600	0.0000	0.1189
22-Jan-98	120,000	23,400	23,400	0.0020	0.1443
26-Feb-98	33,700	8,710	8,710	0.0693	0.5152
18-Mar-98	181,000	12,400	12,400	0.0090	0.0960
15-Apr-98	76,100	12,800	12,800	0.0404	0.2282
18-May-98	67,100	19,900	19,900	0.0334	0.2589
8-Jun-98	111,000	5,670	5,670	0.0761	0.1559
			Average =	0.0307	0.2590

B) July 1998 - June 1998

Survey Date	Cumulative Alongshore Displacement (m)	Onshore Current Displacement (m)	Estimated Offshore Extent of Source Water (m)	Offshore P_S	Alongshore P_S
21-Jul-98	76,300	11,100	3,010	0.2278	0.2278
26-Aug-98	–	–	–	–	–
16-Sep-98	–	–	–	–	–
6-Oct-98	–	–	–	–	–
11-Nov-98	–	–	–	–	–
9-Dec-98	–	–	–	–	–
12-Jan-99	46,200	24,100	3,010	0.3755	0.3755
3-Feb-99	81,900	19,700	3,010	0.2122	0.2122
17-Mar-99	36,900	8,540	4,170	0.4334	0.4709
14-Apr-99	163,000	10,200	8,000	0.0636	0.1068
24-May-99	180,000	21,800	21,000	0.0251	0.0967
23-Jun-99	158,000	5,970	4,380	0.0986	0.1100
			Average =	0.2052	0.2286

The fishes analyzed were separated into three groups based on their adult distributions: fishes that were widely distributed over large oceanic areas included northern anchovy and Pacific sardine, fishes that were primarily distributed in the shallow nearshore included smoothhead sculpin (*Orthonopias triacis*), monkeyface prickleback (*Cebidichthys violaceus*), and clinid kelpfishes (*Gibbonsia* spp.), and the rest of the fishes that were primarily nearshore, but could be found in deeper subtidal areas. The source water population used in calculating P_S was estimated using both alongshore currents and along- and off-shore extrapolation for the last group of fishes resulting in two *ETM* estimates for each analysis period. Only one *ETM* estimate for each analysis period was made for the other two groups depending on whether it was primarily nearshore, or primarily offshore. The *ETM* estimates of P_M ranged from <0.001 (0.1%) to 0.310 (31.0%) with the estimated effects being greatest for the fishes that were distributed primarily as adults in shallow nearshore areas. These fishes such as sculpins (Cottidae), monkeyface pricklebacks, and kelpfishes all had proportional mortalities due to power plant entrainment of greater than 10%. The *ETM* calculations were calculated using both estimates of P_S for snubnose sculpin because they occur slightly deeper as adults than the other nearshore fishes. The results showed that the extrapolated *ETM* estimates were approximately equal to the estimates using only alongshore current displacement because the densities for this species did not increase with distance offshore. The results for DCPP are similar to the other two studies in showing that the greatest effects occur to fishes that primarily occupy habitats in close proximity to the intake and do not occur at the same level in other areas of the source water.

Table 3-18. Results of entrainment monitoring and *FH, AEL,* and *ETM* modeling for fourteen fishes at Diablo Canyon Power Plant. The four analysis periods correspond to 1) Oct. 1996 – Sept. 1997, 2) Oct. 1997 – Sept. 1998, 3) July 1997 – June 1998, and 4) July 1998 – June 1999. Adjusted entrainment (E_{Adj-T}), *FH* and *AEL* not calculated for Analysis Period 4. Nearshore sampling of source waters began in June 1998 so *ETM* estimates of proportional mortality (P_M) was only calculated for Analysis Periods 3 and 4.

Taxon	Analysis Period	E_{Adj-T}	*FH*	*AEL*	P_M Alongshore	P_M Offshore and Alongshore
Pacific sardine	1.	8,470,000	3,170	2,630	–	–
	2.	22,600,000	8,460	7,000	–	–
	3.	22,600,000	8,460	7,000	not calculated	<0.001
	4.				not calculated	not calculated
northern anchovy	1.	136,000,000	16,100	43,200	–	–
	2.	376,000,000	44,700	120,000	–	–
	3.	377,000,000	44,700	120,000	not calculated	<0.001
	4.				not calculated	<0.001
KGB rockfish complex	1.	275,000,000	617	1,120	–	–
	2.	222,000,000	497	905	–	–
	3.	222,000,000	497	905	0.039	0.005
	4.				0.048	0.043
blue rockfish complex	1.	84,040,000	43	353	–	–
	2.	33,800,000	18	164	–	–
	3.	33,900,000	20	142	0.004	<0.001
	4.				0.028	0.002
painted greenling	1.	24,200,000	–	–	–	–
	2.	9,610,000	–	–	–	–
	3.	12,100,000	–	–	0.063	0.051
	4.				0.056	0.043
smooth-head sculpin	1.	57,700,000	–	–	–	–
	2.	115,000,000	–	–	–	–
	3.	129,000,000	–	–	0.114	not calculated
	4.				0.226	not calculated
snubnose sculpin	1.	110,000,000	–	–	–	–
	2.	83,500,000	–	–	–	–
	3.	105,000,000	–	–	0.149	0.139
	4.				0.310	0.310
cabezon	1.	51,900,000	–	–	–	–
	2.	36,300,000	–	–	–	–
	3.	36,300,000	–	–	0.011	0.009
	4,				0.015	0.008
white croaker	1.	305,000,000	5,110	14,700	–	–
	2.	440,000,000	7,380	21,300	–	–
	3.	447,000,000	7,500	21,600	0.007	<0.001
	4.				0.035	0.004
Monkey-face prickleback	1.	83,100,000	–	–	–	–
	2.	61,500,000	–	–	–	–
	3.	60,200,000	–	–	0.138	not calculated
	4.				0.118	not calculated
clinid kelpfishes	1.	181,000,000	–	–	–	–
	2.	308,000,000	–	–	–	–
	3.	458,000,000	–	–	0.189	not calculated
	4.				0.250	not calculated
blackeye goby	1.	128,000,000	12,000	75,200	–	–
	2.	109,000,000	10,300	64,100	–	–
	3.	128,000,000	12,100	75,400	0.115	0.027
	4.				0.065	0.036
sanddabs	1.	7,160,000	426	2,370	–	–
	2.	1,540,000	92	511	–	–
	3.	6,610,000	393	2,190	0.010	0.001
	4.				0.008	0.001
California halibut	1.	8,260,000	–	–	–	–
	2.	15,700,000	–	–	–	–
	3.	15,500,000	–	–	0.005	0.001
	4.				0.071	0.006

4.0 DISCUSSION

The results from these studies demonstrate the importance of a site-specific approach to assessing the effects of CWIS entrainment on marine organisms. Even though Morro Bay and San Diego Bay are both tidally influenced embayments the resulting studies, sampling, and analytical approaches were very different. And both of these studies were dramatically different from Diablo Canyon. The source waters determined to be affected by entrainment were the primary factor responsible for the differences among studies. In San Diego Bay, in the area of SBPP, the turnover in water due to tidal exchange allowed us to treat the source water population as a closed system. A larger number of stations was sampled in San Diego compared to Morro Bay because of the potential for reduced exchange among the various habitats in the San Diego source water study area. Differences in fish composition among habitats in San Diego Bay shown by Allen (1999) were also reflected in some of the differences in larval composition among stations. This resulted in site-specific effects on species such as longjaw mudsuckers which had a relatively high *ETM* estimate of P_M at SBPP. Mudsucker larvae were not particularly abundant in the source waters but were abundant in the SBPP intake canal which provided excellent habitat for adults. Similarly, effects on combtooth blennies estimated using *ETM* were lower than other fishes because they were more abundant in areas of the bay that had extensive pier pilings and other structures that provide habitat for adult blennies. The high level of site fidelity in the community composition in south San Diego Bay was likely due to the lower tidal exchange rates relative to an area such as Morro Bay. The results supported our decision to sample an extensive range of habitats in south San Diego Bay.

The source water sampling in Morro Bay was less extensive than the SBPP study, but included sampling at a nearshore station outside of the bay that was representative of water transported into the bay on flood tides. The less intensive sampling was justified by the large tidal exchange that results in rapid turnover of the water in the bay relative to a large tidal embayment such as San Diego Bay. The shallow mudflats and tidal channels in Morro Bay are drained out through the deeper navigation channel where sampling occurred. Although this may have resulted in under-sampling of larvae from certain fishes that could avoid strong tidal currents, as has been shown for longjaw mudsuckers and other species of gobies (Barlow 1963, Brothers 1975), it was probably representative of the larvae that would be transported on outgoing tides past the plant where they would be exposed to entrainment. The greatest CWIS effects using *ETM* were estimated for combtooth blennies that occur in the piers and other structures located near the plant. This was similar to the SBPP results for longjaw mudsuckers that occur in highest numbers at the entrainment station in the intake canal. These results showed the importance of sampling all habitats and the potential for increased impacts on species with habitats near plant intakes. This also indicates that potential for large impacts exist when habitats are not uniformly distributed in the source water for a CWIS and the potential for larger effects on fishes associated with habitats that may not be abundant throughout the source water.

The nearshore sampling area for DCPP was very extensive to represent the range of habitats along the exposed rocky headland where the power plant is located. The size of the sampling area was also designed to be representative of the distance north and south that larvae could be transported by alongshore currents over a 24 hour period to correspond with the *ETM* model that uses daily estimates of conditional mortality resulting from entrainment to estimate CWIS-related mortality. This extensive sampling showed similar results to SBPP and MBPP by estimating that the greatest CWIS effects using *ETM* occurred on fishes with nearshore habitats that were disproportionately affected by entrainment. In the *ETM* model species that have higher abundances in entrainment samples results in larger *PE* estimates of daily conditional mortality.

We examined the relative distribution of individual species in the sampling areas by comparing the average *PE* to the ratio of the cooling water to source water volumes. For example, in SBPP the average *PE* for CIQ gobies was 0.012 which was very close to the volumetric ratio of 0.015. In contrast, the average *PE* for longjaw mudsuckers was 0.19 which was much greater than the ratio of cooling water to source water. Although this is potentially useful for helping to determine the potential distribution of the larvae in the source water it may not be a good indicator of impacts. When the *PE* is close to the volumetric ratio the resulting impacts are directly dependent on the number of days that the larvae are exposed to entrainment. Therefore, even though the average *PE* was much greater for longjaw mudsuckers, the time (4 days) that they were exposed to entrainment was much less than CIQ gobies because they were in highest abundance in the areas directly around the CWS intake. In contrast, even though the average *PE* for CIQ goby was close to the volumetric ratio, the estimated effects of entrainment based on *ETM* were higher than the estimated effects on mudsuckers (0.215 vs. 0.171) because goby larvae were estimated to be exposed to entrainment for 23 days.

The final source water area used to adjust the *PE* estimates also affected the CWIS effects estimated using *ETM*. The MBPP results for KGB rockfish contrast with those for estuarine fishes such as gobies and blennies. Relative to fishes that are primarily estuarine inhabitants, adult KGB rockfishes are more widely distributed resulting in larger source water body populations and reduced entrainment effects. As a result the *PE* estimates were adjusted using P_S to account for the larger source water population beyond the area sampled for KGB rockfishes. All of the results for DCPP were adjusted to account for the onshore and alongshore currents that can transport larvae over hundreds of kilometers, resulting in very low estimated effects for species, such as northern anchovy, that have widely distributed source populations.

The source water sampling for all three of these studies was done to satisfy the requirements of the *ETM*. Source water sampling would not have been required if the assessments were done using only more traditional demographic modeling approaches. The source water sampling was necessary because the *ETM* directly links mortality to a source population. As a consequence, the habitat occupied by that source population can be described and ecosystem losses can be mitigated. The area of production foregone (APF) is one approach for estimating the amount of habitat that would need to be replaced to compensate for the larval production lost due to entrainment.

Area of Production Foregone (APF) models can be used to understand the scale of loss resulting from an impact and the extent of mitigation that could yield compensation for the loss. It is based on the idea that losses from environmental impacts can usually only be estimated from a group of species and that the true impact results from the sum of direct and indirect losses attributable to the impact. The use of APF allows for the estimation of both the direct and indirect consequences of an impact and provides a currency (i.e., habitat acreage) that may be useful for understanding the extent of compensation required to offset an impact.

Probably the most controversial issue in APF assessment is how it treats the few taxa actually analyzed in the assessment. In most assessments, including Habitat Replacement Cost (HRC) (Strange et al. 2002), estimates of loss of taxa are implicitly considered to be without error. In APF, each estimate is considered to be prone to (sometimes) massive error (indeed, estimates of confidence intervals in ETM calculations often cross through zero). In APF models the assumption is that each taxon represent a sample and that the mean of the samples is representative of the true loss rate. For example, assume 5 taxa and the ETM calculations indicate that for an estuarine system of 2000 acres the loss rates for the 5 taxa are 5, 10, 3, 22 and 15 percent. In APF the estimate of loss would be the average of the 5 values or 11 percent. Because APF considers taxa to be simply independent replicates useful for calculating the expected impact, the choice of taxa for analysis may differ from HRC assessments. In APF the concern is more that each taxon is representative of other taxa that are either unsampled (most invertebrates, plants and holoplankton) or not analyzed (the vast majority of fish). In APF, the average loss across taxa then represents the average loss across all entrained organisms. This is a fundamental difference between APF and economic based models like HRC. The underlying statistical-philosophic basis of APF addresses one of the most problematic issues in impact estimation: the typical inability to estimate impact for unevaluated taxa.

In APF, the next step is to take the average ETM loss rate and turn it into an ecological currency, which then can be used to understand the impact and form a basis for mitigation. This can be quite a simple step. Loss is turned into habitat from which production is foregone. This is calculated as the area of habitat that would need to be added to the system to make up the lost resources. In the example above, the estimate was that 11% of organisms at risk in a 2000-acre estuary were lost to entrainment. The estimate of APF then would simply be 2,000 acres x 11% or 220 acres. Therefore the creation of 220 acres of new estuarine habitat would compensate for the losses due to entrainment. This does not mean that all biological resources were lost from an area of 220 acres, which is a common misunderstanding. Instead it means that if 220 acres of new habitat were created then all losses, calculated and not calculated, would likely be compensated for. Here again is an important feature of APF. The currency of impact (acres needed to compensate) includes all impacts, even indirect ones. One common criticism of the approach of focusing more detailed analysis to only a limited number of taxa is that not only are other taxa directly affected by entrainment not assessed, but that there is also no provision for estimation of indirect impacts (often food web considerations). APF addresses this concern by expressing impact in terms of habitat and assuming that indirect impacts are addressed by the complete compensation of all directly lost resources.

In the given example, APF would predict that the creation of 220 acres of new habitat would compensate for all impacts due to entrainment. What sort of habitat should be created? Again the statistical-philosophic basis of APF contributes to the answer. Because taxa in APF are simply independent replicates that yield a mean loss rate, habitat is not directed by taxa. Instead the approach assumes that habitat should be created that represents the habitat for the populations at risk. If the habitat in the estuary was 60% subtidal eelgrass beds, 15% mudflats and 25% vegetated intertidal marsh, then these same percentages should be maintained in the created habitat. Doing so would ensure that impacts on all affected taxa would be addressed.

The logic of the example would seem to imply that this methodology would only be useful if there were habitat creation opportunities. However even if there are not local opportunities, the approach is useful for other reasons:

1) Opportunities may exist in other locations (such as another nearby estuary);

2) Area of Production Foregone can be useful in understanding the scale and relative importance of the impact, which helps with permitting decisions, and in establishing a cost-basis for the impact; and

3) Often there are alternative mitigation strategies that could be implemented whose scale would be determined by APF. An example would be the size of the creation of an artificial reef or the area of a marine reserve designated as mitigation for entrainment losses.

In the most general model APF is estimated from the product of P_M and the source water area for each taxa analyzed. In the example above the source water area was the same for all taxa as it was the area of the estuary. Clearly, the approach becomes more difficult on the open coast where the source water areas differ across taxa. The task is simplified by the proportional relationship between P_M and the size of the source water population used in calculating P_S. As the size of the source water area increases relative to the sampling area, P_S decreases resulting in a proportional decrease in P_M. If the habitat in the larger source water can be assumed to be distributed in the same relative proportions as the area sampled then you only need to use the areas of various habitats in the sampled area to estimate APF by using the uncorrected P_M. This greatly simplifies the application of APF and also reduces the need to rely on limited current data information to extrapolate beyond the areas sampled. In practice, when many taxa are impacted, each having varying habitat requirements, APF estimation becomes a matter of restoration using an estimate such as

$$\frac{\sum_{i=1}^{N} \frac{1}{F_{S_i}} P_{M_i}}{N},$$

for I = 1 to N taxa.

One of the advantages of the *ETM* model over more traditional demographic approaches towards CWIS assessment is the reduced need for life history data. As the results show, the necessary life history information on reproduction and age-specific mortality for the *FH* and *AEL* models was only available for a limited number of fishes. The life history information was collected from data in the scientific literature, but the level of uncertainty surrounding published demographic parameters was rarely reported. The likelihood is that the uncertainty associated with the information was very large. This needs to be considered when interpreting results from *FH* and *AEL* models, because the accuracy of estimated entrainment effects will depend on the accuracy of age-specific mortality and fecundity estimates. This limits the utility of these modeling approaches especially on the Pacific coast of California where fishes in highest abundance in entrainment samples are small, forage species with limited life history information. We were fortunate that the work of Brothers (1975) provided us with demographic information on CIQ gobies, the most abundant larvae collected in two of the studies.

Unlike demographic models the only life history information required by *ETM*, which it shares with *FH* and *AEL*, is an estimate of the duration of the period of time the larvae are vulnerable to entrainment, estimated in these studies by the age of the larvae entrained. This was estimated in our studies using larval lengths measured from the samples and larval growth rates obtained or derived from the scientific literature. The average length was used to estimate the average age at entrainment (average length – length at 1st percentile) and the maximum length based on the length at the 99th percentile was used to estimate the maximum number of days that the larvae were exposed to entrainment. It is possible that these estimates were biased. Other reported data (e.g., Moser 1996) for various species suggested that hatching lengths could be either smaller or larger than the size estimated from the samples, and indicated that the smallest observed larvae represented 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 larval age.

The extensive weekly sampling at DCPP over more than two years resulted in measurements of almost 10,000 KGB rockfish larvae from entrainment samples. Despite this large data set, we did not have a high level of confidence that these data necessarily provided a more accurate estimate of size at extrusion. The reported size of KGB rockfish at extrusion is 4.0-5.5 mm (Moser 1996) indicating that the average size at entrainment, 4.2 mm, could be a more accurate minimum size for estimating age at entrainment than the much smaller value used in the calculations. Although the minimum and average sizes were different than reported in the literature this shouldn’t present a problem in estimating the number of days of exposure to entrainment as long as the growth rate used in the calculations is valid for that size of larvae. The uncertainty regarding the estimation of the period of exposure to entrainment has resulted in reporting of *ETM* results using larval durations based on the mean and maximum lengths at MBPP and DCPP. This uncertainty can easily be resolved by aging entrained larvae using otoliths. Removing the uncertainty associated with the age of the entrained larvae may justify the additional costs associated with this approach.

The duration that larvae may be subject to entrainment is affected by growth and behavior of the larvae, but also by the hydrodynamic characteristics of the source waters. In closed systems such as south San Diego Bay or freshwater lakes biological factors are probably more important than hydrodynamic factors. In open systems both biological and physical factors affect the length of time that larvae are subject to entrainment. For power plants located in coastal areas, such as DCPP, the effects of currents and larval growth both need to be considered in determining the size of the source population potentially affected by entrainment, but in estuarine areas such as Morro Bay hydrodynamic forces have a much greater effect on exposure to entrainment. The large tidal exchange ratio in Morro Bay results in huge exports of larvae out of the bay and into nearshore waters. Brothers (1975) showed that tidal exchange in Mission Bay, California resulted in much higher larval mortality rates than his calculated values for CIQ gobies. He hypothesized that larval behavior similar to that observed in longjaw mudsucker (Barlow 1963) resulted in the higher observed survival rates. Barlow described that longjaw mudsucker post-larvae are found close to the bottom. The location of MBPP near the harbor entrance of Morro Bay probably results in reduced effects on estuarine fish populations because the large majority of entrained larvae would be exported out to sea. The source water calculations for MBPP did not account for the strong effects of tidal exchange on entrainment exposure which was used to argue that mean larval lengths should have been used in calculating larval exposure to entrainment instead of the length of the 99th percentile. More sophisticated models incorporating hydrodynamic factors should be considered for estuarine systems similar to Morro Bay where hydrodynamic forces strongly affect the period of time that larvae are exposed to entrainment. This could have been done by increasing the source water volume to account for tidal outflow which transport larvae out of the bay into the ocean over the same number of days that the larvae are exposed to entrainment. This would also require that the nearshore area be included in the calculation of the source water population estimate because the larvae transported out of the bay would still be subject to entrainment.

The sampling frequency may be another source of bias associated with our estimate of the age of the larvae being entrained. The potential for biased sampling would be more prevalent in fishes that do not have prolonged spawning periods such as KGB rockfishes or on the East Coast where spawning occurs more seasonally. It would be less of a potential problem in fishes such as CIQ goby that have larvae that are present almost year-round. Entrainment sampling occurring on a monthly or less frequent basis could miss certain periods when certain age classes are present. Although more frequent sampling may not be required in the source water this may argue for more frequent weekly or bi-weekly entrainment sampling.

The frequency for source water sampling also needs to be considered for species with limited spawning periods. This should be one of the considerations in selecting taxa for detailed assessment since species with limited spawning periods will have few estimates of *PE* decreasing the confidence in the *ETM* estimates for those taxa. Unfortunately, the current sampling approach may also result in the selection of taxa that have prolonged spawning durations. This can be avoided if the period of spawning for important taxa can be accounted for in the study design.

In an entrainment assessment being prepared for the Potrero Power Plant in San Francisco Bay, the source water sampling frequency was increased during the spawning season for Pacific herring (*Clupea pallasii*) which was identified as an important species during the study design (Tenera Environmental, unpublished data). If this is not accounted for in the sampling and selection of species for analysis it may result in biased estimates for certain species. This is especially problematical if a species is collected relatively infrequently and in low numbers, but is included in the assessment because of its commercial or recreational value. Examples from these studies include Pacific herring at MBPP and California halibut (*Paralichthys californicus*) at DCPP. Both of these fishes represented less than 1.0% of the total larvae collected during entrainment sampling but were included in the assessments (Tables 2-4, 3-6, and 3-12). In both cases the results of the demographic modeling were important in placing the results for these species in context. In the case of Pacific herring at MBPP the *ETM* estimate of entrainment mortality of 16% represented the estimated loss of 532 adults calculated using the *FH* method (Table 3-11). No demographic estimates were available for California halibut at DCPP (Table 3-18). This problem did not occur at SBPP where the assessment was limited to the most abundant fishes regardless of their commercial or recreational value.

The approach used at SBPP for selecting taxa for analysis is acceptable if the taxa used in the assessment represent the range of habitats and fishes found in the source water potentially impacted by entrainment. If the list of taxa represent a reasonable sample from the fishes in the source water then the P_M estimates for the fishes can be averaged to obtain an estimate of the expected entrainment impacts on other fish and invertebrate larvae, zooplankton, and phytoplankton not included in the assessment. As the examples in the previous paragraph demonstrate, no single estimate of P_M may be particularly reliable, and therefore the use of the average P_M may be more appropriate as a estimator of average losses to the population. As previously discussed, the average value can be also used in calculating APF estimates for scaling restoration projects that could be used to compensate for entrainment losses.

Using averages for APF does not imply that there is an average mortality within the area estimated by the APF, but rather that averages are useful for estimating the amount of habitat affected. In order to view mortality spatially, it may be useful to allocate the mortality estimate over the area of the source population. A first approximation would be to allocate mortality in a linear or Gaussian fashion across the range of the source population. This was the approach used to estimate the cumulative effects of CWIS at all of the power plants in southern California (MBC and Tenera 2005). In this way mortality is equal to zero at the periphery of the source population, the furthest distances from the power plant intake. In addition, the source population is subject to stochastic and variable deterministic processes with a result of a changing source population area. Using current measurements, and numerical or physical modeling can be used to make further refinements.

The simple volumetric approach for estimating cumulative effects (MBC and Tenera 2005) can be expanded using more accurate estimates of P_M for a range of species. This would involve combining source water population, oceanographic, and hydrographic data from individual power plants. Cumulative effects result when the source water populations for the various power plants overlap. The *ETM* is easily adjusted to calculate cumulative effects by expanding the estimates of the source water and entrainment populations (Eq. 18) to include all of the power plants being considered.

The time period that larvae are exposed to entrainment needs to be adjusted for fishes with planktonic egg stages. This was not considered in these studies because the fishes analyzed for entrainment effects were mostly species that did not have a planktonic egg stage. Therefore the durations used in the *ETM* modeling for anchovies, croakers, and flatfishes should have been increased by the average number of days that the eggs for these fishes were potentially exposed to entrainment. Since it would not be feasible to age eggs collected from entrainment samples this adjustment would need to rely on estimates of egg duration from the scientific literature. This requires the assumption that the estimate of *PE* applies to both egg and larval stages and that mortality on passage through the cooling system is 100% for both egg and larval stages. If there is concern that egg stages are less abundant in the source waters than larval stages, separate *PE* estimates could be calculated for egg and larval stages using an approach similar to the original *ETM* concept presented by Boreman et al. (1978 and 1981) which conceptualized an *ETM* model incorporating separate *PE* estimates and durations for each life stage. This approach will be difficult to implement for most fishes because fish eggs can only be identified for a few species on the west coast. Therefore, the most conservative approach would be to assume that fish eggs are entrained in the same relative proportions as fish larvae and account for the egg planktonic duration in the assessment models. For organisms with available life history information, estimates of larval and egg survival can be used to estimate the number of eggs that would have been entrained from abundances of larvae in the samples.

One often proposed method to estimate egg entrainment is to assume a 1:1 eggs to larvae entrainment ratio. However, egg mortality may be significantly different than larval mortality. For example, the estimates of instantaneous natural mortality (M) rates for northern anchovy were 0.191 d^{-1} for eggs and 0.114 d^{-1} for larvae. One million eggs would become 512,477 larvae at the end of 3.5 days, the estimated duration of entrainment for eggs. At the end of a larval duration of 70 days, there would be 175 fish assuming negative exponential survival. The assumption of exponential survival and stable age distribution of eggs and larvae over the 3.5 and 70 day periods can be used to estimate the numbers of all ages by integration as follows:

$$N = \int_0^t N_0 e^{-Mt} dt = \left. \frac{N_0 e^{-Mt}}{-M} \right|_0^t .$$

Separate integration of eggs and larvae results in a 0.568:1 estimated entrainment ratio of eggs to larvae, thus showing a higher risk to larvae due to the prolonged susceptibility.

The focus of our discussion on *ETM* results reflects our belief that entrainment effects from CWIS are best assessed using this approach. Although we focus on *ETM*, the multiple modeling approaches used in these studies was

valuable for several reasons. First of all, the demographic models provide valuable context for assessing effects on commercially and recreationally valuable species that also allows for comparison with *ETM*. For example, DCPP estimates of *AEL* for KGB rockfishes were compared to harvest data assuming 100% catchability of adult equivalents and assuming no compensatory mortality. These assumptions likely result in overestimating fishery values (e.g., price per kilogram). Given these conditions, an estimated economic loss to the local fishery could be based on an average weight of 1.0 kg for a 3-yr old KGB rockfish recruiting to the live-fish fishery. The annual average *AEL* estimate of 1,013 rockfishes translates to a potential direct economic loss of $7,749 based on the average price of $7.65/kg. This value represented approximately 2% of the ex-vessel revenue attributed to KGB complex rockfishes landed at ports in the Morro Bay area in 1999 (PSMFC PacFin Database). Similar conversions to fishery value can be performed using *FH* estimates.

This type of conversion also allows for indirect comparison of demographic model results with *ETM* by similar conversion of *ETM* losses into fishery value. To continue our example using the DCPP results for KGB rockfishes, we assumed that the probable effect of entrainment losses at DCPP on fisheries was likely localized to the ports within the Morro Bay area since most fishes in this complex demonstrate high site fidelity (Lea et al. 1999). In addition, extension of effects based on alongshore currents and larval duration indicate that the area potentially affected was only three to seven times the size of the nearshore sampling area, which was likely within the range of fishers from either Port San Luis or Morro Bay. The estimate of entrainment mortality (P_M) was between 4–5% for this area. Applying this range of proportional reduction to the local catch from the Morro Bay area in 1999 yielded estimated dollar losses to the Morro Bay area fishery of approximately $20,000. In this example the fishery value estimates using *ETM* and *AEL* are reasonably close. The same type of indirect comparison could be done for species without any fishery value by converting *ETM* estimates of P_M to APF. The estimate of APF could be used with data on abundances to obtain estimates of adult populations that could be compared with demographic model results.

The demographic modeling approaches and conversions to fishery value using either demographic or *ETM* model results ignore any potential effects of compensation. We took this approach because there remain conflicting opinions whether larval mortality is compensated in some fashion. One side of the argument is that if compensation occurs, the estimates of *FH*, *AEL* and P_M will overestimate the number of adults lost and ecosystem losses (Saila et al. 1997). The response is that it is difficult to determine if compensation occurs at all (Rose et al. 2001, Nisbet et al. 1996). Additionally, if population mortality is density independent or weakly dependent, then the recruited population size will fluctuate in response to either changes in larval abundances or mortality. In the case of large density dependent mortality, little change due to changes in recruitment might be observed in local population sizes (Cayley et al. 1996). Field experiments on west coast species of fishes have been equivocal (e.g. Stephens et al. 1986) and recent studies on bocaccio (*Sebastes paucispinis*) showed no evidence of compensation in the stock-recruitment relationship (Tolimieri and Levin 2005). Currently, the USEPA and the California Energy Commission consider that compensation does not reduce impacts from entrainment and impingement on adult populations.

Results from demographic models are also necessary for combining estimates from entrainment and impingement unless independent data on adult fish populations are available for comparison with impingement losses. Impingement studies are designed to collect data on juveniles and adult fishes that are used to develop estimates of annual impingement. An *AEL* model is then used to extrapolate the number of impinged fishes either backward or forward to the numbers of adults of a certain age. By using the average age of reproductively mature females in the extrapolation these results can be combined with *FH* or *AEL* entrainment estimates to obtain estimates of the combined effects of impingement and entrainment. This approach assumes that the *FH* and *AEL* entrainment estimates are extrapolated to the same age used in the impingement estimates. Combined assessments can only be done on the few fishes with life history data available for estimating *FH*, *AEL* or one of the other demographic models. Fortunately, the total impingement losses at these three plants were relatively low due to the CWIS designs and species with the highest impingement estimates were not entrained in high abundances (Tenera Environmental 2000, 2001, 2004). This is not always the case and combining impingement and entrainment estimates into comprehensive CWIS assessments remains problematic for most species due to incomplete life history data.

Another approach for combining results from impingement and entrainment would involve using the numbers of impinged individuals for a species to estimate the relative losses to the population. The impingement mortality and entrainment mortality rate estimated by *ETM* can be converted to survival and multiplied to estimate cumulative CWIS effects. This approach involves the assumption that there is no compensatory mechanisms acting on the population between larval and adult stages such that entrainment losses estimated by *ETM* represent losses to the adult population. It also assumes that impingement and entrainment losses apply to the same stock. Although this is reasonable for a closed system such as south San Diego Bay, it would be much more difficult in an open system. In addition, there are few species with adequate data on adult stocks that could be used in this approach.

Finally, demographic model results provide a direct comparison with *ETM* results for both fishery and non-fishery species. It is obviously preferable to present data as both percentages relative to a source population using *ETM* and as absolute numbers of fishes using one or both demographic models. This helps ensure that P_M estimates are properly interpreted and instances where a large P_M that equates to only a few adults fishes are not misinterpreted. Ensuring the species included in the assessment were adequately sampled is the best way to avoid this type of problem. Unfortunately, these types of comparison are only possible for the limited number of fishes on the west coast with published life history data. This approach is also complicated by the uncertainty related to the levels of any compensatory, depensatory, or behavioral mechanisms that may have been operating on the subject populations when the life history data were collected. The availability and uncertainty associated with life history information continue to be the greatest limitations to the use of demographic models for CWIS assessment.

Despite these limitations, the USEPA made extensive use of demographic models in the assessments used in the rule making for 316(b). This was necessary because of the need to determine the economic costs associated with implementing certain technologies that could be used to help meet performance standards for impingement (80-95%) and entrainment (60-90%) reduction mandated in the new 316(b) rule. These methods will continue to be used due to the availability of an option for site-specific compliance. This option involves a cost-benefit analysis that compares the costs of technological or operational measures for achieving the performance standards against environmental benefits calculated using benefits valuation methods. As a result of this requirement there is active research being done to increase the availability of life history data for Pacific coast fishes.

4.1 GUIDELINES FOR ENTRAINMENT IMPACT ASSESSMENT

The three studies presented in this paper make it clear that it is not feasible to use a prescriptive approach to entrainment assessment design. Based on our experiences with these and other studies, we provide some general considerations that might be helpful in the design, sampling, and analysis of entrainment impact assessments. These comments are presented in the hopes that others may benefit from our experiences in conducting CWIS entrainment assessments.

Considerations for Study Design

1. Determine potential species that could be affected by entrainment using historical data on entrainment for the power plant, if available, and data from surrounding waters. Insure that sampling will account for any endangered, threatened, or other listed species that could potentially be affected by entrainment.
2. Determine the source water areas potentially affected by entrainment including the distribution of habitats that might be differentially affected by CWIS entrainment. Different habitats may require use of different sampling gear and methods.
3. We have used oblique tows with bongo and wheeled bongo frames that sample the entire water column for both entrainment and source water because the intake structures for these plants were assumed to withdraw water from the entire water column. Power plants with intakes that withdraw water from a discrete depth in the water column may require the use of pumps or closing nets for entrainment sampling at discrete water depths where water withdrawal occurs. Hydrodynamic studies should be done to verify the intake flow field for sampling at discrete depths. We have not used pumps to sample inside of power plant cooling water systems because of potential bias due to predation by biofouling organisms.
4. Determine appropriate sampling frequency based on species composition and important species that might have short spawning seasons. This could include adjusting sampling frequency seasonally based on presence of certain species. Sampling of entrainment can be done more frequently than source water sampling to provide more accurate estimates of length frequencies of entrained larvae and may also be desirable to provide more accurate estimates for calculating baseline conditions for compliance with new 316(b) rules.
5. These studies were generally conducted over a one-year period except in the case of DCPP where one of the strongest ENSO events of that century occurred during the first year of sampling. The relative effects of entrainment estimated by the *ETM* model should be much less subject to interannual variation than absolute estimates using *FH*, *AEL* or other demographic models. Therefore if source water sampling is done in conjunction with entrainment sampling one year is a reasonable period of sampling for these studies.
6. Use hydrodynamics of source waters to determine appropriate sampling area. In a closed system this may be the entire source water. In an open

system, ocean or tidal currents should be used to determine the appropriate sampling area for estimating daily entrainment mortality (PE) for the larger source water population.

Ad hoc rule 1: Since *PE* is estimated as a daily mortality the sampling area should include the area potentially affected during a 24 h period. This area is a pragmatic way to arrive at a first stage estimate of daily mortality and hence survival. The use of a current meter positioned near the intake but outside the influence of its flow allows the estimation of advection in the nearby source water. The current meter approach can be combined with estimates of larval dispersion (Largier 2003) for an understanding of the magnitude of source water population affected.

Ad hoc rule 2: The *PE* is applied to a larger source population that is potentially affected in the time period of a larval duration. (Another option would be to use the range of the stock.) In an open system, the estimation of P_M includes extrapolating the population of the sampling area to the larger source water population over a larval duration. It is difficult to say that the single current meter accurately reflects the advection of the source water population to the intake. In addition, a single current meter says very little about diffusion processes. Be sure that appropriate physical data are collected during the study to model hydrodynamics and determine size of source population.

7. The uncertainties associated with estimating larval durations, and hydrodynamics used in estimating the size of the source water populations make estimating variance for *ETM* problematic. One approach we have used is to base the variance calculations solely on the sampling variances used in estimating the variance of *PE*. A similar approach would use the CV from the source water sampling (which includes both entrainment and source water data) to estimate the variance for *ETM* or use a Monte Carlo approach using the upper and lower confidence limit values for the *PE* values. These approaches have been considered because of the large unrealistic error terms derived using the Delta method that incorporates all of the multiple intercorrelated sources of error in the model.

Considerations for Sampling and Processing

1. We have used sample volumes of 30-60 m^3 per sample for these and other studies but this volume should be adjusted for the larval concentrations in the source waters. The appropriate sample volume is best determined by preliminary sampling using the gear proposed for the study.
2. Be sure that mesh size used for net sampling is appropriate for taxa that might be the focus of detailed analysis. We have used 335 µm mesh nets

because we have observed fish larvae being extruded through 505 µm mesh nets. Much smaller sized mesh would be needed to sample invertebrate larvae effectively.

3. Although we generally combine the subsamples from the two bongo nets for analysis, preserving one of them directly in 70-80% ethanol allows for genetic analyses to be conducted and analysis of otoliths to determine age and growth rates. Larval fishes are generally easier to identify when initially preserved in 5-10% formalin.
4. If ageing using larval otoliths is not done, be sure that length frequencies measured from entrainment samples are realistic based on available life history. We applied general rules for using the length data for determining mean, minimum, and maximum ages, but would recommend developing criteria based on the length frequency distribution for each species.
5. Be sure to account for egg stages that would be subject to entrainment if fish eggs are not sorted and identified from the samples.

Considerations for Analysis

1. Use multiple modeling approaches to validate results and provide additional data for determining effects at the adult population level.
2. Similar to the approach of using multiple models to provide additional data for determining effects at the adult population level, the *ETM* results can be converted into another currency using APF. This approach is probably most appropriate for scaling restoration projects that could be used to help offset losses due to entrainment.
3. Although *FH* and *AEL* models can be hindcast or extrapolated to the same age they will not necessarily provide the same estimate unless the data used in the two models are derived from a life table assuming a stable age distribution.
4. *FH* and *AEL* are estimates of the number of adults at a specific age. To estimate the number of adult females in the population, N_F, the average fecundity can be used instead of *TLF*. The *AEL* analog is extrapolation to all adult fish ages - *AEL'*. A comparison can be made using the relation $AEL'=2N_F$. This age of entry into the adult population may need to be adjusted to the average age of fishery catch if comparisons are being made with fishery data. The use of *AEL* and *FH* (Horst 1975 and Goodyear 1978), aligning at fishery age, is one method of estimating losses in terms of adult animals.
5. Another estimate would use production foregone or total biomass that would have been produced by entrained or impinged animals, had they not been entrained or impinged (Rago 1984). Production foregone includes all biomass lost through all forms of mortality had the animals survived entrainment or impingement. This measure is most often used for forage species and represents ecosystem losses, e.g. to other trophic

levels. Age-1 equivalent loss is a measure similar to *AEL* and *FH* that is most commonly used for harvested species. The USEPA (2002) used age-1 equivalents to evaluate power plant losses “because methods are unavailable for valuing fish eggs and larvae.” They conservatively estimated fish landings value using the number of age-1 individuals, as the average fishery age is older in most cases. However the USEPA believed the method may underestimate the true value of reducing impingement and entrainment because life history data were not available for most species. If survival rates from the age of entrainment until adulthood are accurate, *FH* and *AEL* underestimate the numbers of lost adults because they are extrapolated to a single age, e.g. age of maturity in the case of *FH*. An improved approach to *FH* will be to use the average annual fecundity to estimate the equivalent number of females N_F removed from the standing stock of adults. Similarly, *AEL* can be extrapolated to all adult ages and summed to estimate the number of adult equivalents *AEL'* and these measures can then be compared with fishery losses. However, the accuracy of these kinds of estimates is subject to the accuracy of the underlying survival and fecundity estimates.

6. Another estimate of the number of equivalent adults lost by larval entrainment is to use the mortality estimate from the *ETM* procedure and apply it to a survey of the standing stock. This accuracy of this estimate is subject to the accuracy of the estimate of the source population affected. This method may result in improvements when there is little confidence in survival estimates or when there is conjecture about compensatory processes that may negate the underlying models of *AEL* and *FH*.

4.2 CONCLUSION

As should be clear from this report, we feel that CWIS impacts are best evaluated using empirically based source water body information and the *ETM* model, and not using demographic models based on life history information derived from various sources with varying, or unknown, levels of confidence. Although demographic models are useful for providing context for *ETM* estimates there is no reason to base an assessment solely on demographic modeling results with the availability of approaches such as the *ETM* that provide estimates based on empirically derived estimates. In contrast to demographic models, uncertainty associated with *ETM* model estimates can be controlled through changes to the sampling design for the entrainment and source water sampling. The CEC and CCC have all required the *ETM* approach in recent studies. Hopefully the information in this paper will assist others in the design and analysis of CWIS assessments that meet the requirements of both 316(b) and regulatory requirements of other agencies.

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APPENDIX A

VARIANCE EQUATIONS FOR IMPACT ASSESSMENT MODELS

A1. Fecundity Hindcasting (*FH*)

The variance of *FH* was approximated by the Delta method (Appendix E2) (Seber 1982):

$$Var(FH) = (FH)^2 \left[CV^2(E_T) + \sum_{j=1}^{n} CV^2(S_j) + CV^2(\bar{F}) + \left(\frac{Var(A_L) + Var(A_M)}{(A_L - A_m)^2} \right) \right]$$

where

$CV(E_T)$ = CV of estimated entrainment,
$CV(S_j)$ = CV of estimated survival of eggs and larvae up to entrainment,
$CV(\bar{F})$ = CV of estimated average annual fecundity,
A_M = age at maturation, and
A_L = age at maturity.

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

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

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

+/-1.645, i.e.

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

A2. Adult Equivalent Loss (*AEL*)

The *AEL* approach uses estimates of the abundance of entrained or impinged organisms to forecast the loss of equivalent numbers of adults. Starting with the number of age class *j* larvae entrained (E_j), it is conceptually easy to convert these numbers to an equivalent number of adults lost (*AEL*) at some specified age class from the formula:

$$AEL = \sum_{j=1}^{n} E_j S_j ,$$

where

n = number of age classes,
E_j = estimated number of larvae lost in age class j, and
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.

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

$$AEL = E_T \prod_{j=1}^{n} S_j ,$$

where

n = number of age classes from entrainment to recruitment and
S_j = survival rate from the beginning to end of the jth age class.

The variance of *AEL* can be estimated using a Taylor series approximation (Delta method of Seber 1982) as

$$Var(AEL) = AEL^2 \left(CV^2(E_T) + \sum_{j=1}^{n} CV^2(S_j) \right) .$$

A3. Proportional Entrainment and *ETM*

The Empirical Transport Model (*ETM*) calculations provide an estimate of the probability of mortality due to power plant entrainment. The values used in calculating proportional entrainment (*PE*) are population estimates based on the respective larval densities and volumes of the cooling water system flow and source water areas. On any one sampling day, the conditional entrainment mortality can be expressed as

$$PE_i = \frac{\text{abundance of entrained larvae}_i}{\text{abundance of larvae in source population}_i}$$
$$= \text{probability of entrainment in } i\text{th time period } \left(i = 1, \ldots, N\right).$$

In turn, the daily probability can be estimated and expressed as

$$PE_i = \frac{E_i}{R_i}$$

where

E_i = estimated abundance of larvae entrained in the ith time period $\left(i = 1, \ldots, N\right)$;

R_i = estimated abundance of larvae at risk of entrainment from the source population in the ith time period $\left(i = 1, \ldots, N\right)$.

The variance for the period estimate of *PE* can be expressed as

$$Var\left(PE_i\right) = Var\left(\frac{E_i}{R_i}\middle| E_i, R_i\right).$$

Assuming zero covariance between the entrainment and source and using the delta method (Seber 1982), the variance of an estimator formed from a quotient (like PE_i) can be effectively approximated by

$$Var\left(\frac{A}{B}\right) \approx Var(A)\left(\frac{\partial\left[\frac{A}{B}\right]}{\partial A}\right)^2 + Var(B)\left(\frac{\partial\left[\frac{A}{B}\right]}{\partial B}\right)^2.$$

The delta method approximation of $Var(PE_i)$ is shown as

$$Var\left(PE_i\right) = Var\left(\frac{E_i}{V_S \cdot \rho_{Si}}\right)$$

which by the Delta method can be approximated by

$$Var\left(PE_i\right) \approx Var\left(E_i\right)\left(\frac{1}{V_S \cdot \bar{\rho}_{S_i}}\right)^2 + Var\left(V_S \cdot \bar{\rho}_{S_i}\right)\left(\frac{-E_i}{\left(V_S \cdot \bar{\rho}_{S_i}\right)^2}\right)^2$$

and is equivalent to

$$= PE_i^2\left[CV\left(E_i\right)^2 + CV\left(V_S \cdot \bar{\rho}_{S_i}\right)^2\right]$$

where

$$R_i = V_S \cdot \bar{\rho}_{S_{ij}} \text{ and}$$

$$CV\left(\theta\right) = \frac{Var\left(\theta\right)}{\theta^2}.$$

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APPENDIX B. Mean larval fish concentrations (larvae per 1000 m^3) by station for monthly surveys from February 2001 through January 2002 in San Diego Bay.

		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**	

APPENDIX C. Estimates of CIQ goby larvae at South Bay Power Plant entrainment and source water stations from monthly surveys conducted from February 2001 through January 2002 used in calculating empirical transport model (*ETM*) estimates of proportional entrainment (*PE*) and annual estimate of proportional mortality (P_M). The daily cooling water intake volume used in calculating the entrainment estimates was 2,275,244 m^3, and the volume of the source water used in calculating the source water population estimates was 149,612,092 m^3. The number of days that the larvae were exposed to entrainment was estimated at 22.86 days.

Survey Date	Entrainment Concentration (#/m^3)	Estimated Number Entrained	Source Water Concentration (#/m^3)	Estimated Number in the Source Water	*PE* Estimate	Days in Survey Period	Estimate of Source Water Population for Period	Proportion of Source Population for Period (*f*)	$=f_i(1-PE_i)^d$
28-Feb-01	2.143	4,877,000	5.712	8.546E+08	0.0057	41	3.504E+10	0.2165	0.1900
29-Mar-01	1.069	2,433,000	3.643	5.451E+08	0.0045	29	1.581E+10	0.0977	0.0882
17-Apr-01	1.997	4,544,000	2.794	4.180E+08	0.0109	19	7.942E+09	0.0491	0.0382
16-May-01	2.036	4,633,000	1.770	2.649E+08	0.0175	29	7.682E+09	0.0475	0.0317
14-Jun-01	3.747	8,525,000	2.311	3.458E+08	0.0247	29	1.003E+10	0.0620	0.0350
26-Jul-01	4.047	9,208,000	2.740	4.100E+08	0.0225	42	1.722E+10	0.1064	0.0633
23-Aug-01	0.648	1,475,000	2.609	3.904E+08	0.0038	28	1.093E+10	0.0675	0.0619
25-Sep-01	1.057	2,406,000	2.307	3.452E+08	0.0070	33	1.139E+10	0.0704	0.0600
23-Oct-01	1.254	2,852,000	2.553	3.820E+08	0.0075	28	1.070E+10	0.0661	0.0557
27-Nov-01	1.655	3,764,000	2.390	3.576E+08	0.0105	35	1.252E+10	0.0773	0.0607
20-Dec-01	1.861	4,233,000	2.745	4.107E+08	0.0103	23	9.446E+09	0.0584	0.0461
17-Jan-02	3.554	8,087,000	3.132	4.686E+08	0.0173	28	1.312E+10	0.0811	0.0545
				Average =	0.0118			P_M=	0.2147

APPENDIX D. Estimates of KGB rockfish larvae at MBPP entrainment and source water stations from monthly surveys conducted from January 2000 through December 2000 used in calculating empirical transport model (*ETM*) estimates of proportional entrainment (*PE*) and annual estimate of proportional mortality (P_M). The daily cooling water intake volume used in calculating the entrainment estimates was 1,619,190 m^3, and the volume of the source water used in calculating the source water population estimates was 15,686,663 m^3. Bay volume = 20,915,551 m^3. The larval duration used in the calculations was 11.28 days.

Survey Date	Estimated Number Entrained	Estimated Number in the Bay	Bay *PE*	Estimated Number in the Offshore Area	Offshore *PE*	Total *PE*	Source Water Population for Period	Proportion of Source Population for Period (*f*)	$=f_i(1\text{-}PE_iP_S)^d$
17-Jan-00	5,500	17,800	0.3097	0	–	0.3097	17,800	0.0099	0.0073
28-Feb-00	2,180	20,700	0.1052	22,100	0.0988	0.0509	42,800	0.0239	0.0227
27-Mar-00	0	6,550	–	186,000	–	–	192,000	0.1076	0.1076
24-Apr-00	38,100	715,000	0.0533	576,000	0.0661	0.0295	1,291,000	0.7218	0.7010
15-May-00	4,460	11,800	0.3785	202,000	0.0220	0.0208	214,000	0.1197	0.1173
12-Jun-00	0	14,900	–	15,000	–	–	30,300	0.0169	0.0169
10-Jul-00	0	0	–	0	–	–	0	–	–
8-Aug-00	0	0	–	0	–	–	0	–	–
5-Sep-00	0	0	–	0	–	–	0	–	–
2-Oct-00	0	0	–	0	–	–	0	–	–
27-Nov-00	0	0	–	0	–	–	0	–	–
18-Dec-00	0	0	–	0	–	–	0	–	–
			$\bar{x}$ = 0.0705		$\bar{x}$ = 0.0156	$\bar{x}$ = 0.0342			P_M = 0.0271

APPENDIX E. Estimates used in calculating empirical transport model (*ETM*) estimates of proportional entrainment (*PE*) for kelp, gopher, and black-and-yellow (KGB) rockfish complex for Diablo Canyon Power Plant. Entrainment estimates and estimates from the nearshore sampling area from monthly surveys conducted for two periods A) July 1997 through June 1998, and B) July 1998 through June 1999. The daily cooling water intake volume used in calculating the entrainment estimates was 9,312,114 m^3, and the volume of the sampled source water used in calculating the nearshore population estimates was 1,738,817,356 m^3. The larval duration used in the calculations was 16.4 days.

A) July 1997 – June 1998

Survey Date	Start Date Based on Larval Duration	Estimated Number Entrained	Entrainment Std. Error	Estimated Population in Nearshore Sampling Area	Nearshore Population Std. Error	PE_i	PE_i Std. Error	f_i	f_i Std. Error
21-Jul-97	5-Jul-97	2,770	2,770	258,000	255,000	0.0107	0.0151	0.0004	0.0004
25-Aug-97	9-Aug-97	0	–	0	–	–	–	–	–
29-Sep-97	13-Sep-97	0	–	0	–	–	–	–	–
20-Oct-97	4-Oct-97	0	–	0	–	–	–	–	–
17-Nov-97	1-Nov-97	0	–	0	–	–	–	–	–
10-Dec-97	24-Nov-97	0	–	216,000	216,000	–	–	0.0003	0.0003
22-Jan-98	6-Jan-98	6,280	6,280	7,775,000	3,345,000	0.0008	0.0009	0.0121	0.0053
26-Feb-98	10-Feb-98	23,900	13,900	11,534,000	2,267,000	0.0021	0.0013	0.0180	0.0038
18-Mar-98	2-Mar-98	1,051,000	503,000	17,903,000	2,903,000	0.0587	0.0297	0.0279	0.0050
15-Apr-98	30-Mar-98	847,000	376,000	111,247,000	12,360,000	0.0076	0.0035	0.1732	0.0214
18-May-98	2-May-98	1,468,000	288,000	409,996,000	51,937,000	0.0036	0.0008	0.6384	0.0334
8-Jun-98	23-May-98	2,940,000	622,000	83,336,000	9,213,000	0.0353	0.0084	0.1297	0.0165
					Mean =	0.0167	Sum =	1.0000	

B) July 1998 – June 1999

Survey Date	Start Date Based on Larval Duration	Estimated Number Entrained	Entrainment Std. Error	Estimated Population in Nearshore Sampling Area	Nearshore Population Std. Error	PE_i	PE_i Std. Error	f_i	f_i Std. Error
21-Jul-98	5-Jul-98	7,000	7,000	2,118,000	636,000	0.0033	0.0035	0.0035	0.0011
26-Aug-98	10-Aug-98	0	–	0	–	–	–	–	–
16-Sep-98	31-Aug-98	0	–	0	–	–	–	–	–
6-Oct-98	20-Sep-98	0	–	0	–	–	–	–	–
11-Nov-98	26-Oct-98	0	–	0	–	–	–	–	–
9-Dec-98	23-Nov-98	0	–	0	–	–	–	–	–
12-Jan-99	27-Dec-98	0	–	14,709,000	3,038,000	–	–	0.0240	0.0053
3-Feb-99	18-Jan-99	6,830	6,830	14,905,000	2,462,000	0.0005	0.0005	0.0243	0.0045
17-Mar-99	1-Mar-99	1,621,000	967,000	49,607,000	5,491,000	0.0327	0.0198	0.0809	0.0108
14-Apr-99	29-Mar-99	1,601,000	825,000	116,783,000	22,089,000	0.0137	0.0075	0.1906	0.0328
24-May-99	8-May-99	4,168,000	868,000	363,131,000	33,925,000	0.0115	0.0026	0.5926	0.0456
23-Jun-99	7-Jun-99	877,000	287,000	51,558,000	33,815,000	0.0170	0.0125	0.0841	0.0509
					Mean =	0.0131	Sum =	1.0000	

APPENDIX F. Regression estimates, onshore and alongshore current meter displacement, source water estimates, and estimates of the proportion of source water sampled (P_S) from monthly surveys conducted for two periods A) July 1997 through June 1998, and B) July 1998 through June 1999 for kelp, gopher, and black-and-yellow (KGB) rockfish complex at the Diablo Canyon Power Plant. The common slope used in calculating source water estimates was 0.000117 for the 1997-1998 period and -0.000367 for the 1998-1999 period. The ratio of the length of the nearshore sampling area (17,373 m) to the alongshore current displacement was used to calculate P_S for each survey (alongshore P_S). The regression coefficients and onshore and alongshore current displacement were used to calculate an estimate of the population in the source water for each survey. The ratio of the estimated population in the nearshore sampling area to the estimated population in the source water was used to calculate an estimate of P_S for each survey (offshore P_S).

A) July 1997 - June 1998

Survey Date	Y-Intercept	X-Intercept	Cumulative Alongshore Displacement (m)	Onshore Current Displacement (m)	Estimated Offshore Extent of Source Water (m)	Extrapolated Number Beyond Nearshore Sampling Area	Total Extrapolated Offshore Source Population	Total Extrapolated Alongshore Source Population	Offshore P_S	Alongshore P_S
21-Jul-97	-0.171	1,460	31,300	4,820	4,820	16,382,000	16,848,234	466,000	0.0153	0.5545
25-Aug-97	–	–	–	–	–	–	0	0	–	–
29-Sep-97	–	–	–	–	–	–	0	0	–	–
20-Oct-97	–	–	–	–	–	–	0	0	–	–
17-Nov-97	–	–	–	–	–	–	0	0	–	–
10-Dec-97	-0.172	1,470	146,000	31,600	31,600	7,772,826,000	7,774,642,009	1,816,000	<0.0001	0.1189
22-Jan-98	-0.015	125	120,000	23,400	23,400	3,753,412,000	3,807,288,976	53,877,000	0.0020	0.1443
26-Feb-98	0.064	-545	33,700	8,710	8,710	144,140,000	166,528,437	22,388,000	0.0693	0.5152
18-Mar-98	0.165	-1,410	181,000	12,400	12,400	1,801,789,000	1,988,251,728	186,463,000	0.0090	0.0960
15-Apr-98	2.115	-18,000	76,100	12,800	12,800	2,264,580,000	2,752,044,506	487,464,000	0.0404	0.2282
18-May-98	8.127	-69,400	67,100	19,900	19,900	10,706,927,000	12,290,666,879	1,583,740,000	0.0334	0.2589
8-Jun-98	1.376	-11,700	111,000	5,670	5,670	559,792,000	1,094,442,999	534,651,000	0.0761	0.1559
								Mean =	0.0307	0.2590

B) July 1998 - June 1999

Survey Date	Y-Intercept	X-Intercept	Cumulative Alongshore Displacement (m)	Onshore Current Displacement (m)	Estimated Offshore Extent of Source Water (m)	Extrapolated Number Beyond Nearshore Sampling Area	Total Extrapolated Offshore Source Population	Total Extrapolated Alongshore Source Population	Offshore P_S	Alongshore P_S
21-Jul-98	0.596	1,620	76,300	11,100	3,010	0	9,299,000	9,299,000	0.2278	0.2278
26-Aug-98	–	–	–	–	–	–	0	0	–	–
16-Sep-98	–	–	–	–	–	–	0	0	–	–
6-Oct-98	–	–	–	–	–	–	0	0	–	–
11-Nov-98	–	–	–	–	–	–	0	0	–	–
9-Dec-98	–	–	–	–	–	–	0	0	–	–
12-Jan-99	0.859	2,340	46,200	24,100	3,010	0	39,166,000	39,166,000	0.3755	0.3755
3-Feb-99	0.859	2,340	81,900	19,700	3,010	0	70,254,000	70,254,000	0.2122	0.2122
17-Mar-99	1.529	4,169	36,900	8,540	4,170	9,113,397	114,452,000	105,339,000	0.4334	0.4709
14-Apr-99	2.936	8,003	163,000	10,200	8,000	744,108,728	1,837,168,000	1,093,059,000	0.0636	0.1068
24-May-99	7.716	21,036	180,000	21,800	21,000	10,709,111,477	14,464,376,000	3,755,264,000	0.0251	0.0967
23-Jun-99	1.605	4,376	158,000	5,970	4,380	54,169,916	522,822,000	468,652,000	0.0986	0.1100
								Mean =	0.2052	0.2286

Edmund G. Brown, Jr.
Governor

VARIATION IN ENTRAINMENT IMPACT ESTIMATIONS BASED ON DIFFERENT MEASURES OF ACCEPTABLE UNCERTAINTY

PIER FINAL PROJECT REPORT

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Public Interest Energy Research Program

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Preface

The California Energy Commission's Public Interest Energy Research (PIER) Program supports public interest energy research and development that will help improve the quality of life in California by bringing environmentally safe, affordable, and reliable energy services and products to the marketplace.

The PIER Program conducts public interest research, development, and demonstration (RD&D) projects to benefit California.

The PIER Program strives to conduct the most promising public interest energy research by partnering with RD&D entities, including individuals, businesses, utilities, and public or private research institutions.

PIER funding efforts are focused on the following RD&D program areas:

- Buildings End-Use Energy Efficiency
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- Energy-Related Environmental Research
- Energy Systems Integration
- Environmentally Preferred Advanced Generation
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- Renewable Energy Technologies
- Transportation

Variation in Entrainment Impact Estimation Based on Different Measures of Acceptable Uncertainty is the final report for the Environmental Effects of Cooling Water Intake Structures Project (Contract Number 500-04-025), conducted by the University of California, Santa Cruz. The information from this project contributes to PIER's Energy-Related Environmental Research Program.

For more information about the PIER Program, please visit the Energy Commission's website at www.energy.ca.gov/research/.

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Abstract

A significant number of California's coastal power plants use once-through cooling. This technology diverts huge amounts of water from a water body into the power plant's cooling system before being discharged back. Millions of small aquatic organisms that are carried along in this water flow are killed as they pass through the power plant; this impact is referred to as entrainment. Power plant operators are required to assess and, if appropriate, mitigate or compensate for entrainment impacts. To determine the size and type of projects, such as wetland restoration, that could compensate for these losses, a method known as the Area of Production Foregone is used. This method has been used in most, if not all, recent power plant entrainment studies in California. The Area of Production Foregone is an estimate of the area of habitat that, if provided, would produce the larvae lost due to entrainment and therefore compensate for the impact. This calculation is based upon another model that estimates the portion of a population lost to entrainment in comparison to the overall population in the water body affected by the cooling water intake. As the number of studies using this approach have increased, two major statistical issues remain unresolved: (1) how to estimate and incorporate statistical error into estimation of Area of Production Foregone and (2) the effect of sample size (number of species used in the assessment) on estimation of Area of Production Foregone. This study found: (1) explicit incorporation of statistical error may lead to an increase in the area of restoration or creation required for compensation; and (2) the number of species sampled dramatically affects the estimation of Area of Production Foregone, but only when the required likelihood of complete compensation is greater than 50 percent. This report documents ways to improve the use and accuracy of this method and therefore benefits California by ensuring appropriate mitigation when entrainment impacts occur.

Keywords: Once-through cooling, Area of Production Foregone, Empirical Transport Model, Habitat Production Foregone, entrainment.

Executive Summary

Introduction
Nineteen power plants in California, representing more than 19,000 megawatts of capacity and located along the state's coast, bays and estuaries, use once-through cooling technology to condense steam used in producing electricity. Once-through cooling technology requires the diversion of millions of gallons of water per day from a water body. This water is then circulated through the power plant's cooling system and then discharged back to marine water bodies.

Power plants in California using this cooling technology are subject to provisions of the U.S. Clean Water Act. Specifically, Section 316(b) of the act requires that the location, design, construction, and capacity of cooling water intake structures reflect the best technology available to protect aquatic organisms from being killed or injured. Cooling water intake structures impact aquatic organisms by either impingement or entrainment. Impingement is where larger organisms are pinned against screens located at the entrance to the cooling water intake structure. Entrainment is where organisms that are small enough pass through the screens are carried by the water into the power plant's cooling systems where they are subjected to thermal, physical, or chemical stresses.

While assessment of impingement impacts can easily be determined through monitoring, the assessment of entrainment impacts presents special challenges. These include that fact that entrained organisms, which include fish and invertebrate larvae, are difficult not only to sample, but also to identify to an informative level. The distribution and variability of these populations in local waters may also be difficult to determine. Finally, there is great difficulty in scaling such losses such that the currency of impact is interpretable and useful when assessing mitigation options.

Project Objectives
The recent history of assessing the impact from entraining small marine organism by power plants has relied heavily on the use of the Empirical Transport Model. The Empirical Transport Model estimates the portion of a population that will be lost to entrainment by determining both the number of larvae from that population that will be entrained as well as the size of the larval populations found in the source water body. The source water body is the area where larvae are at risk of being entrained and is based primarily upon biological and oceanographic factors. Recent determinations using Empirical Transport models have calculated the average mortality across target species and used this number as the best estimate of mortality for all entrained organisms.

Using this information, the Area of Production Foregone (APF) can be calculated. The Area of Production Foregone, also known as Habitat Production Foregone, is an estimate of the area of habitat that, if provided, would produce enough larvae to compensate for those larvae lost due to entrainment. This has usually been based on species specific APF values that were used to generate a mean APF across species. More recently, APF estimation has incorporated the use of statistical error by developing confidence limits in APF calculation. These help provide an approach for addressing the specific question: what is the likelihood the calculated APF is large enough to provide, if used as a basis for mitigation, full compensation for the impact?

Empirical Transport Model and Area of Production estimates are based upon values derived from a limited number of target species and then used as the best estimate for all entrainable species. Target species are selected based on their abundance and the ease of collecting and identifying their larval stages. Because of this, a limited number of fish and, occasionally, crab species have been used for entrainment. The assumption, thus far untested, is that target species are reasonable representatives for the other species not targeted.

The goals of this project are to evaluate the effect of (1) incorporating statistical error in estimating Areas of Production Foregone and (2) the number of species in estimating Area of Production Foregone.

Project Outcomes

There were two major results of this study. First, as expected, explicit incorporation of statistical error leads to an increase in the area required for restoration or creation. As an example, increasing the level of confidence that the mean falls within the specified range from 50 percent to 95 percent increases the required area about 50 percent (across all studies). Using a more conservative increase from 50 to 80 percent produced, on average, an increase in area of about 25 percent. Assuming a direct relationship between area and cost, this means that the cost of increasing the likelihood of attaining full compensation from 50 to 80 percent would add an additional 25 percent to the cost of the mitigation project.
Second, the number of species sampled dramatically affects the estimate of the Area of Production Foregone, but only when the confidence limit is greater than 50 percent. The lack of change for the 50 percent confidence limit is because the expected mean does not change as a function of sample size. Instead, statistical error increases, which, when using confidence limits other than 50 percent, will affect estimates of the Area of Production Foregone. This result points to an important policy implication: if policy mandates that the 50 percent confidence limit for the Area of Production Foregone value (mean) be used to assess impacts and as a measure of compensatory mitigation, sample size is theoretically unimportant, because the expected mean does not vary with number of species assessed. The key implication of this result is that minimizing cost during sampling and assessment may be countered by the increased cost of compensatory mitigation (for example, habitat creation or restoration) due to inadequate sampling, which typically leads to greater statistical error.

Benefits to California

The California State Water Resources Control Board recently adopted a policy for assessing and mitigating the effects of power plants using once-through cooling technology. This policy identifies the use of the Habitat Production Foregone (referred to in this report as the Area of Production Foregone) as the appropriate method to show how power plant operators have achieved reductions in power plant entrainment impacts. Furthermore, other state agencies, such as the California Energy Commission and the California Coastal Commission, have used this method to identify the type and size of wetland restoration needed to address the entrainment impacts of power plants using once-through cooling. This report documents ways to improve the use and accuracy of this method and therefore benefits California by ensuring appropriate mitigation when entrainment impacts occur.

Unless otherwise noted, all tables and figures in this report were generated by the authors for this study.

1.0 Introduction

Nineteen power plants in California, representing over 19,000 MW of capacity and located along the state's coast, bays and estuaries, use once-through cooling technology to condense steam used in producing electricity. Once-through cooling technology requires the diversion through the power plant cooling system and then discharge of millions of gallons of water per day.

Power plants in California using this cooling technology are subject to provisions of the Clean Water Act. Specifically, Section 316(b) of the act requires that the location, design, construction, and capacity of cooling water intake structures reflect the best technology available to protect aquatic organisms from being killed or injured by impingement (being pinned against screens at the entrance to the cooling water intake structure) or entrainment (being small enough to pass through the screens and drawn into cooling water systems and subjected to thermal, physical or chemical stresses).

While assessment of impingement impacts can easily be determined through monitoring, assessment of entrainment impacts presents special challenges. These challenges include that fact that entrained organisms, which include fish eggs and fish and invertebrate larvae, are difficult not only to sample but also to identify to an informative level. The distribution and variability of these populations in local waters are often difficult to determine. There is also great difficulty in scaling such losses such that the currency of impact is interpretable and useful when assessing mitigation options.

The recent history of assessing the impact from entraining small marine organism by the intake of cooling water by power plants has relied heavily on the use of the Empirical Transport Model (ETM). The ETM estimates the portion of a larval population that will be lost to entrainment by determining both the amount of larvae from that population that will be entrained as well as the size of the larval populations found in the source water body. The source water body is the area where larvae are at risk of being entrained and is determined by biological and oceanographic factors. Recent determinations using ET models have calculated the average mortality across target species and used this as the best estimate of mortality for all entrained organisms.

Often ET models have been used in conjunction with demographic models that translate larval losses to adults using either hindcast (Fecundity Hindcast, [FH]) or forecast modeling (Adult Equivalent Loss, [AEL]). However the utility of the FH and AEL models has been hampered by the need for species specific life history information that is lacking for many species entrained in California. These models also suffer from an attribute that is rarely talked about but is fundamentally important and which separates these models from ETM models. Results in FH and AEL models are specific to the species modeled whereas those in ETM models are applicable across species.

To understand this it is helpful to use an example. Assume that an entrainment assessment has been conducted and that all three models were used. FH modeling will estimate the number of adult females that are required to produce the entrained larvae. AEL models will estimate the number of adults that would have resulted from the lost larvae. ETM models will estimate the percent of larvae at risk that were killed due to entrainment (called proportional mortality [P_M]) and the area of the population at risk (called source water body [SWB]). Also assume that the total number of species that were used in modeling was 10. While this is a large number for most 316(b) studies, this is a tiny fraction of the species actually entrained and lost. Hence, the utility of the models must be related to the degree that the model is useful as a proxy for other species not included in the models.

This condition is essential but has never been evaluated. Both FH and AEL models will end up producing numbers of lost adults. Because of the filter of life history, particularly fecundity and early survivorship, there is no expectation that these numbers also estimate species not modeled. By contrast, ETM estimates simply yield the proportional loss of larvae and source water body. The species specific product of P_M and SWB gives the Area of Production Foregone (APF), which is an estimate of the area of habitat that if provided would produce the larvae lost due to entrainment. Importantly, APF estimates should be and have been much more robust to life history variation than either FH or AEL estimates. Hence, it is expected that some estimator of replicate measures of APF (e.g. mean, median, 95% confidence interval) may be a proxy for other species entrained but not directly modeled. Typically, mean APF has been used, but recently the 80% confidence limit was used in a case before the California Coastal Commission (Poseidon Resources [Channelside] 2008). Explicit incorporation of statistical uncertainty (that leads to confidence limits) into APF evaluation has been constrained because of the lack of assessment of the effect of such incorporation and also because the method of incorporation of uncertainty (henceforth called error) has not been vetted.

As noted, the basis of ETM for impact assessment of entrainment is target species, which are used to estimate the general effect on entrainable organisms. Such species are selected based on their abundance, their ease of collection and on the ability to determine their identity based on larval characteristics (Steinbeck et al. 2007). Because of limitation in all these criteria, the vast majority of target organisms in ETM estimation have been a select group of fish species (note, certain species of crabs are also sometimes used). Recent determinations using ET models have calculated the average proportional mortality across target species and used this as the best estimate of proportional mortality for all entrained organisms. The major, thus far untested assumption is that target species are proxies for other species not targeted. Figure 1 schematically represents target organisms as a fraction of species entrained.

The goals of this project were to evaluate the effect of (1) incorporation of statistical error in estimation of APF and (2) sample size (number of species for which APF is assessed) on estimation of APF. For the first goal, both resampling theory and traditional parametric approaches were utilized, while resampling theory was the basis of the approach to address the second goal.

Fundamentals of the Empirical Transport Model (ETM)

A detailed description of the ETM can be found in Steinbeck et al (2007). The following is derivative of that paper. Results of empirical transport modeling provide an estimate of the conditional probability of mortality (P_M) associated with entrainment. P_M requires an estimate of proportional entrainment (P_E) as an input, which is an estimate of the daily entrainment mortality on larval populations in that body of water subject to entrainment, called the source water body (SWB). Empirical transport modeling has been used extensively in recent entrainment studies in California (Steinbeck et al. 2007) and elsewhere (e.g. at the Salem Nuclear Generating Station in Delaware Bay, New Jersey and at other power stations along the east coast of the United States (Boreman et al. 1978, 1981; PSE&G 1993). ETM derivations have also been developed (MacCall et al. 1983) and used to assess impacts at the San Onofre Nuclear Generating Station (SONGS; Parker and DeMartini 1989).

The basic form of the ETM incorporated many time-, space-, and age-specific estimates of mortality as well as information regarding spawning periodicity and larval duration (Boreman et al. 1978, 1981). Much of this type of information is unknown for species entrained in California, Hence, a variation of ETM has been developed for use for coastal once through cooling (OTC) systems in California. The essence of the approach is the compounding of P_E over time, which allows estimation of P_M using assumptions about species-specific larval life histories, specifically the length of time in days that the larvae are in the water column and exposed to entrainment.

On any sampling day *i*, P_E can be expressed as follows:

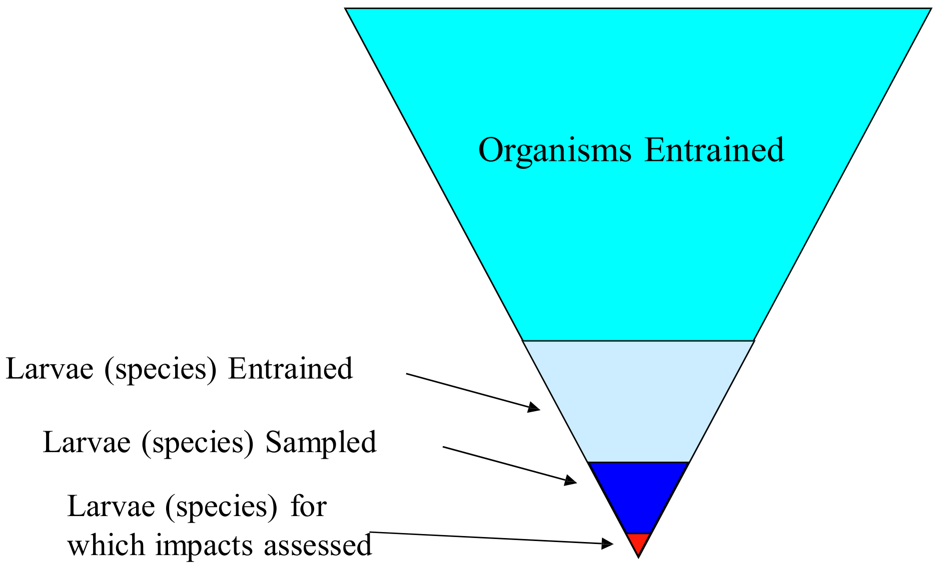

Figure 1. The inverse triangle of entrainment assessment.

$$P_{E\,i} = \frac{E_i}{N_i} \qquad (1)$$

where

E_i = total numbers of larvae of species entrained during a day during the i^{th} survey; and

N_i = numbers of larvae at risk of entrainment, i.e., abundance of larvae in the sampled source water during a day during the i^{th} survey.

Survival over one day = $1\text{-}P_{Ei}$, therefore survival over the number of days (d) that the larvae are vulnerable to entrainment = $(1\text{-}P_{Ei})^d$. Here d is determined based on a derived age distribution of entrained individuals. The derivation is based on the measured size frequency distribution of entrained individuals. Many values of d could be used, but the most common are average age and the constrained maximum (Steinbeck et al. 2007) age of entrained individuals. The difference between these two estimates can have profound effects on the estimate of impact (see below). Methods for estimating E_i and N_i can be found in Steinbeck et al. (2007).

Regardless of whether the species has a single spawning period per year or multiple overlapping spawning, the estimate of total larval entrainment mortality can be expressed as the following:

$$P_M = 1 - \sum_{i=1}^{n} f_i (1 - P_S P_{E\,i})^{d} \qquad (2)$$

Where:

P_{Ei} = estimate of the proportional entrainment for the i_{th} survey
P_S = ratio (sampled source water / SWB)
f_i = proportion of total annual larvae hatched during i_{th} survey
d =estimated number of days larvae vulnerable to entrainment

To establish independent survey estimates, it was assumed that each new survey represented a new, distinct cohort of larvae that was subject to entrainment. Each of the surveys was weighted using the proportion of the total population at risk during the i_{th} survey (f_i) calculated as follows:

$$f_i = \frac{N_i}{N_T} \qquad (3)$$

Where:

N_i = the source population spawned during the i^{th} survey
N_T = the sum of the N_i 's for the entire study period.

As noted above, the number of days that the larvae of a specific taxon were exposed to the mortality estimated by P_E, can be estimated using length data from a representative number of larvae from the entrainment samples. Typically, a point estimate of larval exposure has been used in the calculations (mean or maximum). These point estimates are constrained by using the values between the 1st and upper 99th percentiles of the length measurements for each entrained larval taxon. The constrained range is used to eliminate potential outlier measurements in the length data. Each measurement can then be divided by a species-specific estimate of the larval growth rate obtained from the scientific literature to produce an age frequency distribution. Maximum larval duration is calculated as the number of days between the 1st and 99th percentile. The second estimate uses an estimate of d calculated using the difference in length between the 1st percentile and the 50th percentile and is used to represent the mean number of days that the larvae were exposed to entrainment.

The term P_S represents the ratio of the area or volume of sampled source water to a larger area or volume containing the population of inference (Parker and DeMartini 1989). This allows for sampling of an area smaller than the likely source water body (SWB). If an estimate of the larval population in the larger area is available, the value of P_S can be computed directly.

There are two extreme versions of estimation of the SWB. These are noted for simplicity – the actual estimation is often more complex (Steinbeck et al. 2007). When an intake is withdrawing water exclusively from a contained water body, such as an estuary, the assumed SWB is often that water body for all species entrained. Note that even in these cases, there is often an addition to the SWB that represents tidal flux. For intakes withdrawing water from the open ocean, SWB is calculated separately for each assessed species. This calculation is based on the value of d and an estimate of net current velocity over the period of larval vulnerability. Hence P_S is then calculated as:

$$P_S = \frac{L_G}{L_P} \qquad (4)$$

Where:

L_G = length of sampling area

L_P = length of alongshore current displacement based on the period (d) of larval vulnerability for a taxon

Estimation of Area of Production Foregone and Consideration of Error in its Estimation

For a more detailed treatment of this topic see Strange et al. (2004) and Steinbeck et al. (2007). One problem associated with the use of ETM approaches is in the estimation of impact and potential mitigation opportunities. This is because the currency of ETM is proportional mortality (P_M), which is not an intuitive currency for impact assessment. Calculation of the area of production foregone (APF) is one approach for estimating impact and for giving guidance to compensation strategies because it yields the amount of habitat that would need to be replaced to compensate for the larval production lost due to entrainment.

Area of Production Foregone models can be used to understand the scale of loss resulting from entrainment and the extent of mitigation that could yield compensation for the loss. The basis of APF calculations with respect to entrainment rests on the assumptions that (1) P_M information collected on a group of species having varied life history characteristics can be used to estimate to impact to all entrained species and, (2) the currency of APF (habitat acreage) is useful in understanding both direct and indirect impacts resulting from entrainment, which is essential for understanding the extent of compensation required to offset the loss.

Because APF considers taxa to be simply independent replicates useful for calculating the expected impact, the choice of taxa for analysis may differ from Habitat Replacement Cost (HRC) assessments (Steinbeck et al. 2007). For APF, the concern is that each taxon is representative of others that were either unsampled (most species including invertebrates, plants and holoplankton) or not assessed for impact (most fish species, see Figure 1). The core assumption of APF with respect to estimating impact is that the average loss across assessed taxa is the single best point estimator of the loss across all entrained organisms. This fundamental statistical-philosophic assumption of APF addresses one of the most problematic issues in impact estimation: the typical inability to estimate impact for unevaluated taxa.
The calculation of APF is quite simple mathematically and in concept. Conceptually, it is an estimate of the area of habitat that would be required to replace all resources affected by the impact. Hence, for entrainment, it can be considered to be the area of habitat that would have to be added to replace lost larval resources. As an example, assume that for gobies the estimate was that 11% of larvae at risk in a 2000-acre estuary were lost to entrainment. The estimate of APF then would simply be 2,000 acres (the Source Water Body = SWB) x 11% (P_M) or 220 acres. Therefore the creation of 220 acres of new estuarine habitat would compensate for the losses of goby larvae due to entrainment. This does not mean that all biological resources were lost from an area of 220 acres, which is a common misunderstanding.

Instead it means that if 220 acres of new habitat were created then losses to gobies would be compensated for.

Mathematically then APF is the product of P_M and SWB. This calculation is done separately for each species i.

$$APF_i = P_{M_i}(SWB_i) \tag{5}$$

Clearly the goal should not be to assess impacts to individual species. Rather it should be to estimate all direct and indirect impacts to the system and to provide guidance as to the mitigation that would be compensatory. Indeed one criticism of many assessment methodologies (e.g. Habitat Equivalency Analysis = HEA) is that there is a focus on only a limited number of taxa (Figure 1) of all that are directly affected by entrainment and that there is also no provision for estimation of indirect impacts (often food web considerations). APF, as discussed, addresses this concern by expressing impact in terms of habitat and assuming that indirect impacts are mitigated for by the complete compensation of all directly lost resources. The idea is that the addition of the right amount of habitat would lead to compensatory production of larvae and would also compensate for indirect effects resulting from the larval losses. For example, if one indirect consequence of larval losses was the loss of a food resource for seabirds, the replacement of those lost larvae should mitigate the impact to seabirds. Hence the task is to determine the right amount of habitat.

The most obvious approach, as noted, and one that is consistent with the underlying assumptions of APF is to use species specific APF values to calculate a point estimate of overall effect. The main assumptions of this approach are:

1) Species specific APF values represent random samples from a population of APF values (the family of all possible species specific APF values)
2) Each species specific APF is the mean value of a series of samples and hence has associated measurement error.

Based on these assumptions, the mean (across species) should represent the single best estimate of the impact due to entrainment.

$$\overline{APF} = \sum_{i=1}^{n} APF_i \tag{6}$$

Because species in APF are simply independent replicates that yield a mean loss rate, habitat restored or created should not be directed by species. Instead the habitat monetized or created should represent the habitat for the populations at risk. That is, if the habitat in the SWB estuary was 60% subtidal eelgrass beds, 15% mudflats and 25% vegetated intertidal marsh, the same percentages should be maintained in the created habitat. Doing so would ensure that impacts on all affected species would be addressed. Probably the most controversial issue in APF assessment is how measurement error is accommodated, although such accommodation is part of national policy recommendations (EPA 2006). In most assessments, including Habitat Replacement Cost (HRC) (Strange et al. 2002), estimates of loss of taxa are implicitly considered to be without error. In APF, each species specific estimate is considered to be prone to (sometimes) massive error (indeed, estimates of confidence intervals in ETM calculations often cross through zero). Because of the uncertainty as to how error should be calculated and used in the calculation of estimates of compensatory mitigation, the goals of this project were to evaluate the effect of:

1) Incorporation of statistical uncertainty in estimation of APF – specifically how incorporation of error affects estimates of the likelihood that proposed mitigation acreage will be compensatory.
2) Sample size (number of species for which APF is assessed) on estimation of APF. Here the idea was to test how sensitive APF estimates are to sample size. The results of this portion of the study inform future sampling design.
3)

To address these goals, information (PM, the standard errors of PM, SWB) was collected from entrainment assessments at seven power plants (Figure 2). All assessments included empirical transport modeling and were done consistently with recent 316(b) determinations.
Sources of data are shown in Table 1 below. Note that for some power plants, data sources were corrected addendums to published studies.

Incorporation of statistical uncertainty in estimation of APF: Approach

The goal of this portion of the project was to estimate confidence limits for APF values. Such calculations would inform two questions (that mathematically are equivalent):

1) What is our confidence that the calculated APF accurately describes the impact?
2) What is the likelihood that restoration or creation of a given amount of area of habitat will lead to complete compensation for an impact?

This second question assumes that the measures used to compensate actually work. This assumption should not be left untested − instead there should always be an evaluation of the compensation measures.

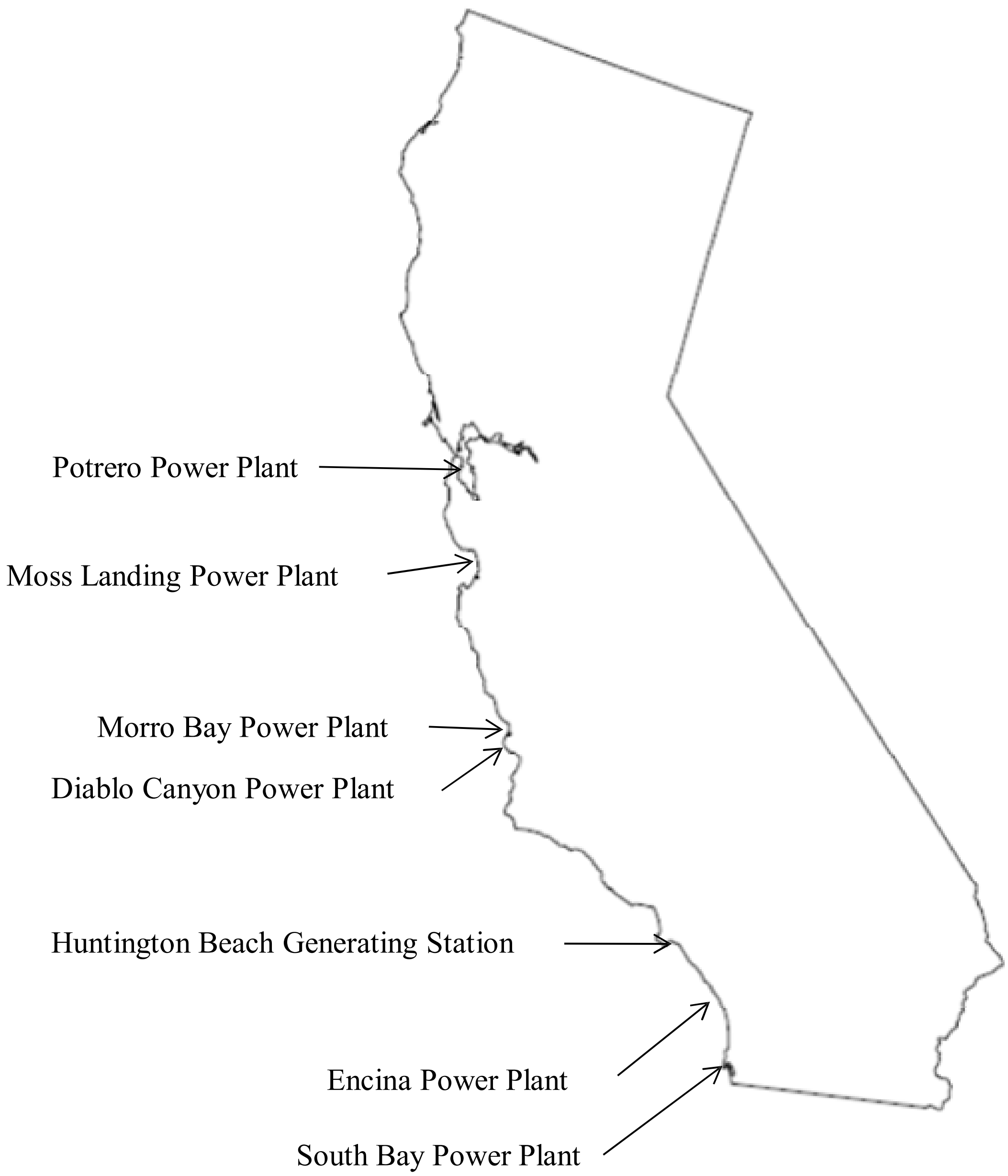

Figure 2. Location of power plants used in this study.

Power Plant	Data Source
South Bay	316(b) demonstration report to San Diego Regional Water Quality Control Board. May, 2004
Encina	316(b) demonstration report to San Diego Regional Water Quality Control Board. January 2008
Huntington Beach	AES Huntington Beach LLC Generating Station impingement and entrainment study. California Energy Commission. April 2005
Diablo Canyon	Addendum to 316(b) demonstration report. Document E9-055.0 to San Luis Obispo Regional Water Quality Control Board. March, 2000
Morro bay	Addendum to 316(b) demonstration report "Morro Bay Power Plant Modernization Project" to San Luis Obispo Regional Water Quality Control Board. July, 2001
Moss Landing	316(b) demonstration report to San Luis Obispo Regional Water Quality Control Board. April, 2000
Potrero	Final Staff Assessment: Potrero Power Plant Unit 7 Project. California Energy Commission. February 2002.

Table 1. Sources of data used in this study.

Two approaches were used to address these questions. First, based on the idea that species specific APF values are random samples from a distribution of values, confidence limits (or intervals) can be calculated using traditional parametric approaches or using resampling methods. There are substantial concerns about the use of parametric approaches (MacKinnon et al. 2004) when the underlying shape of the distribution in question is unknown or known and non-normal. APF values are synthetic not directly measured terms, and even the theoretical shape of the distribution of such values is unknown, hence both parametric and resampling methods were used and compared.

For each (treatment) combination of Power Plant, sample year, larval duration (mean or maximum period of vulnerability) and habitat (open coast or estuarine), $\overline{APF}$ (equation 6) and the standard error of APF (SE_{APF}) was calculated. These were used to generate confidence values based on a normal inverse function (Z inverse).

Generation of confidence limits for the same combinations was also calculated using resampling methods (Simon 1997). Resampling was performed with replacement and a series of 1000 means were generated for each treatment combination. Confidence limits (1, 5, 10, 20, 25, 50, 75, 80, 90, 95, 99) were determined based on the distribution of resampled means. As a reminder, the value at the 50th percentile should approximate the arithmetic mean.

Results from the two methods were compared using ordinary least squares regression for area estimated using confidence values ranging from the 50th to 99th percentiles (50, 75, 80, 90, 95, 99). The lower values (confidence values <50th percentile) were not used as they are inversely symmetric to higher values and would inflate replication.

The second approach was based on the standard errors calculated for each species P_M. See Appendix A. By assuming that the SWB was measured without error (which is probably ok for estuarine species and not ok for coastal species), confidence values for APF could be generated from the product of $P_{M(CV)}$ and

SWB, where $P_{M(CV)}$ is the P_M at a given confidence value. The underlying assumption here was that species specific APF values reflect the impact to that species and are not simply a sample from a distribution of independent measurements of the overall impact. The logic of this approach then is that the impact and confidence interval is species specific and that the net effect should reflect that logic. For example, the mean value of the 80th percentile could be calculated across species for South Bay, estuarine habitat, year one, maximum larval duration. Because parametric and resampling methodologies yielded the same results in the calculations discussed above, only the confidence limits based on the normal distribution were used. Mathematically then for any given confidence value the resulting APF would be:

$$\overline{APF_{CV}} = \sum_{i=1}^{n} APF_{CVi} \qquad (7)$$

Where:

$\overline{APF_{CV}}$ = Mean APF value across species for a given confidence value

APF_{CVi} = APF value for species i for a given confidence value

Incorporation of statistical uncertainty in estimation of APF: Results

Parametric and resampling estimation of area corresponding to similar confidence levels produced very similar results; the equation of the line comparing the two has a slope of 1 and an r^2 of .999. The results for each combination of Power Plant, sample year, larval duration (mean or maximum period of vulnerability) and habitat (open coast or estuarine) are shown in the series of Figures 1a – 1g in Appendix B. While the increase in area varied with each treatment combination, increasing likelihood of compensation resulted in an (exponential) increase in the APF estimate (Figure 3).

Using species specific confidence levels produced dramatically greater number of acres than was found using the approach using species specific APF values as replicates (Figures 2a-2g in Appendix B).

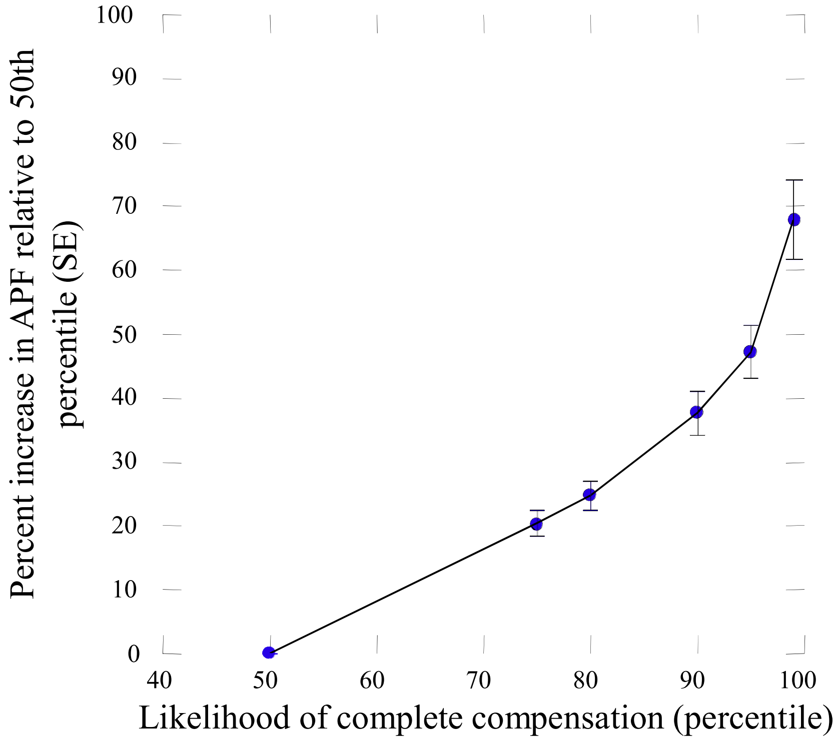

Figure 3. Effect of increasing likelihood of complete compensation on percent increase in APF.

The effect of sample size (number of species for which APF is assessed) on estimation of APF: Approach

Data from Diablo Canyon, in year one of the study, using maximum larval duration was used to assess the effect of replication on estimation of the confidence values for APF. For this treatment combination, P_M and SWB were originally calculated for 12 species and the corresponding APF values were determined as a result of this project (Appendix A). These 12 APF values were subjected to resampling in lots of 12, 11, 10, 9, 8, 7, 6, 5, 4, 3 replicates. During each run of a given level of replication, 1000 means were generated and the distribution of those means was used to determine APF values for a series of confidence values (50, 75,80, 90, 95, 99th percentile).

The effect of sample size (number of species for which APF is assessed) on estimation of APF: Results

The number of species sampled (level of replication) had a huge effect on the area required to attain a given confidence level for all levels above 50%, which is the mean (Figure 4). Using the 80% confidence level as an example, the estimated APF ranged from 3000 hectares (at 3 replicate species) to 2450 hectares (12 replicate species). Using the same line (80th percentile), one can also see that relative to the mean (50th percentile), increasing replication from 3 to 12 species decreased the area required by about 30%.

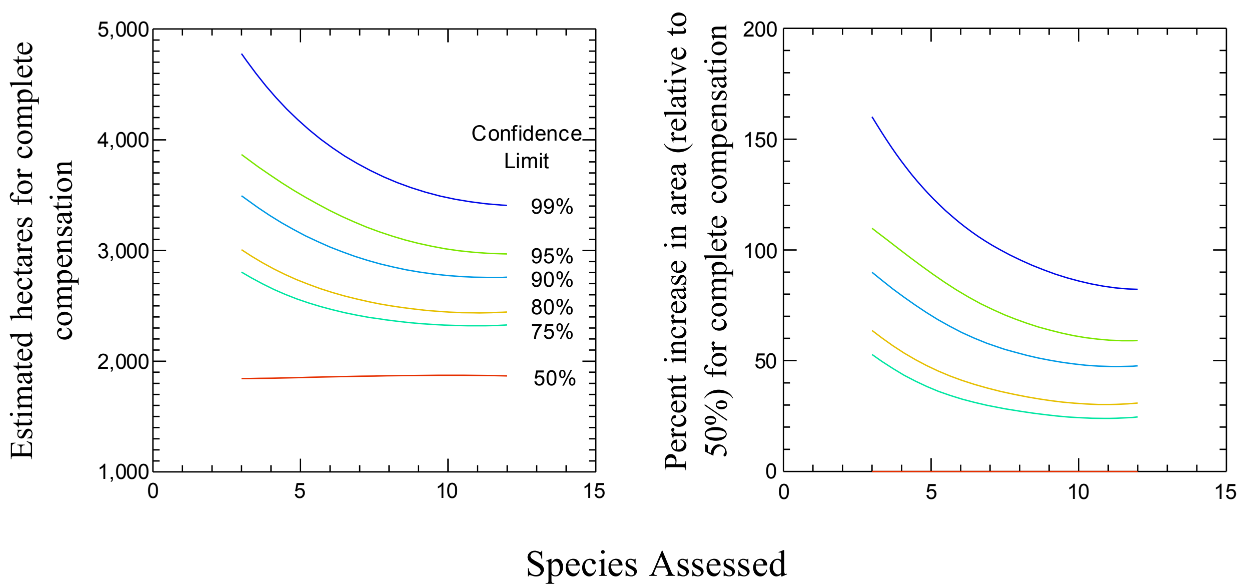

Figure 4: Effect of replication of species assessed on estimated APF.

Synthesis

Area of production foregone (APF, often also called Habitat Production Foregone; HPF) has been used in most if not all recent power plant entrainment studies in the state of California that adhered to 316(b) type assessment methods. In addition it has also been used to assess entrainment in impact studies of desalinization facilities that are co-located with power plants

(Poseidon Resources [Channelside] 2008). Far from being an unchanging approach, it has evolved considerably over the last ten years. While the derived ETM/APF approach was first used in the 316(b) assessment at Diablo Canyon (2000), the first finalized study utilizing APF was that at Moss Landing (Steinbeck et al. 2007, Moss Landing 316(b), 2000). In that assessment ETM was utilized but APF was calculated based on mean larval duration of vulnerability. In subsequent determinations at other power plants, either both mean and maximum larval durations or only maximum values were used for assessment (Appendix A). This evolution reflected the attained understanding that the true period of larval vulnerability was better estimated using maximum larval duration. Other changes in the use of APF have come in the way the SWB has been calculated for both open coast (see Diablo Canyon 316(b) and the use of an offshore gradient approach) and estuarine habitats (see Morro Bay 316(b) and the use of tidal flux). The point is that the use of APF is evolving as we understand both its constraints and the assumptions (often implicit) of the mathematics underlying its calculation.

There has also been an evolution in thinking about the most problematic general issue in impact assessment - how to account for error? In particular, an essential question is how to use confidence values to give a context to assessment of impact. In the specific case of APF, the general approach has been to use species specific APF values in the calculation of the mean APF, which is then used both as a currency of impact and also as a target value for compensatory mitigation. It is rarely if ever noted that the mean APF (from sample APF values) is (making assumption of normality) also the 50% confidence limit for the distribution of possible true population means. In non-statistical terms, this means that the true impact will be greater than or equal to the mean APF 50% of the time and equivalently that the likelihood of complete compensation from the creation of restoration of area equal to the mean APF is 50%. Two important points need to be made here. First, this argument is one about the amount of area; there is the assumption that the restoration or habitat creation actually works as designed. Second, probabilistically, half the possible population means (true impacts) are above and half below the 50th percentile (mean APF). Hence, if the true impact is above the mean APF there will be incomplete compensation, but not none at all. This last point seems obvious, but given the continued misinterpretation about APF (the wrong idea that APF means that existing habitat has been lost), it is important to be clear about the meaning of mathematical / statistical concepts.

Incorporation of confidence levels could have a profound effect on the estimation of habitat (restored or created) required to attain complete compensation for an impact. Ultimately, the confidence level desired is a policy decision that should balance the cost (financial and to society) of underestimating the area required for compensation with the cost (primarily financial) to the permittee or applicant. The results of this study provide guidance to the increase in area associated with increasing confidence that the effort will result in complete compensation. This is in turn should give insight into the trade off in costs noted above.

Conclusions

Parametric and resampling methods yield similar confidence values. Here single species APF values were considered to be independent replicate samples of the overall impact. In every combination of power plant, sample year, larval duration and habitat confidence levels (shown as likelihoods) calculated using parametric and resampling methods yielded similar results (See Appendix B). More importantly, increasing likelihoods of complete compensation were associated with increasing area of restoration or creation. The increase in area varied with treatment combination but the overall relationship revealed an exponential pattern (Figure 3). Increasing the likelihood from 50% to 95%, which is the traditional value used in inferential statistics, increased the required area about 50% (across all studies). Using a more conservative increase from 50-80% produced, on average, an increase in area of about 25%. Assuming a direct relationship between area and cost, this means that the cost of increasing the likelihood of attaining full compensation from 50 to 80% would add an additional 25% to the cost of the mitigation project.

The results of this part of the study can be used to inform other questions. As discussed, early ETM studies used the mean larval duration as the estimate of the period of larval vulnerability instead of maximum larval duration, which is currently used. The ETM study conducted at Diablo Canyon Power Plant was the most thorough investigation of entrainment impacts on the west coast and allows for a robust comparison of the effect of assumed period of larval vulnerability from mean to maximum larval duration. This change fundamentally affected estimated APF values (Figure 5). At all likelihood (of complete compensation) values greater than 50%, the area needed, under the assumption of maximum larval duration, was more than twice that needed under the assumption of mean larval duration.

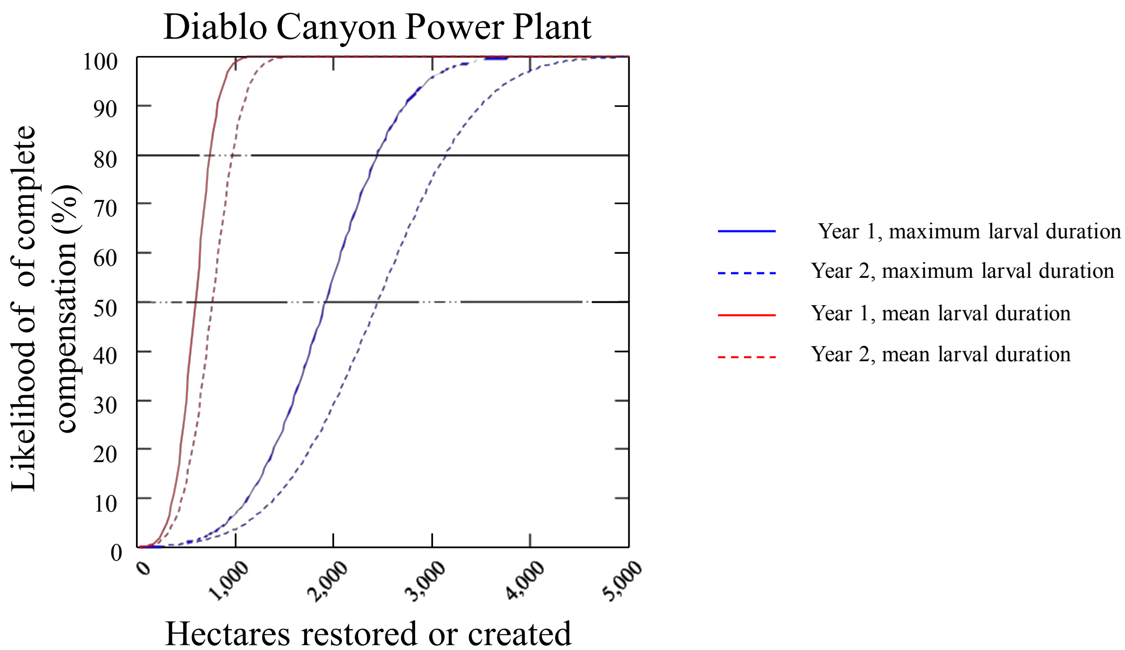

Figure 5: Probability of complete compensation as a function of area restored or created. APF estimates (using parametric approach) based from two years of sampling and two methods of estimating period of larval vulnerability

Species specific confidence values yield APF estimates much larger than those generated under the assumption that species specific APF values are replicate samples. Because standard errors were calculated for each P_M value, it was possible to calculate confidence values for each species. Using the logic discussed above and equation 7, species specific and mean confidence values were calculated. The impact of species specific estimation was large (Appendix B: Figures 2a – 2g). In all cases where the likelihood of complete compensation was greater than 50% this method yielded larger areas than that using mean confidence values; often there was a doubling of area.

The statistical-philosophical basis of this method of incorporation of measurement error is that the calculation of P_M and APF values for each species accurately describes (after error is accounted for) the impact to the species. Hence, APF values are not considered to be independent replicate samples of the overall impact of entrainment across all species be they assessed or not. Under this logic, the goal would be to ensure that the area restored or created was sufficient to compensate for the losses to each species at a given confidence level. While appealing, there are problems with this approach. First, measurement errors associated with P_M are often massive, and likely inappropriate for the task of generation of confidence values. Second, there is no provision for estimation of the impact for species not assessed (which are the vast majority of species). Third, and most fundamental, estimation of confidence values based on species specific error rates is counter to the logic of the calculation of mean APF. That is, the replication for the estimation of mean APF is the species specific APF values (not error rates), therefore the error must be based on the same replication (see Quinn and Keough 2003).

The number of species sampled dramatically affects estimation of APF (Figure 5). This clearly is not an unexpected result and is completely consistent with sampling theory (Quinn and Keough 2003, Zar 1996). Resampling the data for species sampled at Diablo Canyon, year 1, maximum larval duration showed that for all confidence levels above 50% the estimated area required to compensate for entrainment impact decreased as a function of number of species assessed. The lack of change for the 50% confidence limit is because the expected mean does not change as a function of sample size. Instead error changes, which affects the estimates of area at confidence limits different from 50%. Intuitively this is the result of the distribution of expected means broadening at low sample size. This points to an important policy implication. If policy mandates that the 50% confidence limit for the APF value (~mean) be used to assess impacts and as a measure of compensatory mitigation, sample size is theoretically unimportant, because the expected mean does not vary with number of species assessed. Note that the actual mean APF may vary across sample size. Indeed at smaller sample sizes there will be much more variability in the mean if sampled repeatedly. This would lead to a greater probability of under or over estimating the impact than would occur at higher sample size. By contrast to the situation where policy mandates use of the 50% confidence limit for APF, if policy or regulation requires incorporation of confidence values higher than 50% (e.g. Poseidon case where 80% level was used), then sample size becomes even more important. This is because the likely mitigation requirement will decrease with increasing sample size. The key implication of this result is that minimizing cost during sampling and assessment may be countered by the increased cost of habitat creation or restoration due to inadequate sampling.

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Appendix A
Data from Seven Power Plants

Table APA-1 Data from Seven Power Plants

Powerplant	Year	Habitat	Species	larval duration	Pm	Pm (SE)	offshore (km)	SWB (Hectares)	APF (Hectares)
South Bay	1	Estuarine	anchovies	maximum	0.1050	0.3132		3032.66	318.43
South Bay	1	Estuarine	CIQ goby complex	maximum	0.2150	0.4294		3032.66	652.02
South Bay	1	Estuarine	combtooth blennies	maximum	0.0310	0.1774		3032.66	94.01
South Bay	1	Estuarine	longjaw mudsucker	maximum	0.1710	0.3925		3032.66	518.59
South Bay	1	Estuarine	silversides	maximum	0.1460	0.3734		3032.66	442.77
South Bay	2	Estuarine	anchovies	maximum	0.0790	0.2814		3032.66	239.58
South Bay	2	Estuarine	CIQ goby complex	maximum	0.2670	0.4739		3032.66	809.72
South Bay	2	Estuarine	combtooth blennies	maximum	0.0340	0.1849		3032.66	103.11
South Bay	2	Estuarine	longjaw mudsucker	maximum	0.5020	0.5368		3032.66	1522.40
South Bay	2	Estuarine	silversides	maximum	0.1490	0.4121		3032.66	451.87
Encina	1	Coastal	California halibut	maximum	0.0015	0.0024	3	11117.30	16.79
Encina	1	Coastal	northern anchovy	maximum	0.0017	0.0026	3	6299.80	10.39
Encina	1	Coastal	queenfish	maximum	0.0037	0.0049	3	8217.14	29.99
Encina	1	Coastal	spotfin croaker	maximum	0.0063	0.0153	3	5558.65	35.24
Encina	1	Coastal	white croaker	maximum	0.0014	0.0028	3	13499.58	18.63
Encina	1	Estuarine	blennies	maximum	0.0864	0.1347		123.00	10.55
Encina	1	Estuarine	Garibaldi	maximum	0.0648	0.1397		123.00	7.92
Encina	1	Estuarine	gobies	maximum	0.2160	0.3084		123.00	26.39
Huntington Beach	1	Coastal	black croaker	maximum	0.0010	0.0007	4.44	8620.58	8.62
Huntington Beach	1	Coastal	blennies	maximum	0.0080	0.0054	4.44	5687.81	45.50
Huntington Beach	1	Coastal	California halibut	maximum	0.0030	0.0020	4.44	13730.72	41.19
Huntington Beach	1	Coastal	diamond turbot	maximum	0.0060	0.0040	4.44	7509.68	45.06
Huntington Beach	1	Coastal	northern anchovy	maximum	0.0120	0.0080	4.44	31993.92	383.93
Huntington Beach	1	Coastal	queenfish	maximum	0.0060	0.0040	4.44	37726.16	226.36
Huntington Beach	1	Coastal	rock crab megalops	maximum	0.0110	0.0074	4.44	11775.54	129.53
Huntington Beach	1	Coastal	spotfin croaker	maximum	0.0030	0.0020	4.44	7509.68	22.53
Huntington Beach	1	Coastal	white croaker	maximum	0.0070	0.0047	4.44	21240.41	148.68
Diablo Canyon	1	Coastal	blackeye goby	maximum	0.1151	0.0832	3	8560.80	985.69
Diablo Canyon	1	Coastal	blue rockfish complex	maximum	0.0041	0.0479	3	14146.20	58.14
Diablo Canyon	1	Coastal	cabezon	maximum	0.0111	0.1371	3	12058.20	134.21
Diablo Canyon	1	Coastal	California halibut	maximum	0.0047	0.0901	3	21088.80	98.27
Diablo Canyon	1	Coastal	clinid kelpfishes	maximum	0.1894	0.1218	3	29962.80	5674.65
Diablo Canyon	1	Coastal	KGB rockfishes	maximum	0.0388	0.0495	3	20149.20	781.59
Diablo Canyon	1	Coastal	monkeyface prickleback	maximum	0.1377	0.0726	3	31894.20	4390.56
Diablo Canyon	1	Coastal	painted greenling	maximum	0.0629	0.0920	3	26465.40	1664.67
Diablo Canyon	1	Coastal	sanddabs	maximum	0.0103	0.0583	3	12371.40	127.67
Diablo Canyon	1	Coastal	smoothhead sculpin	maximum	0.1139	0.0843	3	36122.40	4115.06
Diablo Canyon	1	Coastal	snubnose sculpin	maximum	0.1494	0.0967	3	31737.60	4741.91
Diablo Canyon	1	Coastal	white croaker	maximum	0.0070	0.0368	3	23437.80	163.60
Diablo Canyon	1	Coastal	blackeye goby	mean	0.0885	0.0774	3	4802.40	425.16
Diablo Canyon	1	Coastal	blue rockfish complex	mean	0.0028	0.0479	3	9657.00	26.75
Diablo Canyon	1	Coastal	cabezon	mean	0.0068	0.1373	3	10179.00	69.12
Diablo Canyon	1	Coastal	California halibut	mean	0.0029	0.0902	3	9291.60	26.95
Diablo Canyon	1	Coastal	clinid kelpfishes	mean	0.1498	0.1248	3	11745.00	1759.40
Diablo Canyon	1	Coastal	KGB rockfishes	mean	0.0242	0.0442	3	12423.60	300.53
Diablo Canyon	1	Coastal	monkeyface prickleback	mean	0.1056	0.0710	3	12319.20	1300.29
Diablo Canyon	1	Coastal	painted greenling	mean	0.0478	0.0920	3	14616.00	698.64
Diablo Canyon	1	Coastal	sanddabs	mean	0.0088	0.0581	3	9239.40	81.49
Diablo Canyon	1	Coastal	smoothhead sculpin	mean	0.0862	0.0767	3	12580.20	1084.16
Diablo Canyon	1	Coastal	snubnose sculpin	mean	0.1045	0.0961	3	12423.60	1297.89
Diablo Canyon	1	Coastal	white croaker	mean	0.0047	0.0368	3	11170.80	52.84
Diablo Canyon	2	Coastal	blackeye goby	maximum	0.0652	0.0576	3	6577.20	429.03
Diablo Canyon	2	Coastal	blue rockfish complex	maximum	0.0277	0.0372	3	15816.60	437.80
Diablo Canyon	2	Coastal	cabezon	maximum	0.0152	0.0651	3	9970.20	151.25
Diablo Canyon	2	Coastal	California halibut	maximum	0.0712	0.0793	3	16547.40	1177.84
Diablo Canyon	2	Coastal	clinid kelpfishes	maximum	0.2497	0.1132	3	22863.60	5709.96
Diablo Canyon	2	Coastal	KGB rockfishes	maximum	0.0480	0.0793	3	22863.60	1098.37
Diablo Canyon	2	Coastal	monkeyface prickleback	maximum	0.1176	0.0894	3	31737.60	3731.39
Diablo Canyon	2	Coastal	painted greenling	maximum	0.0558	0.0666	3	23176.80	1293.96
Diablo Canyon	2	Coastal	sanddabs	maximum	0.0080	0.0749	3	14302.80	113.99
Diablo Canyon	2	Coastal	smoothhead sculpin	maximum	0.2257	0.1133	3	26569.80	5997.34
Diablo Canyon	2	Coastal	snubnose sculpin	maximum	0.3102	0.1383	3	27405.00	8500.48
Diablo Canyon	2	Coastal	white croaker	maximum	0.0347	0.0349	3	20358.00	707.03
Diablo Canyon	2	Coastal	blackeye goby	mean	0.0412	0.0445	3	4489.20	185.00
Diablo Canyon	2	Coastal	blue rockfish complex	mean	0.0293	0.0400	3	6942.60	203.21
Diablo Canyon	2	Coastal	cabezon	mean	0.0117	0.0650	3	6525.00	76.15
Diablo Canyon	2	Coastal	California halibut	mean	0.0606	0.0847	3	5637.60	341.69
Diablo Canyon	2	Coastal	clinid kelpfishes	mean	0.1797	0.1314	3	10022.40	1800.72
Diablo Canyon	2	Coastal	KGB rockfishes	mean	0.0472	0.0798	3	8769.60	413.49
Diablo Canyon	2	Coastal	monkeyface prickleback	mean	0.1153	0.1025	3	9135.00	1053.08
Diablo Canyon	2	Coastal	painted greenling	mean	0.0369	0.0632	3	14824.80	546.89
Diablo Canyon	2	Coastal	sanddabs	mean	0.0101	0.0751	3	7151.40	72.01
Diablo Canyon	2	Coastal	smoothhead sculpin	mean	0.1562	0.1303	3	10544.40	1647.14
Diablo Canyon	2	Coastal	snubnose sculpin	mean	0.1851	0.1091	3	14302.80	2647.59
Diablo Canyon	2	Coastal	white croaker	mean	0.0280	0.0364	3	8091.00	226.87

APA-1

Data from Seven Power Plants (cont.)

Powerplant	Year	Habitat	Species	larval duration	Pm	Pm (SE)	offshore (km)	SWB (Hectares)	APF (Hectares)
Morro Bay	1	Coastal	cabezon	mean	0.0249	0.5373	3	17151.30	427.07
Morro Bay	1	Coastal	KGB rockfishes	mean	0.0271	0.5733	3	15988.50	433.29
Morro Bay	1	Coastal	northern lampfish	mean	0.0253	0.8518	3	20930.40	529.54
Morro Bay	1	Coastal	Pacific staghorn sculpin	mean	0.0513	1.1220	3	45058.50	2311.50
Morro Bay	1	Coastal	white croaker	mean	0.0434	1.0526	3	20058.30	870.53
Morro Bay	1	Estuarine	combtooth blennies	maximum	0.7371	0.6012	3	930.58	685.93
Morro Bay	1	Estuarine	gobies	maximum	0.4333	0.5551	3	930.58	403.22
Morro Bay	1	Estuarine	jacksmelt	maximum	0.4392	0.5451	3	930.58	408.71
Morro Bay	1	Estuarine	Pacific herring	maximum	0.2544	0.4510	3	930.58	236.74
Morro Bay	1	Estuarine	shadow goby	maximum	0.0643	0.2625	3	930.58	59.84
Morro Bay	1	Estuarine	combtooth blennies	mean	0.4972	0.6114	3	930.58	462.68
Morro Bay	1	Estuarine	gobies	mean	0.1158	0.3357	3	930.58	107.76
Morro Bay	1	Estuarine	jacksmelt	mean	0.2172	0.4348	3	930.58	202.12
Morro Bay	1	Estuarine	Pacific herring	mean	0.1642	0.3927	3	930.58	152.80
Morro Bay	1	Estuarine	shadow goby	mean	0.0283	0.1923	3	930.58	26.34
Moss Landing	1	Estuarine	bay goby	mean	0.2144	0.0406		1213.80	260.26
Moss Landing	1	Estuarine	blackeye goby	mean	0.0749	0.0476		1213.80	90.89
Moss Landing	1	Estuarine	combtooth blennies	mean	0.1820	0.0786		1213.80	220.85
Moss Landing	1	Estuarine	gobies	mean	0.1069	0.0067		1213.80	129.76
Moss Landing	1	Estuarine	longjaw mudsucker	mean	0.0894	0.0216		1213.80	108.56
Moss Landing	1	Estuarine	Pacific herring	mean	0.1337	0.0168		1213.80	162.30
Moss Landing	1	Estuarine	Pacific staghorn sculpin	mean	0.1179	0.0198		1213.80	143.09
Moss Landing	1	Estuarine	white croaker	mean	0.1291	0.0242		1213.80	156.73
Potrero	1	Estuarine	bay goby	maximum	0.0025	0.0013		39670.22	99.57
Potrero	1	Estuarine	California halibut	maximum	0.0076	0.0066		39670.22	303.08
Potrero	1	Estuarine	gobies	maximum	0.0048	0.0017		39670.22	191.61
Potrero	1	Estuarine	northern anchovy	maximum	0.0029	0.0020		39670.22	115.44
Potrero	1	Estuarine	Pacific herring	maximum	0.0035	0.0104		39670.22	139.64
Potrero	1	Estuarine	white croaker	maximum	0.0049	0.0037		39670.22	195.57
Potrero	1	Estuarine	yellowfin goby	maximum	0.0017	0.0009		39670.22	67.44
Potrero	1	Estuarine	bay goby	mean	0.0011	0.0005		39670.22	44.43
Potrero	1	Estuarine	California halibut	mean	0.0024	0.0021		39670.22	95.21
Potrero	1	Estuarine	gobies	mean	0.0011	0.0004		39670.22	41.65
Potrero	1	Estuarine	northern anchovy	mean	0.0005	0.0004		39670.22	21.03
Potrero	1	Estuarine	Pacific herring	mean	0.0011	0.0032		39670.22	42.45
Potrero	1	Estuarine	white croaker	mean	0.0011	0.0008		39670.22	44.03
Potrero	1	Estuarine	yellowfin goby	mean	0.0009	0.0005		39670.22	36.50

APPENDIX B
Power Plant Specific Figures

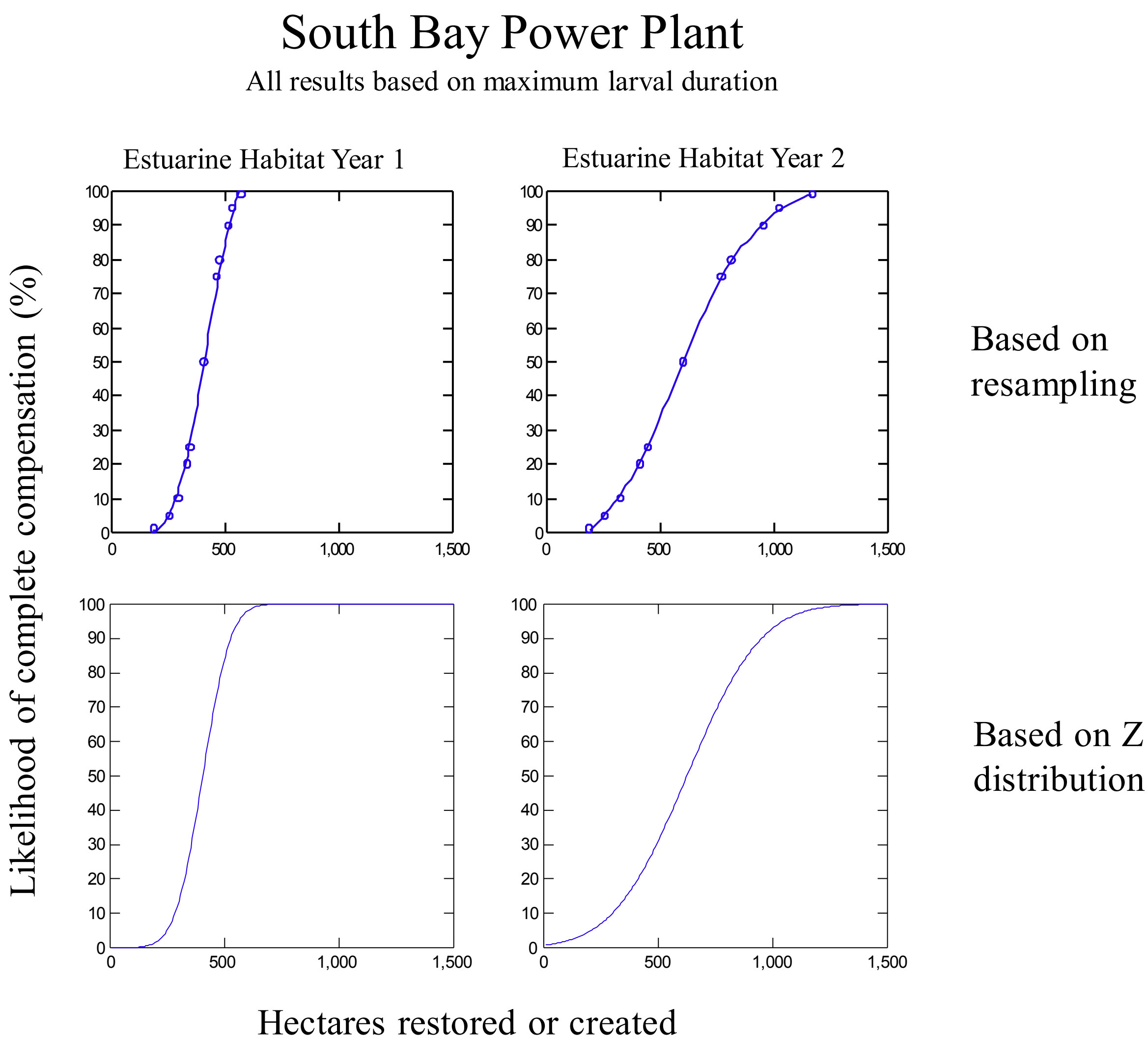

Figure 1a. Hectares restored or created at South Bay Power Plant.

APA-1

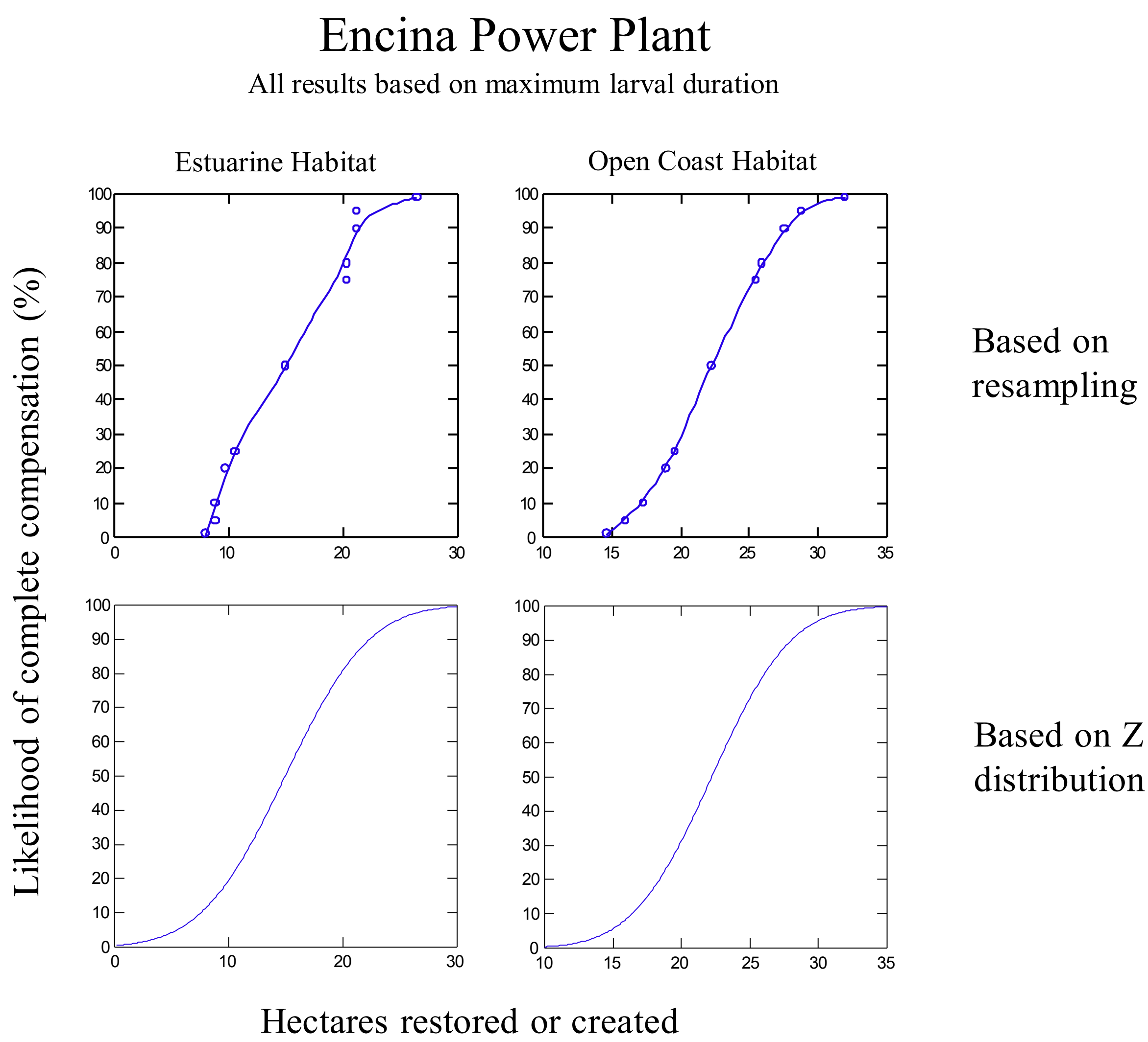

Figure 1b. Hectares restored or created at Encina Power Plan**t.**

APA-1

Huntington Beach Generating Station

All results based on maximum larval duration

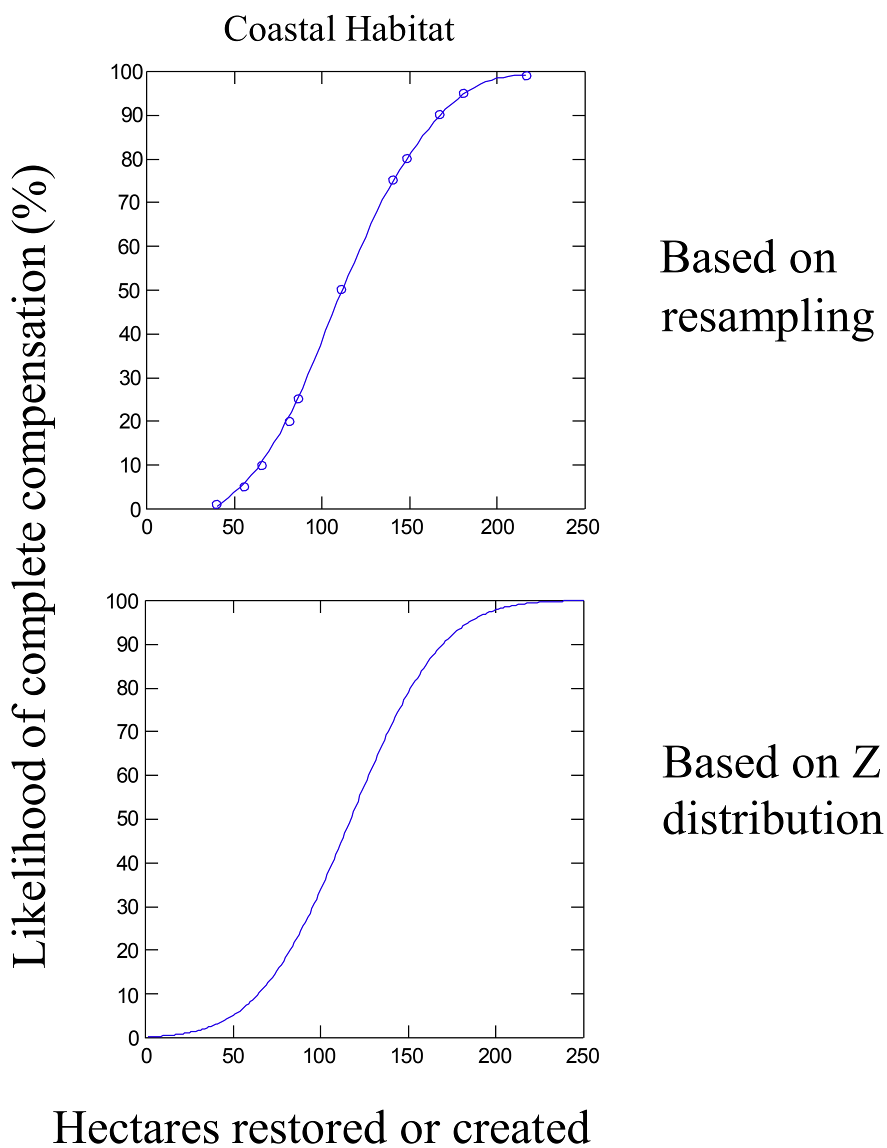

Figure 1c. Hectares restored or created at Huntington Beach Generating Station.

APA-1

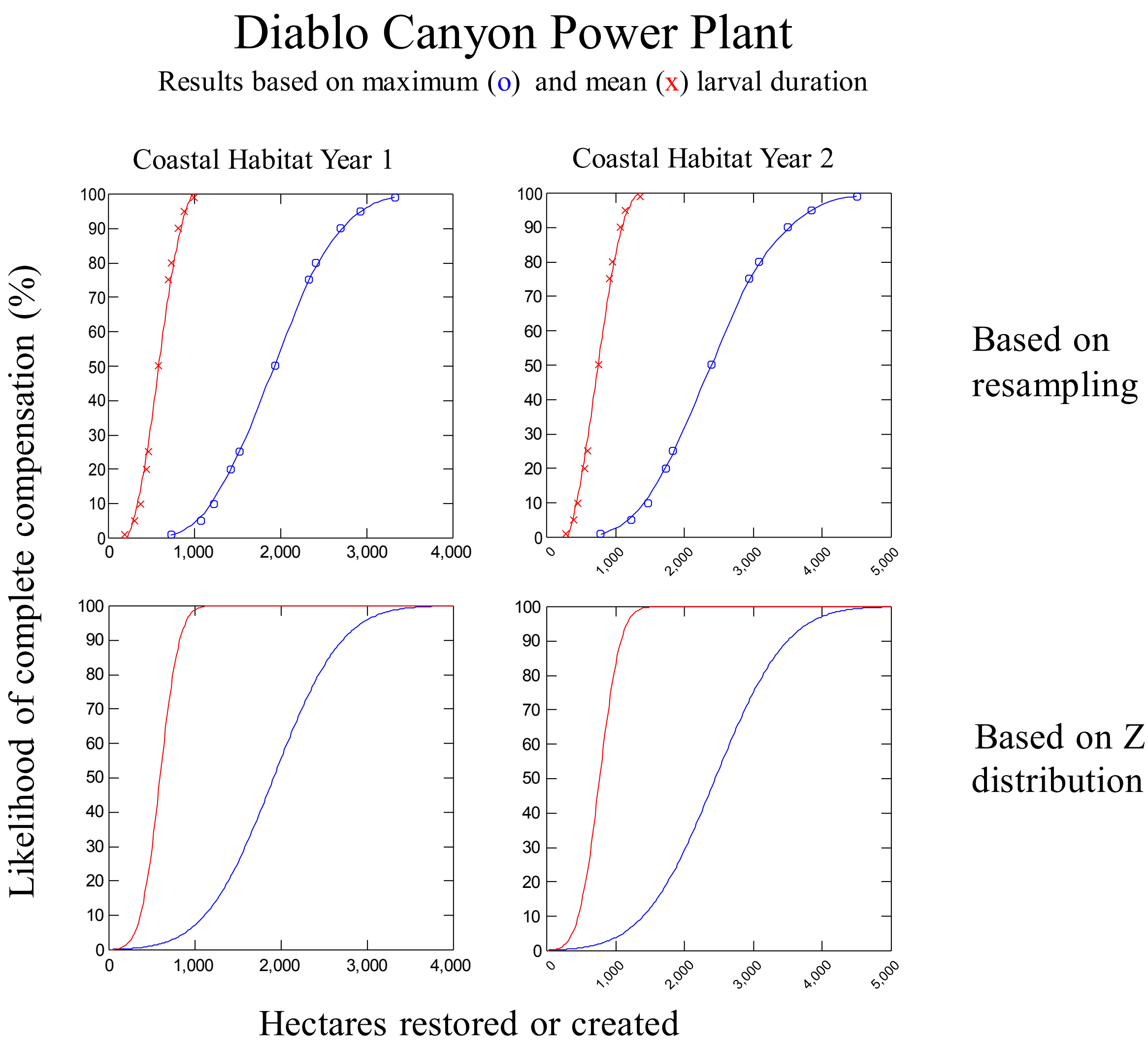

Figure 1d. Hectares restored or created at Diablo Canyon Power Plant.

APA-1

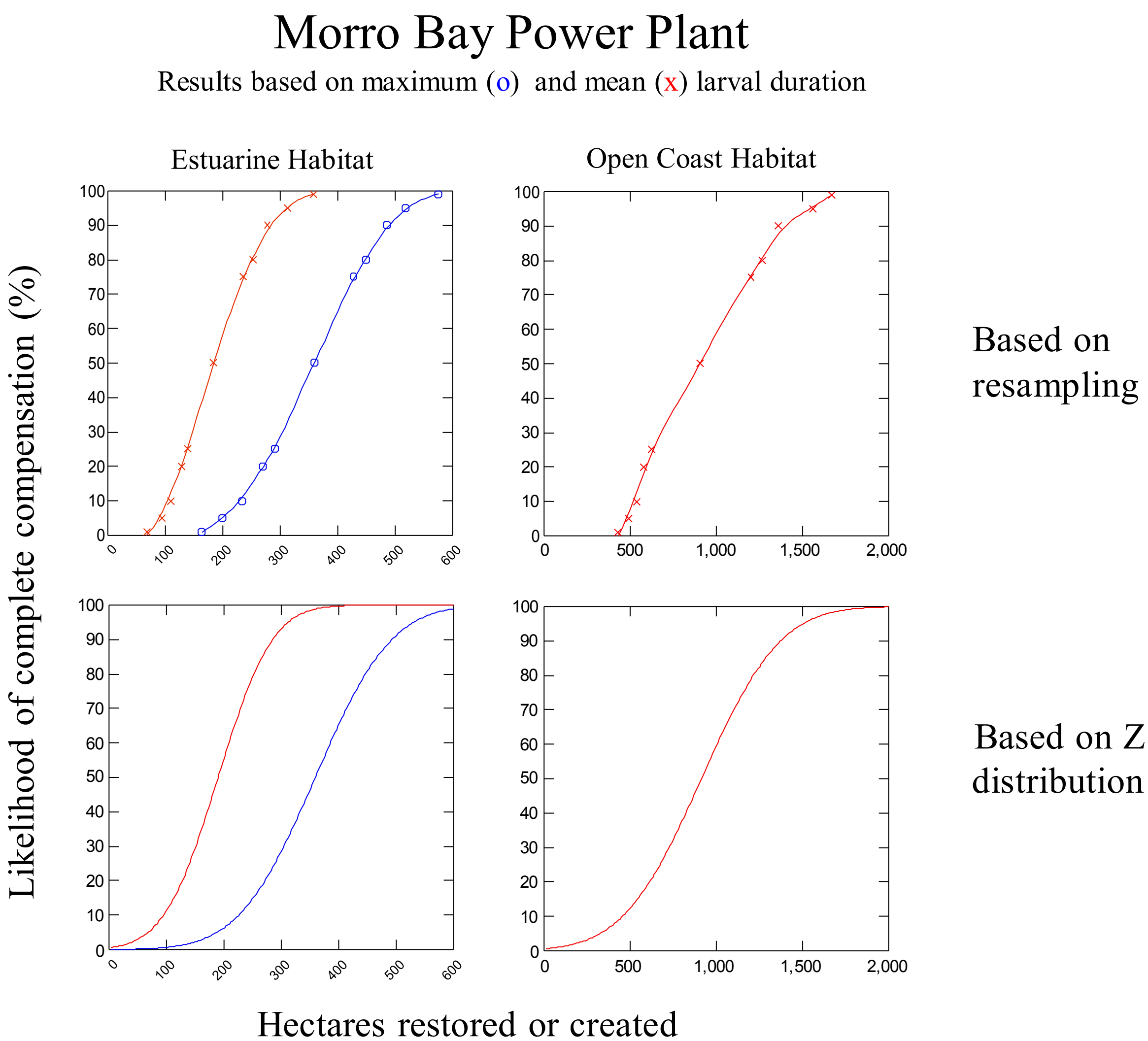

Figure 1e. Hectares restored or created at Morro Bay Power Plant.

APA-1

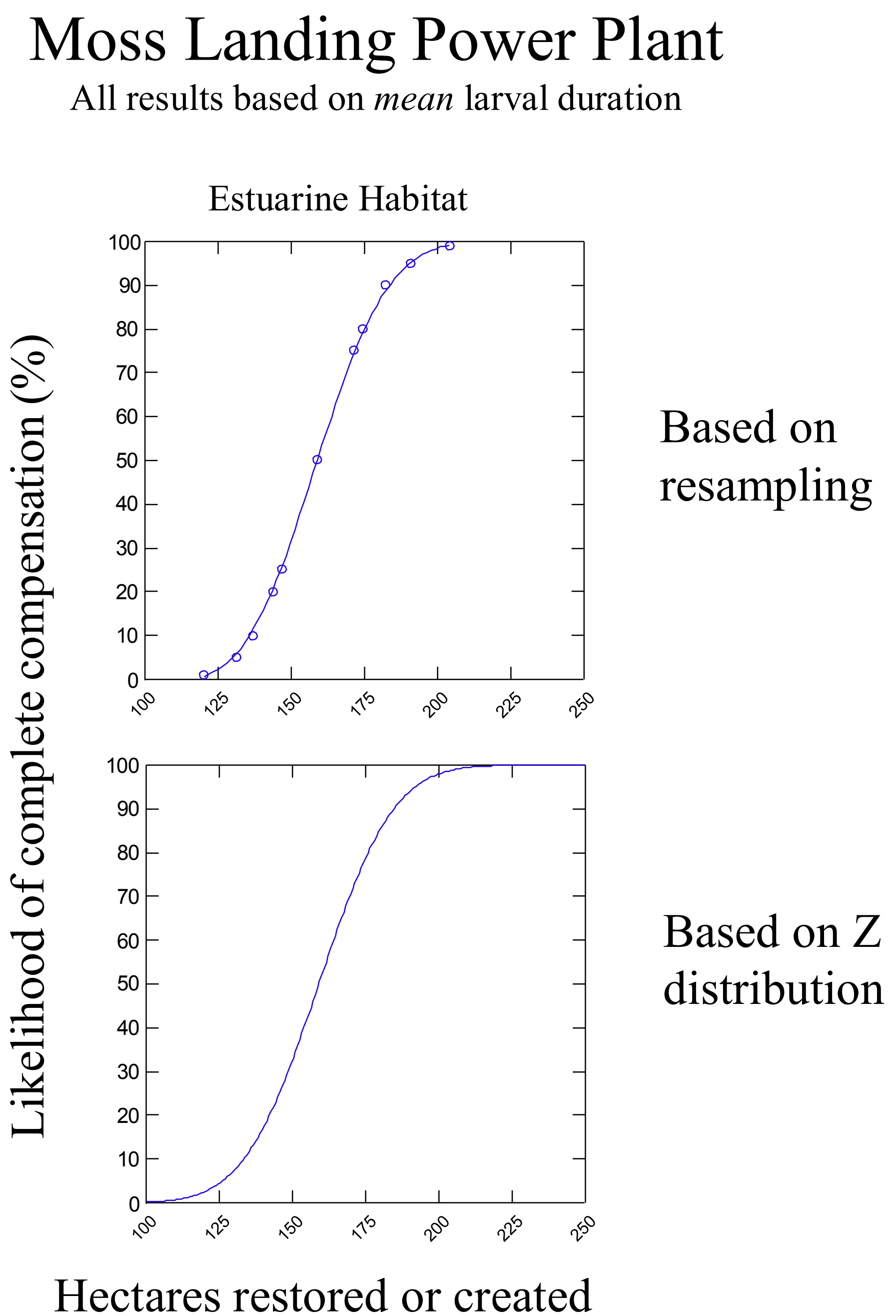

Figure 1f. Hectares restored or created at Moss Landing Power Plant.

APA-1

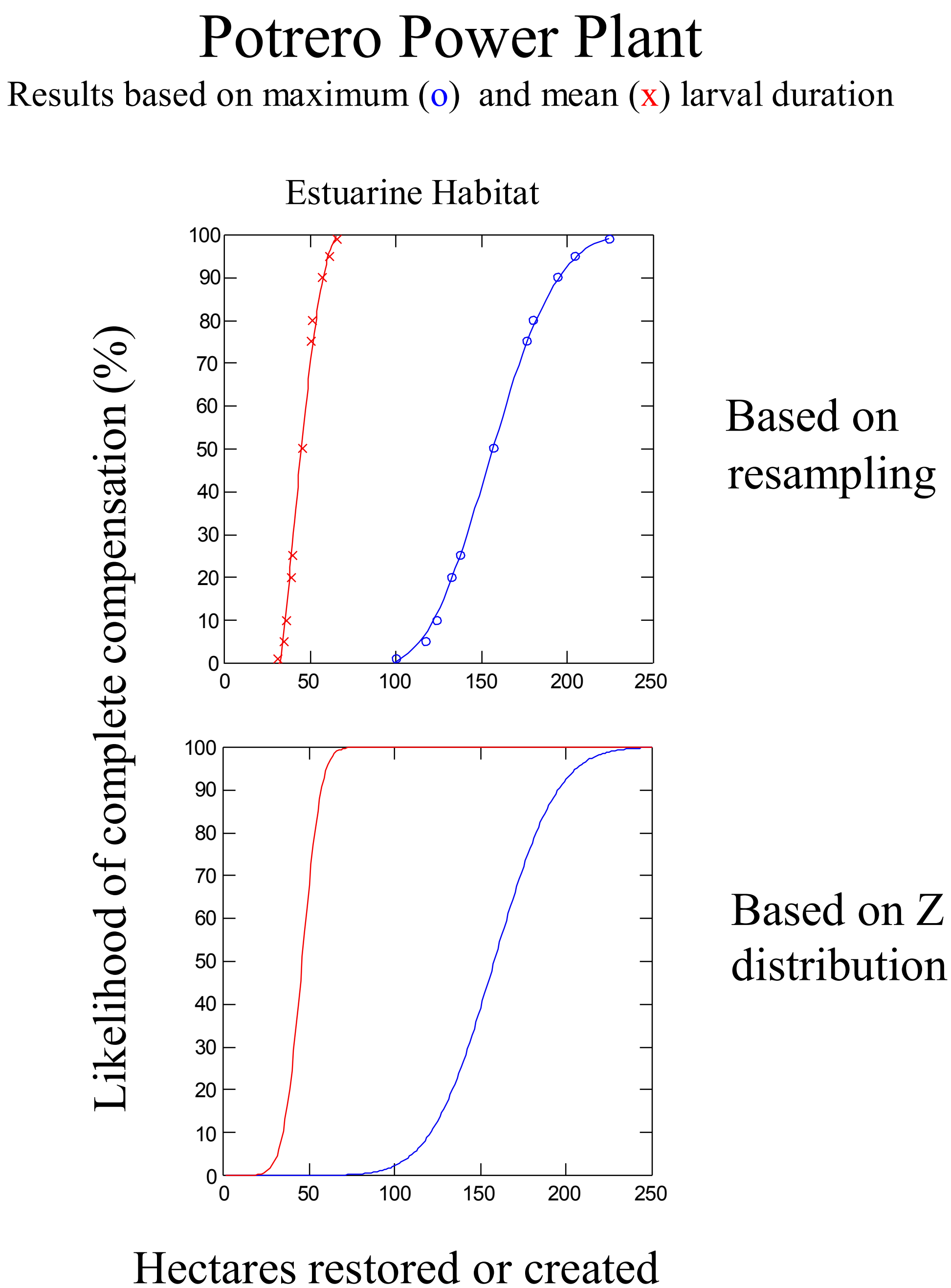

Figure 1g. Hectares restored or created Potrero Power Plant

APA-1

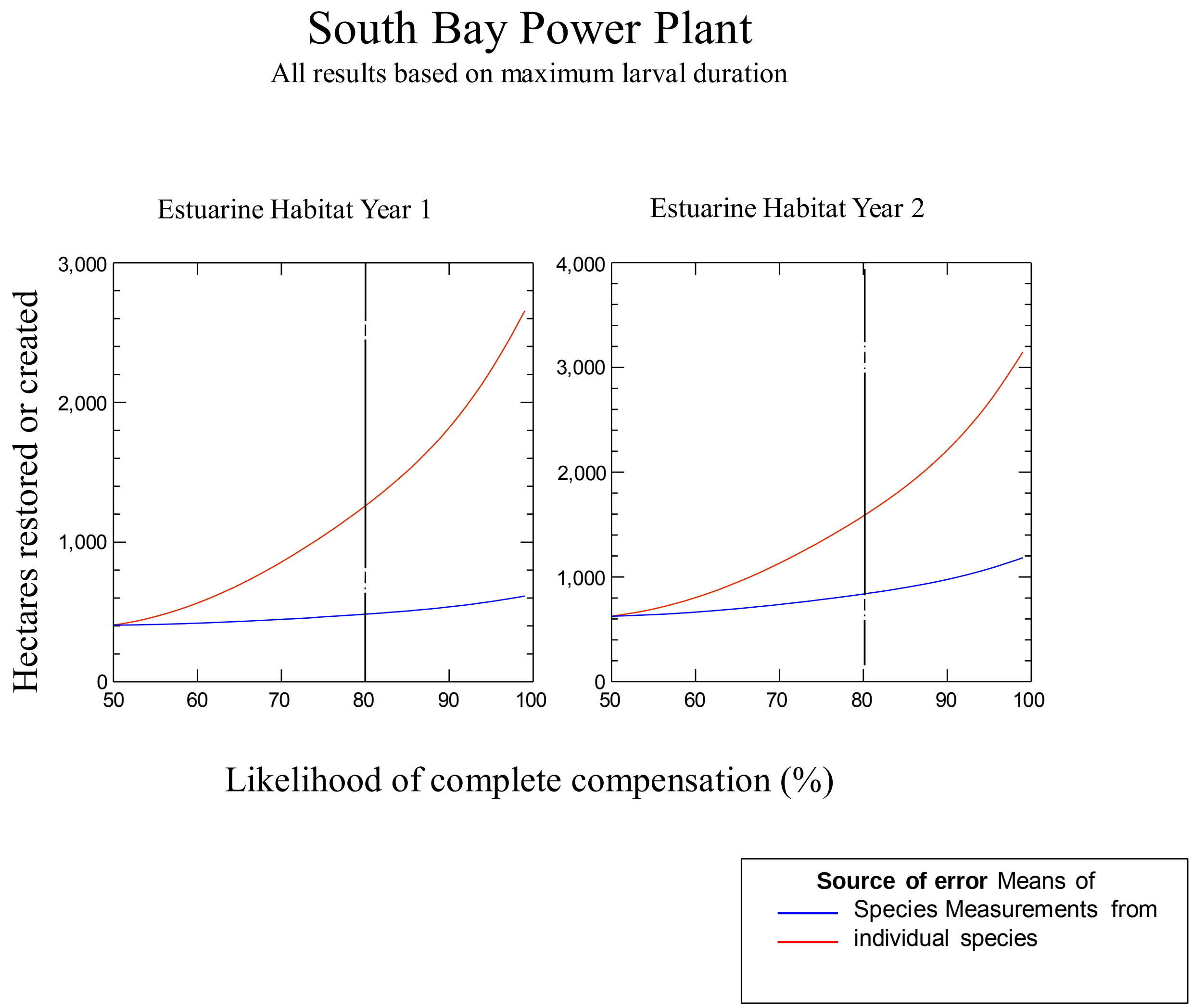

Figure 2a. Likelihood of complete compensation (%) South Bay Power Plant.

APA-1

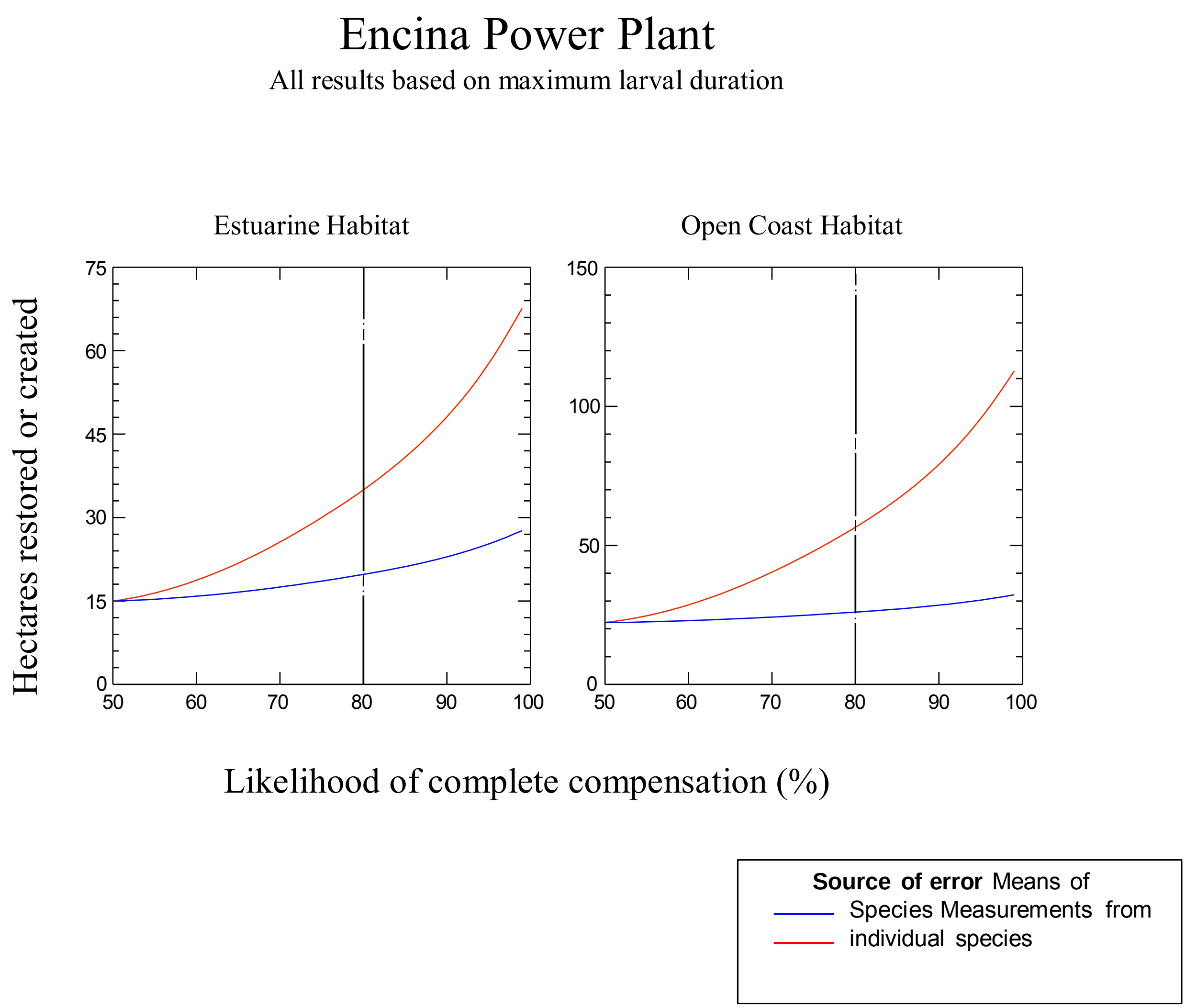

Figure 2b. Likelihood of complete compensation (%) Encina Power Plant.

APA-1

Huntington Beach Generating Station

All results based on maximum larval duration

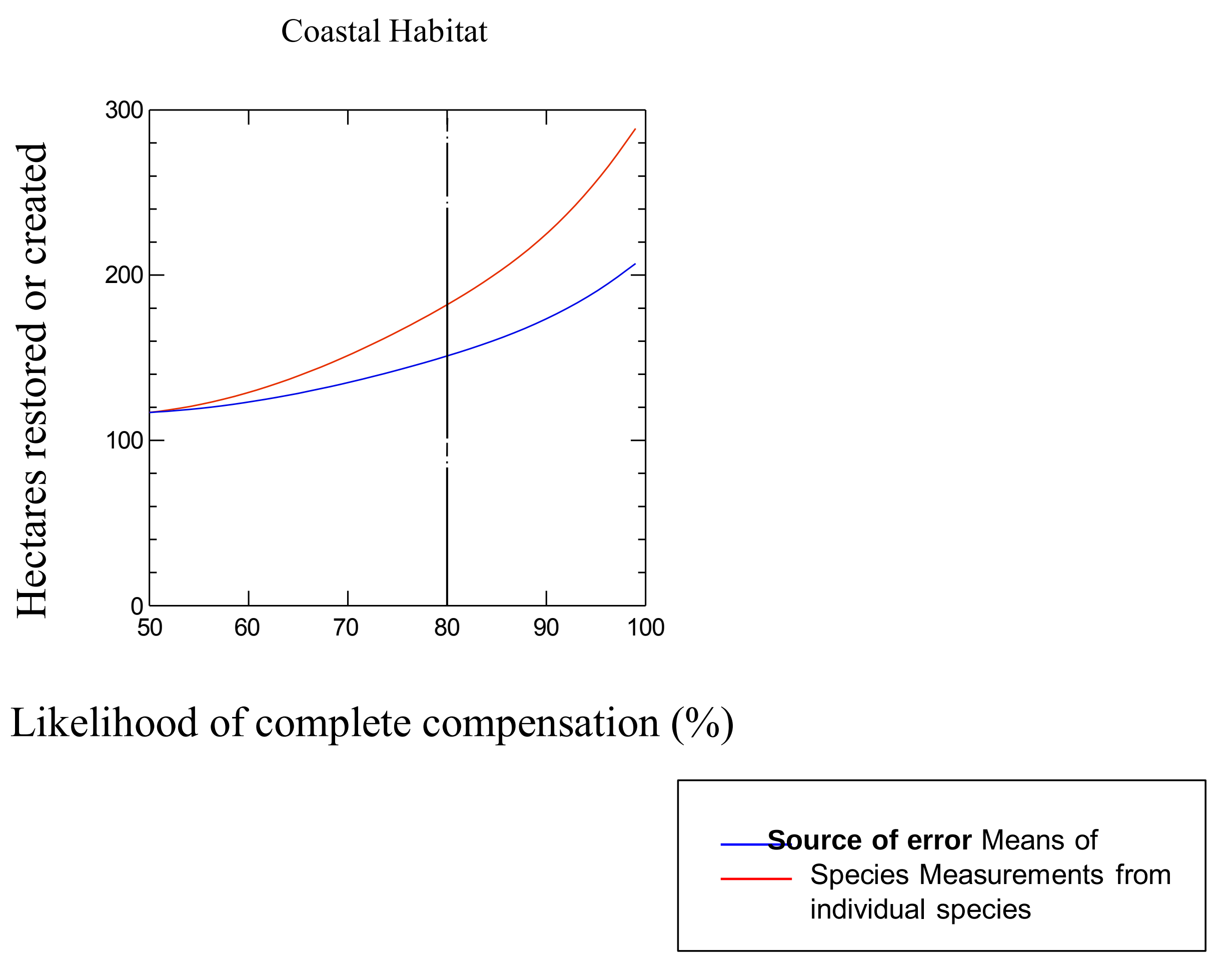

Figure 2c. Likelihood of complete compensation (%) Huntington Beach Generating Station.

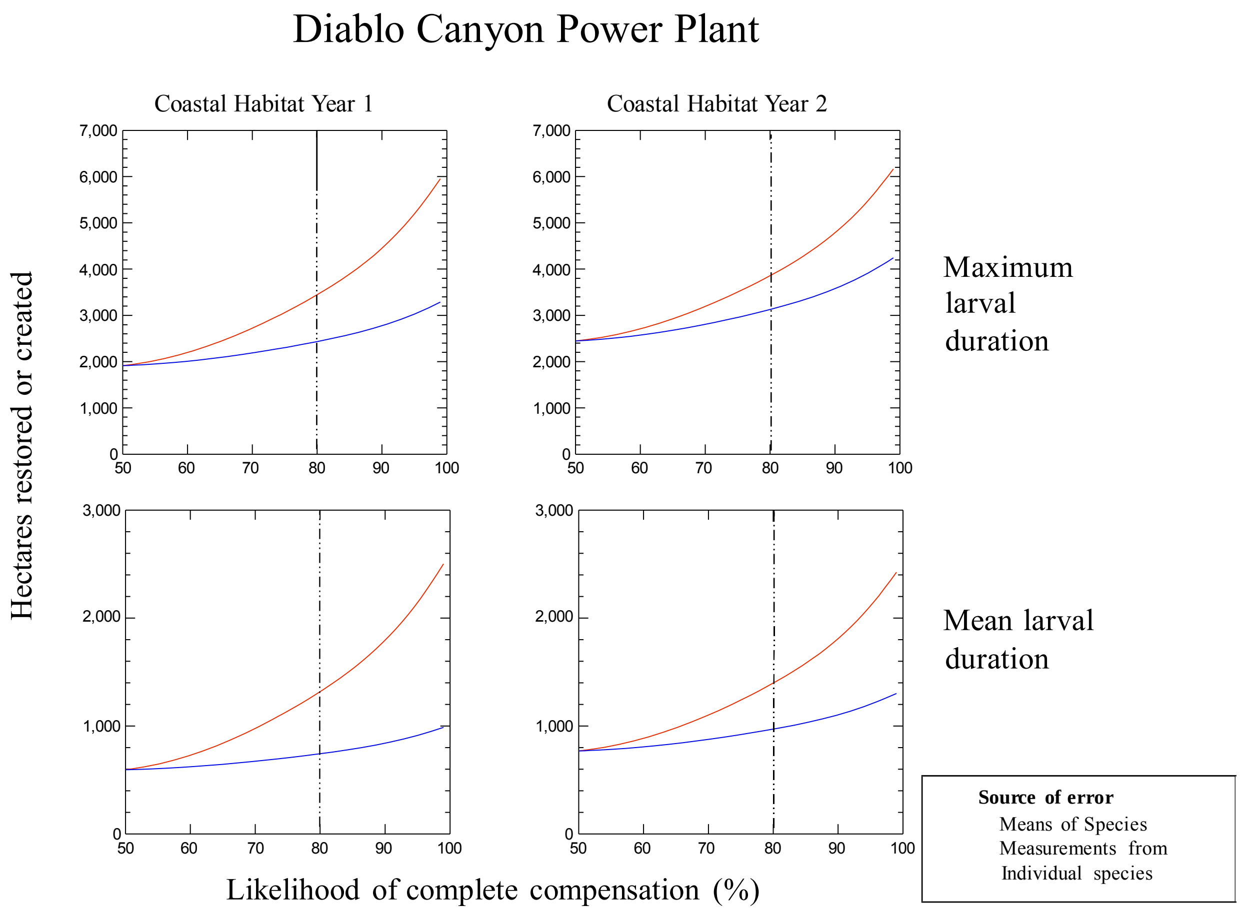

Figure 2d. Likelihood of complete compensation (%) Diablo Canyon Power Plant.

APA-1

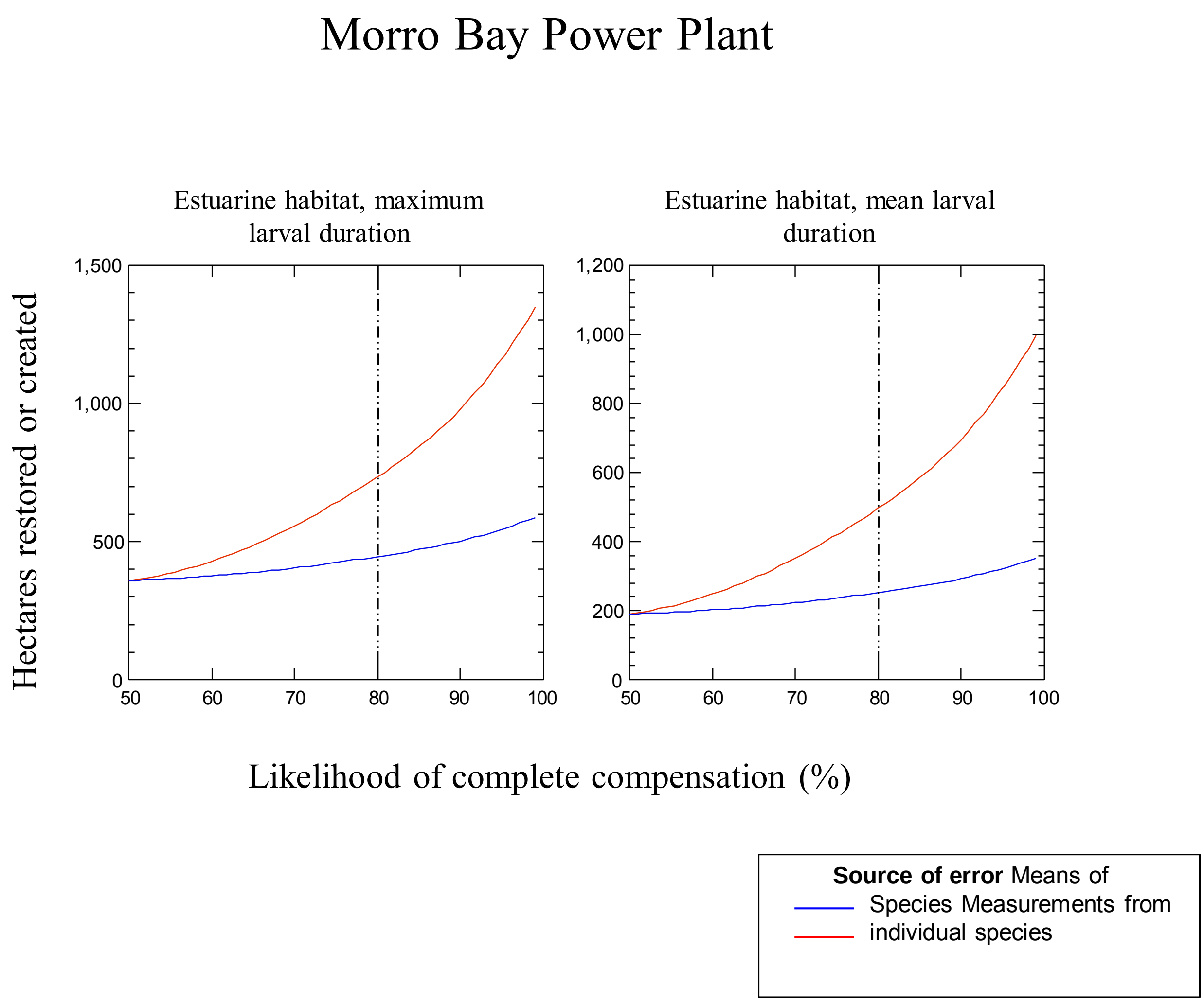

Figure 2e. Likelihood of complete compensation (%) Morro Bay Power Plant.

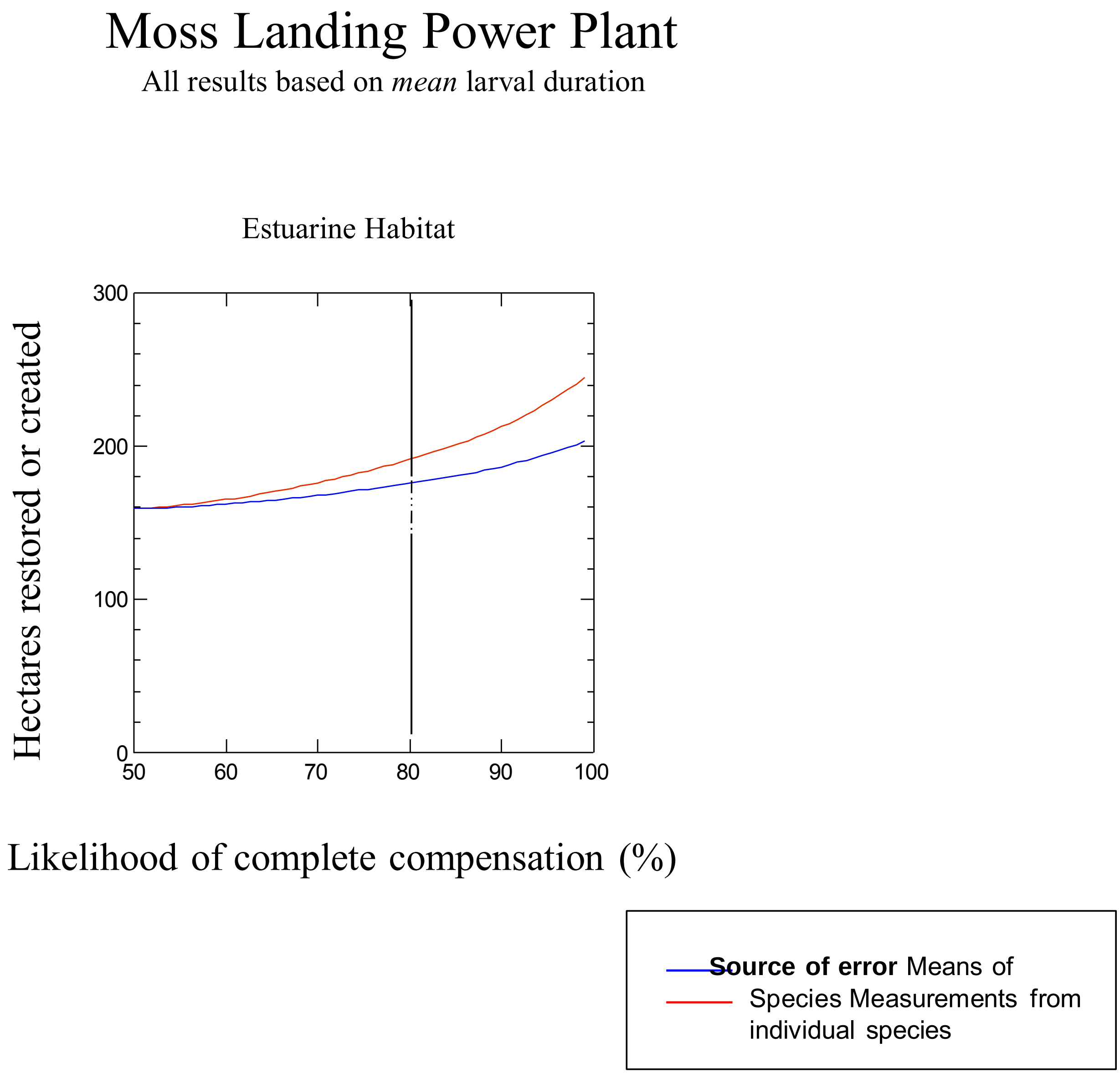

Figure 2f. Likelihood of complete compensation (%) Moss Landing Power Plant.

Potrero Power Plant

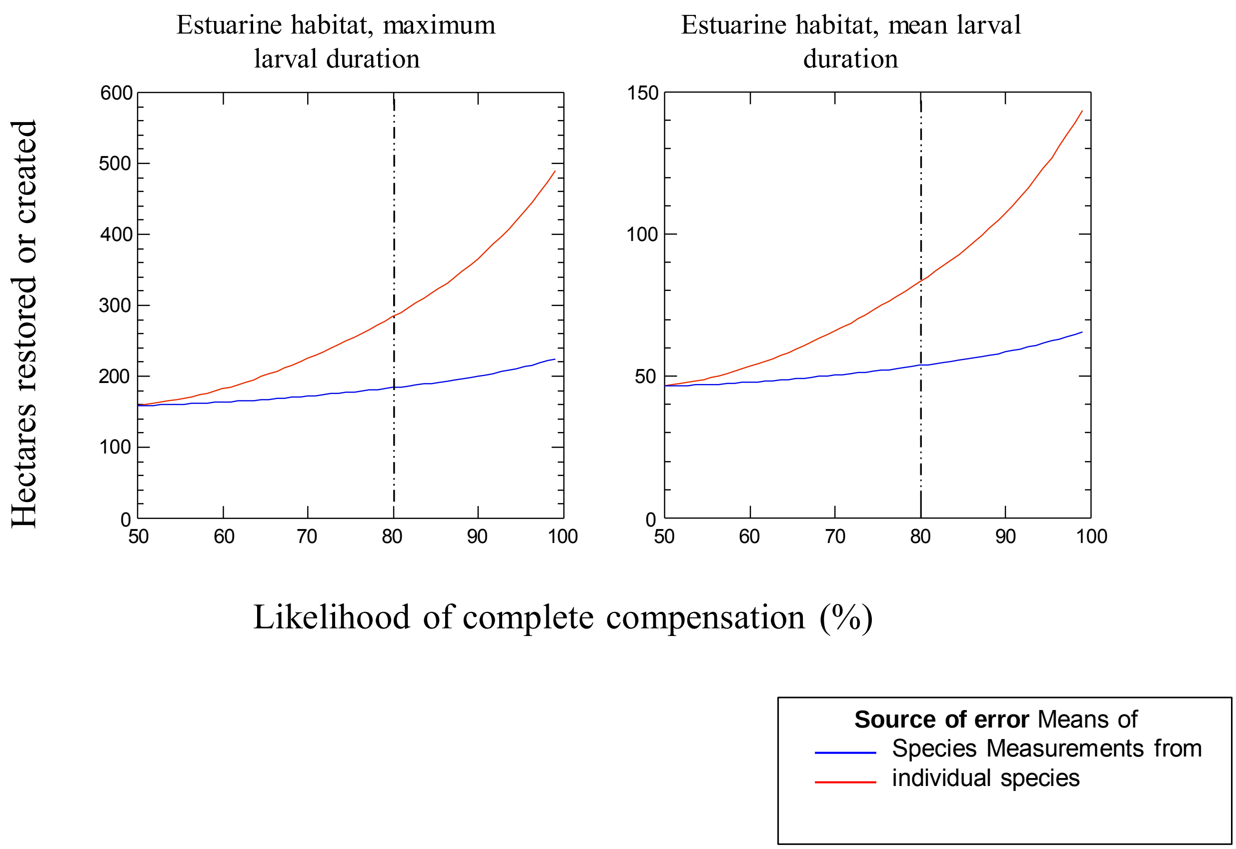

Figure 2g. Likelihood of complete compensation (%) Potrero Power Plant.

APA-1